MORETON BAY
and Catchment

Ian R. Tibbetts
Narelle J. Hall
William C. Dennison

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A publication of the **School of Marine Science**, The University of Queensland

Requests for this book should be made to:

School of Marine Science
The University of Queensland
BRISBANE Qld 4072
Australia.

National Library of Australia
Cataloguing-in-Publication

Moreton Bay and catchment.

Bibliography.
Includes index.
ISBN 1 86499 087 2.

1. Coastal zone management - Queensland - Moreton Bay Region. 2. Estuarine ecology - Queensland - Moreton Bay Region. 3. Water quality - Queensland - Moreton Bay Region. 4. Coastal ecology - Queensland - Moreton Bay Region -. 5. Moreton Bay (Qld.) - Environmental aspects. I. Tibbetts, Ian Rowland. II. Hall, Narelle J. III. Dennison, William, 1954-. IV. University of Queensland. School of Marine Science.

333.917099431

Published December 1998.

Published by the School of Marine Science, The University of Queensland, Brisbane Qld 4072 Australia
Cover photo courtesy Department of Environment and Heritage.
Cover design and graphics by Karen Holloway
Typeset by Narelle Hall and Karen Holloway
Printed by Watson Ferguson & Company

Preface

Moreton Bay is one of the world's great estuarine embayments. It harbours a rich diversity of organisms, some of which are vulnerable to extinction. Ringed by one of the fastest growing urban areas in the developed world (Skinner, p. 67), it is the epitomy of systems that demand urgent attention to the relationship between environment and development. The challenges we face in understanding its resources and processes are made complex by the interaction of the two principal components of change: natural and anthropogenic. It is crucial that we understand both and discriminate between them in our plans for a sustainable future for Moreton Bay.

It is clear from David Neil's paper (p.3) that there is no such thing as a *status quo* for Moreton Bay. On a geological timescale the Bay has experienced major changes both in its extent, through sea level changes due to glaciation, and freshwater input through changes in climate. So great have been some of these changes that periodically the Bay itself ceases to exist. We are fortunate to be present at a time when the Bay is filled and can provide a haven for the plethora of organisms that thrive in its sheltered waters.

For many, the Bay might appear as a timeless, ancient entity that will exist in perpetuity if we govern its resources wisely. This viewpoint is unlikely to be correct. The Bay is ephemeral. It is almost certain that relatively soon, on a geological timescale of tens of thousands of years, the Bay will again cease to exist, the streams of its catchment cutting new paths across the former sea bed and across the continental margin to some distant new shore.

At shorter timescales, tens to thousands of years, change is again a prominent feature of the Bay. It will continue to gradually infill with sediment from tidal deltas to the east and riverine deltas to the west. Banks of sediment will be reworked and mobilised by the effects of waves and tide.

On a more immediate timescale, well within a generation, it is clear that human activities such as land clearing, agriculture, urban development and industry will continue to rapidly change the nature of the Bay. Our challenge is to differentiate between natural and unnatural forces that cause change, and employ this knowledge for an integrated and effective management program.

Complicating this objective is that following 200 years of intensive development, it is difficult for us to divine what the Bay would have been like in the absence of such development. To what extent are the patterns and processes we measure today features of a natural system and to what extent have past and present human activity influenced them?

For example, and with reference to my own discipline, an understanding of what would have been the natural fish community of Moreton Bay in the absence of anthropogenic change is fraught with difficulties. Moreton Bay continues to be heavily fished for the purposes of both recreation and commerce. We know that trawling, for example, not only selectively removes some fishes, it also renders seabed topography less diverse, thereby reducing habitat diversity and as a consequence changing the diversity of fish communities. Moreover, we know that bycatch from fishing operations when returned to the system provides an abundant, if patchy, supply of carrion probably allowing organisms that can make use of such unnatural food falls prosper at the expense of those that cannot. All of these effects act to change the structure of fish assemblages and the processes by which their components interrelate. It is clear that determining what would constitute a natural assemblage of fishes in the absence of human influence is difficult, and next to impossible for less well understood organisms of the Bay.

How then should we set our management objectives for the Bay? Should we aim merely to arrest the processes leading to anthropogenic change and accept the present *status quo* as the bench mark against which the influences of future development are to be gauged, or should we aim instead to achieve some benchmark that includes criteria developed from our understanding of the Bay as it would have been in the absence of human influence? Whichever option (or more likely, compromise) is adopted, it is clear that if the present rate of unnatural change continues, there will shortly be left little to conserve.

If wisely managed, the Bay has the capacity to support industry, tourism and recreation in harmony with rich natural resources. It is my belief that this wisdom cannot be achieved through the activities of a ‘green’ movement alone, but requires a polychromatic approach through the collaboration of science, industry, government and community. We are witnessing the early stages in the development of this mature approach. Funding initiatives, such as the National Heritage Trust, are drawing together such partnerships to address problems at a local scale. Industries are seeking opportunities to support research, and alliances between local government agencies are nurturing major integrated research programs such as the *Brisbane River and Moreton Bay Wastewater Management Study* that will provide the basis for mechanistic management tools.

Such initiatives are likely to both greatly increase our knowledge of how the system works and at the same time increase our ability to exercise fine control over our impact on the Bay. Through control comes choice. The capacity for choice places heightened responsibility on the custodians of the resources of our Bay and its catchment. While the superficial responsibility of custodianship might seem to lie with the ‘managers’, the ultimate custodians of the Bay are the people of southeast Queensland.

Before I leave you to read the book I would like to make two points concerning time and perspective. The first is that warnings concerning the impact of humans on the Bay are not a recent phenomenon. Nearly a century ago, Thomas Welsby wrote of his concerns that the resources of the Bay were being exhausted and should be protected. This makes one wonder how our exhortations for the wise management of the Bay and catchment will be viewed 100 years hence. The second point on time and perspective arises from the apparent myopia associated with our present plans and strategies. We have an obligation, a custodial responsibility, to guard the resources of the Bay in perpetuity. We should be bound by vision that extends across generations. As David Neil so effectively points out, management visions that extend merely to the next election are sadly deficient. It is time we adopt an ethos of ‘management for millennia’. What more appropriate time than at the threshold of a new millenium is there for us to take stock, acknowledge past inadequacies, adopt a new vision for the Bay and challenge ourselves to translate that vision into reality.

Ian R. Tibbetts

Introduction

This compilation of work is for the people of the Moreton Bay region, the future of their Bay and the health of its catchment. While this book had its origins in the *Moreton Bay and Catchment Conference* held at The University of Queensland in December 1996, it is not merely a synthesis of that meeting. The authors, editors and reviewers have worked hard in the two years between the conclusion of the conference and the date of publication to refine and add to the content. Moreover, this book is not the first to draw together summaries of our knowledge of this system. We must pay our respects to forebears, such as *Moreton Bay in the Balance*, *Brisbane River: A Source Book for the Future* and *Future Marine Science in Moreton Bay,* a series of proceedings of conferences that prepared the ground for the present enhanced atmosphere in which marine science and management has begun to blossom in this region.

Regional research initiatives, such as the *Brisbane River and Moreton Bay Wastewater Management Study*, have made important inroads in our understanding of the Bay and how it works. While they increasingly allow us to consider management choices based on detailed knowledge of natural (and unnatural) patterns and processes, actions and reactions, causes and effects, we have yet a long way to go before we can secure the Bay's future.

National initiatives also impact on our investigations of the Bay. For example, the *State of the Marine Environment Report*, *Oceans Policy* and *Marine Science and Technology Plan* have outlined strategies for the investigation, assessment and conservation of our marine resources. Funding through the National Heritage Trust has empowered local communities to play a more active role in research, management and conservation. For individuals and groups, whether they be scientists, managers, politicians, industrialists or ordinary members of our community, synopses of knowledge that allow an objective assessment of the state of play will be an important resource from which they may determine priorities and courses of action. This book represents one such resource. It is not the first, nor will it be the last. It is part of a natural progression in the periodic synthesis and dissemination of information.

The scope of the work is naturally broad, both in the geographical limits of what defines Moreton Bay and Catchment and in the diversity of disciplines on which it touches. Where appropriate we have encouraged the addition of information and examples external to our region that have important lessons for us all.

The book is structured as a series of chapters that very broadly equate to themes used in the conference. Each chapter (theme) is provided with an overview. This allows the reader to gain an overview of what is discussed within the chapter and flags important concepts and gaps in our knowledge. Each overview is followed by a series of invited papers that summarise the state of our knowledge and highlights in more detail critical issues that will need to be addressed by future researchers and managers. These papers are themselves followed by papers or extended abstracts with a narrower focus providing information on specific topics. This structure provides a logical framework whereby the reader can sample the work at a level appropriate for their needs.

The fundamental units of this book are scientific papers. The scientific paper is a succinct and precise method of communication, developed over a long period as the most appropriate means for the communication of scientific knowledge. The basic measures of the quality (=credibility) of a paper are that the method is clearly explained, such that the work may be repeated, and that it is reviewed by experts in the field who have knowledge sufficient to correct errors and challenge inappropriate or unwarranted assertions. While this makes the paper a wonderful vehicle for the storage of knowledge and ideas it does present those not familiar with this form of communication some challenges. Happily the generally accepted structure of abstract (a precise summary), introduction (setting the scene), methods and materials (what was done), results (what was found), discussion (what it means), and references (sources of information) is now well used in the school science curriculum and should be at least vaguely familiar to most readers. While the general structure of scientific papers may be quickly divined, a greater challenge is presented by the jargon and units that tend to pepper such reports. To assist the reader in this respect we have provided a glossary of technical terms. To allow the reader to identify the location within the book of information of particular interest to them we have also provided an index.

Ian R. Tibbetts
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William C. Dennison

Acknowledgements

A large number of people contributed to the publication of this volume. Our particular thanks go to: Jack Greenwood for his foresight in developing the original Conference concept and his patience and encouragement during the production of this volume; Karen Holloway for many hours of hard work and commitment to ensure the high quality of the graphics and manuscripts that appear throughout the body of the book and cover; Heather Frankland for her patience with scientists in her attempts to bring grammatical lucidity to the volume; the reviewers, for donating valuable time to improve the scientific quality of the content; Ian Taylor of the University of Queensland Printery for advice on printing matters; and the School of Marine Science and Marine Botany Group of the University of Queensland who have contributed financially to its production. Ian Tibbetts is thankful for the unceasing support of his wife Wendy and children Daniel, Hannah and Rachel during the many hours spent editing.

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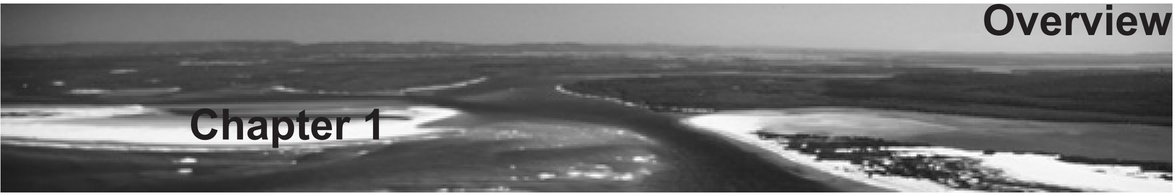

Moreton Bay and Catchment

As is desirable in such a book, the early passages set a scene against which subsequent information may be ranged. The three papers comprising this chapter are unified by their focus on change. They serve to emphasise that the Bay and catchment have undergone major changes in both their fundamental nature and the way in which we use them.

David Neil's paper summarises what is known of the geomorphological history of the region, Aboriginal and European settlement and recent development, and their effects on key habitats. He warns that the integrity of the Bay's ecology is under threat. He challenges as inadequate the short timescales and shifting baseline of previous attempts at managing the system and calls for a management vision of much greater temporal dimension if the Bay and its resources are to be sustained. David summarises an extensive and diverse range of literature, making extensive use of quotations from early explorers and inhabitants to paint a rich picture of how the Bay would have been at the time of European settlement. This allows us to envisage the environmental status (healthy) and capacity (high) that would have prevailed at the beginning of the nineteenth century. This pre-(major)disturbance datum is one of the potential targets of management criteria for Moreton Bay, but turning back the clock on 200 years of development is unrealistic. If this datum is rejected then the challenge for us all to determine what level of degradation from this datum of 200 years ago should we accept as a target for our management of Moreton Bay. As we progressively compromise our standard of quality for the Bay and catchment toward that which prevails at present, our task becomes easier and less expensive, however, we do not know that the Bay in its present state is in either an acceptable or sustainable condition. Of what we can be fairly certain, is that each additional impact on the Bay will render good management more difficult, more expensive and less likely to result in a satisfactory outcome.

Michael Capelin and his coauthors take us a step further in our understanding of the chronology and impact of settlement by discussing changing land use patterns in the region since Aboriginal settlement, with particular emphasis on vegetation cover. The authors confirm the intimate link between land and water, relating changes in landuse with water quality degradation in the catchment and Bay. In particular they highlight the rapid reduction in land cover by natural vegetation following European settlement and its acceleration of land degradation. Capelin *et al*. point out that among the most immediate and serious threats to the system is that of urbanisation, resulting from population growth. They suggest that if population growth in the region is not slowed then, to maintain the present integrity of Moreton Bay and its catchment, population growth will have to be accommodated in existing urban areas, meaning that population density must increase.

The final paper in the chapter, by Jim Skinner and coauthors, provides a review of population growth in the region and a forecast of future growth, enabling us to put the implications of the previous paper in perspective. The astonishing rate of past growth and the trend for continuing and rapid population growth of this region, among the fastest growing urban regions in the world, highlights the immense challenges that face the managers of both the natural and built environments of this system.

Ian R. Tibbetts

Moreton Bay and its Catchment: Seascape and Landscape, Development and Degradation

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Abstract

This paper presents an environmental history of Moreton Bay and its catchment. Aspects of the geological and sea level histories of the area are outlined. The development of the geomorphic and ecological systems of the Bay over the last 10 000 years is described. From 10 000 to 6 500 y BP the Bay experienced a period of filling due to rising sea level followed by a period (from about 6 000 years ago to the present) of relative sea level stability. Environmental changes in both the catchment and the Bay have occurred throughout the past 6 000 years as a result of both natural changes and human activities. The magnitude of the anthropogenic changes increased rapidly in the 200 years since European settlement. The nature of environmental change, both natural and anthropogenic, is outlined. In spite of widespread awareness of environmental degradation in the Bay and its catchment for the last 150 years, of calls for actions to rectify these problems, and of legislation ostensibly with this intent, degradation of the lands and aquatic systems of the catchment, and of the Bay itself, has continued. There is an urgent need to recognise that our obligations for management of Moreton Bay extend beyond the short term to time scales required for survival of systems and species which have survived for millions of years (albeit not necessarily in their present day locations) in the absence of human impacts and to take action to ensure that this can occur.

Introduction

Apart from its significance as home to about 1.4 M people, one of the most desirable places to live (as indicated by high rates of in-migration) and a comfortable climate for human habitation (Auliciems & Kalma, 1981), the southeast Queensland region is widely recognised as having considerable "environmental significance". Aspects of the significance of Moreton Bay include: being located at the overlap zone of tropical and temperate flora and fauna, harbouring coral assemblages unique in the Indo-Pacific region, reef islands probably unique in the world, the only significant population of dugong close to a major city and the southern-most significant dugong population in Australia and a site of great importance for migratory shorebirds (a declared Ramsar site). Moreton Bay is one of few sites in the world where humans can interact closely with 'wild' dolphins, one of the most important fisheries in eastern Australia, one of the fastest growing seaports in the country, and a major destination for recreational boating and fishing. The catchment of Moreton Bay contains some of the fastest growing urban areas in Australia, particularly severe development pressure on coastal and riverine habitats, extensive and expanding rural residential development, considerable areas of intensive cropping of high value produce and large areas of grazing land, as well as significant stands of subtropical rainforest and habitat for numerous vulnerable and endangered species.

Physiographically, Moreton Bay and its catchment could be regarded as a microcosm of the Queensland coast. The catchment consists of several river basins, some of which (Nerang, Coomera, Pine) are relatively small, rising in coastal ranges with short steep courses discharging direct to the coast. Rainfall and runoff in these catchments is relatively high and land use is dominated by closed forests, cropping and urban areas. By contrast, there are two large catchments (Brisbane and Logan) which rise in the hinterland of the coastal ranges, have long, meandering, low-gradient courses, low rainfall and runoff and land use dominated by

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 3-54.

open forests, grazing and limited areas (proportionally) of cropland and urban development. All of these streams discharge to a semi-enclosed water body, with restricted tidal exchange with oceanic waters, and harbouring significant areas of mangroves, seagrasses, saltmarshes and coral communities, and associated populations of fauna and flora. In many respects, these characteristics are typical of the catchments and waters of the Great Barrier Reef Lagoon, albeit on a smaller scale and in a subtropical climate. However, they are in marked contrast to the catchments and coastal waters of southeast Australia.

The southeast Australian coastline lacks the offshore barrier islands which enclose large bodies of relatively protected waters adjacent to low energy coastlines. South of Moreton Bay is a largely open, high energy coastline where major streams discharge direct to oceanic waters. Where large embayments do occur on the southeast coast, the rivers discharging to them generally have small catchments and low total discharges. Furthermore, because rainfall is less variable on both seasonal and long term (i.e. interannual and longer) time scales, river flow is also less variable and coastal ecosystems are likely to be influenced by relatively constant levels of terrestrial input, by comparison with the often extreme, episodic events of the tropics and sub-tropics.

These contrasting physiographic characteristics may be reflected in the perspectives of scientists and managers. For example, the *State of the Marine Environment Report* (SOMER; Zann, 1995) found that declining marine and coastal water/sediment quality, particularly as a result of inappropriate catchment land use practices, is probably the most serious issue affecting Australia's marine and coastal environments. Nevertheless, Underwood & Chapman's (1995) *Coastal Marine Ecology of Temperate Australia* all but ignores the impact of catchments on the coastline. By contrast, in northeast Australia the relationship between catchments and coastal ecosystems has had an increasingly high profile for the last decade. This is evident in the workshops and research projects supported by the Great Barrier Reef Marine Park Authority (GBRMPA), the research profile of the Australian Institute of Marine Science (AIMS), and numerous other activities such as the conference on *Downstream Effects of Land Use* in 1995 (Hunter *et al.*, 1996) and the CRC Reef Research-sponsored workshop on *Great Barrier Reef: terrigenous sediment flux and human impacts* (Larcombe & Woolfe, 1995) and various television documentaries (e.g. Four Corners 1987 *Super reef*).

Moreton Bay is significant in this context, lying at the southern end of the semi-enclosed coastal systems, with economically and scientifically important coastal ecosystems adjacent to both large and small catchments that have been subjected to significant anthropogenic change and degradation. However, an important aspect of the interest in the catchment–coast relationship in Queensland, is the role of the Great Barrier Reef (GBR) as an aesthetic, cultural, scientific and economic icon. By contrast, the values of Moreton Bay are more subtle, there appears to be an element of (superficial) “familiarity breeds contempt”, and for most of the European history of the region the perspective of the Bay has largely been from the intertidal mudflats and often turbid waters of the western Bay.

This paper provides an overview of the changing nature of the environment of Moreton Bay and its catchment through pre-Tertiary and Tertiary, Quaternary and Holocene time periods, and examines both natural and anthropogenic factors influencing environmental change. The paper takes the approach of environmental history, attempting to explain “... how we got to where we are: why is the environment ... like it is?” (Dovers, 1994). The emphasis, however, is on the biophysical environment of Moreton Bay, rather than the socio-economic environment. The sources used range over numerous disciplines and include scientific literature, historical documents and accounts and anecdotal information. Clearly, there is some variability in the reliability of these various sources.

Pre-Tertiary and Tertiary

Significant aspects of this period for modern Moreton Bay include the definition of the main physiographic features of the region and the establishment of the future catchment boundaries. Some Mesozoic sediments have high susceptibility to landslides (e.g. the Heifer Creek sandstone of the Marburg formation), gullying (e.g. the Gatton sandstone of the Marburg formation) and salinity (e.g. the Marburg formation and Walloon Coal Measures). Complex interactions occur between these susceptible landscape elements, patterns of climate variation, and patterns of land use change following human settlement.

Tertiary basalts erupting about 25 million years ago (Ma), now form some important topographic features of the Bay. These now act as substrates for some reef islands (e.g. Mud Island), and high islands (e.g. Russell and some other islands of the southern Bay) which in turn form loci for sediment deposition and colonisation by mangrove and other intertidal vegetation communities. Tertiary lateritic platforms later became important as a substrate for coral colonisation on both mainland and island coasts.

The Tertiary basalts play a role in defining both the catchment boundaries and the location of the mouth of many of the smaller streams discharging to the Bay. The basalts also form the nutrient-rich, fine-textured soils which subsequently influenced patterns of land use and human settlement. In turn, these patterns influence the distribution of sediment and nutrient yields from the catchment to Moreton Bay.

The Quaternary

Coastal landforms in the southeast Queensland region, although of greater antiquity than coastal terrain in most areas of the world, are much younger than those of the catchment, being largely confined to the Quaternary period (the last 2 M years). During this period, the results of tectonic processes, in terms of the positions of land masses on the earth's surface, construction of mountain ranges, and oceanic circulation patterns, led to conditions where oscillations in the earth's orbit, and consequent changes in solar insolation, were expressed as global oscillations in temperature. These temperature oscillations resulted in oscillations of glacial ice accumulation and, as a result, of sea level over a range of about 150 m (e.g. Chappell, 1983a). Present sea level is within a few metres of the upper limit of this range. As a result of these patterns, the Quaternary is a defining period in the evolution of Moreton Bay in several important ways:

1. it was during this period that the dune-island barriers (Stephens, 1982), which define the eastern coast of Moreton Bay, were formed;
2. sea level oscillations during this period defined the geomorphic and ecological character of the Bay;
3. climatic oscillations influenced geomorphic and ecological characteristics and processes in the catchment; and
4. the end of the last glacial phase established the environmental setting and processes within which the present Moreton Bay would form.

Each of these factors is dealt with in turn below.

Dune-island barriers

The bedrocks on which the dune-island barriers have formed are about 60 m and 50 m below sea level for North Stradbroke and Moreton Islands, respectively (Stock, 1990). Without the accumulation of sands on these foundations there would be no Moreton Bay as we know it. Furthermore, it is these Quaternary sea level oscillations which mediated sand island formation.

Sediments carried onto the continental shelf by rivers during periods of low sea level were transported landwards by the prevailing winds to form the coastal dunefields. Whether this occurred at times of low, rising, high or falling sea level has been the subject of some debate.

In eastern Australia, transgressive dunes, which form by far the greatest part of Moreton and North Stradbroke Islands, have been variously reported as forming during low (glacial phase) sea levels (e.g. Ward, 1978; Roy & Thom, 1981; Thom *et al.*, 1994), rising (transgressive phase) sea levels (Stephens, 1982; Pye & Bowman, 1984; Willmott & Stevens, 1992), and high (interglacial phase) sea levels (Kelley & Baker, 1984; Pickett *et al.*, 1984, 1985, 1989). Glacial phase dune building is attributed to relatively arid and windy conditions with lower vegetative cover to stabilise coastal sand masses. Transgressive phase dune building is attributed to destabilisation of the vegetation by the rising sea, and formation of extensive areas of foredunes which are then blown inland. Interglacial phase dune building may be a consequence of the reworking of the abundant foredune sands mobilised during the preceding transgression. There are significant areas of unvegetated blowing sand in the southeast Queensland region in the present interglacial. It is known that they were in existence and active before European settlement because they were observed by Joseph Banks in 1770 (Beaglehole, 1962), and there are parabolic dunes in eastern Australia which have been radiocarbon-dated to the present interglacial (e.g. Lees *et al.*, 1990).

Pickett *et al.* (1985) present various lines of evidence to show that the dune which formed 105 thousand years ago (ka) at Amity Point has maintained its vegetative cover continuously since it was deposited, suggesting that "... the climate here during the last glaciation did not deteriorate beyond that necessary to sustain vegetation ... dune building along the Queensland coast was not necessarily associated with low sea levels or very arid conditions". However, Thom *et al.* (1994) suggest that vegetation cover on sandy surfaces was quite patchy during the last glacial maximum (LGM), i.e. some areas were continuously stabilised by vegetation, some were not. Furthermore, given that the dates for the Amity site have been revised to c. 125 ka (Pickett *et al.*, 1989), it is conceivable that the particular dune in question was initiated during the penultimate glacial maximum and continued to blow landwards (continuously or episodically) until stabilised by increased vegetation cover following amelioration of the climate during the last interglacial. The morphology of some parabolic dunes provides evidence of episodic transgression over long time periods (Neil, 1983; Stock, 1990). Pye (1984) suggested that "... it is conceivable ... that dune formation ... may have continued during both a transgressive and regressive marine phase, or to have exhibited a periodicity which is completely unrelated to sea level conditions." Stock (1990) suggests that "the dune sands visible today could have been formed at **any** sea level position", including levels higher than present.

It seems likely that dune building has occurred during glacial, transgressive and interglacial sea level phases. Furthermore, sea level oscillations during the stadial–interstadial periods may have amplitudes about half that of the glacial–interglacial cycles and rates of change of a similar magnitude, with resulting instability of coastal systems. Consequently, dune building seems possible throughout the glacial–interglacial cycles, although dune building maxima would be likely during the glacial phases, limited by the apparently diminishing (Thompson, 1981) availability of sand on the continental shelf.

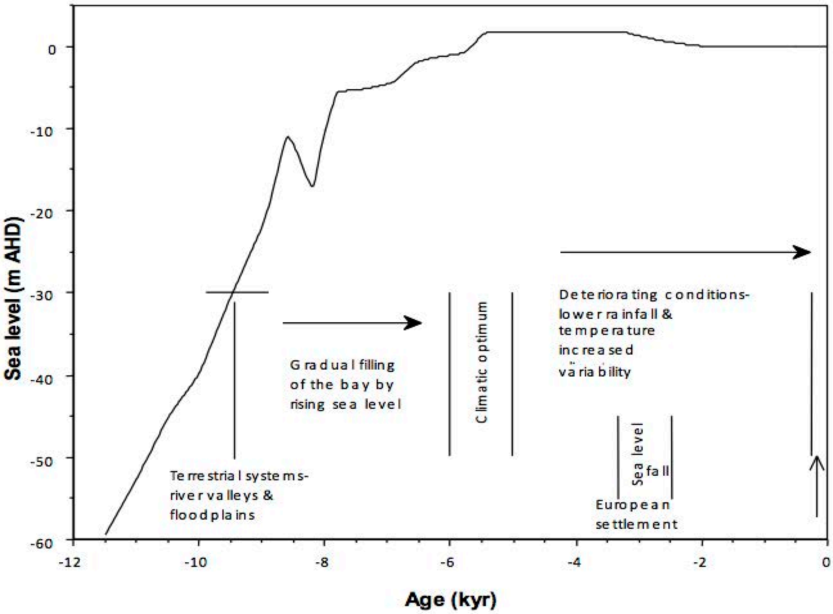
A further consideration in the interpretation of the chronology of the dune islands is raised by Nott *et al*. (1994) who erroneously note that the work of Thompson (1981), Walker *et al.* (1981), and their subsequent publications, assumes that "one soil profile equals one sedimentary unit". The finding that ages for soil A_2 horizons were as much as 200 k years younger than for the underlying B horizon of what was apparently one soil profile, led Nott *et al.* (1994) to caution that "application of these ... [pedologic] ... processes to determine the relative age of deposits which have not been independently dated is fraught with danger..". This conclusion, which questions the validity of the geomorphic framework outlined for the southeast Queensland dunefields, is based on dating of soil profiles in southern New South Wales dunefields. The geomorphic context of the New South Wales sites is sufficiently different from that in most of the southeast Queensland dunefield that their findings may be less relevant to this region, although Stock's (1990) work, by far the most comprehensive on the geomorphology of the southeast Queensland dune fields, demonstrates greater geomorphic complexity in these systems than is recognised in Thompson's model.

Sea level oscillations and the Bay's characteristics

Patterns of sea level change dictate that Moreton Bay can exist in something like its present form for periods of just a few thousand years every 100 000 years or so and, due to factors such as variations in antecedent conditions, the height and duration of each sea level stillstand and any residual inheritance from previous high sea levels, each manifestation of the Bay would differ from previous ones. The morphostratigraphic and dating evidence from the dune barrier islands suggests that they may have been present for only the last several sea level oscillations, although a 700 ka date from the Cooloola dune field raises the possibility that Moreton Bay may have formed episodically over at least this period of time.

During each of the Bay's manifestations an evolutionary sequence, similar to those described by Roy (1984) for New South Wales coastal embayments during the Holocene, is likely to have occurred. Such a sequence generally involves a gradual transition from oceanic, clear water ecosystems (e.g. with coral communities) to communities more tolerant of turbid conditions (e.g. seagrass and mangroves), as infilling by both terrigenous and marine sediments occurs and flushing rates decrease. The oscillations in sea level truncated this sequence, resulting in an embayment of the present form, rather than the shallow coastal swamp which would have developed had sea levels remained at the present level for much longer periods.

The present Moreton Bay formed as sea level rose from that of the Last Glacial Maximum (-150 m at 18 ka) to about the present level (c. 6 ka; Figure 1) (Chappell, 1983a; Lambeck & Nakada, 1990). The sequence is described by Stephens (1992). Conditions in the proto-Moreton Bay are largely unknown, although they are relevant to the Bay's ecological development after sea level stabilised. For example, where were the nearest coral reefs? Woodroffe *et al.* (1986) have shown how mangrove communities tracked sea level rise in north Australian estuaries, forming extensive mangrove swamps in the mid Holocene (Woodroffe *et al.*, 1985). Was proto-Moreton Bay extensively colonised by mangrove communities, or was there a significant timelag as propagules arrived from elsewhere? Flood (1978) has suggested that, at Empire Point, mangrove colonisation occurred only recently, subsequent to the demise of the coral reef there. The most likely scenario is that mangroves were present in proto-Moreton Bay, although the geomorphic setting differs from that of northern Australia and extensive mangrove swamps were unlikely. Their late arrival at Empire Point can be attributed to changing conditions during the late Holocene. The Pleistocene inheritance of Moreton Bay is also important in determining the possibilities for ecosystem development as the Bay filled. For example, interglacial high sea levels during the Pleistocene are likely to have eroded developing soils from the Tertiary lateritic platforms, which are common along the Bay's west coast, maintaining them as a potential substrate for coral colonisation.



Influence of climatic oscillations on the catchment

There is abundant evidence of significant impacts on the geomorphic and ecological characteristics of eastern Australia, as elsewhere on the globe, as a result of marked climatic oscillations throughout the Quaternary (e.g. Bowler & Wasson, 1984; Singh & Geissler, 1985; Kershaw, 1994). Changes which occurred include variations in rates of erosion by water and by wind, of soil formation, and of the area, distribution and composition of vegetation communities. Although these environmental oscillations occurred in the Moreton region, there is little palaeoenvironmental evidence from this region to establish the exact nature of the changes experienced in the Moreton Bay catchment. Geomorphic and ecological responses to these climatic oscillations are likely to have been hysteretic (i.e. not following a reciprocal trajectory during deteriorating and ameliorating climatic conditions; e.g. see Bowler & Wasson, 1984) and complex, with different components of the system responding in different ways at different rates (e.g. Huntley & Webb, 1989; Overpeck *et al*., 1992; Faunmap, 1996) and different time scales (Roy *et al*., 1992; Valentine & Jablonski, 1992). In simple terms, the general pattern is likely to have been one of greater areas of closed forest, relative to open forest and woodland, during interglacial phases.

End of the last glacial phase

Rising sea level at the end of the last glacial phase would have resulted in geomorphic changes in the catchment. Some adjustments in stream morphology would have occurred as the stream profile adjusted to the new base level (i.e. sea level). In the catchment, changing climate is likely to have played a far more significant role than changing sea level, although sea level rise contributed to the changing climate by moving the coastline up to 50 km westward of its position during the last glacial phase, thus changing the distribution of onshore climatic gradients.

The climate became warmer and wetter as the ice caps receded which resulted in the expansion of the more hydrophilic vegetation communities. These changes were occurring during the formation of proto-Moreton Bay, but, by analogy with sites elsewhere and allowing for lag times (Kershaw & Nix, 1988), the climate and vegetation would have largely stabilised by the time Moreton Bay reached its present form.

Another potentially important change at this time was the expansion westward of the regional Aboriginal population, forced to migrate by the rising sea level and more able to occupy the catchment area as a result of ameliorating climatic and environmental conditions. There is now at least circumstantial evidence for Aboriginal occupation from widespread areas of Australia for periods approaching, and in some cases greater than, 100 000 y (e.g. Roberts *et al.*, 1990; Singh & Geissler, 1985; Kershaw, 1994). In the Moreton region, Neal & Stock (1986) document continuous occupation of the Wallen Wallen site on North Stradbroke Island for c. 22 000 y, as well as the changing resource base as the Bay was transformed from a terrestrial to a marine environment. An increase in the number of dated occupation sites (Walters, 1989) as sea level rose is evidence for increased intensity of use of the inland areas of the Moreton Bay catchment. There is evidence from elsewhere in coastal areas of eastern Australia of geomorphic instability, increased erosion and marked changes in vegetation communities as coastal Aboriginal populations were concentrated by rising sea levels (e.g. Hopkins *et al.*, 1996; Thomas & Kirkpatrick, 1996).

The Holocene

At 6 000 years ago: the climatic optimum

Climate and sea level

Throughout the world there is evidence, from numerous locations and various types of proxy records (e.g. palynology, dendrochronology, lake level chronologies, isotopic analysis of ice cores), of a period at about 6 ka of relatively warm temperatures and high precipitation by comparison with the present – referred to as the climatic optimum, or hypsithermal (Bell & Walker, 1992). In the Australian region, too, there is evidence of this pattern from locations ranging from New Guinea (Bowler *et al.*, 1976) to Antarctica (Jouzel *et al.*, 1987). In Australia, as elsewhere (Bell & Walker, 1992), the timing and magnitude of the climatic optimum varies. For example, Kershaw & Nix (1988) have inferred climatic optimum temperatures 2-3.5^{o}C higher than present (21^{o}C) between 3 600 and 5 000 ka on the Atherton Tableland and mean annual rainfall 300-700 mm greater than the present 1 700 mm during this period, and possibly higher during the 6 000-7 500 ka period. The Holocene record closest to Moreton Bay is that of Dodson *et al.* (1986) and Dodson (1987) from Barrington Tops, New South Wales. Their interpretation of the palynological record suggests that the period 8.5-2.3 ka was wetter and warmer than the present, and 6-5 ka was the wettest part of that period.

These climatic patterns are also associated with higher sea level than at present. Sea level in Moreton Bay during the climatic optimum was 1.0-1.5 m higher than present (Flood, 1983; 1984), an estimate consistent with those determined elsewhere for northeast Australia in the mid Holocene (Hopley, 1983; Chappell, 1983b; Larcombe *et al.*, 1995). With the exception of sea level estimates, there is little unequivocal evidence from southeast Queensland to indicate either the occurrence or the magnitude of the mid Holocene climatic optimum. However, a single sample, c. 7 500 ka, from estuarine muds in the Maroochy River estuary is particularly rich in vine forest and *Araucaria* pollen, suggesting higher rainfall and a much less open forest in this catchment in the mid Holocene than at the time of European settlement (Bell, 1978).

The Moreton Bay catchment in the climatic optimum

In the absence of detailed palaeoclimatic data specific to the southeast Queensland region, inferences regarding the effect of climate change, particularly rainfall variations, on catchment processes are made using analogues from elsewhere in eastern Australia. Given the widespread evidence for the climatic optimum and the objective, quantitative estimates of Kershaw & Nix (1988), rainfall in the Moreton Bay catchment is estimated to have peaked at between 18 and 42% greater than present. Assuming that mid Holocene rainfall was 25% greater, and using modern relationships between rainfall and runoff and a simple model relating stream sediment concentrations to catchment runoff for Queensland coastal streams (Neil & Yu, 1996), the mid Holocene streamflow, sediment concentrations and total sediment yield from the Brisbane River were estimated. Results of this analysis suggest that runoff during the mid Holocene climatic optimum may have been about 60% greater than at present. More importantly, the mean flow-weighted suspended sediment concentration is likely to have been about 40% lower than the stream sediment concentrations prior to the effects of European land use intensification, as estimated by Neil & Yu (1996). Given the large proportion of nutrients which are transported adsorbed to inorganic particulates, these estimates would also imply lower nutrient input to the Bay at this time, as well as an increase in the ratio of dissolved to particulate nutrient delivery.

An alternative interpretation of the mid Holocene climate is provided by Shulmeister (1996) which suggests that the mid Holocene climate was wetter in both northern and southeastern Australia, but little different from present in southern Queensland. Under this scenario, the changes in runoff and stream sediment concentrations outlined above would not have occurred.

Shulmeister (1996) suggests that the El Niño–Southern Oscillation (ENSO) may have been "switched off" during the mid Holocene (prior to about 3.5-4 ka) as a result of low intensity Walker circulation at that time. This conclusion is consistent with the proxy palaeoclimate records contained in massive *Porites* corals from Orpheus Island (GBR) which indicate that, at about 5.8 ka, both rainfall and sea surface temperature were less variable than at present, consistent with no, or much reduced, ENSO activity (Gagan *et al.*, 1997). The implication of this finding is that, for a given mean annual rainfall, erosion rates would have been lower than at any time since. In the absence of ENSO, variability of catchment condition (e.g. ground cover, with its critical influence on erosion rates) would have been low.

Under the wetter and less variable environmental conditions postulated, the catchment would have experienced better ground cover, on average, and few severe fluctuations of climate with severe, long duration droughts. Consequently, runoff would have been higher than at present, but more consistent. Sediment concentrations in stream waters are likely to have been lower than at present. Flooding in the Bay would have been more frequent but of lower magnitude and carrying less sediment. The lower reaches of the major streams, drowned by the rising sea, are likely to have been broad, funnel-shaped estuaries, rather than the relatively narrow channels of the modern streams.

Moreton Bay in the climatic optimum

The physical environment of Moreton Bay, as sea level stabilised at the end of the post-glacial marine transgression, was different from that of the present in numerous ways.

- The west coast of the Bay was further landward in many places due to the mid Holocene sea level being higher by 1.0 to 1.5 m. Examples of these mid Holocene coastlines include the Serpentine Creek area adjacent to the north bank of the Brisbane River (coastline up to 9 km landward; Gourlay & Hacker, 1983; Hacker & Ward, 1985) and Deception Bay (coastline up to 4 km landward; Flood, 1981).
- The thickness of bottom muds in the western Bay was much less than at present, both in sediment wedges adjacent to stream mouths and on the platforms which commonly underlie the mudflats of the western Bay. Stephens (1992) estimated that a volume of 1.86×10^9 m^3 of mud has been deposited in the Brisbane River delta in the western Bay, in places > 5 m thick.

- Restriction of tidal circulation through the entrances in the northern and eastern Bay was minimal. In the area of the ancestral Jumpinpin Bar, a transgressive barrier/back-barrier system (the embryonic South Stradbroke Island) enclosed the embayment adjacent to the Coomera and Pimpama Rivers. Baker (1984) argues that, during the transgression of this barrier system, there was a tidal entrance at its northern end so that, in effect, it brought an incipient tidal delta with it when it stabilised in alignment with the then east coast of North Stradbroke Island. The South Passage tidal delta is the product of Holocene sedimentation and, in the North Entrance tidal delta, Holocene sands have been shown to average about 10 m in thickness, although some of this may be reworked Pleistocene sands (Stephens, 1978; 1992).
- Pleistocene parts of the dune-island barriers only were present. The extensive beach ridge sequences on the northwest coasts of both North Stradbroke and Moreton Islands and the Holocene sands at Reeders Point were absent so South Passage would have been much wider. The North Entrance may have been wider in the absence of the Holocene part of the beach ridge sequence at Comboyuro Point and the Holocene sands of Bribie Island.
- In the absence of the "flood-tidal delta islands" which accreted on the tidal shoals adjacent to the Jumpinpin Bar, the deltaic islands of the Logan River estuary and the Holocene accretion on the west coast of Russell Island (see Baker, 1984), the tidal exchange through the area of the present Jumpinpin Bar would also have been greater than at present.
- Sea surface temperatures (SSTs) in the central GBR, were c. 1°C higher than present (Gagan *et al.*, 1996) and this is also likely to have been the case in Moreton Bay waters. Assuming that the 1°C increase occurred in Moreton Bay, and that the thermal gradient on the southern Queensland coast was similar to that at present, summer SSTs are likely to have been similar to those in the latitude of Rockhampton, and winter SSTs similar to those at the latitude of Mackay (see Lough, 1994). Furthermore, the absence of ENSO (Gagan *et al.*, 1996; Shulmeister, 1996) would have reduced SST variability, minimising the temperature extremes which can be lethal to, for example, coral communities.
- Hypothesised low-intensity Walker circulation (Shulmeister, 1996) compared to that today, may have resulted in low intensity southeast tradewinds. Under this scenario, it is likely that mid Holocene bottom sediment resuspension, hence water turbidity, was lower in the western Bay than at present. On the other hand, the wider and deeper openings at ocean entrances would have resulted in exposure of some areas of the Bay to higher wave energies, analogous to the mid Holocene high energy window described by Hopley (1984) for the GBR.

The physical consequences of these conditions include greater tidal exchange with oceanic waters and probably more rapid flushing, and a greater volume of water into which fluvial input could be mixed and diluted. Deeper waters and less bottom sediment, particularly in the western and southern Bay probably resulted in reduced bottom sediment resuspension and turbidity by comparison with present conditions. The combination of deeper water and increased tidal exchange would have reduced the extremes of minimum and maximum temperature experienced in the Bay, by comparison with those at present.

These characteristics of the physical environment of Moreton Bay during the mid Holocene climatic optimum necessarily resulted in a biota which differed from that of the present. Some of these changes may have included the following:

- higher and more stable water temperatures would have made the Bay a more suitable habitat for dugong, perhaps thermally more similar to Hervey Bay or Shoalwater Bay;

- deeper entrances to the Bay, associated with the warmer temperatures, would have resulted in a Moreton Bay more similar, in some respects, to modern Hervey Bay and possibly a suitable resting and breeding area for humpback whales (*Megaptera novaeangliae*);
- more "tropical" in species assemblages, a characteristic particularly evident in the sub-fossil scleractinian coral species assemblages (e.g. Wells, 1955; Lovell, 1989);
- the presence of lateritic rock platforms, rather than mudflats, in the western Bay which are suitable sites for colonisation by corals (these platforms were probably created by erosion of the overlying soils as the sea level rose); and
- restricted areas of mudflats and small mud volumes, less wind energy for sediment resuspension, better tidal circulation, lower fluvial sediment concentrations, and reduced flooding-related salinity fluctuations would have resulted in markedly improved water quality.

As a consequence of these conditions, the populations and patterns of distribution and community structure of, for example, coral, seagrass, fish, marine mammal, and infaunal communities are likely to have been quite different from those of the present. The ecological implications of the characteristics of the physical environment of Moreton Bay at the climatic optimum are probably best seen in the scleractinian corals, given the resistance of their skeletal material to weathering. Turbidity-sensitive acroporid corals are uncommon in the western Bay at present but form the bulk of subfossil corals at sites in western Moreton Bay from Mud Island to Empire Point (Jones *et al.*, 1978; Flood, 1978). In the southern Bay, acroporid skeletal material is common on beaches and intertidal flats of Point Halloran, Coochiemudlo and Lamb Island, and has been found as far south as Redland Bay. Furthermore, coral reefs were more extensive throughout the Bay in the past, occurring in areas where there is little or no modern coral growth (Lovell, 1989). The marked offshore contraction of the distribution of coral growth, and of the acroporid range in particular, is indicative of the severity of the environmental changes which have subsequently occurred.

From 5 000 to 200 years ago: gradual, natural and anthropogenically-induced, degradation

Natural change

The deterioration of the climate during the late Holocene led to drier conditions and, coupled with the onset of ENSO oscillations, increased incidence of climatic extremes, including, for example, oscillations between flood-dominated (more humid) and drought-dominated (more arid) climates (Warner, 1987; Erskine & Warner, 1988) and highly erosive rainfall events. A slight decrease in temperatures would have accompanied these changes. During shifts in climate (particularly rainfall), the soil, vegetation and climate are not at equilibrium, and soil erosion and sediment yield increase while a new equilibrium is established (Knox, 1972). A climatic shift to greater humidity is likely to result in channel incision, while a shift to greater aridity is often followed by hillslope erosion and channel aggradation (Hereford, 1984; Balling & Wells, 1990).

In the Moreton Bay catchment, climatic changes would have led to geomorphic instability, a deterioration of vegetative cover (associated with changes in community composition and the distribution of plant communities), with droughts bringing extremes of vegetative cover decline. Increased variability in streamflow and higher stream sediment concentrations in more extreme and episodic events are likely to have led to significant shifts in the geomorphology and ecology of catchment streams and an increase in the impact of the catchment on the ecosystems of Moreton Bay.

The catchment was experiencing a shift from the "optimum conditions" of the mid Holocene towards the conditions of the present. While this shift in catchment condition following the climatic optimum is expected to have resulted in a significant increase in erosion rates, stream sediment concentrations and sediment delivery to Moreton Bay, it should be noted that examples do occur where no change in inferred erosion rates was detectable in response to the much larger shifts in climate between the last glacial maximum and the present (e.g. O'Hara *et al.*, 1993).

In Moreton Bay, several important changes occurred in response to deteriorating climate, deteriorating catchment condition and, importantly, to the extended period of relative sea level stability. These changes, relative to the climatic optimum, had several consequences for Moreton Bay.

- Infilling of the Bay by fine-grained terrigenous sediments resulting in decreased water depth and volume, more sediment available for resuspension during high energy wind events and, because of the reduced depth, the available sediment being more susceptible to resuspension during high energy wind events. Increased turbidity is likely to have been an important factor in the late Holocene shift in coral community composition and also in the nature and distribution of fish communities in the Bay (e.g. Blaber & Blaber, 1980; Blaber, 1997).
- Progradation of mainland coasts by coarse- and fine-grained terrigenous sediments resulted in decreased water volume in the Bay and the discharge point of terrigenous input (e.g. sediments and nutrients) moved progressively closer to some ecosystems e.g. coral communities at locations such as Mud Island.
- Decreasing sea level resulted in decreased water depth and volume and increased bottom sediment resuspension, and may also have been a trigger for increased erosion of the lower reaches of streams. Opinions differ as to whether sea level fall during the late Holocene occurred gradually over the last 6 000 years (Chappell, 1983b; Lambeck & Nakada, 1990) or rapidly between 3 and 4 ka (Beaman *et al*., 1994; Larcombe *et al*., 1995).
- Decreased, and more variable, sea surface temperatures resulted in increased low-temperature stress for tropical species/communities, increased suitability for temperate biota and increased episodic/acute stress on both tropical and temperate biota due to increased variability.
- Increased intensity of the prevailing southeast tradewind resulted in increased bottom sediment resuspension and turbidity (with associated increases in nutrient transfer between sediments and the water column) and intensification of the wind-forced component of water circulation within the Bay (with implications for the distribution and duration of river plume impacts on marine and intertidal communities).
- Progradation of dune-island barrier coasts by coarse marine sediments (whether due to the sea level fall or simply equilibrating following sea level rise (see Chappell, 1991)) resulted in reduced tidal exchange. This coastal progradation is evident on the northwest coast of Moreton Island (Bulwer Swamp), the north coast of North Stradbroke Island (at Flinders Beach), and much of Bribie Island. An important part of this phase was the establishment of an offshore barrier island (South Stradbroke Island), linking the mainland to the east coast of North Stradbroke Island, probably as a result of the combined effects of two processes: (i) landward movement of an offshore bar as sea level rose, followed by its stranding with the mid Holocene sea level fall, and (ii) longshore transport of sediments forced by the prevailing southeasterly winds. These changes had a significant impact on the openings

between the Bay and the ocean and, consequently, on tidal circulation and water quality.

The important role of the openings in the east coast of Moreton Bay in determining the Bay's physical and biological seascape, as well as playing a role in the commercial and recreational opportunities available, suggests that a brief review of their history and role is warranted, particularly given the inconsistencies and confusion in previous reports.

South Passage, between Moreton and North Stradbroke Islands, has been widely regarded as a recent "breakthrough" (Welsby, 1907; Steele, 1972; O'Keefe, 1975; Clifford & Specht, 1979; Durbidge & Covacevich, 1981; McLeod, 1983), largely as a result of anecdotal evidence from Welsby and misinterpretation of the charts and journals of the early navigators (Cook, Flinders, etc.). Neil (1991) analysed these charts and journals and argued that these records could be relied upon to identify the presence of such an opening but could not be relied on to identify its absence. Stephens (1992), using a simple sediment budget approach, argued that the rate of sediment transport was insufficient to fill and close this opening during the Holocene sea level highstand. Given the geomorphic evidence for a permanent opening and the absence of reliable historical evidence for a recent closure, despite considerable support for this view in previous reports, it seems likely that South Passage has been open throughout the Holocene.

The South Passage opening is important to the geomorphology and ecology of the Bay in several ways.

- The processes and sediments necessary for the formation of the flood-tide delta, comprising the Amity and Moreton Banks, are important to the area, biomass and diversity of seagrass communities in the Bay and to the species which rely on them (e.g. sea turtles, dugongs). The entrance banks are also important in the breeding cycle of some fish species (e.g. Pollock, 1984).
- South Passage is a source of oceanic water of lower turbidity and less extreme temperatures than the waters of the Bay, important to dugongs, both directly (temperature) and indirectly (water turbidity effects on seagrasses). Oceanic waters of low turbidity and low temperature variability are potentially important to sessile organisms such as corals. This source of oceanic water lies adjacent to suitable substrate for coral colonisation, on the laterite platforms of the islands and mainland coast of the central Bay, unlike the Northern Entrance where suitable substrates are uncommon.
- South Passage is a point of entry for coral propagules (Johnson & Neil, this volume), again in the vicinity of suitable substrates for settlement. South Passage also provides access for dugongs to the seagrass beds of the eastern central Bay, important because these are the most extensive seagrass beds in the Bay, because oceanic waters are warmer than Bay waters in winter and tolerable for dugongs, and because there is a relatively low level of disturbance by boat traffic in this area (Preen *et al*., 1992).

Instability of South Passage, including variations in depth and of the orientation of deep channels (Eberhardt, 1978; Davenport, 1986), is also likely to result in temporal variations in tidal flushing and spatial variations in patterns of oceanic dispersal into the central Bay.

The history of the openings to the south of North Stradbroke Island is also important, influencing patterns of sedimentation and ecosystem development in southern Moreton Bay. Through its influence on the circulation of oceanic waters and the fate of Logan River flood waters, the history of the southern openings is probably intimately linked to the collapse of the coral communities in the Macleay, Perulpa and Lamb Island and Pelican Banks areas.

Using essentially the same arguments, based on historical sources, as for the South Passage, Neil (1991) argued that there is insufficient evidence to determine whether the Southport

Bar (South Passage on some early maps and now the Gold Coast Seaway) is only a recent (since 1800) phenomenon or was open, perhaps episodically, prior to that time. However, it is likely that this opening was much less stable than South Passage, and its position has varied considerably since the closure of the northern spit of South Stradbroke Island with the east coast of Stradbroke Island to form the Eighteen Mile Swamp about 600 years ago (Grant *et al.*, 1985). Human activities (notably construction of the Gold Coast Seaway) have altered this entrance considerably and some of the effects are described briefly below.

The opening at Jumpinpin Bar is variously reported as occurring in 1895, 1896 and 1898. Several writers have suggested a role for human activities in this event, viz. detonation of cases of explosives from the wreck of the *Cambus Wallace* which "made huge gaping wounds in the sand" (Welsby, 1907) and trampling of the dune vegetation during salvage operations (e.g. Durbidge & Covacevich, 1981; Salter, 1984). In this context, it is also worth noting that the salvaged cargo of the schooner *Bellinger*, wrecked at Jumpinpin in April 1892, was dragged across the dune ridge, at that time reported to be 35 feet high, to Swan Bay (Welsby, 1907), the area was "for many years a favourite camp for boating men (Anon., 1898) and Welsby's (1907) photograph of 'Jumpin Pin ... before the Break Through' shows small 'blowouts' in the dune ridge, apparently used for access from Swan Bay to the ocean beach. On the other hand, Hanlon (1935), who was aware of the *Cambus Wallace* wreck, notified the police of its occurrence, and was present at the wreck site the morning after it occurred, makes no mention of the role of the explosion in the Jumpinpin breakthrough. He says (Hanlon, 1935) ".. I myself saw the break-through at the narrow neck of Jumpinpin, during which the ever-encroaching seas seemed to melt the sand, with standing scrub on it, as though it were sugar – large areas collapsing in one sweeping surge". The Jumpinpin breakthrough appears to have developed progressively over several years. In his September, 1895 report, Almond (1895) observed that "..during the easterly gales experienced in the early part of the year the sea made a breach of about 350 yards in extent through Stradbroke Island; this has since extended and at high water spring tides the water now washes over into Swan Bay. Should the erosion continue Stradbroke Island will be divided..". The breakthrough was completed in May 1898 when "during the recent gales...a channel has been cut right through Stradbroke Island at Jumpinpin...the break is about 700 yards wide ... and the sea is now breaking right through onto the mangrove islands inside the Bay ... [and]...a bar is forming outside at Jumpinpin... (Anon., 1898).

As a result of the breakthrough, and consequent decreased tidal flows through the opening at Southport, sedimentation in southern Moreton Bay increased markedly. Swan Bay was turned from a "deep-watered fishing haven..[to]..a shallow expanse of water" (Horton, 1983), a mere "gutter through drying banks" (Bell, 1975; "..where it was once deep and blue water, the seaweeds now show on the surface at low tides.." (Hanlon, 1935)). Diminished tidal flows caused shoaling of the southern Broadwater and, by 1905, the Southport Bar had become so shallow that "... a person could easily walk from head to head..." at low tide (Davenport, 1986). Bell (1975) suggested that the oyster banks between Jumpinpin and Southport, including Tippler's Never Fail Island banks, were wiped out at the time of the Jumpinpin breakthrough. On the other hand, Horton (1983) states that the oyster banks in this area were destroyed during the 1893 flood, and that "..Tippler's Never Fail Island escaped." Salter (1984) confirms the effects of both flooding and the breakthrough on the oyster banks, noting the effects of the January 1887 flood and a long campaign (1904-1936) by the oystermen, supported by the Southport Chamber of Commerce, to have the Jumpinpin entrance closed. Kelley (1984) published notes from E.F. Darcy (from 1940) which suggest closure of Jumpinpin in 1917 and a subsequent breakthrough in October 1936. Natural change in the position of the opening at Southport (gradual northward movement forced by net northward sediment transport) also altered geomorphic and ecological conditions within the Broadwater. For example, the area near Narrow Neck, formerly a deep hole known as Shark Bay, was replaced by a "..quagmire... a repulsive mudflat" (Hanlon, 1935).

In summary, it appears likely that the opening between North Stradbroke and Moreton Islands has been open throughout the Holocene, although progressively constricted by progradation of the adjacent islands and deposition of the flood-tide delta and offshore bar. South of North Stradbroke Island, a wide opening was present until the development of a barrier island (South Stradbroke Island) at the time of sea level fall between 3 and 4 ka. Closure of the northern end of this barrier occurred about 0.6 ka, and the barrier may have been quite unstable since then.

During the 6-0.2 ka period, the lower reaches of the rivers were becoming less estuarine and more riverine as sediment infilling and channellisation occurred. Similarly, the Bay ecosystems were becoming less oceanic and tropical, and more estuarine and temperate. The physical environment was changing in such a way as to make it less suitable for some biological activity (e.g. whales, corals) and more suitable for others (e.g. mangroves, seagrass, saltmarsh). For some species these changes could be seen as a "mixed blessing". For example, formation of the South Passage flood-tide delta provided a suitable substrate for colonisation by seagrasses to form the most important habitat for dugongs in the Bay, but they were increasingly restricted in their ability to use it as a result of decreasing temperatures in an area which is now at about the limit of their temperature tolerance. Most of the present seagrass areas can only have formed progressively throughout the late Holocene, although in the past many areas of the Bay may have been colonised by seagrasses at depths much greater than at present due to greater water clarity.

Anthropogenic change

There are two, diametrically opposed, models current which purport to explain the impact of Aboriginal people on the physical environment and ecosystems of Moreton Bay. In the mould of the "noble savage" is the popular view, espoused by Baker (1987), that the "natives lived in harmony with the land, sharing it with animals, plant life and the spirits". By contrast, Walters (1989) suggests a chain of events commencing with forest burning in the catchment, increased erosion rates, and increased sediment delivery to the coast "which would have given rise to large areas of mud and sand flats covered with shallow turbid waters and seagrass beds, permitting the evolution of fish stocks on a scale which today form the basis of large contemporary commercial fisheries". Their success in harvesting "rich fish stocks .. was an unforeseen consequence of the ... activities of their "firestick farming" (Jones, 1969) ancestors in the hinterland" (Walters, 1989). Hall (1990) also supports this model, suggesting that their intensified catchment land use was responsible for "changing the ecosystem to one more suitable to their needs".

Neither the model of Baker (1987) nor that of Walters (1989) and of Hall (1990) is consistent with geomorphic processes in the catchment and Bay, and reality probably lies somewhere in between. Walters supports his argument by analogy with Hughes & Sullivan's (1981) suggestion that late Holocene sedimentation occurred in response to anthropogenic burning. Evidence for this includes sediment infilling occurring at a time of climatic and sea level stability, and "anachronistic phasing" (i.e. asynchronous erosion and sedimentation events). Several arguments can be made against these conclusions.

1. Walters postulates Aboriginal burning as the "ultimate cause ... [of] ... sedimentary loads washed from coastal streams" during the late Holocene. However, catchments yield sediment under natural conditions. "Firestick farming" can only increase, not create, catchment

sediment yield.

2. "Anachronistic phasing" occurs naturally and does not require an anthropogenic trigger (Prosser, 1988; 1989; Taylor & Lewin, 1997).

3. Anthropogenic burning has probably been in the Australian landscape much longer than the Holocene; tens to hundreds of thousands of years (Singh & Geissler, 1985; Kershaw, 1994) rather than thousands. Hughes & Sullivan's (1981) argument was based on apparent intensification of Aboriginal land use during the late Holocene, which has been reported widely throughout eastern Australia (e.g. Ross, 1985). However, Walters' (1989) data (Figure 2) suggest that the increase in the number of dated archaeological sites between the period 5 ka-0 ka and 8-5 ka (by a factor of 2.7) is about the same as the increase between 10-8 ka and 8-5 ka (a factor of 2.9). This pattern is consistent with westward migration of humans in response to a receding coastline throughout the Holocene, analogous to the events described in north Queensland during the same time period by Hopkins *et al.* (1996), rather than a late Holocene intensification of land use. It is also likely that at least some of this increase may be due to greater site visibility, although Ross (1985) presents a case against this. The very large increase in the number of coastal sites (from c. 2 ka) is largely attributable to increased exploitation of the fisheries resources of the Bay (Walters, 1989; 1992), and is unlikely to have significantly altered the catchment's impact on the Bay.

4. Accounts of the early explorers suggest that the grasslands, those areas most frequently burned, were limited in area and, furthermore, were generally on plains rather than hillslopes, thereby minimising the impact of burning on erosion rates. Although these accounts contain frequent mention of grassland and lightly wooded areas, very few of these areas are described as having been recently burned and the journals of the early explorers of the Moreton Bay district also indicate that much of the grass cover was 1-2 m high, which would have provided good ground cover. Furthermore, the general climatic characteristics of the catchment are consistent with a setting for natural vegetation ranging from rainforest to grassland (Figure 3) along a rainfall gradient locally influenced by soil characteristics. The pattern implied by this simple diagram is generally consistent with recent mapping of pre-clearing vegetation in southeast Queensland (Bean *et al*., 1998) which indicates that open forests dominated east of about Ipswich. To the west of Ipswich, the vegetation consisted largely of open forest and woodland with several extensive areas of vine thicket. Throughout the region, the open

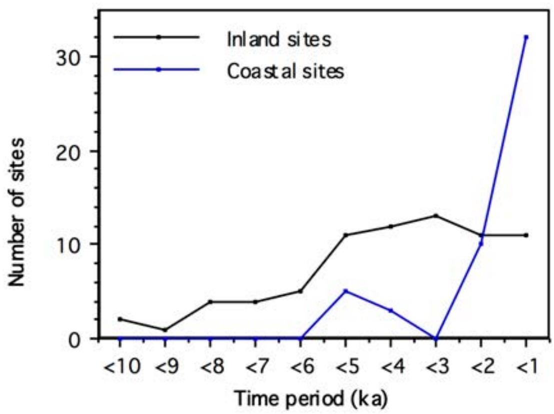


Figure 2. Number of dated archaelogical sites in the southeast Queensland region during the Holocene (data from Walters, 1989).

forest on alluvium generally had an understorey of rainforest species and/or sclerophyllous shrubs.

Braithwaite (1991) argues that, in the monsoon tropics, Aborigines manage their fires, preventing their escape into areas which they have no wish to burn, and Hughes & Sullivan (1986) suggest that Aboriginal burning was largely restricted to "localities which were foci

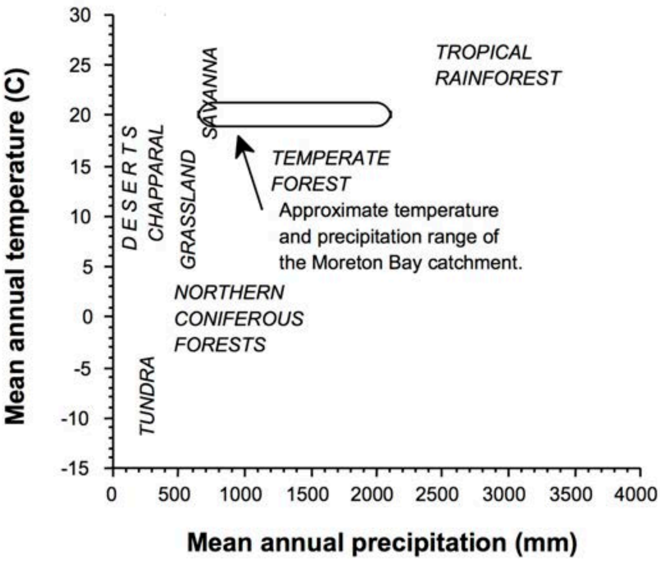


Figure 3. The relationship between mean annual precipitation, mean annual temperature and the major terrestrial biomes, showing the climatic characteristics of the Moreton Bay catchment (After Begon *et al.*'s (1990) modification of Whittaker, 1975).

of Aboriginal economic activities". This is consistent with Leichhardt's (1843a) observation that "only small patches" were burned in southeast Queensland, which contrasts with his observations of extensive burning west of the Great Divide (Leichhardt, 1847). Gresty (1947; 1951) also noted that Aboriginal fire management involved "..only small sections being burnt amidst larger conserved areas". Furthermore, change in species composition in areas of low frequency burning does not necessarily imply a change in forest structure or loss of ground cover. Thus, the geomorphic effects on the catchment are uncertain but would probably have been minimal, a conclusion consistent with the findings of Tulau's (1996a) recent review of the geomorphic effects of Aboriginal land use in eastern Australia.

A simple example may provide some indication of the impact of burning on the catchment. It seems reasonable to assume that:

(i) 10% of the catchment was converted to grassland through anthropogenic burning (50% greater than the area of cropland at the present and several orders of magnitude greater than the estimate of grassland area of Bean *et al.* (1998)), and

(ii) conversion to grassland results in a long term increase in catchment sediment yield by a factor of two. This is about half the factor of increase measured by Neil & Fogarty (1991) for open forest to grassland conversion, but seems appropriate because, in this case, the resulting grasslands are not then trampled by livestock. It should also be noted that the very high sediment yields resulting from burning alluded to by Hughes & Sullivan (1981) are likely to have been both short lived and predominantly organic sediment (e.g. Blong *et al.*, 1982). Furthermore, high intensity fires have much greater erosion and sediment yield consequences (Humphreys & Craig, 1981) than do the low intensity fires associated with Aboriginal burning.

Given these assumptions, it follows that anthropogenic burning increased catchment sediment yield by no more, and probably much less, than 10%. Although this is a crude estimate, it is at least consistent with observations (i), (ii) and (iii) above, and with the known increases in sediment yield associated with "European" land use intensification. Contrary to the model proposed by Walters (1989) and by Hall (1990), it seems highly unlikely that intensified Aboriginal land use was responsible for "changing the ecosystem to one more suitable to their needs". Rather, Aboriginal communities may have benefited by ecological changes in the Bay which were occurring naturally.

Regardless of the cause, natural or anthropogenic, of the ecological changes which occurred in the Bay in the late Holocene, it seems likely that a shift in Aboriginal subsistence patterns was a likely consequence. Alfredson (1984) also suggests that the changes in the archaeological record at St Helena, from fish and bat bone in the lower (older) strata to shell in the upper (younger) strata, may have a been a consequence of a three-fold reduction of the distance from the mainland to the island. This change, caused by progradation of both the mainland and island coasts, reduced the risks associated with accessing the island's resources, shifting exploitation from "specialist parties of experienced adults" to "a more generalised pattern of foraging". Alternative, although not mutually exclusive, explanations for this change in the archaeological record include an increase in regional population (Alfredson, 1984) and a change in the available resources, i.e. increased shellfish stocks as progradation increased the intertidal and shallow subtidal area. The pattern of increased resource accessibility as a result of geomorphic change, suggested by Alfredson (1984), is likely to have occurred throughout Moreton Bay.

The catchment at 200 years ago: country of "the richest description"

An indication of the characteristics of the catchment at the time of European settlement can be obtained from the journals of the explorers. Some caution is needed in the interpretation of these works due to factors such as navigational uncertainties, inconsistencies in terminology, and ulterior motives in their landscape descriptions. (Finlayson (1984) provides an excellent example of Mitchell's need to discover good country strongly biasing his description of the landscape of Lake Salvator in central Queensland). In order to explore this landscape, we will start by travelling upstream along the Brisbane River.

As far as the downstream end of the Hamilton Reach, the river banks were lined with mangroves (four species), referred to by Lang (1861) as "a forest of gloomy mangroves ... cheerless vegetation." Upstream of the mangroves "the soil and scenery on the banks of the Brisbane rapidly improve" (Lang, 1861), giving way to specimens of *Hibiscus heterophyllus* and patches of forest (*Eucalyptus, Callitris, Tristania* [*Lophostemon*] species) similar to that of the adjacent ridges. Breakfast Creek drained a "reedy swamp". The country to the north and south of the lower reaches of the river was described as "Fine open grazing country" and "Open country, hills stony but well covered with grass", respectively (Stirling, in Steele, 1972). John Sweatman travelled up the river to Kangaroo Point in January 1846, and observed that "...the scenery along the banks exceeded anything I had yet seen in Australia ... the bush was rather thick and the banks intersected with numerous small creeks which serve as a resort for swarms of wild duck & plover, but afterwards you pass tracts of beautiful open forest land which remind you rather of a gentleman's park than of wild uncleared bush" (Allen & Corris, 1977). From the vantage point of Mt Coot-tha, "the view from southeast to northwest was extensive and very grand, presenting an immense, thinly wooded plain, whose surface was gently undulating, and clothed with luxuriant grass". To the north was "...a tract of lofty and forest covered hills, interspersed with extensive districts of *Araucaria*..." (Fraser, in Steele, 1972). The "rich flats" (Stirling, in Steele, 1972) at South Brisbane covered with a "...tangled mass of trees, vines, flowering creepers, staghorns, elkhorns, towering scrub palms, giant ferns, beautiful and rare orchids and the wild passion flower, while along the river bank were the water lily in thousands and the convolvulus of glorious hue..." (Convicts' reminiscences, *Brisbane Courier* March 1830, in Johnston, 1988; Gregory, 1996). There was a "chain of ponds watering a fine valley" at Milton (Oxley, in Steele, 1972). Similarly, the Coopers Plains/ Archerfield area consisted of "...excellent land, thinly wooded ... [with] ... a beautiful chain of ponds" (Fraser, in Steele, 1972). The south bank of the river, to Canoe (Oxley) Creek, was "...covered with forests of Pine or Aracauria to a considerable extent. The north bank ... is principally open forest, not reaching very far, beyond which it is clothed with pine brushes" (Fraser, in Steele, 1972). At Long Pocket and upstream of Seventeen Mile Rocks the banks consisted of mudflats vegetated by *H. heterophyllus* and *Casuarina* sp.

Several kilometres upstream of the Bremer confluence, Oxley (in Steele, 1972) described "...a very thick brush abounding with stately and magnificent pines, which towered far above the other timber of the hill ... an entirely new species of the genus *Araucaria* ... decidedly the growth of the interior, and not a coast tree" (*Aracauria cunninghamii*). "It was totally impossible not to halt a few moments to admire this noble tree..." (Cunningham, in Steele, 1972) of which there were "...endless quantities..." (Brisbane, in Gregory, 1996); "Pines" were common on both banks upstream of the Edenglassie (now Brisbane CBD) settlement (Lockyer, in Steele, 1972). About 5 km upstream of the Bremer confluence, Lockyer (in Steele, 1972) described "...delightful, thinly wooded [country] with fine pasturage for any number of cattle." Lockyer recognised the potential for severe flooding of the river with "drift grass and pieces of wood washed ... up into the branches of the trees – Marked the floods to rise here upwards of one hundred feet" (Steele (1972) notes that the recorded 1893 flood height at Mt Crosby pumping station was 94 feet 10 1/2 inches).

Between the confluences of Lockyer Creek and the Stanley River with the Brisbane River, Lockyer described extensive "level country, fine appearance, thinly wooded" with "hills of pines very thick" to the northwest. Water in the streams flowing into the Brisbane upstream of the Stanley confluence was of "excellent" quality. The terrain of the upper reaches of the Bremer was timbered by various *Eucalyptus* and *Angophora*, of "beauty and fertility...[a]... beautiful vale ... excellently watered, and fit for any purpose to which it may be applied" (Logan, in Steele, 1972).

Much of the lowlands of the lower Logan catchment was swampy terrain, often "timbered with forest oak". Further upstream were abundantly watered grasslands, open forest, and extensive scrubs, including "pine scrubs". The upper Logan catchment included large swampy plains and scrub covered slopes (Logan, in Steele, 1972). Some of the plains were dry but well watered by chains of ponds (Fraser, in Steele, 1972).

"From near its source to where it meets the tide the Brisbane [River] retains the same character having a wide gravelly bed with *Casuarina* growing over all these parts not usually covered with water (Burnett, in Gregory, 1996). Cunningham also refers to "the extensive banks of alluvial sand and gravel, with which the channel of the Brisbane is alone* occupied" (*Steele's footnote – "That is, no mud"). Although the banks within the channel may have been exclusively sand and gravel, the banks of the river in the Seventeen Mile Rocks area and at least as far downstream as Long Pocket consisted of mudflats (Oxley, Cunningham in Steele, 1972). Later reports also describe sand beaches at various locations along the lower reaches of the river (e.g. Mandalay, Chelmer and South Brisbane (Gregory, 1996)).

The journals of the early explorers also provide some indication of the extent of anthropogenic burning in the catchment at the time of European settlement (e.g. Cunningham, Lockyer, Logan in Steele, 1972), although any meaningful quantification of this is difficult (see Fensham, 1997), particularly given that burning was often a response to the presence of whites. Fensham (1997) gives examples of accounts of fire being used as a defence against whites (e.g. Leichhardt, 1847), as a means of signalling (Banks in Beaglehole, 1962) and as a mode of attack (e.g. Mitchell, 1848; Cunningham in Steele, 1972; Hann, 1982). Cunningham describes such an attack at Laidley Creek: "... the eldest native set fire to the grass ... ran along the outskirts of the plain, carrying a fire-brand and igniting as he went; the others did the same in an opposite direction ... in a few moments ... an extensive line of flame appeared to windward of us, making rapid advances, towards our little encampment ... [Despite an attempt at burning a fire break] the enemy's flames were with an appalling noise and rapidity approaching us; the columns of black smoke that were driven before the wind almost stifled us, whilst the red hot flakes of burnt stubble flew about us in a most terrific manner ... [however] ... (as by a kind interposition of Providence) the wind veered round more to the Southward and Eastward, and freshening, blew the body of flame past us into the creek ... All was tranquillity again, with the exception that occasionally the natives ... gave us a yell of disappointment, that their diabolical purpose had been thus defeated" (Cunningham in Steele, 1972).

Recently, several authors (e.g. Rolls, 1981; 1994; Flannery, 1994; Ryan *et al.*, 1995) have argued that the Australian landscape, before the coming of the Europeans, was grassland with few trees, that much of the forest now present is due to regrowth subsequent to the cessation of Aboriginal burning, and that modern forest clearing is simply reestablishing the vegetation of 200 y ago. For example, Rolls (1981) states that "Australia's dense forests are not the remnants of energetic clearing, they are the product of one hundred years of energetic growth", "Australia was not a timbered land that has been cleared" (Rolls, 1994), and Ryan *et al*. (1995) suggest that "...much of the tall mixed eucalypt forest with a developed understorey that is a common occurrence these days did not exist at the time of European settlement...the first settlers did not set to and ringbark huge areas. There was no need to. Most of the country was woodland, grassland, savannah and open forest". Furthermore, "the widespread ringbarking that was carried out around the turn of the century was mostly ... regrowth. The landowners were attempting to re-establish the original grazing capacity". A contrary view is put by Norton (1886), whose observations largely relate to the New England Tableland in the 1850s. He notes that "not only have unserviceable classes of trees been intentionally destroyed, but thousands of acres upon which were many of the most valuable eucalypts have been ring-barked, and scarcely a living specimen can be seen in some places. This is the deliberate work of men who persuade themselves that they are vastly improving the country... this artificial mode of destroying has been so extravagantly carried out...". Biases in reporting on the characteristics of the landscape are likely to have arisen because open country was easier to traverse, Aboriginal burning generally concentrated on alluvial flats adjacent to streams which, again, were easier to traverse and provided a source of water, expeditions were often funded by pastoral interests and therefore sought out open country and, in small colonies struggling to attract free settlers and investment, favourable descriptions of the agricultural potential of the landscape were used to promote development (see Tulau, 1996b; Benson & Redpath, 1997).

The controversy and conflicting views are not surprising given the nature of the historical record. For example, in two consecutive paragraphs Gregory (1996) cites an 1845 report of the surveyor Burnett: "The whole of the country on the Brisbane is very thinly wooded forest furnishing a rich and abundant pasturage, patches of scrub intervening in one or two places...", and an 1842 article in *The Australian* newspaper: "...the Brisbane River...traverses a large extent of beautiful country, exhibiting all the luxuriant features of tropical vegetation...". An excellent analysis of the conflicting hypotheses is provided by Benson & Redpath (1997), who use explorers' accounts supported by scientific evidence (e.g. palynology, dendrochronology, geomorphology) to conclude that "... the main causes of change to Australia's vegetation since European settlement have been large-scale clearing and cultivation of land". This also appears to be the case for the Moreton Bay catchment where, although there is limited scientific evidence, the journals of the explorers and the early surveyors' reports (e.g. see Watson, 1988) show clearly that the vegetation was a mosaic of rainforests (which were preferentially cleared (Pettigrew, 1877)), open forests, woodlands and grassland. Bean *et al*. (1998) conclude that only 0.01% of the pre-European vegetation was grassland. The general pattern in southeastern Australia appears to be one of forested or wooded hillslopes with anthropogenic grasslands confined to areas of the alluvial flats (Tulau, 1996b; Benson & Redpath, 1997) which is broadly consistent with long-recognised influences on Aboriginal land use patterns (Peterson, 1973; Meehan, 1982), i.e. resource proximity (vegetable staples, hunting grounds, water) and comfort (e.g. relatively insect free). In the Moreton Bay catchment the explorers' accounts also show that some hillslopes were grassland and that dense forest (scrubs, brushes) was common on the alluvial flats. In Leichhardt's (1844) words: "When you enter the basin of the Brisbane ... [River, from the west]... you are at once aware of the greater vigour of plant growth. Trees are taller and they grow closer together. The flanks of the mountains and the banks of streams and of the river are overgrown by an almost impenetrable brush [rainforest]". "The country between the coast range and the sea ... [from Newcastle to the Moreton Bay district] ... has the advantage of plenty of water, though the rather dense forest and the abundance of scrubs renders them less fit for sheep farming" (Leichhardt, 1843b).

200 years ago to the present: European settlement and environmental degradation

Degradation of the catchment

The problems of extremes of climate and land degradation in Australia were visible and recognised very early in the country's European colonisation. Land degradation, in the form of soil fertility decline, was recorded in the Sydney area as early as 1798 (Collins, 1804). The drought of February 1799 led to ponds becoming "..brackish, and scarcely drinkable ...from which... it was conjectured that the earth contained a large portion of salt" (Collins, 1804). Collins also reported floods and bushfires. In 1810 the *Sydney Gazette* and *New South Wales Advertiser* published regulations to alleviate the problem "that the stream of water which flows through the Town of Sydney, and the Tanks which have been constructed thereon ... for the purpose of procuring an adequate Supply of pure and good Water ... are frequently polluted and rendered totally unfit for those valuable purposes..." (Aplin, 1998). Robertson (1853) describes drainage lines in western Victoria which were rarely entrenched and carried a good cover of perennial tussock grass. Within ten years of settlement (about 1841) overgrazing had resulted in perennial grasses being replaced by annual species, soil exposure, saline runoff and gully incision to 3 m deep. Clarke (1860) observed that "... the deepest injury that could be inflicted...[on the Southern Tablelands of New South Wales]... would be the introduction of the system of swamp drainage which obtains amongst the agriculturalists of Europe. ... I do not know what this beautiful and well-watered country would do if any attempt should be made to drain the swamps and boggy places which so often occur...". Darwin (1845) described the "... sirocco-like wind of Australia ... Clouds of dust were travelling in every direction; and the wind felt as if it had passed over a fire". Sturt (1833) stated that "...the ground on both sides of the [Macquarie] river looked bare and arid" due to overgrazing by cattle; Mitchell (1848) refers to overgrazing by sheep to the extent that "...not a blade of grass could be seen..."; and Trollope (1873) observed that "...salt-bush was disappearing on runs which had carried sheep for many years, ...it certainly receded as the squatters advanced...". Norton (1886) noted the soil compaction, changed infiltration and runoff regimes and eucalypt dieback which followed forest clearing and "excessive" stocking with sheep on the New England Tableland. Clearly, land use intensification in Australia rapidly brought about ecological and geomorphic changes, capable of increasing erosion and sediment yield rates, with consequences for both terrestrial ecosystems and downstream aquatic and coastal systems.

Furthermore, Meredith's (1840) observation that "the system of clearing... [around Sydney] ..., by the total destruction of every native tree and shrub, gives a most bare, raw, and ugly appearance to a new place. In England we plant groves and woods, and think our country residences unfinished and incomplete without them; but here the exact contrary is the case, and unless a settler can see an expanse of bare, naked, unvaried, shadeless, dry, dusty land spread all around him, he fancies his dwelling 'wild and uncivilized'..." suggests that not all environmental degradation in Australia was a consequence of emulation of European land management practices. On the other hand, settlers may simply have been responding to local environmental conditions, e.g. high fire hazard.

In the Moreton Bay catchment, the situation was little different from elsewhere in Australia. In 1824, Oxley (in Steele, 1972) remarked on the uncertainty of "the rules governing the operations of Nature" in Australia, and recognised the problem of drought in the southeast Queensland region. Lockyer (in Steele, 1972) reported the evidence for high magnitude floods in the Brisbane River and surface crusts of salt were reported by Cunningham (in Steele, 1972) in the Lockyer Creek catchment in 1829.

It seems, therefore, that European land management in Australia was based not on ignorance (not knowing, unaware), but on ignore – (to pretend not to see, to set aside) -ance. In spite of the warning signs from elsewhere in eastern Australia, destructive exploitation of the resources of the Moreton Bay catchment proceeded at a rapid rate. The Moreton Bay penal settlement was shifted from Redcliffe to Brisbane in early 1825 and by 1835 there were "no spars within 20 miles"; all the easily available timber had been cut (Johnston, 1988). Jardine (1873) complained that the "Moreton district which, sixty years ago, was covered with valuable red cedar, cannot now boast a single stick of that species". The Government paid a bounty for land clearing and the "...pine and cedar were easily cut, and as easily burned...". The settlers targeted particular landforms and vegetation communities, notably the pine (*Araucaria cunninghamii*) scrubs on the fertile valleys and plains which lay close to the waterways (Webb, 1966; Catterall & Kingston, 1994). "Where this pine grows on anything like level ground farmers know they can grow maize, sugar cane etc.; and therefore many pine scrubs are cleared for cultivation, to the injury of the country..." (Pettigrew, 1877). The clearing of the rainforest scrubs also had its effect on soil fertility which was recognised early – "even the rich scrub soils of Queensland can be exhausted..[and this is]..most observable in scrub farms eight or ten years in cultivation, and from which the roots have all rotted away.." (Anon., 1871). A dubious form of land management in the Brisbane area was "trenching" which involved "..digging down the soil and covering it with the subsoil.." (Pettigrew, 1862). During the 1840s pastoralists moved flocks of sheep from the Darling Downs into the western parts of the catchment which came under increasing grazing pressure, with at least ten stations established in the western catchment by 1842 and 17 by 1844 (Craig, 1925). By the 1850s, Pettigrew (in McLeod, 1990) was reporting that much of the ground in the Brisbane Valley was eaten bare by sheep, and erosion was becoming evident. The cedar scrubs which grew on the banks of the Albert River in the 1840s were cut; the scrubs which remained were swept away in the 1864 flood in which the channel was widened and the river's course altered (Perry, 1923). The channel instability and alteration may, itself, have been a consequence of the previous partial clearing of the littoral scrubs. In the Numinbah Valley (upper Nerang River catchment), there was "ruthless and ... indiscriminate exploitation of the red cedar" by one settler which was "still an unpleasant memory among the descendants of the pioneer timber-getters" seventy years later (Gresty, 1947). Tully (1881) noted that, although timber reserves had been established in the Moreton District, "efforts are continually being made to declare these areas open to selection ... and sometimes these requests are backed by Parliamentary influence." Timber licences, granted under defined conditions, "virtually empower the license holder to cut down as much timber as he likes ... the present regulations encourage greed, and the result is that a large quantity of valuable timber is wasted every year" (Tully, 1881). By 1880, 20% of scrubland in the district was under cultivation and nearly all of the alluvial scrubs had been selected (Smith, 1881).

Lang (1861), following a visit to the Moreton Bay settlement in 1845, was critical of mismanagement of the resources of Moreton Bay and cited several examples.

• "on most of the rivers that fall into Moreton Bay, the cedar has been long since cut away ...[to provide work for convicts] ... large quantities of that timber were actually piled up and left to rot on the beach at Dunwich, Stradbroke Island, after all the labour that had been thrown away procuring it."

• overseers were paid an allowance per acre of land cleared; to maximise this allowance "...it was only necessary to select thinly timbered land, without reference to its quality...[so, areas of Moreton Island]... a mere collection of sand-hills of no use whatever for cultivation, but thinly covered with cypress pine trees, was cleared by the convicts. The timber, which would now have been very valuable...[was]...destroyed.

• "A swamp on the Brisbane River ...was drained ... for the growth of rice, and... sown accordingly; but instead of sowing the grain in its natural state of paddy, it was sown in its manufactured state of rice, procured for the purpose from a merchant's store in Sydney! ... Of course the settlement was proclaimed unsuitable for the cultivation of rice."

During the first decades of settlement, climatic extremes affected production in the Moreton Bay settlement. In 1828 drought resulted in crops which "entirely failed". The next ("thriving") wheat crop was destroyed by blight. In January 1831, wind and hail damaged crops and gardens, followed by frost damage later that year. Harvests were down as a result of drought in 1833 and heavy rain ruined the 1841 wheat crop (Johnston, 1988).

In spite of the extent of the degradation of the landscape during the first few decades of settlement, greater intensification of land use was to occur during the final decades of the 19th century and the first decades of the 20th. Although sheep grazing in the coastal catchments collapsed to negligible numbers by about 1880 (Figure 4), cattle numbers continued to increase. Market size limited the expansion of cattle grazing until the introduction of refrigeration. Livestock numbers then increased dramatically from 1900 to about 1920 and have been reasonably stable since then (Figure 4). Associated with this pattern is a marked expansion of sown pastures, which continued from 1900 to the late 1950s (Figure 5). Neil & Fogarty (1991) have shown that in southeast Australian sheep pastures, soil loss is greater than for native pastures, largely due to the disturbance of the soil during preparation and planting and, in some cases, to a lower tolerance of drought. The area of cropland in the Moreton Bay catchment increased markedly during the period 1895-1925 and has remained reasonably stable since that time (Figure 5). Fertiliser application increased during the 1920s and rather dramatically in the period 1960 to 1990 (Figure 6), the bulk of fertiliser applied being nitrogenous.

The implications of these changes for the condition of the catchment are, in quantitative terms, unclear. There is abundant evidence to indicate that land use intensification to grazing, particularly using sown pastures, increases catchment sediment yields, often by a factor of about four to five. Cropping results in greater increases, from one to two orders of magnitude

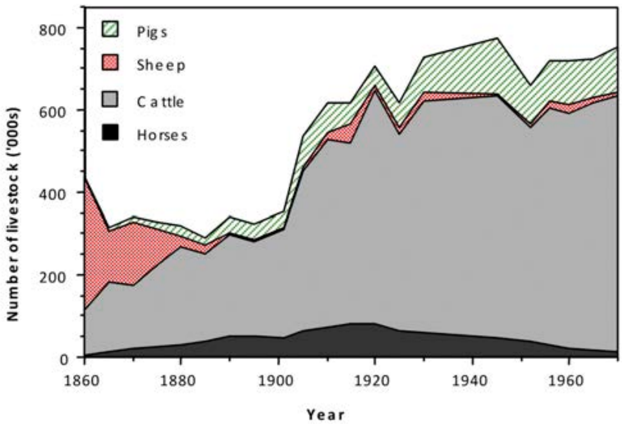


Figure 4. Time series of livestock numbers in the Moreton region (5-y intervals; data from Statistics of the colony of Queensland, Statistics of the State of Queensland and Australian Bureau of Statistics, various years).

depending on the soil, topography and crop type and management. When the changes in land use also result in changes in landforms (e.g. formation of gullies and of salt scalds) the increase in soil loss may be much greater (e.g. Neil & Fogarty, 1991). In southeastern Australia, land use intensification in and adjacent to stable, swampy valley floors with drainage lines characterised by chains of ponds (Eyles, 1977a; b), resulted in gully incision and markedly increased sediment yield. However, incision of these valleys also occurred during the late Holocene (Prosser, 1988; 1989), unaided by European land management. There has been insufficient research in the Moreton Bay catchment to adequately characterise the total landscape response to land use intensification which is likely to have varied along the climatic gradient from the humid coastal catchments to the drier catchments in the west.

The problem of dryland salinity is common in the Bremer and Lockyer catchments (Johnston, 1979; Shaw, 1979). Soil exposure in salt scalds results in high sediment yields (Neil & Richardson, 1990; Neil & Fogarty, 1991). Occurrence of dryland salinity is mediated by climate. High rainfall in the catchment in the 1950s led to rising saline groundwater levels and consequent salinity outbreaks with exposure of bare, erodible soil (Godfrey & Neil, 1993; 1994). However, the contribution of salt scalds to sediment yield from the catchment remains unquantified.

Concern over increasing sediment concentrations in the lower reaches of the Brisbane River led to the commencement of water quality monitoring as early as 1917. The source of the sediments was attributed to upstream transport of sediments resuspended by wave action in

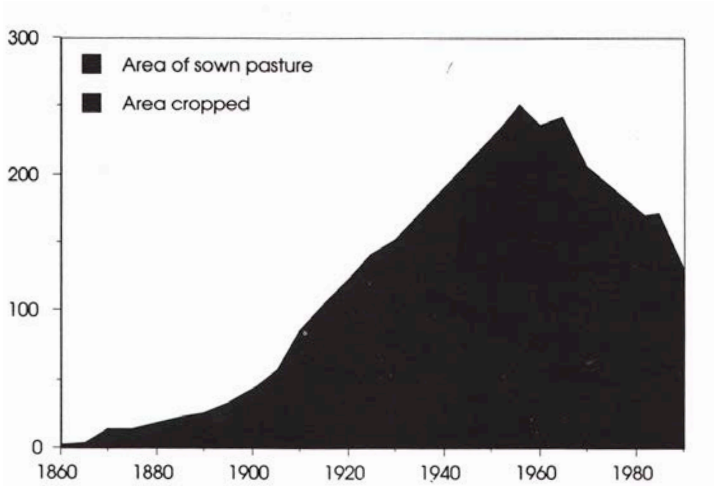


Figure 5. Time series of the area under crops and under sown pasture in the Moreton region (5-y intervals; data from Statistics of the Colony of Queensland, Statistics of the State of Queensland and Australian Bureau of Statistics, various years).

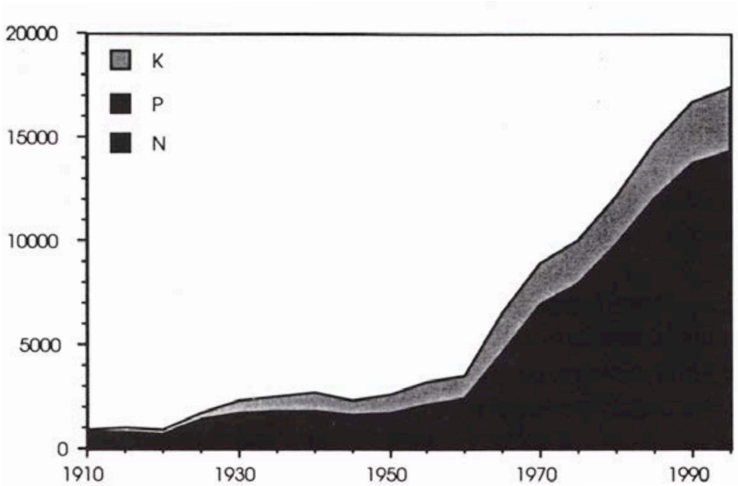


Figure 6. Time series of fertiliser input in the Moreton region (5-y intervals; data from Pulsford & Rayment, pers. comm.).

the western Bay (Cullen, 1917). Human activities are implicated in this problem in two ways. Firstly, dumping of large quantities of dredge spoil from the lower reaches of the river in the western Bay would have increased the availability of this sediment and, secondly, dredging of the river also facilitated the transport of suspended fine sediments upstream. The worst water quality was in the Town Reach, apparently as a consequence of discharges from Brisbane Town (Cullen, 1918). The concern was high sediment concentrations and loss of water clarity; there was "...no question of pollution of the river..." (Cullen, 1918). Although intended as a long term monitoring program, sampling was apparently discontinued within three years.

The land use history suggests that the critical period for sediment impacts on the Bay was probably the first few decades of this century, although this conclusion is based on the rural land use history alone and Stock & Neller (1990) present strong anecdotal evidence for a marked increase in turbidity in the river since the 1940s. The impacts of urbanisation and of river dredging are likely to have been significant contributors to this deterioration (Stock & Neller, 1990).

The early part of this century would also have seen an increase in nutrients associated with the eroded soil. However, the greatest impact of diffuse-source nutrients would have been the period since 1960, the effects of which would have been exacerbated by the marked increase in sewage discharge over the same time period. Clearly, the foregoing indicates the potential of the catchment to yield sediments and nutrients, but actual discharges of diffuse-source sediments and nutrients from the catchment to the Bay are largely determined by extreme rainfall events.

Estimates of the changes in sediment yield from the catchment, in response to both climatic deterioration following the climatic optimum and as a consequence of European settlement are presented in Figure 7. These estimates, based on a simple model for Queensland coastal catchments (Neil & Yu, 1996), suggest that mean flow-weighted sediment concentration in the Brisbane River increase from about 90 mg/L to 150 mg/L in response to deteriorating late Holocene climate, and to 525 mg/L as a consequence of land use intensification following European settlement. These are necessarily crude estimates, with some limitations in relation to both scale and process, but provide an approximation of the contrast between the effects of natural changes during the late Holocene and changes during the last 200 y due to the effects of modern agriculture. Validation of these estimates is difficult. Stephens (1992) calculated an average sediment concentration in the Brisbane River over the last 6 500 y of about 130 mg/L, based on mud accumulation rates in the delta. The estimates presented here imply an average sediment concentration over the same time period of about 170 mg/L, an encouraging similarity. Furthermore, these estimates suggest that only about 3.5% of the Holocene terrigenous sediment in Moreton Bay is derived from the consequences of "European" land management.

Landscape change in coastal areas adjacent to southern Moreton Bay, over the half-century from the 1880s to the 1930s is described by Hanlon (1935): "...rivers denuded of all their scrubs, ... the streams themselves seemed to be sullen and sluggish, and polluted, and wore an air of being ashamed of their nowadays nudity. Utility and ugliness were the dominant notes everywhere. In many places the physical features of the places were changed or entirely obliterated; watercourse and chain of ponds of my day were, nearly all, filled in with the accumulated debris of the past half century or so. ... the thickly timbered hillslope gullies behind the Pimpama Hotel, harboured a great number of families of Bell-birds ... these birds disappeared "ages ago" ... before a big and comprehensive system of drainage of the many swamp areas around Carrara and Benowa, on either bank of the Nerang River, the big birds we call Swamp Redbills were a real pest to the sugar farmers ... Now that their former haunts (the reedy swamps) have been transformed into grazing lands and farms these birds have disappeared entirely." A further 60 y of development have continued the transformation of this landscape since Hanlon's observations (e.g. see Figure 1 and Map 2 of Hyland & Butler, 1988).

In considering catchment impacts on the Bay, it is important to recognise that each catchment discharging to the Bay has its own unique combination of biophysical characteristics, climate history, land use history, and spatial and temporal scale of influence on downstream geomorphic and ecological systems. Consequently, there is no single chronology of catchment impacts on the Bay, rather several chronologies which interact in complex ways.

Degradation of the Bay and its foreshores

If the relationship between "baseline" conditions and those of the present is unclear in the Moreton Bay catchment, it is much less so in the marine ecosystems of the Bay itself. For

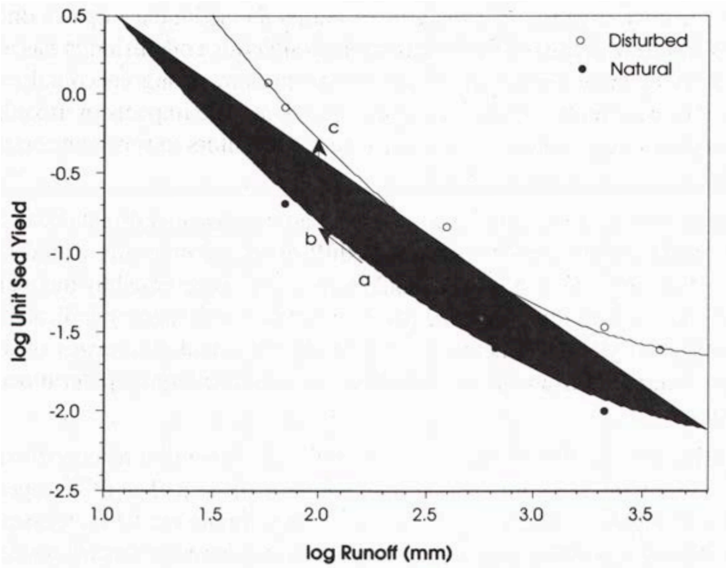


Figure 7. Relationship between catchment runoff and unit sediment yield (USY (t.km^{-2}/y.mm runoff^{1}); USY x 10^3 = mean flow weighted sediment concentration) for Queensland coastal streams. Estimates for the Moreton Bay catchment (Brisbane River): 'a' = at climatic optimum, 'b' = during the late Holocene, and 'c' = present, i.e. including the effects of post-European settlement land use intensification (based on Neil & Yu, 1996).

example, fish catches were reported to have declined by the turn of the century (Petrie, 1904); but made a "remarkable recovery towards what it had been in the golden days of last century" following the reduction of fishing effort during the Second World War. The recovery was short-lived and "since then the decline has been rapid" (Bell, 1975), and fisheries of the Bay continue to be heavily exploited. Attempts to manage the fisheries of Moreton Bay and preserve fish breeding and feeding grounds commenced as early as the Queensland Fisheries Act of 1887. This act also included provision for closures, examples of which included closure to net fishing of the Brisbane River upstream from Bulimba Creek and from Canaipa Passage to the Broadwater, the latter for three years (Davenport, 1986). Bay fisheries in the late 1800s were hampered by the lack of refrigeration and infrastructure. In the absence of refrigeration and motorised transport, overfishing of banks close to markets occurred. The construction of railways improved the efficiency of transporting fish to markets in Brisbane and of subsequent transport to inland towns, thus increasing the level of exploitation of fish stocks. In 1897, 600 t of fish were carried on railways of southern Queensland (Davenport, 1986).

Concern about the sustainability of the Moreton Bay oyster fishery, following decades of burning live oysters for lime, resulted in prohibition of this practice by the Oyster Act of 1863. Further regulation was imposed by the Oyster Act of 1874 and, subsequently, the Oyster Act of 1886 (Davenport, 1986). Nevertheless, as a consequence of the combined effects of disease (possibly introduced in live oysters imported from New Zealand), overexploitation, market competition and land use conflicts, the Moreton Bay oyster fishery collapsed, from the largest fishery in southern Queensland to one of the smallest, by the first decades of this century (Smith, 1982).

Dugong were harvested to near local extinction during the last century. Dugong numbers were reported to be "rapidly decreasing" (Fairholme, 1856a) as early as the 1850s as a result of Aboriginal hunting, made more efficient by the use of European technologies (whaleboats and harpoons) and more desirable by the existence of markets, particularly for dugong oil (Fairholme, 1856a). Petrie (1904) and Welsby (1905) also noted the decline in dugong numbers in the Bay. Although attempts to revive the dugong industry at Amity Point in the period 1901-1910 were hampered because the dugong numbers had been so depleted (Beitz, 1972), some commercial exploitation of dugongs in Moreton Bay continued until 1944 (Preen *et al*., 1992). The range of dugongs within the Bay has largely contracted to the area adjacent to South Passage, probably influenced by a combination of boat traffic disturbance (Preen, 1992) and habitat decline in the western Bay. The dugong is listed as vulnerable to extinction (IUCN, 1990).

Campbell (in Welsby, 1905) noted that "...people who visit the Bay are in the habit of taking shots at the poor old porpoises, both with rifles and shotguns, and the consequence is that they have become shy, and only very seldom can they be got to work to the advantage of the fisherman". This may have contributed to the demise of cooperative fishing between humans and dolphins in the Bay (see Neil & Brieze, this volume). As early as the 1920s, Longman (1926) reported that "...owing to the presence of motor boats ...[dolphins]... are less common than in the past". Both boat traffic and the exploitation of the dolphins' food sources have increased since that time. There is some evidence from locations elsewhere that dolphins avoid approaching boats, possibly as a direct response to the boat itself or in response to prey movements, but their behaviour changes little if boats are merely passing (Acevedo, 1991; Janik & Thompson, 1996). Under conditions of diminished food supply, use of trawler bycatch provides an easily located and captured food source for dolphins. However, this association between the dolphins and human activities may lead to greater vulnerability of the dolphins to attack by sharks which are also attracted to the trawler bycatch (Preen *et al*., 1992).

Petrie (1904) commented on the nineteenth century decline in turtle numbers in the Bay. Although the population of green turtles (*Chelonia mydas*) appears to have experienced some recovery since 1950 (Limpus *et al*., 1994a), when they were protected in Queensland waters, it is unlikely that this recovery has seen a return to the "great numbers" (Petrie, 1904) seen in the early days. In the 1990s the only known concentration of green turtles in the Bay is on the Moreton Banks, in the vicinity of South Passage (Limpus *et al*., 1994a) whereas, in the early days, large numbers were found in the western Bay, particularly in the Humpybong – Pumicestone Passage area (Petrie, 1904). It seems likely that, like the dugongs, the remnant turtle population has largely contracted to the eastern Bay. Harvesting of the Moreton Bay green turtle population occurred both on their feeding grounds in the Bay (from the 1890s) and on their nesting areas on the southern Great Barrier Reef (Limpus *et al*., 1994a). Moreton Bay is also a significant feeding ground for loggerhead turtles (*Caretta caretta*) nesting in eastern Australia, the eastern Australian population being the only significant breeding population in the South Pacific Ocean (Limpus *et al*., 1994b). This is important, given the 50-80% decline in the loggerhead population in the southwest Pacific over the last 10-15 y (Limpus & Reimer, 1994) the major cause of which is incidental killing of turtles by fishing, particularly by prawn trawling. Almost half of all tagged turtles captured by prawn trawlers in eastern Australia (excluding habitat immediately adjacent to nesting areas) were captured in Moreton Bay, with a further 16% captured in waters immediately outside of the Bay. Estimated mortality for this incidental catch is 30% (Limpus & Reimer, 1994). Although loggerhead turtles have not been subject to direct exploitation, about 11% of the Moreton Bay population shows signs of anthropogenic impacts (Limpus *et al*., 1994b). Similarly, about 10% of the green turtle population exhibit these signs, which include fibropapillomas (attributed to anthropogenic changes in marine ecosystems), entanglement in ropes and fishing lines, and boat/propeller strikes (Limpus *et al*., 1994a). These anthropogenic impacts on turtles continue to occur, despite "protection" of all species of sea turtles in Queensland since 1968 under the Fisheries Act (Limpus *et al*., 1984). Limpus *et al.* (1994a) interpret the report of Backhouse (1856) that "... three species of turtle are met with here ...[Moreton Bay]..., one of which is black and unwholesome" as a reference to the leatherback turtle (*Dermochelys coriacea*). At present this species is not resident in Moreton Bay (Limpus, 1995), raising the question of whether this species has been affected by habitat degradation. Green and leatherback turtles are listed as endangered and loggerhead turtles as vulnerable to extinction (IUCN, 1990).

Apart from its influence on the east coast humpback whale population, the Tangalooma whaling station (Bryden, 1978) is a likely indirect cause of anthropogenic impacts on the Bay's dolphins, dugongs and turtles. It has been suggested that "the smell of so much blood ...[from flensing of the whales]... attracted sharks from far and wide to make Tangalooma notorious for them" (Jones, 1980). Increased shark predation on dugongs and dolphins in Moreton Bay is a probable outcome, particularly following closure of the whaling station in 1962. For example, Corkeron (1990) reports that 37% of bottlenose dolphins and 36% of humpback dolphins examined in the Bay carry wounds from sharks. Orams & Deakin (1997), however, suggest that this may be an underestimate of the proportion of dolphins which have suffered shark attack, because of the rapid healing of shark wounds, and that dolphins in the Bay are more vulnerable to shark attack than those outside, e.g. at Point Lookout. A possible explanation is that dolphin groups in the Bay, which are smaller than those at Point Lookout, are too small for adequate predator detection and defence. Concurrently, dolphins in the Bay may be limited to small group sizes by fishing pressure on prey species (see Corkeron, 1997 for discussion). By contrast with the dolphins, turtles in the Bay carry few injuries from sharks (no wounds on 320 *C. caretta* and wounds on one of 826 *C. mydas* examined; Limpus *et al*., 1994a; b)

The avifauna of the Bay has almost certainly suffered a marked decline, although unequivocal evidence is difficult to obtain. Welsby (1907) reported that "black swans were in thousands, ducks the same only more on shore". Such numbers are not seen today, probably as a consequence of such factors as shooting (Welsby, 1907), disturbance and habitat degradation. Few quantitative data are available to document trends in the Bay's avifauna. However, Woodall & Watson (1988) contrasted bird counts during the 1980s at Raby Bay, where coastal ecosystems have been replaced by a canal estate, with counts from the 1950s (Amiet, 1957) and found that marked declines had taken place for most species and that Raby Bay "now is of negligible importance as a wader habitat". A similar pattern is likely to have occurred for most of the "developed" Bay foreshores. This may have a disproportionate effect on particular species because many species which are common in the western Bay (preferring muddy substrate habitats) are not necessarily common elsewhere in the Bay (Driscoll, 1992). Indirect anthropogenic disturbance of shorebirds (e.g. due to nutrient enrichment » seagrass decline » changed habitat structure and food availability) has led to changes in the natural patterns of species composition and distribution in the Bay (Thompson, 1993). Temporal patterns (e.g. seasonal variation) are also important. During their southward migration, shorebirds in Moreton Bay have low energy reserves and tend to utilise a wide range of habitats (Thompson, 1998). Because of the complexity of the spatial and temporal patterns of habitat use, Thompson (1998) argues that protection of all habitats, including those in both the western and eastern Bay, is necessary in order to conserve the full range of shorebird species which use the Bay. Anthropogenic impacts notwithstanding, Moreton Bay remains of sufficient importance to have been listed under the JAMBA and CAMBA agreements and to be declared a Ramsar site (under the *Convention on Wetlands of International Importance Especially as Waterfowl Habitat*). The Bay is "..arguably the most important feeding area for migratory waders along the coastline of eastern Australia (Driscoll, 1992).

The destruction of areas of *Melaleuca* wetlands, mangroves and seagrasses has been reported, however, there is also evidence of increases in the area of mangrove and seagrass communities in some localities. There are anecdotal accounts of the replacement of sandy beaches and intertidal flats by mud, and widespread evidence of shoreline erosion.

Foreshores

Foreshore development on the mainland coast commenced from as early as 1832 with the construction of a stone jetty at Cleveland Point. Mainland coast foreshores have been changed markedly in some areas by development of the early coastal resorts such as Sandgate and Wynnum/Manly. *Melaleuca* wetlands, the vegetation type most severely affected by clearing in southeast Queensland (Catterall & Kingston, 1994), were "reclaimed" at these locations to develop recreation reserves. Greenway (1998) suggests that the loss of these wetlands is largely a consequence of competing land uses and ignorance of the value of wetlands, and wetland degradation has occurred largely as a result of ignorance of the importance of the hydrological regime to their maintenance. The main causes of *Melaleuca* wetland loss and/or degradation, and threats to their future survival, are drainage for agriculture, drainage and filling for urban and industrial development, drainage for mosquito control, use of surface and groundwater for irrigation (e.g. of golf courses), altered hydrological regimes resulting from engineering works (e.g. highway construction), and water pollution, including salt water intrusion (Greenway, 1998). Beaches, which formed the original focus of the bayside resort villages, have changed markedly over the last century.

Artificial shorelines, including seawalls, breakwaters and groins, have been constructed along a large proportion of Moreton Bay's western coast, as well as at several locations on Bay islands. Of these types of structures, seawalls occupy by far the greatest length of the coastline. Given the three main options available for managing eroding shorelines: (i) hard stabilisation (e.g. permanent hard structures in a fixed location such as seawalls); (ii) soft stabilisation (e.g. beach nourishment, beach replenishment); and (iii) relocation or retreat, the traditional response globally has been hard stabilisation (Pilkey & Wright, 1988). This observation also applies in Moreton Bay. Seawalls may degrade or destroy beaches in three ways: (1) walls constructed within the beach zone result in immediate partial or total loss of the beach by burial; (2) walls constructed on an eroding shoreline will result in narrowing or loss of the beach if the erosion continues; and (3) walls may increase rates of erosion by intensifying erosion processes at the beach face (Pilkey & Wright, 1988). Examples of mechanisms (1) and (2) above have occurred in Moreton Bay, although there remains some controversy over the role of seawalls as an active agent of erosion (e.g. Kraus, 1988; Pilkey & Wright, 1988). Determining whether seawalls have played an active role in coastal erosion in Moreton Bay (process (3) above) would require detailed process studies. Terchunian (1988) has argued for beach nourishment programs as a normal companion to seawall construction, comparing this relationship with the use of wetland mitigation to reduce the impact of development on coastal ecosystems. In the > 80 y history of seawall construction on the coastline of Moreton Bay, the need for a concurrent beach nourishment program has been largely ignored. Consequently, many of the beaches of Moreton Bay have been severely degraded.

Redcliffe gradually became popular as a seaside resort from the 1870s, although its growth was limited by poor accessibility from Brisbane. Apparently, one of the attractions was the sand which "...being of brown tint ... there was no glare" (Comyns, 1908; cited in Jones, 1988). Seawall construction at Sutton's Beach, on the Redcliffe Peninsula, commenced in 1918 (Jones, 1988). Reflecting on 40 y of observations (1943-1983), Stanaway (1983) observed that at Scotts Point on the southeast coast of the Redcliffe Peninsula, the "swimming beach" at the foot of

Lahore Hill was reduced from 12 m wide at high tide to nil, following the construction of stone retaining walls. Consequently, "the number of daytrippers has dropped on the Peninsula since the beaches vanished with the advent of the rockwalls, and the best beaches there are the ones which still have a natural shoreline" (Stanaway, 1983). It should be noted that there are still beaches seaward of some rockwalls on the Redcliffe Peninsula. Subdivision practices as early as the 1870s have apparently been a significant factor in the condition of Redcliffe beaches today, limiting subsequent management options.

Real estate posters of the 1920s depict a broad sandy beach at Clontarf, on the southern side of the Redcliffe Peninsula, and describe "a beautiful ... fine sandy beach.. [with] splendid fishing.. [and] .. safe bathing". Today, most of the coastline between Woody Point and Hays Inlet is bounded by seawalls and there are no longer beaches seaward of these walls. A careful evaluation of the shoreline changes which have occurred on the Redcliffe Peninsula, their causes and the nature and effects of management responses, is needed.

In the 1890s, Sandgate, at the terminus of one of the railway lines, became a prosperous bayside resort and the most popular "watering place" near Brisbane. Such was its popularity that, on New Years Day 1899, 8 000 people travelled to Sandgate by train (Lawson, 1973). At Sandgate "the shore was shelving and perfectly free from mud, the beach being composed in some places of sand, and in others of shingle" (Lang, 1861). Construction of seawalls along the coastline between Hays Inlet and Cabbage Tree Creek has altered the character of the foreshore and destroyed the beaches. At present, a small artificial beach is all that remains on a shoreline of concrete walls and intertidal flats.

The other popular bayside resort at the end of the last century was Wynnum. The Wynnum foreshore was characterised by an extensive *Melaleuca* wetland with a sandy beach to seaward. Until the 1930s the foreshore was reportedly "a sandy beach for 100 m out from Lota to Wynnum North" (Ludlow, 1994). A log wall was constructed along parts of this shoreline, apparently, at least in part, from the trees cut from the *Melaleuca* wetland (Ludlow, 1994). During the Great Depression of the 1930s, relief workers replaced the log wall with a stone one (Wetherell, 1993). Today, the Wynnum/Manly foreshore resembles that of Sandgate, with a reclaimed, levelled foreshore park, with a stone seawall and mudflats at its base. About 35% of the coastline between the Brisbane and Logan Rivers is "protected" by such seawalls (Wetherell, 1993).

The early development of Cleveland is associated with its failure as the major regional seaport in the 1850s, as a sugar growing centre in the 1860s, and as a resort in the 1900s. At this time there were "two great Brisbane holiday resorts...Sandgate for those who had to depend upon the train and Cleveland was for those with buggies...In September 1903 the railway line to Tweed Heads was opened ..[and].. Cleveland's hope of becoming the pleasure resort of Brisbane became very dimmed...Local councils ..[on the Gold Coast].. set themselves out to attract holiday visitors, a thing upon which the Cleveland Council frowned" (McKinnon, 1948). A proposal that the Council seek a development loan from the State Government was "turned down flat by the Council, most of its members saying definitely that they did not want Cleveland to become a Brisbane holiday centre.. Cleveland ... had its crowded hour and died. Died without hope even of a glorious resurrection." (McKinnon, 1948). The resurrection of Cleveland has seen the destruction of mangroves, saltmarshes and seagrass beds as a consequence of the construction of the Raby Bay development and of Toondah Harbour. The first seawall on Cleveland Point was constructed in 1875 (Davenport, 1986) and, in the 1990s, Cleveland Point and Raby Bay are entirely enclosed by seawalls.

At Victoria Point, development took place more recently. In the mid 1950s there was some residential development near the end of the point and along the southeast-facing embayment between Victoria Point and Wilson's Point. Most of the remainder was used for small crop farming. A small caravan park and camping ground was located in the reserve at the seaward end of Victoria Point, adjacent to a sandy beach. Construction of carparks and picnic areas by "reclamation" has destroyed the beach and left a seawall and mudflat, littered with angular rock debris from the construction work. In the southern part of the Victoria Point–Wilson's Point embayment a thin line of mangroves protected a narrow, sandy beach. By the late 1970s the mangroves and beach were gone and seawalls abutted the mudflats. In 1997, about 75% of the residential properties along this embayment are "protected" by seawalls. This more recent coastal development at Victoria Point provides some clues as to the process of foreshore degradation of much of the west coast of Moreton Bay.

Southern Moreton Bay was not exempt from seawall construction. A concrete wall about 1 km long was constructed south of the Southport Jetty in 1901-2 (Davenport, 1986).

An important recreational resource has been lost to the communities of southeast Queensland due to beach erosion and seawall construction. Apart from small artificial beaches at Brighton, Wynnum and on the Raby Bay shoreline, little thought appears to have been given to the long term restoration and maintenance of recreational beaches in Moreton Bay. An example of what can be done comes from Tokyo Bay where nine beaches with a combined length of 13 km were constructed during the 1970s and 1980s in partial replacement of recreational beaches lost as a result of land reclamation works (Koike, 1990). It is important to note that these works were undertaken in association with plans for improved water quality and public access. Local authorities should be evaluating restoration programs for Moreton Bay recreational beaches, acknowledging that artificial beaches have a finite life (e.g. May, 1990; Leonard *et al*., 1990) and that their continued viability requires long term, regular maintenance. Although such restored beaches are preferable to seawalls with no beach, the preferred option should be planning which allows for further coastal recession, accompanied by appropriate relocation and retreat.

Further offshore, on the intertidal flats, significant changes have also taken place in Moreton Bay. Stanaway (1983) suggests that sufficient sedimentation had occurred to reduce low tide depths at Scotts Point from 2 m at 5 m offshore to about 1 m at 20 m offshore, and at Clontarf Beach low tide depth at 100 m offshore has decreased from 1.5 to 0.5 m. Such claims warrant sedimentological verification. Changes in the composition of the intertidal substrate have also occurred. At Manly, in the 1920s, there was a "big salt pan at low tide which was used for cricket, football and corroborees" and, to the 1930s, the foreshore was reported to be a sandy beach for 100 m out from Lota to Wynnum North (Ludlow, 1994). Similarly, the intertidal flats at Victoria Point were sandier in the 1930s to 1950s and often used for playing cricket (Arundell, pers. comm.), which is impossible today. This is not to suggest that all muddy areas of the western Bay were once sandy. This is clearly not the case, on the basis of both geomorphic processes (the rivers discharging to the Bay carry large volumes of silt and clay) and observations of the early explorers (e.g. from Redcliffe Point to the Brisbane River there was "no great danger" to ships which ran aground "as the Shoals are of soft mud" (Oxley in Steele, 1972). Anecdotal accounts of changes in substrate have their limitations as the following example shows. In support of the use of Green Island as a resort, Home Secretary Appel described the "..hard sandy bottom..[which].. at high water .. is one of the best bathing grounds imaginable..". Several days later a member of the Moreton Bay League, who were arguing for closure of the prison on St Helena Island to allow it be used as a resort for the people of Brisbane, responded suggesting that "The so-called "hard sandy bottom" resembles an oyster bank more than anything else .. sharp coral and mussel shells and, although I used the utmost care, I returned to the shore with bleeding feet" (Finger, 1987). Both accounts are probably correct, referring to different, but immediately adjacent areas of the intertidal flats.

Changes in water turbidity in inshore areas are even more difficult to evaluate from anecdotal accounts as they are episodic in nature. However, a shift to muddier sediments would result in increased turbidity in response to a given input of wind and wave energy. Again, Bay waters have always been turbid to some degree. For example, Edwardson (in Steele, 1972), from observations made in 1822, reports that "the tide running in various directions at 3 to 4 miles per hour, stirs up mud and sand so thickly as to hide the appearance of shoals and sands".

Mangroves

Mangroves were cleared (e.g. in association with river improvement works at the mouth of the Brisbane River) and exploited (e.g. for smoking fish – "mangrove wood ... gives the best results ... viz., delicious flavour and a rosy, glossy surface" (Fison, 1894)) since the middle of the last century. More recently, changes in the distribution patterns of mangrove vegetation have been documented in Moreton Bay. Hyland & Butler (1988) report the loss of significant areas of mangroves in the Bay (c. 8.4% loss between 1974 and 1987) largely in response to construction of the new Brisbane Airport. Wetherell (1993) examined changes in the coastline between the Brisbane and Logan Rivers between the mid-1950s and the early 1990s using historical air photo analysis. Several areas of direct anthropogenic mangrove loss were recorded, particularly in the north of the study area and some areas of expansion in the area of mangroves (notably at Oyster Point (see also McTainsh *et al.*, 1986)). However, throughout most of this area, the mangrove distribution was generally stable, with only small increases observed, largely in a shoreward direction at the expense of saltmarsh. In the Pimpama/Coomera River region of southern Moreton Bay, Morton (1993) reported a 10% increase in the area of mangroves in the period since 1944, largely shoreward and, in some areas, at the expense of saltmarsh communities. He attributes these changes to several factors including the Jumpinpin Bar breakthrough, and the construction of the tidal barrage on the Pimpama River and the Gold Coast Seaway. Hanlon's (1935) account (above) describes the loss of wetlands on the lower Nerang to the 1930s. Hyland & Butler (1988) contrast the extent of wetlands in this area in 1934 with that in 1987, showing extensive, additional wetland losses. On North Stradbroke Island, Bell (1975) reported mangrove progradation at Myora (on the west coast), although neither rates nor time periods are given, and Flood and Grant (1984) observed that mangroves were encroaching (landward) into the freshwater swamp of Eighteen Mile Swamp, on the island's east coast. These historical changes in mangrove distribution which have occurred on North Stradbroke Island are unlikely to be related to local anthropogenic influences. It should also be noted that a good understanding of mangrove community response to natural and anthropogenic influences is difficult to obtain from a small number of observations through time, given the sometimes rapid fluctuations in mangrove area and condition in response to a diversity of possibly countervailing influences (see Buckney, 1987, for example). Mangrove losses have also been reported from streams discharging to Moreton Bay. For example, Henry *et al.* (1987) reported a loss of 50% of mangroves from the Brisbane River, although the time period and study area boundaries are not defined. Mangroves have migrated upstream in the Brisbane River from about 9 km from the river mouth in the 1840s (Lang, 1861), to at least 35 km in 1928 (Watson, 1928) and 64 km in 1974 (Hegerl, 1975). This process appears to be a consequence of a combination of anthropogenic factors including increased tidal penetration as a result of dredging, dam construction in the catchment and muddier substrates on the river banks. These mangrove communities apparently now play an important role in bank stabilisation (Hegerl, 1975), although they suffer heavy mortality during floods, largely as a result of high rates of silt deposition (Watson, 1928; Hegerl, 1975). Ironically, a significant influence on these mortality rates is likely to be the anthropogenic increase of siltation rates. Mangroves have also colonised tributary streams in the lower reaches and there are anecdotal reports of upstream expansion of mangrove communities in other streams discharging to the Bay (e.g. Tingalpa Creek in response to diminished stream flows following construction of the Leslie Harrison Dam). In the absence of road access to areas of the Moreton Bay catchment south of Brisbane, access for settlers and transport of produce in the late 1800s was largely by the major streams, all of which had shallow bars and shoals in their lower reaches. Dredging for improved navigation on the Logan, Albert, Coomera and Nerang Rivers (e.g. Nisbet, 1887; Davenport, 1986) is likely to have facilitated upstream migration of mangrove communities in these streams and markedly altered the ecology of the lower reaches. Ironically, after decades of lobbying, the dredging of these streams to facilitate improved river boat access was finally carried out as the railway lines, which would soon replace the river boats, were being laid. The same pattern occurred, for example, on the Murray–Darling system and on the Mississippi River and its tributaries in the USA (e.g. Hunter, 1949).

Seagrasses

Seagrass areas in Moreton Bay have both increased and decreased at different locations and time scales. In Deception Bay, a large-scale decline in the 1970s, apparently due to sediment movement (Kirkman, 1978), was followed by a large scale recovery during the 1980s (Hyland *et al.*, 1989). Wetherell's (1993) analysis of the Brisbane-Logan River coastline indicates a general pattern of shoreward movement of the upper seagrass boundary over the three decades to c. 1990. Arundell (pers. comm.) reported that there was no seagrass on the intertidal flats at Victoria Point (between Victoria Point and Wilson's Point) in the 1930s, air photos of the 1950s indicate limited areas of seagrass at the low tide margin and, by the 1970s more than 50% of the intertidal flat was colonised by seagrasses. An exception to this general pattern occurred at Point Halloran where there was a 200 m retreat of seagrass in response to formation and movement of a sandspit. Seagrass losses due to direct human impacts include 60 ha lost due to construction of Manly Boat Harbour, a small area at Thorneside due to dredging of the channel for the Aquatic Paradise canal estate, and 50 ha due to construction of the Raby Bay canal estate. Little change in the area of seagrass between Redland Bay and the Logan River mouth was discernible (Wetherell, 1993). Air photo inspection also indicates that the area of intertidal seagrass has also increased markedly on mudflats south of Dunwich and on banks offshore from Victoria Point. On relatively short time scales, Abal & Dennison (1996) report significant contraction of the area and depth range of seagrass in the Logan River estuary, an effect which diminishes with distance from the estuary and has been associated with increased water turbidity and concentrations of total N and chlorophyll *a* in response to input from the Logan River.

In the Pimpama/Coomera River region, Morton (1993) reported an increase in the area of seagrass in the period 1944 to 1973, followed by a marked decline. Similarly, in the southern Southport Broadwater, about 90% of the dense seagrass present in 1982 was lost by 1987. The overall loss of about 80% is attributed to several factors including freshwater flooding, dredging, increased boat traffic, and altered tidal regimes following construction of the Gold Coast Seaway. Morton's (1993) results are consistent with the observations of Doley (in Hyland *et al.*, 1989) of seagrass decline in the Broadwater.

Cross-Bay variation in seagrass community structure, depth (Preen, 1992), area and depth range (Abal & Dennison, 1996) are associated with a cross-Bay gradient in water quality. It seems likely, therefore, that prior to post-settlement intensification of land use, and consequent decline in water quality, the patterns of seagrass distribution in the western Bay were more like those which presently occur in the east. Such changes have implications for the population and distribution of species which depend on seagrasses. Campbell (in Welsby, 1931) suggested that the best winter dugong catches in the Bay were to be had following a summer of heavy rain and flooding. Flood waters fertilised the seagrass beds resulting in "vigorous sprouting of fresh young shoots of the dugong grass" (nutrient availability has a significant effect on the growth of seagrasses (Short, 1987)). Recent reports of seagrass decline as a result of deteriorating water quality and flooding (e.g. Preen *et al.*, 1993 in Hervey Bay; Abal & Dennison, 1996 in Moreton Bay) may indicate that catchment input to the Bay has crossed the threshold from beneficial to deleterious effects of catchment input.

Anthropogenic effects on seagrass community characteristics are likely to be complex with multiple causality operating at differing timescales. For example, dugongs have a strong dietary preference for the seagrasses *Halophila ovalis* and *Halodule uninervis*, which recolonise disturbed areas. Dugong grazing ("cultivation" grazing (Preen, 1992)) encourages *H. ovalis*, and retards the expansion of *Zostera capricorni* (Preen, 1995). Reduction of the dugong population in western Moreton Bay, whether by past hunting pressure or present disturbance by boat traffic (Preen, 1992), is likely to have changed seagrass community characteristics over the last 150 y. Similarly, given the marked inter-specific differences in seagrass response to nutrient enrichment (Udy & Dennison, 1997), changes in seagrass community characteristics may also have occurred in response to increased nutrient input from agricultural runoff and discharges of inadequately treated sewage effluent. Given the greater capacity of muddy sediments to transport and store nutrients, any change to muddier coastal sediments in the Bay (see above) is also likely to alter seagrass community structure. In summary, anthropogenic influences are likely to have changed the distribution and composition of seagrass communities in the Bay through alterations of light regimes, nutrient input, sedimentation patterns and effects of grazing. Clearly, the spatial (within–site, between–site, cross–Bay) and temporal (decadal, seasonal, episodic) patterns, and interactions between them, will be complex and difficult to unravel.

Coral communities

Coral communities in Moreton Bay largely occur in the central part of the Bay where there are suitable substrates for coral recruitment, where they are exposed to oceanic waters entering the Bay through South Passage and southward down the west coast of Moreton Island, and south of the normal (northward) path of the Brisbane River sediment plume. They are distributed along a strong onshore-offshore water quality gradient, with conditions most stressful for coral growth occurring in the western Bay (Johnson & Neil, this volume). Stresses associated with human activities are also greater in the western Bay. These include increased concentrations of sediments and nutrients in runoff water, sediment resuspension by boat wash and the direct effects of coral dredging. At the present time, there are insufficient data to determine the extent of the effects of human activities on the Bay's coral communities. At Mud Island, where corals are under the greatest stress from Brisbane River outflows and where corals had been collected for limemaking since the 1840s, nearly all of the subtidal reef flat has been removed by dredging (Allingham & Neil, 1995). Dredging has also damaged coral communities at Empire Point by removal of substrate and by sedimentation (Harrison *et al.*, 1991). Anchor damage is inferred from the structural characteristics of corals along the eastern margin of the Green Island reef (Harrison *et al.*, 1991). At sites marginal to coral growth, most colonies of *Favia speciosa* (living or dead) are attached to *Acropora* skeletal fragments, suggesting that the transition from acroporid to faviid coral communities in the Bay may have occurred relatively recently.

Alternatively, relict acroporid skeletons from the mid Holocene may simply provide a suitable substrate for modern coral recruitment.

Extensive coral mortality has occurred as a consequence of major floods (e.g. Slack-Smith, 1959; Lovell, 1989), however, the extent to which these impacts differ from those prior to European settlement is unclear. An encouraging finding for Moreton Bay's coral communities is an apparent increase in diversity since the 1970s. An improvement in water quality has been suggested as facilitating this change (Harrison *et al.*, 1991), although further work will be required to ascertain the extent to which it is influenced by differences in survey methods, species identification and environmental factors such as flood magnitude and frequency. The decade prior to the survey of Harrison *et al.* (1991) was one of low flood impacts on Moreton Bay. In the longer term, McEwan *et al.* (1998) predict increases in chlorophyll *a* concentrations of the order of 40 to 60% at reef sites in the Bay in response to predicted population increase and land use intensification in the catchment to the year 2031. A marked decline in the health of coral communities in association with this increase would be likely if the trend is not arrested.

Although the role of land use intensification in the major catchments (Brisbane and, to a lesser extent, Logan) is generally emphasised in discussion of the degradation of the western Bay, the changes which have occurred in the small coastal catchments cannot be ignored. For example, the sediment laden freshwater plume from Moogurrapum Creek at Redland Bay resulting from the small flood of 15–16 May 1998 reached as far north as Point Halloran and Coochiemudlo Islands (pers. obs.). Similarly, that from Eprapah Creek at Victoria Point extended north of Cassim and Sandy Islands. All of these sites have coral reef communities which are severely degraded by comparison with their sub-fossil equivalents. Both of these catchments have been extensively cleared for agriculture and urban and rural residential development. Sewage treatment plants also discharge waste into many of these streams.

Islands

The islands of Moreton Bay are of four main types:

1. Sand islands, composed of siliceous sands and include Bribie, Moreton and North and South Stradbroke Islands;
2. Reef islands, composed of carbonate sediments and include Mud, Green, King and Bird Islands;
3. Delta islands, predominantly in the area from Redland Bay south to the Broadwater, composed of mud and colonised by mangroves; and
4. High islands, largely composed of Landsborough sandstone, with Tertiary basalts in some areas, and include St Helena, Peel, Coochiemudlo, Macleay, Lamb, Karragarra and Russell Islands (this list is not exhaustive).

The nature of human use, and of consequent changes, on these different types of islands varies considerably, as does their sensitivity to exploitation.

Sand islands

Anthropogenic changes on the sand islands are associated with urbanisation on Bribie, relatively isolated village settlements on Moreton and North Stradbroke, and resort developments on Moreton and South Stradbroke Islands. Extensive degradation of Cape Moreton is associated with the presence of goats and timber felling for the light station. More extensive impacts are associated with plantation forestry on Bribie and, to a lesser extent, on North Stradbroke, and various forms of sandmining on North Stradbroke Island. Human activity on the sand islands generally results in limited impacts on the Bay, although there have been cases of mangrove destruction as a consequence of sand mining (at Amity Point and Blaksley Lagoon) and of mangrove and saltmarsh "reclamation" (at Amity Point) for camping ground construction (see Durbidge & Covacevich, 1981). The early (1880s) land use on South Stradbroke Island was cattle grazing which led to widespread erosion (Salter, 1984).

Reef islands

Mud Island largely consists of a "mangrove park" interspersed with several, roughly concentric sand islets. The island is classified as a "mangrove island" (Allingham & Neil, 1995) which is an unusual type of reef island of which only five examples occur on the Great Barrier Reef, all at < 15°S (Hopley, 1982), although they are relatively common in the Caribbean (Stoddart & Steers, 1977). Anthropogenic alterations to the island are largely a consequence of coral dredging operations by Queensland Cement Limited which commenced in 1937, although removal of corals from Mud Island had been occurring for almost a century prior to this (Saville-Kent, 1893). Dredging operations have removed about 74% of the open reef flat. This has altered the hydrodynamics of the island in two main ways. Firstly, removal of the reef flat has resulted in increased wave energy during high tides, with significant erosion of the northwestern islet. Secondly, around most (75%) of the perimeter of the island, adjacent to areas dredged above the low tide line, the availability of coarse coral rubble generated by the dredging, in combination with the increased wave energy due to removal of the reef flat, has led to the formation of rubble banks which cover an area of about 19 ha. These banks have caused mangrove mortality by restricting tidal circulation in the mangrove park and by direct abrasion and partial burial of trees and seedlings (Allingham & Neil, 1995).

Similar rubble banks, on a much smaller scale, have also formed in response to coral dredging at St Helena Island. Here they occupy 22% of the islands' perimeter (LeProvost, Dames & Moore, 1997a) and have begun to modify drainage of the reef flat (LeProvost, Dames & Moore, 1997b). The observation that "beach ridges are a natural geomorphological feature of Mud, St Helena and Green Islands" (LeProvost, Dames & Moore, 1997a) fails to acknowledge that the geomorphology and sedimentology of the rubble banks resulting from coral dredging are fundamentally different from the natural sedimentary deposits on these islands and their reef flats.

Green Island is a "low wooded island" (Neil, 1994), a type of reef island which has been recorded only at latitudes of < 18°S. At 27°S, Green Island is a unique, subtropical example of geomorphic and biogeographic significance; it consists of three small islets, the two on the eastern side being largely intact, although subject to weed infestations. Most of the natural littoral rainforest has been removed from the largest, northwest sand cay, and replaced by a weed-infested sward, a stand of *Casuarina*, and a fine stand of groundsel (Johnson *et al.*, 1993).

Bird Island was described by Flinders (in Steele, 1972) as one of "two small spots, which being covered with wood look as if they were models for islands; their appearance being very pretty." Welsby (1907) confirmed this – "In those days ..[c. 1880s].. the island was a beauty with numbers of high oak trees all over it....How that pleasant island has changed since then. Few trees, not one of any height, all cut and hewed by the vandal". The island was subsequently reduced to a bare sand bank, however, planting of *Casuarina* by a far-sighted Moreton Bay yachtsman has restored some of Bird Island's former beauty. At present (January 1998) Bird Island is eroding and about two-thirds of the mature *Casuarina* are dead. Goat Island is a small sand cay deposited on a supratidal lateritic platform. Native vegetation includes a mangrove fringe and *Hibiscus* and *Pandanus* spp. The island is heavily infested by weeds including asparagus fern, prickly pear and groundsel.

King Island was described as follows (Ludlow, 1994): "..originally there was about half an acre of thick vine scrub comprising some sizable trees, including dogwood, ironwood, cotton trees etc. .. today, King Island is only a skeleton compared to when we lived there. About half or even more has washed away. This was caused by the local council cutting down all the mangrove trees about 50 years ago.. [i.e. about 1930]". Corals from King Island were exploited for lime making from about the 1840s (Petrie, 1904).

Delta islands

Kelley and Baker (1984) described a sequence of development of delta islands adjacent to the Logan, Pimpama and Coomera River mouths which is intimately linked to the sea level history of the area and to the establishment of an offshore barrier island (South Stradbroke Island). While the barrier was open, a flood-tide delta was established between the incomplete barrier and the mainland coast. Sea level fall in the late Holocene (3-4 ka) facilitated development of the barrier (Flood & Grant, 1984; Kelley & Baker, 1984). Consequent reduction of energy in the back barrier lagoon allowed deposition of the delta islands which were subsequently colonised by mangroves. Closure of the opening probably occurred by northward longshore drift about 600 y ago (Grant *et al.*, 1985). The Jumpinpin breakthrough of 1898 reestablished a tidal delta and led to some erosion of the delta islands (Kelley & Baker, 1984). Recent work on the geomorphology of this area has been carried out (Lang *et al*., this volume; Lockhart *et al*., this volume).

The delta islands of the southern Bay are of considerable ecological importance. In the late 1970s the area between Macleay Island and the Nerang River bridge contained about 50% of the mangroves and 40% of the saltmarshes in the southeast Queensland (Noosa to the New South Wales border) region (Curvengen & Outridge, 1982).

Anthropogenic impacts on the delta islands in Moreton Bay include the development of the Port of Brisbane on Fisherman Islands at the mouth of the Brisbane River, with accompanying changes in hydrodynamics and reductions in the area of mangroves (by 30%) and of saltmarsh/claypans (by 50%) (WBM, 1992). In southern Moreton Bay, delta islands (Woogoompah, Tabby Tabby and Eden) have been used for cattle grazing (Curvengen, 1982). Contemporary pressure on the geomorphology and ecology of the delta islands, however, is largely a consequence of northward expansion of the Gold Coast tourism precinct by reclamation and landfill. The development of Sanctuary Cove/Hope Island, of Griffin and Andy's Islands (now Sovereign Islands) and Ephraim Island may be indicative of the style of things to come.

High islands

A detailed analysis of the physiography and human impact on the high islands of Moreton Bay is beyond the scope of this paper. Nevertheless, it is clear that the effects of European settlement have been considerable, mainly through extensive land clearing, initially for agriculture and more recently for residential subdivision. An example of the effects of clearing is the destruction of the "vast congregations" of fruit bats (Fairholme, 1856a) which existed seasonally on St Helena Island before it was cleared, initially for building and fuelling the dugong oil factory in the 1840s and 1850s (see Finger, 1987) and subsequently for construction of the prison and associated agricultural activities during the late 1860s (Bell, 1975). St Helena was previously "..covered with a dense scrub or jungle" (Fairholme, 1856b).

Moreton Bay: Past, Present and Future

The past

"Optimum conditions" occurred in Moreton Bay and its catchment during the mid Holocene sea level highstand. By comparison with the late Holocene, conditions in the Bay were relatively warm, oligotrophic, deep, and oceanic with high water volumes and less variable temperatures at ENSO and seasonal time scales. Catchment input to the Bay, by comparison with the late Holocene, were less variable at ENSO and seasonal time scales, and of low turbidity and low sediment and nutrient concentrations, although higher flood magnitudes and, therefore, freshwater input would have been common.

During the late Holocene "natural degradation" of both the catchment and the Bay occurred as a result of both climate change and cumulative geomorphic processes including sea level fall, restriction of tidal entrances, sediment infilling from the catchment and decreased rainfall of greater variability. In the catchment, physical changes included lower, more variable runoff and higher sediment concentrations in runoff water.

Physical consequences of these changes in the Bay included decreased tidal flushing, reduced average water temperatures, increased water temperature range, and increased bottom sediment resuspension. Biological consequences included some communities (e.g. corals) coming under increasing stress from these changes, and others (e.g. mangroves, seagrasses) probably facilitated by these changes.

Human intervention was responsible for accelerating the late Holocene "natural degradation" trend, although Aboriginal land management played a very limited role in these changes. European settlement imposed both direct and indirect impacts on the Bay which were imposed on a largely unidirectional trend of deteriorating environmental quality. The effects appear to vary with the biological community. A possible scenario is as follows:

Coral communities		->	further stress		
Seagrass communities	subtidal	->	facilitation	->	stress
	intertidal	->	further facilitation		
Mangrove communities	Bay and riverine	->	further facilitation (local stress)		

The present

Catchment impacts, as a consequence of sediment yields, increased rapidly in the 19th century and have been relatively stable since the 1920s. Nutrient input, on the other hand, have increased rapidly since the turn of the century, particularly during the last three decades. Direct impacts on the Bay were at a relatively low level in the 19th century and increased rapidly this century, with some notable exceptions. For example dugong exploitation, which assumed "the qualities of a minor gold rush" (Bell, 1975), was rife last century and early in the present one. There was a dugong rendering plant at Amity Point until the mid 1930s (Bell, 1975). However, with the exception of indigenous hunting, dugong hunting is prohibited today. Marked changes in intertidal communities have occurred, some related to direct impacts, some to catchment impacts and some, apparently, to global changes. Pressure on the Bay's ecosystems is increasing with more population growth, infrastructure development, housing, resort development, shipping and recreation in and adjacent to the Bay. Whilst there is greater awareness and relevant legislation and regulation, there is little evidence to suggest that the destructive trend has been halted, let alone reversed.

The *State of the Brisbane River, Moreton Bay and Waterways* report (Brisbane River

Management Group, 1997) analyses Moreton Bay and its catchment in the context of the Pressure– State–Response model. Pressures on the Bay arise from "...treated wastewater discharges, urban and rural runoff, dredging, fishing, ship ballast water and the reclamation of coastal land. Because of this, water quality in the Bay is poorer near to the coast, and improves further offshore. Water quality in the western embayments tends to be worse than the recommended water quality guidelines for aquatic ecosystems. Runoff from urban and rural areas during floods can have a dramatic impact on coral and seagrass communities, possibly causing major dieback after large floods ... overall, pressures on the Bay are increasing, the western Bay is the worst affected in terms of water quality and habitat, and this is an area of importance for commercial fisheries and other marine life. So far the system has coped reasonably well in terms of its flushing and flood recovery capacity after small to medium flood events, however, the Bay's long term sustainability is uncertain and a cautionary approach is warranted" (BRMG, 1997).

However, the response to the pressures on Moreton Bay is insufficient to reverse the degrading trend. For example, the Bremer River catchment is one of the most degraded parts of the Moreton Bay catchment. Responses identified in this subcatchment are limited to plans, strategies, schemes, studies, surveys and committees (BRMG, 1997). Similarly, for Moreton Bay itself, the responses are largely plans and strategies. Implementation of the majority of the plans and strategies lies in the future, and there is no moratorium on development in the interim (as of early 1998 implementation of the Moreton Bay Marine Park is occurring, however its scope is limited). Under conditions of increasing pressures on Moreton Bay, delays in making the difficult management decisions can lead to those decisions becoming even more difficult to make in the future, being made within a diminished range of options and/or being made only after some ecological or geomorphic threshold has been crossed from which it may be difficult or impossible to return. Models which conceptualise this problem for coral reef systems include those of Buddemeier & Hopley (1988), in which water quality requirements for reef maintenance may be much lower than for reef establishment, and of Done (1992), in which community trajectories are influenced by disturbance regimes, with recovery following major disturbance controlled by ambient conditions (e.g. nutrient availability).

The recently released *Healthy Waterways – Waterways Management Plan* (BRMG/ BRMBWMS, 1998) provides a possible framework for action to resolve many of the issues confronting the Bay and its catchment. Important initiatives under this plan include sewage treatment plant upgrades, development of wastewater re-use schemes, improved management of urban runoff, and development of catchment management plans. However, the plan will only achieve the desired "water quality and ecological health" outcomes if the plans, schemes and strategies are actually implemented, if there is continuity and strength of commitment on the part of the stakeholders (in perpetuity, rather than for a term of political office), and if action is taken to appropriately manage emerging problems (e.g. acid sulphate soils) rather than ignoring the problem until substantial ecological damage has occurred (e.g. nutrient enrichment and eutrophication). Baudel (1981) observed that "...every centre of population has worked out a set of elementary answers – and has an unfortunate tendency to stick to them out of that force of inertia which is one of the great artisans of history". It remains to be seen whether recent and forthcoming developments in management regimes can overcome 150 y of inertia in the environmental history of Moreton Bay, and chart for the future a new course of realised, not just planned, ecological sustainability and health.

The future

In evaluating the future for Moreton Bay and its ecosystems, and the scope and responsibilities of present and future science and management, we could consider two, apparently antithetical, scenarios:

(i) Global warming and sea level rise in response to artificial "Greenhouse" gas increases.

(ii) Global cooling and sea level fall as a response to orbital forcing (Milankovitch cycles) of the Earth's climate.

Reviewing studies of recent changes in sea level, Pirazzoli (1996) suggests that the most likely estimate for sea level change over the last 100 y is 0.7 ± 1.0 mm/y, of which about 40% might be attributed to thermal expansion of ocean waters and 60% to decreasing ice volumes. Recent scenarios for the next 100 y suggest a sea level rise of 0.5 ± 0.4 mm/y (e.g. Wigley & Raper, 1992) in response to a possible global warming of 1.5-4.5^{o}C. Under the global warming scenario a range of outcomes, many of which may be "beneficial", is possible for Moreton Bay. These are likely to include a deeper Bay with improved water quality through better tidal flushing and reduced sediment resuspension (deeper water and, with increased frequency/magnitude/duration of ENSO events, fewer extreme wind events). Greater diversity of mangrove and coral communities is likely to occur, along with an increase in coral cover. However, if ENSO events are more influential, stream sediment concentrations may increase in response to a deterioration of catchment ground cover. Also on the down side, landward expansion of intertidal communities, particularly mangroves and saltmarshes is likely to be a necessary response for survival of these communities. At the same time, coastal property owners are likely to be fighting shoreline recession caused by the rising sea level. Intertidal communities may be caught, and destroyed, between the rising sea and the sea walls. Management has an opportunity to address these issues now, particularly with respect to the delta islands of southern Moreton Bay. In the longer term, sustained sea level rise at 0.5 mm/y may result in destruction of some coastal ecosystems regardless of direct human intervention (e.g. construction of, or prohibition of, seawalls). Holocene analogues suggest that mangrove communities may collapse at rates of sea level rise only 25% of that predicted (Ellison & Stoddart, 1991). Similarly, although coral reef growth in tropical waters may be sufficient to keep pace with sea level rise (Hopley & Kinsey, 1988), the very low growth rates in Moreton Bay (Lovell, 1975; Roberts, 1997) suggest that, under conditions of sustained sea level rise, coral communities in Moreton Bay may also collapse. In both cases, a marked flow-on effect to the Bay's marine fauna would occur. Forecasting coastal response to rising sea level is, of course, an uncertain pursuit, given that the effects vary greatly according to the rate of rise, interactions with climatic changes and the characteristics of the affected landforms and ecosystems, all of which may vary on a regional and, in some cases, local scale (e.g. Dolotov, 1992; Spencer, 1995).

The basis for the global cooling scenario, in insolation changes induced by orbital forcing, is outlined by Berger (1988) and may occur within a few thousand years. Under this scenario, the following sequence of environmental responses is possible: transition to the next glaciation; sea level falls so Moreton Bay dries out; and temperatures decrease so there is northward range contraction of, for example, turtles, dugongs, corals, mangroves etc.. Under this scenario, if Moreton Bay is just going to dry out, why bother with "science for sustainability" and why manage for the sustainability of Moreton Bay's ecosystems?

In the short term we should bother because Moreton Bay is a valuable resource for both present and future generations, a resource of high economic, ecological, aesthetic, and spiritual values. In the long term (at the scale of Quaternary glacial – interglacial cycles), wise management can make a contribution to the long term sustainability of species such as turtles and dugong, which will be stressed by the falling sea levels and temperatures of the "next glaciation". As a result of relatively high levels of exploitation and habitat loss and low levels of protection and management throughout much of the Indo-Pacific region, many of these species will be getting very little contribution to maintenance of their minimum viable populations from much of their range, particularly where this lies outside Australia. Little is known of these processes; we know something of the evolution of coastal ecological and geomorphic systems under conditions of rising sea level, but relatively little of coastal ecosystem response, or population response of individual species, when sea levels fall. Our obligations for wise and sustainable management of Moreton Bay extend beyond the short term (centuries) view of management for "present and future generations" to the time scales required for survival of systems and species which have survived for millions of years in the absence of human impacts. A lesser commitment is little more than superficial tokenism.

Given that the future may well hold warming and sea level rise in the short term, and cooling and sea level fall in the longer term, science for sustainability demands an understanding of the geomorphology and ecology of the past, the present and, perhaps by using spatial and temporal analogues, the future.

Acknowledgements

I am most grateful to Michael Gourlay and Ian Tibbetts (The University of Queensland), Simon Lang (Queensland University of Technology) and Errol Stock (Griffith University) for their many helpful comments and suggestions.

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Land Use, Land Cover and Land Degradation in the Catchment of Moreton Bay

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Abstract

The condition of a catchment determines the state of its receiving body of water. This statement is true for rivers, streams, lakes, and the inshore areas of oceans. Moreton Bay is the receiving water body for rivers and streams which flow from the catchments of Pumicestone Passage, Deception Bay, Bramble Bay, the Brisbane River, Redland Bay and the southern Bay.

In its pre-European state, the catchment was relatively stable with sediment and nutrients predominantly supplied to the Bay by major flood events. Aboriginal fire management and periodic cyclones and wildfires were the only disturbance to vegetation cover (Hall, 1990).

Clearing of vegetation began soon after exploration by Oxley in 1823, settlement at Brisbane in 1825, and the opening to free settlers in 1842 (McCleod, 1990). Land use in the catchment area of Moreton Bay reflects: (i) the pattern of settlement and land tenure; (ii) the capability of land to support cropping, pastoral and forestry activities in rural areas; and (iii) the occurrence of particular resources such as minerals, sand, rock and timber. Cropping occupies 5%, grazing and private forestry 65%, public land 17% and urban uses occupy 13%.

Land cover correlates closely with land use. The original vegetation cover has been removed from areas used for cropping, plantation forestry and urban purposes; partially removed from grazing and low density residential areas; and is relatively intact in areas set aside for conservation as National Parks and other reserves. Catterall *et al.* (1996) recorded the rate of vegetation clearing in southeast Queensland since European settlement, and found that only 34.6% of the original vegetation cover remains. The overall clearing rate since 1820 has been 8 450 ha per year, with more recent clearing rates (1987-1994) reducing to 3 340 ha per year.

Land degradation is widespread. As early as 1850 there are reports of land eaten bare by sheep and increasing erosion (McCleod, 1990). Severe soil erosion has been recorded from 10% of the Moreton region (Queensland Department of Primary Industries, 1974) where the most affected catchments are the Bremer River and Lockyer Creek. Clearing of vegetation for agricultural activities has almost ceased in the catchment, although some further expansion of coastal horticulture and sugar growing areas may be expected. The most significant land use changes occurring are: (i) the expansion of urban settlements in the north, south, western and Bayside corridors of Brisbane, and (ii) significant rural residential subdivision in the Beaudesert, Laidley and Caboolture commuting areas.

The future condition of Moreton Bay will depend on how this growth is managed to maximise vegetation protection and minimise the impacts of soil erosion and surface runoff into the tributaries of the Bay.

Introduction

Moreton Bay is the receiving water body for rivers and streams which flow from the catchments of Pumicestone Passage, Deception Bay and Bramble Bay in the north; the Brisbane River and its tributaries (Stanley River, Lockyer Creek and Bremer River) in the centre; and Redland Bay and the Logan and Albert Rivers in the south. The condition of these catchments determines

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 55-66.

the state of water quality in the watercourses and ultimately the Bay.

Throughout the catchment, land use and land management practices control the quality of water reaching the rivers, and the level of equilibrium reached between the volume of catchment runoff and the size of stream channels determines stream capacity and bank stability. Changes resulting from fire, clearing and land use cause a change in the volume, period of runoff and sediment load from the catchment, which requires an adjustment in stream channel shape to cope with new conditions. The deposition of sediment and reduced water quality in the lower reaches of streams or inshore areas of Moreton Bay has important implications for river capacity and flooding, and on marine flora and fauna.

This paper examines current land use patterns and practices in the catchments of Moreton Bay and relates these to the present land cover of remnant vegetation and land degradation. Conclusions are drawn on whether the catchments are moving towards or away from stability under the influence of continued land use changes.

History

Pre-European settlement

In its pre-European state, the catchment was relatively stable with sediment and nutrients predominantly supplied to Moreton Bay by major flood events and disturbance by aboriginal fire management, periodic cyclones and wildfires. Hall (1990) estimates aboriginal occupation of the region from 22 000 years ago, with major occupation of the coastal areas occurring since Moreton Bay was formed by the sea level rising to its present level approximately 6 000 years ago. He postulates that the most intensive occupation began between 4 000 to 4 500 years ago when there is evidence of vegetation change from vine forests to open eucalypt forests and grasslands as a result of more intensive ‘firestick farming’. Hall estimates population numbers in the region during this time at over 5 000, divided broadly into a coastal-littoral/marine ‘lifeway’ and an inland-terrestrial/riverine ‘lifeway’. The intensity of the fire management by the inland-terrestrial groups may have caused significantly increased rates of erosion with fertile sediment delivered to the shallow western Bay creating the basis for the rich fishing grounds utilised by the coastal-littoral groups.

European settlement

A much more dramatic change was to result from the arrival of Europeans, led by Oxley who explored the Brisbane River in 1823. Clearing of vegetation began soon after the establishment of Moreton Bay penal settlement in 1824, movement to Brisbane in 1825, and the opening to free settlers in 1842 (McCleod, 1990). Pastoral development of the Brisbane River region began in the 1840s. In the intervening 150 years, successive waves of settlement have developed the forest, land and water resources for human use to supply the needs of the expanding local, state and national population.

As has happened throughout Australia, land development proceeded as an experiment to apply and adapt European methods to local conditions. Understandably, many failures occurred such as the early attempts to grow sugarcane in the Brisbane area. Gradually, however, a large body of knowledge has grown on how these resources can be used in a way which is beneficial to society without diminishing their quality or long term stability. The result is seen in the current pattern of land use in the catchments.

Land Use

Factors affecting land use

Land use in the catchment of Moreton Bay (Figure 1; Table 1) reflects several fundamental factors, including: (a) the pattern of settlement and land tenure; (b) the capability of land to support cropping, pastoral and forestry activities in rural areas; and (c) the occurrence of particular resources such as minerals, sand, rock and timber. Of the total area of the catchment, cropping occupies 5%, grazing and forestry on private land 66%, public land 17% and urban and rural residential uses occupy 11%. Figure 2 shows the distribution of land uses and remnant vegetation.

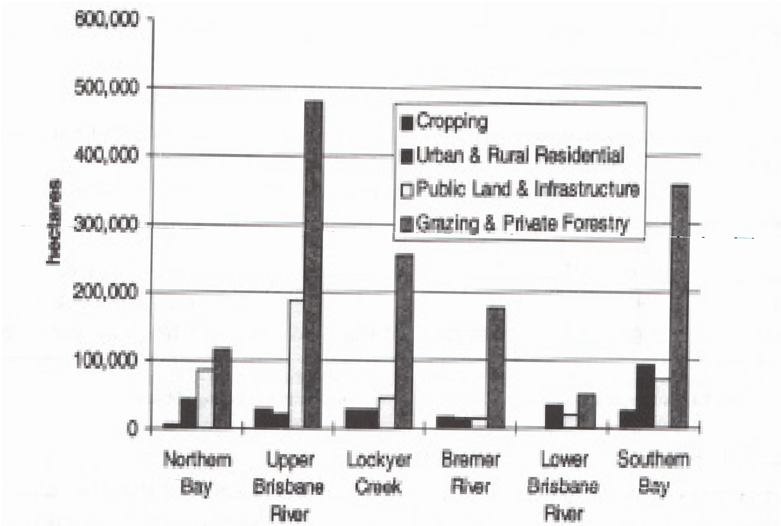

Figure 1. Land use in sub-catchments of Moreton Bay, 1996.

Land used for agriculture, grazing and private forestry dominates, accounting for 71% of the catchment. The emphasis of agricultural production is moving from the development of new areas towards the consolidation and the intensification of existing production areas. The principal issues affecting agricultural producers within the catchment are farm profitability, commodity prices received in a subsidised international marketplace and the need for planning measures to protect land resources. Of equal importance is the management of land resources to avoid land degradation, protect water resources and achieve sustainable production systems.

Urban

Residential development in both urban and rural settings is centred on the Brisbane area as the focus for metropolitan growth for both the region and the State of Queensland. Under the dual influences of population growth and increased commuting distances, urban development will be the most active form of land use change for the foreseeable future. On current lifestyle and urban density trends, the population increases in the region will occupy an additional 155 000 ha over the next 20 years, resulting in 7 500 ha of bushland, agricultural land and other rural land being consumed for housing and other urban purposes each year. Land clearing rates over recent years also indicate that during this time almost half of this area will be developed at the expense of natural vegetation.

Table 1. Land use in the Moreton Bay catchment (hectares), 1996

Sub-catchment	**National parks/ forest reserves**	**Other state land**	**Grazing/ forestry**	**Cropping**	**Water & transport**	**Urban & rural residential**	**Total**	***Remnant bushland (% of sub-catchment)***
Northern Bay	43 140	35 000	115 390	6 000	7 990	41 570	**249 090**	*23.7*
Upper Brisbane & Stanley River	151 760	17 690	479 930	28 540	18 240	20 370	**716 530**	*26.2*
Lockyer Creek/Middle Brisbane River	23 970	15 240	252 770	29 180	5 050	29 570	**355 780**	*31.8*
Bremer River	7 590	3 990	175 240	15 430	3 610	13 710	**219 570**	*15.7*
Lower Brisbane River	5 200	6 150	49 900	930	7 460	33 890	**103 530**	*31.0*
Southern Bay	46 200	13 710	356 720	25 420	10 920	92 380	**545 350**	*26.2*
Total	**277 860**	**91 780**	**1 429 950**	**105 500**	**53 270**	**231 490**	**2 189 850**	***26.0***

Note: Land use figures exclude off-shore islands. Sub-catchment boundaries are shown on Figure 2;
Source: SEQ 2001 Regional Resource Unit, Department of Local Government and Planning, Brisbane.

Agriculture

Cropping occupies five percent of the catchment, and is located generally on alluvial and lower slopes in five major areas: the Lockyer Creek and its tributaries; the Fassifern Valley; the Logan River; the Cressbrook and Buaraba Creeks in the Upper Brisbane River Valley; and the coastal areas of Woongoolba and the Glasshouse Mountains. Small production areas on isolated basalt soils occur at Redland Bay, Rochedale and Mount Tamborine.

The Lockyer Valley, which includes Laidley, Flagstone, Ma Ma and Tenthill Creeks, is the most intensively developed area producing a wide variety of vegetables, cereals, fodder crops and fruit, in particular, potatoes, onions and pumpkins. The Fassifern Valley based on Warrill Creek is the other major agricultural area producing cereals, lucerne, soybeans, pumpkins and potatoes both alone and in conjunction with animal production enterprises. Both areas are dependant on irrigation for reliable production with water resources based on underground supplies in the Lockyer Valley and surface water supplied from Moogerah Dam in the Fassifern Valley. Agriculture in the Logan River area is focused more on feed crops for animal industries, but also produces vegetable and cereal crops.

Cropping on the upper Brisbane River concentrates on cereals, soybeans, hay and fodder crops in association with grazing. In order to provide both water supplies and flood mitigation for the Brisbane metropolitan area, both the Somerset and Wivenhoe Dams have inundated significant agricultural resources.

The coastal agricultural areas with higher rainfall are used for growing sugarcane in the Woongoolba area south of the Logan River, supplying the Rocky Point sugar mill; and a wide range of horticultural crops (pineapples and subtropical tree crops) in the Glasshouse Mountains area.

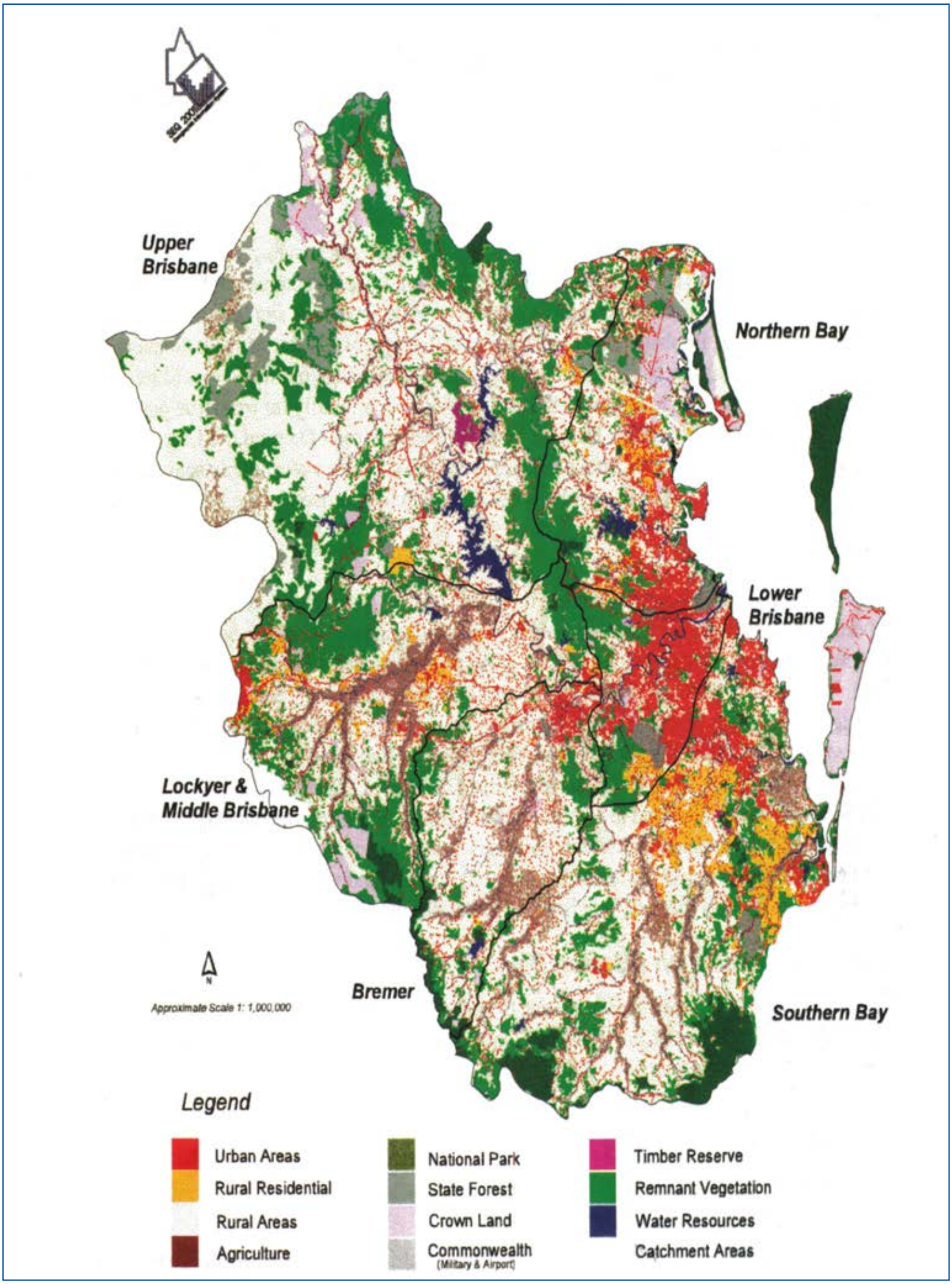

Figure 2. Moreton Bay catchment land use and remnant vegetation, 1996.
Source: SEQ 2001 Regional Resource Unit, Department of Local Government and Planning, Brisbane.

The proportion of land used for cropping (5%) is less than the proportion of arable land (8%), however, 34% of all cropping is located on land which is either marginal or unsuitable for this use (Capelin, 1990). Land used beyond its capability in this way is susceptible to serious soil erosion with implications for downstream water quality and reduced future productivity.

Water supply

Numerous water storages have been constructed to supply the growing population of the catchment with water. These occupy 2% of the catchment. Major structures are: the Wivenhoe and Somerset Dams on the Brisbane - Stanley Rivers; Perseverance and Cressbrook Creek Dams in the Upper Brisbane River catchment (which take water out of the catchment to supply Toowoomba); the North Pine Dam and Lake Kurwongbah in Pine Rivers Shire; and the Leslie Harrison Dam in Redlands Shire. Storages built primarily for irrigation purposes in the catchment are Moogerah Dam on Warrill Creek, Maroon Dam on the Logan River system, and Atkinson Dam on the Lower Lockyer Creek. Future water supply sources have been identified in the Logan River catchment for a planning horizon of the next 100 years.

Conservation

Approximately 58 000 ha is included in an established network of national parks and environmental parks under public ownership, representing almost 3% of the catchment. These parks contain areas representative of 88% of the region's major vegetation types (Regional Planning Advisory Group, 1993). The subcoastal lowlands contain the poorest representation of vegetation types in conservation reserves where only 35% of vegetation types are represented. The coastal lowlands bordering Moreton Bay have the richest diversity of vegetation types (36) of which 72% are represented in reserves. Catterall *et al.* (1996) estimate that 56% of remnant bushland in southeast Queensland is located on freehold land compared with only 35% in national parks, state forests or timber reserves. These figures emphasise the need for conservation strategies for privately owned land where a large proportion of remnant vegetation is located. This issue is further discussed under *Land cover*.

Land use impacts on Moreton Bay

The major impacts on Moreton Bay are due to land development, particularly urbanisation of the catchment. The State of Environment Report for Brisbane City (Brisbane City Council, 1996) lists the following impacts on the Bay:

- significant increase in short term sediment yield, due to the removal of vegetation cover, estimated at up to 30 times larger than before land clearance;
- accelerated stormwater runoff;
- reduced number of aquatic organisms;
- increased time and distance for assimilation of nutrients, and consequently an upward trend in the incidence of pathogens harmful to humans;
- increased tidal prism, tidal discharge, tidal range and tidal velocity; and
- solid and liquid waste from the catchment.

Commercial development at the Port of Brisbane (due to expand from 150 ha to 550 ha) and mainland Bayside residential development has led to the reclamation of significant areas of foreshore for canal developments, marinas and port development with the associated increased threat of oil and petrol spills. The proportion of land uses in Brisbane City are given in Table 2.

Over 40% of the City comprises urban residential, commercial, industrial and transport uses which provide hard impervious surfaces. Such surfaces cause rapid, unconstrained runoff leading to short term flood peaks, giving little opportunity for sediment and pollutants in surface flows to be intercepted before entering streams and ultimately Moreton Bay. A high proportion (50%) of Brisbane City remains either vacant, under bushland, or under some form of agriculture.

Land Cover

Land cover correlates closely with land use and is a useful indicator of catchment condition.

Table 2. Land use in Brisbane City, 1991.

Broad land use category	Area (ha)	% of total
Urban*/Low density residential	29 210	26.0
Commercial*/Industrial*	7 440	7.0
Institutional*/Airport*	6 990	6.0
Currently under development*	1 015	1.0
Utilities*/Major transport*	705	0.6
Major recreational	3 695	3.0
Bushland/Reclaimed land*/Vacant land*	44 645	39.0
Agriculture/arable land	11 700	10.0
Roads* and water courses	9 100	8.0
Total	**114 500**	**100.0**

* Urbanised component of land use in Brisbane City (excluding Moreton Island);
Source: Brisbane City Council (1996).

A fully vegetated catchment with adequate ground cover will be stable and contribute only background levels of sediment and nutrients to runoff. Any disturbance through fire, clearing or cultivation which removes this cover will lead to increased soil erosion and the movement of significant amounts of sediment and nutrients into streams and downstream water bodies with major impacts on water quality and stream capacity.

Vegetation clearing

The original vegetation cover in the catchment has been fully removed from areas used for cropping, plantation forestry and urban purposes; partially removed from grazing and low density residential areas; and is relatively intact in areas set aside for conservation as national parks and other reserves. Catterall *et al.* (1996) has recorded the rate of vegetation clearing in southeast Queensland since European settlement, finding that only 36.5% of the original vegetation cover remained in 1987, declining to 34.6% in 1994 (Table 3). The overall clearing rate since 1820 has been 8 450 ha per year, with more recent clearing rates (1987-1994) reducing to 3 340 ha per year.

Data from the Moreton Bay catchment show that only 26% of the total area remains uncleared, with the Bremer catchment the most affected where only 15.7% of the area remains vegetated (Table 1).

Table 3. Summary of losses of bushland cover in the SEQ 2001 region 1820-1994.

lost	Area of bushland at start of period	Percent of bushland at start of period	Percent of bushland at end of period	Percent of original bushland per year
Net bushland losses	(ha)	(%)	(%)	(%)
1820 - 1987	2 225 148	100.0	36.5	0.38
1987 - 1989	811 713	36.5	35.4	0.28
1989 - 1994	742 820	35.4	34.6	0.47

Source: Table 4.3 in Catterall *et al.* (1996).

Clearing of vegetation for agricultural activities has almost ceased in the catchment with most suitable land now developed, although some further expansion of coastal sugar-growing areas may be expected. The most significant land use changes now occurring are the expansion of urban settlements in the north, south, Bayside and western corridors of Brisbane, and significant rural residential settlement in the Beaudesert, Laidley and Caboolture commuting areas.

A case study of land use change within the Pumicestone Passage catchment between 1974 and 1991 reveals that the single greatest change in land use in that catchment has been the reduction in the area of native vegetation by 8 500 ha or 14% of the total catchment (Table 4). The major land uses which replaced the natural vegetation have been pine plantations (5 448 ha), pastures (1 195 ha) and urban development (1 076 ha) (Queensland Department of Environment and Heritage, 1993).

Distribution of vegetation

Most clearing has occurred and continues to occur in lowland areas on low slopes (Catterall *et al.*, 1996) (Table 5). Only 21.6% of the original vegetation below 60 m elevation remained

Table 4. Pumicestone Passage catchment land use by category in 1974 and 1991.

Land use of category	1974 land use area ha	(%)	1991 land use area ha	(%)	1991 land use 1974 natural vegetation	(%)
Urban	1 392	(2.3)	2 845	(4.7)	1 076	(3.4)
Transport	363	(0.6)	726	(1.2)	195	(0.6)
Pasture	3 632	(6.0)	4 418	(7.3)	1 195	(3.8)
Horticulture	3 874	(6.4)	5 387	(8.9)	415	(1.3)
Littoral vegetation	2 966	(4.9)	2 724	(4.5)	2 724	(8.7)
Native vegetation	28 326	(46.8)	20 095	(33.2)	20 095	(64.2)
Pine forest	19 610	(32.4)	23 363	(38.9)	5 448	(17.4)
Rural residential	363	(0.6)	787	(1.3)	195	(0.6)
Totals	**60 526**	**(100.0)**	**60 526**	**(100.0)**	**31 292**	**(100.0)**

in 1987, and by 1994, this figure had declined to 19.4%. It is this lowland area that is under most threat from urban and rural residential development as a result of population growth. Historically, clearing rates of inland (>10 km from the coast) lowland vegetation for agriculture had been higher, leaving only 17% of the original vegetation in 1987, and 15.1% by 1994. However, more recent clearing rates within 10 km of the coast now exceed the inland rates as a result of urban and coastal development. Only 28.5% of lowland vegetation within 10 km of the coast remained in 1987, and by 1994, this figure had declined to 25.9%.

Littoral vegetation, which forms the western fringe of Moreton Bay, has also suffered significant decline due to development activity. Since European settlement, approximately 20% of mangrove forests and 10% of the saltmarsh/claypans of Moreton Bay have been destroyed by human activity. Major losses have occurred since the 1970s, principally due to the development of the Brisbane Airport. Net losses over this time are estimated at 1 241 ha of mangrove forest and 580 ha of saltmarsh (Quinn, 1992).

These results pose particular challenges to public policy makers wishing to maintain biodiversity, water quality and other indicators of environmental quality in the catchment and Moreton Bay. Whilst not documented in a direct sense, there is clearly a relationship between

Table 5. Bushland clearing within land categories indicative of the risk of loss, within freehold land in the southeast Queensland mainland.

	Bushland lost between 1987 and 1994		% of land area cleared per year	
Risk category	**Area (ha)**	**% of 1987 bush**	**1820-1987**	**1987-1994**
Coastal lowlands+ <10 km, <60 m	5 424	14.1	0.48	0.41
Other lowlands >10 km,<60 m	6 162	13.0	0.51	0.28
Coastal slopes <10 km, 60 - 160 m	120	3.1	0.30	0.22
Other slopes >10 km, 60 - 160 m	5 602	5.2	0.49	0.13
Mountains >160 m	2 728	1.0	0.35	0.06
All freehold land	20 036	4.3	0.44	0.16

\+ Includes Bribie Island due to road and bridge connection;
Source: Table 7.1 in Catterall *et al.* (1996).

population growth and decline in environmental indicators. Land cover, with its related impacts on loss of biodiversity and water quality will continue to decline in the face of population growth in the catchment. As the clearing rates for land held in different tenure show (Table 6), attention needs to be given to the rate of clearing on State controlled land as well as that occurring on freehold land.

Land Degradation

Land degradation is the loss of land quality which may be measured in terms of destruction of the land resource (gullying, land slip), loss of agricultural productivity (soil erosion, salinity) or damage to an external resource (sedimentation of reservoirs and rivers). Land degradation is widespread in the catchment. There are reports as early as 1850, within 10 years of free settlement, of land eaten bare by sheep and increasing erosion (McCleod, 1990). It is beyond doubt that there was some land degradation occurring in the catchment prior to European settlement due to fires, extreme weather events, flooding and earth movements. However, it

Table 6. Bushland clearing by tenure within southeast Queensland.

	Bushland lost between 1987 and 1994		% of land area cleared per year	
Risk category	**Area (ha)**	**% of 1987 bush**	**1820-1987**	**1987-1994**
National parks	110	0.1	0.02	0.02
State forestry timber reserves	820	0.4	0.10	0.05
Lands Dept lease	415	2.1	0.29	0.16
Other Crown	836	4.3	0.39	0.21
Commonwealth/Other	117	1.4	0.23	0.12
Freehold	20 036	4.3	0.44	0.16
All land	22 334	2.9	0.39	0.15

Source: Table 6.6A in Catterall *et al.* (1996).

may be considered to have been negligible compared with the extent of degradation which the changes in land use over the past 150 years have caused.

Soil erosion

The Moreton Region Non-urban Land Suitability Study (Queensland Department of Primary Industries, 1974) revealed that 225 400 ha, or 10% of the rural lands in the region, are subject to severe soil erosion as a result of agricultural and development practices. A further 1 568 100 ha (70%) suffer from less severe erosion and 481400 ha (21%) suffer only negligible erosion (Table 7). Severe soil erosion affects 14% of the Brisbane River catchment (Capelin, 1990). Lands affected by moderate or slight erosion occur throughout but are concentrated in the Upper Brisbane and Lockyer catchments. Subsequent studies into land degradation in the Bremer catchment (Johnston,1979) and the Lockyer catchment (Shaw, 1979) confirmed these figures, defined the problem in more detail and developed land management strategies to control land degradation.

Table 7. Existing soil erosion in the Moreton region, 1974.

Erosion type (%)	**Area (ha)**	**Percentage**
Negligible	481 400	21.2
Slight sheet and rill	1 050 200	46.1
Moderate sheet and minor gullying	517 900	22.8
Severe sheet and minor gullying	197 100	8.7
Severe gullying	23 100	1.0
Severe gullying with lateral extension	5 200	0.2
Total	**2 274 900**	**100.0**

Acid sulfate soils

Intensive development for canals, residential areas and agriculture has resulted in degradation of near coastal areas through habitat destruction and disturbance of acid sulfate soils (ASS). Soils with high concentrations of subsoil sulfate formed under marine conditions are common on the coastal plains below approximately 5 m Australian Height Datum (AHD). When disturbed through excavation or the lowering of the water table and exposure of the subsoil to air, sulfates are converted to sulfuric acid which can move into local waterways and inshore areas causing dramatic loss of water quality and significant fish kills. Methods of identification and management of ASS areas are now available so that susceptible areas can be managed to control the impact of development on water quality. However, as development of the coastal plain intensifies, this issue will need to be adequately addressed by both developers and governments.

Causes and solutions

The causes of land degradation are straightforward and well understood. The equilibrium achieved between rainfall, vegetation, soil and groundwater under natural conditions is disturbed when land is cleared for grazing, cropping uses or other development. Soil is exposed to the impact and runoff effects of rainfall causing erosion and steep lands lose the protective network of tree root systems causing land slip. In addition, in some areas, groundwater levels rise in the absence of moisture-demanding trees causing waterlogging and saline outbreaks.

The methods required to address these problems are now known and available. What is needed is the commitment of the community, government and industries to implement the necessary changes to attitudes, land management practices and land use to achieve a return to a stable, productive catchment. These institutional and attitudinal changes are discussed elsewhere (e.g. Pressland *et al.*, this volume).

Condition of the Catchment

A report on the condition of Queensland river catchments (Queensland Department of Primary Industries, 1993) has recorded that in the grazing lands of the Brisbane River catchment, landslips are common on cleared steep land along the Great Divide. Overstocking has caused extensive sheet and gully erosion on fragile soils, and fires have changed species composition and reduced biomass. Severe gully erosion has occurred on texture contrast soils in the Murphy's Creek and Helidon areas, and tunnel erosion is widespread in coarse-grained sediment (sandstone) areas. Invasion of woody weeds, such as lantana, is also a significant problem.

In the catchment areas of the southern Bay, the upper inland sections of the Logan and Albert Rivers exhibit landslips on steep, cleared grazing land along the Border Ranges, and extensive sheet and gully erosion occurs on fragile soils due to the combination of high stocking rates and excessive burning. The upland scrub soils and lighter textured soils have undergone rapid fertility decline and loss of soil structure. Salinity problems have developed in the mid to upper catchment valleys where vegetation has been cleared resulting in rising groundwater levels.

The catchment of the northern Bay and Pumicestone Passage has suffered similar problems to other grazing lands through soil erosion, woody weed invasion (e.g. groundsel bush) and overgrazing. Horticultural land on steep slopes in high intensity rainfall areas is prone to very high rates of soil erosion in excess of 100 t/ha/yr; however, the trend to perennial tree crops represents a more sustainable use of these soils. Nutrients and high fertiliser inputs are required to alleviate the decline in soil fertility, but these practices can lead to nutrient enrichment of watercourses and storages through erosion and runoff.

Conclusion

The future condition of Moreton Bay will depend on how population growth is managed so as to maximise vegetation protection and minimise the impacts of soil erosion and surface runoff into the tributaries of Moreton Bay. The Moreton Bay catchment has been developed for agricultural, forestry and urban uses up to and, in some areas, beyond its capability for such uses, while industrial and transport uses continue to demand their share of land resources.

The pioneering phase of development has passed and the phase of reassessment and consolidation is currently underway to determine the realistic limits to resources in the region. The SEQ 2001 Project has started this process at the regional level, and the Brisbane River Management Group is continuing it at the catchment level. The tasks of repair and adjustment of land uses to achieve a catchment which is both productive and stable will be the responsibility of landholders with the guidance and assistance of government and industry.

We are still learning how to manage Australia's climate and land resources in a sustainable way. Whilst the health of the rivers and Moreton Bay will be a constant reminder that we are dealing with finite resources with interdependencies between the marine, littoral and terrestrial environments, the catchment community must seek the means to manage population growth and its impacts. While this may mean short term sacrifices in terms of either development controls or increased residential densities, such tradeoffs may be necessary to preserve the quality of Moreton Bay and our lifestyle which are so closely linked.

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The Demographic Future of the Moreton Region

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Abstract

This paper provides a background on the history and future of the human population of the Moreton Region, which is defined as the greater Brisbane region stretching from Noosa in the north to the New South Wales border in the south and west to the Great Dividing Range. This region is experiencing one of the fastest population growth rates of any large city in the developed world and is the fastest growing capital city in Australia. Most of this growth over the last three or four decades has come from migration – not natural increase. Because Brisbane is not a "world city" as defined by Friedman and others, some people question whether its growth can be sustained for long. However, "world city" status appears to have little to do with short term population growth. We expect that greater Brisbane's growth will continue for another decade at least and that most of this growth will be along the coastal strip. Consequently this rapid population growth is likely to place ever greater stresses on the environment of the Moreton catchment.

Introduction

This paper analyses the recent demographic history of the Moreton Region and examines a variety of future scenarios for the population of this rapidly changing region. The Moreton Region is defined as Brisbane and Moreton Statistical Divisions (SDs), stretching from Noosa in the north to the New South Wales border in the south and extending westwards from the coast to the Great Dividing Range. While including some small areas in the north and south which, strictly speaking, are not in the Moreton Bay water catchment area, most of this Moreton Region is in the Moreton Bay "environmental catchment" area. We shall refer to this Moreton Region also as the Brisbane Region, Brisbane Urban Region and Southeast Queensland (SE Qld), the terms being used interchangeably to avoid repetition. Unless otherwise specified "Brisbane" also refers to the Brisbane Region as a whole.

The Moreton Region presently contains just over two million persons and is growing at around 2.9% per annum. As we shall show, this makes the region one of the three or four fastest growing large urban areas in the developed world. Moreover, this rapid growth has extended over several decades and is likely to continue at least another decade or more into the future. For these reasons it is essential to understand something of the demographic growth of the region, in terms of overall numbers and their geographic distribution as well as the components of growth, in order to grasp the significance of possible futures.

In order to accomplish these objectives we look first at a brief history of population growth and its changing distribution in SE Qld. We compare this growth with that of other Australian cities and with the fastest growing cities of the developed world. Finally we examine some scenarios of future growth in SE Qld and the likely outcomes in terms of distributions.

Growth in a World Context

The world's population has grown rapidly since the turn of the century from 1.6 billion to 5.6 billion, an absolute increase of four billion persons (Figure 1). It is presently growing at around 1.6% p.a. Although most areas of the world, including Asia, have slowed their growth over the last two decades, Africa, Western Asia and Central America are still growing at very

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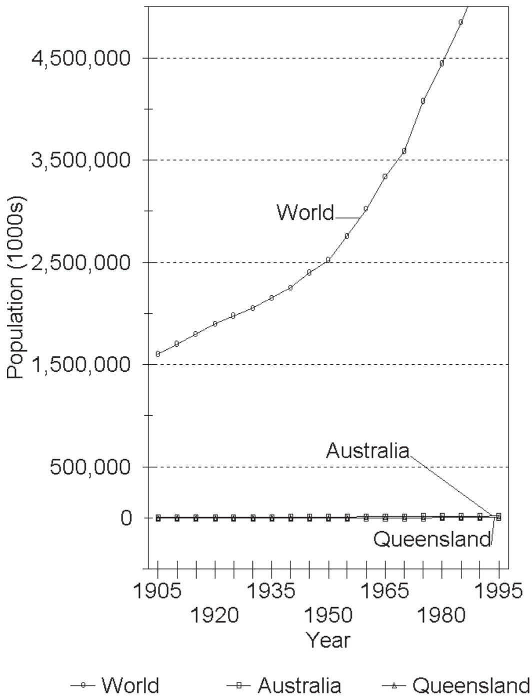

Figure 1. Population of Queensland and Australia compared to world population, 1905-1995. Source of data: United Nations (various), Australian Bureau of Statistics (various A).

Table 1. Average annual growth rates, selected areas of the world, 1990-95.

	%
Least developed countries	2.82
Africa	2.81
Western Asia	2.43
Central America	2.23
Less developed regions	1.88
Latin America & the Caribbean	1.84
Southeastern Asia	1.81
South America	1.74
Asia	1.64
World	**1.57**
Oceania	1.54
Australia	1.37
New Zealand	1.24
China	1.11
Northern America	1.05
More developed regions	0.40
Europe	0.15

Source: United Nations (various).

high rates in excess of 2% p.a. (Table 1).

The population of Australia has more than doubled in the last forty years to over 18 million, with growth at an average rate of 1.7% p.a. or more than 200,000 persons p.a. Despite the fact that Australia's growth has slowed from an average annual rate of 3.3% after World War II to the present 1.4%, Australia is still growing at a faster rate than many countries in East Asia including China (Table 1). Our growth is more akin to that of less developed regions than that of the more developed regions. Nevertheless, as Figure 1 shows, the population of Australia is insignificant when compared with the total number of people in the world and represents only 0.3% of the global population.

However, if the same data are graphed with population on a semi-logarithmic scale, a rather different picture emerges. In Figure 2, slope represents rate of increase and equal slopes represent equal rates of increase. Therefore, it can be seen that the rates of growth of Australia

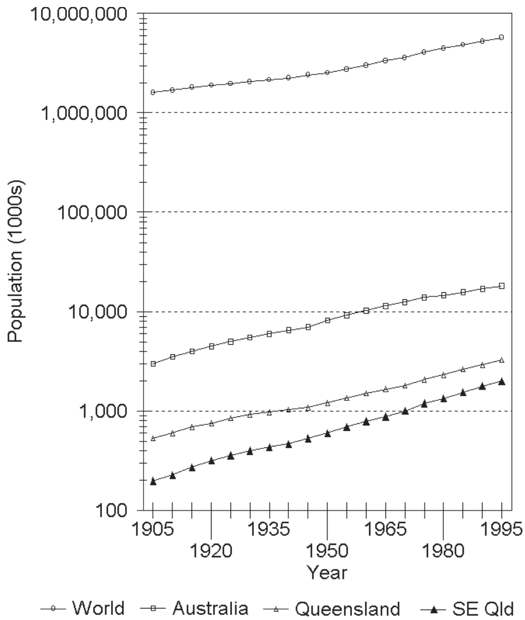

Figure 2. Population of Queensland and Australia compared to world population (semi-logarithmic graph), 1905-1995. Source of data: United Nations (various), Australian Bureau of Statistics (various A), Australian Bureau of Statistics (various B).

and Queensland have virtually paralleled that of the world over the last century.

Queensland, with a total population of more than three million, is the fastest growing state in Australia and is currently growing at over 2% per annum. The growth of the state is expected to continue at a relatively high rate for at least the next decade, with much of this population increase being due to interstate migration from southern states. There has been a substantial redirection of population growth away from the traditional growth areas of Melbourne and Sydney, principally to Queensland. However, some researchers believe that Brisbane's growth will cease or even reverse in the not too distant future (Birrell *et al.,* 1995; Stimson & Taylor, 1996). Nevertheless, there appears to be an important lesson here; the best "predictor" of population growth in Australia is growth in the world, and that also applies to a degree to both Queensland and SE Qld. It is emphasised here that we do not imply that world population growth directly causes population growth in Australia and Queensland but rather, for reasons not yet understood, the growth of these latter areas seems to follow world trends closely.

Components of Growth in Southeast Queensland

Prior to World War II the majority of growth in SE Qld came from natural increase (the excess

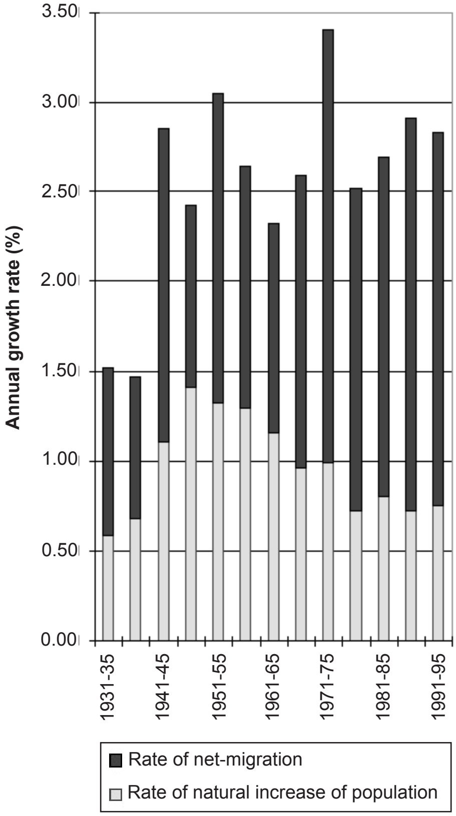

Figure 3. Population growth of SE Qld divided into natural increase and net migration, 1931-35 to 1991-95. Source of data: Calculated from ABS data.

of births over deaths) rather than migration. Since World War II the rate of growth attributed to natural increase has steadily declined and in the last twenty years has been less than 1% annually (Figure 3).

Since the late 1960s, the principal component of growth in SE Qld has been migration. It is primarily internal migration from elsewhere in Queensland and from the southern states that has become more important than natural increase to Brisbane's population growth. The Brisbane Region receives a low level of overseas migration compared to Sydney and Melbourne.

Comparison of SE Qld's Growth with that of other Urban Regions

Queensland's population growth is becoming increasingly concentrated in the southeast corner and Figure 4 shows SE Qld in relation to other selected cities of Australasia. Clearly, SE Qld is not the largest urban area in Australasia since Sydney has dominated in both total population and overall growth since World War II. However, the population of the Moreton

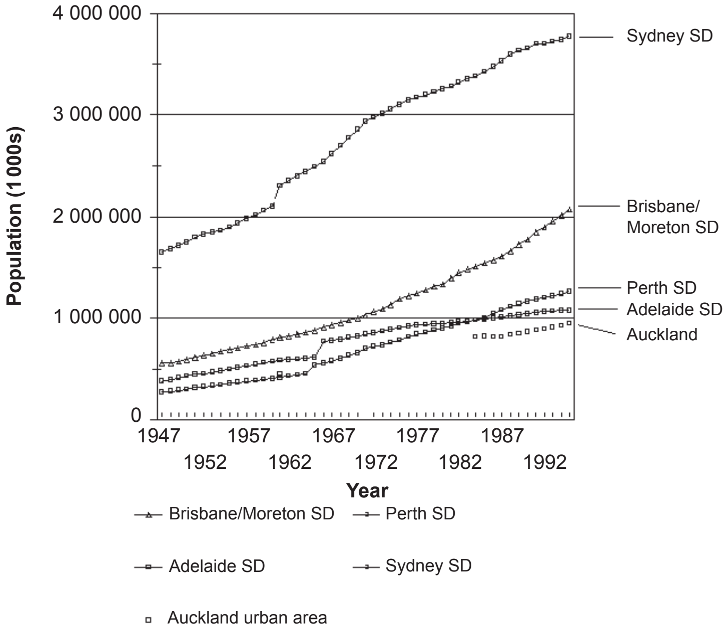

Figure 4. Comparison of population growth rates in selected regions, 1947-1995. Source of data: United Nations (various), Australian Bureau of Statistics (various A), Australian Bureau of Statistics (various B).

Region today approximates that of Sydney in 1957 and the most recent rapid growth rates in SE Qld have surpassed those of Sydney and Melbourne. Compared with cities of comparable size in 1947 (Perth, Adelaide and Auckland), the Moreton Region has far outgrown them.

Over the last decade, the Brisbane Region has maintained steady growth averaging 3% p.a. This compares with an average annual growth of 2.3% for Perth SD, 1.5% for the Auckland urban area, and less than 1% p.a. for both Adelaide and Sydney SDs, between 1985 and 1995. Whilst growth in Perth and Auckland has fluctuated over the last decade, with Auckland declining in population in the late 1980s, the Brisbane Region has remained consistently high.

At this point in time it is difficult to forecast whether this signals a long term change in the relativities of Sydney and Brisbane because of structural changes in the economy, or whether Brisbane's present pre-eminent growth is just a temporary phenomenon. Some commentators suggest that Brisbane is not participating in the world economy to the same degree as Sydney and thus its rapid growth may be only temporary (Birrell *et al.,* 1995; O'Connor & Stimson, 1995).

As Table 2 shows, the Brisbane urban region is probably the third or fourth fastest growing large urban region in the developed world. Here we take the UN classification of the developed world as Europe, North America, Australia, New Zealand and Japan – basically the OECD countries; by large urban regions we mean those that had more than one million persons in the early 1990s.

Of the top ten fastest growing regions, two are in Canada, five are in the USA and two are in Europe. Anecdotal information indicates that Berlin may be growing very fast due to the number of undocumented immigrants, but official UN sources do not confirm this growth. Sapporo is the only large urban region in Japan with growth approaching that of the Moreton Region but it does not reach the top ten list.

Table 2. The fastest growing large urban regions in the developed world, 1990-95.

Rank	Urban Region	1995 Population '000s	Average annual growth 1990-95 %
1	Toronto, Ontario, Canada	4 084[1]	3.66[1]
2	Vancouver, British Columbia, Canada	1 678[1]	3.46[1]
3	Atlanta, Georgia, USA	3 432	2.88
4	**Brisbane-Moreton-Tweed, Australia**	**2 109[2]**	**2.86[2]**
5	Phoenix-Mesa, Arizona, USA	2 564	2.68
6	Denver-Boulder-Greeley, Colorado, USA	2 233	2.37
7	Lisbon, Portugal	1 863	2.33
(7	Orlando, Florida, USA	1 391	2.33
9	Düsseldorf, Germany	3 031[3]	2.31[3]
10	Portland-Salem, Oregon, USA	2 022	2.28

Note: These are cities which had at least one million persons at the beginning of the 1990s;
[1] Data for 1992, growth 1986-1992;
[2] Comprises Brisbane and Moreton Statistical Divisions in Queensland and Tweed Shire Part A in New South Wales;
[3] Düsseldorf includes Düsseldorf, Mönchen-gladbach, Remscheid, Solingen and Wuppertal;
Source: US Bureau of the Census (1996), United Nations (various), United Nations (1995), Australian Bureau of Statistics (1996).

In the 1970s Friedman developed the concept of "world city" to describe those few cities that dominate the economic, cultural and political structure of most of the earth's populations at any given time. While London, New York and Tokyo have remained the pre-eminent world cities of the last few decades, Friedman (1995) has recently identified thirty cities which he designates as "world cities" (Table 3).

Interestingly, only Toronto, Vancouver and Düsseldorf, among the top ten population growth cities, are ones that are designated by Friedman as "world cities". Perhaps being a "world city" is not critical for rapid urban population growth or perhaps this delineation of "world cities" is biased towards historical measures rather than those which measure importance to a present or future world.

When the Applied Population Research Unit (APRU) of the University of Queensland performed a similar exercise in 1993 to determine the ten fastest growing cities of the developed world, it came up with a somewhat different list (Table 4). That list consisted solely of nine cities in North America plus Brisbane, and only five of these cities now appear in the top ten. Whilst the Brisbane Region, Atlanta, Toronto and Vancouver have all increased their ratings, all the Californian cities have now fallen off the list, illustrating the slowing of growth in California. In fact, California has had a net domestic migration loss of more than a quarter of a million persons per annum over the last five years to the other states of the USA.

It is also noteworthy that in terms of population growth, the so-called "sun-belt" cities have fared relatively badly recently and "snow-belt" cities such as Toronto, Vancouver, Portland and Denver have overtaken them. Moreover only three of the top ten cities, perhaps four if Portland is included, could be considered "oceanside" cities.

Table 3. Spatial articulations: 30 world cities, 1990-95.

1995 Articulations	population '000s	Average annual growth 1990-95 %
1 Global financial		
#*London	7 335	0.0
# New York	18 107	0.3
#*Tokyo	26 836	1.4
2 Multinational		
# Miami	3 443	1.4
# Los Angeles	15 362	1.0
# Frankfurt	3 606	0.8
# Amsterdam	1 109	1.0
* Singapore	2 848	1.0
3 Important national		
#*Paris	9 469	0.3
# Zurich	897	1.6
* Madrid	4 072	-0.5
* Mexico City	15 643	0.7
Sao Paulo	16 417	2.0
* Seoul	11 641	1.9
# Sydney	3 590	0.4
4 Subnational/regional		
Osaka-Kobe	10 601	0.2
# San Francisco	6 540	0.9
# Seattle	3 265	1.8
# Houston	4 164	2.1
# Chicago	8 590	0.8
# Boston	2 842	0.5
# Vancouver	1 678	3.5
# Toronto	4 084	3.7
Montreal	3 240	1.8
Hong Kong	5 574	0.7
# Milan	4 251	-1.6
Lyons	1 311	0.7
Barcelona	2 819	-0.6
# Munich	2 238	0.9
# Düsseldorf-Cologne-Essen-Dortmund	12 496	1.0

major immigration target;
* national capital;
Source: Friedman (1995), US Bureau of the Census (1996), United Nations (various), United Nations (1995).

Table 4. Comparison of the fastest growing large urban regions in the developed world, 1980-90 and 1990-95.

Rating (1980-90)	Rating (1990-95)	Urban region
1	5	Phoenix-Mesa, Arizona, USA
2	17	Sacramento-Yolo, California, USA
3	21	San Diego County, California, USA
4	13	Dallas-Fort Worth, Texas, USA
5	3	Atlanta, Georgia, USA
6	**4**	**Brisbane-Moreton-Tweed, Australia**
7	1	Toronto, Ontario, Canada
8	22	Tampa-St Petersburg-Clearwater, Florida, USA
9	20	Los Angeles-Riverside-Orange County, California, USA
10	2	Vancouver, British Columbia, Canada

Source: US Bureau of the Census (1996), United Nations (various), United Nations (1995), Australian Bureau of Statistics (1996).

The importance of these comparisons is that there is nothing to suggest that those things which are commonly held as "attractions" to SE Qld – sun and surf – ensure long term growth elsewhere in the world, at least at very high levels. On the other hand there is also nothing to suggest that cities may not achieve very high population growth over long periods even though they are not presently considered "world cities".

If we compare SE Qld's growth to two cities in the US which are sometimes held up as undesirable models for Brisbane's possible futures, Los Angeles and Miami, we see that Brisbane is remarkable for its consistent growth in the post-World War II period (Figure 5). We can also see that both Miami in the 1950s and 1970s and Los Angeles in the 1970s have had periods of very much more rapid growth than anything the Brisbane Region has experienced.

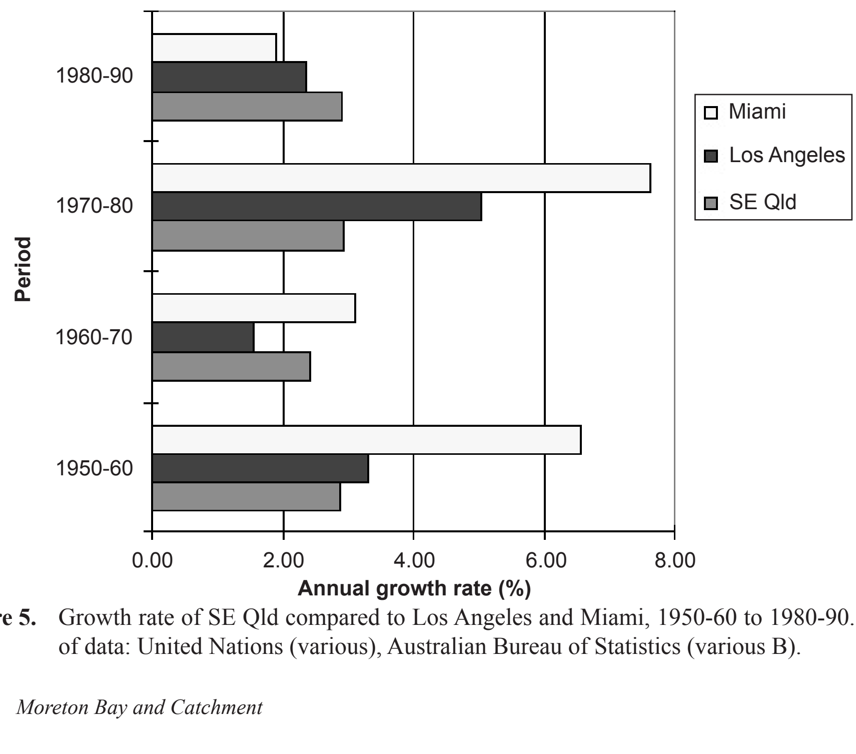

Figure 5. Growth rate of SE Qld compared to Los Angeles and Miami, 1950-60 to 1980-90. Source of data: United Nations (various), Australian Bureau of Statistics (various B).

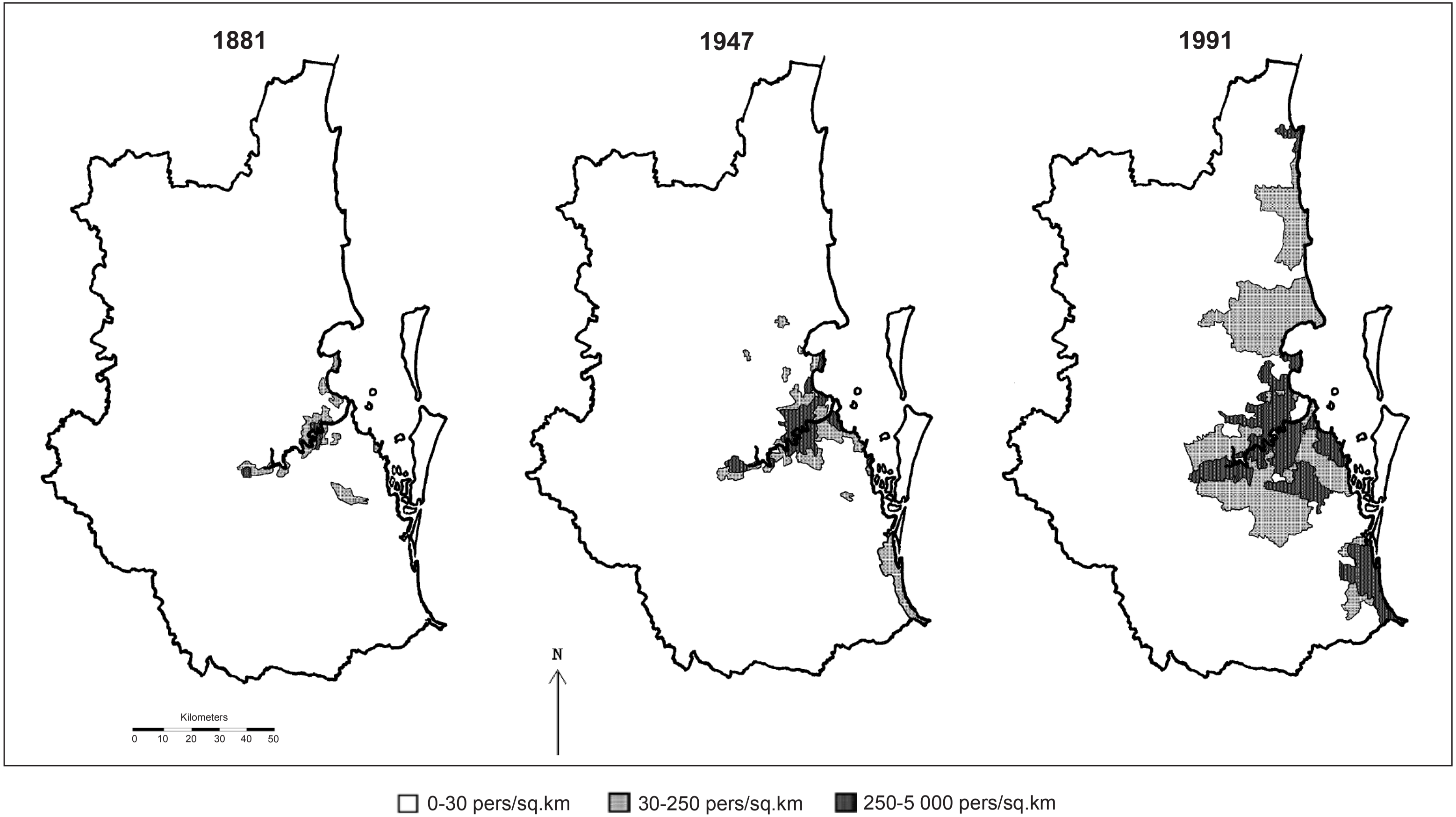

Figure 6. Changes in population density in southeast Queensland over the period 1881-1991. 1881: source of data: Office of the Registrar General 1881, Bureau of Census and Statistics 1969; 1947, source of data: Office of the Government Statistician 1949, Bureau of Census and Statistics 1969; 1991, source of data: Australian Bureau of Statistics 1993.

Finally these data suggest that very fast growth for large cities is most characteristic when their populations are around two to three million. Cities larger than that generally slow down in their population growth although even the largest may have short bursts of rapid growth.

Before proceeding to consideration of future scenarios, let us look at the historical growth of urban settlement in the Moreton Region (Figure 6). As can be seen from the maps, pre-World War II growth was concentrated along the Brisbane River well upstream from the coast. Only after World War II did rapid development begin to take place along the coast itself. The first major growth area was what became the Gold Coast; more recently the Sunshine Coast has had very rapid growth particularly in percentage terms.

Population densities in SE Qld vary from less than 30 persons/km^2 in rural parts to more than 250 persons/km^2 in the cities of Brisbane, Ipswich, Redcliffe and Gold Coast. Australian cities in general have densities comparable with those in the USA and well below those of European and most Canadian cities. Furthermore, in 1980 the population density of Brisbane was the lowest of any Australian capital city, and similar to that of other cities that currently rival it in terms of growth, for example Vancouver, Denver and Phoenix (Newman *et al.,* 1990).

Scenarios for Future Growth

The above analyses and comparisons indicate that most of the recent projections for SE Qld's future population are quite possible. Given the pitfalls of making population projections it is difficult to say which scenarios are the most likely and it is possible that the eventual actual populations may fall well outside the range of population projections provided here.

We have chosen four population projections which are commonly available and are produced by experienced projection agencies utilising accepted methodologies and reasonable assumptions on births, deaths and migration. These projections covering the period 1996 to 2006 are the Medium, High and Low series of projections produced by the Queensland Department of Local Government and Planning (DLGP) which, respectively, are Scenarios 1, 2 and 3. Scenario 4 is that produced by the APRU in 1995 (Figure 7).

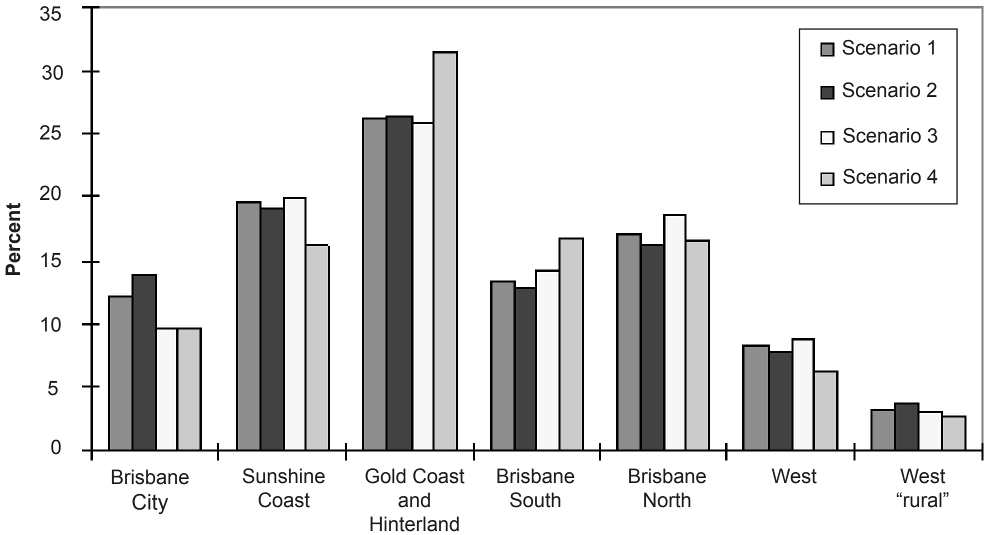

Figure 7. Projected share of population growth under four scenarios, Moreton Region, 1969-2006. Source of data: Queensland Department of Local Government and Planning (1996); Skinner *et al.* (1995).

A glance at Tables 5A,B reveals that the Gold Coast is still the major centre for population growth in SE Qld and will remain so under all four growth scenarios. The APRU scenario maintains Gold Coast's lead more than any of the DLGP scenarios.

The total growth projected for the next decade, under the four scenarios, varies from 430 000

Table 5A. Absolute population growth in the Moreton Region under four selected scenarios, 1996-2006.

1	2	3	4	
Brisbane City	65 060	89 830	41 420	47 920
Sunshine Coast	103 860	123 930	85 770	80 970
Gold Coast and hinterland	139 330	171 270	111 150	156 776
Brisbane South	71 220	83 380	61 480	83 430
Brisbane North	90 730	106 180	79 590	82 562
West	43 290	50 230	37 550	30 620
West "rural"	17 770	24 110	13 010	13 903
Brisbane/Moreton	531 260	648 930	429 970	496 181

Table 5B. Percentage share of population growth in the Moreton Region under four selected scenarios, 1996-2006.

	1	2	3	4
Brisbane City	12.25	13.84	9.63	9.66
Sunshine Coast	19.55	19.10	19.95	16.32
Gold Coast and hinterland	26.23	26.39	25.85	31.60
Brisbane South	13.41	12.85	14.30	16.81
Brisbane North	17.08	16.36	18.51	16.64
West	8.15	7.74	8.73	6.17
West "rural"	3.34	3.72	3.03	2.80
Brisbane/Moreton	100.00	100.00	100.00	100.00

Note: Gold Coast and hinterland includes Gold Coast (C) and Beaudesert (S); Brisbane South includes Logan (C) and Redland (S); Brisbane North includes Pine Rivers (S), Caboolture (S), Redcliffe (C); West is Ipswich (C); West "rural" includes the shires of Boonah, Esk, Gatton, Kilcoy, Laidley. Source: Queensland Department of Local Government and Planning (1996), Skinner *et al*. (1995). [(C) indicates City, and (S) indicates Shire.]

to just under 650 000 persons. This is equivalent to adding a new Newcastle (New South Wales) urban region to SE Qld in the next ten years and implies the need for provision of all the infrastructure that such a new "city" requires – just to maintain present standards.

This Gold Coast dominance in APRU's scenario also produces lower growth in Brisbane City and the west and western rural regions (Ipswich City, and the rural shires from Boonah, through Gatton, and north to Kilcoy) when compared with the scenarios of the DLGP (Tables 5A,B). Nevertheless, all scenarios indicate that the coastal regions will get by far the predominance of population growth in the Moreton Region. In fact under even the "best" scenario (Scenario 3), the west and western rural regions will only receive a little over 11% of the total growth of the region over the next ten years. This is despite the fact that more than two-thirds of the total land area of the Region is in these local government areas.

Conclusion

It was perhaps one of the objectives of the Moreton Bay and Catchment Conference to determine whether the future growth of the Moreton Region should best continue to be distributed along the coast or whether it is better to encourage settlement inland in the agricultural or upland areas. All would appear to have substantial environmental costs.

The acceptance of population growth as *good* rather than *bad* is characteristic of most of the Region's local authorities as well as other major decision makers in the Region such as the State Government and big business. So far the only initiative which could have questioned the *growth is good* syndrome is the SEQ2001 Project. But even that project accepted a pre-existing set of medium to high population forecasts as given, and has structured its policy recommendations around that acceptance.

There is some intention in SEQ2001 to direct growth into certain local authorities, but whether this will become reality is open to speculation – quite literally. Otherwise it appears that the foreseeable demographic future of the Moreton Region is merely a continuation of trends that have been characteristic for three decades or more – **maximum growth with minimum policy**.

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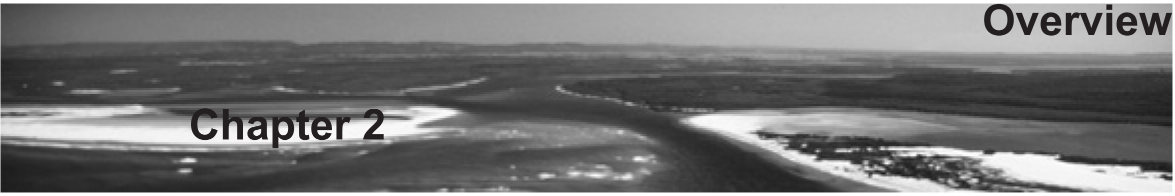

Chapter 2

Geology and Geomorphology

The theme unifying this chapter is our knowledge of that which lies beneath the surface of the land and seabed in the southeast Queensland region. The papers demonstrate that such knowledge can unlock clues to the ancient geomorphological history of the Bay, the position of valuable and as yet untapped mineral deposits, the processes that influence subterranean water resources, and hidden archaeological treasures of value to our cultural heritage.

Simon Lang and Duncan Lockhart and their coauthors extend our understanding of the geomorphological history of the Bay dealing with northern and southern Moreton Bay, respectively. They relate data that confirm David Neil's summary of the great geomorphological changes that have been experienced by the system. They use seismic profiles to track the path of ancient river beds and describe in detail the processes of sediment movement, deposition and infilling that have led to the present hydrography and sedimentology of Moreton Bay. Their confirmation that a mere 18 000 years ago the Brisbane River emptied into the ocean some 25 km east of the present location of Moreton Island provides a graphic illustration of the ephemeral nature of the system, and, as a consequence, the organisms that presently thrive within it.

The paper by Leonard Cranfield and Garry Pascoe highlights the importance of understanding the geology of the region, particularly in prescribing and prioritising land use and for mineral exploration. They describe the process and attendant tasks by which the Department of Mines and Energy are attempting to unlock the geological secrets of the region. Information of this kind can tell us how best to use available land for forestry, agriculture, soil conservation and urban and industrial development, and how to identify the locations of economically important mineral resources. The absence of such knowledge would lead to at best an inefficient use of our terrestrial resources, but at worst to a destructive and wasteful use of lands within the catchment, with dire consequences for its inhabitants, economic prosperity and environmental conservation.

John Harbison and Malcolm Cox relate information on the characteristics of ground water on Bribie Island. The intimate relationship they describe between the quality of the ground water and the surface water, and the influence of local geology on ground water quality marks this paper as a link between discussions of geology and subsequent chapters on water quality of catchment and Bay waters. They catalogue the natural processes that influence the nature of ground waters and describe the impact on this resource of various human activities, confirming the inter-relationship of air, land and sea and the need for consideration of groundwater impacts in our management of the system.

Maria Cotter uses her paper to point out an additional consideration for land use strategies, that of hidden archaeological material and the potential loss of such information through various forms of development. She argues for the development of land use planning strategies that incorporate assessments of the value of buried cultural material. Given the rapid growth of Brisbane's urban satellite areas, both the threat to cultural heritage and the need for assessment are very real.

Sedimentation and Coastal Evolution, Northern Moreton Bay

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A program of seismic profiling and surficial sediment mapping of the Brisbane River delta, Deception Bay and Pumicestone Passage has improved our understanding of sedimentation and coastal evolution in northern Moreton Bay.

Sedimentation in northern Moreton Bay involves three main types: fluvial, tidal and reefal.

(1) Confined to the western margin of the Bay and associated with sedimentation of the Holocene Brisbane River delta and to a lesser extent the Pine and Caboolture Rivers, are fluvial sediments, gravel, sand and especially mud.

(2) Mature quartzose sands dominate the surficial sediments of the eastern and northern tidal deltas. They are derived from either sands undergoing northward littoral transport being deposited in or transported from tidal deltas or sands reworked from older Pleistocene tidal delta deposits, dune island barriers or coastal strandplains.

(3) Carbonate reefal deposits are associated with the fringing reefs (e.g. around Mud and St Helena Islands). No recent sedimentation occurs in the central area of the Bay.

Over 400 km of high resolution continuous seismic reflection data acquired since 1995 from the Brisbane River delta, Pumicestone Passage and northern Deception Bay reveal incised valleys and transgressive surfaces indicating that Late Quaternary sedimentation has been profoundly influenced by sea level changes. The interpretation of the seismic profiles indicates that during the last glacial minimum (18 000 yBP), the Brisbane River drained to the northeast and reached the sea over 25 km east of Cape Moreton. The incised Brisbane River occupied levels between 20 and 40 m below the present sea level in the area now covered by Moreton Bay.

The incised Pine and Caboolture Rivers joined the Brisbane River near the middle of present day Moreton Bay. In Deception Bay a dendritic network of minor creeks drained into the Caboolture River system offshore from Godwin Beach. A southward draining river valley also existed along Pumicestone Passage (Elimbah Creek). Between 18 000 and 6 500 yBP sea level rose rapidly (over 1 m per century) drowning incised valleys and forming broad muddy estuaries. The Pleistocene Bribie strandplain and the Elimbah Creek valley were inundated, causing Deception Bay to extend 2 km further inland than today. At around 6 500 yBP, sea level reached its peak. This was maintained until at least 3 000 yBP and was followed by a slight fall in sea level of around 1-1.5 m. Progradation of a Bay-head delta at the mouth of the Brisbane River resulted. This was fed by the abundant sediment supply, filling the former funnel-shaped estuary between the present location of the suburb of Hamilton and the Brisbane Airport. The shoreline of Deception Bay also prograded rapidly during this time, with at least two distinct beach ridge-tidal flat systems developing. Pumicestone Passage began to fill with flood tidal delta sands derived from either end of the passage, with an estuarine basin developed in the middle. Fluvial sand accumulated in Elimbah and Coochin Creeks. Prodelta muds derived from the Brisbane, Pine and Caboolture Rivers settled out from suspension in restricted circulation areas in the Pumicestone Passage (e.g. Bullock Creek).

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 81-92.

Introduction

Over the past few years a series of geological studies have been undertaken in Moreton Bay by postgraduate students as part of a research program by the Applied Sedimentology and Sequence Stratigraphy Group, School of Natural Resource Sciences, Queensland University of Technology (QUT). The aim of the program is to use Moreton Bay as a field laboratory to develop a sequence stratigraphic framework for late Quaternary coastal deposits that can primarily be used as analogs for developing petroleum exploration models for mixed wave-tide-fluvial settings. Major contributions to the study of sedimentation in northern Moreton Bay have been made by earlier workers (Jones *et al.*, 1978; Hekel *et al.*, 1979; Jones & Hekel, 1979; Jones & Stephens, 1981; Jones, 1992; Stephens, 1978; 1982; 1992; Flood, 1980). This paper aims to fill several small gaps in our knowledge of sedimentation patterns and coastal evolution in three study areas; the Brisbane River delta beneath Fisherman Islands, the Godwin Beach area of Deception Bay, and the Pumicestone Passage (Figure 1). Furthermore, in conjunction with a companion paper (Lockhart *et al*., this volume), it aims to synthesise the coastal evolution of Moreton Bay. It is based on a sequence stratigraphic approach (van Wagoner *et al*., 1988), and discusses the four phases of coastal evolution recognised by Lockhart *et al*. (1996).

Moreton Bay forms a wedge-shaped lagoonal embayment 80 km long, opening to the Coral Sea towards the north (Figure 1). The Bay has an area of 1 400 km^2 and ranges from 35 km wide in the north, to 5 km wide in the south. Maximum water depth varies from 40 m in the north to 6 m in the south.

The Bay experiences both microtidal to mesotidal conditions (up to 2.8 m maximum tidal range), and prevailing winds are from the southeast. Moreton Bay is protected in the east by two large sand islands, North Stradbroke Island and Moreton Island. These islands are predominantly of Pleistocene age and represent aeolian dune-island barriers that protect most of the Bay from ocean swells (Stephens, 1978). The seaward fringes of these islands have a Holocene beach ridge system. Several tidal entrances connect Moreton Bay to the Coral Sea and these contain extensive ebb and flood tidal deltas. The entrances restrict oceanic swells entering the Bay and are associated with extensive sand bar and tidal channel systems.

In both the northern and southern extremities of Moreton Bay, Pleistocene and Holocene strandplains comprising extensive beach ridge systems are evident. These lie adjacent to extensive estuarine complexes at the mouths of several rivers (e.g. Caboolture, Pine and Logan) that drain into the Bay.

The Brisbane River is the largest river that flows into the Bay, and forms a significant bay-head delta. Marine seismic profiling between the Brisbane River and the Logan River in the southern part of Moreton Bay shows a series of incised valleys. These valleys traversed a coastal plain towards palaeo-shorelines lying to the east of the present dune-island barriers during the glacio-eustatic lowstands of the late Quaternary (Figure 2) (Jones *et al.*, 1978; Stephens, 1982; Evans *et al.,* 1992; Lockhart *et al.*, 1996; this volume).

The sedimentary environments of Moreton Bay (Figure 1) have been moulded by a mix of fluvial, tide and wave influences since the onset of the post-glacial transgression that reached its peak 6 500 yBP. The Bay therefore represents an excellent laboratory to examine the differing sedimentary processes and sedimentary sequences resulting from the relative influences of fluvial, tide and wave processes in a transgressive setting. Analysis of these depositional systems will shed light on the environmental response to sea level change in the Late Quaternary, and will assist with refining stratigraphic models for use by petroleum explorers of ancient successions deposited in comparable settings elsewhere.

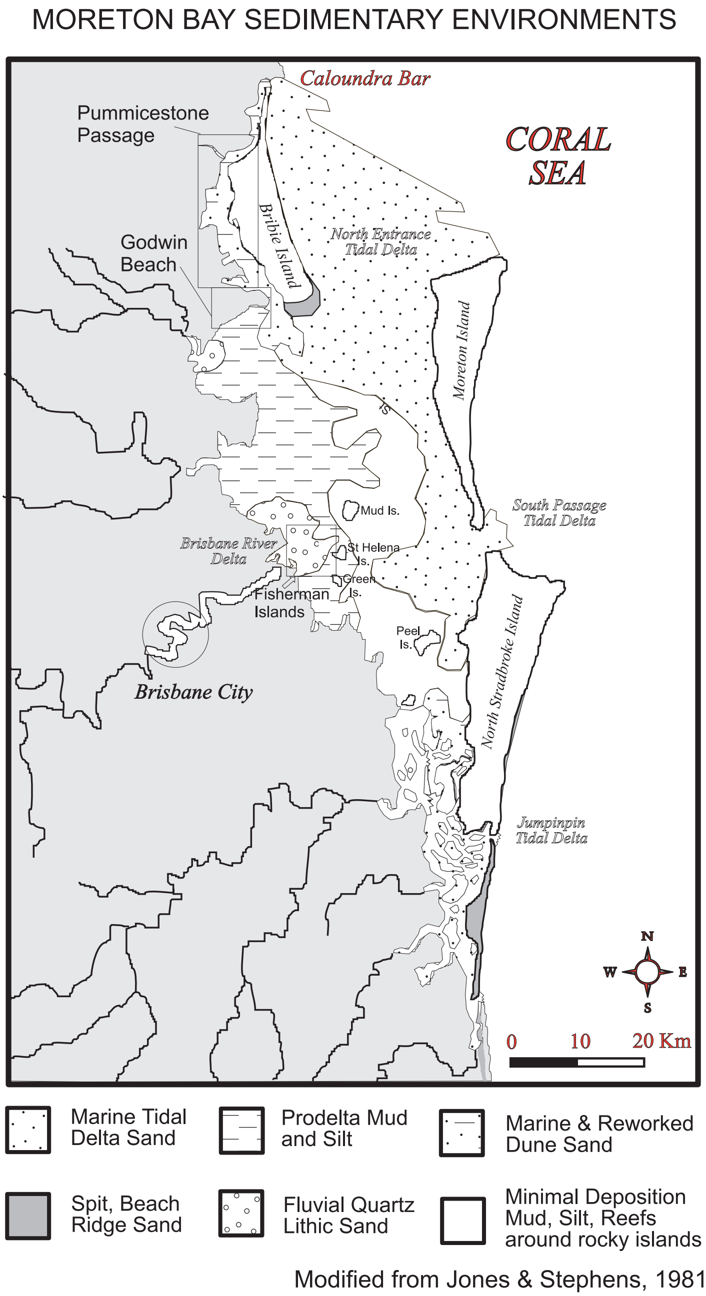

Figure 1. Moreton Bay Sedimentary Environments showing five major types of Holocene sedimentation: (i) Marine tidal delta sands in the eastern Bay; (ii) Marine & reworked dune sand in Pumicestone Passage and the southern part of the Bay; (iii) Spit and beach ridge sands mainly on southern Bribie Island; (iv) Fluvial quartzose and lithic sand from the major rivers, especially the Brisbane River delta; (v) Prodelta mud and silt mainly in the western Bay. Note that the area of minimal deposition is located in the central Bay area, and locally there are coraline reefs developed around and between Mud Island and Peel Island (area too small to show).

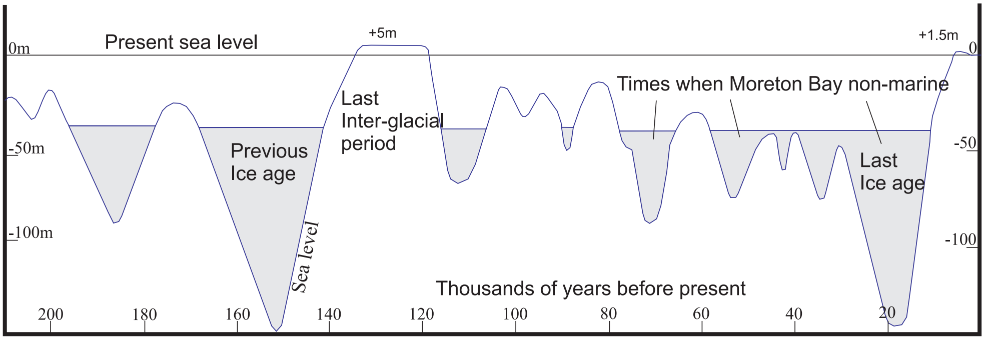

Figure 2. Sea level changes in the Late Quarternary showing periods when Moreton Bay was dry.

Depositional Environments in Northern Moreton Bay

Northern Moreton Bay has several distinct sedimentary environments (Figure 1) that can be divided into three types of sedimentation.

(1) Fluvial gravel, sand and especially mud confined to the western margin of the Bay and associated with sedimentation of the Holocene Brisbane River delta. To a lesser extent the same occurs in the Pine River and Caboolture River estuaries in Bramble Bay and Deception Bay respectively.

(2) Mature quartzose sands ultimately derived from littoral transport along the northern NSW coast, and deposited in or transported in the North Entrance tidal delta and along Pumicestone Passage. This material includes quartzose sands reworked by erosion of older Pleistocene tidal delta sand deposits, dune island barriers (e.g. North Stradbroke Island or Moreton Island), or coastal strandplains comprising extensive beach ridge systems (e.g. Bribie Island).

(3) Carbonate reefal deposits associated with the fringing reefs (e.g. around Mud and St Helena Islands).

The central area of Moreton Bay is a distinctive area of minimal sedimentation, where Pleistocene stiff clays are exposed at the sea bed beyond the reach of the downlapping prodelta of the Brisbane River (Jones & Stephens, 1981).

In the following pages, a summary of the results of sedimentation and coastal evolution studies in three study areas (Figure 1) is presented: (i) the Brisbane River delta; (ii) Godwin Beach in northern Deception Bay; and (iii) the Pumicestone Passage.

Brisbane River Delta

The mouth of the Brisbane River delta lies in southwest Moreton Bay, and is a good example of an incised river valley that has filled a funnel-shaped estuary and built a 300 km^2, fluvially-dominated delta during the Late Quaternary. An extensive offshore delta front and prodelta have developed with a slight elongate surficial distribution reflecting tidal and, to a lesser degree, wave influence. Recent studies have focused on the nature of the incised valley fill and coastal evolution in a mixed fluvial and tidal setting. Evans (1990) and Evans *et al.* (1992) presented a detailed account of the Pleistocene and Holocene incised valley fill beneath the Brisbane Airport and offshore from Nudgee Beach. The Fisherman Islands area has received less attention, however, its stratigraphy may be of key importance in the geotechnical assessment of the area for future Port development. Herdy (1997) undertook a detailed sediment mapping, seismic profiling and drilling program in the area, and identified a complex topographic surface developed on the Pleistocene substrate (stiff clays and sandy clays) beneath the area immediately offshore from the current Port development.

Sediments

An extensive surficial sampling and shallow drilling program over the intertidal and subtidal sediments of the delta in the Fisherman Islands area resulted in recognition of eight different lithofacies (Table 1). These lithofacies represent several sedimentary environments, including the delta front, distributary channels and prodelta, and pass laterally and seaward into fringing reefs, and a zone of minimal deposition. The Brisbane River forms a significant apron of prodelta mud that extends out into the Bay. Wave and tidal influences have caused the prodelta sediments to preferentially accumulate towards the northwest, creating the muddy coastline around Bramble Bay, and supplying fine sediment as far as Deception Bay and possibly to Pumicestone Passage.

Table 1. Lithofacies on and in the vicinity of the Brisbane River delta as determined by mud, sand, and carbonate percentages in each sample.

Lithofacies	Percentage Mud	Depositional environment
Slightly Muddy Sand	0.2 – 10%	Delta Front Sand
Muddy Sand	10 – 50%	Delta Front Sand
Sandy Mud	50 – 90%	Prodelta Muds
Mud	> 90%	Prodelta Mud, Mangrove environments
Shelly Slightly Muddy Sand	0.2% Mud 10-50% $CaCO_3$	Delta Front Sand
Shelly Muddy Sand	10-50% Mud 10-90% $CaCO_3$	Prodelta Mud, Channel Sediment
Shelly Sandy Mud	50-90% Mud 10-90% $CaCO_3$	Prodelta Mud, Channel Sediment
Reefal Carbonate	0-30% Mud 70-100% $Ca\ CO_3$	Confined to Fringing Reefs around Mud and St Helena Islands

Coastal evolution

Based on seismic interpretation, Evans *et al*. (1992) and Herdy (1997) recognised several distinct packages of sediment separated by key surfaces during the Late Pleistocene – Holocene. As age dating of sediments in this area is sparse, the type of sediment and its position relative to the key surfaces was used in conjunction with the sea level curve (Chappel, 1983) to interpret possible age ranges for the sediment packages. Following the approach by Lockhart *et al*. (1996; this volume), each package can be linked to four phases of incised valley fill beneath the Brisbane River delta, representing three depositional systems tracts (van Wagoner *et al*., 1988). Depositional systems developed during discreet phases of the sea level curve in which a sequence is deposited are referred to as "systems tracts" (van Wagoner *et al*., 1988). These include the late lowstand (Phase 1), early and late transgressive (Phase 2 & 3), and highstand/stillstand (Phase 4) systems tracts.

An incised surface marking the base of Phase 1 lies between –20 and –40 m developed on older Pleistocene alluvial deposits during the falling stage in sea level between 30 000 and 18 000 yBP (Evans *et al*., 1992). This incised surface (sequence boundary) is clearly evident on the seismic profiles (Figure 3, SB3) and has been intersected by drilling for the foundations of the Port of Brisbane developments on Fisherman Islands (Herdy, 1997). In this area, the incision has formed a dendritic drainage network on top of consolidated clays and sands of older Pleistocene deposits, producing an irregular, channelled surface between –10 and –20 m AHD. This surface dips seaward joining the main Brisbane River palaeochannel system to the north. A thin package of alluvial sediments (sands and gravels) filled these channels in places during the late lowstand but were transgressed (Phase 2) by estuarine muds during the rapid transgression that reached its peak at 6 500 yBP. A maximum flooding surface traceable throughout the area between –8 and –15 m AHD is visible on seismic profiles (Figure 3, MFS2), separating laminated muddy estuarine deposits from the muddy and sandy delta front, and laminated prodelta deposits of the early highstand. These features dip gently seaward. Rapid delta progradation during the highstand/stillstand (Phase 4) has produced a series of sediment lobes that clearly downlap onto the maximum flooding surface, comprising mouth bar and distributary channel deposits (Figure 3).

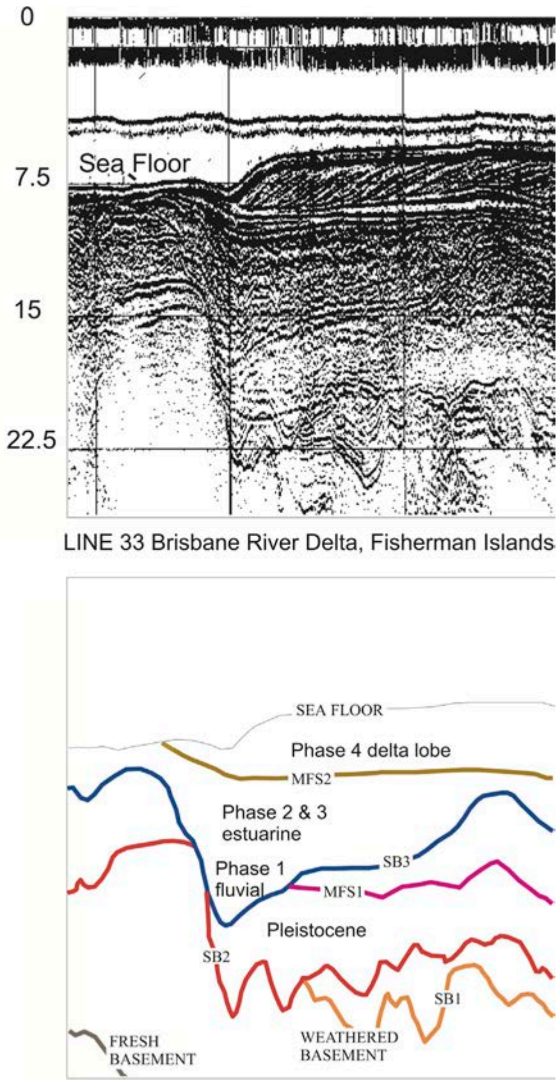

Figure 3.
Seismic reflection profile across a delta lobe of the Brisbane River delta between 1-1.5 km east of Fisherman Islands showing clearly developed progradational delta front. Note the downlapping refectors in the delta lobe that terminate onto the sea floor, which is coincident with the maximum flooding surface (MFS2) that separates highstand/stillstand deltaic sediments (Phase 4) from transgressive estuarine incised valley fill sediments below (Phase 2 & 3). The incised surface (SB3) represents the sequence boundary developed during the last glacial minimum, approximately 18 000 yBP. Immediately above this surface is a thin development of sands and gravel (know from borings around the Port development) representing fluvial sediments deposited in the late lowstand (Phase 1).

Deception Bay

Previous studies in Deception Bay (Flood, 1980) concentrated on the beach ridges in the Beachmere area immediately north of the Caboolture River. McClure (1995) extended this work north to the Godwin Beach-Sandstone Point area (Figures 1, 4) where extensive drilling data has become available as a result of sand resource extraction. The area consists of a strip of unconsolidated Pleistocene and Holocene coastal plain deposits that overlie the early Jurassic

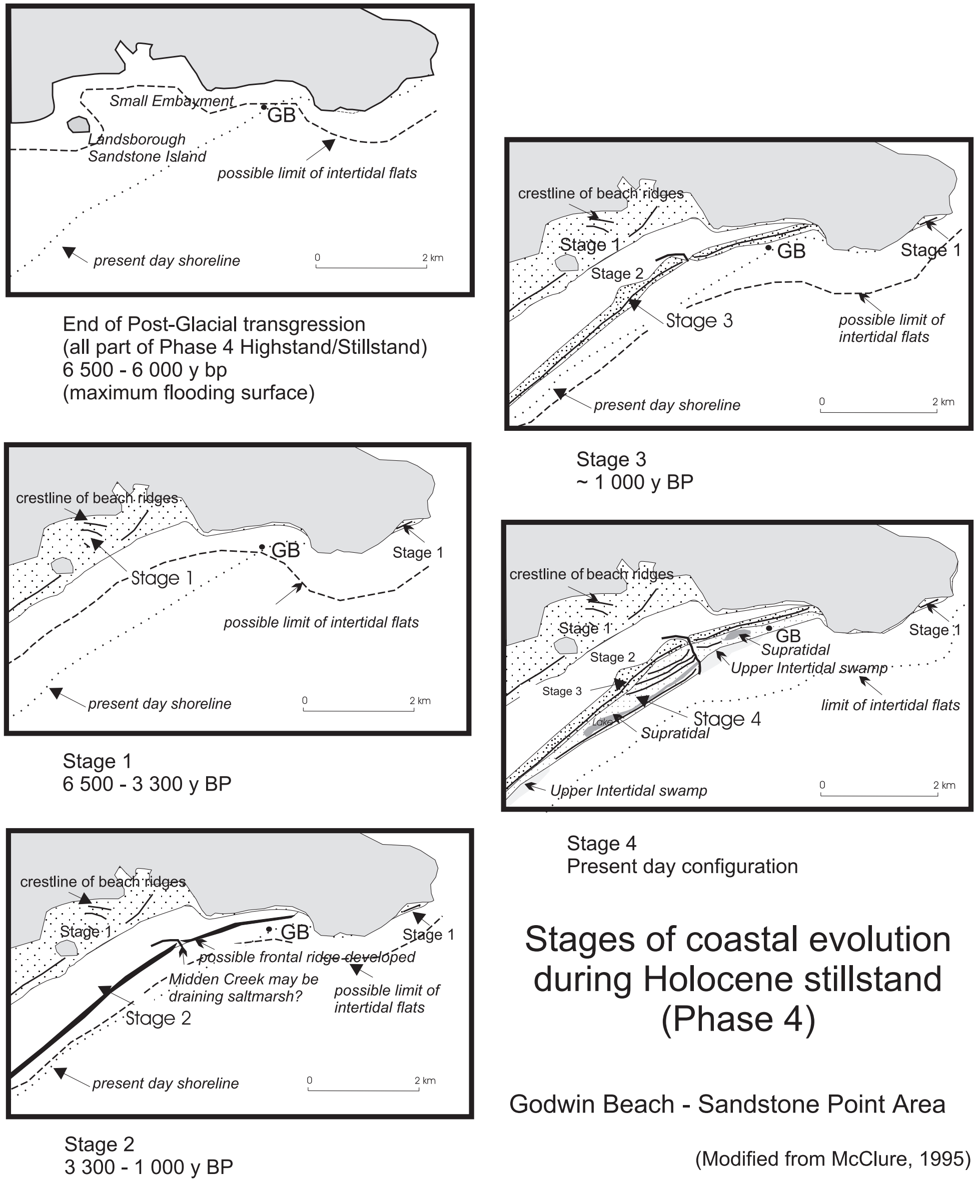

Figure 4. Stages of coastal evolution during the Holocene stillstand/highstand (Phase 4) at Godwin Beach (GB) in northern Deception Bay. Note the two distinct beach ridge systems separated by Stage 2 which mark a slight sea level fall between 3300 and 1000 yBP.

Landsborough Sandstone that crops out at Sandstone Point. The area forms extensive intertidal flats at low tide. The prevailing southeasterly winds play a significant role in redistributing sediment around Deception Bay.

Sediments

Extensive sampling of surficial sediments in the intertidal and subtidal zone has delineated several lithofacies that consist of sand, mud and biogenic carbonate. Mud originates from the Caboolture River, Burpengary Creek, and possibly the Brisbane River, whereas mature quartz sand originates from the North Entrance tidal delta. Two lithofacies dominate the area; a sand lithofacies that is located offshore from Sandstone Point, and further to the southwest a mud lithofacies that locally has a large biogenic carbonate component (shells).

Coastal evolution

A distinctive beach ridge-tidal flat system has developed in four stages in the Godwin Beach-Sandstone Point area (Figure 4) since the end of the post-glacial marine transgression (6 000-6 500 yBP; Flood, 1980). Drilling data indicate the Holocene coastal plain is a 'stillstand' progradational wedge (all Phase 4) that downlaps onto transgressive sand and mud and displays lithofacies similar to those in the nearshore zone (McClure, 1995).

The Holocene coastal plain is over 2 km wide and is dominated by a strandplain comprised of two beach ridge systems separated by 750 m of coastal lowlands and swamps. The most landward system, *Beach Ridge System 1,* developed not long after the end of the post glacial marine transgression (Flood, 1980). The lateral discontinuity of beach ridges in this system indicates that a small outcrop of Landsborough Sandstone located in the middle of the system has influenced their deposition. *Beach Ridge System 2* developed in the late Holocene (>1 500 yBP based on C^{14} dating by McClure, 1995) and consists of a large landward ridge and several smaller seaward ridges. This system is laterally continuous from Godwin Beach through to the Caboolture River (8 km). Between the two systems are coastal lowlands and swamps consisting of supratidal and upper intertidal deposits.

Three different processes may have caused the separation of the beach ridge systems.

(1) A slight sea level fall during the mid to late Holocene (up to 0.7 m based on raised upper intertidal sediments; Chappel, 1983).

(2) The formation of a barrier built up from the southwest inhibiting the development of beach deposits behind it.

(3) A decrease in sand supply between the development of the two systems.

A modern analogue for the development of a barrier or spit is now observed near the mouth of an unnamed tidal creek, where a beach spit has cut off sand supply and supratidal and upper intertidal conditions now prevail behind it. There is clear evidence that a fall in sea level has occurred in the area, however, at this stage it is difficult to assess the influence of the other two processes.

Comparisons of sequential aerial photographs over the last 53 years show the area has experienced short term coastal changes that include encroachment of supratidal environments by intertidal mangroves and beach erosion. These changes may indicate a slight rise in sea level. Urban development along the northern Deception Bay coastline could be threatened if these coastal changes continue.

Pumicestone Passage

Pumicestone Passage in the northern part of Moreton Bay separates Bribie Island from the mainland, and represents a mesotidal, elongate back-barrier lagoon estuary with a tidal inlet at either end. Bribie Island is an extensive strandplain of prograded Holocene and Pleistocene beachridges protecting Pumicestone Passage from the ocean. Within the southern reaches of the Passage lies the Elimbah Creek Bay-head delta, flanked by the shallow estuaries of Bullock and Ningi Creeks (Grosser, 1995). Central Pumicestone Passage is microtidal and contains three small tidally dominated creeks (Glass Mountain, Hussey and Coochin) that enter the lagoon-estuary from the west (Lawless, 1996). The morphology of Pumicestone Passage is inherited from Early Pleistocene and Late Tertiary coastal plain, beach-ridge, tidal delta and fluvial channels. The Pumicestone Passage estuary consists of a tripartite subdivision including a tide delta complex at both ends, a muddy central estuarine basin and a Bay-head delta at the mouths of Elimbah and Coochin Creeks lying along the western margin.

Sediments

Surficial sediment mapping has identified four Holocene sediment populations. Quartzose *Marine Sand* is reworked from Bribie Island and from the tidal entrances at both ends of the Pumicestone Passage. *Terrigenous Sand* is texturally and compositionally immature and is sourced from the fluvial reworking of local basement. *Terrigenous Mud,* settled from the suspended load of local tidal creeks, and from coastal rivers in Moreton Bay. *Biogenic Carbonate* comprises whole mollusc shells, disarticulated valves and fragments, and is either tidally reworked or *in situ*. Sediment transport pathways are linked to the interference of tidal currents entering the lagoon-estuary from both inlets. A narrow migratory zone of convergence and divergence exists in the vicinity of Hussey Creek forming a tidal null point. North of the null point, net bedload sediment transport is northwards, whereas south of the null point, bedload sediment is variable depending on tide time. Transportation of suspended sediment mimics water circulation and has an overall net northward direction (Grosser, 1995; Lawless, 1996).

Coastal evolution

Pumicestone Passage is a shore parallel, tide dominated lagoon-estuary linked to the evolution of the Pleistocene Bribie Island strandplain. Four phases of coastal evolution can be deduced from seismic profiling along the estuary and drilling (Grosser, 1995; Lawless, 1996), combined with interpretation based on the Late Quaternary sea level curve (Figure 2).

The Bribie Island strandplain was largely built between 140 000 and 120 000 yBP, during the last Pleistocene interglacial highstand (Armstrong, 1990). From 28 000 to 18 000 yBP, sea level fell to a glacio-eustatic minimum (Figure 2), during which time small incised valleys developed in the ancestral Bullock, Elimbah and Ningi Creeks, and the exposed surface became oxidised and overconsolidated. These creeks flowed south before heading east to join the palaeo-Caboolture River system. By 18 000 yBP sea level had fallen between 130 and 150 m below its present level and the shoreline had migrated east of Moreton Island (Stephens, 1992). The coastal plain was exposed to erosion and weathering while rivers and creeks incised the bedrock and the exposed coastal plain (Phase 1, lowstand systems tract).

From 18 000 to 6 500 yBP (Phase 2 & 3, transgressive systems tract) the post-glacial transgression, during which sea level rose rapidly to 1-1.5 m above its present position, dominated sedimentation patterns. Fluvial sands and estuarine muds backfilled the deepest part of the incised paleo-channels, overlain by tidal delta and tidal channel sands representing late lowstand/transgressive deposits. By the end of the transgression, the southern margin of Bribie Island was inundated, with the shoreline lying approximately along the present position of the Bongaree-Woorim Road (Armstrong, 1990), and Pumicestone Passage was broader and more extensive than today.

Since 6 500 yBP (Phase 4, stillstand or highstand) sea level has remained relatively stable or fallen slightly by approximately 1.5 m. During this time progradation of fluvial sediments into the passage, and transport of tidal delta deposits, have partially filled Pumicestone Passage in the southern and northern extremities with approximately 3 m of sand containing more mud than the underlying section (Grosser, 1995; Lawless 1996). These have been interpreted as shallow tidal channel and tidal flat sediments of the late transgressive/early highstand systems tract. During this time, numerous beach ridges accreted along the southern end of Bribie Island, the sand sourced from littoral erosion of the ocean beach and from the North Entrance tidal delta in northern Moreton Bay (Armstrong, 1990). In the Passage, large scale bedforms in the deeper parts of the channel show internal geometries and asymmetry indicative of a long term net southward transport of bedload sand (part of an ebb tidal delta system). This contrasts with the net northerly transport of suspended load sediments.

Consistent with observations from Deception Bay (Flood, 1980; McClure, 1995), aerial photography over the last forty years shows a trend of increasing encroachment of supratidal flats by intertidal, mangrove colonised areas. This is possibly indicative of a slight rise in sea level that could result in inundation of low lying areas around the passage.

Towards a Sequence Stratigraphic Framework

Based on seismic stratigraphic interpretation around the Bay (Jones *et al*., 1978; Evans *et al*., 1992; Lockhart *et al*., 1996; this volume), sedimentation is clearly controlled by sea level changes in the Late Quaternary. These represent lowstand, transgressive and highstand systems tracts, each separated by key surfaces (a sequence boundary at the base of the lowstand, a transgressive surface at the base of the transgressive systems tract, and a maximum flooding surface at the base of the highstand systems tract).

In the youngest sequence developed through the last post-glacial transgression in Moreton Bay, four phases of fill can be recognised. The sequence boundary developed around 18 000 yBP, when sea level had fallen to its lowest level and Moreton Bay was completely dry. The Bay area was a broad alluvial plain traversed by several incised valleys centred on the Brisbane River, which flowed out to over 20 km offshore from Cape Moreton (Figure 5A).

The late lowstand systems tract (Phase 1) represents fluvial sand and gravel confined to the thalwegs of the incised valleys (Brisbane, Pine, Caboolture Rivers, and possibly Elimbah Creek) deposited when sea level first started to rise.

The early transgressive systems tract (Phase 2) represents estuarine muds and minor sands deposited in estuaries (drowned river mouths) during rapidly rising sea level.

The late transgressive to early highstand systems tract (Phase 3), is represented by marine dominated quartzose sands entering the Bay via the tidal deltas, and continued estuarine sedimentation. Beach ridges were formed along the early highstand shoreline in northern Deception Bay (Figure 5B). A maximum flooding surface can be clearly recognised in the Brisbane River delta, marking the top of Phase 3. Elsewhere it is difficult to pick with certainty

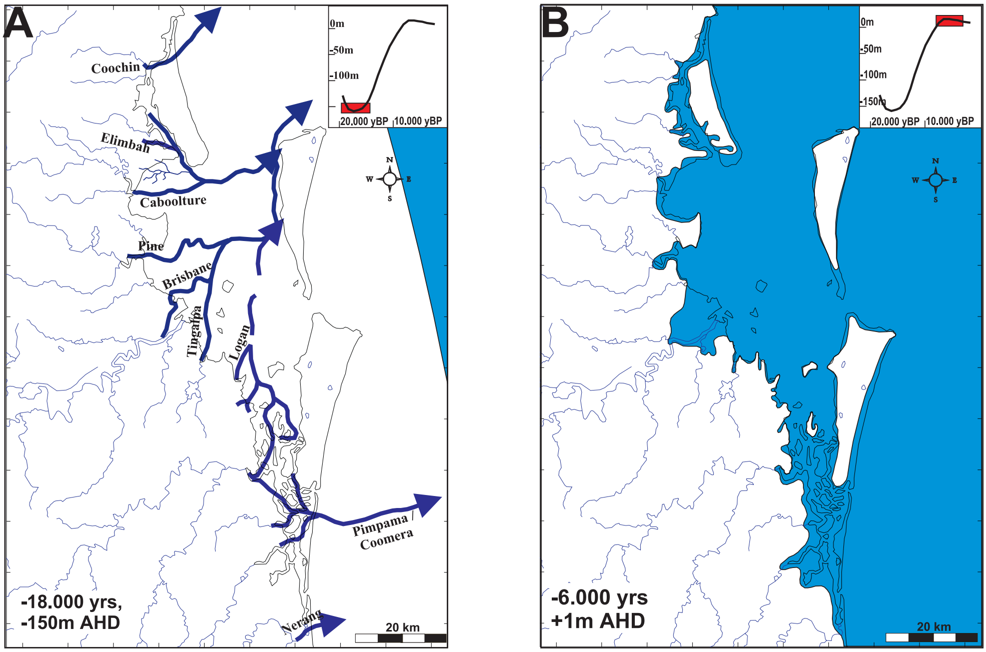

Figure 5. Palaeogeography of Moreton Bay. **A**: Phase 1, late lowstand, 18 000 yBP, showing incised valleys traversing Moreton Bay towards the palaeoshoreline well to the east of North Stradbroke and Moreton Islands. **B**: Beginning of Phase 4, highstand/stillstand, 6 000 yBP. Note the drowning of the river valleys forming extensive estuaries, and a generally broader

(especially where sandy sediments occur) but it is essentially coincident with the seafloor in the central part of the Bay.

The highstand/stillstand systems tract (Phase 4) is predominantly confined to the western part of Moreton Bay, and represents fluvially-dominated deltaic sediments from the Brisbane River, and to a lesser degree the Pine and Caboolture Rivers, and Elimbah Creek. Sea level may have fallen slightly (1-1.5 m), which has promoted the growth of the Brisbane River delta forming well developed progradational delta lobes. It also resulted in a seaward jump of the beach ridge system in Deception Bay between 3 000 and 1 000 yBP. The current retreat of mangrove species in northern Moreton Bay suggests that sea level may be rising slowly, and this may result in coastal erosion in the next few centuries.

Acknowledgements

The authors would like to thank the reviewers for their constructive criticisms of the text, and editors for their patience and assistance. Carolyn Lang is also thanked for her help typing and editing the text. The Port of Brisbane Corporation, Department of Environment, and Department of Mines and Energy are thanked for their logistic support with boats and seismic profiling equipment. Financial support from the AAPG Grants-in-Aid Program, Petroz N.L., Santos Ltd and Queensland University of Technology are gratefully acknowledged.

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Sedimentation and Coastal Evolution of Southern Moreton Bay

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Abstract

This study focuses on the narrow southern extremity of Moreton Bay. The area is bounded on the east by a Pleistocene dune-island barrier, North Stradbroke Island and a Holocene barrier island, South Stradbroke Island. The western part of the area comprises the low lying fluvially-dominated mangrove islands of the Logan River's bay-head delta. A low lying coastal plain south of the Logan River extends up to 10 km inland from the present western Bay shoreline. The central and eastern part of the area is a complex of low lying mangrove-colonised sand shoals and tidal channels, which represents the relict Jumpinpin flood-tide delta.

A combination of continuous seismic profiling, one drillhole and mapping of the palaeo-fluvial systems has enabled the development of a coastal evolution model. Results indicate that during periods of glacial lowstand the Logan River incised the exposed coastal plain of Moreton Bay. This incision created a complex fluvial system that originally flowed east-southeast of the present river mouth (the area now covered by South Stradbroke Island). The Pimpama River flowed into the Logan as a tributary, the confluence being near Jacobs Well. A fluvial system that flowed north through Redland Bay and west of Peel Island to eventually join the Brisbane river near the western shore of Moreton Island also existed at this time.

There are four phases of sedimentation related to rising sea level that filled the valleys and eventually created the modern form of the Bay. Phase 1 is related to the late lowstand when sea level first started to rise, and is comprised of fluvial gravel and sand. These sediments are thin and confined to the valley thalwegs. Phase 2 sediments were deposited due to a rapidly rising sea level and are predominantly estuarine mud and silt with minor estuarine sands. These sediments were deposited in a restricted estuarine environment. Phase 3 sediments were deposited during a late transgressive to early highstand phase. The sediments consist of clean quartzose sands sourced predominantly by marine sedimentary processes through an ancestral Jumpinpin tidal inlet. Conditions at this stage in the southern Bay were less constricted than they are now, with most of the present coastal plain inundated by the sea. The final phase of sedimentation predominates in the western part of the Bay. This phase represents infilling by fluvially-derived sediment from the Logan, Pimpama and Coomera Rivers, and marine sand entering through the flood tide dominated tidal inlet at Jumpinpin.

Introduction

One of the more significant concepts developed by sequence stratigraphy is that of *lowstand* deposits that form during periods of falling or lowered sea level. On the shelf, these comprise two distinct types of accumulations: *incised valley-fills* and *lowstand shorelines*. Incised valleys form when falling base level causes the rivers to extend seaward across the shelf and incise their substratum (Posamentier *et al.*, 1992). Transgression of these incised valleys leads to the formation of estuaries. Sediments become finer grained and depositional environments are controlled primarily by tidal influences. The shoreline retreats landwards because the rate of the increase in accommodation space is much greater than the rate of sediment supply to the basin. During periods of high and stable sea level sediment supply is able to catch up and surpass the amount of accommodation space that has been created by the previous transgression. Consequently, estuaries will begin to infill with a combination of fluvially and marine-derived sediment.

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Modern coastal estuaries and lagoons represent excellent analogues for ancient incised valley-fills. Detailed studies of sedimentary process and evolution of these environments aid in the understanding of natural controls on sediment distribution. These studies are valuable for geotechnical studies in a number of ways. They can delineate the distribution of acid clays and provide the location of palaeo-valleys filled with sediment likely to compact at faster rates than the valley interfluves. Understanding of mud distribution in the estuarine environment also highlights areas in which heavy metals will concentrate in sediments. To date, however, relatively few detailed facies and stratigraphic studies have been carried out in mixed wave and tidal dominated estuarine valley fills (Dalrymple *et al.*, 1994).

Geological setting

The coastal zone of southeast Queensland is a wave dominated coastline with a micro to mesotidal system. This predominance of wave power over tidal power has created the coastal morphology of the region. The southeast Queensland coast is characterised by the presence of barrier islands, coastal spits and the occurrence of tidal inlets (e.g. Jumpinpin, North Entrance and South Passage). The dominance of large floodtide dominated tidal deltas, which act as sediment sinks for massive quantities of sediment transported along the coast by littoral drift, is a feature common to wave dominated coasts. In southeast Queensland this littoral drift direction is to the north and driven by the predominance of southeasterly trade winds and swell waves. In comparison, relatively little sediment is permanently trapped in the subordinant ebb-tidal deltas, which are formed in the seaward position of these tidal inlets.

Moreton Bay itself forms an 80 km long back barrier lagoon system up to 35 km wide and opens to the Pacific Ocean towards the north. The landward side of the Bay is dominated by a series of incised river valleys that grade into estuarine coastal plains accumulated between bedrock highs. The ocean side of the Bay is bordered by a sand dune dominated barrier island system cut by a series of tidal inlets. The lagoon is tide dominated with tides ranging up to 2.4 m. Fluvial outflow is dominant in the summer wet season, especially following periodic cyclonic rainfall.

This study focuses on the southern extremity of Moreton Bay where the lagoon narrows to less than 10 km. The area is bounded to the east by the dune island barrier, North Stradbroke Island, and the barrier island, South Stradbroke Island (Figure 1).

The islands of southern Moreton Bay

North and South Stradbroke Island

Although these two islands were joined in the past, they were both formed by distinctly different geological processes. North Stradbroke Island was formed by aeolian processes during periods of low and rising sea level in the Pleistocene. Strong, dominantly south-southeasterly trade winds (dominant when the climate was more arid, windy and cooler during periods of low sea level (Ward, 1977)), combined with a westerly transgressing shoreline reworking marine sand on the exposed shelf into large migrating parabolic dunes (Stephens, 1982). These dunes eventually anchored close to their western limit against a series of roughly north-south trending bedrock ridges. These ridges, buried beneath the high dunes, have been imaged by a combination of refraction seismic performed by the Bureau of Mineral Resources during the 1960s (Polack & Kevi, 1965; Kevi & Milson, 1966) and subsequent drilling by the Geological Survey of Queensland (Laycock, 1976).

The formation of Eighteen Mile Swamp on North Stradbroke Island is most likely related to a long period of tidal inlet closure at Jumpinpin. Flood *et al.* (1986) determined with C^{14} dating of peat that the swamp was formed approximately 600 yBP. The origin of the barrier beach/

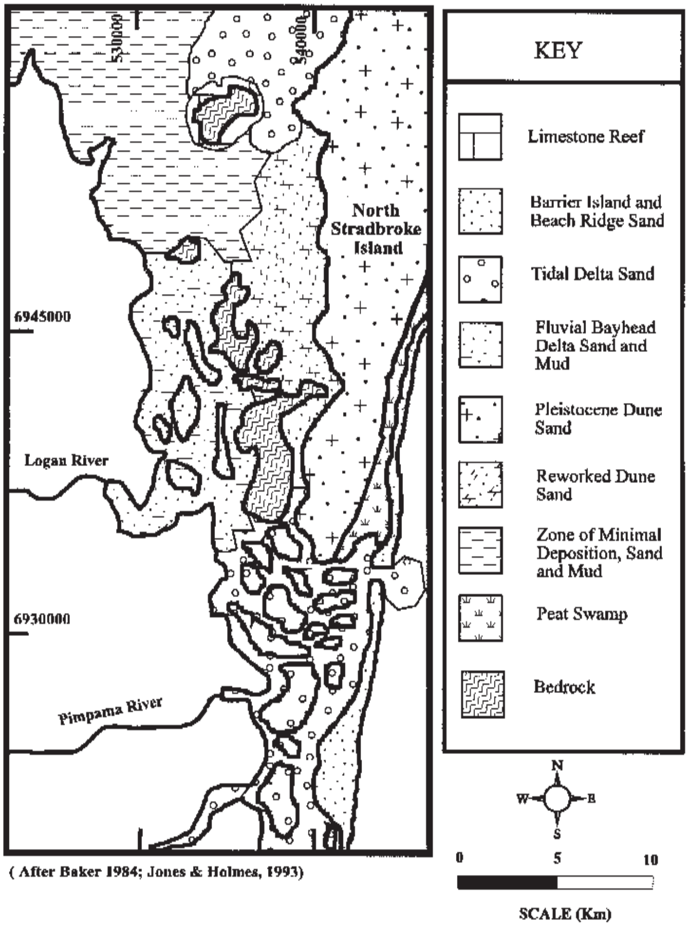

Figure 1. Location of study area and distribution of sediments.

dune system on the easternmost shore of North Stradbroke Island is related to a northward spit progradation evolving from longshore drift-spit accretion (Flood *et al.*, 1986). A modern example of this rapid longshore accretion, progradation and finally coastal attachment of an offshore bar, has been documented at Alva Beach, southeast of Townsville in 1974 (Belperio & Southgate, 1978).

With the spit becoming land-attached a small scale back barrier lagoon formed. With the outflow of freshwater from North Stradbroke Island the water salinities eventually changed from saline to brackish to eventually fresh. This then allowed the development of the extensive peat swamp represented by Eighteen Mile Swamp (Flood *et al.*, 1986).

In contrast, South Stradbroke Island was formed by shoreline processes associated with the Holocene post-glacial transgression. Seismic imaging performed on the continental shelf to the east of South Stradbroke Island shows the development of earlier linear barrier islands formed during periods of sea level stillstand during periods of the Pleistocene when sea level was between 60 to 80 m below present sea level (Searle, 1982). During the last post-glacial transgression (from around 18 000 yBP to 6 000 yBP) sea level rise was rapid, approximately 2.5 m per 100 y, the sand of the beach barrier system was moved landward by storm washover processes in a tank tread fashion (a process described by Swift (1975)). Lagoonal and estuarine sediments buried by the washover processes are assumed to thin because of the rapid progress of the transgression which averaged 1 km landwards per 100 y (Kudrass, 1982).

South Stradbroke Island probably arrived at its present position around 6 500-6 000 yBP (Kudrass, 1982; Baker, 1984) at the time of the maximum Holocene sea level. Since this time the Island has experienced active beach-ridge accretion associated with the slight fall in sea level (Flood, 1984).

Mangrove-colonised islands

The eastern side of southern Moreton Bay is affected by fluvial influx from the adjacent Logan River. This fluvial sediment is presently building a regressive bayhead delta within the estuary and lagoon. A series of mangrove colonised silt and sand islands of fluvial, and mixed fluvial and marine sand origin has developed. Comparisons of aerial photographs taken over a period of decades in the area indicate that mangrove colonisation of shoals at the mouth of the river is taking place.

Low lying mangrove-colonised islands of the Logan River bay-head delta are associated with a network of likewise mangrove-colonised mud and sand islands near the Jumpinpin tidal inlet. Shallow vibrocoring confirms that the islands near the Jumpinpin tidal inlet have a marine sediment source and represent abandoned tidal delta shoals. These tidal delta shoals became abandoned during an undetermined period of tidal inlet closure during the late Holocene during which time mangrove colonisation was able to take place. During the period of tidal inlet closure the only sources of sediment in the lagoon were the Logan, Pimpama and Coomera Rivers. The sediment supplied was predominantly mud and silt, with fluvial sand being deposited only near the mouth of the rivers. Jumpinpin inlet subsequently re-opened in May 1898 due to catastrophic southeast gales (Welsby, 1913) and has remained opened since. As a consequence, the tidal conditions in southern Moreton Bay have changed and the relict tidal delta is experiencing active erosion. Prior to the opening in 1898, the tidal channels were so shallow that even rowing a small boat around the area was a difficult task (Gordon Maas, pers. comm.) The re-opening of Jumpinpin and the associated change in tidal conditions caused active erosion and scouring of tidal channels, along with daily inundation of some areas on the coastal plain during the high tide. To remediate the tidal inundation problem a number of canals were dug on the coastal plain (Gordon Maas, pers. comm.).

Bedrock islands

A number of bedrock islands are located within southern Moreton Bay. These islands are predominantly comprised of Palaeozoic metamorphics (Neranleigh-Fernvale Beds, Rocksberg Greenstone), Jurassic sandstones (Woogaroo Sub-Group) and Tertiary basalt. Remote sensing data show a number of strong lineaments throughout the area. These lineaments have a strong northwest to southeast grain and are likely to be related to thrust faulting which was initiated during the Mesozoic. Fault reactivation occurred during the Tertiary with the onset of volcanism in the southeast Queensland region and created a small graben in the northwestern portion of the study area. Basaltic eruption was associated with the faulting and subsequently the graben was filled with up to 300 m of predominantly basaltic rock (Freiderich, 1978).

The coastal plain

The coastal plain adjacent to the Logan and Pimpama Rivers was formed during periods of higher sea level in both the Pleistocene and the Holocene. Around 120 000 yBP sea level along the east coast of Australia was approximately 6 m higher than present sea level (Chappel, 1983).

At the peak of the last post-glacial transgression sea level peaked at a level approximately 1-1.5 m higher than present around 6 500 yBP (Chappel, 1983) (Figure 2). Numerous deep auger holes have been drilled on the coastal plain by the Geological Survey of Queensland during the mid-1970s as part of an evaluation of sand extraction potential of the area. Sediments intersected by these auger holes reveal that the entire area of the coastal plain was part of a large shallow, predominantly sandy bay at the end of the Pleistocene transgression and at least a large part of the area was also inundated at the end of the Holocene transgression (Figure 3).

Numerous molluscs and gastropods are contained within the sediments of the auger holes. Numerous shells were collected from a 3 m deep pit dug at the Palm Farm on Hotham Creek, near the Pacific Highway at Pimpama. Species collected included *Anadara trapezia* (blood cockle); *Pyrazus ebeninus* (mud whelk); *Pecten fumatus* (scallop); *Antigona chemnitzii* (venus clam); *Saccostrea commercialis* (oyster); *Corbula* spp. (basket shell); *Bedeva paivie* (oyster drill); *Nassarius burchardi* (dog whelk) and *Velacumantus australis* (mud whelk). Several well preserved specimens of the crustacean *Scylla serrata* were also collected. Many of the bivalve specimens were articulated, indicating rapid death *in situ* and therefore are representative of the sedimentary environment in which they were recovered. This faunal assemblage supports the interpretation of a predominantly sandy mud to clean sand estuarine tidal flat environment in the vicinity of sea grass and rocky headlands (Coleman, 1988). Indicated water depths vary from subtidal to intertidal.

An articulated sample of *Anadara trapezia* was recently sent for dating by Amino Acid

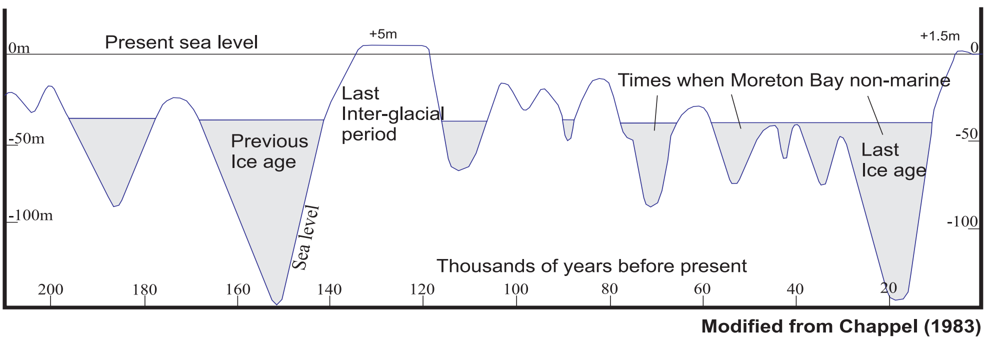

Figure 2. Quaternary sea level curve with reference to Moreton Bay.

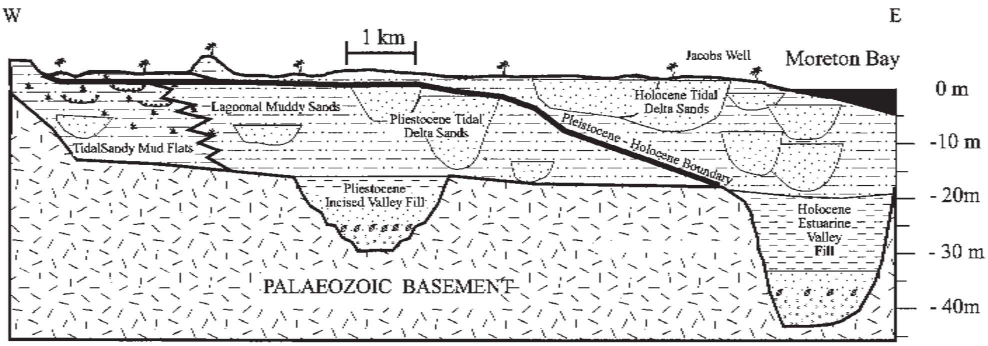

Figure 3. Interpreted cross section based on extensive deep auger drilling of the southern Moreton Bay coastal plain.

Racemitisation (AAR) at the University of Wollongong. A map of the coastal plain produced by Stephens (1982) indicates that the sample locality is Pleistocene in age and related to a higher period of sea level during the last interglacial (approx 120 000 yBP). Given the potential of an age around 100 000 yBP it was decided to use AAR dating in preference to C^{14} . The sample returned an aspartic acid ratio of 0.415 ± 0.021, a value for this species that indicates an age between 6 000-7 000 yBP (Colin Murray-Wallace pers.comm.). The result correlates closely with aspartic acid ratios for a sample of the same species taken from Hervey Bay (ratio of 0.44 ± 0.005). This sample was dated using both AAR and C^{14} techniques. The C^{14} technique returned an age of 6 400 ± 140 yBP for the sample (Murray-Wallace, 1979).

The age of the sample is coincident with the peak of the postglacial marine transgression (Chappel, 1983), the topographic location of the sample being adjacent to the 2.5 m contour of the coastal plain. From the depth of the sample and the dominantly intertidal to subtidal faunal assemblage of the sample locality, it may be inferred that at least a thin veneer of Holocene aged sediment covers the coastal plain in areas below the 2.5 m topographic contour (i.e. the majority of the coastal plain).

The majority of the clean sands intersected on the coastal plain are of a fine to medium grain size, well sorted, subrounded to rounded, with minor feldspar and lithic content. The composition of these sands and their distribution are consistent with deposition in a large flood tidal delta system active during both the last interglacial highstand and at the peak of the Holocene transgression. A complex combination of sandy shoals and tidal channels were deposited during these periods when the tidal inlet was much wider than present and similar in form to that seen around South Passage today.

The fluvial influence in the coastal plain sediments is restricted to the mud and silt component of the sediment. The present day fluvial system follows this pattern, with the majority of river transported sand being deposited in the immediate vicinity of the mouth of the Logan River. It is therefore expected that the predominance of fluvial derived sands on the coastal plain will be confined to an area around the ancestral mouth of the Logan and, possibly, Pimpama Rivers.

The majority of the sand which fills the present lagoon is marine in origin, being sourced through Jumpinpin by a combination of northward longshore drift and the predominance of wave power over tidal power along the coast. This dominance of wave power over tidal power creates a wave dominated coastline. This part of the coast has been supplied by vast quantities of marine sand throughout the Quaternary on a scale that renders the fluvial sediment contribution of the Logan and Pimpama Rivers less significant with respect to infilling the Bay during periods of high and stable sea level.

Seismic profiling

A total of 250 km of continuous high resolution seismic profiling has been acquired to date in southern Moreton Bay by the Queensland University of Technology (QUT) School of Natural Resource Sciences. Shallow water depths encountered in the area (between 0.5 m-8 m) confined the survey to the network of narrow tidal channels, and subsequently an optimal grid pattern was unable to be obtained (Figure 4).

Data were acquired using a E.G. & G. Uniboom High Resolution System borrowed from the Queensland Department of Mines and Energy. Position fixing was by means of a Trimbole

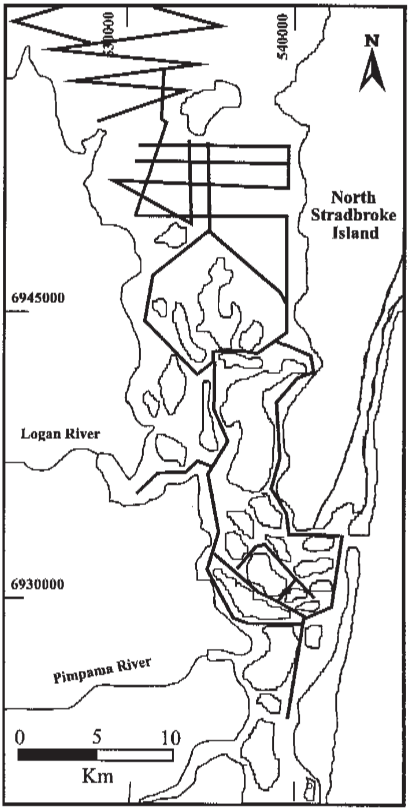

Figure 4. Location of seismic lines acquired by the Queensland University of Technology from 1994-1996 in southern Moreton Bay.

Pathfinder Basic GPS unit in non-differential mode using AMG '84 as the datum. Locations were checked with known positions and accuracy was determined to be within ± 50 m.

Seismic stratigraphy and interpreted coastal evolution

Seismic stratigraphy involves dividing seismic reflection profiles into distinct packages. These are separated by key seismic reflection surfaces, and characterised by distinct reflection signatures or facies based on amplitude contrast and orientation. These packages can then be used to interpret geological processes that were unique to that area at the time of sediment deposition.

Four distinct seismic facies constitute the valley fill seen in southern Moreton Bay. These seismic facies represent four distinct phases of sedimentary fill deposited within the system in response to sea level changes during the late Quaternary (Figures 5 & 6).

The lower-most reflector represents the base of late Quaternary sedimentation in the area and is characterised by a dramatic erosional surface that may be correlated regionally throughout Moreton Bay. This reflector corresponds to a major Type 1 sequence boundary (Posamantier *et al.*, 1992) related to glacial lowstand conditions when sea level was at its lowest leading to incision of the exposed Bay floor by rivers such as the Logan, Brisbane, Pine and Caboolture.

To test the seismic stratigraphic interpretation, a deep fully-cored drill hole was drilled in 1996 by QUT at Rudy Maas Marina near Rocky Point. The hole known as Logan River #1 was located based on the seismic interpretation of a large incised valley in close vicinity (approximately 100 m from the drill hole location). The hole intersected three of four interpreted seismic facies (Figure 7).

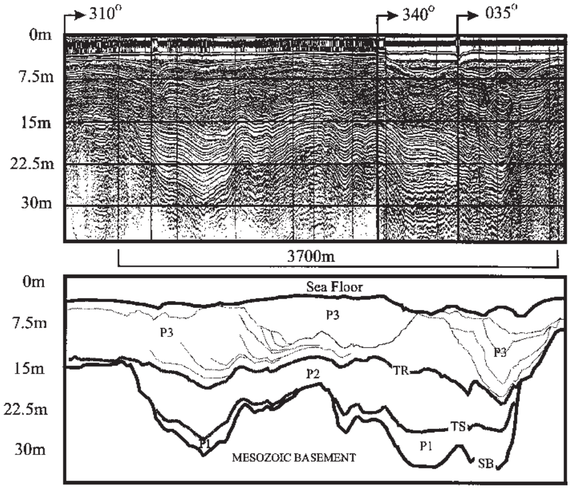

Figure 5.
Interpreted seismic profile located near Behm Creek.
SB = sequence boundary;
TS = transgressive surface;
TR = tidal ravinement surface;
P1 = Phase 1;
P2 = Phase 2;
P3 = Phase 3.

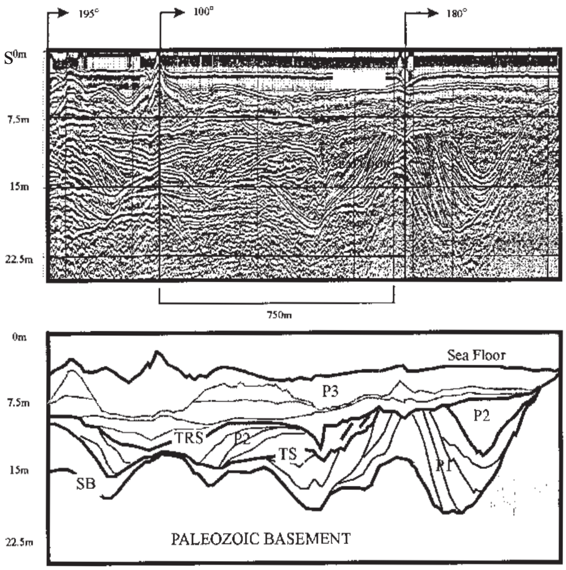

Figure 6.
Interpreted seismic profile located near Canaipa Point.
SB = sequence boundary;
TS = transgressive surface;
TR = tidal ravinement surface;
P1 = Phase 1;
P2 = Phase 2;
P3 = Phase 3.

Phase 1 (late lowstand fluvial channel)

The Phase 1 seismic facies characteristically displays variable amplitude contrasts and varies from chaotic to clinoform where lateral accretion appears obvious (Figure 6). The reflectors are confined to and around the thalweg of the valley and onlap the valley walls. Deposits are generally in the order of 4-5 m thick. Results of deep auger drilling on the coastal plain near Behm Creek, confirm the presence of fluvial quartzite gravel fining upwards into peat at depths of between 29-36 m. These reflectors are interpreted as being representative of fluvial point bars, deposited after sea level had reached its lowest stage. The lack of significant vertical aggradation in this phase suggests that sea level was constant at this stage.

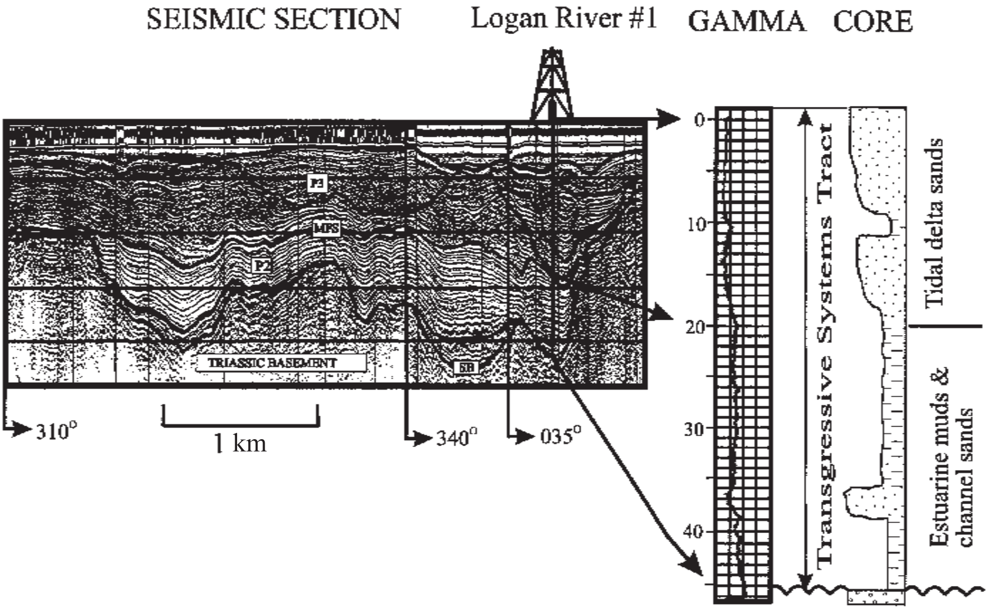

Figure 7. Location of Logan River #1 with respect to seismic line. Also displayed are the gamma ray log and lithological log for the hole.

Phase 2 (early transgressive restricted estuarine)

Phase 2 deposits are characterised seismically by strong amplitude reflectors which onlap the broader valley walls and can be seen most clearly in Figure 5. There are significant vertical aggradation and weakly to strongly developed clinoformal point bar geometries evident in some areas (Figure 6). These deposits continue on from the previous phase without any significant erosional surface being evident. The reflector separating the two phases is interpreted as representing the initial transgressive surface and, although not correlatable in a lateral sense (it is confined to within the valley), it is seen on all sections where incision is evident. The reflector is usually characterised by a significant amplitude contrast to the underlying sequence. These deposits are interpreted as representing an early transgressive succession of heterolithic estuarine point bars. Sediments intersected by deep auger drilling on the coastal plain considered to represent this facies are characterised predominantly by finely laminated mud and minor sandy mud.

These sediments are characteristically olive green to black in colour and are commonly associated with the presence of shells and pebble lags representing tidal influenced channel bases. Towards the top of the Phase 2 sediment unit the muds become distinctly laminated with alternating black and buff-coloured layers. Biologically, one species of bivalve predominates and there appears to be low species diversity indicating stressful conditions in a reducing environment. This is consistent with restricted estuarine circulation. Sponge spicules and foraminiferans are common in the sandy mud facies.

Minor occurrences of mud crab (*Scylla serrata*) claws and body fragments are found in both the mud and sandy mud facies. Shells recovered from phase 2 sediments intersected by Logan River #1 were dated using C^{14} – AMS. Ages ranged from 13 650 ±50 yBP at the base (-42 m) to 7430 ±50 yBP at the top of the phase (-20 m). The age dates from the shells also reveal the presence of two depositional series within this phase that have distinctly different sedimentation rates.

The interval between -20 m and -39 m was deposited between 9130 ± 50 yBP and 7430 ±50 yBP. The lower 3 m of the phase (-39 m to -42 m) was deposited at a much slower depositional rate between 13 650 ±50 yBP and 9130 ±50 yBP.

Phase 3 (late transgressive to early highstand-open estuarine/marine)

Phase 3 sediments are represented by weakly defined noisy seismic reflectors with only isolated well developed strong amplitude reflectors present. The nature of these reflectors indicates the predominance of sand in the sedimentary sequence with the higher amplitude reflectors representing the presence of mud-filled abandoned tidal channels. The reflector which separates the underlying Phase 2 sediments from this more marine influenced phase is significantly erosional and is representative of a tidal ravinement surface created when the tidally dominated lagoon has transgressed the estuaries (Figures 5 & 6).

Within Phase 3 there are several erosional surfaces that may be recognised. These surfaces most commonly mimic present day tidal channels in form, dimension and location. In many examples the location of many present day tidal channels appears to be determined by the pre-existing location of major bedrock incised valleys.

Phase 3 sediments are not confined to the incised valley walls and represent a change from restricted estuarine conditions into open shallow lagoonal conditions similar to those experienced in the present day lagoon. The majority of the sediment for this phase was supplied by the existence of a large flood and tide dominated tidal delta situated between an evolving barrier island (South Stradbroke Island) and the Pleistocene dune island barrier of North Stradbroke.

Sediments are characterised by well sorted, clean to minor muddy quartzose marine sand. The sands are uniformly fine to medium grained, the only grain size variation being provided by relative quantities of mud. The sand is compositionally mature with approximately 90-95% quartz, ± 5% lithic fragments (metasiltstone, chert), ± 5% heavy minerals (ilmenite, rutile, zircon, hornblende), > 1-2% feldspar (orthoclase and plagioclase). Reworked shell fragments of fine to medium grain size vary in abundance between 0-5% and are associated with the muddier facies.

The tidal ravinement surface separating Phase 2 and Phase 3 was intersected at -20 m in Logan River #1. The surface is represented by a sharp basal contact between clean marine sand and black, laminated mud with abundant rip-up clasts immediately above the contact. Bases of tidal channels in this phase are marked by the presence of pebble and shell lags associated with clay balls and rip-up clasts of mud. Shells recovered from the base of Phase 2 at -20 m were dated at 7430 ± 50 yBP providing an age for the tidal ravinement surface.

Phase 4 (highstand-mixed fluvial/marine)

The sediments representative of Phase 4 are confined to the landward edge of the lagoon, the bayhead delta islands and the mangrove-colonised tidal flats surrounding the bedrock islands in the lagoon. Generally this phase is absent from the seismic data, a fact that may be exacerbated by water depths insufficient to allow boat access at any stage of the tide around the bayhead delta area. Shallow auger drilling on the bay-head delta islands in the mouth of the Logan River confirm the existence of at least 5 m of fluvially-dominated sand, silt and mud with a high organic content and peat. A minor component of marine sand is also present.

The maximum flooding surface for the Holocene transgression is a diachronous surface between Phases 3 and 4 which may be recognised on some seismic sections as a downlap surface. Phase 4 (and most likely the latter stages of Phase 3) represents the regressive infilling of the lagoon during the present highstand.

Incised valley dimensions

Incised valleys seen on the seismic records have significant dimension. Logan River #1 drilled at Rudy Maas Marina near the mouth of Behm Creek reached a depth of -46 m G.L. before reaching basement. The seismic line in the vicinity indicates that the valley is possibly 2.7 km wide. Even allowing for the strong possibility that this section is a traverse oblique to the direction of the valley and arbitrarily halving the width to 1.35 km indicates a large fluvial system comparable in size to the Brisbane/Pine River system (Evans *et. al*., 1992). Other examples have dimensions of smaller but not insignificant proportions varying from around 300-500 m wide and at least 22 m deep.

These depths are based on seismic depth conversion using a velocity of 1500 m/sec based on seismic velocity work performed around Moreton Bay by the Geological Survey of Queensland (Searle, 1980). Drilling results from Logan River #1 displayed that this depth conversion method is too simplistic in some areas, as the predicted depth of Logan River #1 was 35 m and the actual depth was 46 m. To account for this depth discrepancy the seismic velocity is believed to have increased to at least 2000 m/sec through the Phase 2 sediments as the contact between Phase 2 and the overlying Phase 3 sediments came in at the predicted depth. It is therefore possible that some of the incised valleys may be deeper than they are presently mapped.

Other surfaces representative of tidal ravinement surfaces with subsequent tidal channel style morphologies are comparable to present day tidal channels seen within the bay. Generally these tidal channels are in the vicinity of 2-8 m deep and vary from 100-400 m wide.

Basement structural control

Mapping of bedrock basement based on seismic and drill hole data show an incised valley trend for the palaeo-Logan system that is strongly controlled by the pre-existing structural grain of the region. Mapping of the palaeo-Brisbane River system in the northern sector of Moreton Bay indicates that this river has also been controlled by regional structural lineaments. The main grain in the area is a north-west to southeast trend, a structural grain determined by thrust faulting during the Mesozoic in conjunction with a conjugate set of lineaments that trend north-east to south-west.

There is a close relationship between the present day tidal channels and the location of incised valleys in the vicinity of Peel and Macleay Islands and Redland Bay. This is in an area away from the influence of the relict tidal delta near Jumpinpin. Sediment supply in the northern sector of southern Moreton Bay during the Holocene transgression has been low when compared to these tidal and bay-head delta areas. Consequently the majority of sedimentation in this northern sector has been confined to the incised valleys and only a thin veneer of sediment overlies the interfluves in the northern sector. As a result the presence of major tidal channels in the northern sector reflects the position of the underlying incised valleys and highlights the close relationship between the present bathymetry and the palaeo-drainage pattern.

Mapping the course of the palaeo-Logan River

A map of the depth to basement for the area has been made by mapping the recently acquired seismic and incorporating BMR refraction seismic data (Laycock, 1976) covering North Stradbroke Island. The map covers an area from the southern tip of Green Island in the north to just south of the mouth of the Coomera River in the Broadwater in the south. The top of the basement was mapped by hand and two-way times measured at each position fix. Extra two-way times were recorded in the vicinity of valley thalwegs if they did not coincide with position fixes. These two-way times were then converted to depth using a velocity of 1 500 m/s. The depths were then corrected for tidal variation and datumed to AHD.

Positions and corresponding depths were entered into computer contouring packages (Surfer for Windows and a program written by John Laycock, QUT). Extra points were included to indicate areas of bedrock outcrop and the mainland. The maps were then checked for correctness and adjusted accordingly.

The results indicate that a complex fluvial network existed in southern Moreton Bay during periods of lowstand (Figure 8). A major fluvial system was separated by a bedrock divide created by a topographic high located south of the present mouth of the Logan River and Russell Island. This directed a major fluvial system to the passage through the Behm Creek area and continuing east-southeast under the present South Stradbroke Island. The palaeo-Pimpama River joined the system as a tributary in the vicinity of Jacobs Well.

North of this divide a major fluvial channel existed to the west of Russell and Macleay Islands and flowed northward through what is now Redland Bay and into the area south of Peel Island. The system meandered towards the west around the very northern tip of Coochiemudlo Island and then continued north on the western side of Peel Island with tributaries joining the system from Eprapah and Hillards Creek. Green Island and Wellington Point were joined at low sea level and acted as a divide between Tingalpa Creek and the north-flowing system. Tingalpa Creek flowed northwards to join the palaeo-Brisbane River in the vicinity of St Helena Island.

The North Stradbroke Island data reveal the presence of a large valley running approximately east-west under the island. There is no apparent incision on the seismic lines running north-south on the western side of the island. It is possible that this represents an older Tertiary incision into the basement rocks that has not been imaged on the seismic due to its depths (up to -90 m MSL) and the inability of the uniboom seismic signal to penetrate these depths.

Conclusions

High resolution continuous seismic profiling in the southern Moreton Bay area has revealed a palaeo-fluvial system that was originated by incision during a period of glacial-induced lowstand during the Pleistocene. The results indicate that a complex fluvial network existed in southern Moreton Bay during periods of glacial lowstand. A major fluvial system was separated by a bedrock divide created by a topographic high located south of the present mouth of the Logan River and Russell Island. This directed a major fluvial system to the south through the Behm Creek area and east-southeast under the present South Stradbroke Island. The palaeo-Pimpama River joins with this system as a tributary in the vicinity of Jacobs Well. North of the divide a major fluvial channel existed to the west of Russell and Macleay Islands and flowed northward through what is now Redland Bay. The river then meandered to the west of Peel Island continuing northward to probably join the Brisbane River system.

The incised valleys that were created by lowstand incision are about 46 m deep and potentially up to 3 km wide. These valleys have been infilled in four stages each related to a specific period of sea level history. The first phase was deposited during the late lowstand and comprises fluvial quartzite gravel fining up into peat. These sediments were deposited as fluvial point bars under relatively stable sea level conditions. Phase 2 sediments are estuarine in origin and were deposited during early transgression in wide but restricted estuaries. There is significant vertical aggradation and the sediments were deposited as heterolithic estuarine point bar deposits. Phase 3 sediments are related to the late transgression and are more marine in nature than the previous deposits. They were deposited in a tidally-dominated back barrier lagoon with abundant marine

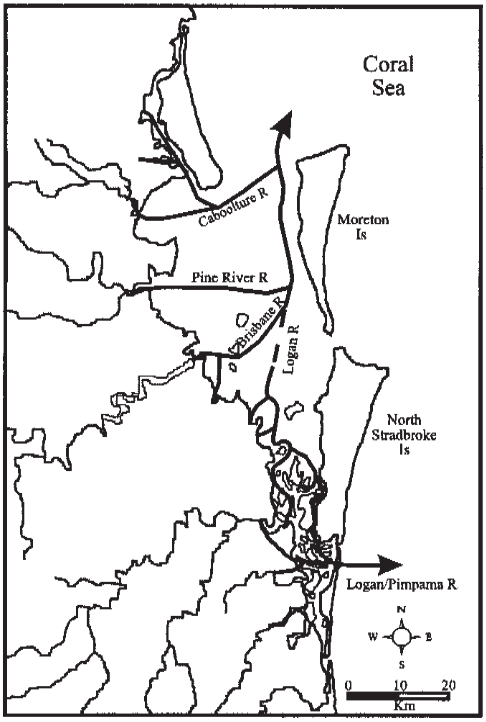

Figure 8. Interpreted fluvial drainage network for the Moreton Bay area during the last glacial period.

sand supply via a large active flood-dominant tidal delta. Phase 4 sediments are restricted to the landward edge of the present lagoon, and are predominantly fluvial in origin. The sediments are a mixture of fluvial sands and muds. They represent a prograding bay-head delta which downlaps onto the underlying Phase 3 sediments and is gradually infilling the present lagoon.

The coastline of southern Moreton Bay developed as a barrier coastline similar to that of today, with South Stradbroke Island migrating westward across the continental shelf by washover processes as sea level rose rapidly. South Stradbroke Island arrived at its present location about 6 500 yBP when sea level was close to its present height. The island has since expanded by lateral beach-ridge accretion.

Prior to the closure of Jumpinpin the low lying mangrove islands of southern Moreton Bay were the subtidal to intertidal sand shoals of a large flood-tide delta. At the peak of the transgression most of the present coastal plain was inundated by the sea and was part of the tidal delta complex. Closure of Jumpinpin for an unknown period of time prior to European settlement allowed for the mangrove colonisation of the sand shoals and the formation of the islands of southern Moreton Bay (Baker, 1984). Eighteen Mile Swamp probably formed at this time by the redistribution of sand contained in the ebb-tide delta complex seaward of the tidal inlet at Jumpinpin. The sand would have been redistributed as part of a mobile longitudinal sand spit which eventually accreted onto the shore of North Stradbroke Island.

The islands around the mouth of the Logan River are comprised predominantly of sand and mud of fluvial origin and form a bay-head delta complex. They have formed in response to a slightly falling or near stable sea level since the peak of the post-glacial transgression. The re-opening of Jumpinpin in 1898 has led to both the erosion of some of the islands in the relict tidal delta complex and further deposition of sand in southern Moreton Bay.

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Development of a Digital Geological GIS for the Moreton Region

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Abstract

Digital Geographical Information System (GIS) technology gives much greater scope to update geological map data over Queensland. Queensland Department of Mines and Energy's experience in this technology led to the proposal of a pilot project to generate a seamless digital geological GIS database over the SEQ 2001 planning region.

Currently this project has resolved differences and similarities between geological units in different areas. A unique map legend has been produced for the whole region. A polygon cover of geological units based mainly on 1:100 000 scale published mapping has been produced for the whole region. Geological map symbols have been generated by manual digitising of symbols from published maps and extracting digital data from the SURFACE GEOLOGY component of DME's MERLIN database. A layer of mineral occurrences and mineral deposits exists and is being updated.

Future goals include the separation of digital linework into individual components such as geological boundaries, faults, lineaments, etc. for spatial modelling and entering of digital geological point-attribute information from field notebooks. Spatial modelling to create new layers relating to mineral potential and quarry rock potential is also planned.

Introduction

Systematic geological mapping of the southeast Queensland region commenced as standard 1:250 000 Sheet areas using 1:85 000 photography as a basis for compilation from the early to mid 1970s in the production of the Ipswich, Brisbane, and Gympie 1:250 000 Sheet areas. Mapping of the Caboolture and Ipswich 1:100 000 Sheet areas using 1:25 000 aerial photographic bases dates from the mid to late 1970s and these areas were mostly joined across common boundaries. Caboolture (1978), Ipswich (1980), and Brisbane (1986) are the only full colour 1:100 000 Sheet areas completed in this region. Other 1:100 000 sheet areas were produced in the late 1970s and 1980s as preliminary maps. These maps were generally based on the 1:250 000 locally updated from information from auger drilling and point observations related to quarries. They mainly focused on land use planning and development; in particular the provision of basic locations of known sources of industrial minerals, quarry rock and including sand and gravel resources. To provide an overview of the regional geology of the region, geological boundaries were rationalised at 1:500 000 scale resulting in the production of the MORETON GEOLOGY (1980). This map is available in digital format and has been used for a range of land use issues including regional modelling of forest types by other government agencies. Little work was undertaken in this region in the mid-late 1980s due to commitment of most of the Geological Survey to north Queensland projects. A new mapping project was commenced in the Gympie and Laguna Bay 1:100 000 Sheet areas in 1990 and was completed in 1993 (Cranfield & Scott, 1993). This mapping was accompanied at the same time by mineral occurrence mapping of the same region (Barker *et al.,* 1993). In 1994 and 1995 geological mapping mineral occurrences that focused on Palaeozoic metamorphic rocks and plutonic rocks of the North D'Aguilar Block was undertaken by Donchak *et al.* (1995) and Crouch *et al.* (1995). Mineral occurrence mapping in this area was completed by Randall *et al.* (1996). These projects included large parts of the Goomeri, Nanango and Nambour 1:100 000 Sheet areas. New maps produced of these areas include the new mapping integrated with

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 107-110.

Cainozoic units. Geological boundaries were then joined in the ARC/INFO GIS across map sheets and a single ORACLE key was used for common units.

Checking map boundaries

To ensure geological units joined across map boundaries and that units had been correctly classified it was necessary to colour these units. Colouring of map units was accomplished by customising the ARC/INFO polygon shades routine. In the Department of Mines & Energy customising of this routine has been accomplished by adding standard (Pantone) colours for map publication with a range of screens to create an in-house shade set library. Individual shade sets were allocated to each geological unit to allow for checking of the geological boundaries and units. At present most of this area has been successfully joined to complete seamless digital linework for this region.

Future Work

To generate a digital geological GIS for the region the linework needs to be split into its components for a range of modelling (e.g., separating a boundaries layer from a layer containing faults and lineaments). For modelling the digital linework and map polygon information needs a layer of point-collected geological attributes to create the GIS.

A number of tasks are required to complete the digital geological GIS database for this region. These tasks are:

- seamless digital linework separated into unique components such as geological boundaries, faults, lineaments etc.;
- a polygon cover of geological units based on the best available mapping scale for the whole region (probably 1:100 000);
- a geological symbols layer linked to the SURFACE GEOLOGY component of the MERLIN system;

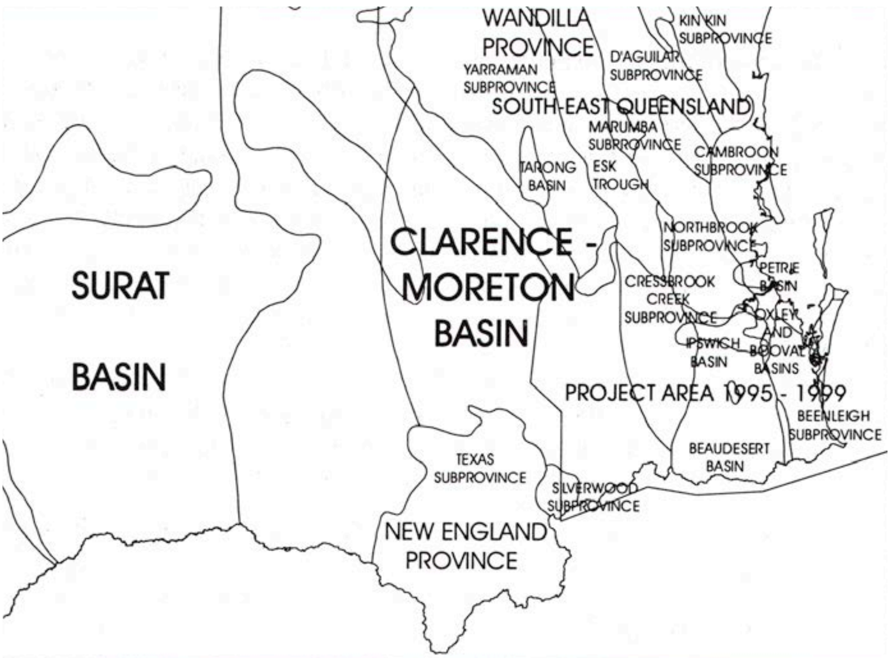

Figure 1. Structural units of southeast Queensland.

old 1:250 000 mapping over the Gympie 1:250 000 Sheet area.

The importance of SEQ 2001 Queensland Departmental planning area for land use studies led to the proposal of an initial pilot project to generate a seamless digital Queensland Geological GIS database. The SEQ 2001 area includes the Moreton Bay and catchment region extending from the Queensland border to Noosa in the north and Toowoomba to the west. The seamless GIS will cover the area of the Southeast Queensland Project area that contains several different structural units (Figure 1).

Geological provinces in the Moreton Bay and catchment region include the Palaeozoic Wandilla Province (comprising the North and South D'Aguilar and Beenleigh Subprovinces), the Mesozoic Esk Trough, Ipswich, Nambour, and Clarence-Moreton Basins, and the Tertiary Booval, Oxley, and Beaudesert basins. The main depocentre for post-Tertiary Cainozoic rocks is Moreton Bay. Here the sediments include fluvial, marine shelf, fringing coral reefs and marine tidal delta deposits that are of Pleistocene and Holocene age.

Issues

Because of the different publication dates from individual 1:100 000 geological maps in southeast Queensland and prevailing geological knowledge, inconsistencies of interpretation have occurred. Linework did not join up across most map boundaries and geological units were commonly different across map boundaries. To generate a seamless digital database it was necessary to resolve differences and recognise similarities in geological units in different areas, and to generate a unique map legend that would be suitable for the whole region. For the Palaeozoic and Mesozoic geology with named stratigraphic units this proved less difficult than for Cainozoic units that were morphostratigraphic in nature (i.e. related to landforms). For the first time in Queensland, the Brisbane 1:100 000 Sheet areas provided map information for unconsolidated Cainozoic offshore units in Moreton Bay. This map information was basically a summary of a range of offshore drilling and seismic data accumulated over more than 10 years of field investigation including sampling and drilling on the northern part of Moreton Bay by the Geological Survey of Queensland.

Methodology

Production of a seamless digital geological GIS for southeast Queensland had to be linked to the corporate MERLIN system because DME's corporate direction is to capture all map data in ARC/INFO graphical format using map production standards that link to the corporate ORACLE relational databases for public access. MERLIN is a developing digital system based on ARC/INFO for graphical data combined with an ORACLE relational database for geological attributes. Geological map production is undertaken in ARC/INFO and all geological linework (geological boundaries of various categories, lineaments, faults etc.) is put on a single layer for this purpose. Geological map units are given a unique key in the ORACLE database and geological unit polygons are tagged with their unique key. Cartographers have designed generated routines in ARC/INFO called AMLs (Arc Macro language) to plot geological map boundaries, legends and map symbols in the production of final maps.

Resolution of geological units into a unified scheme and joining of linework

The initial data source was published 1:100 000 and 1:250 000 map sheet areas of southeast Queensland. These units were captured as a database in EXCEL format. The spreadsheet initially captured existing geological units and was updated to include new mapping in the area. Equivalent units were given the same ORACLE key and shade-set colour. Cainozoic units from different map sheets were simplified into a unified classification. The Brisbane 1:100 000 Sheet area as the latest area mapped was generally chosen as a template for classification of

- a layer showing mineral occurrences and mineral deposits;
- geological attributes recorded during field work as a number of point-related observations on rock type, structure, boreholes, and geochemistry; and
- spatial modelling to create new layers relating to mineral potential and quarry rock potential.

Work has commenced on Tasks 1 and 2. To complete Task 3 two processes are necessary:

- digitising of existing map geological structural symbols; and
- creation of structural symbols through the creation of geological site information that includes structural symbols from previous mapping in MERLIN's SURFACE GEOLOGY system. DME has ORACLE data browser routines that can extract structural symbols as an ASCII file. The symbols are then created using a in-house customised ARC/INFO AML.

The mineral occurrence layer (Task 4) exists from recent investigations and digital capture of previous compilation of mineral occurrences. This layer is being updated and verified for the region in selected areas. Data for Task 5 will be compiled using data entry consultants and cooperative student research as applicable. Task 6 is in the initial planning phase and will be developed over time with a land use planning and exploration focus.

Data Sources

The geological digital GIS database for the SEQ 2001 region requires the collection and integration of a range of data from different sources. These include:

- published 1:100 000, 1:250 000 and other geological map data from various sources (University theses and student projects, mining company exploration, research and other government agencies);
- field notebook information from previous 1: 250 000 and 1: 100 000 mapping projects and resource studies;
- borehole information from a range of diamond drill and auger holes and offshore drilling in Moreton Bay; and
- comparison and updating of geological map databased on interpretation of signatures on geophysical (radiometrics, magnetics and gravity) images and contour maps and satellite imagery.

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General Features of the Occurrence of Groundwater on Bribie Island, Moreton Bay

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Abstract

Bribie Island is situated in the northwest part of Moreton Bay, 65 km north of Brisbane. It is a large coastal barrier island with a permanent population of 12 000 people which increases in summer. Groundwater has been used as a municipal water supply since 1962, and the rate of groundwater extraction has increased substantially from 65 million litres (ML) per annum to approximately 1 500 ML per annum. In addition, treated water is also provided from mainland surface reservoirs.

Recent changes in land use relevant to groundwater conditions are the clear-felling of exotic pine plantations in the northern part of the island in order to restore native vegetation, the expansion of residential areas, canal developments, and the relocation of sewage outlets in the southern part of the island.

In this paper, the shallow groundwater regime of Bribie Island is defined in terms of geology, water level fluctuations, response to rainfall, and geochemistry. Recharge to the aquifer is primarily via precipitation. Discharge is via evapotranspiration, evaporation, stream runoff and groundwater discharge to sea, with evapotranspiration likely to be the predominant mechanism. Critical factors in the behaviour of the groundwater system are the elevation of the seawater boundary above mean sea level, the degree of semi-confinement of the regional aquifer by horizons of indurated sand (“coffee rock”), and the extent of other low permeability layers.

Introduction

Bribie Island is situated at the northern end of Moreton Bay within Caboolture Shire and is separated from the mainland by the shallow Pumicestone Passage (Figure 1). The island is a popular retirement and recreation centre with a permanent population of 12 000, concentrated along the southern beaches; the population rises to around 40 000 during the summer holiday season. The island is approximately 143 km^2 in area and is additionally the location of a weather research station, a fisheries research station, a water reserve, national park areas and commercial pine plantations.

The sand deposits which make up Bribie Island produce supplies of potable water, principally for domestic use. The Bribie Island Water Treatment Plant is within a designated Water Reserve in the southeast of the island. As it is a shallow groundwater system in an area of episodic rainfall, standing water levels are important in maintaining areas of phreatophytic vegetation, including mangroves, woodlands, heathlands and swamps. The effect of the current removal of pine plantations on groundwater levels and the interaction of the pines with groundwater is not fully determined. In the very southern part of the island, there is a municipal sewerage treatment plant which disposes of the treated effluent water on land.

This paper provides an overview of the occurrence of groundwater and groundwater chemical types on Bribie Island, and is part of a more comprehensive study. Broadly, the island aquifer has rapid infiltration and response to rainfall, and so surface waters and groundwater are closely related. As Bribie has a different depositional history and geomorphology to the nearby

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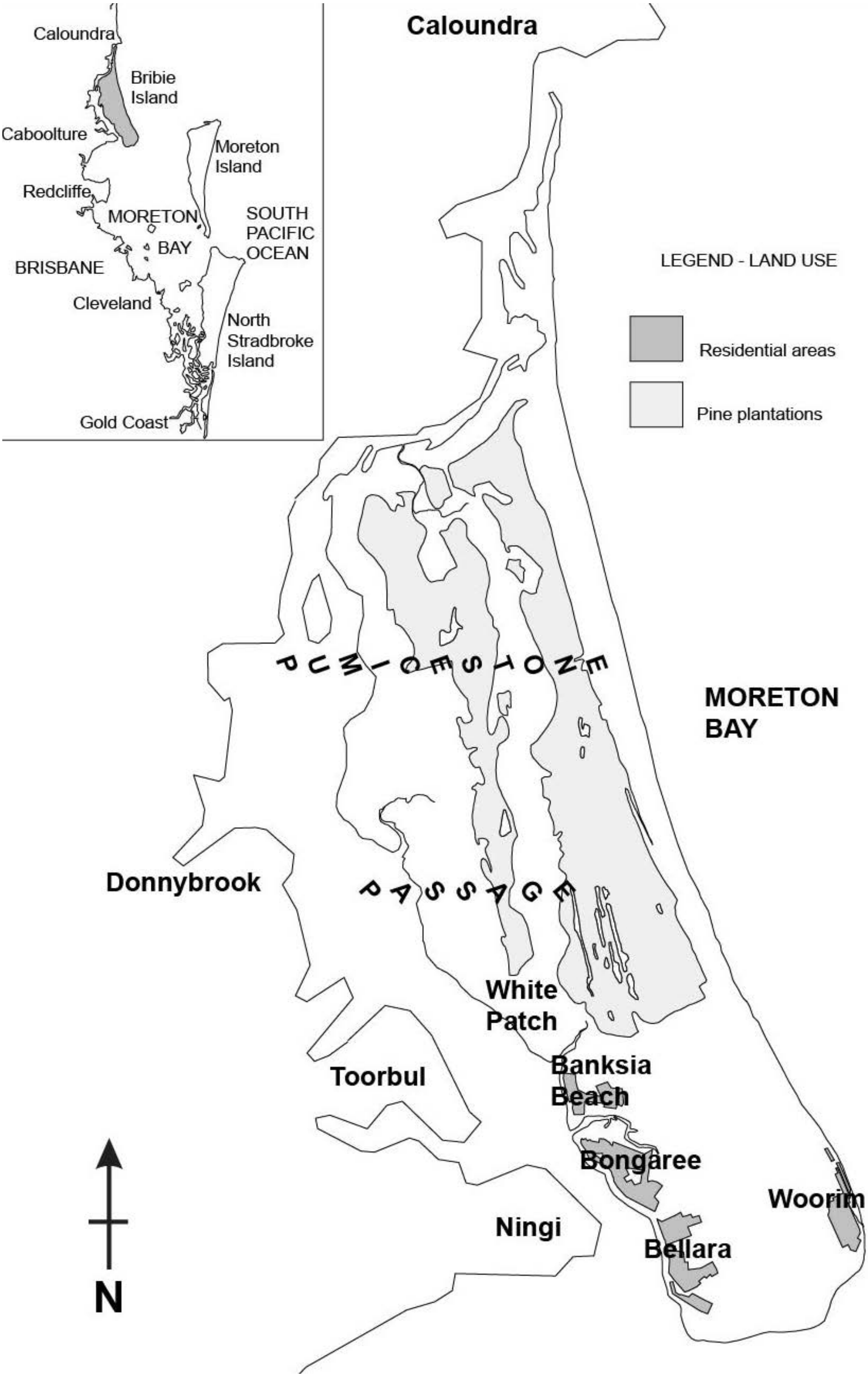

Figure 1. Locality plan showing place names referred to in the text (inset – Moreton Bay).

large sand islands, Moreton and North Stradbroke, there also are discernible differences in its hydrologic character. Suitable hydrogeological methods for measuring groundwater behaviour in this particular environment are also considered.

Previous Groundwater Investigations and Development

Six production bores and a water treatment plant were installed southwest of Woorim in 1962. The Geological Survey of Queensland carried out hydrogeological investigations in the southern part of Bribie Island in 1963-64 (Lumsden, 1964) and in 1980 (Ishaq, 1980). During the former investigation, 31 observation boreholes were drilled, typically to a depth of 14 m, in the southern part of the island. In 1971, due to iron-fouling of bore screens, groundwater extraction was converted to an excavation approximately 3 km long and 5 m deep extending west from the previous borefield.

In order to determine the response of the saline groundwater interface adjacent to the main water extraction trench, Isaacs (1983) developed a two-dimensional dynamic model of the interface using assumed hydraulic parameters and average aquifer dimensions. Isaacs & Walker (1983) developed a numerical model for the southern part of the island to examine the effect of relocating the Council's sewage outlets.

In a general study of tidal response in groundwater, Kang & Nielsen (1995) surveyed a conductivity-depth transect for the northern spit of Bribie Island. Superelevation of the water-table by 0.6-0.9 m Australian Height Datum (AHD), tidal phase lags and temporal skewing were observed in bores proximal to the beach.

As part of a study of the Pumicestone Passage and the peripheral area (DEH, 1993), a total of 23 observation boreholes were drilled along five east-west transects to provide more comprehensive monitoring. These holes were geologically and gamma-logged, which defined clay lenses within the sand profile. With inclusion of boreholes from previous studies, additional drilling and borehole attrition, 39 observation bores are now currently being monitored by the Department of Natural Resources (DNR). These boreholes are numbered 079 to 124 (Figure 2).

In late 1995, the Groundwater Assessment Group of DNR was requested by the Department of Environment (DoE) to develop a predictive computer groundwater model of Bribie Island (DNR, 1996). The main purpose of the model was to better manage the clear-felling of pine plantations and subsequent regeneration of native vegetation. As part of this assessment, six additional bores were drilled in the northern part of the island.

Bribie Island is not a proclaimed groundwater area under the provisions of the Water Act, and at present there is therefore no control over groundwater use or the siting or construction of new boreholes. Borehole extraction by private spears for garden irrigation and irrigation of recreation areas from excavations is common, but is not monitored. Overall, the effect of total extraction on the water balance is not known.

Background

Present water supply and sewerage

The residential and recreation areas in the south of the island have increased rapidly, resulting in greater water demand. The municipal extraction rate has recently increased from approximately 3 ML per day to 6 ML per day, the new capacity of the water treatment plant.

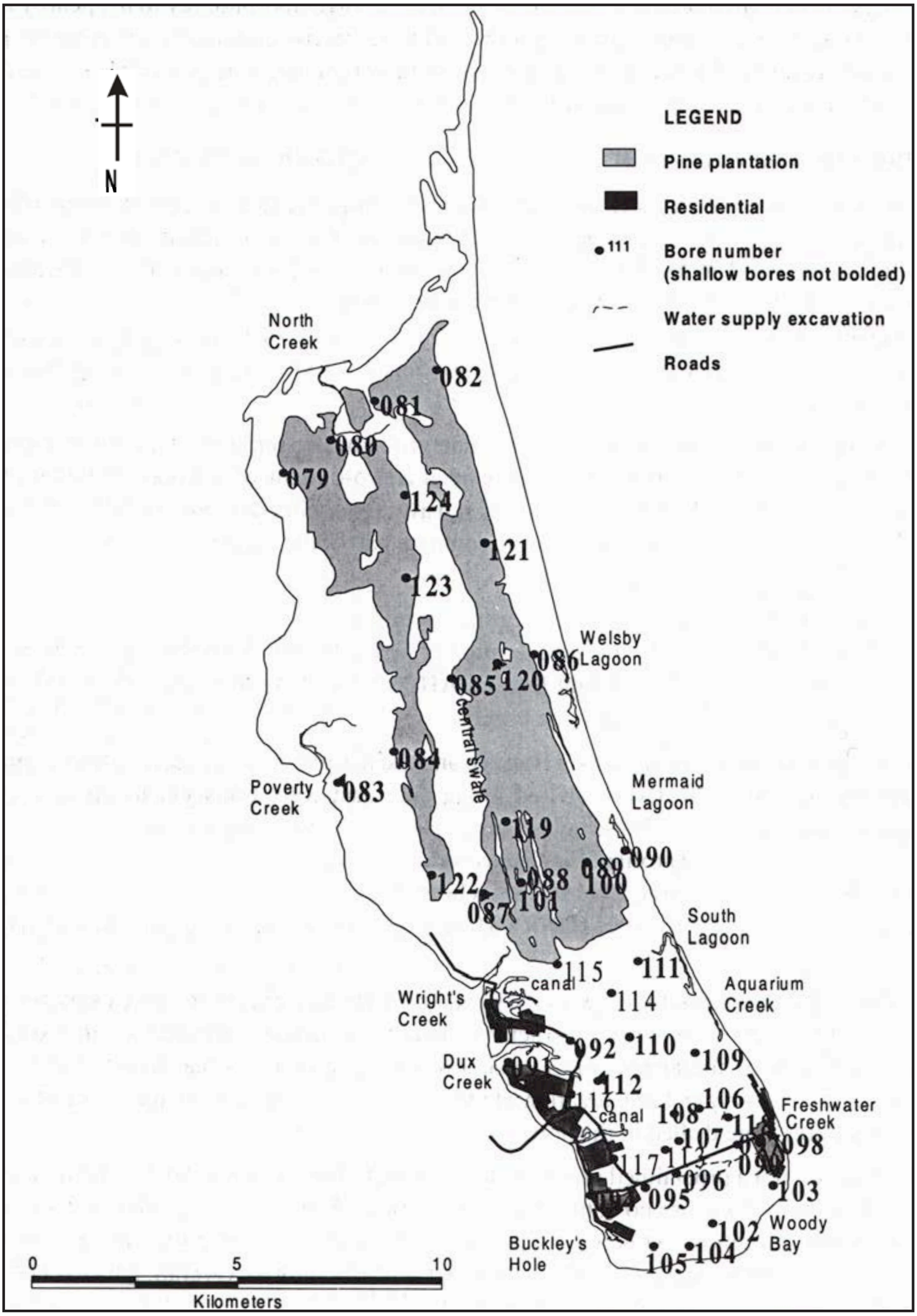

Figure 2. Borehole locations, drainage features and surface water bodies on Bribie Island.

Between 1 and 1.5 ML per day is currently being pumped from a second, more recent water supply excavation into the main excavation. The lower water quality of the second excavation (high iron and manganese) is diluted sufficiently by the better quality water in the primary excavation to enable successful water treatment. The current drawdown due to pumping in the second excavation is 4 m, and the water level is minus 2 m AHD.

Sewage is directed to four infiltration ponds located within the Water Reserve. Approximately 3 ML per day of secondary treatment sewage infiltrates from these ponds in the recent sand deposits and is intended to act as a buffer between fresh groundwater and the saltwater interface. Three further infiltration ponds are planned to be installed seaward of the current ponds.

Monitoring network

The DNR maintain the current observation borehole network. Water levels have been monitored on a monthly basis since 1992. Water quality for registered boreholes were assessed in February 1992, March 1992, September 1995 and August 1996. A series of shallow bores were also installed by DNR in 1994 to monitor perched water levels.

The DNR borehole monitoring network includes three pairs of bores, located so as to monitor the effect of clay layers on the behaviour of the sand aquifer. One bore on the central eastern coast is screened at two depths, and is therefore not representative of a water body at a particular depth. This bore is automatically monitored and includes a rain gauge with a tipping-bucket system. A gauge-board situated at the second water extraction trench is also monitored. Some boreholes in this network are inoperative as they have been damaged by vandalism or heavy machinery.

Physical setting

Bribie Island is one of three large sand islands in Moreton Bay. Unlike the two barrier bay islands, Moreton Island and North Stradbroke Island, which contain massive dune systems up to 200 m above sea level, Bribie Island consists predominantly of successive beach ridges. The island is therefore relatively flat with mean elevation of 5 m and a maximum elevation of 17 m above sea level. These features indicate a different history of sediment deposition. Bribie Island is separated from the mainland by a shallow and narrow tidal estuary, Pumicestone Passage (Figure 1).

Climate

The average annual precipitation for southern Bribie Island for the period 1977-1996 is 1 400 mm. Beerwah Forest to the north-east and Caloundra to the north receive approximately 1 600 mm rainfall per annum, while Redcliffe Airport and other Redcliffe Peninsula weather stations to the south receive approximately 1 250 mm per annum. Previous weather stations located on southern Bribie Island recorded annual rainfalls of approximately 1 350 mm. Presumably, the northern part of Bribie Island receives as much as 20% more rain than the southern part.

While the local rainfall trend is partly orographic, the predominant influence is a rain shadow of the high outer sand islands of Moreton and North Stradbroke, which receive considerably more rain. The wettest months are in summer and autumn with long dry spells in winter. Autumn is the period of greatest discrepancy between northern and southern rain stations and coincides with the period of southeasterly winds. Under these conditions the windward, higher sand islands are likely to reduce precipitation on the southern part of Bribie Island.

Landforms

The estuarine, deltaic and dune systems on Bribie Island each have characteristic sediment type, topography and vegetation. The island is relatively low lying and primarily formed of longitudinal sand dunes, with a maximum elevation of 17 m. Dune systems are composed of aeolian sands deposited with the longitudinal axis of the dunes aligned with prevailing southeast winds. The dunes are fully vegetated and therefore stabilised. Deltaic deposits consist of low-lying poorly sorted sediments, generally found in the north and west of the island.

Vegetation

Vegetation types in the pristine areas on the island are related to soil water availability, soil nutrient status, salinity and waterlogging tolerance, and landform age. On Pleistocene sediments, low *Banksia* woodlands dominate beach ridges while open *Melaleuca* woodland and open heathland dominate in swales. Open forests of *Eucalyptus tereticornis* and *Lophostemon suaveolens* dominate the deltaic sediments in the west of the island. Some dry rainforest occurs on ridges on the western coast. On Holocene sediments, *Acacia, Callitris*, and *Eucalyptus* forest is dominant. *Melaleuca* forest occurs in freshwater swamps with a groundcover of waterfern (*Blechnum* sp.). Salt-tolerant *Allocasuarina littoralis* surrounds the salt scalds on tidal sediments in the Poverty Creek area and also occurs on Holocene coastal foredunes. As shown in Figures 1 & 2, exotic pines (*Pinus elliottii*) have replaced large tracts of native vegetation, predominantly on beach ridges.

Sedimentary Geology

The system of dunes, swales and plains on the island is somewhat complex and is a result of variations in sea level over recent geological time. The clayey sands in the western section of the island have deposited as a result of recent lowering of sea level. Pumicestone Passage has incised these sediments, possibly in response to infilling of the central swale. The Pleistocene dune system to the west of the central swale and the larger eastern Pleistocene dune system have been produced by earlier sea level rises.

The typical soil profile on Bribie Island is deep aeolian sands overlying either green-grey mangrove mud, or a red-grey mottled clay, which is probably weathered sandstone. The sands are well sorted and fine to medium grained, consisting predominantly of quartz. The Triassic Landsborough Sandstone consists of sandstones, with shale bands and conglomerate lenses, dipping seaward.

The mangrove muds are principally kaolinite and are grey to grey-white-green in colour. In two of the boreholes drilled, near White Patch and at Woorim, this mud is absent and deep sands directly overlie decomposed sandstone bedrock. Thin lenses of grey to grey-white clays and fine gravels occur at intermediate depths in deeper sand profiles. The Landsborough Sandstone basement outcrops on the adjacent mainland at Ningi and Caloundra, but there is no outcrop of this unit on Bribie Island. The overlying sand profile is deepest on the central east coast of the island, at 40 m depth. The shallowest profile is 5 m in the northwest of the island, where the Pumicestone Passage is shallowest.

Younger Holocene beach ridge accretions in the southern part of the island show longitudinal alignment of dunes, indicating recent changes in longshore current direction (Armstrong, 1990). Current processes are beach ridge accretion such as at Skirmish Point, south of Woorim and erosion of the eastern coastal dunes. The alignment of the longitudinal dunes is likely to be a result of the prevailing wave and wind direction at the time of deposition.

The degree of podzolisation of sand profiles is indicative of the age of the formation. Holocene beach ridges do not show significant profile development, but “coffee rock” formation is more extensive in the older dunes. Preliminary examination shows that this sand rock consists of quartz sand grains cemented predominantly by clays, with filling of pore spaces by organic silts. Ishaq (1980) estimated the maximum depth of the sand rock to be 9 m in the southern part of the island. While this indurated sand is of very low permeability and so influences groundwater movement, the volume of the indurated sand also forms a significant proportion of the aquifer. Previous investigations have not assigned a lower porosity value to this mass of sand.

Hydrogeology

Recharge

A comparison of rainfall with borehole water level readings indicates that four general situations occur:

1. Recharge is well correlated with rainfall. This is typical of well-drained Holocene ridges (e.g. the area south of the Woorim-Bongaree Road).
2. Recharge is poorly correlated with rainfall (i.e. the water table "under-responds" to rainfall events). This situation is found in deeper boreholes in Pleistocene sand deposits overlain by indurated sands.
3. Recharge "over-responds" to rainfall. This is typical of swales where recharge has been laterally directed from topographically higher areas, either as surface runoff or as shallow seepage across indurated layers.
4. Recharge and rainfall are poorly correlated, but there are large hydrograph fluctuations. These fluctuations are common in areas under strong tidal influence.

The above trends have been quantified by curve matching (first degree correlation) of mean cumulative rainfall with water levels, and the range of water level fluctuations (Table 1).

Water levels in bores in the southern part of the island are also subject to anthropogenic influences, such as injection and extraction of water for irrigation and domestic uses. These factors can complicate natural discharge and tidal effects.

Table 1. Correlation of water levels with rainfall for 21 boreholes at Bribie Island.

Physical setting	Number of bores	Correlation with rainfall (R^2)	Minimum water level (m AHD)	Maximum water level (m AHD)	Ranges of fluctuation in water level (m)
Holocene ridges	5	0.83 - 0.89	0.69	3.03	1.42 - 1.85
Pleistocene ridges	7	0.002 - 0.54	1.10	3.24	0.63 - 1.04
Freshwater swamps, swales & deltaic sediments	5	0.55 - 0.83	0.22	2.49	1.10 - 1.75
Tidally influenced	4	0.01 - 0.47	-0.15	1.19	0.30 - 1.34

After rainfall, recharge to the regional water table occurs within 48 h, which allows the response to be studied at spatially discrete points. The study of discharge is more difficult due to considerations such as:

1. The proximity of boreholes to the groundwater divide; and
2. Lack of knowledge about discharge processes (i.e. the relative importance of evapotranspiration, evaporation, stream flow and groundwater discharge to sea).

Chloride accretion is a possible method for estimation of aquifer recharge (Sharma & Hughes, 1985). The use of this method to calculate recharge relies on the conservative nature of the chloride ion and the assumption that recharge is a one-dimensional process (i.e. both lateral flow of infiltrating water and groundwater inflow are assumed to be negligible). The relation is:

$$R = P \times C_P/C_G$$

where R is recharge, P is precipitation, C_P and C_G are chloride concentrations for precipitation and groundwater, respectively.

Mean Cl values of the basal freshwater body on Bribie Island are higher than for equivalent bores on North Stradbroke Island (65 mg/L Cl cf. 40 mg/L Cl), although borewater salinity is spatially and temporally variable due to preferred recharge pathways. The estimated recharge value for North Stradbroke Island is 50% of precipitation (Laycock, 1975), so by inference, the recharge to Bribie Island's regional aquifer should be in the order of 30%. Previous estimates of recharge to the southern part of Bribie Island are 40-45% (Lumsden, 1964) and 13% (Ishaq, 1980).

Variability in chloride accretion within and between sites is a further complication. Atmospheric accession, as measured at the Cooloola sand mass, are markedly greater for the first kilometre inland from the coast and then relatively constant for 20 km inland (Thompson & Bowman, 1984). Therefore, elevated chloride levels in bores adjacent to the eastern coast of Bribie Island may be attributed to the proximity to the saltwater-freshwater interface or relatively high chloride accretion.

Interpolated pan evaporation figures for Bribie Island (unpublished data, DNR) show the mean annual pan evaporation for 1992-1995 was 1 795 mm. While pan evaporation shows variation of 1-12 mm per day, there is a definite seasonal trend with mean monthly maxima in the summer months (225 mm in January) and minima in winter (83 mm in July).

Runoff and surface waters

The creeks draining into Pumicestone Passage (Figure 2) reduce to a trickle during the drier winter months. The unnamed creek draining the central swale to the north-west, and Dux Creek are probably the most significant drainages. The lagoons along the east coast contain brackish tannin waters and are tidal, separated from the bay by small sand bars that can be breached by storm surges. As its name suggests, Freshwater Creek is less saline than the more northerly drainages and flushes out any saline water after rain. Likewise, Buckley's Lagoon, at the entrance to Pumicestone Passage, contains fresh water (500 µS/cm conductivity). Numerous smaller areas within swamps and swales contain tannin-stained freshwater during normal seasons.

Surface-ponded water occurs in drains, fire-fighting dams and wheel ruts and is elevated above the perched water table. Organic silts and clay are transported to these local low points during storms. Upon settling, the mixture of clays and silts provides a seal which reduces the infiltration rate of surface water.

Waterlogging apparently results in poor growth in many of the pine areas. Furthermore, the location of pine plantations generally coincides with areas of perched water. As a result, the variations in long-term or short-term regional groundwater levels cannot be correlated to growth rates of the pines, nor used to estimate evapotranspiration.

Infiltration and hydraulic properties

Previous estimates of hydraulic conductivity have been concerned with the aquifer in the vicinity of the water supply excavations and a large range of values have been calculated for the southern part of the island (typically 17 to 75 m/day). Pumping tests are complicated by the long pumping times required for satisfactory results. No porosity values are available at this stage for clean or indurated sand on Bribie Island and previous studies have assumed a porosity of 0.2.

The basal aquifer has a mean water table elevation of less than 2 m AHD with only slight elevation beneath beach ridges. In previous contouring of the regional water levels, only the water levels in deeper boreholes have been used, i.e. those holes penetrating indurated sand or in Holocene sands where induration is not significant. In the peaty sands north of the Woorim-Bongaree Road (Figure 2), water levels in shallow bores are equivalent to or only slightly higher than water levels in deeper bores.

Further north, standing water levels are further elevated. Water levels in shallow bores 114 and 115 are around 5 and 6 m AHD. The standing water levels in September 1996 at Bores 088 and 089 are 7.7 and 5.0 m AHD respectively which is 3 m above the level of the basal groundwater aquifer and so represents perched water.

The response of groundwater levels at two Pleistocene ridge sites (088 and 124) in the central area (Figure 3) to an isolated rainfall event in June 1996 was automatically monitored using pressure transducers. At 088 there are two nested bores (088 and 101) with screens at depths of 40 m and 20 m respectively. Initial recharge is rapid for all bores (less than 2 days), but other responses are also evident in the shallower Bore 101. It can be demonstrated by curve-matching with hourly barometric pressures that long term barometric fluctuations cause these water level variations. Short term semi-diurnal fluctuations noticeable in the pressure signals are caused by barometric tide; 10am and 10pm barometric highs produce synchronous water level lows.

For the deeper bores, the barometric fluctuations are smaller, with a semi-monthly fluctuation in amplitude due to reinforcement and dampening by an earth tide signal (Todd, 1959). Assuming equal precipitation at both sites, the larger water level rise in Bore 124 (on the western dune system) compared with Bore 088 (on the eastern dune system) may represent less infiltration. A similar variation in water levels was observed for the other Pleistocene ridge nested bores at 089 (Bores 089 and 100).

Groundwater discharge

Coastal discharge via seepage probably occurs mainly at low tide, especially after heavy rain. Turner *et al*. (1996) suggest that superposition of the water table in coastal zones and wave runup are significant in the quantitative assessment of coastal groundwater discharge. Preliminary investigations of conductivity profiles on Bribie Island's southern beaches reveal that shallow groundwater preferentially discharges at the low tide mark. The point of discharge

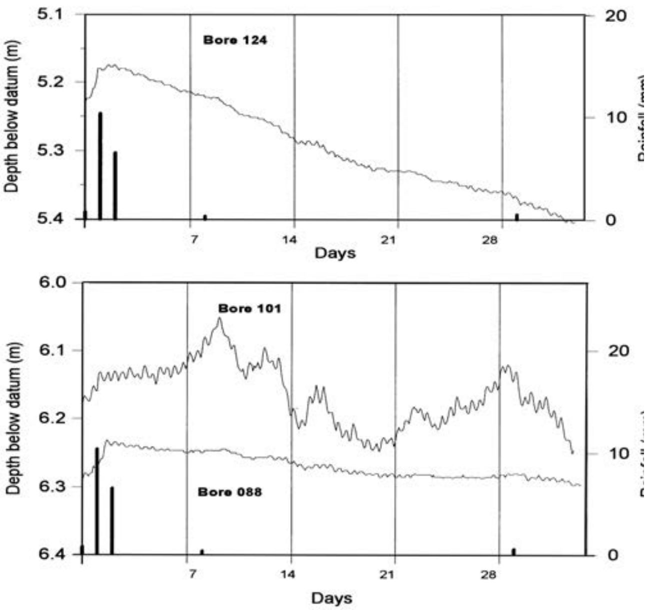

Figure 3. Hydrographs for Bores 088, 101 and 124 showing response to an isolated 30mm rainfall event.

depends upon beach morphology, with freshwater discharge noticeable along a beach face that is concave in plan view, e.g. at Woody Bay and Buckley's Hole (Figure 2). Longshore drift southward along the east coast causes a straighter beach face which may reduce this effect, and hence reduce the opportunity for groundwater discharge to the ocean.
[Note: Longshore drift on Bribie Island is southward, as opposed to the typical northward direction of south Queensland beaches.]

Water Chemistry

Physico-chemical properties

Field measurements of temperature, pH, conductivity and Eh were determined for 47 groundwater and surface waters. Rainwater chemistry was deduced from rainwater tank samples from Bongaree and Woorim. The mean conductivity for boreholes deeper than 5 m (and not subject to saline intrusion) is 370 µS/cm; for shallower bores, the mean conductivity is 245 µS/cm. Groundwaters have a typical pH of 5, with tannin-stained surface and shallow groundwaters being more acidic (pH of approximately 3.5). Redox potential varies between 530 to -40mV for surface water to 180 to -30 mV for deeper groundwater.

As is typical of waters associated with podzolised sands, colour serves as an indicator of a water's source (Reeve & Fergus, 1982). Organic-staining is typical of water which has moved laterally over perched layers; deeper groundwater is clear and colourless. After rain, a mixture of stained and clear water can be observed flowing over white sands at creek crossings. Perched and ponded waters in Pleistocene dunes are stained and of low turbidity. Higher turbidity waters can be found in bores in or adjacent to deltaic sediments, generally in the western part of the island.

Major ion chemistry

In a sand island like Bribie, different bodies of water can be characterised by their salinity and composition however, a logarithmic plot of Na versus Cl concentration (Figure 4) shows that the ratio of these two major ions is constant, confirming local recharge of Na-Cl type precipitation and limited ion-exchange reactions.

The plot of Ca versus Cl (Figure 5) shows Ca enrichment for a significant number of samples, principally the Holocene dune groundwaters, indicating marine carbonate material. Pleistocene sedimentary deposits contain negligible carbonate fragments, and Ca and HCO_3 enrichment in groundwater is a function of flow path migration distance. According to Thompson & Bowman (1984), nearly all of the carbonate in coastal dunes is removed from near the surface within 300 years, with an age-dependent leaching front that increases in depth with distance from the sea.

Both Na/Cl and Ca/Cl plots (Figures 4 & 5) indicate the range of concentrations between seawater and rainwater. Higher salinities tend to occur in coastal lagoons, a tidal creek and

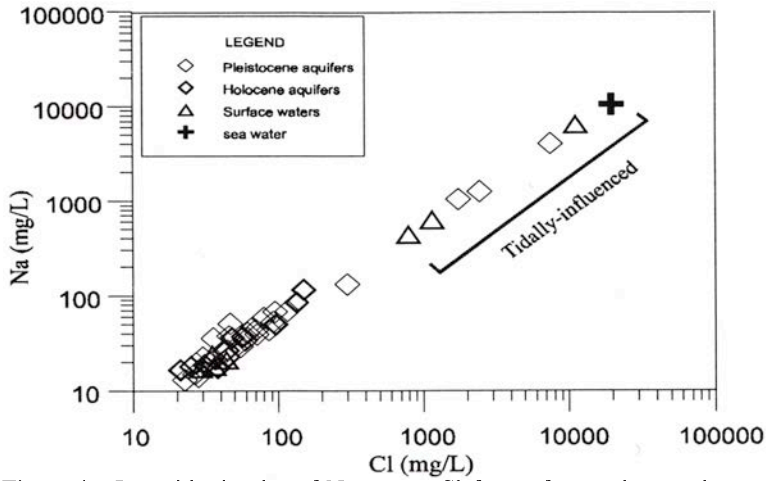

Figure 4. Logarithmic plot of Na versus Cl for surface and groundwaters compared to mean seawater.

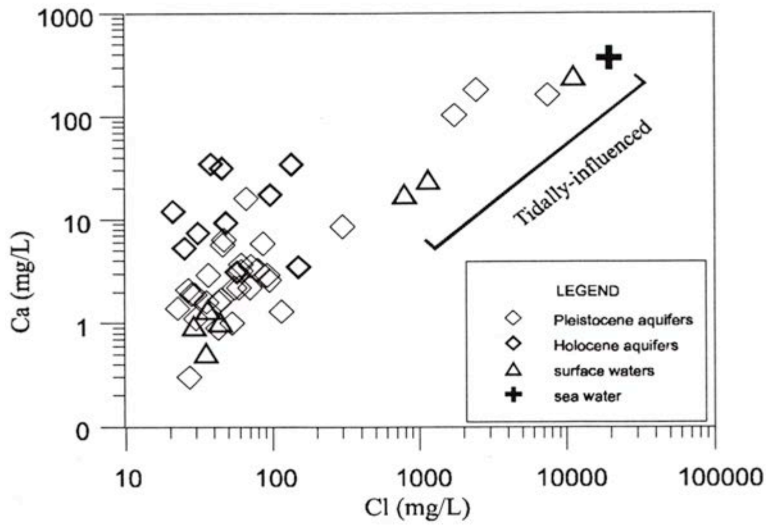

Figure 5. Logarithmic plot of Ca versus Cl for surface and groundwaters of Bribie Island compared to mean seawater.

some tidally-influenced boreholes. Ponded rainwater and water from flowing creeks are of lower salinity.

The relative ratios of the major elements as represented in the Piper diagram in Figure 6 show the following trends:

- the majority of water samples are of Na-Cl character, as are the endpoint rainwater and seawater samples;
- Ca and HCO_3 enrichment in deeper bores in the Holocene ridges is due to layers of shell fragments. Some of these waters are saturated with respect to carbonate minerals, especially dolomite and calcite. Shallower bores in the Holocene ridges and the southern Pleistocene ridges are SO_4-enriched;
- deeper bores in the central-eastern part of the island exhibit HCO_3 enrichment with lesser Ca, indicating less marine material but longer migration; and
- increase in the proportion of Na occurs in some bores proximal to the coast. This may be indicative of either fresh water flushing of saline intrusion, or depletion of Mg. Variable conductivity over time is also evident in these bores.

Minor ion chemistry

As is the case for Fraser Island waters (Little & Roberts, 1982), dissolution of quartz is the main reaction in quartzitic coastal sands. The majority of samples are saturated with respect to silica (SiO_2), the degree of saturation serving as an indication of residence time in deeper bores. Current understanding of trends in Fe and Mn concentrations is not conclusive due to uncertainty of sampling procedures employed; Mn however tends to persist to lower depths while Fe is less mobile. Zinc levels are high in three southern boreholes (3.5, 29 and 12 mg/L), located at an abandoned tip (Bore 108), a disused sewage outlet (Bore 104) and south of Woorim

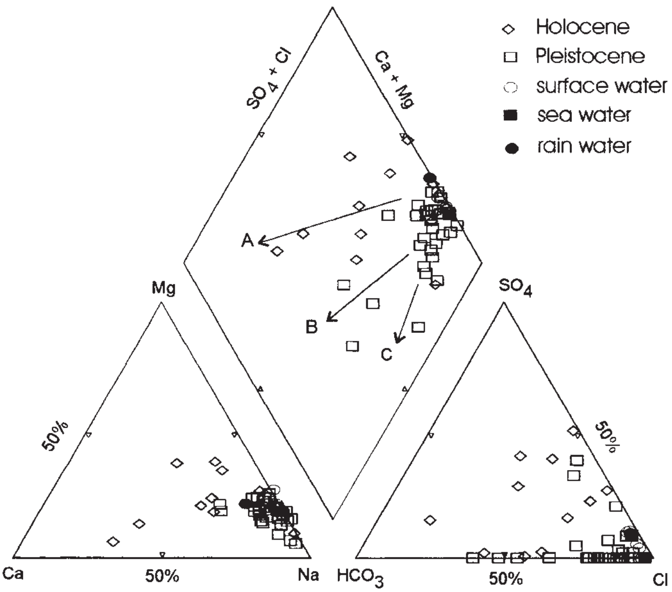

Figure 6. Piper diagram for a selection of surface and groundwaters of Bribie Island showing indicated trends of chemical evolution. (A) Ca-HCO_3 enrichment; (B) HCO_3 enrichment; (C) Mg depletion.

township (Bore 103). The elevated Zn is presumably due to the population concentration in this part of the island: NO_3 and F levels are low and do not show significant trends.

Discussion and Conclusions

The concept of a groundwater mound following the local topography is inadequate to describe the hydrology of Bribie Island since the relief and consequently the hydraulic gradients are negligible. Due to the presence of horizons of indurated sands, infiltration on the higher Pleistocene dunes is reduced. The schematic diagram (Figure 7) illustrates flow directions for a central east-west transect of the island. The higher than predicted water level at Bore 090 is attributed to (a) superelevation by the ocean tide and (b) drainage of perched water from the adjacent dune.

Layers of indurated sand ("coffee rock" or "beach rock") appear to form in coastal areas. The extent of sand induration on Bribie Island is probably a result of sea level control of inland water levels. Both the volume and porosity of indurated sand, and the extent of saltwater intrusion are significant in assessing the volume of the freshwater resource and are currently being considered. While indurated sand can be assumed to be relatively impermeable compared to clean medium-fine grained sand, discontinuity of induration and secondary porosity may enhance permeability.

Acknowledgements

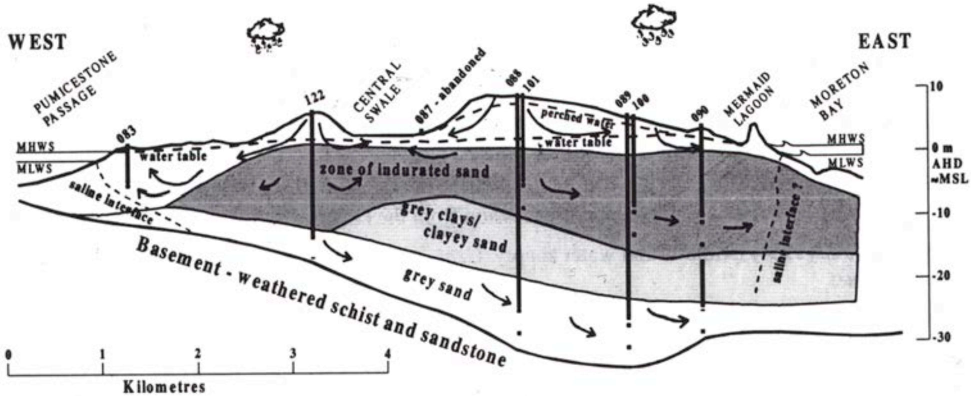

Figure 7. Conceptual model of groundwater flow for a broad central transect of Bribie Island.

Assistance by staff of the Department of Natural Resources, particularly Peter Cochrane, John Hillier, Jerome Arunakumaren and Stuart Wade, in providing data, expertise and field equipment, is greatly appreciated.

Thanks are also extended to John Laycock of the Queensland University of Technology, John Broadbent of Queensland Transport – Maritime Division, and Caboolture Shire Council staff involved in the water management of Bribie Island for helpful discussion.

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The Holocene Coastal Evolution of Deception Bay, Southeast Queensland: Implications for Prehistoric Cultural Heritage Management within Northern Moreton Bay

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Introduction

Deception Bay forms the most northern mainland element of the larger, more widely known Moreton Bay. It is a low energy tidal embayment with a shoreline characterised by a relatively flat coastal plain itself dominated by estuarine mudflats and prograded beachridges. The local prehistoric archaeological record of Deception Bay features a diverse range of site types including a ceremonial bora ground, a possible stone-fish trap, the well-documented Sandstone Point midden complex, a probable swamp fern processing site, and numerous undated shell middens (Cotter, 1996). The chronological evidence suggests that prehistoric occupation and resource exploitation of Deception Bay has occurred only within the last 2 000 years in contrast to regional evidence indicating occupation of the southeast Queensland coastal lowlands for the past 20 000 years (Hall, 1987).

In the context of an environmental archaeological study this paper reports summary details of palaeobotanical and geochemical evidence for the mid to late Holocene coastal evolution of Deception Bay. Radiocarbon and thermoluminescence age determinations of several sediment facies within the Bay provide a chronometric framework for the coastal evolution of this region that is directly comparable to the archaeological record (Cotter, 1996). In light of this comparison discussion is directed towards an assessment of the likely impact of this Holocene coastal evolution on the formation, preservation and current visibility of the archaeological record of the Bay. Finally, an assessment is made of (a) the importance of understanding coastal geomorphic processes for the effective management of prehistoric cultural heritage within the coastal zone and (b) the specific implications of the Holocene evolution of Deception Bay on the management of the prehistoric cultural heritage of northern Moreton Bay.

Geomorphic Processes

The evidence suggests that coastal progradation, wind-blown sand transport, and the intermittent formation of tidal lagoons adjacent to the shoreline are features of the geomorphic history of the Bay. In particular, influxes and subsequent declines in mangrove pollen types (i.e. Rhizophoraceae, *Avicennia & Aegiceras*) through time, in sediments situated 1.75 km landward of the present coastline, provides clear evidence both for sea level fluctuations in the region during the mid-Holocene as well as for subsequent progradational events. Geochemical evidence of acid-sulphate soil conditions concurs with this palaeobotanical evidence for a past marine transgressive event within the Bay. Importantly AMS dating of associated sediments indicates that the most significant positive change in sea-level/tidal regime occurred at ca 4 600 BP. Later, as sea level began to drop to present levels, shoreline progradation was facilitated by a relatively stable supply of marine sediments so that a characteristic ridge and

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 125-127.

swale system has developed parallel to the present coastline. In addition particle size analyses indicate that aeolian sands have accumulated over the earliest prograding dunes obscuring much of the microtopography.

Implications for the archaeological record

This geomorphic history provides some explanation of both the spatial and temporal patterns observed in the local archaeological record of Deception Bay and has several implications with regard to the formation, preservation and interpretation of that record. These are summarised below.

(a) It is unlikely that evidence of early Holocene or late Pleistocene occupation of the coastal lowlands will be found in this region. Moreover if Pleistocene occupation occurred in the area all evidence would now be overlain by several metres of Holocene sands.

(b) The palaeobotanical evidence suggests that the tidal lagoon system presently featured along the Deception Bay shoreline is a modern consequence of an intermittent but repeated geomorphic process operational in the area throughout the mid to late Holocene. It is probable therefore that the resource zones presently observable within Deception Bay (including estuarine mudflats, tidal lagoons, and *Melaleuca* swamps) existed in some form throughout the past 5 000 years and hence were available throughout this period for prehistoric human exploitation despite the lack of archaeological sites of this antiquity.

(c) Artefactual material noted eroding out of dunes dated to ca 5 000 years BP suggest that elements of the mid-Holocene component of the regional archaeological record have been overlain with aeolian sands, rendering them invisible in the present landscape, and providing at least one environmental reason for the lack of observable sites of more than 2 500 years old in the area.

(d) The unconsolidated sediments of the estuarine mudflats and prograded dunes of the coastal plain do not provide the raw material resources for many of the lithic artefacts found within Deception Bay. As a consequence artefacts found within these dunal systems must have been transported to the area, perhaps through trading networks. It is likely therefore that a lack of suitable geological outcrops for stone tool manufacture within northern Deception Bay impinged upon the nature and type of social interactions in which the prehistoric occupants of the area engaged.

Implications for prehistoric cultural heritage management within Moreton Bay

It is clear that a full appreciation of the spatial and temporal patterns observed in the archaeological record of Deception Bay is concomitant on a detailed understanding of the nature, timing and extent of coastal geomorphic change within the Bay. This finding has a number of implications for cultural heritage management initiatives and practice within the coastal zone, and specifically for northern Moreton Bay.

(a) Prehistoric cultural heritage management within the coastal zone requires the adoption of interdisciplinary studies for the effective determination of the significance and vulnerability of the local and regional archaeological record. Of paramount importance is the determination of the effect of coastal processes on the form, preservation and visibility of the known archaeological record (cf Boyd, 1982; Boyd *et al.* 1995; 1996a; in press).

(b) Within Deception Bay it is apparent that there exists a subsurface component to the prehistoric archaeological record. Therefore cultural heritage management strategies which are designed to ascertain and protect surface exposures of cultural material only, are likely to be inadequate.

(c) Northern Moreton Bay exists within one of the fastest growing regional areas of Australia (Tarte, 1993). To maximise the effectiveness of cultural heritage management in this rapid urban growth area, prior to new development, subsurface assessments for cultural heritage material should be carried out. This risk assessment of buried archaeological material (cf Boyd *et al.,* 1996b) would thereby allow for the most appropriate development strategies to be adopted for the coastal zone of northern Moreton Bay.

Conclusions

The effective management of a rich and diverse prehistoric archaeological record within Northern Moreton Bay is clearly dependent on a thorough knowledge of landscape evolution and the dynamic geomorphic processes in operation within this coastal zone. This paper demonstrates that the geomorphic evolution of Deception Bay has impinged on the preservation and current visibility of the known archaeological record. Moreover, by indicating the existence of a subsurface component to the archaeological heritage of the area the paper highlights the need for cultural heritage management strategies which incorporate buried cultural material risk assessments in their design and implementation. With a thorough knowledge of such risks informed and appropriate strategies for development can be formulated for this rapid urban growth area.

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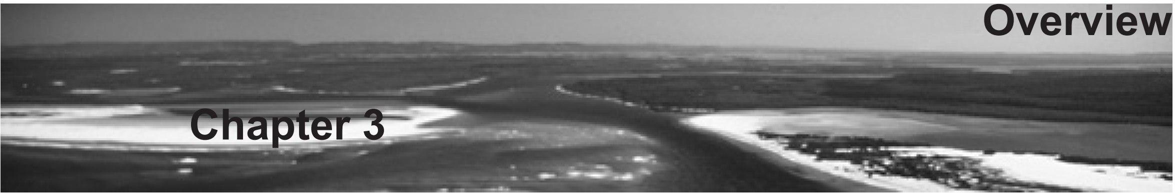

Overview

Chapter 3

Catchment Rivers and Lakes

This book has as its focus Moreton Bay and its catchment. Physical, chemical and biological systems in the Bay and catchment are strongly influenced by the inputs from the terrestrial catchment to the rivers and water storages and then the output from these water bodies to the Bay itself. Thus the waters of the Bay and catchment are seen as an interdependent system where activities in one part of the system would have consequences for other areas, particularly those downstream. Accordingly the prime objectives for this chapter were to:

- Obtain an overview of the operation of the overall functioning of the catchment from the headwaters to the estuary and the Bay in terms of linking factors such as energy, nutrient and water flows; and
- Evaluate the influences of activities in the catchment on the water quality in the catchment waterways.

Stuart Bunn outlines the various concepts which could be used for describing functioning of the Brisbane River system are outlined, including the River Continuum concept, Flood-pulse concept and the Riverine Productivity Model. Which of these apply to the Brisbane River system is not known at present and it is possible that they all do to some degree. The role of riparian vegetation in introducing energy and nutrients into the system is emphasised. On the other hand riparian zones can trap sediments, nutrients and other detritus from the catchment, enhance bank stability and prevent erosion. Also the riparian zone is noted for its diversity and is of considerable biological value in itself.

Angela Arthington's paper discusses the timing, frequency, duration, rate of change and predictability of water flows governing the nature and characteristics of the aquatic ecosystem. The construction of two large dams, Wivenhoe and Somerset, have introduced major discontinuities into the river system. Now there exist upper and lower components in which the upper component is highly controlled in accord with the requirements for raw water for Brisbane and flood control while the lower component is essentially uncontrolled. This, together with a range of other modifications to the system, has resulted in a considerable loss of habitat, and modifications to the aquatic ecosystem, in response to the modified flow regime.

My paper draws attention to the water quality characteristics in specific zones of the catchment which share similar problems. These were the water storages (Wivenhoe, Somerset and North Pine Dams) and associated catchments, the rivers affected by agricultural and rural activities (Bremer River and Lockyer Creek) and the urbanised estuary (lower Brisbane and Bremer Rivers). The water storages were reported to have difficulties in the control of blue-green algal blooms while the Bremer River and Lockyer Creek contained excessive amounts of sediments and nutrients derived from soil erosion. The urbanised estuary was affected by similar problems together with depletion of dissolved oxygen and widespread contamination with toxicants.

In considering the overall implications of the comments made in the papers above, some general observations can be made. To manage water quality and protect the natural aquatic ecosystem a better understanding of the functions of the system is needed. The linkage from catchment to rivers, or water storage, to Bay in terms of energy, nutrients and water flows needs to be evaluated. This will then allow the role of energy, nutrients and water flows from point and nonpoint sources to be understood and control of adverse effects from these sources maximised. Of particular importance is the evaluation of the environmental flows of water for the whole system. Water quality in many water bodies in the catchment is degraded. Sediments and nutrients are major contaminants affecting water quality although oxygen reduction and toxicants are significant in some areas.

Des Connell

Riparian Influences on Ecosystem Function in the Brisbane River

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Abstract

Riparian vegetation is known to have a strong direct influence on the function of forest stream ecosystems, as a regulator of in-stream primary production through shading, and as a supplier of energy and nutrients in the form of leaf litter and other organic debris. Logs and branches from the riparian zone provide important structural habitat and contribute to the diversity of flow and substrate conditions that favour high faunal diversity. In addition to these direct ecological roles, riparian zones have the potential to trap sediments, nutrients (and other contaminants) from the surrounding catchment and, under some circumstances, the vegetation may also enhance bank stability and prevent erosion and slumping. Given these multiple roles, there is little doubt that protection and, where appropriate, rehabilitation of riparian lands in the upper catchments of the Brisbane River will lead to the protection of stream ecosystems and maintenance of high water quality.

We are much less certain, however, of the degree to which riparian zone management can influence ecosystem function in the main-stem Brisbane River. Current models describing ecosystem function of larger rivers (e.g. River Continuum Concept, Flood-Pulse Concept and Riverine Productivity Model), differ considerably in their emphasis on the strength of direct riparian linkages. We need to establish which of these models, or some combination of the three, best describes the Brisbane River ecosystem. Without this, we will not be able to predict the consequences of changes to lateral and longitudinal exchanges of energy and nutrients resulting, for example, from river regulation and/or changes in land-use. Nor will we be able to identify an appropriate strategy for restoration.

Introduction

Understanding the flux of organic carbon in streams and rivers is an essential requirement for the sustainable management of riverine environments as healthy and natural ecosystems. In part, this is because carbon is the principal building block of all living tissue, and the fundamental element that drives ecological systems. Perhaps equally if not more important, however, is the recognition that human activities result in considerable changes to the carbon cycle (see Degens *et al.,* 1991). The need to understand such basic ecosystem processes in Australian rivers has been emphasised in recent years (e.g. Bunn, 1995; Arthington *et al.,* 1996a; Robertson *et al.,* 1996).

Much of the research on stream and river ecosystem function has highlighted the importance of riparian vegetation, as a regulator of in-stream primary production through shading, and as a supplier of energy and nutrients in the form of leaf litter and other organic debris (e.g. Hynes, 1975; Vannote *et al.,* 1980; Cummins *et al.,* 1995). While this is certainly true for many forested stream ecosystems, there is some debate as to the importance of direct riparian links in large river ecosystems (see Robertson *et al.,* 1996). Current models describing ecosystem function of larger rivers, e.g. River Continuum Concept (Vannote *et al.,* 1980), Flood-Pulse Concept (Junk *et al.,* 1989), and Riverine Productivity Model (Thorp & Delong, 1994), differ considerably in their emphasis on the strength of direct riparian linkages.

Arthington *et al.* (1996a) proposed that 'river hydrology' and 'carbon dynamics' were the two most important ecological 'drivers' to be considered in the management of the Brisbane River.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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The major aims of this paper are to: (i) describe key ecosystem processes in the Brisbane River, in the context of current models of stream and river ecosystem function, with an emphasis on riparian linkages; (ii) examine how past and present human activities may have altered these processes, and (iii) consider the implications of riparian restoration to river health. The direct influence of river regulation on carbon dynamics will also be discussed, as these two are tightly linked. Flow-related issues are dealt with in more detail by Arthington & Mosisch (this volume).

Tributary Streams and their Catchments

Sources of organic carbon

Factors that influence the nature and extent of catchment vegetation cover (e.g. climate, extent of land-clearing) ultimately determine the quantity and quality of carbon sources to river systems (Degens *et al.,* 1991; Robertson *et al.,* 1996). In forested catchments, the major terrestrial inputs of carbon to rivers include:

(i) logs, branches and other large woody debris (LWD), primarily as direct inputs from riparian vegetation;

(ii) litter inputs and other coarse-particulate organic matter (CPOM) directly from riparian trees, or washed or blown in from elsewhere in the catchment;

(iii) inputs of fine-particulate organic matter (FPOM); and

(iv) dissolved organic matter (DOM).

There is little doubt that food webs in many forested streams are dependent on these terrestrial sources of carbon (e.g. Hynes, 1975; Vannote *et al.,* 1980; Rounick *et al.,* 1982; Bunn, 1986; Rosenfeld & Roff, 1992). It is often not clear, however, as to which of the major components of terrestrial carbon flux (CPOM, FPOM or DOM) are most important. Fruits from riparian tree species (e.g. species of *Ficus* in the tropics and sub-tropics), and insects and other terrestrial arthropods may provide a smaller, but much higher quality, source of carbon for higher order consumers in rivers (see Bunn, 1993; Pusey & Kennard, 1995). Australian freshwater fish, particularly those in the temperate southern systems, show a high dependence on direct inputs of insects from the riparian zone.

The influence of riparian vegetation on stream function

In addition to the direct supply of these carbon sources, catchment vegetation can play an important role as a 'regulator' of in-stream primary production through reduction of incident light to stream channels and lowering of water temperatures (Gregory *et al.,* 1991). This dual function of terrestrial supply and regulation of in-stream production is most obvious at the riparian zone and is especially important in smaller tributary systems. Terrestrial inputs of carbon to forest streams generally exceed aquatic sources within the stream channel (e.g. algae and macrophytes), and such systems are generally accepted to be heterotrophic (see Bunn, 1986; 1993; Cummins, 1993).

Shading by riparian vegetation may not be the only mechanism, however, of regulation of in-stream primary production. Some rivers in Australia have high turbidities and water clarity may limit production of phytoplankton and macrophytes (see Walker, 1986; Lake, 1995). The overall nutrient status of catchments is also a major determinant of in-stream primary production. Given the low nutrient status of many Australian catchments, primary production in undisturbed streams historically would have been low. This is most apparent in the dry sclerophyll forest streams of southwestern Australia, which have some of the lowest levels of net primary production recorded for any stream (Bunn & Davies, 1990; Davies, 1994).

Riparian vegetation can also indirectly affect stream ecosystem function. Inputs of larger woody debris (LWD), such as logs and branches, provide important structural habitat and contribute to the diversity of flow and substrate conditions that favour high faunal diversity. This material also offers a considerable substrate for primary and secondary production (Benke *et al.,* 1984; O'Connor, 1991), as well as influencing the rate of transport of leaf litter and other more mobile sources of carbon (Bilby, 1981). Much of the supply of logs to rivers is from the adjacent riparian zone (Cummins, 1993), though more catastrophic events can periodically delivery large quantities of LWD. Riparian zones also may 'buffer' streams and other water bodies from adjacent land uses, and protect receiving water bodies, by trapping sediment, nutrients and other contaminants (see Bunn *et al.,* 1993, Woodfull *et al.,* 1993; Haycock *et al.,* 1997). Under some circumstances, the extensive root systems of the vegetation may also enhance bank stability, and prevent erosion and slumping.

Longitudinal and Lateral Linkages in Large Rivers

As we move downstream from small tributaries through the channel network, the relative importance of terrestrial and aquatic sources of carbon will vary because:

(i) the direct (lateral) contributions from riparian vegetation decrease relative to inputs from up-stream processes; and

(ii) at the same time, the increased channel dimensions reduce the direct effect of vegetation as a regulator of in-stream primary production. Three major models have been proposed to describe the function of larger, 'receiving' rivers.

River Continuum Concept

The River Continuum Concept (Vannote *et al.,* 1980), emphasises the importance of carbon and nutrients "leaked" from upstream processes to the structure and function of larger river reaches (Figure 1a). In this model, the middle order reaches, where the direct effects of riparian shading are diminished, have an increased dependence on in-stream primary production. FPOM is the principal carbon source for food webs in downstream reaches, and much of this is derived from upstream processing. Direct inputs of CPOM from adjacent riparian vegetation are insignificant and, in larger more turbid rivers, in-stream primary production is limited by turbidity and depth (Figure 1a).

Flood-Pulse Concept

More recent studies have emphasised the importance of allochthonous sources of organic matter, other than those derived from upstream processes, to the function of some large (floodplain) rivers. The Flood-Pulse Concept (FPC) (Junk *et al.,* 1989) emphasises the important river-floodplain interactions and proposes that riverine food webs are driven by production from the floodplain rather than by transported organic matter from upstream (Figure 1b).

Riverine Productivity Model

An alternative theory for carbon fluxes in some large rivers is the Riverine Productivity Model (Thorp & Delong, 1994). This model emphasises the importance of (i) local autochthonous production (phytoplankton, benthic algae, other aquatic plants); and (ii) direct inputs from the adjacent riparian zone (CPOM, FPOM, DOM) (Figure 1c). Thorp and Delong (1994)

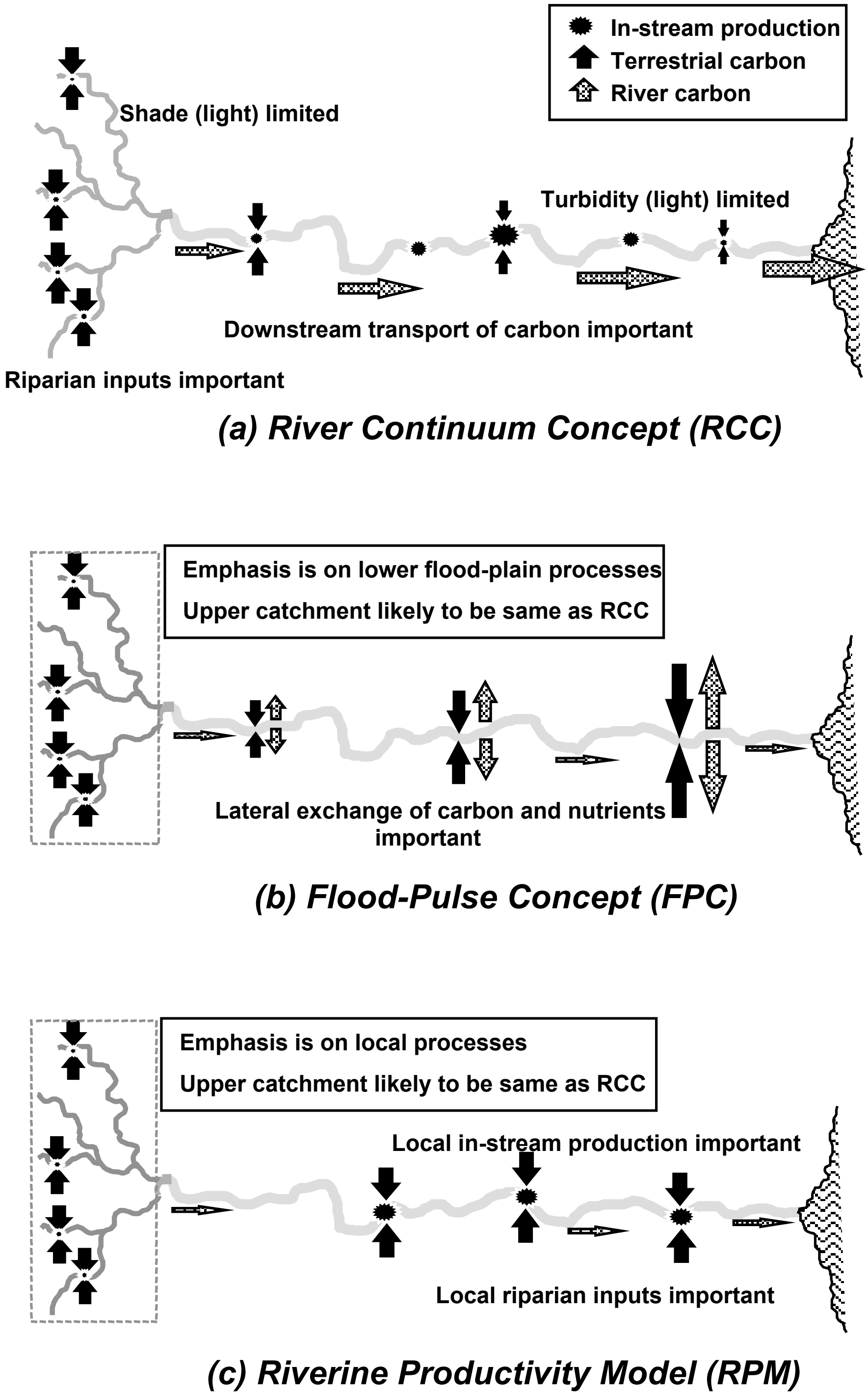

Figure 1. Comparison of major carbon fluxes emphasised in three models of river ecosystem function: (a) River Continuum Concept (Vannote *et al.,* 1980); (b) Flood-Pulse Concept (Junk *et al.,* 1989); and (c) Riverine Productivity Model (Thorp & Delong, 1994). In each case, the size of the arrows and stars is indicative of the relative importance of the carbon flux.

argue that the previous two models have underestimated the role of local sources and have instead overemphasised the transport of organic matter from headwater streams (RCC) or floodplains (FPC).

A Model for the Pre-European Brisbane River

It is likely that the most appropriate model to describe ecosystem processes in the Brisbane River before European settlement is one that incorporates elements of at least the first two of these concepts (Figure 2a). The RCC is likely to be an appropriate model for tributary forest streams and the smaller branches of the Brisbane River. Our recent studies of ecosystem processes in upper tributary streams of the Conondale Ranges, indicate a strong riparian control of in-stream primary production and a food web based on terrestrial inputs of organic carbon (unpublished data). For larger, 'receiving' rivers, we might expect an increase in autotrophic contributions to aquatic food webs (Vannote *et al.,* 1980). However, if light was limited by high turbidity and/or depth in larger streams and rivers, then terrestrial inputs would be the major source of carbon, simply because of the low availability of higher quality, algal food sources. Under such circumstances, sources of carbon derived from upstream processes (sensu Vannote *et al.,* 1980) and lateral exchange, either from direct riparian inputs (Thorp & Delong, 1994) or pulsed inputs from the floodplain (Junk *et al.,* 1989), would be of greater importance.

The lateral perspective of the FPC may have been more appropriate for the lower floodplain sections of the Bremer, Oxley and Lockyer, as well as the lower sections of the Brisbane River itself. However, estuarine processes may also have played a major role in the flux of carbon and nutrients in the lower tidal sections of the river (Figure 2a). None of the current models

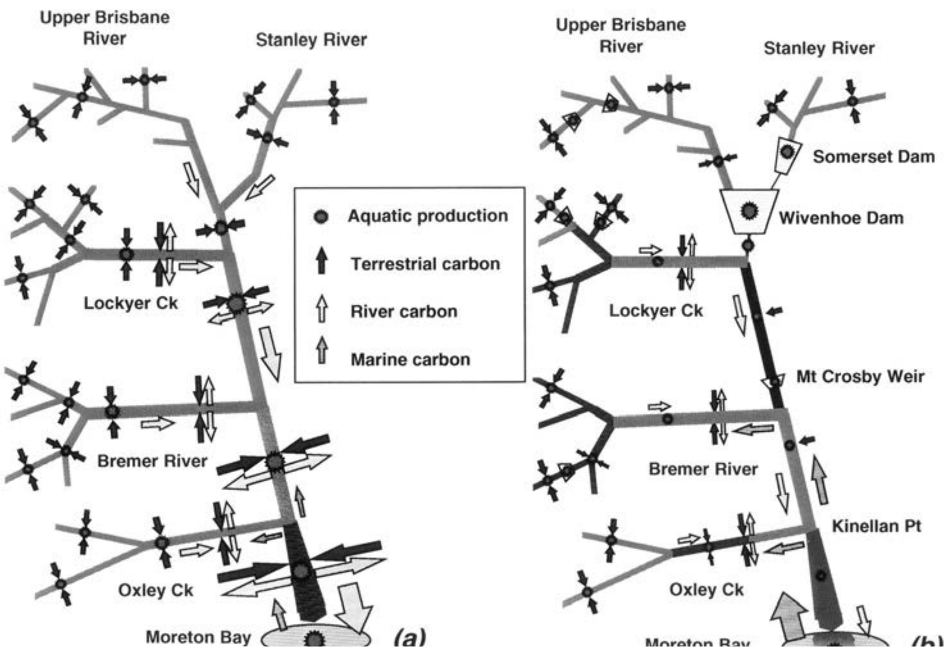

Figure 2. Schematic diagram of the Brisbane River and major tributaries (a) before European settlement and (b) present day, showing likely changes to the major carbon fluxes associated with catchment and riparian clearing, and river regulation. In each case, the size of the arrows and stars is indicative of the relative importance of the carbon flux.

Changes to Carbon Flux in the Brisbane River

Catchment vegetation

The clearing of catchment vegetation has a considerable effect on the quantity and nature of carbon inputs to rivers. At the simplest level, clearing of forest vegetation considerably lowers annual litter inputs to streams and rivers (e.g. Pidgeon, 1978; Campbell *et al.,* 1992). Replacement of native vegetation with agricultural crops can also have a major effect by changing the type of carbon source entering streams. For example, benthic detritus in caneland streams in north Queensland is primarily from C4 (cane plus invasive pasture grasses), rather than C3 plants (Bunn *et al.,* 1997). Given the apparent inability of aquatic consumers to assimilate C4 carbon (see Forsberg *et al.,* 1993), this can have a significant effect on carbon flux in these disturbed systems. Much of the catchment of the Brisbane River has been cleared for agriculture (60% of total catchment) and urban plus rural residential development (28% of total catchment) (Arthington *et al.,* 1996a; b).

Changes in land use can lead to substantial changes to the quantity, composition and timing of catchment inputs of organic matter. One noted feature of many Australian rivers and farm dams is the sudden flux of livestock manure that follows intense rain events (e.g. Bunn & Davies, 1992). Summer thunderstorms can deliver large quantities of such material at times of otherwise low flow, and its subsequent decomposition can lead to anoxic conditions and the death of fish and macrocrustaceans.

Clearing of riparian vegetation

Clearing of riparian vegetation can result in a major shift from heterotrophy to autotrophy, particularly in smaller streams and especially when accompanied by nutrient additions. To some extent, increases in light and nutrients associated with land and, particularly, riparian clearing may have a positive effect on productivity in rivers by stimulating high quality algal sources (e.g. diatoms). Such sources tend to be more palatable than terrestrial detrital material and make a disproportionately high contribution to the diet of aquatic consumers. However, particular problems occur when the system shifts from these palatable plant species, to filamentous green algae, macrophytes and cyanobacteria. The condition of riparian vegetation in developed areas of the Brisbane River catchment is generally poor with pockets of higher quality found only in headwaters and National Parks (Arthington *et al.,* 1996a; b). Such degradation has undoubtedly resulted in similar shifts in ecosystem function (Figure 2b).

Large vascular plants and filamentous algae can proliferate, restricting flow, trapping sediment and ultimately resulting in marked changes to available habitat and lowered water quality. A spectacular example of this is the excessive growth of para grass (*Brachiaria mutica*) in stream channels in the canelands of northern Queensland (Bunn *et al.,* 1997). Clear relationships have been established between the extent of riparian cover and aquatic plant biomass (e.g. Canfield & Hoyer, 1988) or production (e.g. Gregory *et al.,* 1991).

Removal of riparian vegetation can also directly reduce the inputs of litter and, perhaps more importantly to fish and other higher order consumers, fruits and insects. For example, herbivorous turtles (*Elseya dentata*) in the Daley River are found associated with dense riparian vegetation, particularly containing species of *Ficus* (see Sattler, 1993).

River regulation

Flow regulation and impoundments can have a major influence on the health of riparian forests. For example, recruitment of river red gum along the River Murray is encouraged by flooding, though trees are intolerant of continued submersion in the backwaters of weirs and reservoirs (Walker, 1986). The potential effects of river regulation on carbon dynamics are, however, far more pervasive. Elimination of periodic flooding may disconnect the river from its floodplain, a potential major source of carbon and nutrients. Isolation of the floodplain from the main channel can lead to the accumulation of litter on forest floors and an increased risk of fire (Molles *et al.,* 1995). Regulation of the Brisbane River, particularly by Wivenhoe Dam, and urbanisation of much of the lower catchment has dramatically limited potential floodplain exchange of energy and nutrients in the lower reaches (Figure 2b).

Major storages and even smaller weirs enhance retention of organic debris in transport, and such material may be 'lost' to consumers in the deeper, anoxic regions of stratified impoundments. Somerset and Wivenhoe Dams will have greatly altered the quantity and quality of carbon exported from the Upper Brisbane and Stanley River catchments to the lower Brisbane River.

At the same time, impoundments and flow reduction in general encourage the development of phytoplankton assemblages. Phytoplankton represents a high quality carbon source compared with terrestrial detritus and can represent a major carbon source for riverine consumers downstream of impoundments (e.g. Glen Canyon, Colorado – Angradi, 1994). The often observed result in nutrient enriched systems is the development of noxious cyanobacteria blooms (Boon *et al.,* 1994). Stable isotope studies have confirmed that little carbon from blue-green algae is incorporated into planktonic food webs in lentic systems, although they can be a major contributor to the nitrogen pool (Estep & Vigg, 1985; Bunn & Boon, 1993).

The more lentic character of the regulated reaches of the river also encourages the proliferation of aquatic and semi-aquatic macrophytes, and compounds many of the problems associated with land clearing, indicated above. There is increasing evidence to suggest that such conspicuous primary producers are not major contributors to aquatic food webs in general (Bunn & Boon, 1993; France, 1996). Elimination of peak flows may prevent scouring of sediments and allow the accumulation of large vascular plants which modify habitat but do not contribute greatly to food webs. Similarly, standing crop and production of filamentous algae and its contribution to aquatic food webs are linked to hydrologic regime (e.g. Power, 1992).

Impoundments

Major changes to ecosystem processes occur with the conversion of lotic (flowing-water) to lentic (still-water) habitat with the construction of large impoundments. There is often considerable argument that the loss of productive riverine habitat is more than compensated for by the creation of lake habitats. However, fluctuations in water levels are often far more extreme in impounded water bodies than in natural lakes and this has profound effects on the availability of habitat and, importantly, on ecosystem processes.

The conversion of lotic to lentic habitat is accompanied by a major shift from heterotrophic processes (i.e. energy from surrounding forest) to autotrophic processes (i.e. energy generated in situ by aquatic plants). Rather than being almost entirely 'driven' by forest sources of carbon (energy), aquatic food webs become dependent on algae and other aquatic plants. One consequence of this is a major shift in the composition and dynamics of the fauna. The other is that the system becomes far more sensitive to changes in nutrient status and excessive primary production (of aquatic plants) can become a management issue.

With the exception of very large, deep lakes with a relatively small littoral margin, few fresh water-bodies are driven by planktonic sources (Wetzel, 1990). Instead, most lakes and wetlands are dependent on primary production occurring in the littoral zone. Many natural lakes are fringed by extensive and highly productive beds of submerged and emergent macrophytes, but much of this production does not appear to enter aquatic food webs, either directly through herbivory or indirectly as detritus (Bunn & Boon, 1993). Epiphytic algae, however, are thought to be a major source of carbon supporting aquatic food webs and macrophytes provide important substrate.

In many impoundments, rapid fluctuations in water levels can prevent the establishment of productive littoral vegetation. Even if the macrophyte beds can withstand periods of exposure, their productive epiphytic algae cannot. As a consequence, the basic ecosystem processes that contribute to the natural functioning of the lake are affected.

Cyanobacterial blooms can dramatically change the functioning of impounded lake systems. It is generally accepted that much of the primary production of cyanobacteria, particularly the toxic varieties, is not grazed by zooplankton or other consumers (Boon *et al.,* 1994). This means that blue-green populations cannot be controlled by 'top-down' grazing pressure and their accumulated biomass leads to deterioration in water quality (both from toxins and decomposing organic matter). Nitrogen fixed by the cyanobacteria can, however, enter aquatic food webs and contribute to the overall nutrient enrichment of the impoundment.

Future of the Brisbane River

There is little doubt that protection and rehabilitation of riparian vegetation along the smaller tributary streams of the Brisbane River will go a long way toward restoring basic ecosystem processes, and improving aquatic habitat and water quality. Given the extent of land clearing and riparian degradation in some parts of the catchment (Arthington *et al.,* 1996b), control of stock access and re-vegetation will be required. Careful design of riparian buffer zones on tributary streams may also help to reduce inputs of inorganic sediment and nutrients from diffuse run-off (Woodfull *et al.,* 1993; Haycock *et al.,* 1997).

In order to identify an appropriate strategy for restoration for the main-stem of the Brisbane River and the lower reaches of its major tributaries, we first need to resolve which of the three current models (RCC, FPC or RPM) best describes ecosystem function. To some degree, better management of tributary streams will help to protect these receiving water bodies by lowering water temperatures and reducing inputs of nutrients and sediment. There are no available data to indicate whether downstream food webs are dependent on catchment-derived organic matter "leaked" from upstream processing (RCC, Vannote *et al.,* 1980). The fact that there is very little evidence of assimilation of riverine carbon into estuarine food webs (e.g. Haines & Montague, 1979; Loneragan *et al.,* 1997) suggests that much of the catchment-derived carbon in larger rivers is of poor quality for aquatic consumers, relative to other sources.

Even if flood-driven processes were important to river ecosystem function prior to European influence, there is little opportunity for restoration given the extent of urbanisation of the lower Brisbane River and the lower reaches of the major tributaries. Of the three models, the Riverine Productivity Model has been proposed as most relevant for large rivers with constricted channels and firm substrates in the photic zone (Thorp & Delong, 1994). To some extent it may also provide an appropriate framework for examining the function of large rivers, such as the Brisbane River, that have been effectively isolated from their lower floodplain by river regulation. Deprived of access to nutrient and energy sources on the floodplain, the river may now be more reliant on local in-stream and riparian sources of primary production.

Loss of riparian vegetation along the lower reaches of the Brisbane River and the lower Lockyer and Bremer means that local inputs of carbon are reduced. High turbidity, from erosion processes upstream, dredging and high tidal energy, limit the potential for significant sources of high quality in-stream production. This is probably quite fortunate given the extremely high levels of nutrients in the lower river, which would otherwise lead to explosive growths of aquatic macrophytes, blue-green algae and other nuisance aquatic plants. These are not readily incorporated into aquatic food webs and contribute to further loss of habitat and degradation of water quality. Removal of large woody debris and rocky outcrops has further compounded the problem by reducing the available stable substrate for important microalgae in the lower reaches of the river.

Restoration of the lower Brisbane River should focus on revegetation of degraded stream banks to provide local inputs of organic debris and shade, and provision of large woody debris in the littoral margins. Any efforts to reduce turbidity, from upstream diffuse sources and dredging, must be accompanied by a significant reduction in nutrient inputs from both diffuse and point sources (e.g. sewage treatment plants). Accordingly, management of the lower river cannot be undertaken in isolation from the upper catchments.

Acknowledgements

Many of the issues presented here are drawn from recent reviews by the author for the Murray Darling Basin Commission (Carbon Scoping Study) and the Brisbane River Management Group (Ecology Foundation Paper), and their financial support is gratefully acknowledged. The work was also supported by the Land and Water Resources R&D Corporation, as part of their commissioned program “Rehabilitation and management of riparian lands” (GRU17). I thank Thorsten Mosisch for assistance in the preparation of the original reports and the figures used in this paper, and two anonymous referees for their constructive comments.

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Flow Modifications and Flow Management Scenarios in the Brisbane River Catchment

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Abstract

The construction of two large impoundments, Wivenhoe Dam and Somerset Dam, water abstraction, and changes in land use have substantially modified the natural flow regimes of the Brisbane River. River flows have decreased overall and become more predictable in a system which is naturally highly variable and unpredictable. Wivenhoe and Somerset Dams have introduced major discontinuities into the once free-flowing river and effectively divide the Brisbane River system into two separate components, an 'upper' component flowing into Wivenhoe Dam, and a 'lower' component flowing into Moreton Bay. This major discontinuity has isolated the lower river from its natural supplies of inorganic and organic materials originating in the upper Brisbane and Stanley catchments, and the tidal river and Moreton Bay are accordingly deprived of materials and freshwater flows. Discharges from the Lockyer, Bremer and Oxley systems have a disproportionate influence on the lower Brisbane River but their ecological value to the river is probably reduced because they drain such degraded catchments. Floodplain systems along the lower reaches have also suffered due to flow regulation by Wivenhoe Dam. The barrier effects of the two dams, the conversion of riverine to lake habitat, the regulation of river flows and the upstream advance of the tidal prism have collectively contributed to significant losses of freshwater habitat. The ecological implications of riverine discontinuities and barrier effects, habitat loss and water quality degradation must be understood throughout the Brisbane River catchment if protection of the riverine ecosystem is to be an underlying goal of river management.

Introduction

In late 1996, the Brisbane River Management Group (BRMG) commissioned several issue-specific Foundation Papers to assist the development of a Management Plan for the River. The Centre for Catchment and In-Stream Research prepared the Ecology Foundation Paper as a two-part document. Stage One (Arthington *et al*., 1996a) presented information about the condition of the main catchments (ecosystem components) making up the Brisbane River system as a whole, and forecast the trends in condition to be expected as pressures on the river basin increase. Stage Two (Arthington *et al*., 1996b) used the factual material provided in Stage One to develop conceptual models of the river system in its pre-European state and as it is today, with particular emphasis on key "ecosystem drivers".

River ecologists agree that two sets of processes drive riverine ecosystems – those associated with the sources, fluxes and sinks of organic carbon and those associated with the river's flow regime (Degens *et al*., 1991; Esser & Kohlmaier, 1991; Poff & Ward, 1990; Sparks, 1992). The present condition of the Brisbane River, like any river ecosystem, is largely a function of these two sets of driving processes, whereas other pressures, such as water pollution, physical disturbances and exotic species act merely to confound the outcome of modifications to these fundamental driving processes (Arthington *et al.,* 1996b). Although these confounding pressures may be more important in some areas than others, they are thought to be far less important than the main drivers of the Brisbane River system – the flow regime and carbon (energy) flux. Fixing the minor pressures is not irrelevant, but fixing the major drivers should be the primary goal of river rehabilitation and management in the future.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 143-152.

This paper is focused on flow regimes in the Brisbane River catchment. It presents an overview of changes made by humans to the River's natural flow regime, taking the system apart catchment by catchment and then stepping back to view the river ecosystem in its entirety. Indirect indicators of change (e.g. number of dams and weirs, length of river habitat lost) have been used rather than quantitative flow data because the analysis of flow regimes is still in progress (DNR, 1996; Water Studies, 1996). The second part of the paper suggests some possible options for the management of flow regimes, with the objective of improving ecosystem condition in the river, the estuary and Moreton Bay. The implications of each option for water management are then discussed in broad terms, followed by a quick summary of how environmental flow objectives might be achieved within existing policy frameworks and with the tools, techniques, and knowledge base available.

Changes to the Flow Regime of the Brisbane River

Natural flow regimes within the Brisbane River system have been modified by a range of human activities and water management systems (Water Studies, 1996; DNR, 1996). These changes are summarised briefly below, with the details for each catchment of the Brisbane River presented in Figures 1-5 (extracted from the Ecology Foundation Paper Stage 2, Arthington *et al.,* 1996b).

Run-off characteristics have been modified particularly in agricultural catchments and urban areas (Water Studies, 1996). The volume of floodwaters in agricultural catchments has increased, resulting in an increase in peak flood discharges and flood levels, particularly for low to moderate floods. Rates of rise and fall of flood peaks have also changed. Urbanisation has had even more pronounced effects on flood discharge, flood peaks and levels and rates of water rise and fall (Water Studies, 1996).

Figure 1. Flow regulation in components of the Brisbane River catchment: Stanley River.

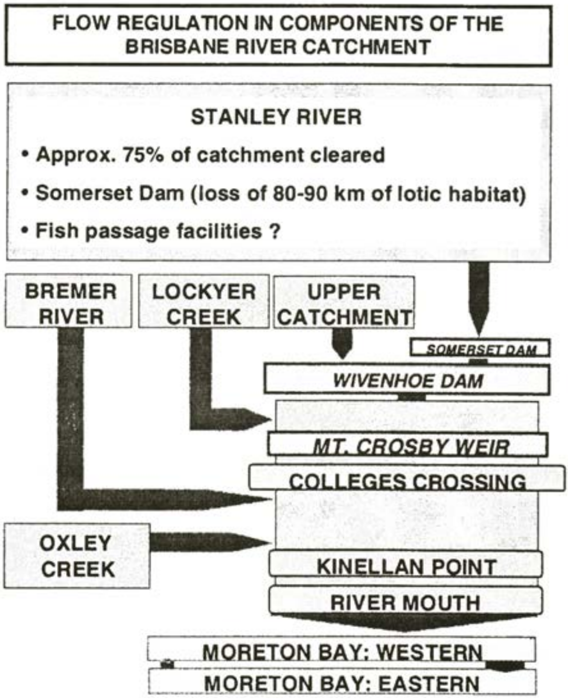


Figure 2. Flow regulation in components of the Brisbane River catchment: Upper Catchment.

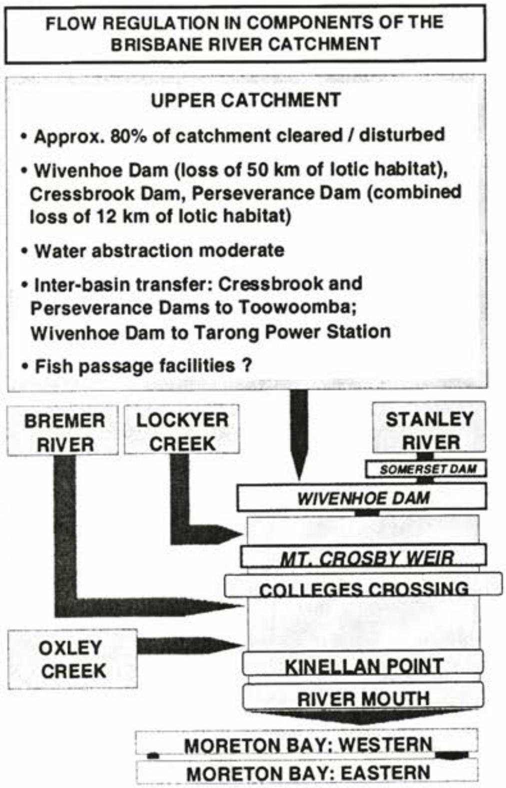


Figure 3. Flow regulation in components of the Brisbane River catchment: Lockyer Creek.

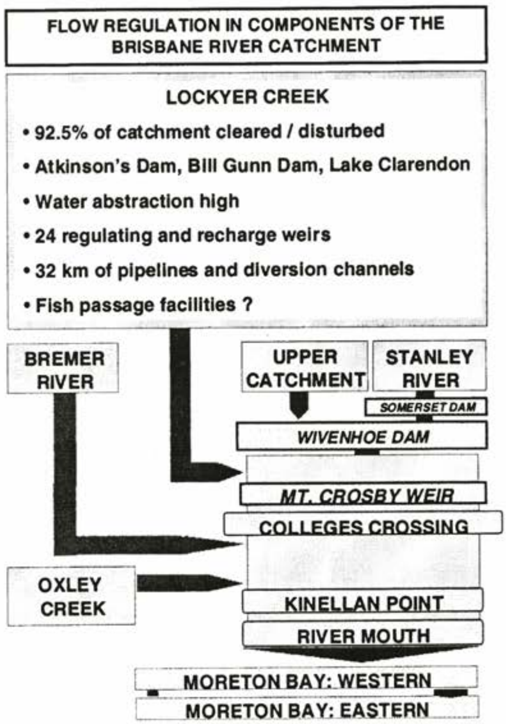


Figure 4. Flow regulation in components of the Brisbane River catchment: Bremer River.

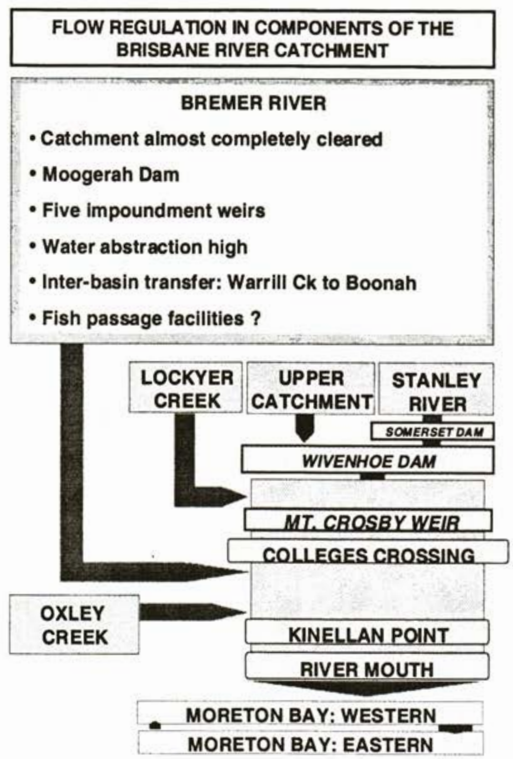
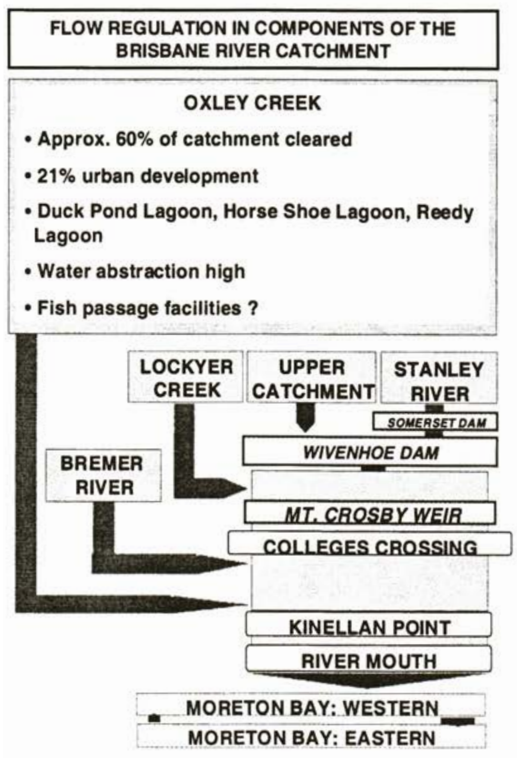


Figure 5. Flow regulation in components of the Brisbane River catchment: Oxley Creek.



The construction of major flood control impoundments, Wivenhoe Dam and Somerset Dam, and several smaller impoundments has further modified natural flow regimes (Water Studies, 1996). The quantities and temporal patterns of water release from these storages produce downstream flow regimes that differ significantly from the former natural regime. River flows have tended to become more predictable in a system which is highly variable and unpredictable in its natural state, as are other river systems in southeastern Queensland (see Pusey *et al.,* 1993). Water abstraction by pumping has also altered the quantity and the pattern of river flows in many part of the Brisbane River system (DNR, 1996). The effects of inter-basin transfers of water to nearby catchments have not been assessed but have the potential to modify the flow regimes of both the donor tributaries and the receiving waterways (see Davies *et al.,* 1992).

In addition to the changes within each major catchment, and inter-basin transfers of water, the construction of Somerset Dam and Wivenhoe Dam has had a major impact on the river as an ecosystem. These dams have introduced major physical discontinuities into the once free-flowing river. In the Stage Two Ecology Foundation Paper (Arthington *et al.,* 1996b), it is suggested that Somerset and Wivenhoe Dams effectively divide the Brisbane River system into two major ecological components which now function largely independently. These components are the 'upper river' (Upper Catchment of the Brisbane River and Stanley River) feeding into Wivenhoe Dam, and the 'lower river' (Bremer River, Lockyer Creek and Oxley Creek) feeding into the main channel of the Brisbane River, its estuary and Moreton Bay (see Figure 1).

The consequences of this major discontinuity in the Brisbane River system are considered to be the following. Firstly, the Brisbane River below Wivenhoe Dam has been isolated from its natural supplies of inorganic and organic materials (including biota) originating in the upper Brisbane and Stanley catchments. Somerset and Wivenhoe Dams have become 'sinks' for these materials, and the tidal river and Moreton Bay are accordingly deprived of inorganic and organic materials, and of freshwater flows which may perform critical geomorphological and ecological functions. For example, there is evidence from the Logan River system that high summer flows can be correlated directly with the productivity of coastal fisheries, allowing for a timelag associated with recruitment (Loneragan & Bunn, 1997).

As a consequence of the capture of many flood flows by Wivenhoe Dam, flows from the Lockyer, Bremer and Oxley Creek systems are thought to have a disproportionate influence on the lower Brisbane River, except when large floods are passed through or released from Wivenhoe Dam. Our assessment of river condition (Arthington *et al.,* 1996a) indicates that flows from Lockyer, Bremer and Oxley Creek drain the most degraded catchments within the Brisbane River system. These flows are modified in terms of their quantity, temporal pattern and water quality, and we suggest that their value to the 'lower' River ecosystem is proportionately reduced. This was amply demonstrated during the May 1996 floods which originated mainly in the Lockyer and urban catchments with major pollution and siltation problems.

The present mode of operation of Wivenhoe Dam has led to a highly modified flow regime in the 'lower' Brisbane River (Water Studies, 1996). A river condition survey by the Queensland Department of Primary Industries (DPI, 1995a; b) indicates that the spatial and temporal heterogeneity and quality of aquatic habitats in the freshwater sections of the 'lower' River have been reduced by flow regulation and channel degradation. The upstream movement of the tidal prism has also contributed to loss of freshwater habitat in the main river channel. Finally, higher than natural flows during many months of the year have effectively 'drowned out' most of the major riffle habitats between Lake Wivenhoe and Mt Crosby Weir (Arthington & S. Brizga, unpublished data). Riffle habitats are significant because of their major contributions to the biological diversity and productivity of streams and rivers (Benke *et al.,* 1984).

Wivenhoe and Somerset Dams, and to some extent Mt Crosby Weir, are significant barriers to upstream migration of riverine biota. There has also been significant loss of freshwater habitat in the 'upper' river due to conversion of riverine habitat to lake habitat (Lakes Wivenhoe and Somerset, and many smaller storages). Fluctuations in water level prevent large impoundments from functioning as natural lakes with highly productive littoral zones (Wetzel, 1990), and many riverine species adapted to flowing water conditions are unable to survive and/or reproduce in this standing water habitat. Fish stocking is one response to the loss of biological diversity and productivity in man-made lakes. Unfortunately such waterbodies tend to provide ideal habitat for introduced species which may out-compete indigenous flora and fauna. An exotic species of Tilapia (fish family Cichlidae) introduced to dams in the Brisbane catchment is causing some concern in this regard (McKay & Johnson, 1990).

In summary, the Brisbane River no longer functions as a free-flowing river linked from its headwater areas to the tidal estuary and Moreton Bay by river flows and longitudinal, flow-dependent processes. It has suffered two major, main-channel discontinuities – Wivenhoe Dam and Somerset Dam. The lower reaches of the River have been disconnected from their alluvial floodplains, pulses of flow in the river are now largely confined to the active channel, and the estuary and Bay are deprived of freshwater flows except during periods of high flow releases from Wivenhoe Dam and flooding in the lower tributaries. The quality of these tributary flows as ecosystem drivers may be relatively low due to water quality problems and changes to the quantity, timing and frequency of river flows.

Scenarios for Management of Flows in the Brisbane River Catchment

This paper has proposed that Somerset Dam and Wivenhoe Dam divide the Brisbane River system into two major components. Flow management must be addressed separately for each component, and the aim of the following section is to outline some possible options for the management of river flows both upstream and downstream of Wivenhoe Dam. We emphasise that these options were developed to inform the Brisbane River Management Group and do not represent any official position with respect to the management of the river.

Flow management in the 'upper' Brisbane River

Scenario 1 – Manage flows as in 1996

Under this scenario, it is assumed that current water management practices will continue (i.e. abstraction, flow regulation by dams and weirs, groundwater pumping, IBTs to other catchments). If these practices continue, catchments in the 'upper' Brisbane River system will have:

- modified flow regimes;
- modified spatial and temporal heterogeneity and quality of aquatic habitats due to flow regulation and channel degradation;
- modified aquatic communities and ecosystem processes due to flow regulation and channel degradation;
- limited (or no) fish passage at all dam and weir sites so that migratory fishes will be unable to gain access to aquatic habitats upstream from barriers; and,

- flows from the Stanley and upper Brisbane River catchment will have a significant influence on the quality of water in Somerset Dam and Wivenhoe Dam and hence on aquatic communities and ecosystem processes in Lake Wivenhoe.

Scenario 2 – Modify water management practices in the 'upper' Brisbane River to meet environmental flow targets

Under this scenario, it is assumed that dams, water abstraction and IBTs will be managed so that environmental flow targets are met to the maximum extent possible given other operational constraints. This scenario implies:

- introduction of more natural distributions of flows with appropriate timing of low flows, small pulses of flow, small floods etc. in all major tributaries of the 'upper' Brisbane River. These more natural flow regimes would be of benefit to the tributaries themselves as well as Lake Wivenhoe; and,
- co-ordinated management of tributary flows, groundwater flows and other contributing flows/losses.

Implications of Scenario 2

If Scenario 2 is to be adopted then the following should obtain:

- no further development of water resources will be possible in some tributaries;
- some water allocation and/or management practices would have to be reviewed and modified to achieve more natural flow regimes in the 'upper' catchment;
- it may be necessary to remove or modify weirs and other minor barriers along streams and reduce groundwater pumping; and,
- the environmental impacts of IBTs should be reviewed both for the donor catchment and the receiving catchment.

Flow management in the 'lower' Brisbane River

Scenario 1 – Operate Wivenhoe Dam as in 1996

Under this scenario, it is assumed that Wivenhoe Dam will be managed as at present for flood mitigation and provision of water to Mt Crosby for urban supplies, plus IBTs to the Burnett and other catchments. If this management regime continues, the 'lower' Brisbane River system will have:

- highly modified flows (for details see Water Studies, 1996) in the freshwater reaches extending from Wivenhoe to the upper limits of tidal influence;
- modified spatial and temporal heterogeneity and quality of aquatic habitats in the 'lower' River due to flow regulation and channel degradation;
- modified aquatic communities and ecosystem processes in the 'lower' River due to flow regulation and channel degradation;
- limited fish passage through Mt Crosby Weir. The fish ladder at Mt Crosby Weir frequently does not function efficiently under the existing flow regime (Simpson & Jackson, 1996);
- migratory fishes unable to gain access to aquatic habitats upstream from Mt Crosby Weir, Wivenhoe Dam and Somerset Dam;

- flows from the Lockyer, Bremer and Oxley Creek systems exerting a disproportionate influence on the lower Brisbane River (except when large floods are passed through Wivenhoe Dam) because these flows are presently modified in terms of their quantity, temporal pattern and water quality, their value to the river ecosystem is believed to be reduced;
- urban creek flows continuing to influence the lower Brisbane River, but this influence is probably conditional upon the level of flow regulation, water pollution etc.;
- the 'lower' river remaining confined to the main channel by flow regulation, depriving the system of natural floodplain flows (except during uncontrolled flooding); and
- freshwater flows to the tidal river and Moreton Bay remaining modified in quantity and possibly not performing their normal ecological functions (e.g. transport of nutrients and organic carbon, flushing of substrates).

Implications of Scenario 1

The Brisbane River downstream from Wivenhoe Dam, the estuary and the Bay will continue to degrade in terms of water quality, habitats and aquatic communities. How far these systems will degrade in the future cannot be stated from the data available. Their present ecological condition is very poorly understood.

Scenario 2 – Modify Wivenhoe Dam operations to meet environmental flow targets

Under this scenario, it is assumed that Wivenhoe Dam will be managed so that environmental flow targets are met to the maximum extent possible given other operational constraints (flood mitigation, provision of water to Mt Crosby for urban supplies, plus IBTs to other catchments). Environmental flow targets might include:

- introduction of greater variability into flow regime downstream from Wivenhoe Dam (i.e. a more natural distribution of flows with appropriate timing of low flows, small pulses of flow, release of some flood waters downstream etc.);
- larger flows over Mt Crosby Weir. These would serve several functions (improved fish passage, water quality and habitat improvements downstream, larger flows to estuary and bay, maintenance of ecological linkages between high flows and estuarine/coastal productivity);
- under this scenario, the following matters will still present problems:
- flows from the Lockyer, Bremer and Oxley Creek systems will continue to have a disproportionate influence on the lower Brisbane River (except when large floods are passed through Wivenhoe Dam); and
- urban creek flows will continue to have some influence on the lower Brisbane River but this influence is probably conditional upon the level of flow regulation, water pollution etc.

Scenario 3 – Manage all sources of flow to the 'lower' river to meet environmental flow targets

This scenario implies that Wivenhoe Dam will be managed as in Scenario 2, and also that flows from the Lockyer, Bremer and Oxley Creek systems, and in urban streams, would be managed to maximise their beneficial contributions to the Brisbane River, contingent upon other constraints.

This scenario implies:

- introduction of more natural distributions of flows with appropriate timing of low flows, small pulses of flow, small floods etc. in all major tributaries into the 'lower' Brisbane River. These more natural flow regimes would be of benefit to the tributaries themselves as well as the Brisbane River; and,
- co-ordinated management of Wivenhoe flows, tributary flows, groundwater flows and other contributing flows/losses.

Implications of Scenario 3

If Scenario 3 is adopted, the following would obtain:

- no further development of water resources will be possible in most tributaries;
- some water allocations and/or management practices would have to be reviewed and modified to achieve more natural flow regimes in the Lockyer, Bremer and Oxley systems, and in urban creeks; and,
- it may be necessary to remove or modify weirs and other minor barriers along streams and cease groundwater interference.

Concluding Comments

The ecological implications of riverine discontinuities and barrier effects, habitat loss and water quality degradation throughout the Brisbane River catchment must be assessed if maintenance of aquatic ecosystems is to be an underlying goal of river management. This paper has summarised changes to the flow regimes of Brisbane River tributaries and the main river channel using indirect indicators of change rather than quantitative flow data and ecological responses because there is limited biological data on the condition of waterways in relation to these modified flow regimes.

Until recently, little if any consideration has been given to the environmental flow requirements of the Brisbane River and the ecological consequences of modifications to stream and river flow regimes . It is only at the initiative of the South East Queensland Water Board (SEQWB) that environmental flows to the 'lower river' are now being considered in a major study (Arthington, 1996). This study, although timely and comprehensive, represents only part of the action required to manage flows sustainably in the Brisbane River catchment.

There can be only one conclusion from the above assessment. A catchment-wide environmental flow study must be conducted within a suitable framework such as WAMP (Water Allocation and Management Planning) developed by the Queensland Department of Natural Resources. This should be designed to extend and enhance the environmental flow study commissioned by the SEQWB. The tools and techniques required for environmental flow assessment are well-developed in Queensland, the hydrological data base is very good and IQQM and flood models are becoming available for the catchment. An environmental flow study would appear to be fundamental to the achievement of key objectives in the Brisbane River Management Plan.

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Moreton Bay Catchment: Water Quality of Catchment Rivers and Water Storage

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Abstract

The Queensland Parliament passed the Clean Waters Act in 1971 and this coupled with the Pollution of Waters by Oil Act passed in 1973 resulted in a significant improvement in water quality within the State particularly in the Brisbane urban area, since the 1960s. However, the population of the Moreton Bay catchment is in excess of one million and growing at a relatively rapid rate. This population produces steadily increasing volumes of wastewaters from domestic and industrial sources as well as contaminants associated with runoff water. A major water quality management problem in the Moreton Bay catchment is the management of blooms of blue-green algae (Cyanobacteria) in the various large water storages in this area. These blooms already pose a major management problem and could increase in frequency and in intensity. The dissolved oxygen over a major section of the urbanised estuary is often severely depleted. The factors causing this require evaluation and a program instituted to allow improvements. In addition, excessive levels of nutrients occur in the estuary, which does not exhibit the adverse effects due to excess growth of plants, due to turbidity, inhibiting plant growth. A nutrient management program is required, coupled with a turbidity reduction program, to enhance the water quality in the estuary and protect the quality of waters in Moreton Bay.

Streams in the catchment, such as the Lockyer and Bremer Rivers, exhibit adverse effects due to stormwater runoff containing sediments, nutrients and possibly pesticides from rural and agricultural activities. Catchment management programs developed in association with community groups, are needed to manage these problems. A range of toxicants occur in the river estuary in measurable concentrations including persistent organochlorine pesticides, petroleum hydrocarbons, and polycyclic aromatic hydrocarbons. Several of these categories of toxicants, particularly petroleum hydrocarbons and polycyclic aromatic hydrocarbons are associated with urbanisation and possibly are increasing. An evaluation of the occurrence of toxicants in the system is needed which will allow an interpretation of the significance of the various substances involved and the institution of remedial measures.

Introduction

The population of the Moreton Bay catchment is in excess of one million and growing at a relatively rapid rate. This population produces wastewaters from domestic and industrial sources as well as contaminants associated with runoff water. It can be expected that as the population increases the volume of wastewater will correspondingly increase. Water quality has been of interest in the past but not of major concern until about the last 30 years. In the 1960s, as a result of previous population increases with little attention to the control of wastewater, the quality of water in the Brisbane urban areas was in some cases very low. In response to this, the Queensland Parliament passed the Clean Waters Act in 1971 and this coupled with the Pollution of Waters by Oil Act passed in 1973 resulted in a significant improvement in water quality within the State particularly in the Brisbane urban area.

In recent years, there has been concern regarding water quality in the Moreton Bay catchment stimulated by continued population growth. In addition the occurrence of 'blooms' of blue green-

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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algae (Cyanobacteria) has resulted in increased attention to water quality in the Moreton Bay catchment. Anecdotal evidence points to a deterioration of water quality in the receiving water body for the Moreton Bay catchment which is Moreton Bay itself. All of these factors have indicated an increased need for water quality management, however, water quality management is expensive and therefore there must be action on the basis of the best information available. The most effective control and management program to ensure that scarce resources are spent to give the best result in water quality terms should be put in place.

There have been a number of conferences and symposia which have attempted to draw together the existing information on water quality and other aspects of aquatic conservation in Moreton Bay and its catchment and these have proved useful in placing water quality and aquatic conservation into an enlightened perspective (Davie *et al*., 1990; Crimp, 1992).

Recent efforts in the development of water quality management strategies (WBM-SKM, 1995) have highlighted the need for a clear knowledge of which values are perceived by the community to apply to the rivers, streams and lakes of the Moreton Bay catchment and Moreton Bay itself. We have knowledge in a general sense of the uses of these water bodies but need to know more accurately the values the community places on them so as to guide and direct management into the most appropriate areas. The National Water Quality Strategy (AWQ Guidelines, 1992) is a national attempt to guide State and local governments and regions toward the development of water quality management programs on a consistent and logical basis. The Strategy involves the community in the setting of values to be used to guide the Strategy itself. Many guidance documents have been prepared by the Australian and New Zealand Environment and Conservation Council (ANZECC) to assist in the management of water bodies throughout Australia (AWQ Guidelines, 1992).

This paper provides an overview of water quality of streams and lakes in the Moreton Bay catchment. It draws on a wide range of information available including scientific papers and reports, consultants' reports, government reports and others.

The Nature of Water Quality

The quality of water in a stream or lake can be related to the uses made of that water. The uses of water in the Moreton Bay catchment fall into the following categories:

- domestic raw water for domestic water supply;
- water for industry, including rural industries;
- recreation and aesthetics; and
- conservation and protection of aquatic ecosystems including natural fisheries.

There are physical, chemical and biological characteristics of the water which have an influence on the uses that can be made of that water and certain indicators defining these characteristics against which quality can be judged. Table 1 includes some water quality indicators which are important in water bodies in the Moreton Bay catchment and in addition, Table 2 includes some applications of the indicators used in the evaluation of the suitability of water for different uses. Table 2 also includes some guideline values from the *Australian Water Quality Guidelines for Fresh and Marine Waters* (AWQ Guidelines) (1992) which form part of the National Water Quality Management Strategy. It is important to note that the nutrients, nitrogen and phosphorus are specified only as a general range of concentrations which require further specific evaluation of the area under consideration to be made more precise. This applies particularly to water bodies in the Moreton Bay catchment and as yet no specific nutrient levels for this area are available. The values the community wishes to apply to specific water bodies in the catchment also influences which value is applied in different areas.

Table 1. Some important water quality indicators in the Moreton Bay catchment.

Physical parameters	• Temperature • Light penetration – turbidity • Conductivity
Chemical parameters	• Dissolved oxygen concentration • Suspended solids content • Biochemical oxygen demand (BOD) • Nutrient concentrations (principally N & P) • Dissolved salts • Chlorophyll *a* concentration • Toxicant concentrations e.g. pesticides, petroleum etc. • Occurrence of blue-green algal toxins
Microbiological and biological characteristics	• Faecal coliform counts • Blue-green algal counts • Ecological and biological characteristics

Table 2. Some applications of water quality indicators with different use and corresponding guideline values*.

Raw water for domestic use	• Faecal coliforms (none in 2 consecutive 100 mL samples) • Dissolved salts (many levels) • Blue-green algal counts (usually <1 000 cells/mL) • Occurrence of algal toxins (usually <1 μg/L microcystin) • Toxicants (many levels)
Industrial use	• Dissolved salts (many values related to use) • Faecal coliforms (many values related to use)
Recreation and aesthetics	• Dissolved oxygen concentration (not specified) • Faecal coliforms (median value 150 organisms/100 mL)
Conservation and ecosystem protection	• Dissolved oxygen concentration (> 6 mg/L) • Toxicants (many values) • Suspended solid content (<10% decline from mean) • N & P concentration (site specific but generally < 10-100 μg/L. Total P and 100-750 μg/L Total N)

* *Australian Water Quality Guidelines for Fresh and Marine Waters*, 1992.

Water quality parameters generally relate to the contamination of water which makes it unsuitable for particular uses. For example, Miller & Connell (1990) have identified the major contaminants in the Brisbane River and these can be considered to apply to the other streams in the Moreton Bay catchment as well as:

1. **Organic substances**, generally derived from sewerage and food processing wastes which can create an oxygen demand in a water body generally related to Biological Oxygen demand (BOD);
2. **Plant nutrients**, such as N & P forms which cause the excess growth of plants. (Of particular interest in this area are the 'blooms' of blue-green algae which create problems for domestic water supply);
3. **Suspended matter**, such as silt and other particulate material;
4. **Toxic substances** such as petroleum oils, pesticides, solvents, chlorine residues, detergents, toxic metals, disinfectants, plasticisers and polychlorinated biphenyls;

5. **Pathogenic microorganisms**, such as faecal bacteria, viruses and parasites; and
6. **Thermal discharges** which are another potential source of pollution, however while those derived from power generation are believed to be relatively unimportant in the Moreton Bay catchment, the potential ecological effects of discharge water at relatively low temperatures from dams are unknown.

The wastewaters giving rise to these contaminants in a water body are derived principally from two broad sources: point and non-point discharges. Point discharges derive from sewage plants, and industry, for example, whereas non-point sources include storm water derived from rural, urban and other catchments. In the Moreton Bay catchment sewage discharges, mainly treated to a secondary standard, contribute significantly to the nutrient and BOD load of water bodies as well as in the toxicant load. Non-point discharges to water bodies in the catchment can contribute to increased nutrients, BOD, as well as toxicants such as agricultural pesticides in runoff from rural areas and toxic metals from road runoff from urban areas. Accidents occur relatively infrequently but can involve spillage of oil or toxic substances which may be have a major detrimental impact over a short period of time. Although the time period involved in such incidents is relatively short their environmental impact may extend over a considerable period of time.

The water quality in streams and lakes in the Moreton Bay catchment is influenced by climatic conditions (Miller & Connell, 1990). In this group of influences there are two major factors: annual and seasonal fluctuations in flow.

Seasonal flow variations, with high rainfall and water flow in summer and low rainfall and water flow in the winter, lead to different characteristics of water quality from season to season (Miller & Connell, 1990). This variation needs to be taken into account in evaluating water quality characteristics. Complicating these influences of flow regulation from the water storages leads to alteration of the natural flow regime with consequent impacts on water quality.

Large annual variations in rainfall, characteristic of the Australian environment, lead to variations in water quality from year to year which, in turn, relate to water flow and the availability of water for dilution in water bodies. These large fluctuations in water quality characteristics can make difficult the evaluation of water quality and the evaluation of the success of different water quality management strategies, (based on the physical, chemical and biological characteristics as well as management situations). Water bodies of the Moreton Bay catchment can be divided into several groups which are influenced by major discharges. These are:

- **Water storages and related catchments** such as Wivenhoe Dam, Somerset Dam and North Pine Dam;
- **Water bodies subject to rural and agricultural discharges** such as the Lockyer and Bremer Rivers; and
- **Tidal zones subject to urbanisation** such as zones of the Brisbane and Bremer Rivers subject to tidal influence.

Of course these categories are not exclusively influenced by the factors nominated since there can be a combination of discharges in some situations, however, the categories nominated are the major discharges and they help to identify influences and factors causing contamination and deterioration in water quality. These various categories are discussed below.

Water Storages and Related Catchments

Within the Moreton Bay catchment the three major water storages are Wivenhoe Dam, Somerset Dam and North Pine Dam. Wivenhoe Dam is the largest of these storage bodies with a catchment of 7 020 km^2 and a water supply storage of 1.15 x 10^6 ML; Somerset Dam has a catchment of 1 330 km^2 and a storage of 0.369 x 10^6 ML and North Pine Dam has a catchment of 347 km^2 and a storage capacity of 0.202 x 10^6 ML.

There are several small rural townships, such as Esk, in the Wivenhoe catchment with other townships distributed below the dam itself. Unsewered wastewater discharge in the Wivenhoe catchment is considered to be minor in terms of its impact on the Dam but may be significant in local areas, particularly in terms of faecal Coliform contamination.

The major land uses in the upper Brisbane River catchment have been estimated as grazing (75%), cropping (4%) and forests (16%). WBM-SKM (1995) have estimated the pollutant loads entering Wivenhoe Dam (Table 3). It can be seen that rural and agricultural activities make a significant contribution to the total load. The size of the loads is primarily influenced by factors such as grazing practices, forestry on steep slopes, horticultural practices and livestock industries (feedlots and piggeries). Total nitrogen for the streams ranged from 0.1-0.4 mg/L (Table 4) which is somewhat similar to Moreton Bay but much less than the Brisbane River estuary which typically ranges up to about 4 mg/L (Moss & Mortimer, 1996) reported for 1996 and up to about 12.8 mg/L in 1985-1986 (Miller & Connell, 1990). All of the observed values for the input streams are below the upper limit specified in AWQ Guidelines (1992), although they generally exceed the lower limits. In general the total phosphorus in the streams was below AWQ Guidelines (1992) with the exception of Cressbrook Creek. The dissolved oxygen in these streams is within the guidelines.

Table 3. Estimates of pollutant loads in stormwater runoff from catchment of Brisbane River above Wivenhoe Dam*.

Land use	Total sediments (Tonnes/yr)	Total nitrogen	Total phosphorus
Forest	8 900	161	18
Rural	89 600	448	90
Agriculture	11 200	224	34
Total	**109 700**	**833**	**142**

* WBM-SKM, 1995

Table 4. Some water quality characteristics of water in upper Brisbane River Catchments and Wivenhoe Dam (1994-1995)*.

Characteristics	Brisbane River	Cressbrook Creek	Murray's River	Wivenhoe Dam	AWQ Guidelines** (protection of ecosystems)
Dissolved oxygen (mg/L)	6.4 - 10.8	5.6	8.5 - 9.8	–	> 6
Total N (mg/L)	0.1 - 0.5	0.2	0.1 - 0.4	0.1 - 0.91	0.1 - 0.75 (stream) 0.1 - 0.5 (reservoir)
Total P (mg/L)	0.01 - 0.052	0.22	0.024 - 0.043	<0.01 - 0.08	0.01 - 0.1 (stream) 0.005 - 0.05 (reservoir)

* GHD, 1996; ** AWQ Guidelines, 1992.

Reservoirs generally exhibit lower concentrations of nutrients than AWQ Guidelines due to different physical conditions in these water bodies (Table 4). The Wivenhoe Dam had a tendency to exceed or be towards the upper limits of the Guidelines for total N and total P. However, there is a need for caution in interpreting these results since stratification and other seasonal factors should be taken into account. There is also a requirement for knowledge of site specific nutrient characteristics for different water bodies, without which only general comparisons can be made.

Elevated nutrient concentrations often cause excessive growth of plants. Of particular concern in the Moreton Bay catchment water storages is the excess growth of blue-green algae (Cyanobacteria). Cell counts for blue-green algae in 1995-1996 reflect a serious water quality problem (Table 5). Counts greater than 10 000 cells/mL are generally considered to reflect a 'bloom' situation with a requirement for concerted management action to protect the domestic water supply. All of the major storages had cell counts which would reflect bloom situations in 1995-1996. In addition, the bloom organism during this period was *Cylinalrospermopsis* which is a relatively uncommon species. This species produces a toxin, Cylindrospermopsin, the toxicology of which is little known. Urgent work is required to develop a knowledge of the toxicology of Cylindrospermopsin to allow the setting of appropriate water quality standards and the development of methods for treatment of water to remove this toxin.

The management of 'blooms' is a major water quality problem and is made difficult by the present lack of means whereby the occurrence of blooms can be prevented or their effects ameliorated. Some measures that could be taken, which may be beneficial, include the more intense management of the catchment to reduce discharges of nutrients and sediment, and the control of phosphorus in detergent.

Table 5. Counts of cells of blue-green algae in water storages (cells/mL) in 1995-1996.

Water storage	*Cylinderospermopsis raciborskii*	*Microcystis aeruginosa*
Wivenhoe Dam	0 - 56 700	0 - 1 000
Somerset Dam	0 - 121 000	0 - 1 500
North Pine Dam	0 - 100 000	0 - 110 000

Water Bodies Subject to Rural and Agricultural Discharges

Almost all of the water bodies in the Moreton Bay catchment receive some discharges from agricultural and rural activities. Generally, rural industries such as intense agriculture, pasture improvement and grazing produce water discharges containing contaminants affecting local streams during storm events. The main types of contaminants are:

- Soil/sediments, derived from erosion;
- Nutrients (N and P), derived from natural sources and the application of fertilisers; and
- Toxicants, derived from the use of pesticides.

A considerable proportion of the nutrients and toxicants bound to soil and sediment particles is moved from agricultural areas to streams in this form. The relationship between pollutant loads and catchment activities is illustrated by the data in Table 6.

The importance of these discharges, in terms of the impact on local streams, can be readily observed. In many streams throughout the Moreton Bay catchment there is evidence of

Table 6. Estimated pollutant loads in stormwater runoff from Lockyer Creek catchment.

	Area* (ha)	Total sediments	Total nitrogen	Total phosphorus
Crops and improved pastures	28,506	14,253	285.00	42.76
Settlement	151	151*	1.50	0.23
Barren	146	29	0.15	0.03
Water	385	72	0.40	0.07
Plantation forest	82	8	0.15	0.02
Grasslands	116 035	23 207	116.00	23.21
Woodland	31 211	3 121	56.00	6.24
Open forest	99 405	9 940	179.00	19.88
Closed forest	21 449	2 145	39.00	4.29
Total	**297 370**	**52 926**	**677.00**	**97.00**

increased siltation and turbidity. In addition, the beds of silt resulting from such discharges provide a substrate for weed growth but may damage the aquatic ecosystem and establish a reservoir for nutrients and toxicants from which the overlying water may become contaminated.

Jones (1993) has carried out an investigation of Laidley Creek which is probably indicative of conditions which exist in many other streams in agricultural and rural areas. Jones (1993) concluded that Laidley Creek has high nutrient levels probably due to the extensive erosion in the catchment. The total N and P levels in the water exceed the recommended levels for the protection of aquatic ecosystems as defined in the AWQ Guidelines (1992). Laidley Creek also receives particulate organic matter in the form of leaf litter, animal manure and crop residue. In general terms, the known and likely water quality impact on agricultural and rural streams can be summarised as:

1. increased siltation of water ways and subsequent growth of weeds and an increase in the level of nutrients;
2. pressure on local receiving waters due to sewage effluent discharge and sullage waters from rural townships and subdivisions; and
3. exports of highly turbid waters containing high loads of sediment, soluble nutrients, nutrients sorbed to sediment and possibly pesticides into the Brisbane River and other local streams.

Management of these problems will only be effectively achieved through a long-term program which could include some of the following factors:

- increased use of integrated catchment management (ICM) practices with the associated involvement of community groups;
- monitoring of land use practices for feedback into ICM plans;
- the development of water quality performance targets related to the specific conditions in each catchment; and
- monitoring of water quality and other characteristics of the aquatic system for assessment of the effectiveness of water quality performance targets.

Tidal Zones Subject to Urbanisation

The most important tidal zone influenced by urbanisation in the Moreton Bay catchment is the Brisbane River estuary which extends from the river mouth to Colleges Crossing and the 22 kms of the Bremer River upstream from its junction with the Brisbane River. Many tributaries enter this complex tidal estuary including Bulimba, Oxley and Breakfast Creeks and Kedron Brook. These drain urban, industrial and semi-rural catchments.

Wastewaters originating from domestic, trade and industrial activities in the catchment discharge into the Brisbane and Bremer River estuaries. The wastewater treatment plants in these urban areas, their capacity in terms of the resident population and the industrial equivalent population for industrial discharges are indicated in Table 7. Stormwater runoff contains contaminants derived from accumulated street litter, motor vehicle residues deposited on roadways and residues of fertilisers and pesticides used on urban properties. A comparison of non-point and point source loads of sediments, total nitrogen and total phosphorus is shown in Table 8. This indicates that total nitrogen and total phosphorus are derived in significant proportions from both point and non-point sources in the tidal Brisbane River and the Bremer River whereas sediments are principally derived from non-point sources.

The effects of these discharges on the water quality characteristics of the Brisbane River estuary are indicated by the data in Figure 1. It is interesting to note that the dissolved oxygen over a significant part of the river is below 75% saturation (or less than 6 mg/L) which is the minimal level in the AWQ Guidelines (1992). This is a significant detrimental effect on water quality which needs to be addressed. An evaluation of the impact of higher standards for discharge at local plants and other factors, such as water flow, is desirable to indicate levels which would allow a more acceptable saturation level of dissolved oxygen to be maintained in the river water. Nutrient levels are significantly elevated in much of the estuary (Figure 1) probably due to the discharge from various sewage treatment plants. Acceptable concentrations of principal nutrients in streams range from 0.1 to 0.75 mg/L for total N and 0.01 to 0.1 mg/L for total P (AWQ Guidelines, 1992). However, the AWQ Guidelines (1992) recommend the development of site-specific guidelines. In the Brisbane River estuary total N reaches a concentration of 4 mg/L and phosphorus 1 mg/L (Figure 1). These levels of nutrients might be expected to result in excessive plant growth reflected in high chlorophyll *a* levels in the estuary however the chlorophyll *a* levels are in fact quite low. This is due to high turbidity of these waters limiting plant growth, a relationship reflected by the low chlorophyll *a* levels (Figure 1).

Various classes of toxic substances (e.g. persistent pesticides, metals and petroleum hydrocarbons) have been shown to occur in the Brisbane River system. These substances originate from sewage and industrial discharges as well as from runoff from urban, industrial, mining and agricultural activities. Their occurrence and behaviour in the Brisbane River system have been reviewed by Connell & Bycroft (1990). In aquatic systems, many toxicants become preferentially deposited in the sediments rather than the water column although it is common practice to measure the concentrations in the water column. Heavy metals (and arsenic) in the Brisbane River estuary have been measured by the Queensland Department of Environment (Table 9). Sediment levels are available but are not included here since are no guidelines for acceptable concentrations are available. In most cases the values tend to show a maximum around about Brisbane City centre and decline in concentration upstream and downstream. The median levels of zinc, lead and chromium are detectable but are well below their respective AWQ Guidelines (1992). On the other hand the median levels of copper at 6 mg/L exceed the AWQ Guidelines.

Table 7. Capacity of sewage treatment plants in Brisbane and Ipswich.

Plant	Capacity (Total equivalent population served)
Luggage Point	530 000
Oxley Creek	24 000
Fairfield	10 000
Inala	16 000
Gibson Island	149 000
Wynnum	32 000
Bracken Ridge	8 900
Sandgate	56,000
Nudgee Beach	280
Wacol	18 100
Bulimba	45 000
Tivoli	62 000
Rosewood	1 660

Table 8. Sources of Non-Point (runoff) and point (effluent) load in Moreton Bay Catchment*.

Receiving water	Sediment			Total nitrogen			Total phosphorus		
	Total (t/yr)	PS (%)	NPS (%)	Total (t/yr)	PS (%)	NPS (%)	Total (t/yr)	PS (%)	NPS (%)
Caboolture River	23,842	0.3	99.7	298.0	39.7	60.3	64.0	60.0	40.0
Burpengary Creek	5,602	0.0	100.0	360.0	0.0	100.0	5.0	0.0	100.0
Boundary Creek	12,954	0.0	100.0	71.0	0.0	100.0	10.0	0.0	100.0
North & South Pine Rivers	61,898	0.2	99.8	572.0	33.2	66.8	118.0	53.9	46.1
Cabbage Tree Creek	7,773	0.0	100.0	39.5	0.0	100.0	5.8	0.0	100.0
Brisbane River (tidal)	118,042	0.5	99.5	1,150.0	59.4	40.6	357.8	75.2	24.8
Brisbane River (fresh)	15,230	0.0	100.0	171.0	1.6	98.4	23.8	4.6	95.4
Bremer River (tidal)	18,999	0.7	99.3	280.4	63.3	36.7	94.3	84.7	15.3

The levels of persistent organochlorine insecticides in the Brisbane River estuary have been reviewed by Connell & Bycroft (1990). During the 1970s and 1980s monitoring programs initiated by the Water Quality Council of Qld and other investigations demonstrated widespread contamination of the estuary and also the tidal Bremer River with the common organochlorine insecticides, DDT and dieldrin. Much of the dieldrin contamination in the upper reaches and the Bremer River could be attributed to woollen mills which have now closed. Moss & Mortimer (1996) from the Queensland Department of Environment reported on organochlorine insecticide residues in mud-crabs collected in the Brisbane River estuary during 1995. Detectable levels of DDE, dieldrin and heptachlor epoxide were found in many samples.

The total residues were found to range up to about 1.5 mg/kg wet weight which is well below the National Health and Medical Research Council maximum residue limits for compounds in edible fish tissue. The pattern of urban and agricultural pesticide use in the Moreton Bay catchment has changed substantially since the 1970s and 1980s. Persistent organochlorine

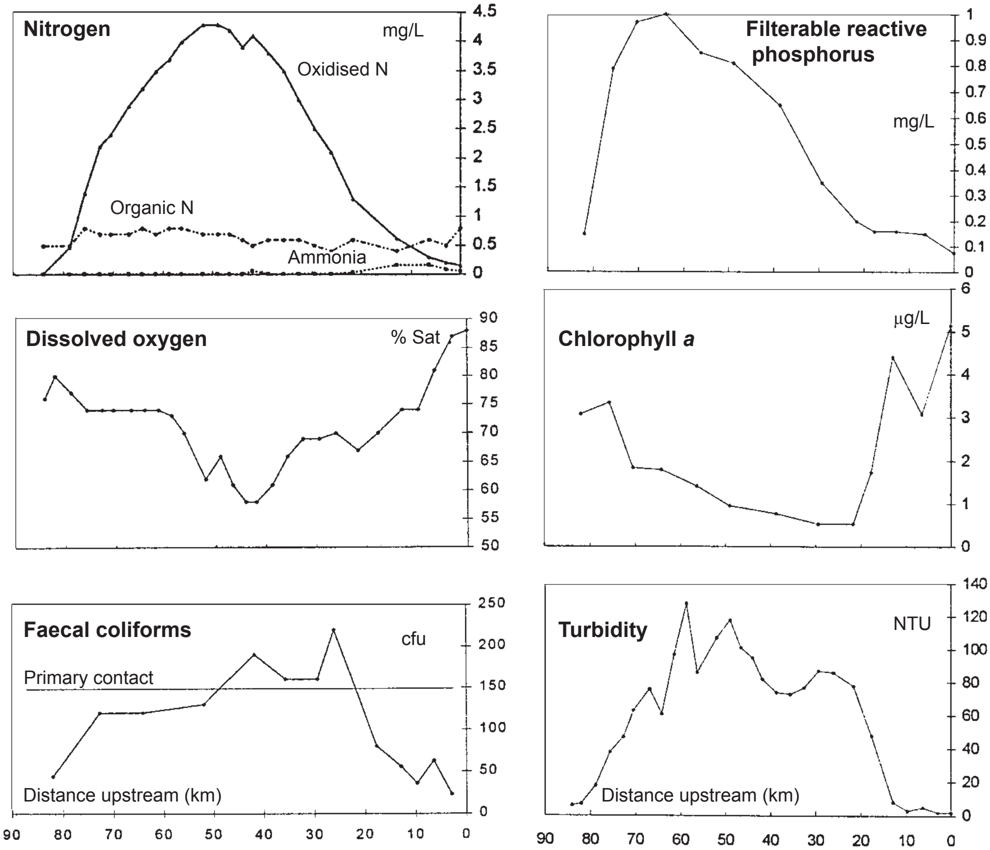

Figure 1. Typical water quality characteristics of Brisbane River Estuary (from Moss & Mortimer, 1996).

insecticides have been phased out and more recently removed from termite control in buildings. This may explain the decline in concentration of these substances, however, a much more thorough evaluation of the occurrence of these substances is needed.

A range of petroleum hydrocarbons have been reported in waters, sediments and biota collected in the estuary since the 1970s. These substances include kerosene-like hydrocarbons, diesel and polyaromatic hydrocarbons (PAHS) as reviewed by Connell and Bycroft (1990). These substances can be attributed to oil spill incidents as well as sewage discharges and storm water runoff containing petroleum residues from roadways and so on. The current information on the occurrence of petroleum hydrocarbons in sediments from the Brisbane River indicates that a range of concentrations from about 24 mg/kg up to 161 mg/kg is present in the sediments in the Brisbane River estuary. Since there are no standards for acceptable concentrations of these substances an evaluation of potential impact is difficult. Also in this case, the highest sediment concentrations were observed in the Brisbane City reaches of the estuary in accord with the origin of these substances from urban activities, particularly motor vehicles. Particles and liquid hydrocarbons fall onto the road surface and are flushed into waterways during storm events where they attach to particles in the water mass and bottom sediments.

Polyaromatic hydrocarbons (PAHs) are mixtures of substances formed from the combustion of fuels in motor vehicles, or present in refined petroleum oils, and can occur in sewage discharges

Table 9. Heavy metal concentrations in estuarine waters in the Brisbane River (3-2-88 to 9-5-90)*.

Locations (km from mouth)	No. of samples	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
Waters				(µg/L)**				
2.90	9	<1	<1	1	<2	<0.5	<5	<10
9.70	9	<1	<1	<1	<2	<0.5	<5	<10
38.70	8	<1	2	6	4	<0.5	<5	14
46.60	8	<1	3	6	7	<0.5	<5	20
70.50	8	<1	<1	3	<2	<0.5	<5	<10

* Source: Moss & Mortimer, 1996;
** 50th percentile values.

and stormwater runoff as well as originating from spillage of petroleum products in the River. They accumulate in bottom sediments and biota, and levels observed by Kayal & Connell (1995) are shown in Table 10. The highest total PAH concentration (16.1 mg/kg dry weight) which occurred in sediments was in the estuary. Total PAH concentrations in sediments range from 3.94-16.1 mg/kg (dry weight) and the carcinogen, benzo(a)pyrene (BaP), comprised 4-7% of this total. Soluble water concentrations of PAHs measured by these authors varied with means of 0.103-0.131 mg/L. This is somewhat less than the AWQ Guidelines (1992) which are set at 3 mg/L for protection of natural ecosystems. However these substances are well known carcinogens and the level in the protection of human health from contaminants in edible food fish is 0.3 mg/L in the ambient water. The mean concentrations are approximately half the maximum levels, and have the potential to increase with increased use of motor vehicles. Thus there is a clear need to ensure that these levels do not rise to or exceed the guideline value.

Conclusions

One of the major water quality management problems in the Moreton Bay catchment is the management of blooms of blue-green algae in the various large water storages in this area. These blooms already pose a major management problem and without remedial action are likely to increase in frequency and in intensity. In this area there is a need to evaluate measures which could be expected to result in an improvement in the situation in the long term. Such measures could include integrated catchment management, control of phosphorus in detergents and more stringent controls of land use in the catchments. In addition, there are a range of short term problems to be addressed including evaluation of the toxicology of the toxin, Cylindrospermopsin. This is needed to allow the establishment of guidelines for its occurrence in drinking water to protect public health as well as occurrence in natural waters to protect the natural ecosystems. Streams in rural and agricultural areas throughout the catchment have visible signs of deterioration in terms of sediment loads and damage to benthic ecosystems. The remediation of these problems will require attention to a set of measures similar to those outlined above for the management of blue-green algal bloom problems.

Much of the estuarine portion of the Brisbane River exhibits a significant decline in dissolved oxygen. To attain good quality water, this fall would need to be corrected through measures such as improved discharge standards for sewage treatment plants in the urban estuary. In addition, elevated levels of nutrients occur in the estuary, however, due to turbidity, the estuary does not suffer the adverse effects of excess plant growth. The impact of these nutrients on Moreton Bay when river waters discharge into it needs to be considered.

Table 10. Polyaromatic hydrocarbons in estuarine biota from the lower Brisbane River (1987/88)*.

Species	Total concentrations (ng/g wet weight)**				
	Section 1***	Section 2	Section 3	Section 4	Section 5
Boney bream *(Nematolosa come)*	64.4	116.5	96.8	177.2	116.1
Blue catfish *(Arius graffii)*	42.8	61.2	68.2	53.7	156.1
Sea mullet *(Mugil cephalus)*	93.4	131.9	91.3	83.8	195.0
Mud crab *(Scylla serrata)*	55.5	74.8	123.1	83.8	96.4

* Source: Adapted from Kayal & Connell (1995);
** Muscle tissues of fish and soft tissues of crabs;
*** Sections of river from mouth to 40 km upstream.

A range of toxicants occur in the river in measurable concentrations including persistent organochlorine pesticides, petroleum hydrocarbons and polycyclic aromatic hydrocarbons. Many of these appear to be below the Australian Water Quality Guideline values and in the case of the organochlorine pesticides may be declining due to the cessation in usage. More information is needed to establish this latter point. However, several of the other categories of the toxicants, particularly petroleum hydrocarbons and polycyclic aromatic hydrocarbons are potentially increasing. According to AWQ Guideline values, the polyaromatic hydrocarbons are now at levels where protection of consumers of fish should be considered. Attention must be directed towards a clearer evaluation of toxic substances and their control to ensure the protection of public health and the health of the urban ecosystem and the adjacent Moreton Bay.

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Pimpama River, an Example of a Catchment Draining into Moreton Bay

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Abstract

The Pimpama River catchment although of limited size possesses many of the features that typify coastal drainage systems of southeast Queensland. The river originates in the Darlington Range at an elevation of 300 m and flows 30 km to the extreme southern part of Moreton Bay. The steep headwaters are incised into the metamorphic Late Paleozoic Neranleigh-Fernvale Beds which form the basement of the entire catchment. The gradient decreases rapidly to the middle section with a widened valley, from which the river crosses the flat coastal plain. On the plain the river meanders through various sedimentary deposits and the channel is relatively shallow with steep sides. Wetlands and swamps also occur in this section and the lowest part of the river is estuarine and up to 100 m in width.

The successive terrestrial-marine conditions produced by Late Pleistocene-Holocene sea level fluctuations led to the deposition of the sequences of sand, clay and gravel of the coastal plain. Farming and agricultural activities have been common and now there is a continued expansion of urbanisation. The development of sugarcane required construction of networks of drains; some bypass meanders on the middle section, and tidal gates were built on the lower river. There are also other developments which are common in these coastal areas, such as hard rock and sand quarries, nurseries and marinas. A result of these developments is some degree of change to natural systems. An important environmental issue that has emerged is the formation of acid sulfate soils and waters, due to exposure of pyritic clays formed in the reducing conditions of the wetlands and tidal flats. There is local impact on aquatic life, but downstream transport of such acidity appears limited as it becomes buffered by the more saline estuarine conditions.

Introduction

Although the Pimpama River and its tributaries drain only 100 km^2 of the Moreton region, this drainage system possesses many features characteristic of coastal catchments in southeast Queensland. In summary, the river has a steep headwater zone in coastal ranges, a central low lying section with a meandering course and an estuary sheltered by a sandy barrier island. The coastal plain over which the mid-lower course of the river meanders has been shaped by Quaternary sea level fluctuations and currently has been modified by a variety of agricultural and urban land use activities. Significant in this catchment are the types of changes and associated impacts that are occurring in much of the Moreton Bay peripheries. Typical of the Pimpama area is the occurrence of acid sulfate soils and the subsequent impact on local development.

Physical Setting

The Pimpama River is located 80 km south of Brisbane, in the Gold Coast City Council area (Figure 1). It originates on the eastern flank of the Darlington Range at an elevation of about 300 m and flows over 30 km into the southern part of Moreton Bay. In its central section the Pimpama River is joined by its major tributary, Hotham Creek. In the lower course, the estuary is formed by flat lying mud-sand islands such as Woogoompah Island and is protected from the open ocean by the sand dunes of South Stradbroke Island.

The climate of the Pimpama region is subtropical humid with a hot wet summer and a mild dry winter. On the coastal plain the average precipitation is 1 300 mm, increasing in the ranges to

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 165-172.

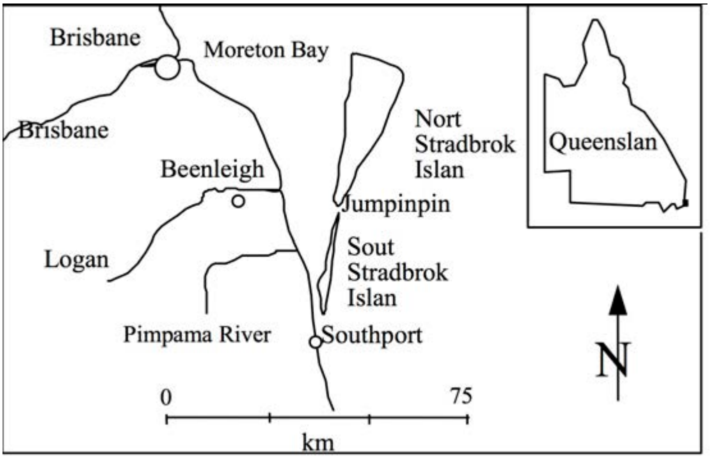

Figure 1. Map of the study area showing the location of Pimpama River.

1 600 mm. About 70% of this rainfall occurs during the wet season, from November to April (Bureau of Meteorology, 1983). Prolonged precipitation associated with tropical storms often produces flooding in the lower sections of the catchment. This seasonal regime of rainfall is an important component of the geochemical processes that occur in the area.

The ranges of the upper catchment are covered in native vegetation which is largely species of *Eucalyptus*, with pockets of bamboo, figs and ferns. The coastal plain has been cleared for agricultural purposes and the native bushland remains only on hills and slopes. Pines spread along the river banks in both fresh and brackish parts of the catchment. In the estuarine section of the Pimpama River and at its mouth the shores are covered with mangroves.

Land Use

The Pimpama River catchment reflects many of the features of land use typical of the Brisbane-Moreton area, as well as the type of changes that have been occurring over the last 10-15 years.

The upper course of the Pimpama River is quite pristine with little human intervention (Figure 2: location 1 and 2). Near the source, there is a bush nursery and banana plantations. The few residential areas use the river water for domestic purposes. From Kingsholme (Figure 2: location 3) downstream small farms are common and the flatter areas of the river valley have been cleared for grazing. Quarries for greywacke (Figure 2: location 4) have been established on the eastern flank of the ranges. The Pacific Highway tends to follow the western edge of the coastal plain and its location is significant as development becomes concentrated along it.

The mid-lower catchment has developed as an agricultural area with sugar cane and fruit plantations. During the 1960s and 1970s the Woongoolba Flood Mitigation Scheme was developed to help the local agricultural industry by restricting the tidal flooding. Drainage canals and tidal gates were constructed between the Pimpama and Logan Rivers. A fifteen cell structure with hanging gates was constructed on the Pimpama River (Figure 2: location 13). One of the larger canals built in that period is Eggersdorf drain (Figure 2: location 10) which became an artificial tributary of the Pimpama River (Holz, 1979). The drain is linked to other canals and, at the northern end, this network is connected to the Logan River. Locally, the Eggersdorf drain cuts off one meander of the river which eventually became inactive.

The upper and middle courses of the tributary Hotham Creek have been extensively cleared for grazing; the lower section is cultivated with sugar cane and used for growing cattle. There

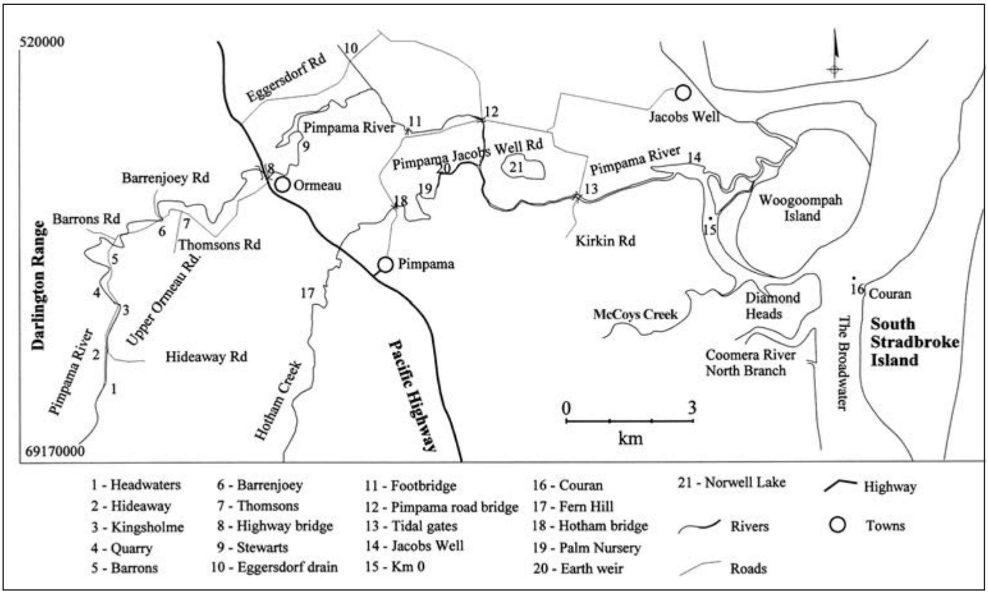

Figure 2. Study area and sample locations.

is also a quarry that extracts chert (Figure 2: upstream of location 18) and a palm nursery (Figure 2: location 19) which uses creek water for irrigation. The creek water at this location is suitable for cattle and plants as an earth weir limits upstream migration of salt water (Figure 2: location 20). East of the junction with the Hotham Creek a small natural lake, Norwell Lake (Figure 2: location 21) has been enlarged artificially as a result of ongoing sand mining operations. Another sand mining operation is located adjacent to the estuarine section of the river, immediately south of Jacobs Well.

The entire catchment is under development pressure especially around the highway at Ormeau and on the coastal strip at Jacobs Well. The rapid residential expansion led to improvement of the transport system such as the reopening of the Brisbane-Southport railway and upgrading of the Pacific Highway, the main motorway of eastern Australia.

Outline of Geological Evolution

Paleozoic lithological character

The Pimpama River originates in highlands of Late Paleozoic rocks. These Neranleigh-Fernvale Beds form the basement of the area and are exposed in the Darlington Range. The rocks are dominantly flysch-type sediments and were deposited on the continental slope and the adjacent abyssal plains of the Late Devonian continental margin. The Neranleigh-Fernvale Beds are formed by different types of sandstones interbedded with shale and chert; in the Early Carboniferous period the rocks were folded and metamorphosed to greenschist facies (Day *et al.,* 1983).

The basement formation is an anticline with an axis trending north-northwest and the eastern limb dipping towards the Pimpama River. In the headwaters, greywacke is the dominant rock type exposed along the valley. Here the rock is massive, moderately foliated, with minor interbeds of shale. It is dark grey when fresh, medium to coarse grained, containing quartz, K-feldspar, plagioclase and hornblende. The formation can be highly weathered depending on jointing and foliation (Cooper, 1979).

In the central section of the catchment, the basement is mainly formed by phyllite and greywacke and is overlain by deposits of younger sediments with thicknesses varying between 6 and 30 m (Quarantotto, 1979).

Quaternary geological history

The coastal plain is formed of Quaternary age sediments of fluvial and marine origin which are deposited as layers over the older bedrock. The most important geological events that ultimately shaped the area were the Quaternary sea level fluctuations. The coastal plain of southeast Queensland and Moreton Bay and its islands were all subjected to the same sequence of terrestrial and shallow marine processes. The impacts of these events were, however, different from place to place, depending on local morphology and dynamics.

During the last Pleistocene low sea level the valleys widened dissecting the coast and the newly exposed continental shelf. Erosion of the pre-Quaternary deposits and palaeo-deltas supplied alluvial material for the river valleys and for the growing islands of the Bay.

This is probably the time when thick layers of gravel accumulated along the central sections of the Pimpama and Hotham valleys. Adjacent to the Pacific Highway, around Ormeau and Pimpama (Figure 2) the basement rocks are overlain by deposits of fluvial gravel (Quarantotto, 1979). The uniformity of the occurrences suggests that the mid courses of the rivers have not changed much over the last several thousand years, however, the thickness of these gravel deposits (up to 5 m) indicates a more voluminous source of material and a more dynamic river system than exits today.

During the same period, series of sand ridges began to accumulate and form a barrier system at the Pimpama River mouth. At that time, Diamond Heads (near the mouth) formed the eastern shoreline as a rock platform backed by a sea cliff. To the north of this escarpment there was a large embayment where the Pimpama mouth was an inlet. Northward longshore drift of sand produced a series of spits; the waves and currents entering the embayment caused the northern end of the spits to migrate to the northwest. As successive ridges accumulated to the east and north, the embayment filled and the Woogoompah Island began to form (Friederich, 1973).

When sea level rose at the beginning of Holocene, the river valleys flooded and the coastline prograded with the development of estuarine conditions (Kelly & Baker, 1984). In the lower catchment, thick deposits of sand with horizons of marine shells cover the estuarine clays (Willmott *et al*., 1976). This quartzose sand occurs mainly in the Norwell Lake area and south of Jacobs Well and suggests deposition in a near-shore marine environment with little estuarine influence.

As the sea level fell to the present position, the onshore sediment supply produced the closure of ancient Jumpinpin bar, between North and South Stradbroke Island. This caused the development of low deltaic islands and channel migration. Landward, sets of levee banks and minor flood plains developed along the rivers as they adjusted to the new base-level (Kelly & Baker, 1984). As a result of sediment input and longshore currents, the barrier systems developed enclosing the southern part of the Bay and protecting the coastal plain and the low islands. It is only during recent times that the beaches and estuarine islands have been vegetated and stabilised (Beckmann, 1975).

The modern Jumpinpin opening (Figure 1) occurred in 1896 and changed the dynamics of the southern part of Moreton Bay. The rapid flow through this opening introduced oceanic sediments, induced erosion in the southern part of South Stradbroke Island and flooded large areas of the Pimpama River valley. As the region adjusted to the new dynamics caused by tides, fresh marine sediments were supplied to the Bay islands and the coast.

Geochemical processes

The coastal wetlands and tidal flats of the Mid-Holocene rising sea level provided ideal conditions for the production of sedimentary pyrite. The formation of this iron sulfide (FeS_2) requires sources of sulfur, iron and organic material. Subtropical estuarine environments can provide abundant organic material and its decomposition generates energy for the activity of sulfate reducing bacteria. These anaerobic bacteria reduce sulfate from the sea water to sulfide and ferric iron in sediments to ferrous iron. Pyrite is the end product of the reaction between detrital iron oxides and hydrogen sulfide (Dent, 1986; Dent & Pons, 1993; Melville *et al*., 1993).

In the mid-courses of Pimpama River and Hotham Creek the fluvial gravel is overlain by typical estuarine deposits, silty or sandy clays (Quarantotto, 1979) which in this area contain high amounts of pyrite.

Morphology of the Drainage Area

In its upper several kilometres the Pimpama River has a deep valley, 200 m wide and 40-50 m deep, with very steep sides, waterfalls and numerous side gullies (Figure 2: location 1). The gradient of the river bed is about 10% and the overall flow direction is from south to north. During the dry season, flow is largely groundwater seepage and the active channel is less than 1 m wide; local high energy flow is typical during floods of the summer wet season.

At Hideaway (Figure 2: location 2) the river gradient decreases to 2% and the channel widens to 5-7 m, although the flow is limited during the dry season. Active alluvial deposits formed of angular and subrounded pebbles and cobbles are common along the river forming riffles and pools.

From this point to Kingsholme (Figure 2: location 3), the river flows more slowly, forms irregular meanders and the width of the channel is 2-3 m. The valley enlarges, the slopes are gentler and the flatter areas are cleared of native vegetation. Further downstream (Figure 2: location 3 to 6), the gradient of the river bed decreases to 1% and the flow direction changes gradually to east-northeast. The river has many sharp bends and during the dry season it barely flows. From Thompsons Rd (Figure 2: location 7) the river enters the coastal plain where it is less than 20 m above sea level; the gradient decreases to $1^0/_{00}$ and the river channel widens to more than 5 m.

At Stewarts (Figure 2: location 9) the river enters swamps which are often dry during winter and the freshwater input at the junction with the Eggersdorf drain (Figure 2: location 10) ceases. The lower course is then supplied mainly by the tidal waters that enter the river channel through two paths, (a) the Pimpama mouth from the east and (b) Eggersdorf drain from the north. Downstream the river is tidal; the width gradually increases and the depth varies from 50 to 150 cm.

From location 11 to location 13 (Figure 2) the Pimpama River develops the largest meander of its course, 5 km long and 2 km in amplitude. Hotham Creek joins the river at the point of inflection of this meander, 7.2 km from the Pimpama mouth. At the Pimpama tidal gates, the river is around 20 m in width and 3-4 m deep (Figure 2: location 13).

Downstream from the tidal gates the river is open and dominated by tides. The estuary is typically 50 m wide or more (Figure 2: location 14) and up to 6 m deep, with steep banks covered in mangroves. The river flows into The Broadwater adjacent to Couran on South Stradbroke Island (Figure 2: location 16).

Character of Shallow Sediments and Water

General features

In the upper several kilometres the river channel is exposed bedrock. At the Hideaway (Figure 2: location 2) and downstream, the gradient decreases and there is deposition of fine sediments. The typical presence of subrounded cobbles and boulders confirm the occasional high flow velocity during floods. These coarse sediments are covered by a thin layer of silty clay formed of a variety of clay species: kaolinite, illite and irregular mixed-layers of smectite-illite. From Barrons (Figure 2: location 5) downstream the river channel contains an increasing proportion of clayey sands. The dominant clay species here are illite, kaolinite and smectite. The water of the upper catchment is fresh with salinities around 500 mg/L and neutral pH. Downstream of the greywacke quarry (Figure 2: location 4) minor increases in amounts of Ca, Mg and Zn in the water may be caused by this operation.

Shallow cores collected along the lower Pimpama River and Hotham Creek revealed mainly silty or sandy clays with high contents of pyrite. The clay species are comprised of mixed-layers of smectite-illite with smaller amounts of kaolinite and illite. The water quality of the lower catchment is influenced by a variety of natural and anthropogenic factors.

Aspects of environmental geochemistry

The lower catchment of the Pimpama River is the primary zone where pyrite-rich sediments occur. Pyrite is usually quite stable under reducing conditions. Oxidation and hydrolysis of the layer containing sedimentary pyrite result in production of sulfuric acid which can destroy the structure of the sediments and affect the quality of river water.

The character of the lower catchment water is influenced by the chemical reactions which take place in the shallow sediments of the area, and are affected by seasons, tides and land use. The seasonal regime of rainfall controls the process of pyrite oxidation and therefore the amount of sulfuric acid that is washed into the rivers. The tides provide continuous dilution of acidity as the saline waters have a strong buffering effect. Various activities such as the quarries, drain construction and clearing may directly expose layers with high acid-producing potential or cause oxidation by lowering the water table.

The tidal Pimpama River has salinities of 20 000-30 000 mg/L and a low pH around 5. Occasional rainfall induces seepage from the adjacent sediments and the river pH can drop to 3. The acidity breaks down the sediment structure releasing Al and Fe. During the dry season when the process of oxidation is more intense and the production of acid more conspicuous, the water can contain around 15 mg/L Al and 10 mg/L of Fe. Prolonged rainfall after a long dry period may produce water with very high amounts of Al as well as low pH which are toxic to fish. Hotham Creek is fresh for most of its course due to the earth weir and the absence of the tidal buffering effect. The pH can remain around 3 for many months; and concentrations of Al and Fe in solution can both reach 40 mg/L. During the wet season the water quality improves (Preda & Cox, 1996).

The sediments of the estuary are sandy clays similar to those of the river, except they do not contain pyrite. The water quality is typical of seawater and is not influenced to a noticeable degree by the periodic toxicity of the lower catchment.

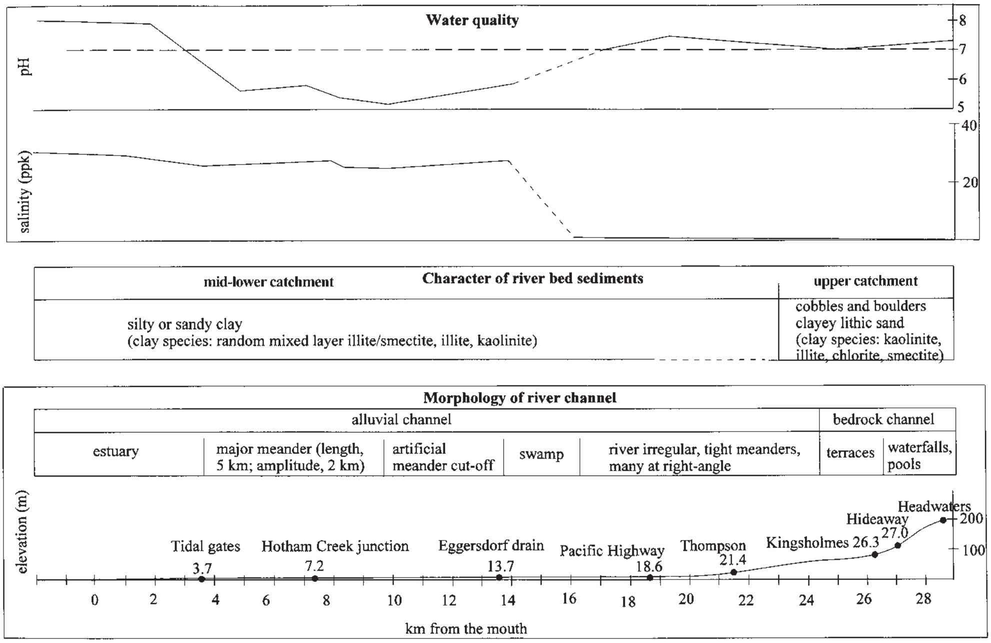

Figure 3. Character of Pimpama river channel, sediments and water.

Conclusions

The Pimpama River catchment possesses features of geomorphology, geology and land use that typify coastal drainage areas of southeast Queensland (see summary in Figure 3).

The river originates in coastal ranges, meanders over the coastal plain and flows into Moreton Bay. The Late Paleozoic basement is overlain by sequences of various Quaternary sedimentary deposits. The successive terrestrial-marine conditions produced by Late Pleistocene-Holocene sea level fluctuations led to deposition of layers of fluvial gravel, estuarine silts, clays and sands typical of a nearshore marine environment. Of these sediments, the silty clays throughout the mid- and lower catchment represent a very recent episode in the evolution of the coastal plain; these are typical locations for the formation of sulfuric acid by the oxidation of sedimentary pyrite. Produced toxicity does impact on local aquatic life, but in the estuary and Bay the impact is limited as the acidity produced upstream becomes buffered by the saline estuarine conditions.

Acknowledgements

We thank staff of School of Natural Resource Sciences for assistance with laboratory work: S. Price, chemical analysis of waters and sediments; W. Kwiecien, scanning electron microscopy analysis; T. Raftery, X-ray diffraction. B. Powell and C. Ahern from Department of Natural Resources encouraged the research of the Pimpama River area and clarified aspects related to acid sulfate soils. We thank J. Ferris from the Department of Environment for assistance in the field. The managers of the Palm Nursery on Hotham Creek are thanked for access to the creek and rainfall records.

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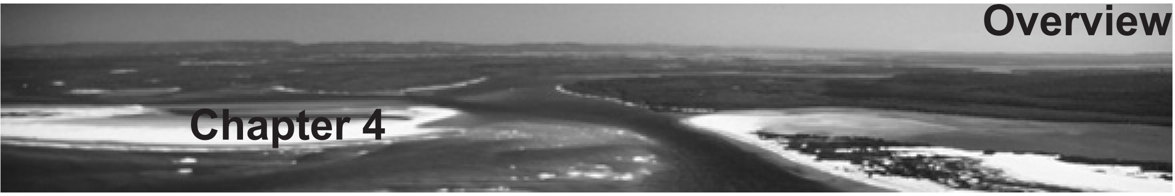

Overview

Chapter 4

Water Quality of Estuarine and Bay Waters

The water quality of Moreton Bay and its associated estuaries is in part a consequence of "freshwater" inflow, described in the previous chapter, and various physical and biological processes which prevail in the euryhaline and marine reaches of the system. Water quality controls the distribution and health of organisms described in subsequent chapters. As a consequence, the first major attempt at quantifying the parameters necessary for the integrated management of the system has as its focus water quality: The Brisbane River and Moreton Bay Wastewater Management Study (BRMBWMS). The present chapter provides analyses of the factors that influence water quality, it impacts and several fairly specific considerations.

Malcolm Cox provides two papers that relate the present characteristics of water quality for the tidal reaches of the Brisbane River, the most important input to the Bay. Pauline Semple *et al*. present a study of a small creek that provides a clear picture of the potential influence of various nutrient point sources on the biota of the system and the Bay into which it flows. James Udy and William Dennison describe the potential of seagrass as a bioindicator of not only the presence of nutrient impacts on the Bay system but also their source. This latter role of seagrass as a sophisticated detective, able to yield information on what is impacting it, is particularly exciting. It allows us to determine unequivocally the relationship between nutrient source and impact, providing an additional level of subtlety to future environmental monitoring and management practices.

While Udy and Dennison provide hope that the influences of nutrient point sources can be identified and thus managed, Wesley Walden and Brian Bycroft review the complexity of models that will be required to manage non-point source input to the system. They highlight the fundamental importance of stormwater runoff in non-point source impacts on water quality, and emphasise that models developed for catchments other than Moreton Bay are of limited use. They argue that the presently available models of non-point source impact are as yet insufficient to allow prediction of cause and effect – a fundamental requirement for effective management of this challenging problem.

James McEwan's paper takes the theme of water quality modelling a step further by describing a mathematical model with direct application to managing nutrient loads entering the rivers and Bay. Developed through the BRMBWMS, the model represents the application of investigations into water movement through the system and physical and biological processes which interact to modify nutrient and toxicant loads in associated waters and sediments. McEwan points out that we must advance rapidly our understanding of nutrient inputs and their subsequent physical and biological transformations if we are to " determine critical nutrient loadings beyond which irreversible damage to the functioning of the Bay ecosystem would occur".

Following McEwan's paper are three papers that describe particular facets of water quality issues in some detail. Munro Mortimer relates the distribution of organochloride pesticides in the system and ponders their ecotoxicological consequences. Micaela Preda and Malcolm Cox provide details of an acid sulfate forming soil in the southern Bay. The acid from such soils, when released by poor land management practices, can have swift and serious consequences for the waters into which it discharges. There have been several recent media reports about this phenomenon and the fish kills that are one of the more visible effects of this form of pollution. Michael Brown and co-authors evaluate the effect of various insecticides, either used widely or planned for use in the control of mosquito populations, on other animals that share habitats favoured by the larvae of certain mosquitoes. Their laboratory experiments indicate that the use of two agents should be discontinued and that field-based experiments are required to determine acute and chronic effects of larvicides on non-target organisms. Such work is particularly important because the saltmarsh and mangrove habitats that are breeding grounds for health threatening mosquitoes are also of major importance as nursery habitats for juvenile fish and crustaceans. The application of larvicides that kill non-target species could have serious ecological and economic side effects.

This chapter ends with a paper that draws the human dimension into the management of water quality. Cesar Luna describes a consultative process which aims to maximise the acceptance/efficacy of management actions concerning water quality through harmonising the needs of stakeholders with the available scientific information.

Ian R. Tibbetts

Chemical and Turbidity Character of the Tidal Brisbane River, Moreton Bay

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Abstract

Water sampling was conducted on established traverses approximately 5 km apart in the tidal Brisbane River. Sampling was during a low flow regime after a prolonged dry period, and at high water. Trends for turbidity and suspended matter are similar to each other, with values typically of 5-70 NTU and 20-120 mg/L; both decrease downstream but are lowest where salinities are > 20.0. Of note, both turbidity and suspended matter are greater (typically 20% for turbidity) in near-bottom waters than at 0.5 m depth. Chemical analysis of water samples shows the estuary contains diluted marine water but has a much lower Cl/K (15 compared to 50 for seawater) reflecting dissolved and suspended K. Total SiO_2 (< 13 mg/L) and total Fe (< 3.4 mg/L) both steadily decrease downstream and reflect dissolved and suspended forms. Mn is variable, but typically low (< 0.064 mg/L). Total Zn is relatively low, the highest value 0.084 mg/L, and was detected only in the lower reaches where most maritime-industrial activity occurs, and appears to be primarily in dissolved form.

Introduction

General

A range of studies of the lower Brisbane River has been reported that considers various aspects of water quality and physico-chemical parameters (e.g. Straughan, 1967; Rankin & Milford, 1979a; b; 1980; Moss, 1979; 1987; Bennett, 1990; Connell & Bycroft, 1990; Razzel, 1990). This paper presents part of a study undertaken to establish a more detailed distribution of chemical parameters within the estuary, its chemical composition and the extent of the tidal effect. Here the main findings for suspended material and chemistry are discussed. High water was selected as the tidal phase, to provide both consistent conditions, and the greatest penetration of marine water.

Changes to the Brisbane River

Various physical modifications have been made to the river, but significant to chemical character are several large dams and flood mitigation schemes. The weir at Mt Crosby (Figure 1) incorporates a major water supply treatment plant and the large Wivenhoe Dam, in the upper section of the Brisbane River began effective operation in 1983. Razzel (1990) notes that water quality at Mt Crosby is far superior to that within the urban limits of the river, which reflects that many causes of lower water quality arise downstream. A factor that has now been recognised is the increase in chemical and sediment loads from the urban zone and urbanising activities themselves (e.g. Stock & Neller, 1990) which can be seen to impact on the central reaches of the estuary. Maritime and other industry had developed along the river up to 18 km from the mouth, but over the last 10 years industry has been relocated to the lower 5-6 km. The Port of Brisbane has been relocated to the mouth and is currently being extended.

Dredging has changed much of the lower 55 km of the river, and is for both navigation purposes and supply of sand. Dredging for navigation is now mostly within the lower 15 km, and in several sections "swing basins" are maintained to enable large vessels to manoeuvre. Such dredging is particularly needed after flood events, as much fine sediment is deposited in the

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 175-184.

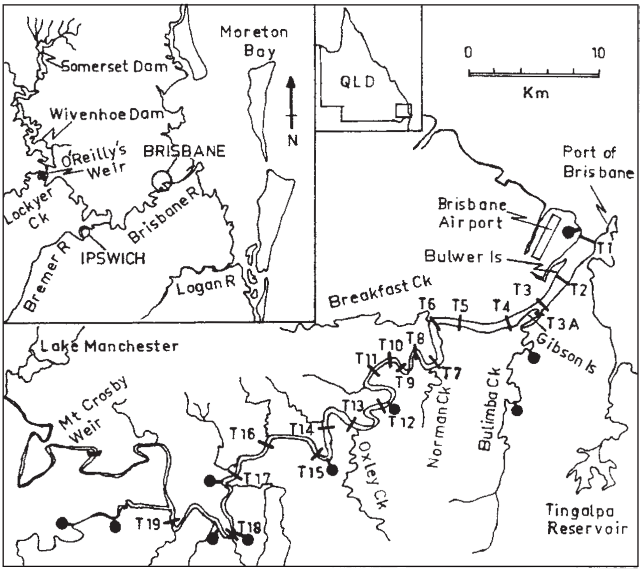

Figure 1. Location of Brisbane River and sites of traverses within the tidal section. Sewage treatment plants discharging to the river are shown by closed circles. The locations of the cities of Brisbane and Ipswich are shown in insert.

lower sections, particularly in swing basins (J. Dobson, pers. comm. 1996).

In the Brisbane City reaches the river has also been widened on the inner section of four large meanders. Another significant modification has been the construction of training walls along much of the lower 15 km. Of note, these walls also blocked off the upstream sections of channels around Gibson and Bulwer Islands (Figure 1).

Production of sand and gravel deposits within the river reached a peak of 1.45 x 10^6 m^3 per year in the mid-1970s, and has declined since. Through the 1980s annual production in the tidal reaches was 0.68-0.95 x 10^6 m^3 but was less than half of this in 1990. Currently dredged material is now around 80% sand and 20% gravel (O'Flynn & Thornton, 1990). The total amount of solid material removed is difficult to establish, but indications are that 140 x 10^6 m^3 have been dredged over the last 130 years (Dobson, 1990).

Several outcomes of dredging activities are generally accepted (e.g. Stock & Neller, 1990): (a) dredging has changed the cross-sectional area of many places in both the lower and upper estuary; (b) the estuary has been widened, producing greater penetration of saline water and larger tidal exchanges; (c) dredging has changed the relative proportions of gravel, sand and silt; and (d) dredging is a major factor in re-suspending particles.

Hydrological setting

This lower section of the river is a coastal plain estuary formed as a drowned river valley. The river course and channel cross-section are strongly controlled by the geology. Much of the lower course of the river, flows within fine-grained Triassic-Jurassic age sedimentary rocks (e.g. Cranfield *et al.,* 1976; Stevens, 1990). Near Brisbane City several northwest trending rhyolite flows have been cut by the river, and strongly influence some meanders (e.g. T8). In other places, meanders have been produced by the resistant metamorphic basement rock. The lower 5 km or so around the mouth of the current river, is a delta formed of fine-grained sands and silts which extends an equal distance out into Moreton Bay (e.g. Jones *et al.,* 1978;

Grimes *et al.,* 1986; Evans *et al.,* 1992).

The region has a sub-tropical humid climate, with a hot wet summer and a mild dry winter. As a consequence the Brisbane River is characterised by a low ratio of rainfall/runoff, large volume floods, and occasional severe droughts with very low flow. The Mt Crosby Weir, approximately 98 km from the mouth, and at the head of the estuary, traps the whole flow of the main river. Other creeks below the Bremer River have limited contribution during drought and freshwater flow into the river is now principally controlled by releases from Wivenhoe Dam.

The configuration and position of the saltwater "wedge" is dependent upon the amount of freshwater discharged into the river system; in dry periods the saline water penetrates far upstream, while in extreme floods the freshwater is pushed downstream or even out into Moreton Bay (e.g. Stephenson, 1968; Cossins, 1990). The tidal limit in the Brisbane River is now about 95 km from the mouth, and extends past the Bremer River almost to Mt Crosby, as well as around 20 km into the Bremer.

Methods

The survey was at high water and during a low flow regime. It was carried out in the period from December 1993 to January 1994, and was confined to a 4 h period straddling high tide at each traverse location. There was no rainfall during the survey or in the preceding two-month period, which followed 3-4 years of very low to nil rainfall.

A series of traverses across the river was made to produce cross-sectional profiles of the channel itself and the main physico-chemical parameters. Seventeen traverses (T1-T17) were established, and single measurements made at a further two sites (T18 and T19). Surveys on the traverses were carried out from either a 6 m runabout or survey barge. The channel cross-section was measured as bottom echo profiles, using a Raytheon Survey Bathometer. These profiles were then adjusted to the width of the river at the traverse, taken from orthophotomaps (QDT, 1990; 1991). The cross-sections are reproduced in Figure 2 with a vertical exaggeration of 20 times.

Water samples were collected central to each traverse at a depth of 0.5 m directly into polypropylene sample bottles, one for chemical analysis and one for turbidity/suspended solids. Near-bottom water samples were collected on traverses T1, T2, T3 and T4, using a submersible pump and weighted hose. Samples for chemical analysis were unfiltered and were stored under refrigeration. Cations and metals were determined by ICP analysis, as were SiO_2 and SO_4, Cl by Mohr titration and HCO_3 by HCl/sodium tetraborate titration. Turbidity was measured with a Hach Turbidimeter; suspended matter was determined in mg/L by filtering 250 g water through a glass microfibre membrane, and obtaining the dry weight difference to five decimal places.

Results and Discussion

Morphology of the river channel

Figure 2 shows transverse profiles of the river channel to the high water mark. The "U" shaped profile of the swing basins is evident at traverses T1 (Luggage Point) and T5 (Hamilton). The dredged navigational channels can be seen in profiles T2 and T3. The narrower profiles of the channel occurring upstream of T7 (New Farm Park) are within the area where dredging for sand is conducted. Several meanders demonstrate geological controls having a steep outer side and a sloping inner bank (e.g. T7, T8, T10 and T11). The changes in morphology of the river channel can be examined by comparing depth to width (Figure 3a). The variation in depth reflects the dredging in the lower 50 km of the estuary.

The distribution of salinity in the estuary is largely a result of the longitudinal dispersion by tide, currents and wind which produce mixing of freshwater flowing into the river with the

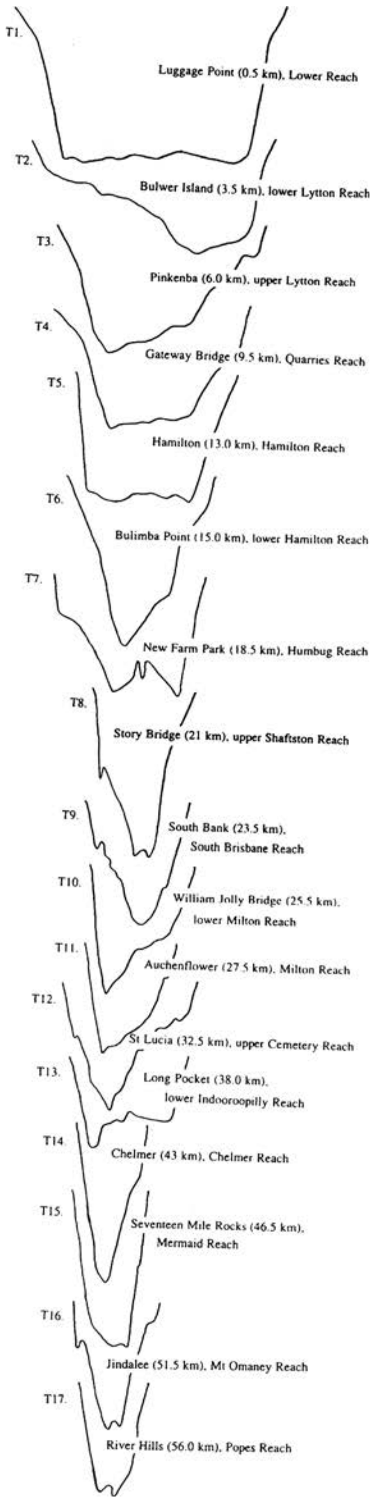

Figure 2. Cross sections of river at high water from echo sounder profiles, looking downstream with "north" bank on left. Vertical exaggeration is 20 x.

saline water. Major factors in mixing of constituents in rivers are horizontal and vertical shear in the body of water, and importantly in the lower Brisbane River, turbulent mixing. However, it has been shown (e.g. Sooky, 1969) that in comparatively wide rivers horizontal shear is more dominant than vertical in producing dispersion. It can be seen from Figure 3b that although the lower reaches are wide, with a width/depth > 20, the section from around 20 to 55 km has a ratio of < 20. These different dispersive mechanisms are reflected, for example, in the distribution of salinity and temperature.

Summary profiles of chemical parameters

Results over the survey area are summarised as longitudinal profiles in Figure 3. Salinity (Figure 3b) decreases steadily from the mouth to around 57 km by 5.0‰ per 10 km, then to a lesser rate. The temperature profile (Figure 3c) shows that under high tide conditions, temperatures along the river are lowered by water from Moreton Bay up to around 20 km; pH continually decreases upstream with mixing, at around 0.28 pH units per 10 km (Figure 3d). Values of Eh (Figure 3e) reflect different redox conditions within upper and lower layers. Dissolved oxygen in the estuary has an obvious zone of lower value (< 4.5 mg/L) from around 18 to 52 km upstream (Figure 3f). A comparison is made with DO values for October 1974, which followed the major flood of January 1974; the results suggest that the DO profile has been pushed downstream, with high DO values subsequently measured in the central section.

Chemical composition and character of water

Turbidity

Values of suspended matter and turbidity are given in Table 1, and plotted against distance (Figure 4a). Measured turbidity is typically 70 to 5 NTU, and suspended matter 120 to 20 mg/L. The strong trend for both is to decrease downstream, and also to be higher in bottom than near-surface samples. Turbidity is typically 11-32% greater in near-bottom samples; suspended matter also tends to be higher, but is variable. This finding has implications for assessment of volume of sediment transported, especially as the conditions of this study were of low flow. Measured turbidity shows a generally similar trend to mean Secchi depths (Figure 4b) measured at other times (Queensland Water Quality Council, 1987 in Stock & Neller, 1990; Moss, 1990). Of note is the reverse correlation of turbidity and salinity (Figure 4c) with lower NTU values in higher salinity water.

The turbid character of the lower Brisbane River has been concluded (Stock & Neller, 1990) to be caused by the excessive production of inorganic and organic suspended sediments from rural and urban sources, and continued re-suspension by processes such as tidal action and dredging. Turbidity is also seen to be related to high rainfall and associated runoff events. Razzel (1990) reports typical intake water at Mt Crosby Weir (February 1988) at 0.8 to 2.0 NTU, with values downstream of Wivenhoe Dam after storm runoff of around 2, up to 55 NTU (February - March 1988); after storms of April 1988 turbidity reached peaks of around 420 and 890 NTU.

Maximum turbidity is nearly always located around the freshwater/saltwater interface, and is considered due to cycling and particle concentration (e.g. Postma, 1967; Moss, 1990). Downstream, turbidity decreases markedly, which is generally accepted as a function of river-stable colloids depositing, or floccing due to the increased ionic strength (higher electrolytic effect) of seawater. It is also related to a lower energy environment and less eddying within the river. Figure 4c shows that at around 15-25 km (T4-T8) there is substantial deposition of suspended material within the zone of salinities of 25.0-30.0. An important aspect of turbidity is that strong tidal currents keep the particles in suspension, while low rates of advection result in long residence times. These factors are added to by the on-going dredging.

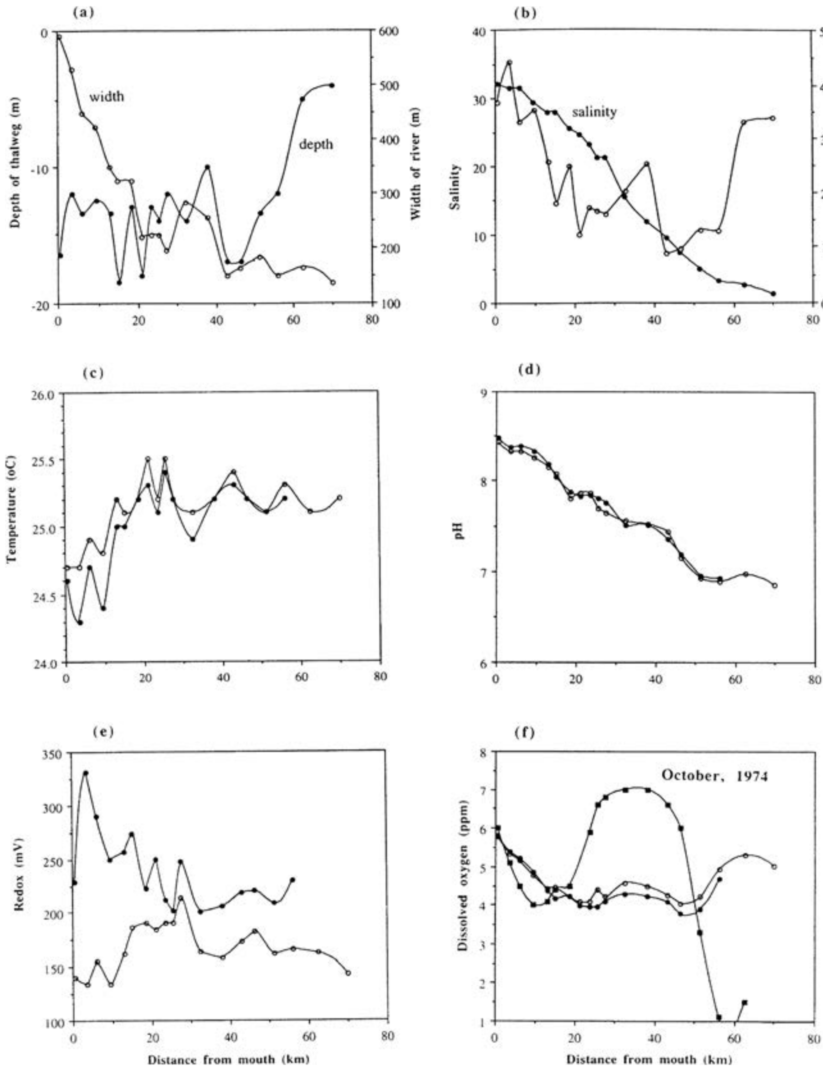

Figure 3. Longitudinal profiles for central section of channel: (a) depth and width (m); (b) mean salinity compared to ratio of width/depth; (c) mean temperature (^{o}C); (d) mean pH; (e) mean Eh (mV); (f) dissolved oxygen (mg/L), values shown by squares are from October 1974 following the major flood in January 1974 (Rankin & Milford, 1980). For plots c, d, e and f, open circle is value at 3 m, and closed circle is at 7 m depth.

Major ions

Brisbane River water becomes progressively more saline downstream with increasing amounts of marine water. Analyses of the major dissolved constituents (Table 2) show, however, that there are some differences in relative proportions of major ions when compared to average seawater. Significant is the much higher amount of K, even up to 30 km from the mouth; average seawater has Cl/K of 50, but the Brisbane River estuary is relatively constant around 15. The higher K is due to dissolution of K from soils and silts, and very fine suspended matter. HCO_3 is appreciably higher reflecting the meteoric water origin, as does the lower proportion of Cl. The Cl/HCO_3 for average seawater is 147; for the Brisbane estuary the ratio increases steadily from around

Table 1. Minor dissolved elements in water and suspended load (Suspd), Turbidity (Turbid) and total dissolved solids (TDS) for stations T1-T17 on the Brisbane River.

Location	SiO_2	Fe	Mn	Zn	Suspd	Turbid	TDS
T1 (0.5)	1.6	bld	bld	bld	35.20	6.5	32473
T1 (16.0)	0.4	bld	bld	bld	35.60	7.6	34521
T2 (0.5)	0.7	0.01	0.008	0.025	31.60	5.3	31514
T2 (8.0)	0.6	0.02	0.020	0.052	46.80	7.0	33209
T3 (0.5)	0.8	0.01	0.014	0.084	40.80	6.3	31761
T3 (10.0)	0.8	0.01	0.010	0.072	44.40	7.0	32898
T4 (0.5)	1.4	bld	bld	0.074	41.60	5.5	29449
T4 (10.0)	0.9	bld	0.006	0.084	27.20	7.3	31575
T5 (0.5)	1.3	bld	0.004	0.044	19.20	4.5	28096
T7 (0.5)	2.0	1.07	bld	bld	53.20	14.0	25682
T9 (0.5)	3.3	0.77	bld	bld	46.80	24.0	23140
T11 (0.5)	2.5	0.24	0.064	bld	31.60	15.0	22078
T12 (0.5)	5.4	1.56	0.014	bld	72.80	41.0	15052
T13 (0.5)	10.8	2.10	0.016	bld	106.0	50.0	11907
T14 (0.5)	8.2	1.82	0.014	bld	123.6	62.0	9389
T16 (0.5)	8.7	1.63	0.01	bld	62.80	40.0	5075
T17 (0.5)	12.7	3.36	bld	bld	92.80	68.0	3523

All units are μg/L, except turbidity which is in NTU;
bld = below limit of detection.

3 at T17 to 35 at the mouth. SO_4 is also in lower proportion than in seawater, except in bottom waters near the river mouth, reflecting the salt water "wedge".

Minor ions

Analyses of total content of the minor constituents SiO_2, Fe, Mn and Zn are given in Table 1. Both SiO_2 (mostly < 10 mg/L) and Fe (mostly < 2.0 mg/L) decrease downstream, and are higher in near-surface waters. Mn during this survey is overall low, the highest value being 0.064 mg/L, and typically < 0.02 mg/L (20 μg/L); distribution is variable although it tends to be higher in the section of highest suspended load. Razzel (1990) reported that total Mn in Mt Crosby intake waters is typically 10.0 to 65.0 μg/L; after storm runoff concentrations downstream of Wivenhoe Dam can increase to several hundred mg/L or more. The highest Zn concentration in this current study was 0.084 mg/L, Zn only being detected in the lower reaches. Abstracted data from the Water Quality Council (1976) report (in Milford, 1978) shows mean total Zn concentrations (μg/L) in the Brisbane River of < 29.3, and in tributaries as, Bremer < 27.5, Oxley < 42.9, Bulimba < 29.1, and 176 in Norman Creek. Moss (1979) concluded that there were no discrete discharges of metals into the river; the sources would include sewage effluents and storm water runoff.

A plot of SiO_2 and Fe in river waters against turbidity (Figure 4d) shows that both strongly decrease with suspended load, which indicates their presence as both dissolved and suspended forms. Loss of SiO_2 and Fe from estuarine waters with increasing salinity is well known, although the form of the Fe and the mechanisms of its removal can be complex (e.g. Boyle *et al.,* 1974; Sigleo & Helz, 1981). A plot of Fe and Zn (Figure 4e) against distance (and therefore salinity) shows a different distribution for Zn to Fe (and SiO_2). This indicates that Zn concentration is not controlled by suspended material, but is primarily dissolved. Further, the detectable Zn content of the river water is restricted to the section between T2 and T5 (around 3 to 13 km upstream) the section with the greatest concentration of industry and maritime activity.

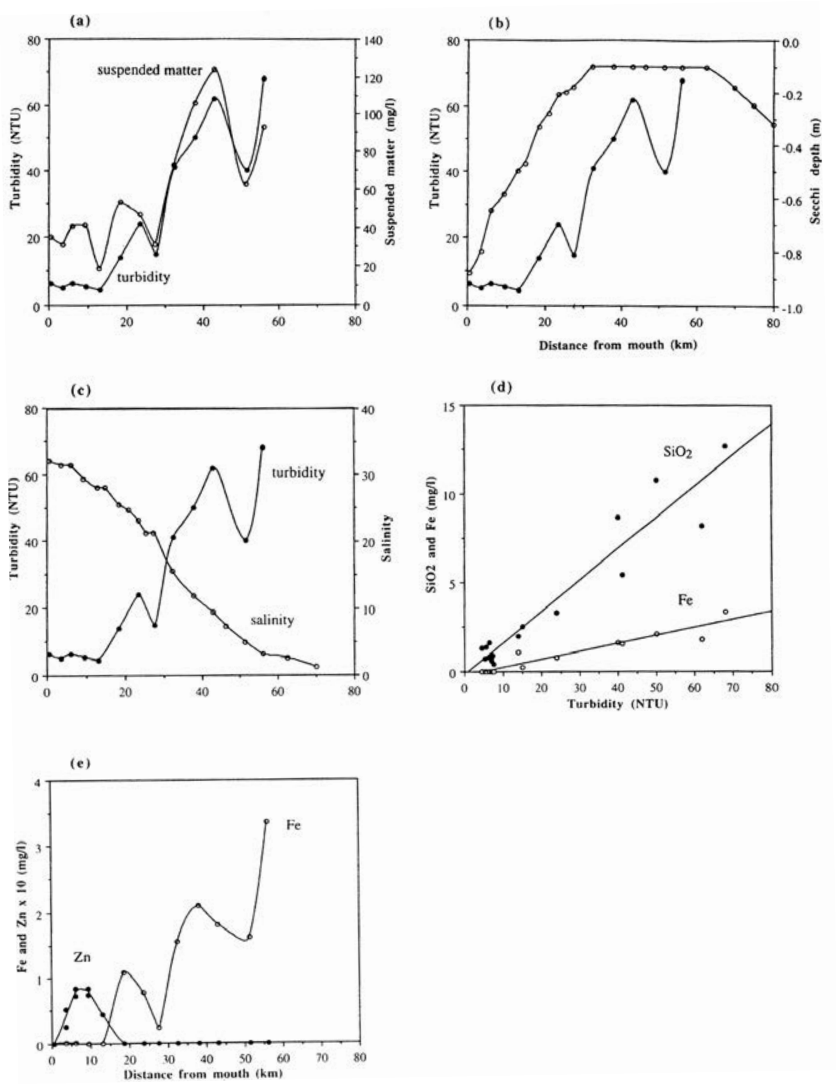

Figure 4. Longitudinal profiles of suspended load and trace elements in water at 0.5 m depth: (a) turbidity, NTU (closed circle) and suspended matter, mg/L (open circle); (b) turbidity, NTU (closed circle) and Secchi depth, m (open circle); (c) turbidity, NTU (closed circle) and salinity at 3 m depth (open); (d) SiO_2 (closed) and Fe (open) in mg/L compared to turbidity, with lines of best fit; (e) Zn x 10 (closed) and Fe (open) in mg/L compared to distance from mouth. Secchi depths are mean values taken from Moss (1990).

Conclusions

Chemical analyses show that the estuary contains diluted marine water, but relative to average seawater it has a much lower Cl/K, reflecting dissolved and suspended K sourced from sediments. The Cl/HCO_3 is also appreciably lower than seawater, and increases steadily towards the mouth. The extent of intrusion of the salt water wedge is clearly shown by SO_4 concentrations. Turbidity generally decreases downstream, and drops rapidly in the zone 10-20 km from the mouth where salinities are 25.0-30.0. Both turbidity and suspended matter are overall appreciably higher in near-bottom water than in the near-surface layers. Measured concentrations of SiO_2 and Fe are low and both suspended and dissolved. Zn is overall low, and

Table 2. Major dissolved ion composition of water samples at each traverse (T1-T17) on the Brisbane River.

Location	Na	K	Ca	Mg	Cl	HCO_3	SO_4	TDS
T1 (0.5)	9608	1104	397	1122	17301	492	2449	32473
T1 (16.0)	10190	1188	379	1186	18419	516	2643	34521
T2 (0.5)	9437	1115	360	1123	16401	503	2575	31514
T2 (8.0)	10114	1176	377	1176	17342	516	2508	33209
T3 (0.5)	9558	1110	359	1120	16456	508	2650	31761
T3 (10.0)	10045	1156	372	1164	17122	525	2514	32898
T4 (0.5)	8960	1024	333	1044	15284	495	2309	29449
T4 (10.0)	9465	1094	353	1113	16574	506	2470	31575
T5 (0.5)	8037	919	299	937	13726	460	3718	28096
T7 (0.5)	7658	913	297	933	13498	456	1927	25682
T9 (0.5)	6805	821	276	830	12107	499	1802	23140
T11 (0.5)	6476	785	258	786	11523	546	1704	22078
T12 (0.5)	4411	532	183	541	7788	505	1092	15052
T13 (0.5)	3544	415	152	421	5848	535	992	11907
T14 (0.5)	2779	327	123	336	4592	483	749	9389
T16 (0.5)	1431	175	75	178	2314	476	426	5075
T17 (0.5)	921	116	59	117	1583	517	210	3523
Av Seawater	**10620**	**380**	**400**	**1280**	**19100**	**130**	**2650**	**34560**

One sample was taken at each site at 0.5 m depth. Deeper samples taken on traverses T1 to T4 were pumped to the surface. A comparison is also made with mean seawater. TDS = total dissolved solids. All units are μg/L.

largely in dissolved form and in the zone 3-13 km from the mouth, the site of most industry and maritime activity.

Acknowledgements

The study was carried out under Queensland University of Technology Research Grant NRGS93.FRM. The Port of Brisbane Authority is thanked for making available a suitable boat and skipper, and providing access to their shore facility. Assistance during the survey was provided by Rob Virtue, and further analytical assistance by Sharyn Wooderson.

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Distribution of Salinity and Other Physico-Chemical Parameters in the Brisbane River Estuary, Moreton Bay

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Abstract

The tidal section of the Brisbane River is a coastal plain estuary in which channel morphology is controlled by geology but has been modified by dredging. A detailed study of physico-chemical parameters of the river on 17 traverses, 4-5 km apart provides an understanding of the vertical and longitudinal distribution of salinity, pH, redox (Eh), dissolved oxygen (DO) and temperature. The surveys were conducted after a prolonged dry period, and at high water. Under these conditions of a low flow regime, the estuary had a character of a slightly stratified type. The cross sections of salinity and pH show that the estuary displays a distinct salt wedge only in the lower 10-15 km. Comparison with periods of high flow in floods shows that although salinities decrease, stratification becomes more developed. Longitudinal profiles show a steady decrease upstream of salinity and pH from typical seawater values, by around 5.0 per 10 km and 0.28 units per 10 km, respectively. Eh is broadly distributed as lower values (150-180 mV) in the upper 6-8 m, and below this higher values of 200-300 mV. Dissolved oxygen shows low values, < 4.5 mg/L, in the most urbanised zone from 13-55 km upstream.

Introduction

General

Numerous studies have been conducted on the Brisbane River including various aspects of water quality and chemical parameters (e.g. Straughan, 1967; Rankin & Milford, 1979a; b; 1980; Moss, 1979; 1987; Bennett, 1990; Connell & Bycroft, 1990; Razzel, 1990). Studies which consider physico-chemical parameters such as salinity and dissolved oxygen have been largely based on single measurements in longitudinal profiles, with little consideration of vertical and cross-sectional variations. An aim of this current study is to establish a more detailed distribution of chemical parameters within the estuary, and determine the form of the seawater intrusion. High tide was therefore selected to provide consistent conditions, and the greatest penetration of marine water.

Changes to the Brisbane River

A variety of physical modifications has been made to the river, all of which produced some re-adjustment; significant are several large dams and flood mitigation schemes. The weir at Mt Crosby (Figure 1) was built in 1926 and now incorporates a major water supply treatment plant. Somerset Dam on the Stanley River tributary began to store water in 1943 and was completed in 1959. The large Wivenhoe Dam, in the upper section began effective operation in 1983.

The lower section of the river is a coastal plain estuary formed as a drowned river valley. Although the river course and channel cross-section are strongly controlled by rock type and geological structures, the effect of the urban developments must also be considered.

Dredging has impacted on much of the lower 55 km of the river, for both navigation purposes and supply of building materials (e.g. Dobson, 1990). Dredging for navigation is now mostly within the lower 15 km, and in several sections “swing basins” are maintained to enable large vessels to manoeuvre. The river mouth has been significantly altered: in 1912 a channel was

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 185-202.

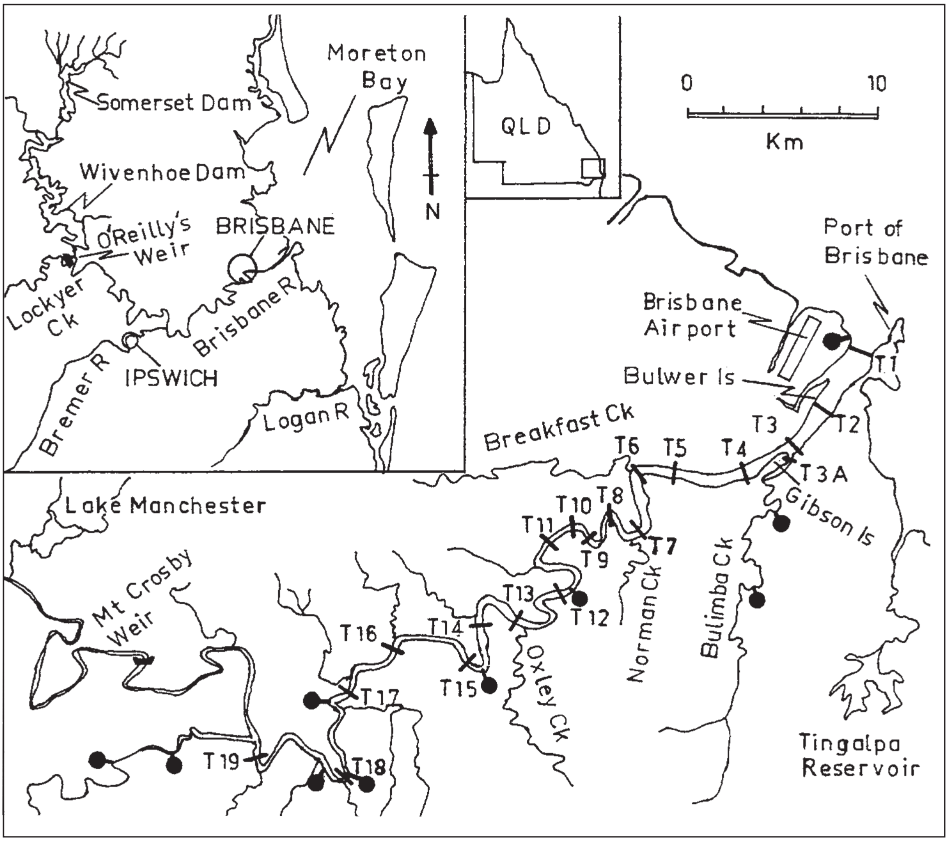


Figure 1. Location of Brisbane River and sites of traverses within the tidal section. Sewage treatment plants are shown by closed circles.

cut to a depth of 7.1 m, deepened in 1965 to 11.6 m, and in 1987 to 13.0 m deep and 180 m wide. This channel now joins the swing basin dredged in 1965, the site of T1.

Between 1901 and 1941, the river was also widened on the inner section of four meanders: Gardens Point (800 m downstream of T9), Kangaroo Point (T8), Kinellan Point (400 m downstream of T7) and Bulimba Point (T6). Another significant modification has been the construction of training walls along much of the lower 15 km, designed to straighten channel reaches and to promote scour. These walls also blocked off the upstream sections of channels around Gibson and Bulwer Islands (Figure 1).

Physical setting and hydrological character

The climate of Brisbane is sub-tropical humid, with a hot wet summer and a mild dry winter. During summer and autumn (November - April) winds are from the southeast, and thunderstorms and heavy rainfall are common. For the city of Brisbane mean annual rainfall is around 1 750 mm and evaporation around 1 630 mm; the mean minimum – maximum temperature range is 8-29 ^{o}C (Australian Bureau of Meteorology, 1983; Auliciems, 1990). High rainfalls and resultant flooding are usually associated with tropical cyclones.

In an Australian context the Brisbane River is of medium size, and is characterised by a low ratio of rainfall/runoff, large volume floods, and occasional severe droughts with very low flow. The average annual streamflow, allowing for abstractions, is 1.35 x 10^6 ML (e.g. Cossins, 1990). The Mt Crosby Weir, approximately 98 km from the mouth, and at the head of the estuary, traps the whole flow of the main river. Other creeks below the Bremer River have limited contribution during drought and freshwater flow into the river is now principally controlled by releases from Wivenhoe Dam.

Three freshwater flow regimes have been described for the lower Brisbane River (e.g. Bennett, 1990):

1. high flow, during which the estuary is flushed and there is a significant improvement in water quality (this is the flood condition);
2. an intermediate flow regime, which produces short-term deterioration of quality due to non-point sources of pollution, such as from storm water drains, and flushing of smaller, more polluted tributaries (this is great enough to flush significant loads of pollutants into the river, but not to flush the river itself); and
3. a low flow regime, during which the effects of point source discharges predominate, and dissolved oxygen values oscillate along the river due to tidal influence. During dry months, freshwater inflow to the tidal section is very limited, and the only dilution of any effluent is by tidal flows. This regime probably best reflects the conditions of this current study, and that existing for the longest period in any year.

Most of the features of chemical character/water quality of the estuary are strongly influenced by the nature of its hydrodynamic regime (see Moss, 1987; 1990). This is characterised by the movement of large volumes of water up and down stream, but only a relatively slow net downstream advection. Overall, tidal currents have a greater influence on the dispersal of salinity or other components than does freshwater flow, except in periods of high discharge. The tidal excursion, the total distance moved by the water between changes of direction, has been estimated for the lower Brisbane River at around 10 km (Steel, 1990). The tidal limit in the Brisbane River is now about 95 km from the mouth, and extends past the Bremer River almost to Mt Crosby as well as around 20 km into the Bremer.

Methods

The survey was of the high water regime. It was carried out in the period from December, 1993 to January, 1994, and was confined to a 4 h period straddling high tide at each traverse location. There was no rainfall during or in the period preceding the survey, which followed very low to nil rainfall conditions over the previous 3-4 years.

A series of traverses across the river was made to produce cross-sectional profiles of the channel itself and the main physico-chemical parameters. Seventeen traverses (T1-T17) were established, and single measurements made at a further two sites (T18 and T19). Surveys on the traverses were carried out from either a 6 m runabout or survey barge. The channel cross-section was measured as bottom echo profiles, using a Raytheon Survey Bathometer. These profiles were then adjusted to the width of the river at the traverse, taken from orthophotomaps (QDT, 1990; 1991). The cross-sections are reproduced in Figure 3 with a vertical exaggeration of 20 times. Traverse sites were located by identification of specific features on the banks.

Vertical measurements of physico-chemical parameters were made at 1 m intervals (except on T2 which were at 2 m intervals). The upper measurement of each sounding was at 0.5 m depth. Locations of the individual soundings were referenced to features on the river banks, and to depths on the echo profiles. Measurements were made using a TPS Field Analyser (Model 90-FL) with separate probes for conductivity-temperature, dissolved oxygen (DO), pH and redox (Eh). All probes were calibrated daily by standard methods. The Eh electrode used was a platinum-calomel and was referenced to a ZoBell standard solution. To compare

Eh values to a Standard Hydrogen Electrode +187 mV should be added to reported Eh. The probes were tied together and attached to a steel wire cable with a weighted end, and were housed in a conical cage at the cable end. A measuring tape was also attached to the steel cable. The maximum depth of measurement was limited to 9 m as the probes were attached to cables of 10 m length. One deeper sounding of conductivity was made on T1 using a probe with a longer cable. Test measurements showed that a special casing was required for the original pH probe used, as it was affected by pressure at depths over 4 m.

Drift of readings was tested randomly by five repeat measurements for each parameter, at different times, depths and locations. Typical repeatability was salinity/conductivity < 0.5%, temperature < 0.3%, pH < 0.2%, Eh 2-3% and DO 1-2%. Relocation at the same site on subsequent days (at the same tidal period) typically produced values 5-10% different, but with similar trends vertically.

Results and Discussion

Longitudinal profiles of physico-chemical parameters

Figure 2 displays longitudinal profiles of the parameters measured based on the most central sounding for each traverse. The distribution of salinity (Figure 2a) shows that at high tide, under the very dry conditions of summer 1993-1994, saline water had encroached well up the estuary, with values of 20.0 at Auchenflower 30 km from the mouth. Salinities of nearly 2.0 were measured at Moggill Ferry, 70 km upstream. The profile reflects slight stratification only and no distinct salt "wedge" except for the lower 5 km. Overall, vertical salinity gradients are small or uniform. In contrast, Figure 2b shows high tide salinity under high flow conditions after a flood in April, 1963 (Stephenson, 1968). It can be seen that very saline water is confined to the lower 5 km, and at 30 km salinity is only around 2.0; due to the greater outflow of freshwater the vertical salinity "layering" is better developed. Of note in the 1963 data is that the water depths are around 8 m; this is appreciably less that present depths as is it before the swing basins were dredged at T1 and T5.

The temperature profile (Figure 2c) shows the cooler (< 24.8^{o}C) marine waters at depth in the lower 15 km. Generally, shallow waters are warmer and temperatures at 0.5 m depth are typically 25.0-25.5^{o}C reflecting summer solar radiation. The distribution of pH (Figure 2d) reflects mixing and the proportion of marine water, and a pattern similar to salinity distribution (Figure 2a). The pH at the mouth is around 8.4, typically that of seawater, and 50 km upstream past Jindalee is < 7.0, reflecting the dominance of fresh river water.

An interesting distribution was measured for Eh (Figure 2e) with the deeper, more saline layers being more oxidising, with appreciably higher redox values. Shallower waters were commonly 130-180 mV, while deeper waters were 200-300 mV, with the deeper waters of the lower 25 km having Eh values > 250 mV. Dissolved oxygen (Figure 2f) displays a more systematic trend than does Eh, with less variability and a broad zone of low DO (< 4.50 mg/L) from around T5 (Hamilton; 13 km) to T16 (Jindalee; 50 km). The low DO zone represents both diffuse and point source input to the river from the city and the most urbanised section.

Cross-sections of physico-chemical parameters

Figure 3 (a to e) are contoured cross-sections of the traverses. Traverse 3A is 15 m into Aquarium Passage, and at the mouth of Bulimba Creek. Generally cross sections display a marked variation at their edges relative to the central parts, which is due to differences in velocity and discharge to the river. The main features on these cross-sections are noted below.

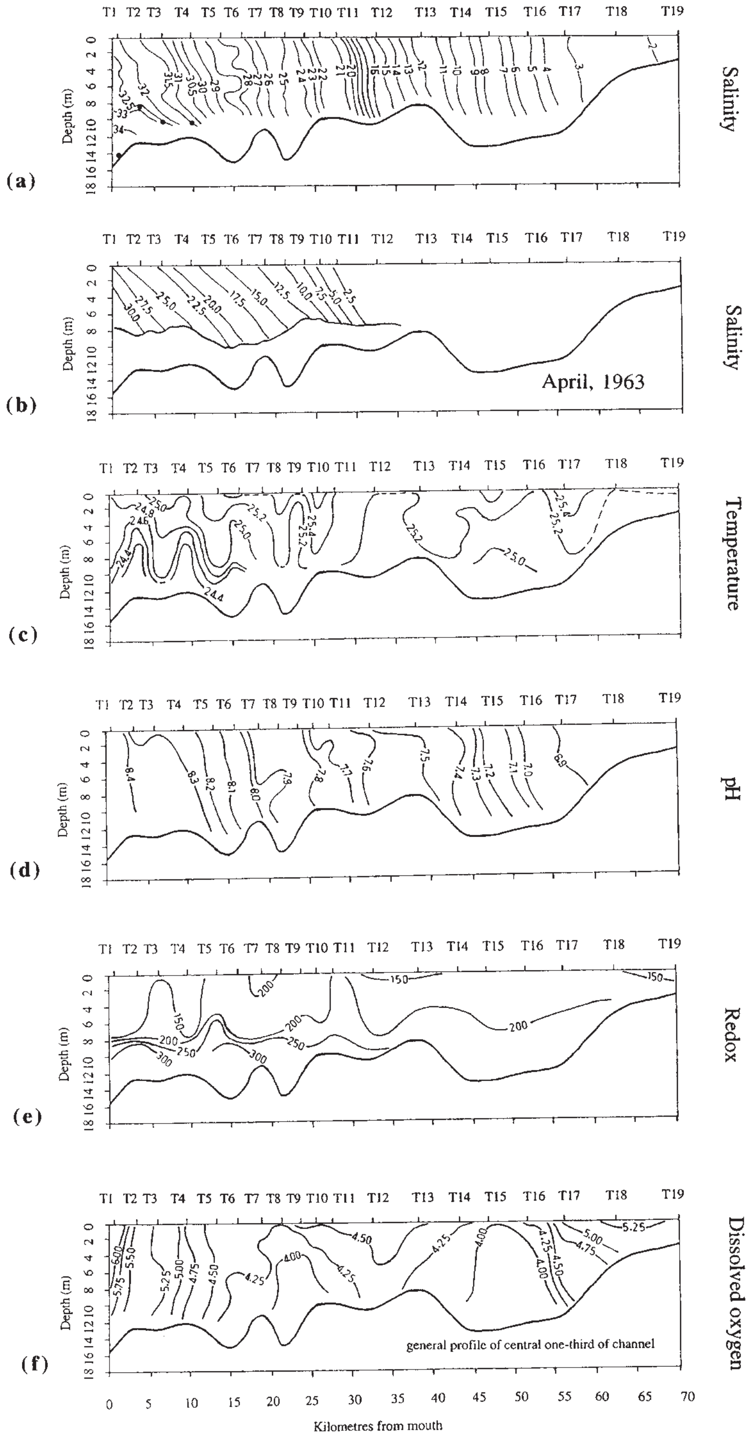
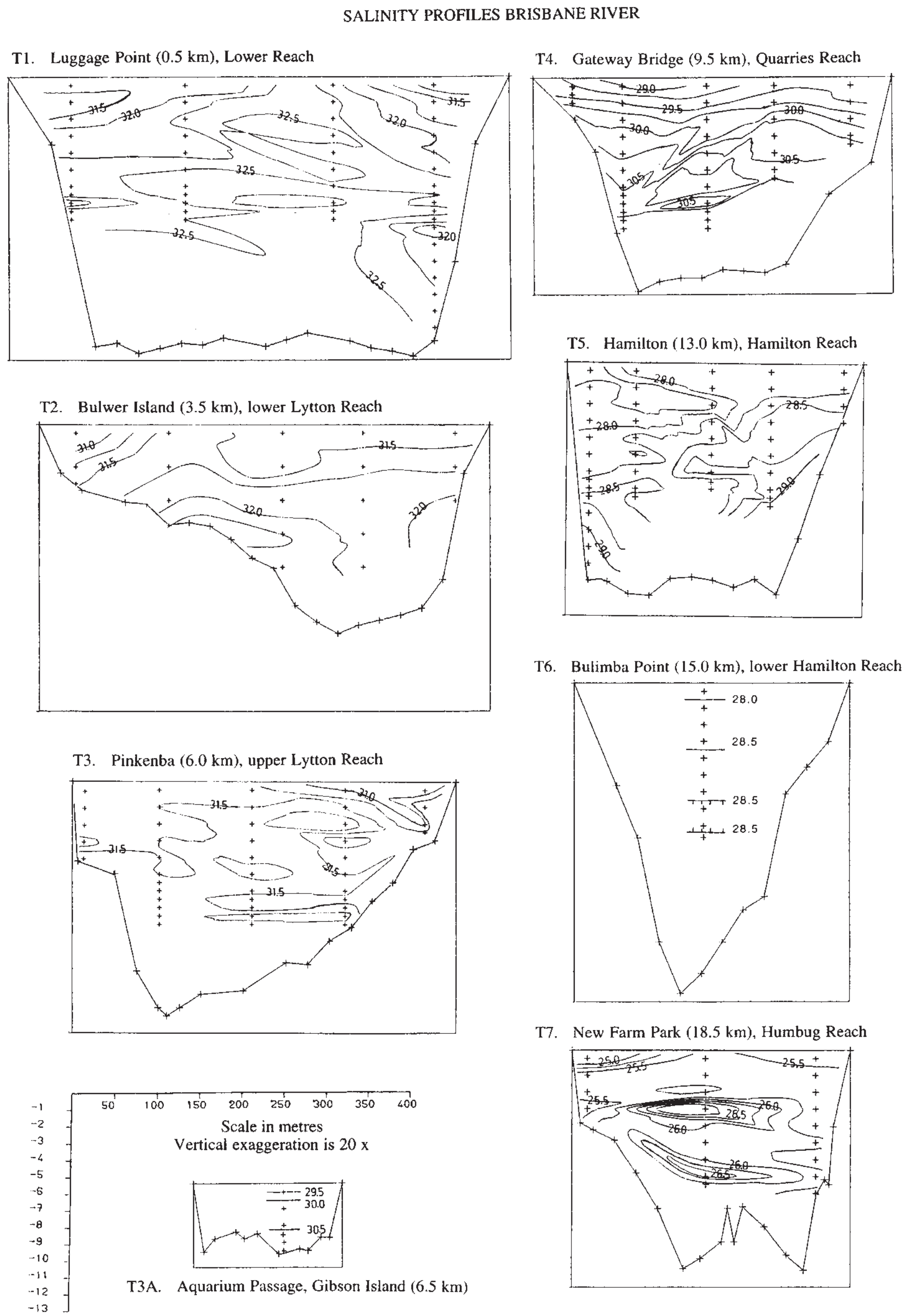
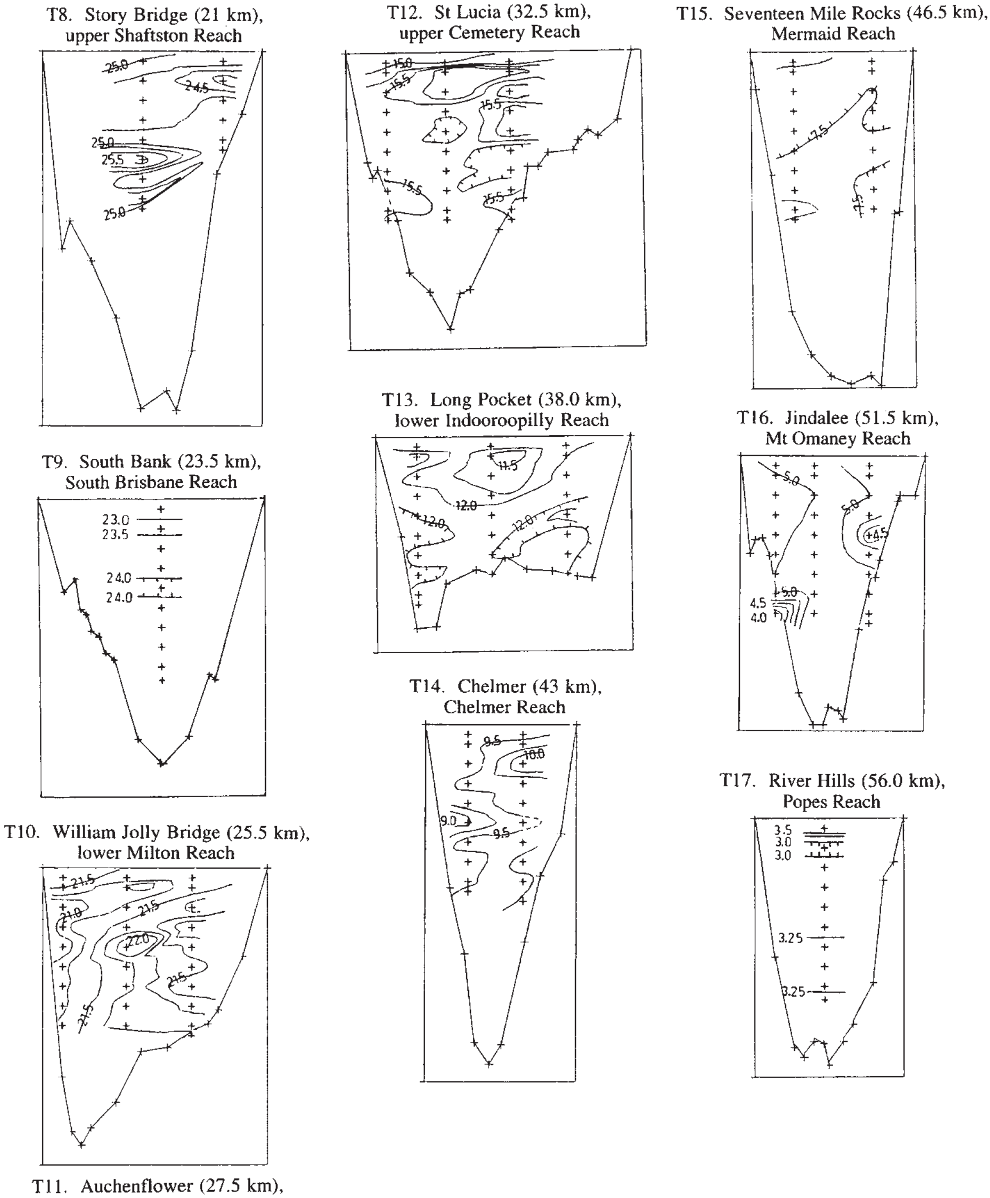


Figure 2. Longitudinal vertical profile at high tide, based on most central sounding in each traverse. Values at T18 and T19 are at 0.5 m depth: (a) salinity; (b) salinity and depth, 1963; (c) temperature in °C; (d) pH; (e) redox, mV; (f) dissolved oxygen, mg/L. Vertical exaggeration is x 1000. River bed profile is from soundings on traverses.

Salinity

The dynamic nature of the estuary is evident by vertical mixing, but some slight horizontal stratification is evident in the cross sections from the mouth to around T11-T12. T1 shows lower salinity to about 4 m depth on the north bank, probably related to the Luggage Point sewage outfall (discharging treated sewage has a salinity of 1.5) (Figure 3a). There is no extensive





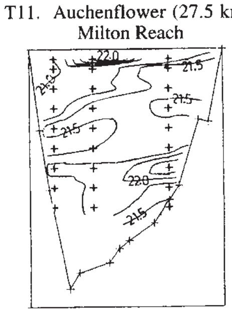
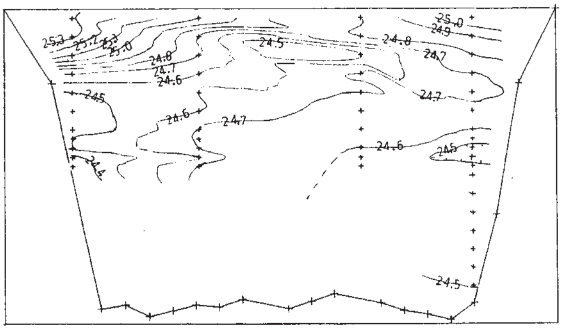


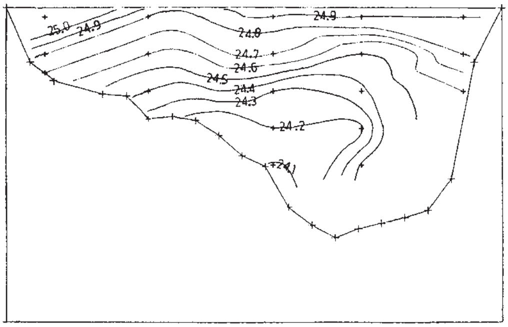
Figure 3a. Salinity profiles across the tidal Brisbane River at high tide. All profiles are viewed downstream, with the "north" bank on the left of profile. Cross sections are drawn from spot points on continuous echo sounder profiles; + shows site of parameter measurement. Traverse numbers and locations are given as the distance upstream along the river from the mouth at Luggage Point. The survey period was December 1993 and January 1994.

TEMPERATURE PROFILES BRISBANE RIVER

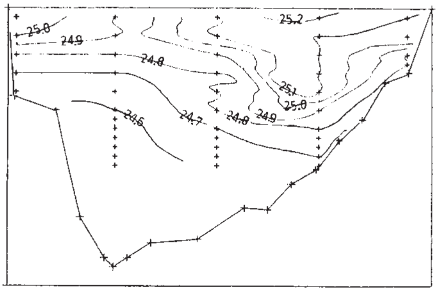
T1. Luggage Point (0.5 km), Lower Reach

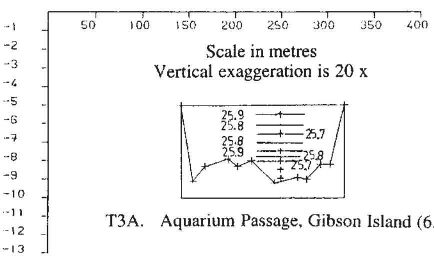


T2. Bulwer Island (3.5 km), lower Lytton Reach



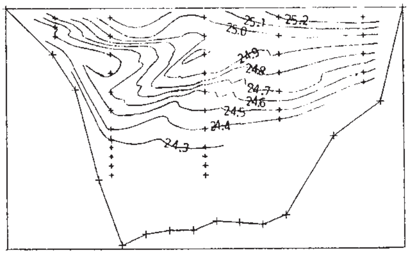
T3. Pinkenba (6.0 km), upper Lytton Reach



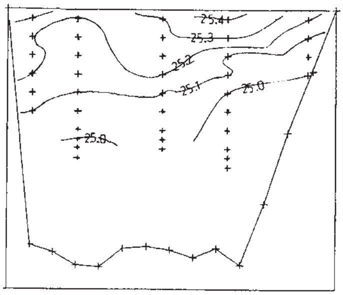


T3A. Aquarium Passage, Gibson Island (6.5 km)

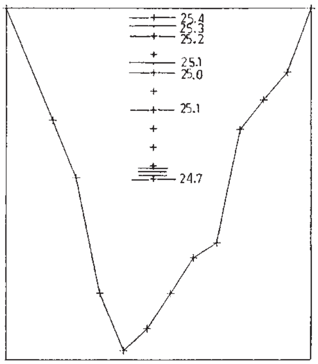
T4. Gateway Bridge (9.5 km), Quarries Reach



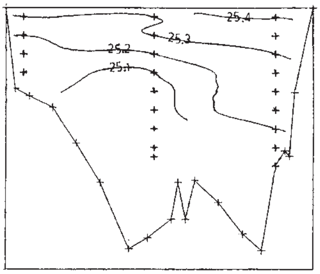
T5. Hamilton (13.0 km), Hamilton Reach



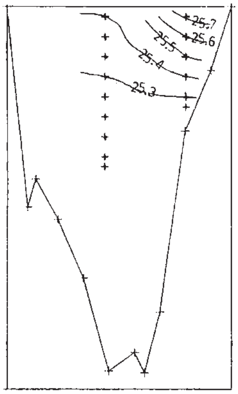
T6. Bulimba Point (15.0 km), lower Hamilton Reach



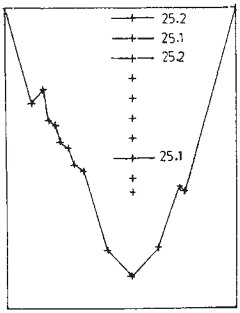
T7. New Farm Park (18.5 km), Humbug Reach



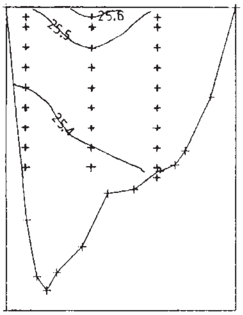
T8. Story Bridge (21 km), upper Shaftston Reach



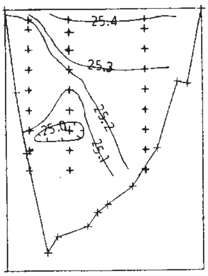
T9. South Bank (23.5 km), South Brisbane Reach



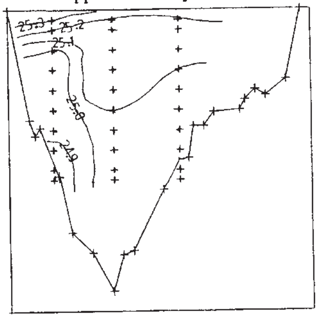
T10. William Jolly Bridge (25.5 km), lower Milton Reach



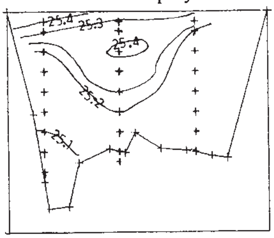
T11. Auchenflower (27.5 km), Milton Reach



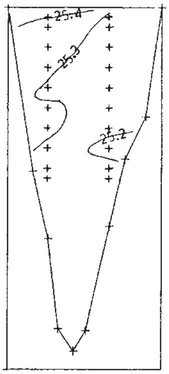
T12. St Lucia (32.5 km), upper Cemetery Reach



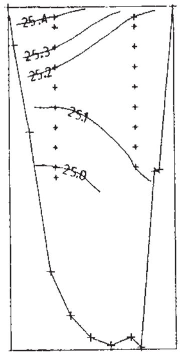
T13. Long Pocket (38.0 km), lower Indooroopilly Reach



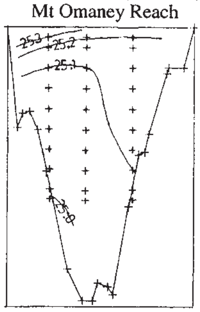
T14. Chelmer (43 km), Chelmer Reach



T15. Seventeen Mile Rocks (46.5 km), Mermaid Reach



T16. Jindalee (51.5 km), Mt Omaney Reach



T17. River Hills (56.0 km), Popes Reach

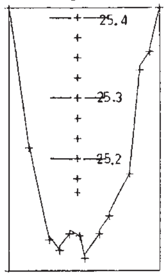
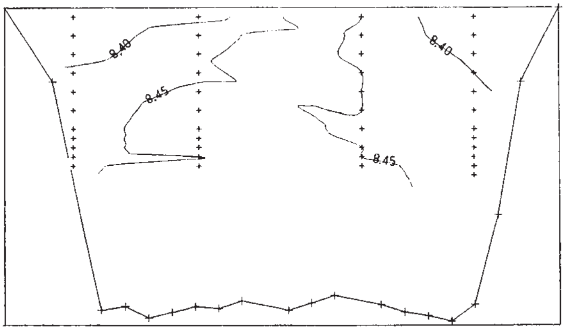


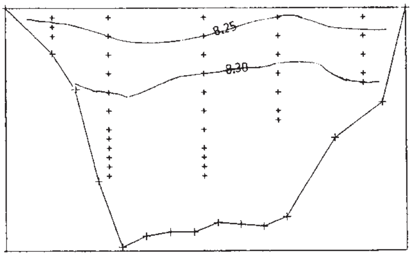
Figure 3b. Temperature profiles across the tidal Brisbane River at high tide. All profiles are viewed downstream, with the "north" bank on the left of profile. Cross sections are drawn from spot points on continuous echo sounder profiles; + shows site of parameter measurement. Traverse numbers and locations are given as the distance upstream along the river from the mouth at Luggage Point. The survey period was December 1993 and January 1994.

pH PROFILES BRISBANE RIVER

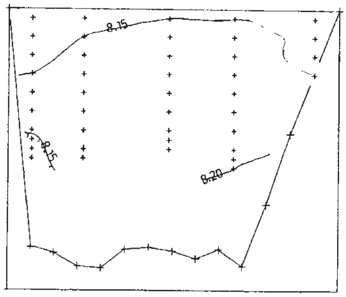
T1. Luggage Point (0.5 km), Lower Reach



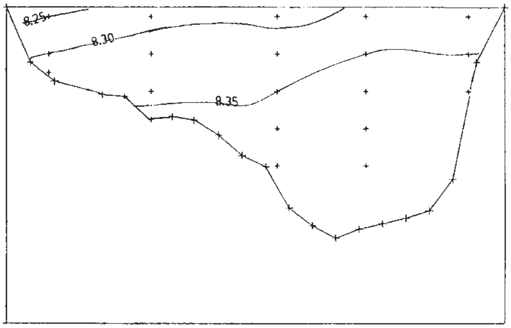
T4. Gateway Bridge (9.5 km), Quarries Reach



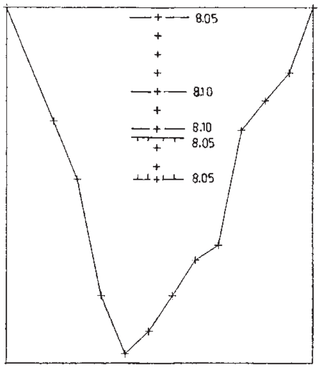
T5. Hamilton (13.0 km), Hamilton Reach



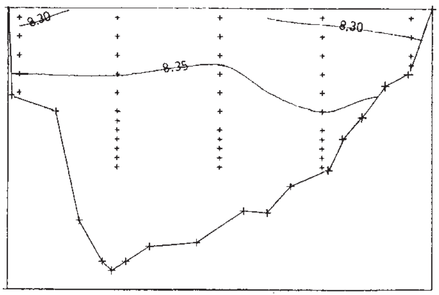
T2. Bulwer Island (3.5 km), lower Lytton Reach

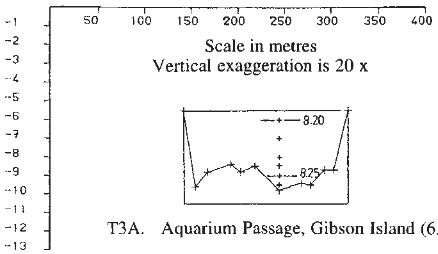


T6. Bulimba Point (15.0 km), lower Hamilton Reach



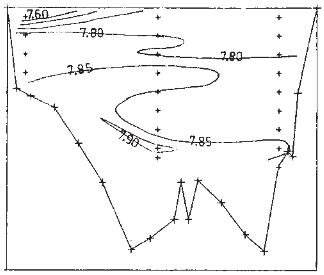
T3. Pinkenba (6.0 km), upper Lytton Reach

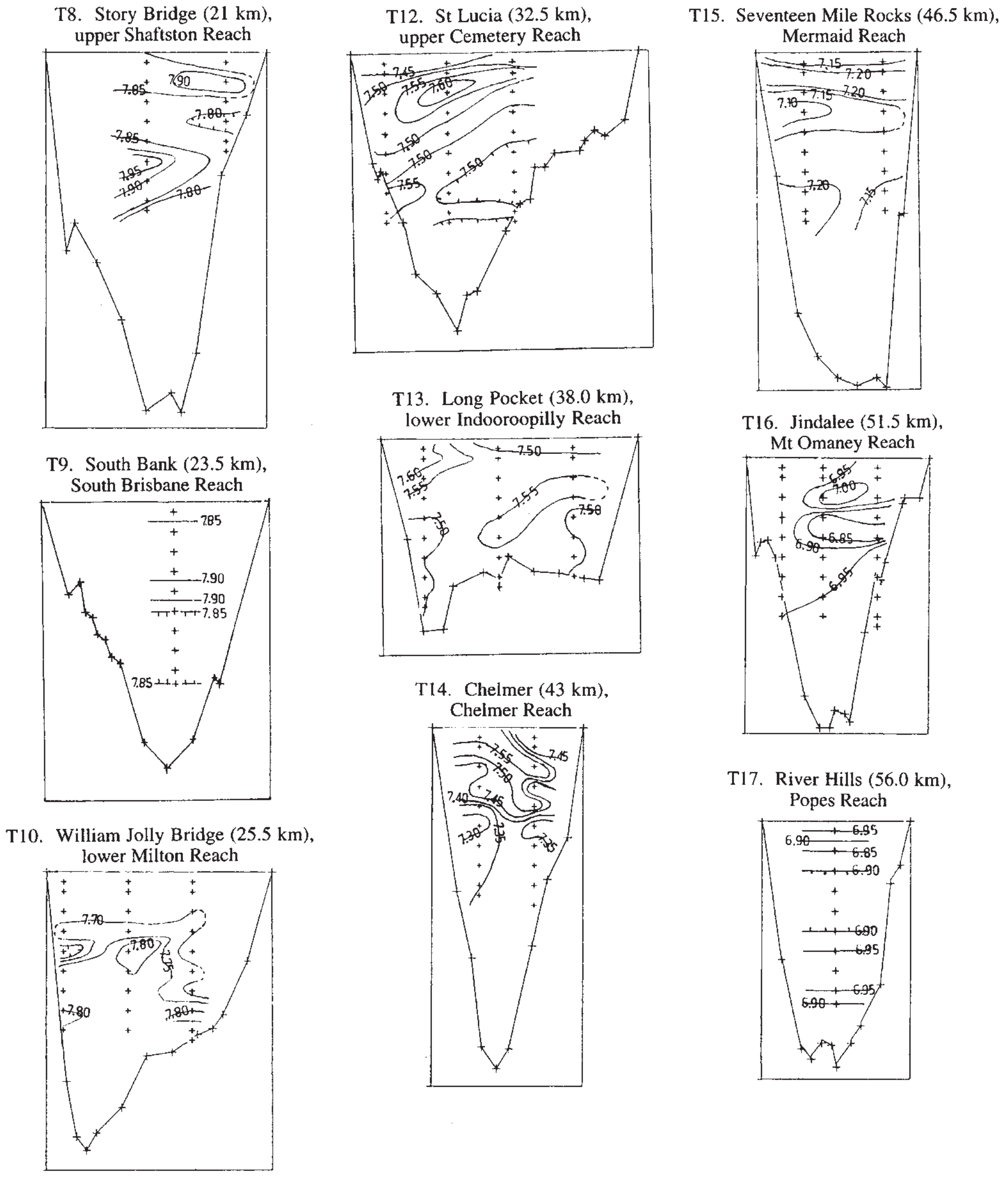




T3A. Aquarium Passage, Gibson Island (6.5 km)

T7. New Farm Park (18.5 km), Humbug Reach





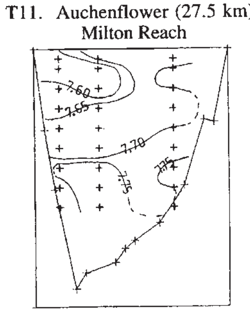
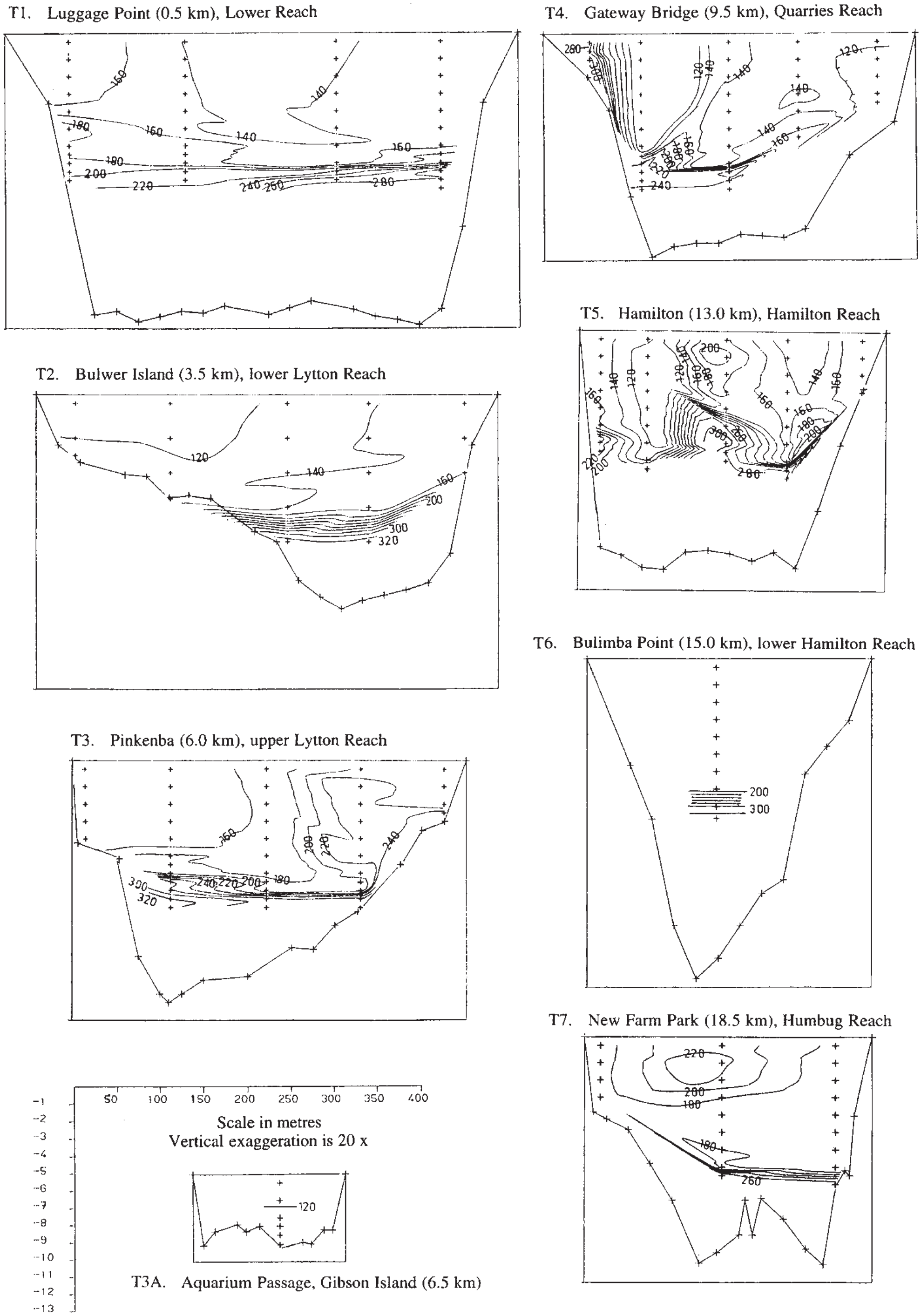


Figure 3c. pH profiles across the tidal Brisbane River at high tide. All profiles are viewed downstream, with the "north" bank on the left of profile. Cross sections are drawn from spot points on continuous echo sounder profiles; + shows site of parameter measurement. Traverse numbers and locations are given as the distance upstream along the river, from the mouth at Luggage Point. The survey period was December 1993 and January 1994.

Eh PROFILES BRISBANE RIVER



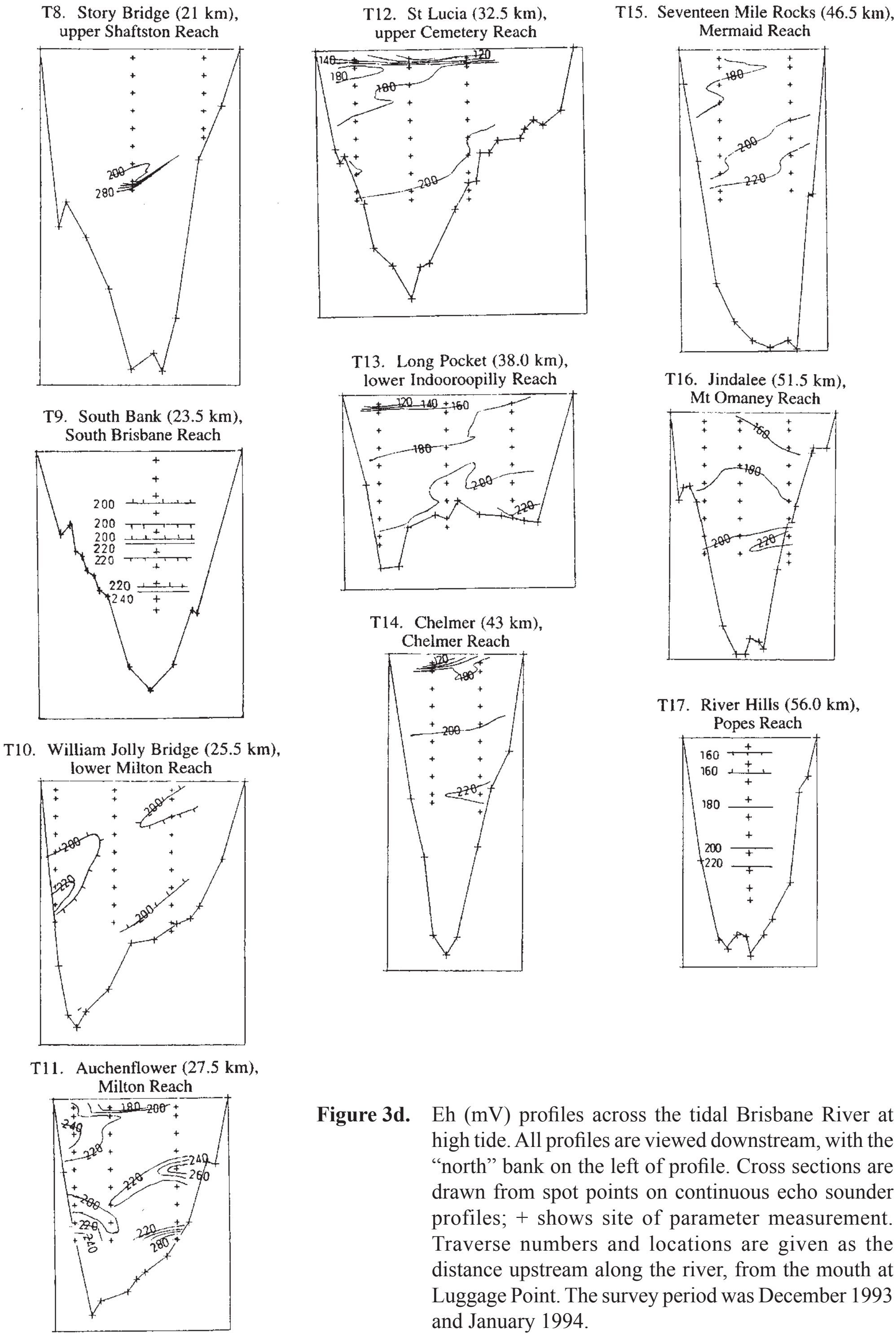
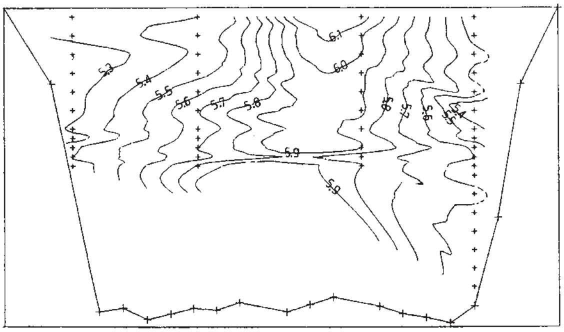


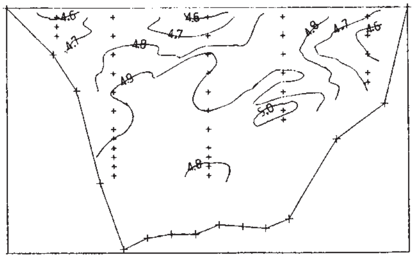
Figure 3d. Eh (mV) profiles across the tidal Brisbane River at high tide. All profiles are viewed downstream, with the "north" bank on the left of profile. Cross sections are drawn from spot points on continuous echo sounder profiles; + shows site of parameter measurement. Traverse numbers and locations are given as the distance upstream along the river, from the mouth at Luggage Point. The survey period was December 1993 and January 1994.

DISSOLVED OXYGEN PROFILES BRISBANE RIVER

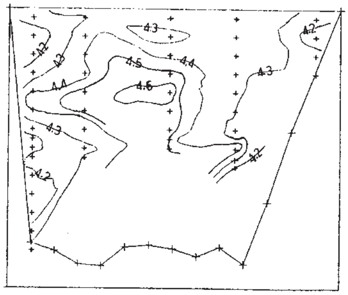
T1. Luggage Point (0.5 km), Lower Reach



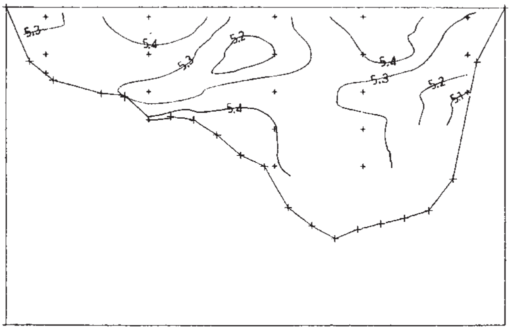
T4. Gateway Bridge (9.5 km), Quarries Reach



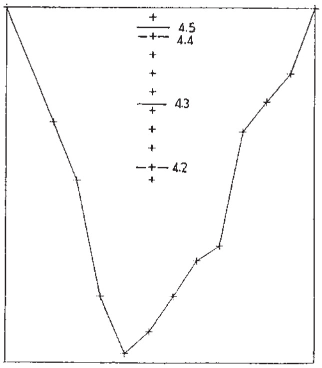
T5. Hamilton (13.0 km), Hamilton Reach



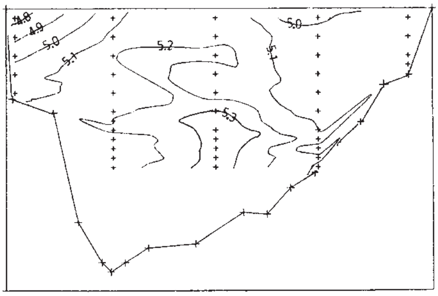
T2. Bulwer Island (3.5 km), lower Lytton Reach



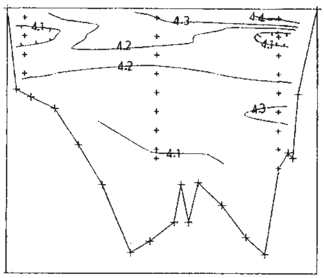
T6. Bulimba Point (15.0 km), lower Hamilton Reach

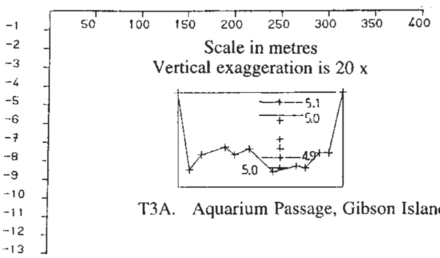


T3. Pinkenba (6.0 km), upper Lytton Reach

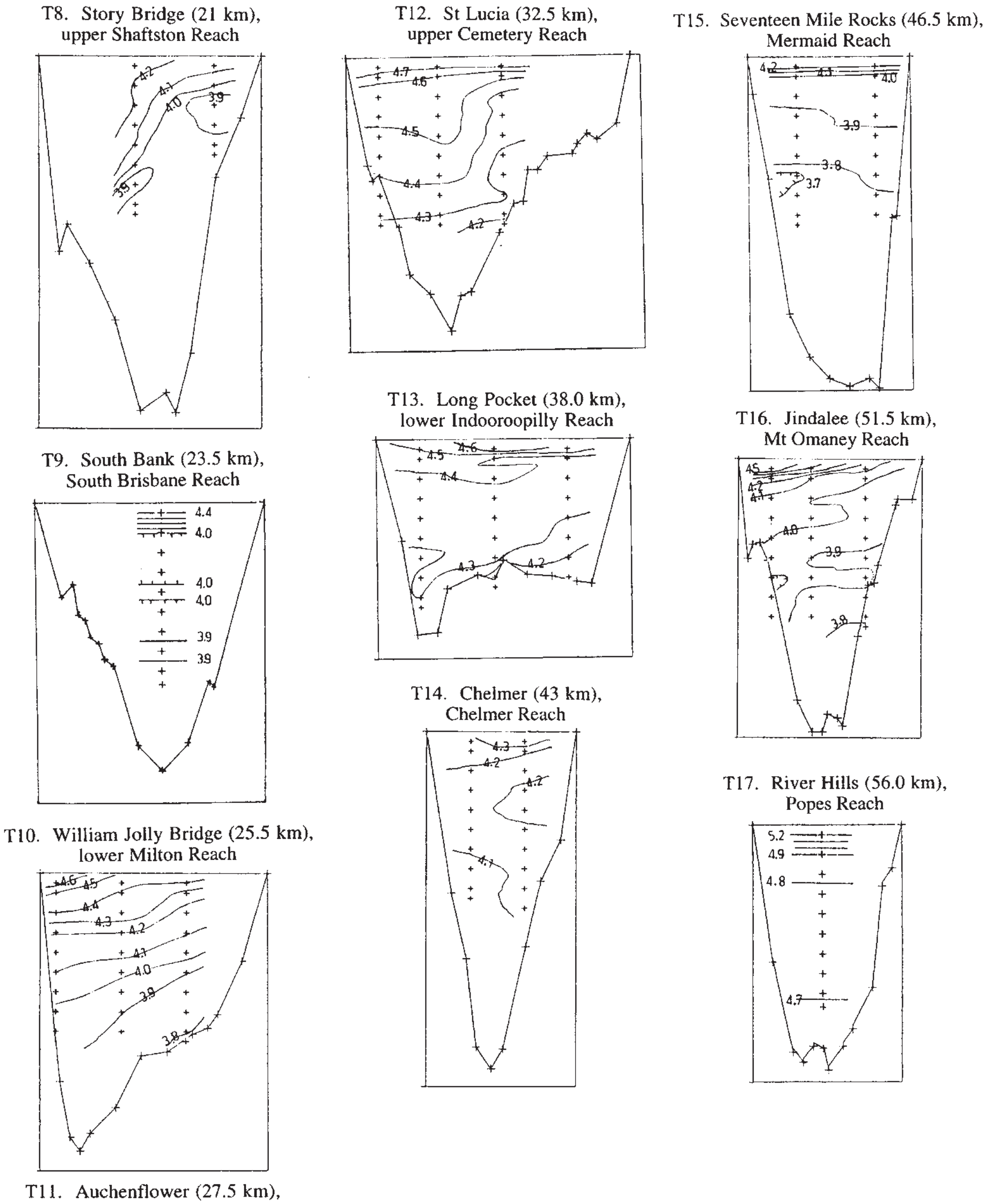


T7. New Farm Park (18.5 km), Humbug Reach





T3A. Aquarium Passage, Gibson Island (6.5 km)



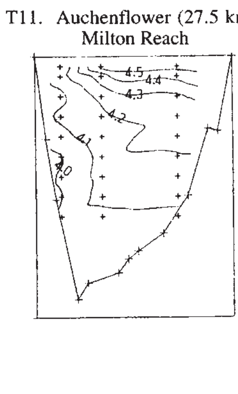


Figure 3e. Dissolved oxygen (mg/L) profiles across the tidal Brisbane River at high tide. All profiles are viewed downstream, with the "north" bank on the left of profile. Cross sections are drawn from spot points on continuous echo sounder profiles; + shows site of parameter measurement. Traverse numbers and locations are given as the distance upstream along the river, from the mouth at Luggage Point. The survey period was December 1993 and January 1994.

deep wedge of salt water, and a distinct deep saltwater mass is only evident near the mouth (T1, T2). On T3 there is lower salinity water from Bulimba Creek on the south side, and at T4 the salinity is clearly stratified. T5 has a definite wedge of lower salinity on the north on the outside of the meander and presumably sourced from Breakfast Creek; this is also present in T6. T7 at New Farm Park has a lower salinity surface layer from park drainage and Norman Creek. At William Jolly Bridge (T10) there is a distinct lower salinity zone on the outer side of the meander, which is also present on the outer side of T11; these features both reflect a faster flow on the outer, northern bank. From T13 upstream, no layering of salinity is evident as a greater vertical component of mixing has occurred.

Temperature

On T1 at the mouth cooler temperatures show the deeper saline wedge; the warmer near-surface water on the north bank is probably in part related to discharge of ponded, treated sewage water (Figure 3b). Temperature in a discharge channel was 24.8^{o}C in August, 1993. Very uniform temperature stratification is evident at T2, with a typical vertical gradient of 0.1^{o}C/m. At T3 the discharge from Aquarium Passage-Bulimba Creek has a strong effect. Cooler water occurs at the north bank of T4 and may reflect groundwater seepage from the golf course on this flat-lying area. Warmer surface layers at T5 are related to Breakfast Creek, and those on the south bank at T7 to Norman Creek. The warmer water of the inner meander at T8 is presumably related to a sand bar and park on that point. A common trend is for warmer waters to occur on the inner side of meanders, which is slower moving, often more saline (except where tributaries discharge); this is the case at T8, T10(?) and T11. At other sites, the temperature profiles reflect the dynamic mixing within the river.

pH and Eh

The pH values measured in the estuarine section are largely controlled by the proportion of marine water, therefore reflects a similar pattern to salinity (Figure 3c cf Figure 3a). Upstream of T8 the distribution is more variable due to turbulence and probably some input from groundwater baseflow. Eh is also related to salinity, but commonly displays a strong increase at depths of 5-7 m and with lower temperatures (Figure 3d). This presumably reflects the aerated nature of marine waters of the bay, and that the concentration of dissolved molecular oxygen is close to saturation relative to the atmosphere (e.g. Mason & Moore, 1982). This more positive Eh is postulated to also reflect the greater salinity and the suspended material of the deeper layer of water. The sewage outfall at Luggage Point had a pH of 6.85 and Eh of 180 mV.

Dissolved oxygen

The longitudinal profile of DO (Figure 2f) shows the lowest values (< 4.0) are in the central section, T8 to T10, which spans the city proper, and T14 to T16, an area of newer suburbs. In both of these zones lower values occur in the deeper layers, but also towards channel sides. In lower traverses (T1-T5) the deeper, more central wedge of marine water is clearly more oxygenated (Figure 3e). The overall distribution pattern of DO is due in part to discharge of poorer quality water from various industrial sites, storm water drains and sewage treatment plants (the sewage outfall at Luggage Point had a DO of 4.1 mg/L). This is evident on cross sections T12 and T15 where there is discharge from sewage plants. The section T14 to T16 is also a zone of higher turbidity, central to which is the depot of dredging operations.

A DO value considered an acceptable standard for the lower river during conditions of low flow, is a minimum of 5 mg/L (Bennett, 1990), which is the minimum oxygen concentration normally considered adequate for fish (e.g. Alabaster & Lloyd, 1982). Razzel (1990) shows intake waters at Mt Crosby typically have values around 2.5 to 5 mg/L DO. Temporal variations do occur, such as DO dropping to near zero in February - March, 1988 following a marked increase in turbidity from storm runoff. Levels of DO have also been found to be dependent on the magnitude of freshwater inflows to the estuary, and that the low flow regime was the critical condition. Reports by the Water Quality Council of Queensland (see Miller & Connell, 1990) show DO values in 1976-1977 as a mean of 3.9 mg/L (range 0.5-6.4), and for 1985-1986 as 6.3 mg/L (4.1-11.9). An oxygen deficiencient zone exists in the Bremer River downstream of Ipswich, particularly under dry conditions; this has been reported as DO (mg/L) of 3-4 at high tide, falling to 2-3 or even to 1 at low tide (Rankin & Milford, 1979b; Steel, 1991).

Implications for a salt wedge

The distribution of salinity shows that under dry conditions the estuary is of a slightly stratified type in its lower part, but is vertically mixed upstream of around 15 km. This distribution is largely a result of longitudinal dispersion by processes such as tide and current. In contrast, the Derwent Estuary in Tasmania for example, is highly stratified, especially in the upper estuary, and has a salt wedge in the lower estuary (e.g. Davies & Kalish, 1994). The amount and seasonal distribution of rainfall to a catchment is therefore of major importance.

In the high flow regimes of the Brisbane River freshwater passes over the top of the body of denser saltwater. The configuration and position of the saltwater body is temporally variable, and is dependent upon the amount of freshwater discharged into the river system. In dry periods the saline water penetrates far upstream, while in extreme floods (such as 1974) the freshwater is pushed well out into Moreton Bay (e.g. Stephenson, 1968; Cossins, 1990). Note the conditions after flooding in April 1963 (Figure 2b) when stratification is more evident. This latter condition was also found to be the case with the flood of May 1996 (e.g. Davies & Eyre, this volume; Moss, this volume).

Conclusions

Comparison of the results of this study (December 1993 - January 1994) with available results of other investigations confirm that at high tide and under low flow conditions there is very little input of freshwater to the tidal section. Longitudinal profiles and cross-sections show that the character of the estuary at the time of the survey is of a slightly stratified type and that formation of a distinct salt wedge is spatially limited. Cross-sections show some horizontal stratification of salinity in the lower 10 km and that the upper part of the estuary, from 30 to over 70 km, is well mixed and stratification is longitudinal (i.e. vertical).

Redox values are of note and display the strongly oxidised nature of the Bay water relative to near-surface river water. Dissolved oxygen content has apparently improved from previous years, but there is a broad low DO zone (< 4.5 mg/L) 13 to 55 km upstream, which is related to various point source industrial and sewage discharges, but does appear in part related to dredging induced turbidity and washing of dredged material. The lowest DO values characteristically occur in the deeper parts of the channel and along the sides.

Acknowledgements

The study was carried out under Queensland University of Technology Research Grant NRGS93.FRM. I am grateful to the Port Authority of Brisbane for its cooperation, making available a suitable boat and skipper, and providing access to their shore facility. Valuable assistance during the survey and analytical part of the work was provided by Rob Virtue. The assistance of John Laycock in computer contouring the cross-section data is greatly appreciated.

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Tingalpa Creek Water Quality Study

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Abstract

Following an extensive study of the Tingalpa Creek estuary in southeast Queensland in 1991-1992 (John Wilson and Partners Pty Ltd, 1994), the Department of Environment and Redland Shire Council have continued a monitoring program to assess the impacts of point source discharges on the water quality and biota of the creek. A number of stresses are imposed on this creek in the southern Moreton Bay catchment. The estuarine reaches drain a predominantly urban area and receive effluent discharges from two sewage treatment plants and a canning factory. The creek is dammed for urban water supply at 11.5 km from the mouth. The monitoring program consists of a number of components. Two four-week periods of intensive water quality sampling have been carried out in winter and summer each year. This part of the program aims to assess the nutrient status of the estuary and its impact on the plankton community. The associated plankton study has mapped the changes in abundance of key groups of phytoplankton and zooplankton, and related these to changes in water quality. In addition to the intensive surveys, monthly surveys have been carried out at fewer sites to determine trends in ambient water quality. During the monthly surveys, water samples have been collected from Tingalpa Creek as part of a joint study with The University of Queensland to document the phytoplankton communities in the estuarine creeks of Redland Shire. This program is aimed specifically at identifying the taxa present, their seasonal changes in abundance and species diversity, and response to the nutrient status of the waterway.

Introduction

The Department of Environment has carried out a water quality monitoring program in Tingalpa Creek each summer and winter since November/December 1994 as part of a continuing monitoring program with Redland Shire Council. Surveys were carried out in August 1991, April/May 1992, July/August 1992 as part of an intensive study of the creek (John Wilson and Partners Pty Ltd, 1994). The initial study was used to determine the health of the estuarine ecosystem, develop and calibrate a hydrodynamic model of the estuary and develop strategies for future wastewater management in the catchment.

The monitoring program aims to:

- monitor the status of ecosystem health; and
- assess localised impacts of wastewater disposal on the ecosystem.

Specific monitoring objectives of each intensive survey aim to:

- detect changes in water quality parameters since the previous surveys; and
- detect changes in populations of plankton.

This paper describes the components of the long-term study, discusses briefly some of the findings to date and draws some conclusions about the status of ecosystem health and impacts of wastewater disposal.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 203-212.

Description of study area

Tingalpa Creek forms part of the boundary between Redland Shire and Brisbane City Council in southeast Queensland and drains into southern Moreton Bay (see Figure 1). The overall length of Tingalpa Creek is approximately 24 km with approximately 11.5 km of this being estuarine downstream from the Leslie Harrison Dam (Redland Shire potable water supply). The creek receives occasional fresh water releases from the dam when water levels are high. The estuarine part of the creek drains a mostly urban catchment with some industry, flower and turf farms and a golf course. Coolnwynpin Creek, a significant tributary, enters at 8.7 km from the mouth.

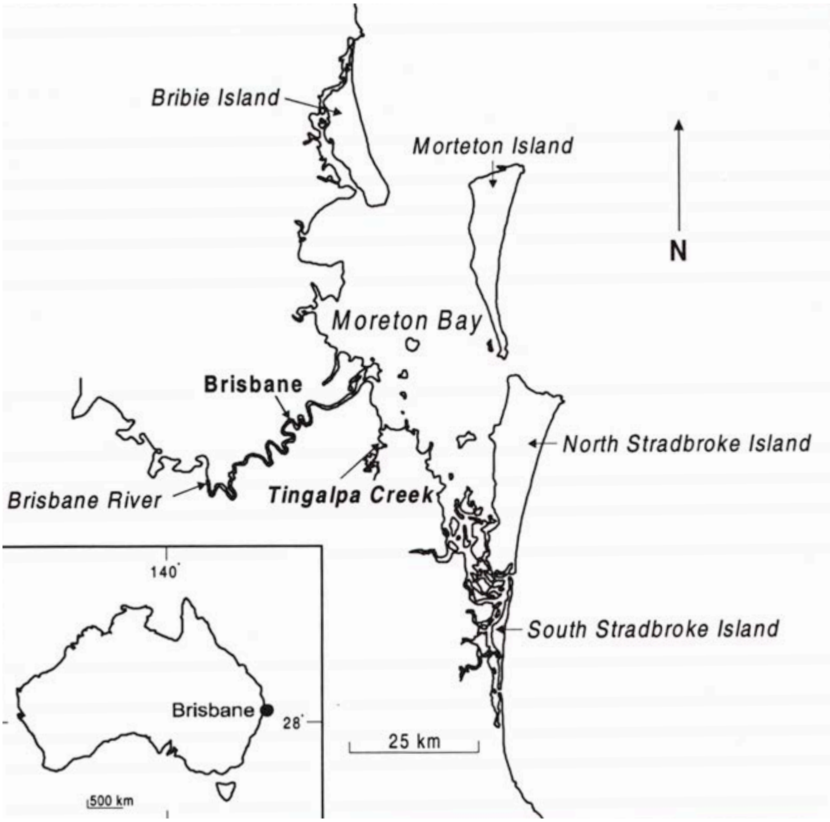


Figure 1. Location of Tingalpa Creek in southeast Queensland. (Figure courtesy

Eleven sampling sites are located on Tingalpa Creek and one on Coolnwynpin Creek. Sites are identified by the distance from the mouth in kilometres. This is known as the Adopted Middle Thread Distance (AMTD), e.g. 1.2 km AMTD. Sites are also identified by a site code, e.g. TGC-2.

Current point source discharges into Tingalpa Creek include:

(i) Thorneside sewage treatment plant discharging a daily maximum of 5 400 m^3 at approximately 1.3 km AMTD;

(ii) wastewater discharge from the maturation lagoon at Edgell's Brisbane food processing plant discharging a daily maximum of 3 300 m^3 at approximately 1.9 km AMTD; and

(iii) Capalaba sewage treatment plant discharging a daily maximum of 10 000 m^3 at approximately 8.7 km AMTD.

Before 1994, the potable water treatment plant at 11.5 km AMTD had an additional point source discharge. Backwash water and sludge from this plant are now disposed to sewer and landfill. A small sewage treatment plant at Greenacres Caravan Park discharges into a tributary of Coolnwynpin Creek which drains into Tingalpa Creek at approximately AMTD 8.7 km. A number of stormwater drains also discharge into Tingalpa Creek. The middle and lower estuarine reaches are used for boating, canoeing and fishing.

Methodology

Intensive water quality monitoring program

Since 1994, intensive water quality sampling has been carried out each winter and summer. Each sampling period includes four surveys at weekly intervals. This part of the program aims to assess the physico-chemical characteristics and nutrient status of the estuary and the impact of point source discharges on the water quality. Table 1 lists the sites by site code, km AMTD

Table 1. Sites by site code, kilometres AMTD and parameters measured at each site in Tingalpa Creek. All sites are estuarine.

Code No.	Location	Parameters measured			
	(AMTD)	Suspended solids	Nutrients	Chlorophyll a	Faecal coliforms
TGC-1	0.0 ✓	✓	✓	✓	
TGC-2	1.2	✓	✓	✓	✓
TGC-3	1.7	✓	✓	✓	
TGC-4	2.4	✓	✓	✓	
TGC-5	3.8	✓	✓	✓	
TGC-6	5.9	✓	✓	✓	
TGC-7	7.6	✓	✓	✓	
TGC-8	8.5	✓	✓	✓	
TGC-9	9.6	✓	✓	✓	
TGC-10	10.4	✓	✓	✓	
TGC-11	11.0	✓	✓	✓	

and the parameters measured at each site. Figure 2 shows the twelve sites at which sampling is undertaken in the Tingalpa Creek water quality study and point source discharge locations.

Field measurements of pH, temperature, conductivity, turbidity and dissolved oxygen were carried out at all sites. Readings were taken near the surface (0.2 m) and at 1 m intervals to the bottom. Secchi depth was measured at all sites. All samples were collected at midstream collection points commencing at 11.0 km AMTD on an outgoing tide when possible, and analysed for the physical and chemical parameters shown in Table 2. Triplicate samples were

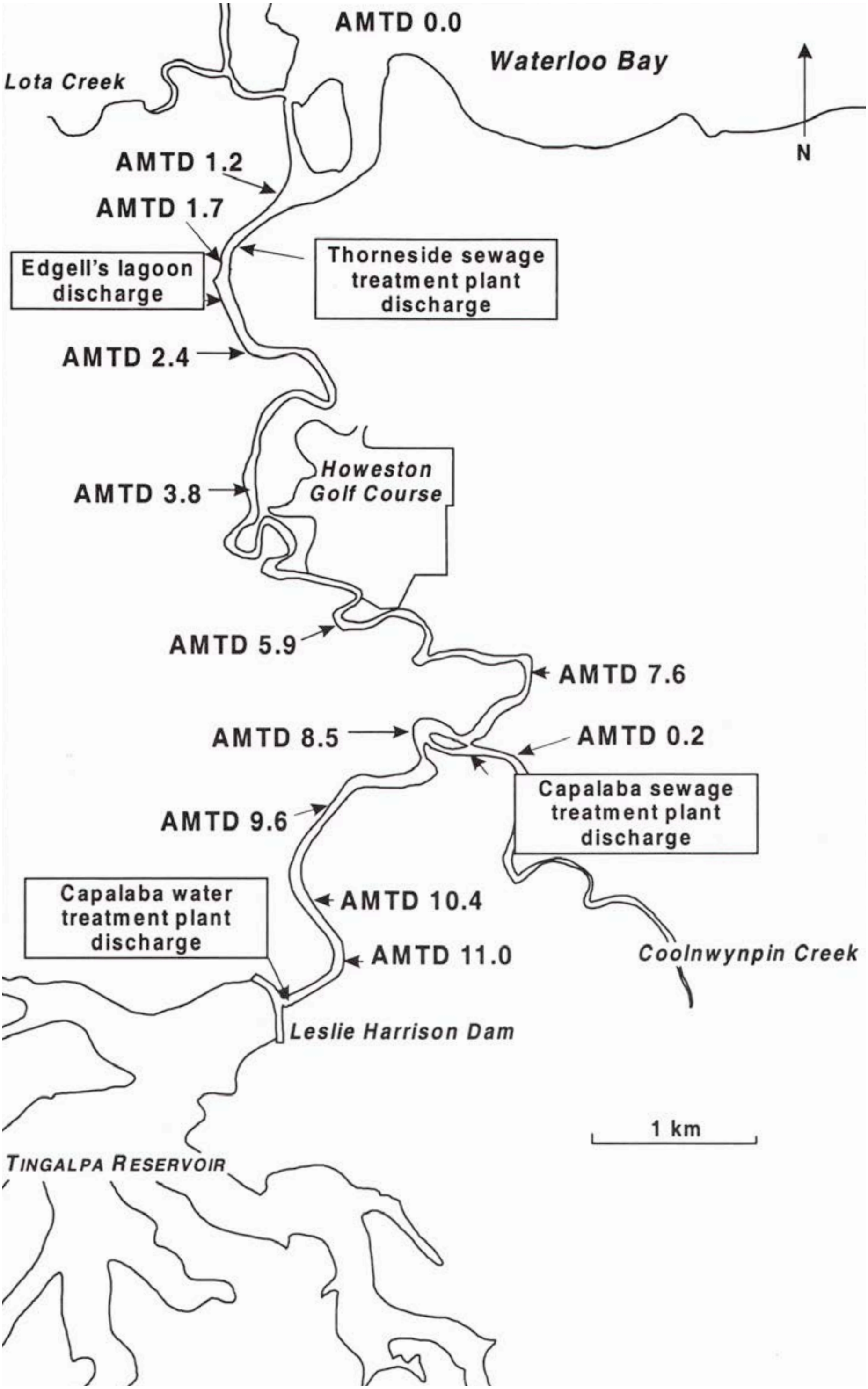


Figure 2. Location of Tingalpa Creek estuary sampling sites and discharge points (boxes) with distances from the mouth in kilometres AMTD. (Figure courtesy P. Uwins.)

Table 2. Parameters measured in the Tingalpa Creek study.

***In situ* measurements**	**Laboratory analysis**
pH	suspended solids
temperature	chlorophyll *a*
turbidity	phosphorus (total)
dissolved oxygen	phosphorus (dissolved reactive)
conductivity	nitrogen (oxidised)
secchi depth	nitrogen (ammonia)
	nitrogen (organic)

Plankton study

During the intensive water quality monitoring surveys twice a year, unfiltered water samples were collected at the same sites for plankton identification and counts. These samples were delivered chilled and unfixed within 24 h of collection to the laboratory. Data from these counts were used to map changes in abundance of key groups of phytoplankton and zooplankton. This information is used to assess the health of the plankton community and correlate this with physico-chemical water quality. One sample was also collected at each of the Thorneside sewage treatment plant, Edgell's lagoon and Capalaba sewage treatment plant to assess the biological quality of the effluent.

To date plankton analysis has been performed by B. Bowles of Microcosm Consulting Pty Ltd (Melbourne) for Redland Shire Council. However, negotiations are under way with The University of Queensland to perform this work.

Ambient water quality monitoring

In November 1995, seven long-term monitoring sites were set up in Tingalpa Creek as part of the state-wide ambient water quality monitoring program. These sites coincide with seven of the sites from the intensive monitoring program. Sites adjacent to discharges have not been included in the ambient monitoring as water quality in the mixing zone is highly variable and does not reflect the general water quality of the stream. Monthly surveys were carried out at these seven sites in the estuary to assess long-term trends in ambient water quality. Samples were collected in the same way as the intensive monitoring data and analysed for the same parameters. Data were analysed using the Seasonal Kendall test to identify trends.

Phytoplankton characterisation

During the monthly surveys, water samples were collected from Tingalpa Creek as part of a joint study with The University of Queensland to document the phytoplankton communities in Redland Shire estuarine waterways. This program is aimed at identifying the taxa present, their seasonal changes in abundance and species diversity, and response to the nutrient status of the waterway. A possible outcome of this base-line plankton study is identification of plankton species which could become a future indicator species for monitoring water quality.

Water quality criteria

Guidelines published by the Australian and New Zealand Environment and Conservation Council (ANZECC) (1992) were used to describe the health of the waterway in terms of physico-chemical characteristics. These guidelines, however, do not specify levels for nutrients, the parameters of most interest in the case of Tingalpa Creek. A single set of nitrogen and phosphorus concentrations to prevent nuisance algal problems cannot be recommended because many other factors, e.g. light attenuation, can also limit nuisance growth. The guidelines provide a concentration range, primarily as an indication of levels at or above which problems have been known to occur, dependent on a range of other limiting factors.

Results and Discussion

Physico-chemical water quality

Dissolved oxygen

Dissolved oxygen levels are markedly variable and are generally lower in the middle and upper reaches of the estuary where the discharge from Capalaba sewage treatment plant has its greatest impact. Measurements of temperature and conductivity through the depth profile indicate that the water is well mixed. Figure 3 shows the surface level dissolved oxygen for sites included in the ambient water quality monitoring program for 1995-96 as box-and-whisker plots.

The top of the box is the 75th percentile, the middle line is the median and the bottom of the box is the 25th percentile. The whiskers extend to those data points which fall up to one-and-a-half times the height of the box either above or below the box. The asterisks represent points falling up to three times the height of the box either above or below the box, and the far outlying values are shown as circles.

Figure 3 shows that surface level dissolved oxygen in the middle and upper reaches is generally lower than the ANZECC-recommended level of 80-90% saturation due to the organic pollution from the various wastewater discharges. Department of Environment data indicate that levels of dissolved oxygen in south-east Queensland streams often range from 60-80% saturation, even in relatively undisturbed streams. However, Tingalpa Creek levels regularly fall below 50% saturation in the middle reach. Although there are two discharges of wastewater in the lower reach of the estuary, dissolved oxygen levels were within the recommended ANZECC range due to the high level of tidal exchange in the lower estuary.

Nutrient status

ANZECC guidelines do not recommend a concentration range for levels of total phosphorus or total nitrogen in estuaries or marine waters, however, a range is recommended for dissolved inorganic nutrients.

Figures 4 and 5 show the total nitrogen and phosphorus levels for the sampling period November 1995 to December 1996. The impact of the wastewater discharge from Capalaba sewage treatment plant is evident in the high nutrient levels in the middle to upper reach. Levels of all nutrients decreased downstream of the discharge location and dissolved inorganic nutrients were within the ANZECC-recommended ranges in the lower estuary.

The dissolved inorganic nitrogen levels followed a similar profile to that of total nitrogen. However, organic nitrogen levels increased from the mouth to the upper estuary. The higher organic nitrogen levels in the upper estuary are due to high levels of algae and organic matter at the former disposal site for backwash material from the potable water treatment plant.

Figure 6 shows the mean concentrations of total phosphorus for six intensive monitoring surveys for 1991-1996 as a series of line graphs joining the mean values for each sampling period of four surveys a week apart. Differences between phosphorus concentrations at each site during the winter surveys were tested using the Mann-Whitney U Test which showed there is no significant difference ($P > 0.05$) in values at sites in the lower or upper estuary. Phosphorus levels, however, have increased slightly in recent surveys in the middle estuary. Only winter data were used in this instance to eliminate the influence of seasonal variation.

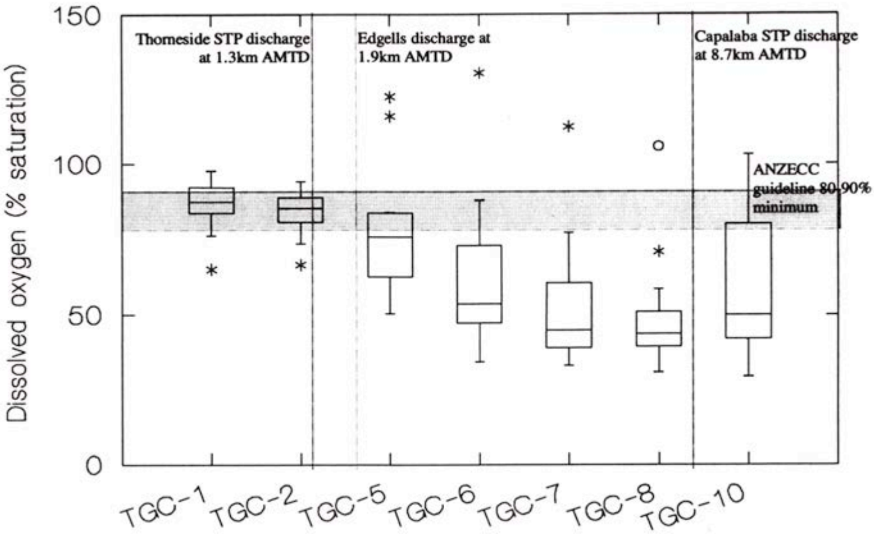


Figure 3. Surface level dissolved oxygen for 1995-96 at the seven Tingalpa Creek sites included in the ambient water quality monitoring program. The box-and-whisker plots in this graph are based on fourteen data points from monthly surveys.

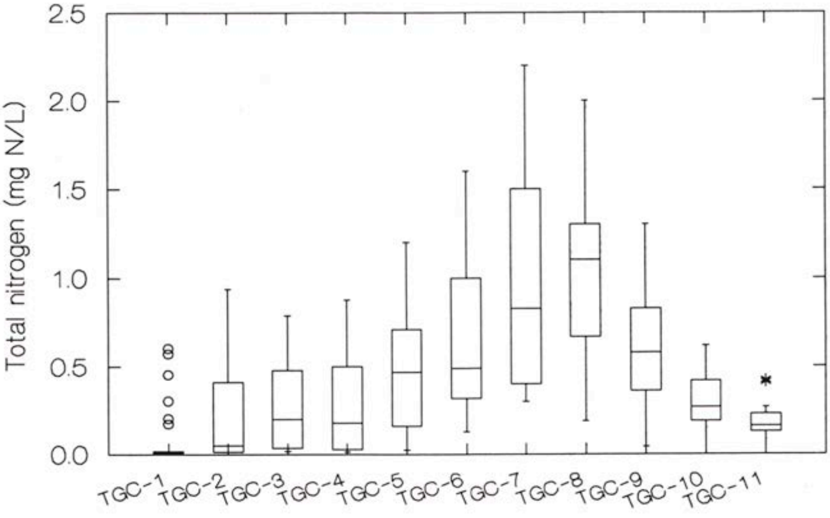
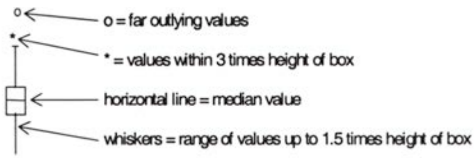


Figure 4. Total nitrogen levels for all data collected in 1995-96 at the Tingalpa Creek sites. The box-and-whisker plots in this graph are based on 30 data points from monthly and twice yearly intensive surveys.



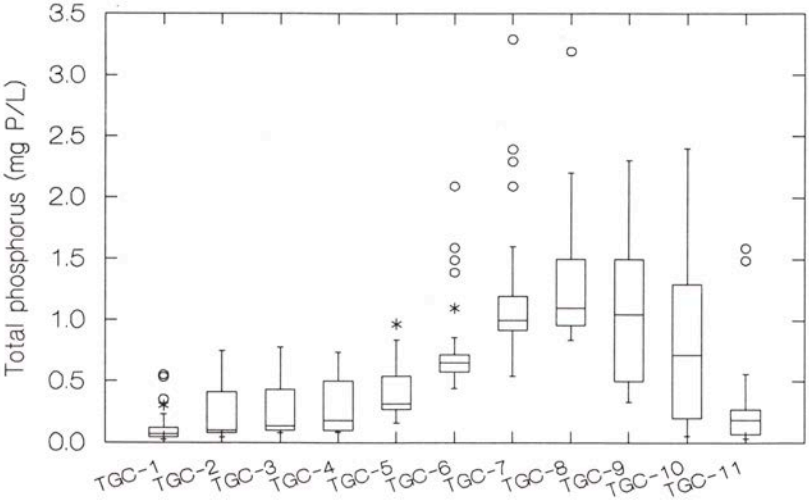


Figure 5. Total phosphorus levels for all data collected in 1995-96 at the Tingalpa Creek sites. The box-and-whisker plots in this graph are based on 30 data points from monthly and twice yearly intensive surveys.

Higher phosphorus levels would be expected in the vicinity of the effluent discharge as phosphorus is not removed at the Capalaba sewage treatment plant at present and the population has increased over the last five years. Generally, potential nuisance algal growth in local marine waters is limited by nitrogen availability rather than phosphorus availability (Dennison, pers. comm., Sinclair Knight, 1994). Hence, elevated levels of nitrogen are of more concern than phosphorus.

Figure 7 shows the mean concentrations of total nitrogen for six intensive monitoring surveys for 1991-1996 as a series of line graphs joining the mean values for each sampling period of four surveys a week apart. The impact of the discharge from Capalaba sewage treatment plant is evident at site 8.5 km AMTD. Differences between data from intensive surveys were tested using the Mann-Whitney U Test and showed there has been a significant decrease (P= 0.002) in total nitrogen levels since nitrogen removal processes were installed at Capalaba sewage treatment plant. The graph shows the significant decrease in total nitrogen at the discharge location. Higher levels during the June/July 1995 sampling period were caused by operational problems at the sewage treatment plant for the first two surveys of this period.

Biological water quality

Chlorophyll *a*

Algal problems in estuarine areas occur generally in upper and lower reaches where turbidity is lower (Dennison, pers. comm., Department of Environment and Heritage, 1993). In Tingalpa Creek chlorophyll *a* levels increased from the mouth to the upper estuary, being dominated in the low to middle reach by high inputs of blue-green algae, predominantly *Microcystsis aeruginosa*, from Edgell's lagoon and in the upper reaches by occasional large blooms of very small green algae (picoplankton).

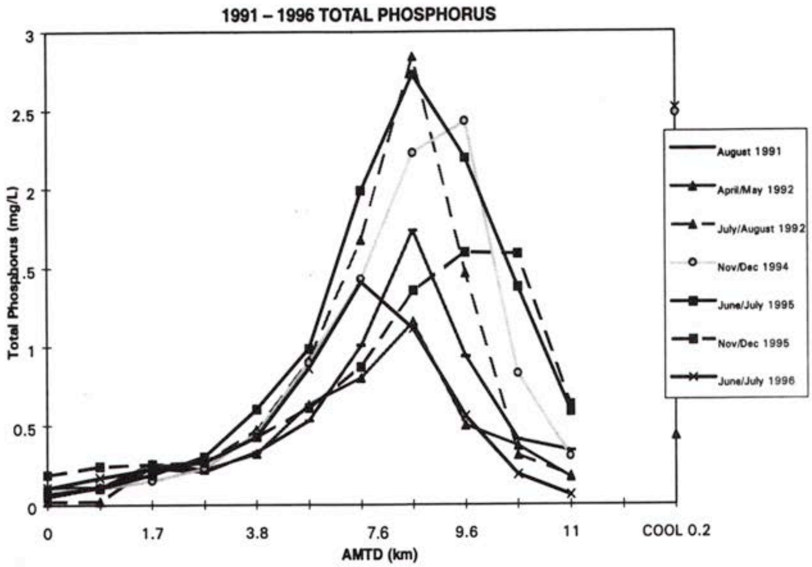


Figure 6. Total phosphorus levels for six intensive monitoring surveys for 1991-1996 based on mean values for each sampling period. COOL = Coolwynpin Creek.

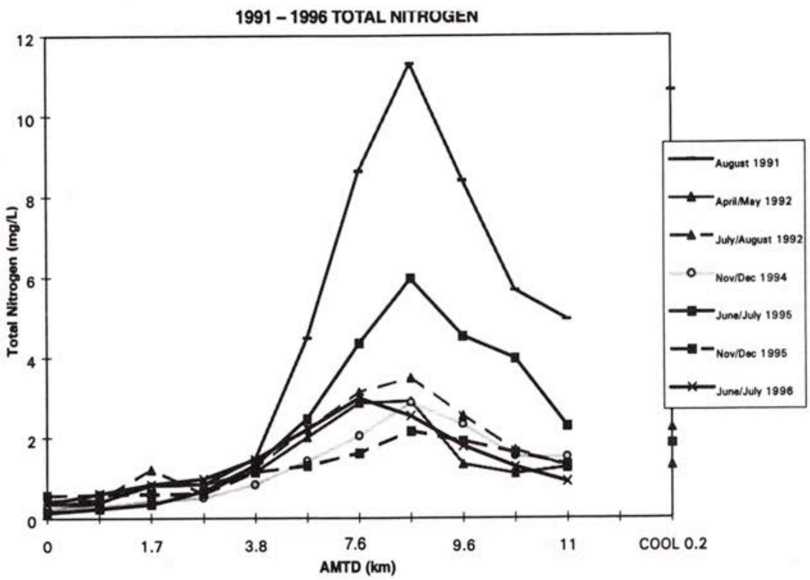


Figure 7. Total nitrogen levels for six intensive monitoring surveys for 1991-1996 based on mean values for each sampling period. COOL = Coolwynpin Creek.

Plankton study

In recent surveys many more oceanic and neritic diatoms have been observed, indicating a stronger marine influence, and a cleaner estuarine zone, allowing better survival of these organisms (Bowles, 1995).

Since 1991 populations of large benthic diatoms which occurred between 3.8 and 7.6 km AMTD have decreased. Short lived blooms of non-toxic dinoflagellates *Peridinopsis* and *Exuviella* and the diatom *Skeletonema* seem to have replaced the large benthic diatoms (Bowles, 1995). This change from benthic organisms to marine/estuarine plankton suggests that a major change has taken place in the creek. The change might result from an improvement in this area of the creek and might be permanent.

Bowles (1995) noted that the contribution of these dinoflagellate blooms to chlorophyll *a* levels would be low. The genera present are heterotrophic feeders and often have very little chlorophyll *a*.

Occasional dense algal blooms observed in the upper estuary (Bowles, 1995) have been identified as very small organisms (picoplankton), probably supported by residual nutrients from the potable water treatment plant backwash material.

Conclusions and Recommendations

While this creek suffers the stresses of being in an urban catchment and receiving discharges of urban wastewater, water quality seems to have improved since the study commenced. However, the changes cannot yet be attributed to better management rather than natural events such as tidal and climatic influences.

While nutrient levels are higher and dissolved oxygen levels are lower in Tingalpa Creek than the recommended range suggested in the ANZECC Guidelines (1992), the diverse plankton community is indicative of a healthy estuarine ecosystem (Bowles, 1995). Generally, sampling results indicate water quality in the creek, especially nitrogen levels, has continued to improve.

The beneficial effects of removing nutrients from discharge effluents might take much time to become noticeable as past discharges have left a large reserve of residual nutrient rich sediment in the creek and the creek is poorly flushed due to the upstream impoundment.

Conditions at 11 km AMTD have continued to improve since the potable water treatment plant backwash/sludge discharge was removed.

The nutrient status of Tingalpa Creek can be expected to continue to improve. A new biological-nutrient-removal sewage treatment plant at Thorneside has been completed recently and work has started on building a new biological-nutrient-removal sewage treatment plant at Capalaba.

The physical structure and flow regime of the creek appear to be very important in preventing nuisance algal blooms and maintaining the current healthy ecosystem. The building of any structures inhibiting or changing the flow regime could be expected to cause algal problems.

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The Use of the Seagrass, *Zostera capricorni,* to Identify Anthropogenic Nutrient Sources in Moreton Bay

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Abstract

Fertilisation experiments in eastern Moreton Bay have established that the growth of the dominant seagrass species (*Zostera capricorni* Aschers.) is limited by the supply of both N and P. The addition of nutrients to seagrass sediments also resulted in changes in the canopy height, shoot density, biomass, growth, tissue nutrient content, amino acid concentrations and $\delta^{15}N$ values of *Z. capricorni*. These morphological and physiological characteristics were compared between sites in Moreton Bay, close to and distant from nutrient sources to determine if they can be used to identify natural gradients of nutrient availability. Neither sediment nutrient concentration (NH_4^+ and PO_4^{-3}), seagrass morphology nor growth demonstrated a consistent trend relative to the proximity of a nutrient source. However, *Z. capricorni* from the four sites close to nutrient sources (sewage, septic, prawn farm effluent and river discharge), demonstrated physiological characteristics (nutrient content and amino acid concentrations) indicative of high nutrient availability (similar to fertilised seagrass), whereas *Z. capricorni* at the five sites distant from nutrient sources had physiological characteristics indicative of low nutrient availability. The tissue nutrient content (%N and %P) of *Z. capricorni* leaves from plants located close to a nutrient source were 50 - 100% higher than nutrient concentrations in leaves distant from a nutrient source. The amino acids glutamine and asparagine were the most responsive to elevated nutrient availability, with concentrations five times higher at sites close to nutrient sources relative to distant sites, whereas $\delta^{15}N$ values of *Z. capricorni* reflected the source of N rather than the nutrient load. The results from this study demonstrate that physiological characteristics of *Z. capricorni* can be used to identify both the load and source of nutrients affecting marine ecosystems.

Introduction

Seagrasses are marine angiosperms which occur in both intertidal and subtidal photic environments. They generally have high rates of areal productivity (in Moreton Bay up to 6.4 g C m^{-2}/day; Udy & Dennison, 1997a) which can be limited by the availability of nitrogen, phosphorus or iron (Dennison *et al.,* 1987; Short, 1987; Short *et al.,* 1990; 1993; Bulthuis, *et al.,* 1992; Fourqurean *et al.,* 1992*a*; Kenworthy & Fonseca, 1992; Duarte *et al.,* 1995). However, high nutrient loads can also be detrimental to the growth and distribution of seagrasses by indirectly reducing light availability by stimulating phytoplankton and epiphyte growth (Tomasko & Lapointe 1991; Short *et al.,* 1995; Taylor *et al.,* 1995; Abal & Dennison, 1996).

For seagrasses, high nutrient availability has been correlated with increased leaf size, higher shoot densities, growth rates, tissue nutrients and amino acid concentrations (Orth, 1977; Short, 1983; Duarte, 1990; Short *et al.,* 1990; Bulthuis *et al.,* 1992; van Lent *et al.,* 1995; Udy & Dennison, 1997a,b). Seagrass distribution, growth, morphology and physiology can, however, also be affected by light availability in Moreton Bay (Abal, 1996; Abal & Dennison, 1996). Reducing the light available to seagrasses can result in morphological and physiological changes such as increases in leaf length, tissue nutrient content (%N, %P) and amino acid concentrations similar to those that result from increases in nutrient availability (Abal 1996; Udy & Dennison, 1997a).

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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The relative abundance of the ^{15}N isotope, $\delta^{15}N$, compared to an atmospheric standard, has been used to trace the movement of nitrogen through ecosystems (Schimel, 1993) and to estimate the contribution of different nitrogen sources to the growth of plants (Lajtha & Marshall, 1994). The effectiveness of this technique relies on minimal discrimination between nitrogen isotopes (^{14}N, ^{15}N) occurring during their acquisition and incorporation by the plant. Sources of anthropogenic N, such as sewage, result in higher plant $\delta^{15}N$ values (close to 10‰) and N from N_2 fixation results in lower $\delta^{15}N$ values (close to 0‰) (Lajtha & Marshall, 1994; Grice *et al.,* 1996).

Moreton Bay supported about 25 000 ha of seagrass meadows in 1987 (Hyland *et al*., 1989), however, Abal & Dennison (1996) have suggested that this area is declining because of continuing degradation in water quality, especially in the southern and western Bay. Moreton Bay is bordered by the mainland on the western side and three large sand islands on the eastern side; Moreton, North Stradbroke and South Stradbroke Islands (Figure 1). Nutrient input to Moreton Bay reflects the development focus on the western side where nutrients come from agriculture and urban sources, whereas the eastern side receives low-nutrient oceanic water and few anthropogenic nutrients. The growth of some seagrasses in the eastern Bay is nutrient limited, with *Halodule uninervis* being exclusively N limited and *Z. capricorni* being co-limited by N+P (Udy & Dennison, 1997a). This suggests that certain morphological and physiological characteristics of seagrasses (described above) may be useful in determining regions of Moreton Bay that are exposed to high nutrient loads.

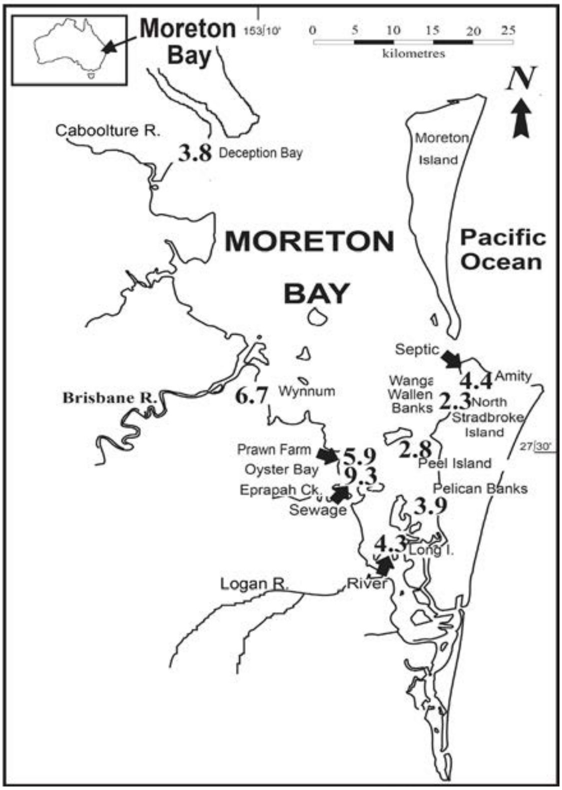

Figure 1. Moreton Bay, Queensland, Australia, showing the $\delta^{15}N$ values of *Zostera capricorni* from nine study sites and the likely source of nutrient input (➔) for anthropogenic point sources (Modified from Udy & Dennison, 1997b).

The present study aimed to identify morphological and physiological characteristics of *Z. capricorni* that can be used to indicate the nutrient availability to this seagrass at different sites around Moreton Bay. Characteristics of *Z. capricorni* (growth, morphology and physiology) that change in response to nutrient additions in field experiments (Udy & Dennison, 1997a) are compared at four sites close to nutrient sources (sewage, septic and prawn farm effluent, river discharge) and four sites distant from nutrient sources (Udy & Dennison, 1997b). Both studies were conducted in December (1994 and 1995) as preliminary research (unpublished) suggested that December is a period of heightened growth for *Z. capricorni* in Moreton Bay, which should result in maximum differences between sites with different nutrient loads.

Materials and Methods

Study sites

Fertilisation experiment

The fertilisation experiment was conducted in December 1994 on the Wanga Wallen banks (153°22' E, 27°25' S), off Amity Point on North Stradbroke Island. These banks consist of a shallow, subtidal lagoon (0-1.5 m deep at mean low water) fringed on the eastern side by intertidal *Zostera capricorni* Aschers. and mangroves and on the western side by an unvegetated intertidal sand bar. The Wanga Wallen banks are comprised of predominantly sandy siliceous sediment and receive oceanic water on a semi-diurnal tidal cycle. Seagrasses were fertilised *in situ* using slow release fertiliser in September 1994 and sampled in December 1994 (5-6 month release Osmocote with phosphorus as 18% PO_4^{-3} and nitrogen as 11.5% NH_4^+ and 11.5% NO_3^- Scotts Australia). Three replicates of four treatments were used; Control (C) – sediment disturbed but no fertiliser added, phosphorus (+P; 23 g P/m^2), nitrogen (+N; 88 g N/m^2) and nitrogen plus phosphorus (N+P; 23 g P/m^2, 88 g N/m^2). These treatments were randomly assigned to 12 circles (1 m diameter) located along a line of equal water depth. The surface sediment was removed from each circle by gentle hand-generated water currents until rhizomes were visible. Granules of fertiliser were then administered (as above) to the rhizomes and between 5 and 10 mm of sediment was then added to recover the rhizomes and the fertiliser.

In December 1995 *Z. capricorni* was sampled from an additional eight sites in Moreton Bay to test if the growth, morphology or physiology of local seagrass communities reflected the nutrient availability in their environments (Figure 1). All sites examined were subtidal except for Eprapah Creek, which was intertidal. Four sites were chosen for their proximity to specific nutrient sources. These were:

- Oyster Bay, located at the mouth of a small creek which receives up to 3 000 m^3/d of prawn farm effluent (2.0 mg N/L, 0.2 mg P/L; Jones, unpublished);
- Eprapah Creek, located at the mouth of this large creek which receives 2 400 m^3/d of sewage effluent (4.5 mg N/L; Redland Shire Council);
- Amity, a small creek next to the Amity Point township which receives nutrients from local septic tanks and the surrounding mangrove forest; and
- Long Island, the first site north of the Logan River where seagrass currently occur (Abal & Dennison, 1996), receives chronic loadings of total suspended solids and nutrients from the Logan River plume.

The remaining four sites were in areas of western and eastern Moreton Bay which had no known adjacent anthropogenic nutrient point sources. These were:

- Pelican Banks, in the eastern Bay distant (> 10 km) from any urban settlement and anthropogenic nutrient sources;
- Wynnum, in the western Bay next to a medium density urban settlement, probably affected by rain events, but distant from any constant nutrient point sources;
- Deception Bay, in the north western Bay, distant from (> 5 km) any urban settlements and anthropogenic nutrient sources, receiving low nutrient oceanic water from North Passage; and
- Peel Island, in the central Bay distant from urban settlements (~ 5 km) and anthropogenic nutrient sources, receiving low nutrient surface water from South Passage.

Sediment nutrients

Biologically available sediment nutrients at each site were determined by measuring two nutrient pools: (1) nutrients dissolved in the sediment porewater, which are immediately available to the seagrass roots; and (2), nutrients adsorbed to sediment particles, which are assumed to be in equilibrium with the dissolved nutrients and hence potentially available to seagrass.

To sample porewater a "sipper", constructed of an external PVC pipe and an inner 10 μm screen (Udy & Dennison, 1996), was inserted into the sediment to a depth of 10 cm to collect *in situ* porewater samples. Nine replicate samples from each treatment were collected in the fertilisation experiment (Udy & Dennison, 1997a) and 10 replicate samples were collected from the other sites (Udy & Dennison, 1997b). The samples were collected in acid-washed, N_2 purged, evacuated serum bottles, immediately placed on ice and frozen within 4 h of sampling. These samples were later analysed colorimetrically for [NH_4^+] and dissolved reactive [PO_4^{-3}] (Parsons *et al.,* 1989).

In the fertilisation experiment, one sediment core was taken from the top 10 cm of each replicated plot, whilst four replicate sediment cores were obtained at the eight sites, subject to ambient nutrient concentrations. Cores were placed on ice and returned to the laboratory within 2 h of collection for extraction of the adsorbed nutrient fraction. The sediment from each core was mixed into a homogeneous slurry, with subsamples being taken for adsorbed NH_4^+, adsorbed PO_4^{-3}, and porosity (see Udy & Dennison, 1996; 1997a).

Biomass, morphology and shoot density

Three 15 cm diameter cores, from each site or nutrient treatment, were collected from the middle of the *Z. capricorni* depth range for determination of *Z. capricorni* biomass, canopy height and shoot density. Seagrass tissue was separated into new leaves (the youngest leaf, plus the bottom section of the second youngest leaf required to make up the equivalent of one full size leaf), old leaves (all other leaf tissue), rhizomes and roots, dried (60°C for 3 d) and weighed. The longest five leaves in each core were measured to estimate seagrass canopy height and the number of shoots counted to estimate shoot density.

Growth rates

Growth rates of *Z. capricorni* were determined using the leaf hole punch technique (Dennison, 1990). Twenty shoots from three replicate areas at each site were leaf hole punched and recovered one week later to measure growth (g/shoot/d). These growth rates were converted to areal growth rates (g/m/d) using the mean number of shoots present at each site.

Tissue nutrient content and isotopic analysis

Biomass samples were ground to a fine powder in a vibratory ball mill (Retsch MM-2, Haan, Germany). Subsamples were then used to determine tissue nutrient content and $\delta^{15}N$. Percent nitrogen (N) and phosphorus (P) were determined for new leaves and rhizomes by digesting 200 mg of dry, ground tissue using a Kjeldahl digest (Oweczkin, & Kerven, 1987) and analysing the digested samples in a Chemlab Mark I autoanalyser for N as NH_4^+ and P as PO_4^{-3}. For new leaf tissue, $\delta^{15}N$ and %N were also determined by automated combustion and analysed on a continuous-flow isotope ratio mass spectrometer (CF-IRMS, Tracer Mass, Europa Scientific, Crewe, UK).

Amino acid analysis

Fresh leaf tissue was collected at each site for amino acid analysis. As with the tissue nutrients, leaf tissue was separated into new and old leaves with only the new leaves being used in the analysis. Leaves were finely chopped with a razor blade and added to methanol in a 10:1 methanol: tissue wet weight ratio. A minimum of three leaves was used for each sample, with three samples being collected from each site and treatment. Samples were held for 24-48 h at room temperature and then stored at -4^{o}C. Amino acid content was determined on a Beckman 6300 High Performance Amino Acid Analyser. Amino acids were separated into four groups for interpretation. These groups consisted of the three amino acids with the highest concentrations, proline (Pro), asparagine (Asn), glutamine (Gln) and a fourth group including all other amino acids.

Statistics

One-way analyses of variance (ANOVA) were used to determine if the site or treatment were significant in determining the seagrass morphological or physiological characteristics. If significant differences ($P < 0.05$) occurred, Tukey's pairwise comparisons of means was used to identify which sites or treatments were significantly different ($P < 0.05$) from other sites or treatments. Data from the fertilisation experiment (treatments and control) and those from sites subject to ambient nutrients only were analysed separately.

Results

Sediment nutrients

The use of slow release fertiliser to enrich sediment nutrients resulted in highly significant increases in nutrient concentration in all treatment plots (Table 1). Increases occurred both in interstitial nutrient concentrations ($[NH_4^+]$ increased 100 fold; $[PO_4^{-3}]$ increased 10 fold) and in adsorbed nutrient concentrations ($[NH_4^+]$ increased 1 000 fold; $[PO_4^{-3}]$ increased 5 fold). The majority of biologically available N and P, at all sites, was adsorbed to the sediment particles (Table 1). Interstitial NH_4^+ separated into three statistically different ($P < 0.05$) groups: sites with low concentrations (1.5-2.8 µM), sites with intermediate concentrations (6.7-8.1 µM) and sites with high concentrations (10.7-12.7 µM). However, this statistical grouping was not related to the proximity of the site to a nutrient source. The adsorbed NH_4^+ concentrations ranged from 50-146 µmol $NH_4^+/L_{sediment}$ and were not significantly different ($P > 0.05$) between sites.

Both the interstitial and adsorbed $[PO_4^{-3}]$ differed significantly ($P < 0.001$) between sites (Table 1). Two sites which receive large amounts of nutrients and suspended solids (Oyster Bay and Long Island) had the lowest concentration of interstitial PO_4^{-3} (0.7-0.9 µM) and highest adsorbed $[PO_4^{-3}]$ (787 and 1 033 µmol $PO_4^{-3}/L_{sediment}$). This pattern was reversed at sites distant

Table 1. Interstitial and adsorbed nutrient concentrations at sites in Moreton Bay.

Site	Interstitial	Interstitial NH^{+}_{4} (µM)	Adsorbed PO^{+}_{4} (µM)	Adsorbed NH^{+}_{4} (µmol/L_{sed})	Adsorbed: PO^{+}_{4} (µmol/L_{sed})	Adsorbed: Interstitial NH_4^+	Interstitial PO_4^{-3}
Wanga Wallen fertilisation experiment							
Control	7.4^{a}	4.7^{b}	66^{a}	447^{a}	20	217	
+P	10.4^{a}	38^{c}	65^{a}	2 044^{b}	14	123	
+N	1 025^{b}	3.5^{a}	11 969^{b}	496^{a}	27	323	
N+P	719^{b}	86^{c}	7 842^{b}	2 064^{b}	25	55	
Close to nutrient sources							
Oyster Bay	6.7^{ab}	0.7^{a}	98^{a}	787^{c}	24	1 874	
Eprapah Creek		10.7^{b}	5.2^{bc}	146^{a}	523^{b}	25	183
Amity	8.1^{ab}	6.3^{bc}	87^{a}	260^{ab}	24	94	
Long Island	11.5^{b}	0.9^{a}	109^{a}	1 033^{c}	17	2 014	
Distant from nutrient sources							
Pelican Banks		2.8^{a}	1.9^{ab}	50^{a}	360^{ab}	40	421
Wynnum	7.2^{ab}	4.1^{b}	92^{a}	339^{ab}	28	184	
Deception Bay		2.9^{a}	7.9^{c}	80^{a}	117^{a}	67	36
Peel Island	1.5^{a}	5.7^{bc}	54^{a}	75^{a}	84	31	

Values in a column followed by the same letter are not significantly different using Tukey's tests ($P > 0.05$);
(Modified from Udy & Dennison, 1997a & 1997b).

from anthropogenic nutrient sources all of which had high interstitial [PO_4^{-3}] (5.7-7.9 µM) and low adsorbed [PO_4^{-3}] (75-154 µmol $PO_4^{-3}/L_{sediment}$). Interstitial and adsorbed [PO_4^{-3}] (5.2-6.3 µM, 87-146 µmol $PO_4^{-3}/L_{sediment}$) at Eprapah Creek and Amity, which receive treated sewage and septic effluent, respectively, did not differ significantly ($P > 0.05$) from concentrations at sites distant from nutrient sources.

Canopy height, shoot density and biomass

In the fertilisation experiment *Z. capricorni* biomass, canopy height and shoot density only increased significantly ($p < 0.01$) in the N+P treatment, compared with the control, +P and +N treatments (Figures 2 & 3). Leaf biomass increased by 3-4 fold in the N+P treatment, however, root and rhizome biomass were not significantly different ($p > 0.05$) from the control in any of the treatments. Canopy height of *Z. capricorni* in the N+P treatment (29 cm) was approx. 2.5 times that of the control (13 cm). This large increase in shoot size in conjunction with a doubling in shoot density, in the N+P treatment, resulted in the ratio of above: below ground biomass being significantly ($p < 0.01$) higher in the N+P treatment.

At the eight sites receiving only ambient nutrients, canopy height, shoot density and biomass differed significantly ($P < 0.01$) between sites but differences in these characteristics could not be related to the seagrasses' proximity to anthropogenic nutrient sources. Canopy heights at Long Island and Amity (59 cm), were significantly higher than at other sites ($P < 0.001$; Figure 3A). However, canopy heights at sites close to nutrient sources (Wanga Wallen (N+P), Oyster Bay and Eprapah Creek; 13 cm to 29 cm) did not differ significantly from sites distant from nutrient sources (11 cm to 29 cm).

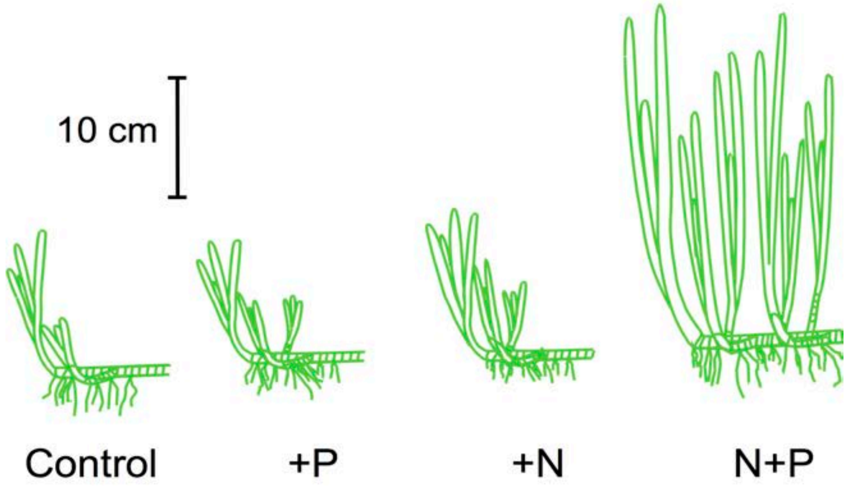

Figure 2. Growth response of the seagrass *Zostera capricorni* to fertilisation with phosphorus (+P), nitrogen (+N) and nitrogen plus phosphorus (N+P). Height of the leaves represents the average leaf canopy height, number of shoots represents the mean number of new shoots formed since the rhizomes were tagged (92 days) and the rhizome leaf scars are equal to the mean number of leaves produced since the rhizomes were tagged (modified from Udy & Dennison, 1997a).

Shoot density also varied greatly between sites but did not demonstrate any consistent trend in relation to a site's proximity to anthropogenic nutrient sources. Highest shoot density (4 900 shoots / m^2) was found at Peel Island, a site distant from nutrient sources, and lowest shoot density (370 shoots / m^2) occurred at Long Island, which receives high nutrient loads and suspended solids from the Logan River plume (Figure 3B). Shoot density was also negatively correlated with canopy height ($r^2 = 0.65$).

The leaf, rhizome and root biomass values of *Z. capricorni* differed significantly (P < 0.001) between sites (Figure 3C). Highest total biomass of *Z. capricorni* occurred at sites distant from nutrient sources: (Wanga Wallen Banks (Control) and Peel Island). At these sites, leaves accounted for 24-26% of total biomass. In contrast, the highest biomass of leaves occurred at two sites close to nutrient sources (Long Island and Amity) and the N+P fertilization treatment at Wanga Wallen, with leaves at these sites accounting for 55 to 74% of total biomass. However, there was no significant difference (P > 0.05) in the proportion of leaves to total biomass at the two sites close to nutrient input (Oyster Bay and Eprapah Creek) and three sites distant from nutrient input (Pelican Banks, Wynnum and Deception Bay).

Growth rates

Leaf growth of *Z. capricorni* was stimulated in the N+P treatment (3 times control), however, showed no significant difference (p > 0.05) between the control, +P and +N treatments (Figure 2). In contrast to this large difference in leaf growth rate of fertilised *Z. capricorni* relative to unfertilised *Z. capricorni* in the clean water of the Eastern bay, there was no significant difference (P > 0.05; Figure 3D) in areal leaf production of *Z. capricorni* at the nine sites in Moreton Bay exposed to ambient nutrient concentrations despite large differences in the leaf sizes and growth rates per shoot (Figure 3A).

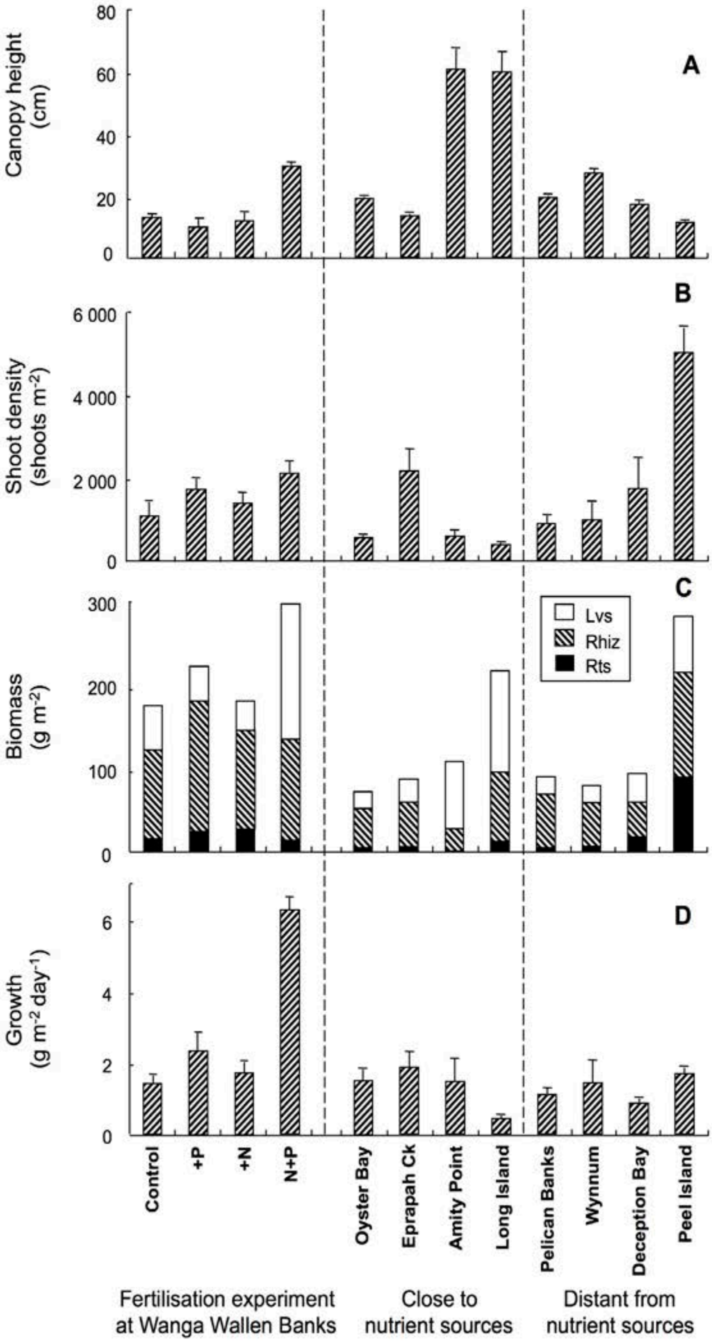

Figure 3. Characteristics of *Zostera capricorni* which were influenced by fertilisation, but were not influenced by proximity to a nutrient source: (A) canopy height, (B) shoot density, (C) biomass and (D) growth rate. (Combined data from Udy & Dennison, 1997a &1997b).

Tissue nutrient content

Tissue nutrient content (%N and %P) of *Z. capricorni* leaves and rhizomes was significantly higher in the fertilised sites relative to the control and at the sites close to nutrient sources relative to the sites distant from nutrient sources (Figure 4A). The leaves at sites close to nutrient sources (including fertilised) had 2.2-2.7%N and 0.36-0.49%P, and the rhizomes contained 0.72-1.1%N and 0.18-0.34%P. In contrast, the leaves and rhizomes of *Z. capricorni* from sites with no adjacent nutrient source ranged between 1.3-1.8 %N and 0.18-0.26 %P in the leaves and 0.38-0.54%N and 0.09-0.20%P in the rhizomes.

Amino acids

In the fertilisation experiment the concentration of all amino acids increased significantly in *Z. capricorni* leaves, with total amino acid concentrations being five times the control in the +N treatment and three times the control in the N+P treatment (Figure 4B). The relative contribution of amino acids to total leaf nitrogen also varied from 1.9% in the control to a maximum of 9.9% in the +N treatment. Glutamine increased from 0.5 $\mu mol/g_{wet\ wt}$ (18% of total amino acids) in the control leaves to 8.2 $\mu mol/g_{wet\ wt}$ (59% of total amino acids) in the +N treatment and 3.5 $\mu mol/g_{wet\ wt}$ (38% of total amino acids) in the N+P treatment. Proline, asparagine and other amino acids also increased in concentration in the +N and N+P treatments, however, their concentration was less variable than that of glutamine.

Zostera capricorni leaves from the four sites close to a nutrient source also had significantly ($P < 0.001$) higher concentrations of glutamine (1.6-4.9 $\mu mol/g_{wet\ wt}$) and asparagine (0.6-1.4 $\mu mol/g_{wet\ wt}$) relative to the concentrations at sites distant from nutrient point sources (glutamine 0.5-0.6 $\mu mol/g_{wet\ wt}$, asparagine 0.0 0.1 $\mu mol/g_{wet\ wt}$). In contrast, the proline concentration of *Z. capricorni* leaves varied greatly between the different sites in Moreton Bay with no consistent trend between sites close to or distant from anthropogenic nutrient sources. The other amino acids reached their highest concentrations in the four sites adjacent to a nutrient source with the sites distant from nutrient sources having significantly ($P < 0.001$) lower concentrations of these amino acids.

δ^{15}N

The δ^{15}N values of *Z. capricorni* leaves were significantly different ($P < 0.001$) between sites (Figure 1). The highest δ^{15}N (9.3‰) occurred at Eprapah Creek, which receives treated sewage effluent directly from the treatment plant. The lowest δ^{15}N values occurred at sites remote from anthropogenic nutrient sources, Wanga Wallen Banks and Peel Island (2.4 and 2.8‰). The remaining sites had δ^{15}N ratios ranging between 3.8 and 6.7‰.

Discussion

Sediment nutrients

Interstitial and adsorbed nutrient concentrations in the seagrass sediments of Moreton Bay are low in comparison to values reported for seagrass sediments in other parts of the world (see summary in Udy & Dennison 1996; Udy, 1998). However, the NH_4^+ and PO_4^{-3} concentrations in the present study were similar to values reported from studies in which seagrass biomass and productivity is limited by either N or P (Short *et al.,* 1985; Fourqurean *et al.,* 1992b; Bulthuis *et al.,* 1992; Udy & Dennison, 1997a), suggesting that these nutrients can potentially limit seagrass productivity in Moreton Bay.

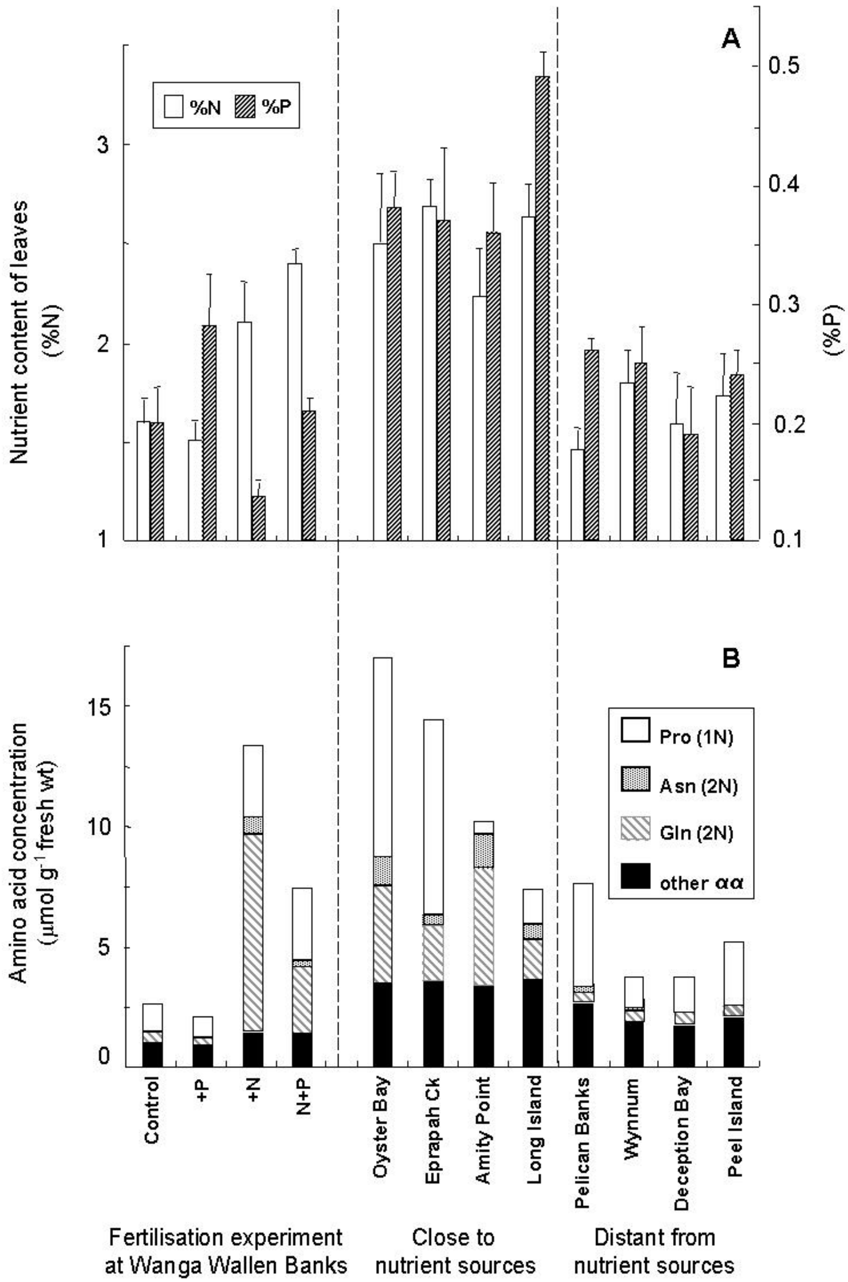

Figure 4. Characteristics of *Zostera capricorni* leaves which were influenced by both fertilisation and proximity to a nutrient source: (A) nutrient content, (B) concentration of amino acids; proline (Pro), asparagine (Asn), glutamine (Gln) and the other amino acids (other αα). (Combined data from Udy & Dennison, 1997a &1997b).

The high correlation between interstitial and adsorbed NH_4^+ concentrations indicates that a similar equilibrium exists between these chemical phases at each site (see Rosenfeld, 1979). In contrast, interstitial PO_4^{-3} was negatively correlated with adsorbed PO_4^{-3}, suggesting that the environmental factors and/or sediment characteristics which affect PO_4^{3-} adsorption vary between sites, resulting in site-specific equilibria between interstitial and adsorbed PO_4^{-3} concentrations (see Udy & Dennison, 1996). Long Island and Oyster Bay had the lowest concentrations of interstitial PO_4^{-3} and relatively high adsorbed PO_4^{-3}. Both these sites receive large quantities of suspended particles either from the Logan River or as prawn farm effluent, which may contain large quantities of iron, calcium, aluminium, or clay. If these particles are not saturated with PO_4^{-3} ions they would adsorb additional dissolved PO_4^{-3} at these sites, reducing the concentration of PO_4^{3-} in the interstitial water (Carpenter & Smith, 1984; Fox, 1990). These same two sites also had very low concentrations of sulfide (Udy & Dennison, 1997b), resulting in the sediment at these sites being more oxidised than the other sites and favouring the adsorption of dissolved PO_4^{-3} by iron hydroxides onto sediment particles (Lijklema, 1977).

Sediment nutrients relative to tissue nutrient content

If plants are nutrient limited their tissue nutrient content usually correlates with available sediment nutrient concentrations (Andrew & Robins, 1969; Mears & Humphreys, 1974). Although seagrasses adsorb nutrients from both the water column and the sediment, they receive most of their nutrients from the sediment when nutrient concentrations in the water column are low (Thursby & Harlin, 1982; Short & McRoy, 1984; Brix & Lyngby, 1985; Zimmerman *et al.,* 1987). Based on the knowledge that water column nutrients are low in Moreton Bay (Abal & Dennison, 1996) and that seagrasses in eastern Moreton Bay are nutrient limited (Udy & Dennison, 1997a), a strong correlation was expected between the sediment nutrients and the seagrass tissue nutrient content. However, our results suggest that the relative importance of sediment N for seagrass growth probably differs between sites. Leaf %N of seagrasses close to nutrient sources were higher than predicted by the regression with sediment N (Udy & Dennison, 1997b), suggesting they supplement their nutrient requirements with additional N from the water column, whereas seagrasses distant from nutrient sources had similar or lower leaf %N than predicted by the regression with sediment N. Seagrass at the Wanga Wallen Banks had the lowest %N, yet relative high adsorbed NH_4^+ concentrations in the sediment. This site receives oceanic water from the adjacent South Passage and has the highest water clarity and lowest water column nutrients of all the study sites (Jones *et al.,* 1996). The low turbidity at this site results in a high potential growth rate and hence a high N demand (Udy & Dennison, 1997a). Previous studies have suggested that physiological characteristics of seagrasses, such as tissue nutrient content, are determined by the rate of nutrient supply relative to demand (Abal, 1996; Udy & Dennison, 1997a,b; Unpublished data). This may explain why seagrasses at the Wanga Wallen site have lower %N than seagrasses from other sites with similar nutrient pools.

The high correlation between the adsorbed PO_4^{-3} in the sediment and the %P in the seagrass leaves and rhizomes supports the applicability of the Fe strip method to measure bioavailable P (Udy & Dennison, 1996). Fourqurean *et al.,* (1992b) suggested that interstitial PO_4^{-3} concentrations could also be used to estimate bioavailable P. However, the present study found a negative correlation between interstitial PO_4^{-3} and %P in the seagrass, indicating that interstitial PO_4^{-3} concentrations are not a reliable estimate of bioavailable P in all systems. This is probably due to differences in the sediment characteristics (sediment grain size, mineralogy and oxidation state) resulting in site-specific adsorption-desorption equilibria and hence differences in the relationship of bioavailable P to interstitial P at each site (Carpenter & Smith, 1984; Froelich, 1988).

Seagrass responses to nutrient availability

In general, methods that measure sediment nutrient pool sizes, as above, will only be useful in understanding plant responses if they correlate to the rate of nutrient supply to the plant. The application of seagrass physiology, as outlined in this paper, allows for a qualitative, but not a quantitative, estimate of nutrient availability in the environment. More importantly it directly measures the plants' response to nutrients in its environment and indicates if the seagrass is exhibiting signs of nutrient saturation due to high nutrient loads. This ability to provide an early warning to managers before a decline in seagrass biomass occurs may prove invaluable in managing coastal environments.

Udy & Dennison (1997b) proposed a model of how nutrient availability and other factors interact to influence *Z. capricorni* growth, morphology and physiology (Figure 5). This model suggests that nutrient availability determines potential seagrass growth rate at any site, however, light and other environmental variables can also limit seagrass growth and influence seagrass morphology and physiology. In the present study, sites close to nutrient sources experienced reduced light availability due to higher concentrations of nutrients and total suspended solids in the water column (Abal *et al.,* 1996; Jones, 1996), epiphyte growth or self shading (Amity site). This suggests that the seagrass growth rates at these sites were light limited. In contrast, sites distant from nutrient sources experienced higher light availability, with their growth rates being potentially nutrient limited. The fertilised *Z. capricorni* at the Wanga Wallen site had a much higher growth rate than any of the other sites because it had the clear water typical of nutrient limited sites and the nutrient limitation was removed by adding fertilizer to its sediment. This allowed the seagrass to grow at its maximum potential growth based on light availability and other environmental variables. The model also explains why seagrasses can demonstrate similar morphological and physiological changes in response to shading and fertilization experiments. As seagrasses in eastern Moreton Bay are nutrient limited prior to experimental manipulation of their environment (Udy & Dennison, 1997a), both shading and fertilizing alter the growth limiting factor from nutrient limitation to limitation by light or other environmental factors. This results in seagrasses exhibiting features characteristic of nutrient saturation, such

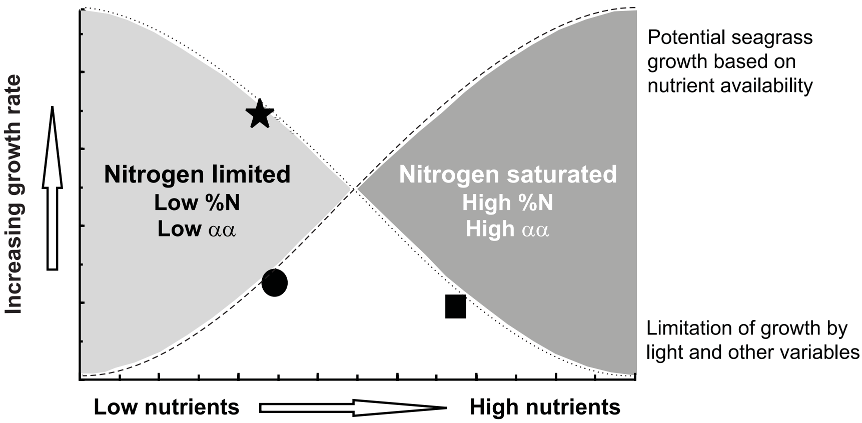

Figure 5. Model of how nutrient availability and other environmental variables affect seagrass growth. Dotted lines represent the maximum potential growth rate based on nutrient availability (- - -) or other environmental variable such as light (·····). The actual seagrass growth is on the lower of the two lines. (★) fertilised Wanga Wallen site; (■) sites close to nutrient sources; (●) sites distant from nutrient sources; amino acids (αα) (from Udy & Dennison, 1997b).

as high tissue nutrients and amino acid concentrations (especially asparagine and glutamine) (Abal *et al.,* 1994; Abal, 1996; Udy & Dennison, 1997a; b; Longstaff, pers. comm.).

Previous manipulative studies have demonstrated that seagrass growth and morphological features, such as canopy height, shoot density and the proportion of leaf biomass relative to root and rhizome biomass, increase with increasing nutrient availability (Udy & Dennison, 1997a) or decreasing light (Abal *et al.,* 1994; Abal, 1996; Longstaff, pers. comm.). In the present study, an increase in leaf length and the proportion of leaf biomass to root and rhizome biomass was observed at some of the sites close to nutrient sources, however, there was no consistent trend for all sites based solely on their proximity to a nutrient source. The growth rate of seagrasses was also similar at sites close to nutrient sources and distant from nutrient sources. This suggests that light availability and/or other environmental variables influence the growth and morphological characteristics of seagrasses more than the physiological characteristics (tissue nutrients and amino acids), that primarily reflect the nutrient availability at a site.

The nutrient content of *Z. capricorni* leaves at the five sites distant from nutrient sources were close to the critical values of 1.8%N and 0.2%P which suggest nutrient limitation of seagrass growth (Duarte, 1990). It has been demonstrated, using fertilization experiments, that *Z. capricorni* on the Wanga Wallen Banks is co-limited by both N and P (Udy & Dennison, 1997a), suggesting that N and P may co-limit *Z. capricorni* growth in large sections of the Bay. The higher nutrient content of *Z. capricorni* leaves and rhizomes at sites close to nutrient sources indicates increased bioavailability of nutrients relative to demand, and suggests that *Z. capricorni* growth and physiology is saturated for both N and P at these sites.

The concentrations of asparagine and glutamine in *Z. capricorni* were strongly influenced by proximity to a nutrient source; concentrations at sites close to nutrient sources were 4-40 fold higher than at sites distant from nutrient sources. Asparagine and glutamine have been shown to increase by 6 and 19 fold in *Z. capricorni* fertilised with N+P or N, respectively (Udy & Dennison, 1997a). Glutamine is a critical amino acid in the GS/GOGAT pathway, which incorporates NH_4^+ as organic N (Forde & Woodall, 1995), hence increases in its concentration may reflect an increase in N assimilation by seagrass leaves at sites with high nutrient availability (Short & McRoy, 1984; Zimmerman *et al.,* 1987). Asparagine and glutamine, which both have an amino and amide group, may also be accumulated for metabolic N storage (Rosenthal, 1982). The concentration of "other amino acids" were also higher at the four sites close to nutrient sources, yet, the differences between sites close to and distant from nutrient sources was much less than for asparagine or glutamine. This disproportionate increase in the concentrations of asparagine and glutamine relative to other amino acids, in relation to nutrient availability, provides an effective bioindicator of N availability in seagrass ecosystems.

In contrast to asparagine and glutamine, proline concentrations in seagrass leaves were independent of proximity to a nutrient source, suggesting that proline concentrations within *Z. capricorni* are determined by site-specific differences. Proline may be associated with osmoregulation in seagrasses, as for terrestrial plants (Davies, 1987), or it may be important in other unidentified metabolic functions.

The use of $\delta^{15}N$ ratios to establish the contribution of different N sources to seagrasses relies on knowledge of the $\delta^{15}N$ values of various N sources and the amount of fractionation involved in nitrogen uptake by the plant. The $\delta^{15}N$ values for several N sources in Moreton Bay are 9.2‰ for treated sewage particulates, 5.1‰ for raw sewage and by definition close to 0‰ for N from N_2 fixation (Loneragan, unpublished data). The range of $\delta^{15}N$ values observed in the present study was similar to $\delta^{15}N$ values reported previously for seagrasses in Moreton Bay

(2.6‰ - 8.8‰; Grice *et al.,* 1996). The high $\delta^{15}N$ values at Fisherman's Island (8.8‰; Grice *et al.,* 1996) and Eprapah Creek (9.3‰, present study) suggests that both these seagrasses receive N from nearby sewage discharges. Conversely, the similar $\delta^{15}N$ values of seagrass from the Wanga Wallen Banks (*H. uninervis*, 2.6‰, Grice *et al.,* 1996; *Z. capricorni*, 2.4‰, present study) suggests that both species of seagrass have a similar N source, with a large portion of N probably coming from N_2 fixation. The present study also demonstrated that changes in $\delta^{15}N$ values can be localised. For example, the adjacent sites Oyster Bay (5.9‰) and Eprapah Creek (9.3‰) had different $\delta^{15}N$ values reflecting the influence of prawn farm and treated sewage effluent, respectively.

In conclusion, seagrass growth, morphology and physiology in Moreton Bay are probably regulated by the interplay of light and nutrient availability. Manipulative experiments found that seagrass growth, morphology and physiology change in response to varied light and nutrient treatments (Abal *et al.,* 1994; Abal, 1996; Udy & Dennison, 1997a; Dennison *et al.,* 1997). However, physiological characteristics of seagrass (tissue nutrient content and amino acids) are the most accurate indicators of their nutrient status. Additionally, $\delta^{15}N$ values of seagrasses may be used to identify the source of N impacting on particular marine environments.

Acknowledgements

Special thanks to the many volunteers (especially: B. Dudley, E. King, S. Costanzo, B. Longstaff, N. Robinson and P. Hoxey who helped with the extensive field work and Drs N. Loneragan and C. Heil for constructive comments on the manuscript. The research was supported through postgraduate funding from the Botany Department, The University of Queensland, and as part of a collaborative research effort supported by Department of Environment, Great Barrier Reef Marine Park Authority, Queensland Department of Primary Industries, CSIRO and Australian Research Council funding to Dr W.C. Dennison.

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Non-Point Source Pollutant Estimation in Brisbane River and Moreton Bay

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Abstract

A review of non-point source pollutant estimation techniques and available pollutant loading data that are applicable to estimate non-point source pollutant loads in Brisbane River and Moreton Bay has been undertaken. To identify deficiencies in available information, current trends in southeast Queensland regarding the application of non-point source pollutant estimation techniques have been considered. The overview is intended to highlight the capabilities and limitations of the various estimation techniques and approaches, particularly in terms of their application to Queensland catchments for which recorded stormwater quality data are rarely available.

Non-point source pollutant estimation and modelling techniques were considered in relation to their complexity to reproduce catchment processes, their potential accuracy when applied to ungauged catchments and their ability to consider management strategies and Best Management Practices (BMPs). The review identified a number of limitations associated with approaches in each category. These limitations highlighted the caution which must be exercised when applying generic non-point source loading data to ungauged catchments.

A review of available non-point source data collection programs in southeast Queensland disclosed an extensive data set for sediment and nutrient runoff from urban landuses collected by Brisbane City Council. Available non-point source loading data are limited for non-urban landuses, however, some aerial loading rate data have been derived for rural and undisturbed landuses in different regions of Queensland.

Due to a general lack of locally specific data, there is a tendency for practitioners to apply loading data collected from areas outside of the catchment area of interest. There is therefore a need to collect non-point source data for additional landuse types in the Brisbane River and Moreton Bay catchment to enable formulation of effective management strategies and plans.

Introduction

Population growth and landuse changes in the Brisbane River and Moreton Bay Catchment since European settlement in the early 1800s have resulted in increased loadings of both sediment and nutrients to the receiving waters of the Brisbane River and Moreton Bay. These loadings have resulted in the general deterioration of water quality in the lower reaches of the Brisbane River (Brisbane River Management Group, 1996a).

Management of receiving water quality requires the effective assessment and understanding of the nature and volume of non-point source loads entering the waterways so that targeted management strategies can be formulated to achieve water quality objectives. Assessments in Queensland, the Australian Capital Territory and New South Wales have concluded that the successful implementation of such management strategies is increasingly dependent on the ability to model the impacts of changes in landuse (Phillips *et al.,* 1993). A range of estimation techniques and modelling approaches is available to simulate the effects of landuse changes, however, these approaches are often applied without locally specific data and a limited understanding of the limitations of the various approaches.

This paper provides an overview of landuse runoff data and non-point source modelling

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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studies relevant to the estimation of non-point source pollutant loads in the Brisbane River and Moreton Bay in southeast Queensland). The aims of this paper are to identify factors which influence non-point source pollutant export from a catchment, provide an overview of the various non-point source pollutant estimation techniques and modelling approaches which are commonly applied in Australia and southeast Queensland, review non-point source load data for southeast Queensland relevant to the estimation of non-point source pollutant loads in southeast Queensland, and review current trends in the application of non-point source estimation and modelling techniques as part of the preparation of stormwater and catchment management strategies in southeast Queensland.

Factors Influencing Non-Point Source Loads

Non-point source pollution refers to the polluted stormwater resulting from diffuse catchment runoff. The nature of non-point source pollution is determined by a range of interrelated factors which are generally site specific, and include the meteorological, physical and landuse attributes of a catchment. The factors which influence the quantity and quality of non-point source pollutant loads can be broadly divided into two groups: (i) factors which affect wash-off characteristics, and (ii) factors affecting pollutant availability.

Factors influencing wash-off

Factors which influence wash-off characteristics are generally associated with the meteorological, hydrological and physical characteristics of a particular catchment. These factors include rainfall intensity, rainfall volume, inter-event duration (i.e. time between rainfall events), catchment size, topography (i.e. slope and form), soil type (moisture storage characteristics), vegetation cover (runoff efficiency) and impervious area (runoff efficiency). With the exception of vegetation cover and impervious area, these factors can generally be considered as being site-specific and not necessarily related to a generic landuse.

Pollutant sources

Pollutant availability within a catchment is more closely associated with landuse. Table 1 provides an example of different pollutant sources for three typical landuse categories (i.e. urban, rural/agricultural and natural), and shows that pollutant availability varies between different landuse types. For example, a rural or agricultural type landuse may generate elevated levels of suspended sediment due to the increased area of exposed earth compared to a natural or undisturbed landuse.

Table 1. Pollutant sources associated with typical landuse types.

Urban Landuse	Rural/Agricultural	Natural/Forest
Vehicle emissions	Atmospheric deposition	Atmospheric deposition
Atmospheric deposition	Fertiliser wash-off	Decaying vegetation
Animal wastes	Soil erosion	Animal wastes
Refuse/Garbage/Litter	Decaying vegetation	
Fertiliser wash-off	Animal wastes	
Leaf litter, decaying vegetation matter	Agricultural chemicals	

The magnitude of the actual pollutant load washed off from each landuse will also be dependent on site specific characteristics of the catchment. Hence, based on the landuse types within the catchment, the pollutant export potential of a catchment is a function of both the catchment wash-off characteristics and pollutant availability. This is an important consideration, particularly in relation to the capabilities of the various estimation techniques and whether these processes can be separately replicated.

Methods of Estimating Non-Point Source Pollution

A wide variety of techniques and modelling approaches is available for the estimation of non-point source pollutant loads from a catchment. These techniques range in complexity, potential accuracy and data requirements. Some of these techniques have been developed to simulate non-point source pollutant export from particular catchment types, such as agricultural or urban catchments, whereas others have been formulated to replicate the individual physical processes which occur in any catchment.

A large number of studies have provided comprehensive reviews of surface runoff quality models (e.g. Huber & Heaney, 1982; Whipple *et al*., 1983; Donigian & Beyerlein, 1985; Codner, 1989; Donigian & Huber, 1991; Heidtke & Auer, 1993; Novotny & Olem, 1994; Walton & Hunter, 1996). The majority of these reviews have considered non-point source numerical models which have been developed and extensively applied in the USA.

The various estimation techniques can be segregated into three broad categories, or levels. These categories are based upon: (i) the estimation or modelling objective; (ii) complexity of the approach and the way in which it simulates the processes which contribute to non-point source pollutant runoff; (iii) the capabilities of the approach for considering changes in landuse; (iv) the potential accuracy of the approach when being applied in ungauged catchments; and (v) the ability of the approach to consider various management and mitigation strategies (e.g. Best Management Practices (BMPs) and Stormwater Quality Improvement Devices (SQIDs)). The techniques in these categories range from simple desktop calculation methods to more sophisticated computer models which attempt to replicate the individual physical processes that cause non-point source pollutant export. Each approach is valid in its own right, depending on the objectives of the modelling effort and how the adopted approach is applied.

The segregation of techniques into three categories is consistent with the approach used by other authors (e.g. Codner, 1989; Willing & Partners Software, 1996), however, a fourth level which relates to Operational Models has also been considered. Operational modelling refers to the ‘real time’ application of the three categories of estimation techniques discussed in this paper using input from telemetry stations in the catchment (e.g. rainfall).

Level 1 non-point source pollutant estimation approaches

Level 1 approaches are often referred to as either ‘simple models’, ‘screening models’ (Codner, 1989; Novotny, 1995; Willing & Partners Software, 1996) or ‘community awareness models’ (Walton & Hunter, 1996). This type of approach is generally useful for broad scale planning and includes such methods as unit loading rates, statistical techniques and regression/rating curve approaches (Novotny, 1995). They can be used as ‘first pass’ techniques to consider ‘order of magnitude’ changes in pollutant export from a catchment for changes in landuse characteristics.

The parameters and relationships used in Level 1 approaches are typically formulated using a data set obtained from a specific catchment. Site-specific characteristics, such as catchment topography and climatic condition, are not represented separately within a Level 1 approach. Therefore, the application of these approaches outside the location where the original data set was obtained may result in error.

The general characteristics of a Level 1 approach include the following:

- the approach can generally be undertaken by using desktop analysis techniques and does not necessarily require a computer;
- in the case of unit loading rates (e.g. kg/ha/yr), the approach is generally landuse-based (e.g. rural, urban, industrial);
- other methods, such as statistical or rating curve methods are utilised to provide relationships of concentration or load against flow for a specific catchment;

- there is generally no specific representation of flow or pollutant routeing within the catchment (i.e. simulation of in-stream and lagging processes); and
- the approach is generally used for calculation of long term averages (e.g. annual-kg/ha/yr).

Some examples of Level 1 estimation techniques and modelling approaches include:

- unit loading rates (kg/ha/yr) for various landuse types;
- constant concentration methods where all runoff is assumed to have the same concentration for a given pollutant;
- statistical methods, such as the derivation of frequency distributions for Event Mean Concentrations (EMCs), and rating curve approaches, such as regression analyses in which concentrations and/or loads are related to flow rates and/or volumes; and
- Catchment Management Support System (CMSS).

Level 1 approaches have a number of inherent limitations and potential inaccuracies when applied to ungauged catchments. These are due essentially to their inability to account for catchment-specific characteristics, such as topography or climatic condition. Limitations of a Level 1 approach include inability to:

- account for site-specific catchment characteristics;
- account for variations of rainfall intensity or volume;
- consider management strategies such as BMPs and SQIDs; and
- be accurately applied beyond the catchment where the original data set was collected.

Level 1 approaches are among the most commonly applied non-point source pollutant estimation techniques. They have been applied with varying degrees of success throughout Australia and overseas. Level 1 approaches provide a valuable tool for landuse planners to consider broad scale impacts associated with changes in landuse. They also provide an excellent basis for considering whether more complex simulation models are needed (Novotny, 1995). When considering this kind of approach it is important to bear in mind its inherent limitations and applicability for detailed landuse planning and mitigation assessment in ungauged catchments.

Level 2 non-point source pollutant estimation approaches

Level 2 non-point source pollutant estimation techniques are often referred to as ‘planning models’ (Codner, 1989; Willing & Partners Software, 1996). These approaches are often used to more accurately assess the impact on pollutant export associated with variations in landuse types within the catchment.

The algorithms which are applied in Level 2 approaches generally incorporate some specific representation of major site-specific hydrological processes in the catchment, such as soil moisture storage characteristics and rainfall volumes, when applied to ungauged catchments. The general characteristics of Level 2 non-point source pollutant estimation approaches are generally computer-based, and usually include the following:

- some kind of hydrological model representation;
- a simple pollutant export algorithm based on the hydrological representation within the catchment;
- generation of runoff hydrographs and pollutographs at various locations throughout the catchment;
- continuous simulation of watershed behaviour over a long period;
- a time step ranging from hours to days;

- provide facilities for simple routing of flows and pollutants through the catchment; and
- basic integration of management measures (e.g. BMPs) and their operation within the catchment.

Examples of the Level 2 approach include the following commercially available software packages:

- Storage, Treatment, Overflow, Runoff Model (STORM);
- Integrated Water Quantity and Quality Model (IQQM); and
- AQUALM-XP.

Level 2 modelling approaches generally provide greater accuracy than Level 1 approaches when applied to ungauged catchments, however, data requirements are more intensive. To effectively apply these models, it is necessary to define parameters which describe the hydrological response of the catchment and the pollutant export characteristics of the various landuses it supports. Where possible, the accuracy of Level 2 approaches should be confirmed by calibration and verification to available stormwater quality data within the catchment.

Level 2 models also have a number of inherent limitations, which include the following:

- an inability to accurately account for rainfall intensity;
- a limited ability to account for catchment slopes or topography;
- a tendency to generally be more data-intensive than Level 1 models;
- an inability to accurately simulate pollutant export for small duration, individual events; and
- somewhat site-specific parameters.

One of the major advantages of Level 2 approaches is their ability to include the effects of BMPs and management strategies within the catchment. Phillips *et al.* (1993) point out that the need to go beyond annual load type models (i.e. Level 1) occurs whenever water quality management strategies are to be formulated or control measures are to be optimised.

Level 2 approaches can be used very effectively as planning tools to include the impacts of future development in a catchment and the subsequent formulation of management strategies. Commercially available models such as AQUALM-XP and IQQM have been used extensively throughout Queensland and other parts of Australia. In many instances a lack of locally specific data has resulted in the application of Level 2 approaches using generic loading data from other catchments. Under these circumstances, the inherent limitations of Level 2 approaches should be considered when interpreting results.

Level 3 non-point source pollutant estimation approaches

Level 3 approaches are generally referred to as either 'complex routeing models' or 'design models' (Codner, 1989; Novotny, 1995; Willing & Partners Software, 1996). These techniques enable a detailed simulation of storm events, the rainfall runoff response of a catchment and its subcatchments, and the associated pollutant export relationships for specific landuses within the catchment. Level 3 estimation techniques and models tend to be more deterministic in nature than Level 1 and 2 approaches. Some Level 3 models are extremely detailed and attempt to replicate the individual physical processes that occur in the catchment.

Typical characteristics of a Level 3 approaches include the following:

- detailed description of flow and pollutant routeing from the point of rainfall to the receiving waters;
- representation of extensive drainage networks (including gutters, channels, overland);

- short time steps (e.g. minutes or smaller);
- single event or continuous simulation;
- more sophisticated pollutant export simulation, such as build-up and wash-off algorithms or simulation of individual physical processes;
- numerous parameters and coefficients describing hydrological and pollutant export relationships; and
- computerisation to enable fast processing of multiple inter-relationships.

Examples of commercially available software packages which apply Level 3 approaches include:

- Stormwater Management Model (SWMM);
- Hydrological Simulation Program – Fortran (HSPF); and
- Chemical, Runoff and Erosion from Agricultural Management Systems Model (CREAMS).

Level 3 modelling approaches provide the greatest potential for accurate estimation of the magnitude and characteristics of non-point source pollutant export from ungauged catchments because they provide an accurate representation of catchment-specific physical and meteorological characteristics. Level 3 approaches, however, also have a number of inherent limitations associated with the complexity of the models and their data requirements. Due to the number of process parameters in Level 3 models, there is a need for an extensive data set of in-field process measurements or stormwater quality data for parameter calibration within the catchment of interest. In the past, the lack of available water quality data in Australian catchments has limited the full use of more sophisticated models (Phillips *et al.,* 1993).

Summary of non-point source estimation approaches

Each of the modelling approaches described above is valid and useful for non-point source pollutant estimation, however, when selecting an estimation approach it is important to consider the relative limitations of each approach and the overall objective of the study being undertaken. The following guidelines (Novotny, 1995) provide a useful basis for considering non-point source estimation requirements and the subsequent selection of an appropriate technique which will satisfy the study objectives:

1. Have a clear understanding of the project objectives and carefully consider the need for pollutant export estimation or modelling.
2. Adopt the simplest estimation technique possible to satisfy the project objectives (a screening model, e.g. Level 1, is often useful for considering whether a more complex approach is required).
3. Adopt an estimation technique that is consistent with the available data. (The limitations of particular estimation techniques should also be considered in this regard.)
4. Select a time scale that is compatible with the study objectives.
5. Predict only the water quality parameters that are of interest to the study (e.g. when considering impacts associated with urbanisation or agricultural development, suspended sediment and nutrient parameters will be of key interest).
6. Perform a sensitivity analysis on the selected model and familiarise yourself with the model characteristics.

7. Wherever possible, calibrate and verify model parameters and results to data collected within the study area catchment.

Existing Non-Point Source Load Data in Southeast Queensland

A review of known major non-point source data collection programs for southeast Queensland has been undertaken by the authors, and is based on available literature, discussions with the personnel involved in the various data collection programs and the authors' experience in similar studies. In identifying key data collection programs, we considered the nature of the programs and the subsequent analysis and utilisation of the data collected. In the majority of cases these data collection programs focused on collecting suspended sediment and nutrient data.

Brief descriptions of major stormwater data collection and analysis programs in southeast Queensland and other relevant locations are listed below.

Department of Natural Resources (DNR) have been collecting non-point source loading data for agricultural landuse on the Darling Downs over the past 20 years (WBM Oceanics Australia & Sinclair Knight Merz, 1997). The aim of this data collection program has been to identify sediment washoff rates for agricultural landuses and farming practices. To date, data have been collected for approximately 30 small catchments.

DNR have collected a limited amount of stormwater quality data from a site on Laidley Creek near the Warrego Highway (WBM Oceanics Australia & Sinclair Knight Merz, 1998).

Cosser (1989) carried out a data collection program in the South Pine River over a period of approximately two years (October 1983 to September 1985). Water quality data were collected using a flow-weighted sampling approach and the aim of subsequent analysis undertaken was to investigate nutrient concentration-flow relationships and loads. Non-point source loading rates were derived using a unit loading approach (i.e. $kg/ha/km^2$).

Brisbane City Council commenced an extensive stormwater data collection program in 1994 (WBM Oceanics Australia, 1995). Stormwater quality data for suspended sediments and nutrients have been collected for two established urban catchments: (i) a commercial/industrial catchment, and (ii) a rural residential catchment. The aim of this data collection program is to develop nutrient and sediment load runoff relationships for a range of landuses in the Brisbane region to enable predictive assessment of landuse changes. For the urban sites, four years of stormwater runoff data for phosphorus, nitrogen and suspended solids have been collected, and are largely consistent between the two catchments. The data have been used to develop export parameters for use in AQUALM-XP (Level 2) pollutant export models. An example of the accuracy of the model calibration for the load versus runoff relationship for total nitrogen in one of the urban catchments is presented in Figure 1.

Department of Environment installed instrumentation on Enoggera Creek upstream of the Enoggera Creek Dam in 1997 to collect non-point source pollutant export data for an undisturbed landuse (WBM Oceanics Australia & Sinclair Knight Merz, 1998). To date, data have only been collected for several major storm events, and detailed analysis of this information has not yet been possible.

Ipswich City Council has been collecting stormwater data for approximately two years (WBM Oceanics Australia & Sinclair Knight Merz, 1997). Stormwater samples have been collected using rising stage samplers for developed urban, developing urban and undisturbed catchments. Samples have been analysed for suspended sediment levels. At the time of preparing this paper, analysis was being undertaken to determine unit load relationships for the various landuses.

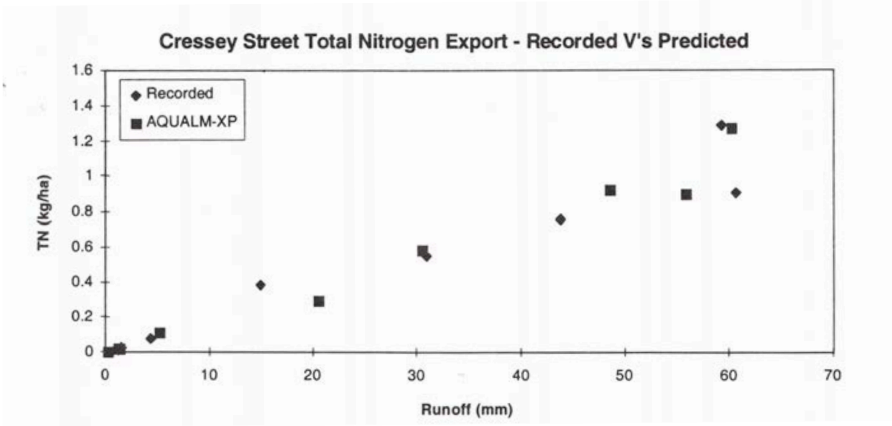

Figure 1. AQUALM-XP Model Calibration Using BCC Stormwater Quality Data (WBM Oceanics Australia, 1995).

Southern Cross University collected field data from various locations in the Brisbane River and Moreton Bay catchment during 1996 (WBM Oceanics Australia & Sinclair Knight Merz, 1998). This information has been used to derive pollutant load estimates for the Brisbane River, Logan River and Caboolture River catchments.

A number of other detailed data collection programs and landuse runoff assessments have been undertaken in north Queensland (Prove & Hicks, 1991; Walton & Hunter, 1996). These are not considered relevant to southeast Queensland, however, due to the elevated annual rainfall volumes and peak intensities which are experienced in north Queensland.

In addition to the data collection and assessment programs detailed above, a number of studies have also been undertaken which have applied generic unit loading rates to derive non-point source pollutant export data for Queensland catchments (e.g. Moss *et al.,* 1992; Brisbane River Management Group, 1996b). These studies exhibit considerable variability in the range of loading rate values which have been adopted for the same landuse. This highlights the caution which must be exercised when applying aerial loading rates which are derived from other studies.

The stormwater data collected by Brisbane City Council (WBM Oceanics Australia, 1995) represent the most comprehensive urban stormwater quality data set available in southeast Queensland. This data set has been extensively analysed to derive suspended sediment and nutrient export relationships for predictive assessments of urban runoff using a Level 2 modelling approach (i.e. AQUALM-XP). The data would also be suitable for deriving parameters for a Level 3 approach, however, this has not yet been undertaken.

The previously mentioned data collection programs indicate that a limited amount of data has also been collected for suspended sediment and nutrient loads from other landuse types in southeast Queensland (e.g. rural and undisturbed landuses). The data collection programs that have been used to collect this information are less rigorous in nature than the Brisbane City Council urban stormwater data collection program. In addition, the analysis and interpretation of these data have limited their suitability for application to predictive assessments beyond Level 1 estimation techniques.

Application of Non-Point Source Loading Data – Current Trends

The release of the Environmental Protection (Water) Policy (EPP) (Queensland Government, 1997) placed an obligation on local authorities in Queensland to prepare environmental plans that aim at the improvement of stormwater quality to meet the water quality objectives of downstream waters. In response to the EPP (Water), many local authorities in southeast Queensland, and other parts of the state, have embarked on a process of developing catchment and stormwater management plans.

As part of the development of catchment and stormwater management plans, it is generally necessary to consider the effects of non-point source pollutant export loads for existing and future landuses to enable development of management strategies. This has generated considerable interest in the availability of non-point source loading data to undertake such assessments.

Due to the general lack of locally specific data, many local authorities have applied the data derived by Brisbane City Council, which, as noted above, currently represents the most comprehensive urban data set in Queensland. These data are being used to predict any impacts of urbanisation on downstream water quality and to develop management strategies accordingly. This generally includes the utilisation of AQUALM-XP models (i.e. a Level 2 approach) as part of the preparation of stormwater management plans.

Although the Brisbane City Council data set may represent the most comprehensive urban runoff data collected in Queensland, it is important to note that potential inaccuracies can result when applying this information generically to catchments with different climatic and physical characteristics. In addition, there are limited non-point source loading data on rural and undisturbed catchments in Queensland, and this has led to a tendency to apply generic data derived in southern states (e.g. Sydney landuse data). These practices highlight the need to gain a better understanding of the limitations of the various estimation approaches being applied and the potential inaccuracies which can result when extrapolating data sets beyond their catchment of origin.

Conclusions

This paper has provided a review of various non-point source pollution estimation techniques and available pollutant loading data for southeast Queensland. In addition, current trends in the estimation of non-point source pollutant loads as part of the development of catchment and stormwater management plans have been discussed.

Each of the estimation techniques and modelling approaches discussed can be highly effective, depending on the objective of the study being undertaken and the availability of local data sets. Due to a general lack of locally specific stormwater quality data for certain landuse types in southeast Queensland, there is a growing tendency for practitioners to apply generic loading parameters to ungauged catchments with a limited consideration of the capabilities of the adopted estimation technique. If predictive assessments are to be undertaken with any degree of accuracy there is a need to validate the models and estimation techniques using data collected within the catchment of interest.

In the context of the Brisbane River and Moreton Bay catchment, there is a need to collect detailed data sets for additional major landuse types within the catchment (e.g. rural and undisturbed). This process has been commenced by Brisbane City Council, however, only limited data have been collected to date. Once these data sets have been collected and analysed to a level similar to the existing urban stormwater data set, it will be possible to derive non-point source pollutant loading relationships for all major landuses. This will enable more accurate predictive assessments of the impacts associated with changes in landuse and the identification of management strategy requirements to meet the water quality objectives for receiving waters.

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Water Quality and Modelling in Moreton Bay

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Moreton Bay and associated waterways receive an ever-increasing load of pollutants, principally nutrients, sediments and toxicants, as a result of human activities in the catchment. Our understanding of the pathways, impact and eventual fate of these inputs is currently at a fairly elementary level, but it is clear that they have the potential to endanger the environmental health of the system.

Management of the Bay and its waterways requires a sound scientific understanding of the processes regulating water quality and the interactions between the different components of the ecosystem. The Brisbane River and Moreton Bay Wastewater Management Study, now entering Stage 2 of a 4-year study, is designed to provide a framework for the environmental management of the Moreton Bay catchment for the next 20-30 years.

A core component of the Study is the development of a deterministic mathematical model of the important physical, chemical and biological processes (Gabric *et al*., 1998; McEwan *et al*., 1998*)*. This model provides a quantitative description of:

- water movement throughout the Bay and rivers in response to tides, winds and freshwater flows,
- pollutant transport pathways in response to advective and dispersive processes, oceanic exchange, settling and resuspension from the underlying sediments, and
- biological uptake and recycling of nutrients in the water column and benthos.

Model results indicate that water quality in the northern and eastern sections of Moreton Bay is maintained at or near background oceanic levels by vigorous tidal flushing, the residence time of the Bay as a whole being in the region of 50 days. Nearer the western shores of the Bay residence times increase substantially due to reduced tidal flushing, making them more vulnerable to the numerous point-source pollutant discharges in this region. Due to the Bay's shallowness and width, winds can have a major influence on transport processes.

For most of the time (~95%) freshwater flows are minimal and the Bay acts as a fully-developed marine embayment. During flood events the Bay is more like a highly-turbid estuarine system with large inputs of catchment-derived nutrients and sediment. These have only a transitory effect however and the Bay returns to its normal state within two to three weeks.

Ambient nutrient concentrations appear to have remained fairly constant over the last 20 years despite the significant increases in inputs (due to population growth) which have occurred over this period. There is insufficient historical data to delineate long-term trends in phytoplankton concentrations. Diatoms appear to be the most abundant type of phytoplankton in the water column of the Bay.

Field and model results together identify nitrogen as the limiting nutrient for primary production. This results from a low N:P ratio in point-source nutrient loads coupled with vigorous denitrification in sediments. Model runs estimate almost half the external inputs of

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 239-240.

nitrogen are lost through sediment denitrification, with approximately 10% of nitrogen loads accumulating in the benthic compartment in organic form. In contrast almost 75% of external P loads are flushed to the outer ocean.

To account for current levels of water column primary production during the predominant baseflow regime requires an extra 1500T/yr of nitrogen to supplement point source loads (~3100T/yr). Possible sources of this additional N include *in situ* benthic N-fixation, atmospheric or groundwater loads, but our knowledge of the system is insufficient to determine the relative importance of each. A better understanding of nutrient inputs and their transformations is essential in order to determine critical nutrient loadings beyond which irreversible damage to the functioning of the Bay ecosystem would occur.

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Organochlorine Pesticide Residues in Brisbane Waterways

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Abstract

Urban landscapes today carry background levels of a range of persistent contaminants such as organochlorines and their breakdown products, a legacy of the past use of this class of pesticides. Trace concentrations can readily be detected in sediments and the tissues of biota sampled from near urban waterways. This study is an overview of organochlorine residues in superficial sediments and aquatic biota quantified in a recent survey of typical urban Brisbane waterways, and an estimation of the ecotoxicological significance of the concentrations present.

Introduction

Organochlorine pesticides including DDT, chlordane and dieldrin were used extensively during the 1950s and 1960s in urban areas of Australia for a wide range of domestic, public health and agricultural purposes. Recognition of their intransigent persistence and long-term effects on ecosystems led to their replacement with other chemical options for pest control.

Consequently, urban landscapes today carry background levels of a range of organochlorine contaminants and their breakdown products, some of which have persistence and potential toxicity similar to those of the parent compounds. Trace concentrations can readily be detected in sediments and the tissues of biota sampled from near urban waterways. The extent of contamination in Australia as a whole was reviewed in a study by the Commonwealth Environmental Protection Authority (Hamdorf, 1992), and globally by Simonich & Hites (1995).

Methodology

A hand-operated stainless steel grab was used to collect composite samples of superficial bottom sediment at various sites in the Brisbane River and several urban creeks during September - October 1996. All sampling sites were within the urban area; upstream and downstream sites were sampled on all creeks except Boggy Creek and Jacksons Creek which are short (see Figure 1).

Samples were stored on ice in solvent-washed glass jars until analysis by the Queensland Government Chemical Laboratory, Coopers Plains, for pesticide content (organochlorines, organophosphates and synthetic pyrethroids) and total organic carbon.

Results

Organochlorines were present in all sediment samples, and an organophosphate (chlorpyrifos) was present at two sites, as was a synthetic pyrethroid (bifenthrin).

Table 1 lists organochlorine concentrations and organic carbon fractions.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 241-246.

Table 1. Organochlorine concentrations (µg/kg dry weight) and total organic carbon fractions. Sediments from Brisbane waterways.

	Boggy Creek	**Jacksons Creek**	**Kedron Brook Floodway**	**Brisbane River**	**Norman Creek**	**Enoggera-Breakfast Creek**	**Bulimba Creek**	**Oxley Creek**
DDE	1.0	0.2	0.2 - 0.5	4.5 - 6.5	5.0 - 7.5	5.0 - 6.0	9.5 - 18.0	2.5 - 4.5
DDD	2.0	< 0.2	0.3 - 0.8	3.0 - 3.5	5.0 - 8.0	6.5 - 40.0	7.0 - 16.5	5.5 - 7.0
DDT 49.0	5.5	< 0.2	< 0.2 - 0.3	1.0 - 1.5	1.5 - 3.5	1.0 - 2.0	2.0 - 2.5	9.5 -
dieldrin	< 0.2	0.4	0.5 - 1.3	2.5 - 3.0	4.0 - 16.5	4.0	1.5 - 3.0	2.5
chlordane	< 0.2	< 0.2	< 0.2	1.5 - 2.0	6.5 - 23.5	9.5 - 11.0	< 0.2	< 0.2
heptachlor epoxide	< 0.2	< 0.2	< 2.0	< 0.2	2.5	0.2	< 0.2	< 0.2
Total organic carbon fraction (% dry weight)	**15.3**	**2.34**	**2.88 - 10.32**	**10.75 -11.04**	**8.04 -11.8**	**4.98 - 9.51**	**9.59 - 12.3**	**8.62 -**

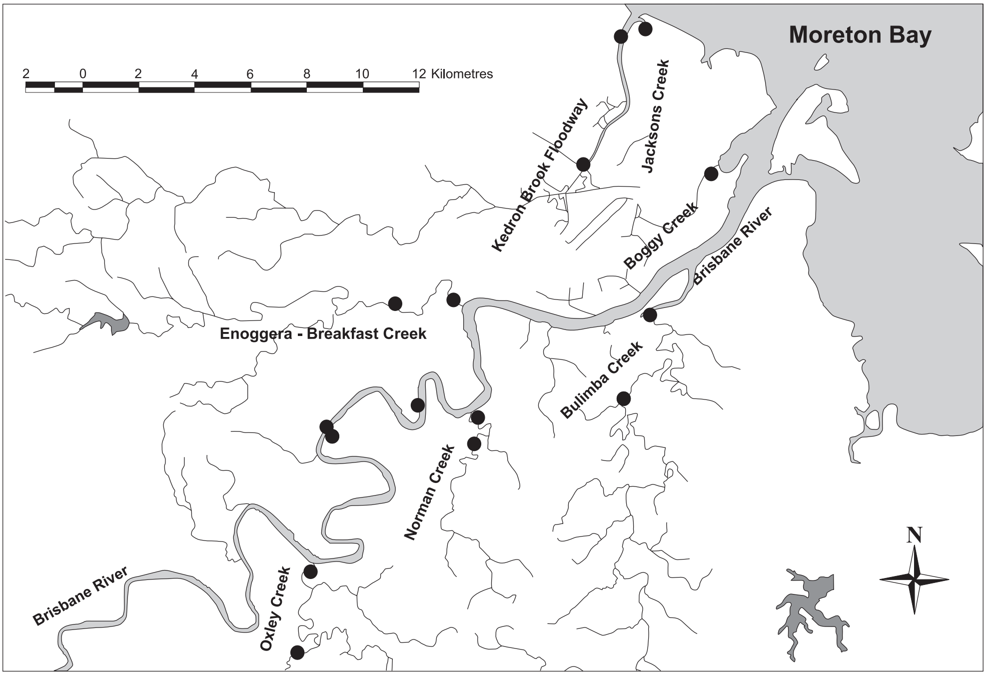

Figure 1. Sediment sampling sites for pesticide detection in Brisbane waterways.

Discussion

Source of organochlorines

The organochlorine pesticides detected in Brisbane sediments have been banned in Australia and are no longer legally available. DDT has not been used legally in Queensland since the early 1970s. The widespread occurrence of organochlorine residues in sediments is evidence of background contamination from past use rather than the result of illicit use or dumping of residual stock. Dumping may contribute, however, as suggested by the relatively high level of DDT at one site in Oxley Creek.

Organochlorines were used widely in the Brisbane area to protect buildings against termites and to control other household pest species. They were also used domestically and commercially for plant pest control. DDT was used for mosquito control before the introduction of organophosphates and synthetic pyrethroids.

The quantity of organochlorines used in Australia is not documented. Most material was imported under the category of ‘agricultural chemicals’. The exact nature of the chemicals was suppressed on the grounds of ‘commercial sensitivity’ and there was no central registry (Voldner & Li, 1995). However, these authors report that before DDT was banned, up to 10 000 tonnes were used in Australia.

Organochlorines bind strongly to soils and sediments, are very resistant to environmental degradation, and have half-lives of up to 10 years (Howard *et al.*, 1991). As a result, waterways can be contaminated by stormwater run-off containing soils from areas where organochlorines have been used legitimately in the past.

Ecotoxicological significance

Organochlorines in sediments are potentially toxic to aquatic organisms. The movement of degradation-resistant lipophilic compounds between sediments, water and biota is controlled by their physico-chemical properties, in particular partitioning behaviour (Connell, 1988).

Accordingly, ambient levels of organochlorines in sediments can be associated with an equilibrium concentration in the interstitial water to which benthic organisms are exposed (C_w). This concentration can be compared with levels accepted as non-harmful to aquatic ecosystems under ANZECC guidelines (Table 2). These calculations can be done using the relationship

$$C_w = \frac{C_s}{K_{oc}.f_{oc}}$$

where C_W is the interstitial water concentration, C_S the contaminant concentration in sediment, K_{oc} the sorption coefficient (the ratio of contaminant per unit mass of organic carbon), and f_{oc} the fraction of organic carbon in the sediment.

Values for K_{oc} were estimated from the octanol/ water partition coefficients for the pesticides (K_{ow}) using the Karickhoff relationship which is widely accepted for this purpose (Table 3).

Taking the highest contaminant concentration (49 µg/kg of DDT in Oxley Creek), the calculation of C_w gives an equilibrium interstitial water concentration of less than 1 ng/L which does not exceed the ANZECC guideline for this contaminant. Accordingly no significant toxicological effect on biota can be attributed to the presence of residual organochlorines in these sediments.

Table 2. ANZECC guidelines for protection of aquatic ecosystems.

Pesticide recommended	Maximum concentration (ng/litre)
DDE	14
DDD	not set
DDT	1
Dieldrin	2
Chlordane	4
heptachlor	10

Table 3. Values[a] for organic carbon sorption coefficients (K_{oc}) used in calculations of ambient water concentration.

Pesticide	Sorption coefficients ($\log K_{oc}$)
DDE	6.09
DDD	5.60
DDT	5.95
Dieldrin	3.93
Chlordane	5.13

[a] Estimated using the relationship $\log K_{oc} = 0.989 \log K_{ow} - 0.346$ (Karickhoff, 1981).

Comparison with organochlorine-contaminated sediments elsewhere

Concentrations of organochlorines in Brisbane sediments are low compared with those which have caused concern elsewhere. For example, Young *et al.* (1976) reported concentrations in sediments offshore from Los Angeles exceeding 200 000 µg/kg of DDT, and Iwata *et al.* (1994) up to 1700 µg/kg of DDT in sediments from Sydney Harbour.

Some authors have attempted to estimate the age and source of DDT contamination from the ratio between the parent compound and its metabolites (DDE and DDD). However the rate of conversion depends on a number of environmental variables, and little is known concerning these in local waters and sediments. In all but two waterways the ratio of metabolites to DDT is >1, suggesting that these contaminants are not of recent origin. By contrast, in Oxley Creek and Boggy Creek, the concentrations of parent compound are in excess of metabolites. The location of these sampling sites suggests the possibility that the organochlorines in these sediments may be related to the influence of significant inputs of organochlorines from nearby sewage treatment plant discharges (Whyte, 1987).

Expected organochlorine concentrations in fish and seafood

The bioaccumulation of lipophilic compounds in fish, prawns and crabs is dominated by partitioning relationships (Connell, 1988). Consequently, concentrations of organochlorines in tissue lipids are comparable with those in the organic carbon fraction of bottom sediments and we would expect concentrations in locally caught species to be in the order of a few µg/kg wet weight, well below the National Food Guidelines (typically 1 mg/kg). This is confirmed by recent data which show median measured concentrations of total organochlorines in edible mudcrabs caught in the Brisbane River are in the range 3-11 µg/kg wet weight (see Moss & Mortimer, 1996, Figure 2).

Export of organochlorines to Moreton Bay

The organochlorine concentrations listed in Table 1 were measured in the top layers of sediments of waterways which discharge into Moreton Bay. Accordingly, these contaminated sediments

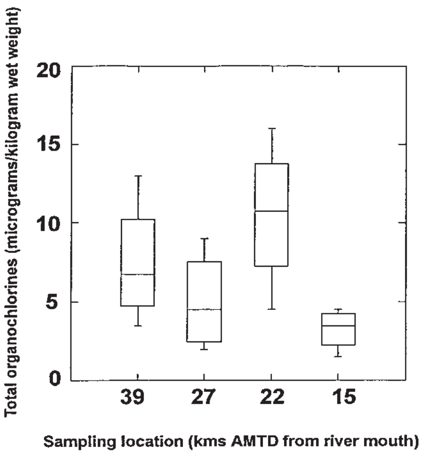

Figure 2. Total organochlorines in mudcrab (*Scylla serrata*) from the Brisbane River. Source: Moss & Mortimer, 1996.

will be periodically resuspended and carried to the Bay. The quantity of sediments carried into Moreton Bay was recently estimated to be 375 000 tonnes/year (WBM-SKM, 1995).

Table 4 gives the mean concentrations of organochlorines in the sediments and the potential rates of transport into the Bay. Based on these values, the total annual amount of sediment-bound organochlorines reaching Moreton Bay is about 10 kg. However, because of dilution by the sediments and waters of the Bay, the resulting ambient levels are well below those which might be of environmental concern.

Dissolved organochlorines will also be carried into Moreton Bay in creek and river water discharges. Although the volume of water, estimated at 900 gigalitres per annum from the Brisbane River alone (GHD, 1996), is much greater than that of the contaminated sediments, the partitioning relationships referred to above indicate that ambient concentrations of

Table 4. Estimates of annual organochlorine discharge to Moreton Bay.

Pesticide bound Bay	Mean (standard deviation) of source waterway sediment concentrations (µg/kg) (n = 14)	Estimated sediment- discharge to Moreton (kg/year)
DDE	5.1 (4.7)	1.9
DDD	7.5 (10.3)	2.8
DDT	5.7 (12.7)	2.1
Dieldrin	3.3 (4.0)	1.2

organochlorines in water discharges are approximately six orders of magnitude less than those in suspended solids. Accordingly, the input attached to sediments is the most significant source.

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Chemical Character of a Potential Acid Forming Sediment, Pimpama River, Moreton Bay

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Abstract

Acid conditions of soils and waters have been observed in many developments around Moreton Bay, from early sand mining to new marina construction. To determine typical geochemical processes that produce the acidification, a representative sediment core was selected from an acid section of the Pimpama River on the southern Moreton Bay coastal plain. The sediments that are of concern are Holocene estuarine muds, mainly clays, rich in organic matter, Fe, Al, and S. In these areas the occurrence of acid sulfate soils and the subsequent Al toxicity are important environmental considerations.

Sampled as a shallow core from the intertidal zone of the river, 9.5 km from the mouth using a 5 cm diameter PVC pipe driven into the river bed, the sediment is typically silty clay: reddish from 0 to 11 cm, black from 11 to 24 cm, and dark grey from 24 to 131 cm (bottom). Eight sections of the core were taken (every 20 cm) and analysed by total digestion (hydrofluoric-aqua regia) for ICP determination of major elements, Leco for total sulfur, and X-ray diffraction for clay mineral identification and quantification. The major element composition of the core sections shows no significant chemical variation within the sample depth. Composition ranges as oxides are: Si 46-57%, Al 15-25%, Fe 10-15%, Na 1.5-5%, Mg and K both 1.5 to 2%, Ca and Ti both 1 to 1.5%. Total S varies from 2.2% at the surface to more than 10% in the lower sections. The clay minerals are irregular mixed-layers of smectite-illite, kaolinite, and illite. The texture and shape of the clay particles and pyrite grains was determined by scanning electron microscopy. The mixed- layers of smectite-illite are well crystallised with tabular particles of 2 to 5 microns. Pyrite develops in two distinctive forms, round framboids (~5 microns) and elongate clusters of crystals (~50 microns). The crystals are well rounded and dodecahedrons, sometimes coated with iron oxides.

Introduction

Acid conditions in sediments and some stream and groundwaters are common on many coastal plains which were subject to Late Pleistocene-Holocene sealevel fluctuations and the lower course of the Pimpama River is a typical example. The river is located 80 km south of Brisbane. It originates in Darlington Range and flows east into southern Moreton Bay. The headwaters consist of low ranges with steep slopes covered in native vegetation, the central course meanders over a flat coastal plain, which in this area is cultivated with sugarcane, and the estuary is sheltered by a sandy barrier island.

The area is characterised by subtropical humid climate with an average rainfall of 1 300 mm per year. About 70% of this amount falls during the wet season from November to April and is often related to storms. This seasonal rainfall regime is an important component of the geochemical processes that occur in the mid-lower course of the river.

The geology of the area consists of two main units: (a) Devonian-Carboniferous rocks represented by greywacke, different sandstones and shale interbedded with chert, which form the high parts and the basement of the catchment; and (b) Pleistocene-Holocene sediments represented by a large range of deposits from clean sands to clean clays. The recent geological evolution of the Pimpama region is linked to the development of the Moreton Bay and its islands, and influenced by Late Quaternary sealevel fluctuations and changing fluvial regimes.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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An important lithological feature of the Holocene history is the formation of sulfidic sediments with high potential to produce acidity. The sediments of concern are Holocene estuarine muds, mainly clays, rich in organic matter, iron, aluminium, and sulfur. The occurrence of acid sulfate soils and the subsequent aluminium toxicity are important environmental issues of the area

Geochemical Processes

The coastal wetlands and tidal flats related to the Holocene fluctuations of sealevel provided ideal conditions for the production of sedimentary pyrite (FeS_2), a source of sulfur and iron. In the reducing conditions of these subtropical estuaries, the decomposition of organic material generates energy for the activity of sulfate reducing bacteria. These anaerobic bacteria reduce sulfate from seawater to sulfide, and sedimentary ferric iron to ferrous iron. The end product is fine-grained pyrite which accumulates within sediment particles (Dent, 1986; Dent & Pons, 1993).

Pyrite is stable under anaerobic conditions, however, oxidation and hydrolysis of the layer containing sedimentary pyrite can result in production of sulfuric acid (H_2SO_4). This acid destroys the structure of the sediments where it forms and leads to leakage of elements such as Al, Fe, and other minor components. The water quality of nearby rivers can be affected not only by sulfuric acid but also by the associated presence of toxic metals.

Sampling Procedures and Analytical Methods

The sediments have been examined by shallow cores (about 1 m) collected from the intertidal zone of the river. Several water quality surveys have been conducted throughout the catchment and during different seasons. In this paper we examine in detail one core (C9), located 9.6 km from the mouth of the Pimpama River, few metres upstream from a footbridge (Figure 1.) The core was collected manually by a 5 cm diameter PVC pipe driven into the soft material of the river bed, parallel to the steep side. After collection cores were split into two, sealed, and stored under refrigeration. One half was retained as an archive split. Soon after sampling, pH and conductivity were measured along the core, directly into the wet sediment. Eight vertical sections of the core were sampled (one every 20 cm) and analysed by a variety of methods: total digestion (hydrofluoric-aqua regia) using ICP for determination of major chemical

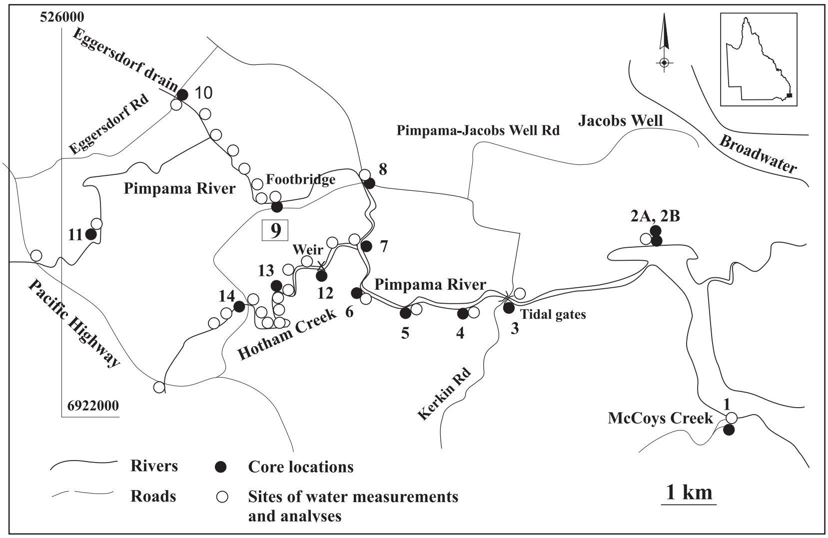

Figure 1. The lower Pimpama River and locations of sediment core and water samples. Location of core C9 is shown by the square.

elements, Leco to determine the percentage of total sulfur, X-ray diffraction (XRD) for clay mineral identification and quantification, and scanning electron microscopy (SEM) for study of the texture and shape of the clay particles and pyrite grains.

Results

The shallow sediment of the river bed is represented by silty clay: reddish from 0 to 11 cm, black from 11 to 24 cm, and dark grey from 24 to the bottom, 131 cm. The silty clay of the river bed was "ripe" (Dent, 1986) at the surface (pH = 4.7) and "unripe" (Dent, 1986) below 10 cm (pH = 5.3-7.2). Conductivity dropped significantly from the surface, 10.5 mS/cm, to 1.3 mS/cm at the depth of 120 cm, showing a rapid decrease in the influence of the saline river water with depth in the sediment (Table 1).

The major element composition of the core sections (C9-0, C9-20, C9-40, C9-60, C9-80, C9-

Table 1. Lithological and physical character of Core 9 from the Pimpama River.

Lithology	Depth (cm)			pH	Conductivity (mS/cm)
Reddish clay	11	0		4.7	10.5
Black clay	24		20	6.5	7.5
			40	7.2	6.2
			60	7.2	3.7
Dark grey silty clay			80	6.8	2.3
			100	6.7	1.8
			120	5.7	1.3
		131		5.3	1.3

100, C9-120, and C9-131) shows no significant chemical variation with depth. The content of major elements as oxides is: 46-57 % silica, 15-25 % aluminium, 10-15 % iron, 1.5-5 % sodium, 1.5-2 % magnesium and potassium, 1-1.5 % calcium and titanium. Sulfur varies from 2.2 % at the surface to more than 10 % in samples such as C9-20 (black clay), C9-80 and C9-100 (grey clay).

Material at C9-20 and C9-60 reflect the character of the entire core and have been investigated by X-ray diffraction. Both samples contain similar amounts of irregularly mixed-layers of smectite-illite, kaolinite, and illite. Quantified XRD results for C9-60 are: 55.3 % montmorillonite, 19.5 % illite, 4.8 % kaolinite, 12.6 % quartz, 4.5 % feldspars, and 3.3 % pyrite. Scanning electronic microscopy revealed that the mixed layer of smectite/illite is well crystallised with tabular particles of clays ranging in size from 2 to 5 microns.

Pyrite develops in two distinctive forms, framboids and clusters. The framboids are spherical, some only several microns diameter and grow within the clay particles (Figure 2). Other pyrite framboids are larger and grow close together in groups. The clusters have rounded elongate shapes and are larger, around 50 microns in length (Figure 3). The crystals of pyrite are well developed and mainly dodecahedrons in form with rounded sides and corners; sometimes they are coated with iron oxides (Preda & Cox, 1996).

Discussion

Core 9 is representative of sediments in the central and lower catchment of the Pimpama River. Many of the cores collected along the river and its tributary, Hotham Creek revealed the same grey silty clay with similar chemical composition and mineralogy, and similar pyrite content.

Acid production is controlled by two main factors: (a) the seasonal regime of rainfall and (b) the pyrite content. During the dry season the water table drops and the pyritic layers within

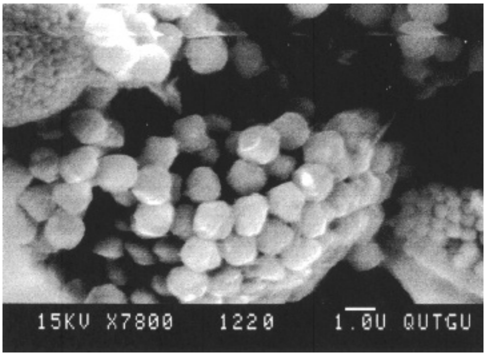

Figure 2. Framboids of pyrite from Pimpama River sediments.

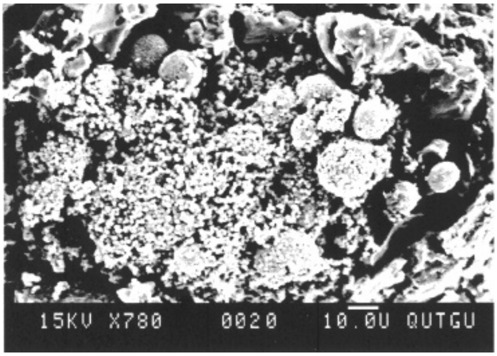

Figure 3. Cluster of pyrite surrounded by framboids from Pimpama River sediments.

the clays oxidise. Occasional showers during the dry season, then followed by the heavy rains during the wet season flush the acid into the rivers, along with high amounts of sulfates of iron and aluminium.

Therefore, the water quality of the rivers is a good indicator of the processes which take place in the adjacent sediments. A good example is represented by the water measurements conducted at the footbridge on the Pimpama River (the core site) on 15 & 16 August 1995. On the 15th, the surface water was characterised by pH = 7.5 and salinity = 30 000 mg/L. The next day, after the heaviest rain of that dry month (40 mm) the surface water had pH = 3.7 and salinity = 12 500 mg/L.

The Core 9 sediments are typical of many other subtropical catchments which were subject to Holocene sealevel fluctuations. These coastal deposits largely exist in an unripe condition and represent the original estuarine muds. Over time and with exposure, the development of acid condition has enhanced the breakdown of silicate minerals with subsequent formation of soil.

Acknowledgements

We thank the following for assistance with analytical work: S. Price for chemical analysis of waters and sediments, W. Kwiecien for scanning electron microscopy analysis, and A. Raftery for X-ray diffraction analysis. We also thank J. Ferris from the Department of Environment for assistance in field monitoring. The managers of the Palm Nursery on Hotham Creek are thanked for rainfall records.

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The Impact of Mosquitocides on Non-Target Estuarine Fauna from Moreton Bay: Acute Toxicity Trials

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Abstract

Many compounds are being used to attempt control of larval mosquitoes in aquatic habitats, often with hazardous or unknown effects on associated non-target species. Properties which initially made many of these chemicals appear useful, such as long residual action and broad spectrum activity, have in retrospect been found to create serious environmental problems.

In the Moreton Bay region, local authorities regularly spray mangrove and salt-marsh habitats with chemical insecticides during summer months. These same habitats are important as nursery environments for a wide diversity of fish and crustaceans of ecological and commercial importance. Results are presented from acute static toxicity tests on four species of organisms which are common in these habitats (copepod *Metis holothuriae*, mysid shrimp *Haplostylus udrescui*, shrimp *Leander tenuicornis*, fish *Pseudomugil signifer*), using five different mosquito larvicides. These include Altosid Liquid Larvicide® (active ingredient [AI]: 20% s-methoprene), Sumilarv® (AI: 2% pyriproxyfen), VectoBac12AS® (AI: 1200IT U/mg *Bti*), Abate 100E® (AI: 10% temephos) and Actelic® (AI: 90% Pirimiphos-methyl).

The organophosphate compounds temephos and pirimiphos-methyl were the most toxic of larvicides tested, with LC_{50} values for the shrimp species being at concentrations well below the estimated field concentration (E.F.C.). All species tested were tolerant to the other larvicides at the E.F.C. and well above. Field trials with caged populations, and sub-lethal and chronic long-term effects on growth, survival and fecundity of these and other organisms continue.

Introduction

An increasing range of compounds is being used in attempts to control larval mosquitoes and midges in aquatic habitats, often with hazardous or unknown effects on associated non-target species (Mulla *et al.,* 1978; Pierce *et al.,* 1988; Nipper *et al.,* 1993; Hershey *et al.,* 1995). Knowledge of the toxicities of pesticides used in control is essential for responsible management. Properties which initially made many of these chemicals appear useful, such as long residual action and broad spectrum activity, have in retrospect been found to create serious environmental problems. In response to legislation and public pressure, local authorities in southeastern Queensland, regularly spray mangrove and salt-marsh habitats with chemical insecticides, in attempts to control mosquito breeding. These same habitats are important nursery environments for a great diversity of non-target species and their food organisms, including fish and crustaceans of ecological and commercial importance (Morton *et al.,* 1988; Sumpton & Greenwood, 1990; Coull *et al.,* 1995). Any adverse disturbance to populations of such species may have impacts at higher trophic levels, threatening the integrity and survival of the fundamental estuarine food web.

For the past 25 years in eastern Australia, saltmarsh mosquito control programs have been reliant

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 251-256.

on the application of various formulations of Abate® (active constituent: temephos) to control *Aedes vigilax* (Skuse) and *Culex sitiens* Wiedemann larvae (Kay *et al.,* 1973). Surprisingly, despite the widespread and prolonged usage of temephos, there is little published information available on the susceptibility of Australian non-target species to this compound (Kay *et al.,* 1973; Gehrke, 1988; Mortimer & Hughes, 1991; Mortimer & Chapman, 1995) and much of that has only become available *a posteriori*. There are growing environmental concerns over temephos usage, and mounting evidence of mosquito resistance to its use (Cousineau, 1992). Alternative compounds are now being evaluated for mosquito control purposes.

This study examined the acute toxic effects of a number of pesticides used for mosquito control purposes in Australia, on four non-target species from the Moreton Bay region of south-eastern Queensland. The species were selected because: (1) they are the more common inhabitants of the shallow estuarine and wetland margins of the region (Wadley, 1978; Morton *et al.,* 1988); (2) they are of known importance in estuarine food webs (Sumpton & Greenwood, 1990; Coull *et al.,* 1995; authors' observ.); (3) there is a coincidence of these species' maximum breeding seasons with that of mosquitoes and hence dosing programs; and (4) they are readily maintained in laboratory culture.

Materials and Methods

Collection, maintenance and identification of test species

The fish (*Pseudomugil signifer* Kner), palaemonid shrimp (*Leander tenuicornis* Say) and copepod (*Metis holothuriae* Wilson) species were collected from salt-marsh pools near Coomera Marina (27°54S, 153°17E) in southeastern Queensland. Late juvenile to adult *P. signifer* and *L. tenuicornis* were collected in 25 x 25 x 45 cm, 2 mm mesh bait fish traps (Mossop's Tackle Pty Ltd, Brisbane, Australia). *M. holothuriae* was collected using hand held (10 x 20 cm) 280 µm mesh nets. The mysid shrimp (*Haplostylus udrescui* Greenwood *et al.,*) were collected from the Brisbane river estuary in the St. Lucia region by towing a sledge-mounted net (mouth 0.5 x 0.3 m, 500 µm mesh) along the shallow mangrove margins.

Collected specimens were placed in aerated habitat water for transport to laboratories in Brisbane. Additional habitat water was also collected for subsequent maintenance and experimental purposes. To remove detritus, all water for experimentation and maintenance of test animals was passed through a 100 µm mesh net prior to use.

In the laboratory, shrimp and fish were sorted and then transferred to separate 24 x 22 x 46 cm aerated aquaria for a 3-4 day acclimation period. Recently hatched *Artemia salinus* nauplii (Marine Laboratory, Hayward California, USA) and Wardleys's Goldfish Food (Wardley Corporation, Seaucus, New Jersey, USA) were provided as food. The copepods were maintained in 2 L sorting trays on detritus present in the habitat water. Only specimens active after the 3-4 day acclimation period were used in trials. *P. signifer*, *L. tenuicornis*, *H. udrescui* and *M. holothuriae* were identified according to the descriptions in Grant (1982), Wadley (1978), Greenwood *et al.,* (1991) and Wilson (1932).

Pesticides evaluated

In order to evaluate the effects of pesticides utilised in field applications, we tested: (1) Abate 100E® (Active Ingredient [AI]: temephos, Cyanamid Australia Pty Ltd); (2) Altosid Liquid Larvicide® (AI: s-methoprene, Sandoz Ltd., Dallas, USA); and (3) VectoBac 12AS® (AI: 1 200 International Toxic Units [ITU]/mg *Bacillus thuringiensis* var. *israelensis* (*B.t.i.*), Hoechst Schering AgrEvo Pty Ltd, NSW Australia). Dilution with a solvent was unnecessary with these products.

Acute toxicity trials

To provide an estimate of the concentration of pesticide that causes direct irreversible harm to non-target species, a series of static exposure assays were designed according to criteria specified by the American Society for Testing and Materials (ASTM 1980), for acute toxicity testing of fish and macroinvertebrates. In these assays, the test animals were exposed to serial dilutions of a larvicide in filtered habitat water, with no change of water for the duration of the assays. Three replicates each of 20 late-juvenile to adult specimens were introduced into 20 x 20 x 30 cm (12 L) glass aquaria containing 5 L of test concentration. Three negative control containers holding 20 test specimens each in habitat water without pesticide were used for each bioassay. Testing was conducted in two stages. Firstly, range finding tests with widely spread exposure concentrations were conducted. These were then followed by tests focusing on a narrow range of concentrations which straddled the effect range deduced from the first stage. In total a minimum of 10 concentrations were used. Test specimens were individually removed from the holding aquaria and distributed randomly amongst the test containers. To minimise variability due to nutritional and metabolic condition, shrimps were not fed for 24 h prior to, or during testing. The assays were conducted at 25°C under a light dark cycle of 12:12 h. Death or the lack of reaction to gentle prodding with a glass pipette, was the measured deleterious response. The numbers surviving were counted at 24 h intervals for 96 h and dead animals were removed from the test containers at each evaluation.

Initially, due to current environmental concerns, all species were tested against temephos. Based on these evaluations, the most sensitive species was then selected for evaluation against the other pesticides being evaluated.

Statistical analysis

Probit models were used to model mortality as a function of pesticide dose. To avoid infinite logarithmically transformed values, zero concentrations were analysed as concentrations of 0.001 µg/L. The relationship between pesticide concentration and the percentage of exposed organisms affected was determined, and concentration-mortality curves plotted. The SPSS--PC+ version 4.0 PROBIT (Norusis, 1990) was used for these analyses. The LC_{50} values and associated 95% confidence intervals are presented.

Results

The estimated field concentrations (E.F.C.) of active ingredient in 15 cm deep pools for the pesticides evaluated, ranged from 0.008 µ/L for s-methoprene to 0.03 mg/L for temephos (Table 1). Temephos was highly toxic, with LC_{50} values for the three crustacean species being at or well below the E.F.C. (Table 2). *Haplostylus udrescui* and *L. tenuicornis* were especially sensitive, with LC_{50} values being at 4% and 33% of the E.F.C., respectively. *Pseudomugil signifer* was tolerant to temephos at concentrations up to 20x the E.F.C. *Haplostylus udrescui*, the most sensitive of the crustacean species tested against temephos, was tolerant to s-methoprene and *B.t.i.*, at >200 times the E.F.C. (Table 3).

Discussion

Temephos had high acute toxicity to all crustaceans tested, and for the mysid and shrimp, at concentrations well below application levels. This is consistent with findings elsewhere on the toxicity of temephos to non-target species of crustaceans, including shrimp, mysids, cladocerans and crabs (Ward & Busch, 1976; Mulla *et al.,* 1978; Pierce *et al.,* 1989; Australian EPA, 1994; Mortimer & Chapman, 1995). Indications are that temephos should only be used in situations where such species do not occur.

Table 1. Selected pesticides and the estimated field concentration (E.F.C.) of active ingredient (AI) for 15 cm deep pools.

Pesticide	AI	Application rate	E.F.C.[a]
Abate (100E)	10% temephos	0.1 kg (AI)/hectare	0.03 mg/L
VectoBac 12AS	1 200 ITU/mg *B.t.i.*	288 mg (AI)/hectare	345 600 ITU
Altosid Liquid Larvicide	20% s-methoprene	0.06 kg (AI)/hectare	0.008 μg/L

[a] calculation based on the application rate of active ingredient in a 15 cm deep pool.

Table 2. LC_{50} values (expressed in mg/L temephos) for 96 h Abate (100E) bioassays of three crustacean and one fish species, based on probit models of mortality.

Species	LC_{50}[a]	Slope (SE)	*P*[b]
Leander tenuicornis	0.0112 (0.0099, 0.0126)	4.050 (0.369)	0.262
Haplostylus udrescui	0.00013 (0.000007, 0.00020)	1.461 (0.136)	0.808
Metis holothurae	0.0290 (0.0101, 0.0798)	1.530 (0.115)	< 0.001
Pseudomugil signifer	0.590 (0.570, 0.611)	0.723 (1.075)	2.37

[a] values in parentheses are the 95% lower and upper fiducial limits;
[b] refers to the probability corresponding to maximum likelihood chi-square statistic for goodness of fit of the model.

Table 3. Concentrations of pesticides causing 50% mortality of *Haplostylus udrescui* after 48 h exposure.

Active ingredient	E.F.C.[a]	LC_{50}[b]	Slope (SE)	*P*[c]
Temephos < 0.90 (mg/litre)	0.03	0.00039 (0.00031, 0.00048)	3.382 (0.310)	
B.t.i. < 0.25 (ITU)	0.34×10^6	91.9×10^6 (58.4×10^6, 144.6×10^6)	1.281 (0.295)	
s-Methoprene < 0.25 (μg/litre)	0.008	2.28	1.088	

Results from these acute static tests show that s-methoprene and *B.t.i.* can be safely applied in situations where the test species occur. In their comprehensive review of the non-target effects of *B.t.i.*, Lacey & Mulla (1990) and Hershey *et al.* (1995) have similarly found that there are few adverse consequences of *B.t.i.* use. These studies detailed population effects on 75 genera of stream fauna (Lacey & Mulla, 1990). Chironomids have been found to be susceptible to *B.t.i.*, and this is probably the result of their filter-feeding behaviour (Charbonneau *et al.,* 1994). The literature contains no evidence of impact on crustaceans or fish.

Mysids and shrimps make up a significant proportion of the diet of estuarine juvenile stages of many fish species in the region from which the present test species were collected (Sumpton & Greenwood, 1990). Of the four species tested here, the two malacostracan crustaceans were the most sensitive, with the mysid *H. udrescui* being especially so. Many other studies have made similar findings with different mysid species, with *Mysidopsis bahia* having become a standards choice for evaluating toxic effects in saltwater systems (reviewed by Nimmo & Hamaker 1982; Nipper *et al.,* 1993; Rand, 1995a). *Haplostylus udrescui* appears to equally well satisfy the criteria desirable for both acute and chronic toxic test procedures (Rand, 1995a) and is clearly a species which should be further explored in this context.

It is acknowledged that there are other considerations to be explored, both biological and environmental, which may affect concentrations of treatment compounds in the field situation. These include tidal flushing and dispersal (Pierce *et al.,* 1989), adsorption to organic matter (Cooney, 1995), the size and life-history stages of the non-target species (Mian & Mulla, 1992), and synergistic effects with other pollutants (Rand, 1995b; Elcin, 1995). Conclusions drawn from laboratory tests therefore tend to be conservative. Also, in our static tests no analysis of the concentrations of pesticide in the exposure tanks throughout the 96 h exposure period was completed. Accordingly, results tend to underestimate toxicity, as the available toxicant is very likely to reduce with time through absorption to the glass, evaporation and degradation. However, our "pulse" exposure method does simulate field applications of pesticides for mosquito control.

Similarly, acute lethal tests cannot indicate likely chronic sublethal influences on fecundity, growth and survival of life-history stages. These chronic effects may be equally deleterious to the ecology of these species. In the present case, field trials and sub-lethal chronic long-term effects remain to be tested.

Acknowledgements

We thank the officers representing the Local Authorities Research Committee for their support and encouragement. H. Standfast (International Vector Consultant), S. Ritchie (Tropical Public Health Unit, Cairns), P. Mottram (State Health Department, Brisbane), H. Chapman, P. Ryan and C. Jennings (Queensland Institute of Medical Research) provided useful discussion and guidance. P. Mason (Gold Coast City Council), J. Milton and members of the Department of Zoology Department, The University of Queensland provided technical assistance.

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Evaluating Water Quality Management Strategies for Moreton Bay: A Decision Analytic Framework for Integrating Scientific Information with Stakeholder Values

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Abstract

This paper describes a decision analytic framework to assist stakeholders in systematically providing inputs to the development of a water quality management strategy for Moreton Bay. With assistance from a decision analyst, stakeholders will collectively define and structure the decision problem, then proceed individually with the rest of the analysis. The problem confronting a stakeholder may be viewed as deciding what combination of management proposals to recommend, given the available scientific information. To approach this problem, a stakeholder will be guided through a portfolio decision analysis. The ultimate result would be the identification of a portfolio of proposals that is consistent with the values of the stakeholder and the available scientific information. The process is expected to generate insights on the values of stakeholders, the scientific information necessary to evaluate alternatives for water quality management, and the features of desirable alternatives.

Introduction

The Councils of Brisbane, Caboolture, Ipswich, Moreton, Pine Rivers, Redcliffe and Redlands and the Queensland Departments of Environment and Heritage and Natural Resources, have launched the Brisbane River & Moreton Bay Wastewater Management Study (BRMBWMS). The ultimate goal of the study is to develop a water quality management strategy for Moreton Bay and its catchment. The study includes two major components: (1) intensive scientific studies of the Bay; and (2) a program of public consultation and community involvement.

The BRMBWMS faces the task of meshing the activities and results of its scientific investigations and its community involvement program so that the two project activities support each other in the development of the water quality management strategy. This paper outlines an exercise that could facilitate such integration.

The exercise would use decision analysis as a framework within which stakeholders could use their values and the available scientific information to create and evaluate management alternatives. The design of a water quality management strategy for the Bay is a decision problem, and one that is made complicated by several factors:

Multiple objectives. Water quality management involves balancing several objectives. For example, it would be desirable to have a management system that simultaneously maximises environmental protection, minimises restrictions on economic activities and minimises implementation costs.

Multiple stakeholder groups. There are several sectors in the catchment and coastal area with stakes in the outcome of the water quality management strategy.

Uncertainty. At this time much is unknown about the Bay as a system and it will probably remain so in the near future. Thus, uncertainty would complicate the evaluation of any proposed management activity.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 257-266.

No clear-cut alternatives. At this stage management alternatives have yet to be specified and there is much room for innovative strategies.

Decision analysis offers an approach for systematically dealing with the above complexities. It provides methods for making decisions using a "divide and conquer" strategy. A complex problem is divided into parts that are separately analysed. Results of separate analysis are later integrated to suggest the preferred course of action (Keeney, 1982). Decision analysis is widely regarded as the method of choice for dealing with multiple objectives and uncertainty. Recent developments have extended its traditional role from evaluating alternatives to creating them (see Keeney, 1992).

Decision analysis is used in a wide range of environmental decisions. Its methods or aspects of decision analysis have been used to enhance communication among decision makers and stakeholders in decisions regarding national energy policy (Keeney *et al*., 1990), offshore oil drilling, siting of dams for flood control (Edwards & von Winterfeldt, 1987), coal mining in an environmentally valued area (Gregory & Keeney, 1994), demand-side management by a utility company (Hobbs & Horn, 1997) and fisheries management (Luna, 1994; Luna *et al*., 1995).

The Proposed Procedure

The objective is to provide insights that would aid in the development of a water quality management strategy for the Bay. Decision analysis would be conducted with each stakeholder to select the combination of management activities that could best achieve the objectives of a stakeholder. [This formulation of the problem differs from what is encountered in a typical decision analysis. Decision analysis is normally used to select the best alternative from a set of mutually exclusive alternatives. When the task is to select the best combination of non-exclusive alternatives, the analysis is modified into a portfolio decision analysis (Kirkwood, 1996).] In the process, insights are expected to be generated regarding the values held by stakeholders, the scientific information necessary to evaluate alternatives, and the desirable properties of alternatives.

An analyst is required to assist each stakeholder individually in going through the decision analysis. It is the analyst's role to explain all procedures and results to the participants throughout the exercise and to adjust techniques so that they suit individual participants.

With each stakeholder the analyst would start with a simple decision model based on easily understood methods and best available estimates. The analyst would then conduct sensitivity analysis to determine if additional modelling and information gathering are needed in subsequent iterations. Modelling would stop when the stakeholder indicates that the decision problem has been sufficiently clarified and they agree with the resulting mix of proposals. The procedure is divided into the following steps: problem structuring, estimating impacts of proposals, modelling preferences of stakeholders and determining an optimal portfolio or mix of proposals. In the initial phase of analysis, all steps are conducted without considering uncertainty. Sensitivity analysis is then conducted to determine if particular steps have to be repeated with the inclusion of uncertainty, and possibly other factors (Figure 1).

The analytical approach borrows much from Philipps' (1982) *requisite decision modelling* and Matheson & Howard's (1983) *decision analysis cycle*. These iterative approaches begin with simple model structures and methods and introduce more complex modelling only when justified by sensitivity analysis. The aim is to use sufficient analytical resources to clarify the problem for a decision maker and to prevent an over analysis of the problem.

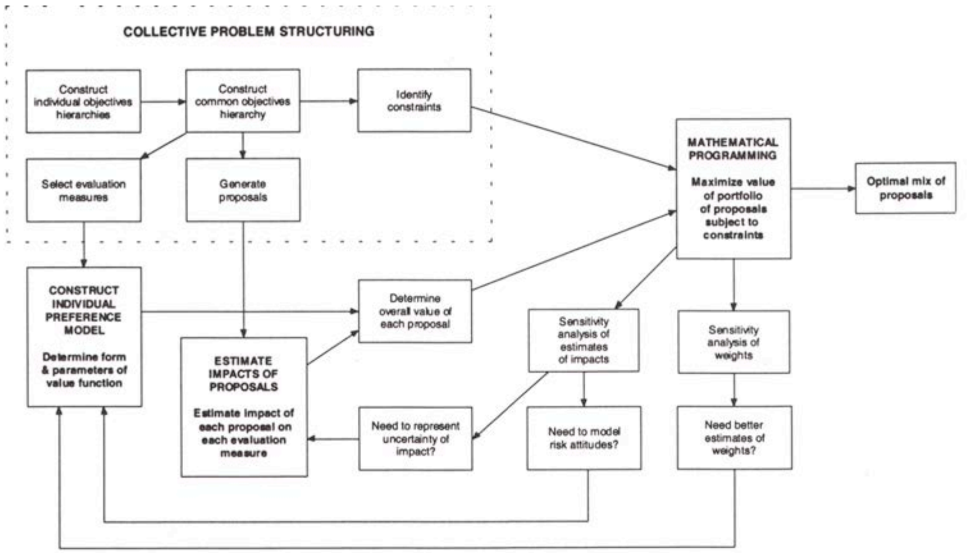

Figure 1. The proposed decision analytic framework.

Problem Structuring

Problem structuring includes (1) identifying and structuring objectives, (2) specifying evaluation measures, (3) identifying and creating management proposals and (4) identifying constraints. The initial phase of problem structuring would be done individually with each stakeholder. The analyst would then facilitate agreement on a common problem structure for all stakeholders to make the rest of the analysis manageable.

In dealing with a complex decision problem with no clear alternatives, it is preferable to commence by clearly defining the objective. Keeney (1988) describes the procedure for structuring of the objectives of multiple stakeholders. The first step is to elicit objectives separately from each stakeholder. In the initial interview the analyst discusses areas of concern with a stakeholder and attempts to identify all objectives that a stakeholder would like to achieve in the situation, without ranking or priorities. The result of the first interview is an unstructured listing of objectives.

The objectives are then separated into fundamental objectives and means objectives. Fundamental objectives are ultimate ends that are important in themselves while means objectives are important for achieving fundamental objectives. For each objective in the unstructured list the analyst repeatedly asks the stakeholder why it is important in order to establish means-ends relationships and ultimately to identify fundamental objectives. To clarify and structure them, they are arranged into a hierarchy where objectives in higher tiers are explained by objectives in lower tiers (Figure 2).

The next step is to combine the fundamental objectives of the stakeholders into a single hierarchy. To do this, the analyst lists all top-tier objectives from the individual hierarchies. The list may be condensed by removing redundancies and identifying natural hierarchies. This results in a common set of top-tier objectives. Next, each individual hierarchy is examined and the objectives in its next tier are listed under an appropriate objective in the common set of top-tier objectives. The common hierarchy will now have two tiers. The process is repeated for all tiers until a single hierarchy is created. The draft hierarchy is then checked and revised so that it conforms with the desired properties of a fundamental objectives hierarchy (see Keeney, 1992). The analyst then presents the draft hierarchy for approval to stakeholders who will check if their concerns are included and suggest ways to improve the common hierarchy.

Next, evaluation measures (also called either criteria, attributes or performance measures) are defined for each objective at the lowest level of the fundamental objectives hierarchy (Figure 2). An evaluation measure is indicative of the degree to which the associated objective is achieved. For example, if the objective is to "minimise costs", then the evaluation measure might be "cost in dollars". Sometimes there is no convenient evaluation measure to describe the achievement of an objective. In this case it is necessary to construct a stepwise scale where each distinct level of impact is assigned a scale number. The set of evaluation measures should be complete, non-redundant, decomposable, operational and minimal in size (Keeney & Raiffa, 1976). Inputs from scientists are needed to identify appropriate evaluation measures. Stakeholders must approve the set of evaluation measures as representative of their concerns and, more importantly, as tools they can use for evaluation.

The next step in problem structuring is to generate proposals. Proposals have been suggested for managing water quality in the Bay. It may be desirable to enlarge this initial set of proposals to ensure that promising ones are not left out of the analysis. For this exercise a proposal does not have to be a complete project proposal that is ready for submission to a funding agency. Rather, a proposal should have sufficient detail that it can be characterised by the set of evaluation measures. The analyst must strive to involve the stakeholders in the creative process of generating proposals. After all, proposals are the means for achieving their objectives. Keller & Ho (1988) compiled and reviewed various creativity-enhancing strategies for generating proposals. For example, each objective can be considered separately to suggest a proposal. The common purpose of these strategies is to stimulate creativity and prevent premature evaluation and censoring. Initially a large set of proposals should be generated after which a winnowing process is done so that the analysis can focus on promising alternatives.

Stakeholders must agree on a set of proposals for evaluation, at least initially. The generation of additional proposals throughout the course of the analysis is a normal and most welcome event. In fact, the analyst should actively encourage the creation of proposals or alternatives because sometimes the creation of a good alternative is enough to solve a decision problem. However, an initial set of proposals must be agreed upon to get the process underway. During the course of problem structuring constraints must be identified. For instance, it would be advantageous to have an estimate or a range of estimates of the ceiling cap on the total cost of proposed management activities.

At the end of problem structuring, the stakeholders must agree on a set of proposals and a set of evaluation measures. This will make the next step of estimating impacts of proposals manageable. After this the stakeholders will individually evaluate the proposals with guidance from the analyst. Stakeholders who object to particular proposals or objectives during the problem structuring stage will be informed that they will later have the chance to rate these proposals or objectives accordingly when their preferences are quantified.

Estimating Impacts of Proposals

Once the problem has been structured, the analysis proceeds to the forecasting of impacts of each proposal, as measured by each evaluation measure. A matrix of impacts can be prepared with the alternatives as column (or row) headings and the evaluation measures as row (or column) headings (Table 1). The objective of this part of the analysis is to fill up the cells of the matrix with estimates of impacts. The intent is to summarise the scientific information needed for decision making in a single matrix.

Information to fill the matrix can come from available studies, initial results of ongoing modelling work or the professional opinion of scientists knowledgeable about the Bay. In the initial round of analysis a best point estimate and range of each impact are required. The range of the uncertain impact can be regarded as the 10th percentile and 90th percentile of its probability distribution (Matheson & Howard, 1983).

Quantifying Preferences of Stakeholders

In a decision problem with multiple and often conflicting objectives, not all objectives can be achieved simultaneously. Trade-offs are thus unavoidable, and it is the analyst's job to assist the stakeholder in expressing these trade-offs precisely. The analyst must capture the stakeholder's trade-offs in a form that is suitable for combining with the summarised scientific information. To do this, the analyst constructs a quantitative model of the stakeholder's preferences. The parameters of this model are elicited from the stakeholder and should reflect the stakeholder's values or priorities. The input to this model is the estimated impacts of a proposal. The output is a single measure of the overall predicted performance of the proposal. Higher levels of this measure are preferred. By applying this model to all proposals in the set, their ranks in terms of overall performance can be established.

Essentially, the analyst asks the stakeholder to weight each objective and thus establish its relative importance. A weighted average of the estimated impacts of a proposal can then be obtained as an overall measure of the proposal's predicted performance, as long as the impacts are made comparable by re-scaling them into a common dimension or unit. In symbols, the overall performance or value of a proposal is given by:

where x_i is the predicted impact level of the proposal on the i-th evaluation measure, w_i is the weight that a stakeholder assigns to the i-th objective and v_i is a single dimensional value function.

The latter converts the original unit of an impact into a common unit, among other things.

The above is called an additive value model. It computes the overall value of a proposal as the weighted sum of its impacts, where the weights reflect the preferences of the stakeholder. The w_i's are "swing weights" that indicate a stakeholder's assessment of the relative importance of changing (swinging) an evaluation measure from its worst to its best level compared to a similar change in another evaluation measure. For example, suppose that two evaluation measures are total cost of a proposal with a range of 0-10 million dollars and concentration of a pollutant with a range of 5-25 ppm. In an elicitation session with an analyst, a stakeholder might indicate that reducing pollutant concentration from 25 to 5 ppm is two times more important than reducing the cost of a proposal from 10 million dollars to nothing. Von Winterfeldt & Edwards (1986) and Edwards & Barron (1994) describe the procedure for eliciting swing weights, which is called the Simple Multiattribute Rating Technique with Swings (SMARTS).

In the first round of analysis, it is assumed that the single dimensional value functions are linear. That is, the impacts are simply re-scaled by proportional scoring, thus converting all impacts into a common unit and making them comparable. An analyst could elicit a curved single dimensional value function in order to model a stakeholder's notions about returns-to-scale with regard to an impact. However, the next step in the analysis would be more complicated if non-linear functions are introduced.

Table 1. An example of a framework for a matrix of impacts for decision making, relevant to the management of Moreton Bay.

Proposals / Evaluation measures	Water quality controls	Water quantity controls	Recycling/ re-use	Catchment management	...
Volume of point source discharges					
Types & concentrations of point source contaminants (constructed scale)					
Percent of catchment in erosion-prone land uses					
Stormwater impacts (constructed scale)					
Probability of seafood contamination					
E. coli MPN					
Restrictions on economic activities (constructed scale)					
Cost of proposal					
Efficiency of water quality management (constructed scale)					
No. of adversely affected groups					
General public support (constructed scale)					

In most cases an additive value model would be an appropriate model of the stakeholder's preferences. There are other models of preferences and in practice, the analyst would first check independence conditions to select the appropriate preference model.

Determining an Optimal Portfolio of Proposals

For each proposal, the preference model computes an overall measure of value based on the proposal's predicted impacts and the preferences of the stakeholder for those impacts. We can thus rank-order the proposals according to overall value. We can also imagine that in the portfolio of proposals the overall values of the component proposals are combined into a total value of the portfolio. Suppose that one constraint identified in problem structuring is a total budget ceiling for all management initiatives. The question then is, which proposals should be included in the portfolio to maximise the total value of the portfolio, given that the total cost of proposals included should not exceed the budget constraint?

This optimisation problem can be solved using 0-1 integer programming because the stakeholder's preferences were previously modelled by linear relationships (Kirkwood, 1996). The result of this step is a combination of proposals that gives the highest total value of the

$$v(x_1, \ldots, x_N) = \sum_{i=1}^{N} w_i v_i (x_i)$$

portfolio while satisfying the budget constraint. Golabi *et al*. (1981) used this approach to help the US Department of Energy select an optimal combination or portfolio of research projects. The approach can also handle other constraints (e.g. diversity in type or location of projects). In an interactive session with a group of decision makers, Golabi *et al*. successively introduced constraints to demonstrate the effect of each on the portfolio of projects. The exercise produced many insights and eventually the decision makers selected a portfolio that balanced their major concerns.

Sensitivity Analysis

The previous step would produce a first-cut combination of proposals that a stakeholder might recommend because it was based on their examination of the pertinent scientific information and their reflection on their priorities. Because the calculations that produced the combination of proposals involved many estimated variables, a stakeholder might (or should) ask what would happen if one or more variables were to change. Would this change the composition of the recommended portfolio of proposals?

The question can be answered by sensitivity analysis. In sensitivity analysis, the value of one variable is swept through its range while other variables remain at their base values. A variable that causes a change in the portfolio while being swept through its range must be re-examined and more carefully estimated if we are to have confidence in our decision. The most important things that should be subjected to sensitivity analysis are the estimated impacts of the proposals and the stakeholder's weights.

Additional analysis after sensitivity analysis

If the decision is sensitive to a particular impact estimate, two things must be done: first, we must represent the uncertainty of this impact and second, we must assess the stakeholder's risk attitude towards this uncertainty and incorporate this risk attitude in the stakeholder's preference model.

Representing the uncertainty of an impact means assessing a probability distribution over its possible levels. Scientists could build models or modify existing models so that their outputs are probability distributions, but this is not always possible. It is expected that scientists will not have considered the impact of a particular proposal, because specific proposals were not defined when they started their research. Thus, in most cases the stakeholder will have to rely on the opinion of scientists.

In decision analysis formal procedures are used to quantify expert opinion or to elicit so-called subjective probability distributions. The most widely recommended procedure is described by Spetzler & Stäel von Holstein (1975) and Merkhofer (1987). The analyst formally interviews a scientist in a session consisting of five phases: motivating, structuring, conditioning, encoding and verifying. The uncertain quantity is unambiguously defined so that the expert's definition of it does not change while it is being assessed. The latter is said to be the major source of difficulty in probability encoding (Howard, 1988). Before encoding probabilities, the analyst and the expert discuss possible sources of biases, both intentional and cognitive. An example of a cognitive bias is the tendency to anchor on and adjust from a particular value, which is usually the one first mentioned or the one most easily recalled. During encoding the analyst pays attention to cues that suggest bias. The analyst then uses an appropriate debiasing strategy, if necessary. The interview procedure draws heavily on psychological research on biases and heuristics that people are prone to employ when estimating uncertain quantities.

After the uncertainty of an impact is represented, the next task is to model the stakeholder's risk attitude towards the uncertainty. With regard to a particular impact a stakeholder can either be risk-averse, risk-neutral or risk-taking. To quantify this aspect of the stakeholder's preference, the analyst presents the stakeholder with trade-off questions involving both sure and uncertain outcomes. The implications of the stakeholder's responses are captured in a single dimensional utility function, which replaces the single dimensional value function that was earlier assumed to be linear. Keeney & Raiffa (1976) describe the assessment of a utility function.

Sensitivity analysis of weights

We can also conduct a sensitivity analysis of the weights in a stakeholder's preference model. A weight range may be defined by the next higher and the next lower weights. A weight is swept through its range and judged as sensitive if a change in the portfolio occurs. If the decision is sensitive to a weight, then a careful re-examination of the weight should be considered. The analyst could use elicitation techniques that force the stakeholder to think harder about his values. For example, Keeney & Raiffa (1976) present techniques that attempt to establish how much a decision maker is willing to give up in terms of the achievement of an objective for a specified improvement in another objective.

The sensitivity analysis of the weights of stakeholders could yield interesting insights. For example, in their normal deliberations some stakeholders might take a pro-conservation stance while others take a pro-development stance. We can examine the sensitivities of weights related to these objectives. It may turn out that the decisions are insensitive to these weights and that other factors have more influence. Then, as a step towards reducing conflict between the protagonists, it might be pointed out that their stances have little influence on their decisions. On the other hand, if the weights are sensitive, then this reaffirms that diametrically opposed objectives are at the heart of the conflict.

Conclusion

The development of a water quality management strategy for Moreton Bay is a decision problem involving multiple objectives and multiple stakeholders. The collective problem structuring procedure described in this paper emphasises the multiple-objective nature of the problem, thus ensuring that proposals for water quality management will be evaluated from a broad range of criteria, including environmental, economic and social criteria. The individual analyses by each stakeholder will generate insights on the values held by stakeholders and the information necessary to evaluate proposals. The analyses might also identify particular proposals that are ranked highly in most instances, thus providing a direct answer to the question of which management actions should be included in the final strategy. In any case, the analyses will identify the features of a water quality management strategy that would make it more likely to be effective and widely accepted.

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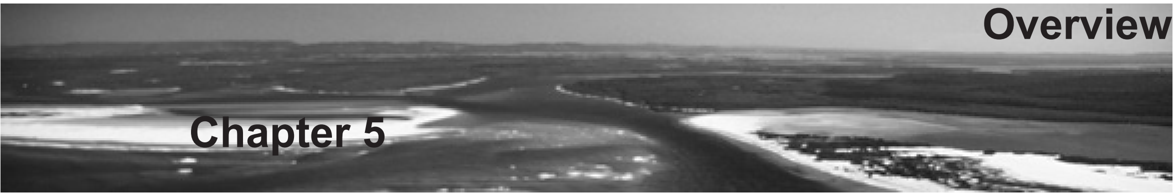

Overview

Chapter 5

Marine Plants

This chapter on marine plants, and the two chapters on animals that follow, summarise our knowledge of the biota of Moreton Bay. Eva Abal and her coauthors describe the ecological importance of two groups of plants that have become symbols of subtropical and tropical wetlands: seagrasses and mangroves. Media attention and environmental education have raised the level of our general awareness of these communities, however, specific information is seldom related. This paper quantifies the ecological value of these communities, the dimension of changes in their extent, and the causes and ecological consequences of such change. Seagrasses and mangroves make a substantial contribution to the productivity of Moreton Bay. The loss, since European settlement, of 20% and 8% of seagrasses and mangroves, respectively, is of concern and further losses must be prevented.

Julie Phillips provides an overview of the ecological importance of macroalgae and a detailed description of their distribution and biodiversity in the Moreton Bay region. She points out that the present extent of research on macroalgae falls short of being commensurate with their ecological importance, and further detailed taxonomic, ecological and biogeographic data are vital.

A set of three papers then assesses our knowledge of phytoplankton in Moreton Bay. Adrian Jones and his coauthors examine the factors that limit phytoplankton biomass along a salinity gradient in the Brisbane River, from Wivenhoe Dam to the eastern Bay. They warn that the models of nutrient limitation may not be applicable to Moreton Bay and catchment. Two papers by Philippa Uwins, one with coauthors, describe phytoplankton bloom associations, and emphasise the importance of good sample preservation techniques and scanning electron micrography for the accurate determination of bloom-associated phytoplankton. Phytoplankton blooms are characteristic of eutrophic systems and can have important consequences for the health of aquatic animals and, potentially, human health.

The chapter concludes with an extended abstract by Thomas Pillen and coauthors. They present a summary of the drastic environmental and economic consequences of the invasion of the Mediterranean Sea by a macroalga, *Caulerpa taxifolia* that is also present in Moreton Bay. The accidental translocation of marine organisms can have major impacts on marine systems. The invasion of the Mediterranean by *C. taxifolia* was particularly rapid and it has become a widespread problem. *C. taxifolia* has been known from Queensland waters for over a century, and as yet no major destructive expansion of its population has occurred, however, until we know more about the factors that lead to outbreaks of macroalgae (we know of the importance of bloom events for phytoplankton) *C. taxifolia* should be monitored. This paper highlights not only the need for vigilance concerning the accidental introduction of exotic species into the Bay (e.g. via ballast water), but also the need to monitor some species that are in this context apparently benign but have undergone major outbreaks and caused major environmental problems elsewhere (cf. crown of thorns).

Ian R. Tibbetts

Seagrasses and Mangroves in Moreton Bay

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Abstract

Seagrass meadows and mangrove communities are of considerable ecological importance in Moreton Bay, which supports approximately 25 000 ha of seagrasses and 14 000 ha of mangroves. A total of seven seagrass and seven mangrove species are found in Moreton Bay. *Zostera capricorni* is the dominant seagrass species in the Bay, however, *Halophila* spp. and *Halodule uninervis* predominate in areas intensively grazed by dugong and green sea turtles. These areas are characterised by low biomass but high productivity and have self-sustaining N inputs through microbial N fixation associated with seagrass roots. In areas protected from grazing, *Z. capricorni* dominates, with *Syringodium isoetifolium* and *Cymodocea serrulata* also occurring in monotypic stands. These seagrasses are characterised by high biomass and high productivity but require exogenous nutrient inputs. The dominant mangrove species in the Moreton Bay region is the grey mangrove, *Avicennia marina*, growing along river banks, surrounding Bay islands and along intertidal fringes of the Bay. *Avicennia marina* is replaced by the river mangrove, *Aegiceras corniculatum*, in upriver, low salinity reaches of the rivers.

Preliminary productivity values computed from seagrass growth rates, shoot densities and mangrove litter fall reveal that seagrasses and mangroves contribute 105 tonnes C/d and 96 tonnes C/d, respectively, to primary production in the Bay. Ongoing seagrass losses in Moreton Bay have been documented near the Logan River mouth, and past seagrass losses in Bramble and Deception Bay have been inferred. Together, this has resulted in an estimated 20% loss of seagrass habitat. Development activities occurring in the Brisbane airport region, as well as the development of canal estates and marinas along the foreshores of Moreton Bay, have resulted in at least 8% loss of mangrove habitat. The sensitivity of mangroves and seagrasses to changes in water quality has led to the use of these plants as biological indicators of long-term trends in water quality.

Introduction

Seagrass and mangrove communities form important components of nearshore ecosystems (Larkum *et al.,* 1989; Mall *et al.,* 1991). Both communities are extremely productive ecosystems, representing critical habitat areas for juvenile fish and invertebrates, including commercially important prawns (Robertson & Blaber, 1982; Hyland *et al.,* 1989).

Moreton Bay supports approximately 25 000 ha of seagrass meadows and 14 000 ha of mangroves (Dowling, 1986; Hyland & Butler, 1988; Hyland *et al.,* 1989). The greater extent of mangrove occurs in the islands and in the mainland coasts of southern Moreton Bay (Dowling, 1986) while the most extensive seagrass beds occur in the sandbanks of eastern Moreton Bay (Hyland *et al.,* 1989) (Figure 1). Seagrass on the eastern side of Moreton Bay generally grow to greater depths than those on the western side of the Bay (Abal & Dennison, 1996), correlating to the east-west gradient in water quality of the Bay. The tidal influx of large volumes of ocean water via the northern and eastern entrances, together with terrigenous inputs from the rivers discharging along the western margin, result in this east-west decline in water quality across the Bay (Moss *et al.,* 1992).

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 269-278.

This paper provides a synthesis of existing information of mangroves and seagrasses: their distribution, productivity and roles as biological indicators of long-term trends in water quality in Moreton Bay. Synthesis information was obtained either from results of direct parameter measurements (e.g. seagrass growth rates, phytoplankton ^{14}C productivity values, mangrove litter fall) by the Marine Botany Group, The University of Queensland or existing published literature.

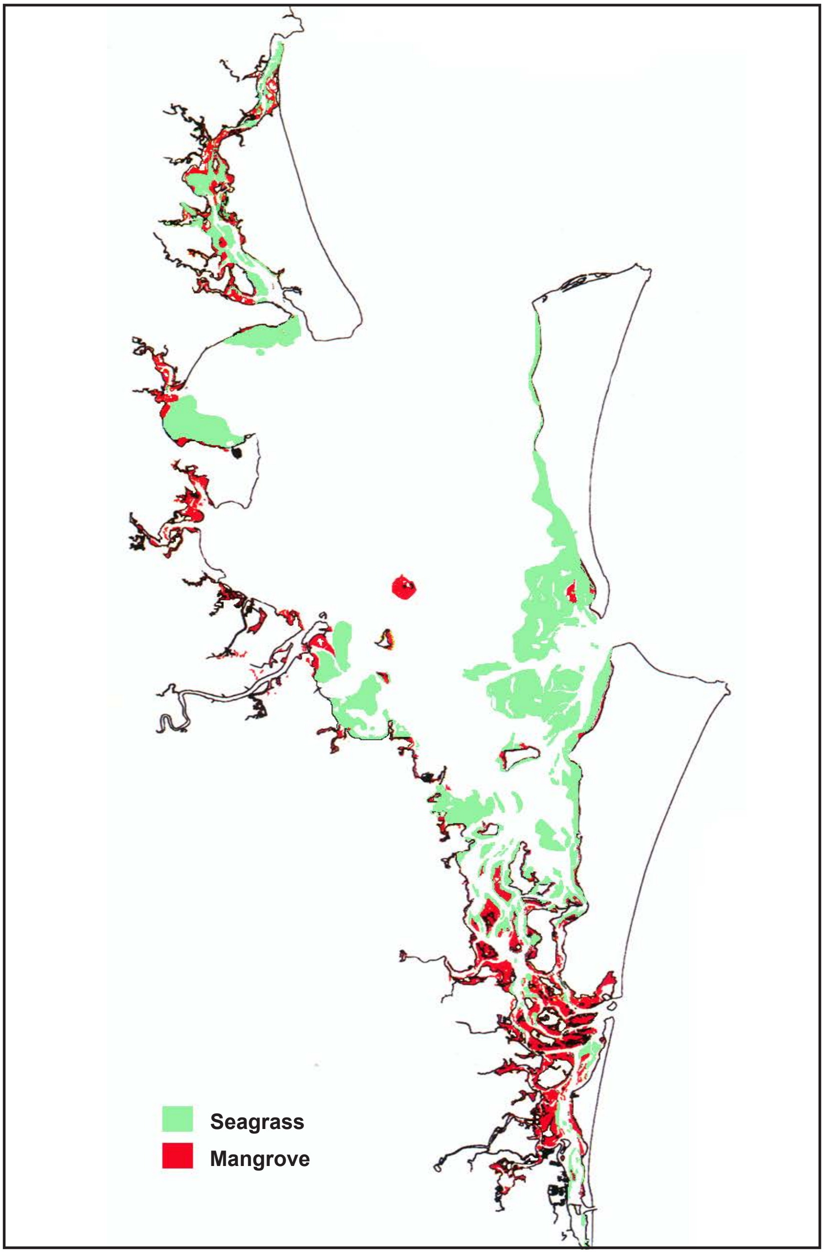

Figure 1. Distribution of seagrass and mangroves in Moreton Bay (modified from Quinn, 1992).

Seagrass Species Composition and Zonation

A total of seven seagrass species (*Zostera marina, Halodule uninervis, Cymodocea serrulata, Syringodium isoetifolium, Halophila ovalis, Halophila spinulosa* and *Halophila decipiens*) are found in Moreton Bay (Figure 2). *Zostera capricorni* is the dominant species, often mixed with *H. ovalis, H. spinulosa and H. uninervis*. Although *Z. capricorni* occurs all-year-round and does not exhibit seasonality in distribution, the species is least preferred as food by dugongs (Preen, 1992). *Halophila* and *Halodule* species predominate in areas extensively grazed by dugongs and sea turtles.

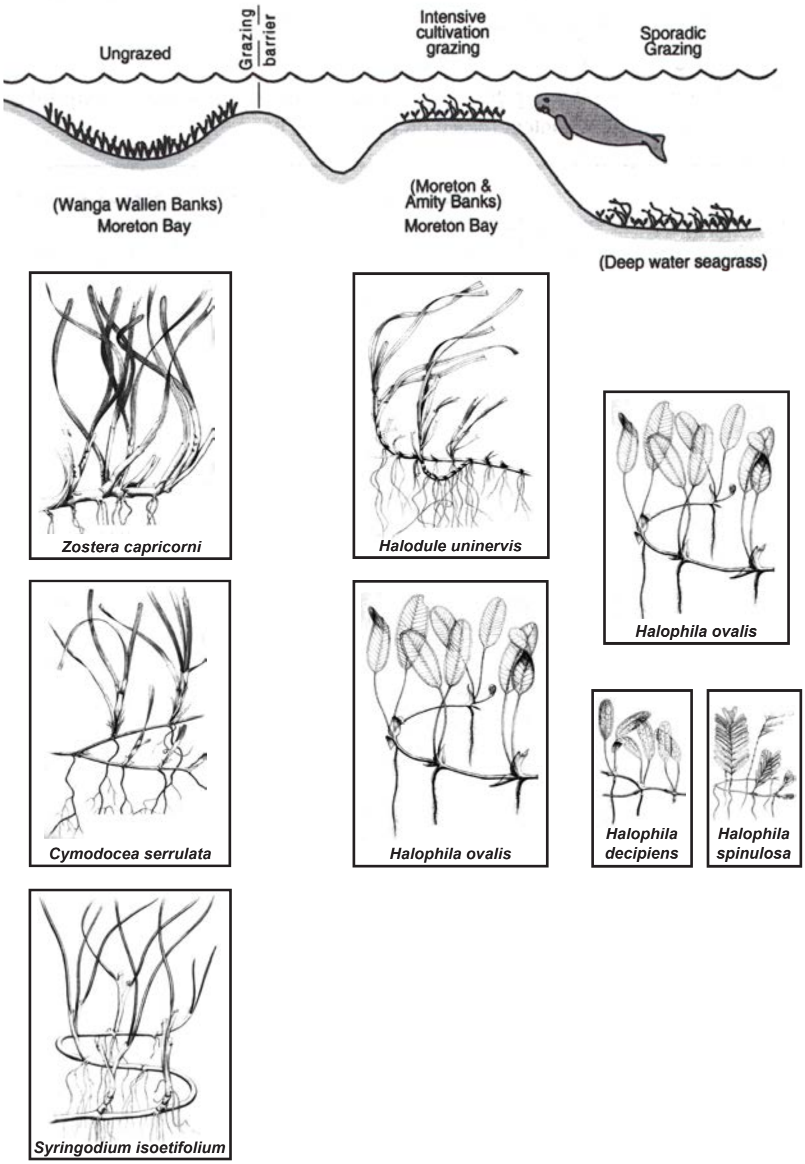

Figure 2. Seagrass species composition in grazed and ungrazed areas of Moreton Bay (seagrass diagrams from Lanyon, 1986).

Grazing by dugongs has been observed to profoundly impact on the ecology of seagrasses in the Bay, affecting seagrass species composition, distribution and sediment nutrient cycling (Table 1). Rates of microbial nutrient cycling in seagrass sediments increased following intensive grazing by dugongs (Perry, 1997). Dugong grazing aerates the sediments and buries plant detritus, increasing sulfate reduction and nitrogen fixation in the sediments. More nutrients become available for seagrass regrowth, elevating the nitrogen tissue content for future grazing efforts. Grazed areas are characterised by low seagrass shoot biomass, but high productivity. The high percentage nitrogen content of *Halophila* and *Halodule* species in grazing areas are indicative of self-sustaining nitrogen inputs through microbial nitrogen fixation. The high nitrogen and low fibre contents of these species contribute to the preference of these species by dugongs and turtles (Preen, 1992).

Table 1. Impacts of dugong grazing on seagrass species composition, physiology and nutrient content (based on Perry, 1997).

	Grazed areas	**Ungrazed areas**
Dominant species	*Halophila* spp. *Halodule uninervis*	*Zostera capricorni* *Syringodium isoetifolium* *Cymodocea serrulata*
Above-ground biomass	low	high
Productivity	high	high
% Nitrogen	high	low
Fibre content	low	high
Nitrogen fixation	high	low
Sulfate reduction	high	low

In areas protected from grazing, *Z. capricorni* dominates, with *S. isoetifolium* and *C. serrulata* also occurring in monotypic stands (Perry, 1997). These species are characterised by high biomass, high productivity, however, their high fibre and lower percentage N content make them least preferred for food by dugongs and turtles.

The upper distribution limit of seagrass is influenced by a number of factors which include ultra-violet radiation (UV) and photosynthetically active radiation (PAR) intensities, desiccation, temperature, salinity, tidal and wave action (Walker & McComb, 1990; Dawson & Dennison, 1996), while the lower distribution limit is determined by light availability (Abal & Dennison, 1996). *Zostera capricorni* and *Halophila ovalis* grow in the intertidal and subtidal regions while *S. isoetifolium* and *C. serrulata* are restricted to a small sub-tidal depth range (Young & Kirkman, 1975; Young, 1978*)*. In Moreton Bay, *Halophila* spp. and *H. uninervis* grow in the deepest water where seagrass is found (up to 7 m) (Figure 3).

Distribution of Mangroves

In Moreton Bay and its rivers, mangroves are represented by seven species *(Aegiceras corniculatum, Avicennia marina, Bruguiera gymnorrhiza, Ceriops australis, Excoecaria agallocha, Lumnitzera racemosa* and *Rhizophora stylosa)* (Figure 4*)*. These communities are found around mean sea level (MSL) to the high tide line, penetrating into the rivers and creeks. Where mangroves form a fringe along the mainland, they occur in areas that are protected from strong wave action by extensive mud flats (Dowling, 1986).

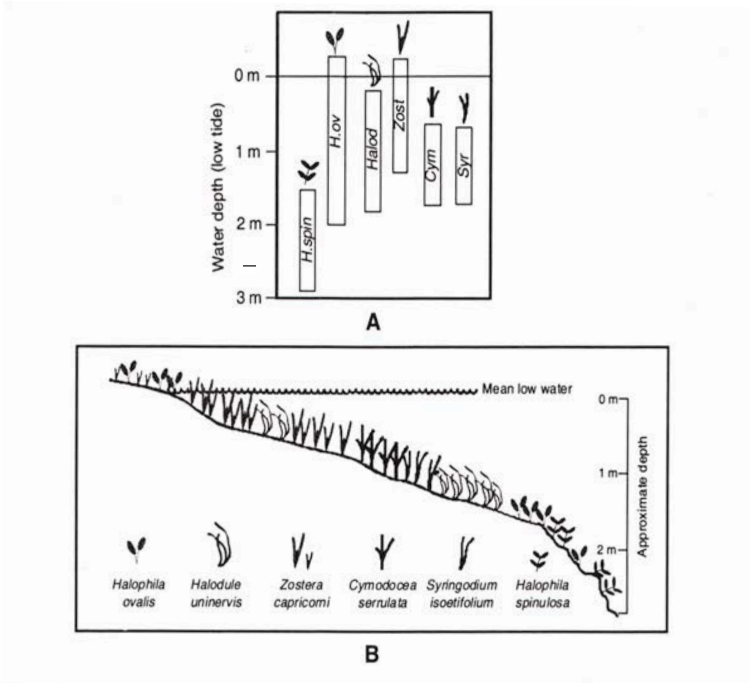

Figure 3. Schematic diagram of a seagrass transect in Moreton Bay (A), and generalised seagrass depth distribution (B) showing depth distribution of each species (Dawson & Dennison, 1996).

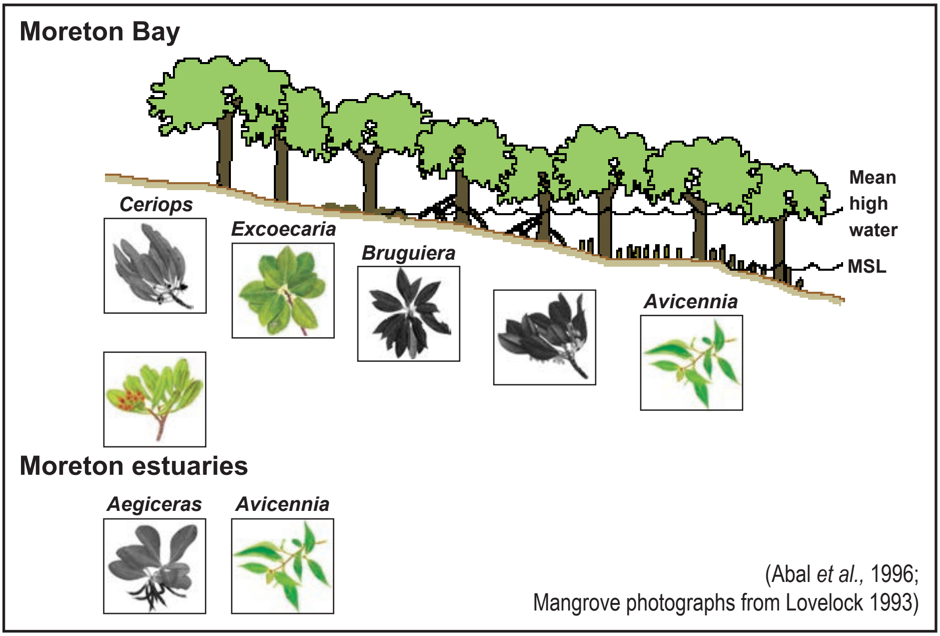

Figure 4. Zonation of mangrove species in Moreton Bay and its estuaries.

The most widespread and common mangrove species in the Moreton Bay region is the grey mangrove, *A. marina*, growing along river banks, surrounding bay islands and along intertidal fringes of the Bay. *Avicennia marina* occurs along the seaward edge of the intertidal zone and along with the other mangrove species, grows to the upper intertidal areas. *Rhizophora stylosa* typically occurs in a very narrow zone behind the seaward fringe of *A. marina*, however, it may also be found forming the seaward fringe in shallow, protected areas, extending to the upper reaches of inlets and creeks where they may form pure stands. *Ceriops australis* is found mainly in the upper tidal zone and is consequently inundated by only a few tides each month. *Bruguiera gymnorrhiza* is scattered in the most landward fringe, as understorey in *A. marina* forests and sometimes growing in association with *E. agallocha*. *Avicennia marina* is replaced by the river mangrove, *A. corniculatum*, in upriver, low salinity reaches, growing along the edges of the rivers and creeks. The rarest of all the mangrove species in Moreton Bay is *L. racemosa*, which occurs in the landward fringe in areas only reached by the highest tides (Dowling, 1986; Lovelock, 1993).

The distinct zonation pattern characteristic of tropical mangrove forests, which is more or less parallel to the adjacent sea, river or creek (Davie, 1984) is not particularily distinct in Moreton Bay, except for some species in which vertical distribution appears to be related to tidal planes and soil types (Dowling, 1986).

Productivity of Mangroves and Seagrasses in Moreton Bay

Mangrove productivity is influenced by a number of factors including nutrient regimes, tidal flushing, salinity, and substrate characteristics (Bunt, 1995). The productivity of seagrasses is regulated by a similar variety of environmental factors such as light availability, nutrients, temperature, salinity, and substrate characteristics (Walker & McComb, 1990; Duarte, 1995; Abal *et al.*, 1994; Udy, 1997).

To estimate the contribution of each of the major groups of primary producers to primary productivity in Moreton Bay, we computed preliminary productivity values using seagrass growth rates, shoot densities (Udy, 1997), phytoplankton ^{14}C productivity values (O'Donohue & Dennison, 1997), and mangrove litterfall (Rogers, 1998). All values were converted to g C/ha/d, then these values were extrapolated using estimates of the total area (ha) occupied by each group, to derive total Bay values. Results reveal that seagrasses and mangroves contribute a third of the total productivity of Moreton Bay (seagrasses: 105 tonnes C/ha/d and mangroves: 96 tonnes C/ha/d). Because total productivity is relative to the total area, phytoplankton communities, which occur Baywide, contribute 68% to total Bay productivity (424 tonnes C/ha/d) (Table 2).

Table 2. Primary productivity of seagrasses, mangroves and phytoplankton in Moreton Bay and the percentage contribution of these communities to the overall Bay productivity. Values computed from growth rates and shoot densities (seagrass), litter fall (mangroves), and C^{14} productivity (phytoplankton). Growth rates were then related to the total area occupied by each group to obtain their relative productivities in Moreton Bay, as a whole.

	Area (ha)	Primary productivity (tonnes C/day)	Contribution total Bay primary productivity (%)
Seagrasses	25 000	105	16.8
Mangroves	13 604	96	15.3
Phytoplankton	140 000	424	67.9

Seagrass and mangrove communities are highly productive and make a substantial contribution to Baywide primary productivity, however, due to their proximity to urban areas they are particulary vulnerable to human activities (Cambridge *et al.,* 1986; Tomlinson, 1994). Over a period of five years, decreases in seagrass depth range, which is the maximum depth of seagrass growth, were recorded in western Bay areas affected by river plumes. Results from an intensive seagrass and water quality monitoring program indicated high correlation ($r^2 > 0.88$) between seagrass depth limit and water quality parameters (total suspended solids, light attenuation coefficient, total nitrogen and chlorophyll *a*). Based on these correlations the following water quality minima for seagrass have been formulated: 0.9 K_d/m; 10 mg/L total suspended solids; 1.0 µg/L chlorophyll *a* and 21 µM total Kjeldahl nitrogen (Abal & Dennison, 1996).

Seagrass loss has been documented over a five-year period near the mouth of the Logan River, a turbid river with increased land use in its watershed (Abal & Dennison, 1996). The ongoing loss of seagrass in southern Moreton Bay and inferred seagrass loss in Bramble Bay and Deception Bay (Figure 5) areas result in an estimated 20% loss of seagrass habitats since European settlement.

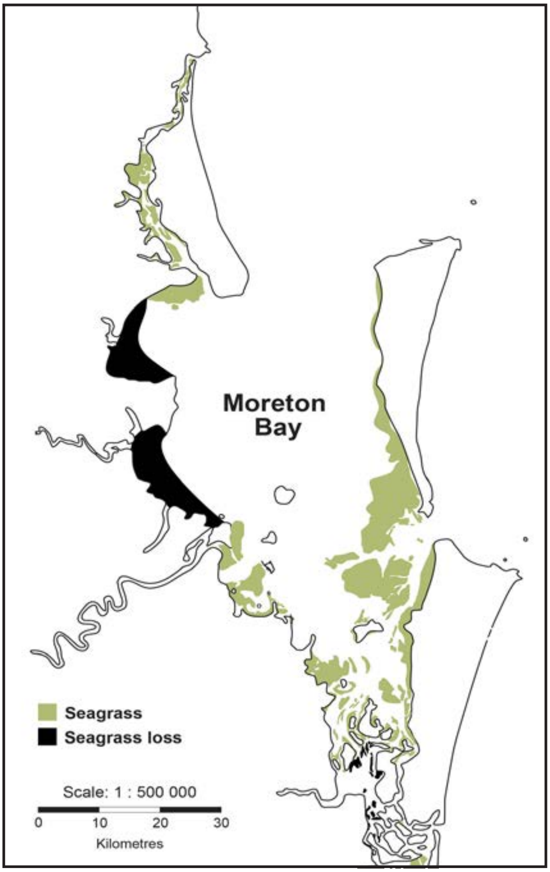

Figure 5. Seagrass distribution map indicating documented loss of seagrass beds in southern Moreton Bay (1987 to 1992) (Abal & Dennison, 1996) and inferred seagrass loss in Bramble Bay and Deception Bay (pre-European settlement to present). Seagrass loss is indicated by black areas. Diagram is modified from Hyland *et al*. (1989).

In contrast to seagrass, mangrove losses have been more localised and attributed to the physical removal of mangroves accompanying development. Large areas of mangroves have been lost through the development and construction of airport, wharf, port, and private real estate developments, contributing to an estimated 8% loss of these plant communities in Moreton Bay (1974-1987) (Table 3). However, it is difficult to ascertain mangrove loss accurately as at the same time as clearing has been taking place in some areas, limited expansion of mangrove communities has also been occurring in Moreton Bay (Hyland & Butler, 1988). This 'natural' expansion of mangrove area has been attributed to either a seaward advance of the mangrove front (Dowling, 1986) or to their landward encroachment (McTainsh *et al.,* 1988).

Table 3. Net loss of mangrove communities in Moreton Bay from 1974-1987 (from Hyland & Butler, 1988).

Region	1987 Mangrove area (ha)	1974-1987 Mangrove area decline (ha)
Southern Moreton Bay:		
Coomera River to Empire Point	7 211	49
Northern Moreton Bay:		
Empire Point to Caboolture River	3 892	962
Eastern Moreton Bay:		
Myora to Moreton Island	332	nil
Pumicestone Passage:		
Caboolture to Caloundra	2 169	121
Total	**13 604**	**1 132 (8.3%)**

In conclusion, mangroves and seagrasses are important primary producers in Moreton Bay, contributing to one-third of total productivity of the Bay. However, losses of these communities, related to changes in water quality (seagrass) and localised physical removal (mangroves) as a consequence of rapid development in the Moreton Region, continue to occur and need to be monitored and regulated. A balance between the extensive development of coastal wetlands and catchment which impacts on over all water quality in the Bay, and the existence of healthy seagrass and mangrove communities, must be sought.

Acknowledgements

This paper incorporates the results of research by Christine Perry (grazing impacts), James Udy (seagrass productivity), Jane Rogers (mangrove productivity), Shane Dawson (seagrass zonation) and Department of Environment – Coastal Management (seagrass monitoring program). Karen Holloway of the University of Queensland Department of Botany provided assistance with the diagrams.

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Macroalgae of Moreton Bay: Species Diversity, Habitat Specificity and Biogeography

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Abstract

There have been few publications which have documented the macroalgae of Moreton Bay. In the absence of published information, herbarium collections can provide some indication of the status of poorly known groups. Database searches of the algal collection lodged at Queensland Herbarium reveal that approximately 275 species of Chlorophyta, Phaeophyta and Rhodophyta have been recorded for Moreton Bay and that 43 localities along the mainland coast from Caloundra to Victoria Point and on the offshore islands have been sampled over the past 100 years. Collections have tended to be haphazard and biased with different localities recording from two (Margate) to approximately 100 species (Redcliffe, Caloundra). Tropical/subtropical species predominate in Moreton Bay, some having their southern distribution limit in the region while other species extend further south. A smaller number of temperate species range from southern Australia into the Bay and further north. Some species occur in many different habitats whereas others are restricted in their ecological distribution to coasts exposed to ocean swells, to sheltered areas, mangrove communities, polluted areas and possibly either to the northern or southern parts of the Bay. The macroalgae of Moreton Bay should now be surveyed to accurately determine species diversity and distribution patterns. Furthermore, ecological data on the availability of suitable habitat, relative abundance of species, composition of macroalgal assemblages and patterns of spatial and temporal variability need be collected. Ideally, a monitoring program to document the changes in the macroalgal communities should be initiated. Conservation and management strategies for the marine environment must be based on detailed systematic, ecological and biogeographic information.

Introduction

Macroalgae are important components of coastal marine ecosystems. They are major primary producers in shallow seas, with 10 % (kelp communities) to 60-97 % (algal turf communities on coral reefs) of algal production entering grazing food chains (Mann, 1973; Carpenter, 1986; Klumpp & Mc Kinnon, 1989). Even in seagrass meadows, 20-62 % of the primary productivity of algal epiphytes on seagrass leaves, compared to a maximum of 10% of seagrass primary productivity, are consumed by herbivores (Klumpp *et al.,* 1992; Orth, 1992). Frondose macroalgae increase habitat complexity in marine communities by providing a greater number of niches which are inhabited by diverse and abundant assemblages of crustaceans, mites, worms, echinoderms and molluscs (Gunnill, 1982; Naim, 1988; Duggins *et al.,* 1990; Poore, 1994; Smith *et al*., 1996).

Some macroalgal species are good indicators of water quality. Blooms of *Ulva*, *Enteromorpha*, *Cladophora*, *Dictyosphaeria* and *Rhizoclonium* generally occur in eutrophic waters, often in response to massive injections of nutrients from a point source (Smith *et al.,* 1981; Walker & Ormond, 1982; Lapointe & O'Connell, 1989). Other species, such as large brown algae, are sensitive to declining water quality and disappear from habitats subjected to nutrient enrichment and toxicants (Brown *et al.,* 1990; Reed *et al*., 1994; Doblin & Clayton, 1995). Furthermore, macroalgal communities are just as severely impacted by prolonged pollution as other marine communities: they show dramatic decreases in species diversity, reduced vertical stratification and community complexity and there is a marked shift in dominance from larger macroalgae

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 279-290.

to blooms of fewer opportunistic and stress-tolerant species, often referred to as nuisance algae (Littler & Murray, 1975; Brown *et al.,* 1990; Hardy *et al.,* 1993). In order to identify anthropogenic-induced changes to macroalgal communities which are naturally dynamic, it is necessary to have detailed knowledge and a good understanding of the relative abundance of various species and their temporal and spatial variability in natural communities.

Little is known of the macroalgae of Moreton Bay; there is no published flora or checklist. Studies which focus on macroalgae in the region are few, published in three scientific papers (Askenasy, 1894; Johnston, 1917; Cribb, 1979), and a report from an environmental impact study (Atherton & Dyne, 1977). Some information is available from publications which cover wider geographical regions such as checklists (Grunow, 1874; Askenasy, 1888; Okamura, 1904; Levring, 1946; 1953; May, 1965), taxonomic revisions (Cribb, 1954; 1956; 1958a; 1958b; 1960; Huisman, 1986; 1996; King & Puttock, 1989; Huisman & Borowitzka, 1990) and an ecological account (Endean *et al*., 1956). The paucity of information for Moreton Bay is further compounded by the lack of a macroalgal flora for Queensland which would facilitate species identification, although a recent naturalist's guide (Cribb, 1996) is a valuable step towards compiling information. Checklists for Queensland (Bailey, 1883; 1913) and northern Australia (Lewis, 1984; 1985; 1987) provide an indication of species records for the State but many are erroneous. A catalogue (Phillips & Price, 1997) lists all records of brown algal species for Queensland, provides an update of nomenclatural and taxonomic changes and indicates correct species identifications supported by vouchered herbarium material. In addition, the current Queensland plant census (Henderson, 1997) includes the algae (Phillips, 1997), although many species other than those in the Phaeophyta require further taxonomic reappraisal.

By contrast, the macroalgae of Port Phillip Bay, Victoria, are well known having been studied in the 1886-1892, 1957-1963, 1968-1971, 1975-1978 and 1992-1996 surveys (Light & Woelkerling, 1992; Harris *et al.,* 1996). The surveys, as well as other scientific investigations conducted in Port Phillip Bay, have produced at least 16 scientific papers and 22 reports to government and statutory authorities that specifically deal with macroalgae. Various aspects of taxonomy and ecology are documented, including systematic lists of intertidal and subtidal macroalgae, descriptions of intertidal macroalgal assemblages, the effect of sewage on macroalgal communities and macroalgal standing crop and the nutrient content of macroalgal species.

In the absence of published information, herbarium and/or museum collections may provide some indication of the status of poorly known groups. The macroalgal collection at the Queensland Herbarium comprises approximately 17 000 specimen sheets and includes many specimens from Moreton Bay. The collection has recently been curated and databased to facilitate the retrieval of information. This paper aims to demonstrate that macroalgae are important components of the marine communities of Moreton Bay, to review the current status of the knowledge base and finally to make recommendations for further research.

Methods

Searches of the HERBRECS database at Queensland Herbarium provided the following data on the macroalgae of Moreton Bay: species recorded, collection localities, date of collection and the collector's name. The data was then used to compile a species list and a map of the number of species recorded at each locality. Preliminary analyses have also been undertaken on the biogeographic affinities and habitat preferences of the macroalgae recorded from the Bay.

Clearly, much research needs to be undertaken on macroalgae of Moreton Bay, and for Queensland generally, but there is sufficient information available from the herbarium collection to begin the documentation process. Revisionary studies are urgently required. For the purpose of the present analyses, species identifications have not always been verified. Records known to be erroneous have been excluded, and specimens identified only to genus have been included only when other species in the genus are not recorded.

Results and Discussion

Approximately 275 species of Chlorophyta, Phaeophyta and Rhodophyta (Appendix 1), or about 40 % of the macroalgal species reported for Queensland (Lewis, 1984; 1987; Phillips & Price, 1997), have been recorded for Moreton Bay (Table 1). Macroalgal species richness for the Bay far exceeds the richness of seagrasses and mangroves (seven species each) found in the region. Comparisons between the total number of species recognised for Moreton Bay and for Port Phillip Bay, Victoria, are interesting. Port Phillip Bay records 404 macroalgal species, has been surveyed five times with the macroalgae included in each survey (Light & Woelkerling, 1992), and is located on one of the four most species-rich coastlines in the world (the southern Australian coastline) (Womersley, 1990; Bolton, 1994). In contrast, the macroalgae of Moreton Bay are poorly known, have never been surveyed, and yet still show remarkably high species richness, comprising approximately 70% of the total number of species recorded for Port Phillip Bay.

Forty-three localities from Caloundra to Victoria Point along the mainland coast and on some offshore islands have been sampled over the past 100 years (Figure 1). Many collections made late last or early this century give "Moreton Bay" as the collection site and are uninformative.

Table 1. Comparisons of species richness at selected localities (data from Lewis, 1984; 1985; Light & Woelkerling, 1992; Phillips & Price, 1997; this study).

Division	Number of species		
	Queensland	Moreton Bay	Port Phillip Bay
Chlorophyta	196	65	64
Phaeophyta	126	51	93
Rhodophyta	385	161	247
Total	**706**	**275**	**404**

Rock platforms on the Redcliffe Peninsula and in the Caloundra and Point Lookout regions are relatively well collected, a result of collections most notably by G. & J. McKeon in the late 1940s and early 1950s, and by A.B. Cribb over the last five decades. Most collections have been made in the intertidal zone, in the shallow subtidal zone or from the drift. Offshore subtidal localities within the Bay have not been sampled, although collections have been made at popular SCUBA diving sites near Moreton Island (Flinders Reef, Smiths Rock, Brennan Shoal, Flat Rock) during the past two decades.

Localities with the highest species richness are Redcliffe (109 species) and Caloundra (103 species). Ten localities record from 20-68 species and 31 localities (more than 70% of those sampled) less than 20 species (Figure 1). Species numbers reported at each locality appear to be related to sampling effort. The macroalgae of Redcliffe and Caloundra have been sampled on more than 30 different occasions. Localities with low species numbers have been collected infrequently. For example, the only collection for Peel Island (23 species) was made in 1938 by S.T. Blake, a botanist from the Queensland Herbarium (Everist, 1982). Differential sampling effort, rather than real differences between the localities, may also explain the differing species

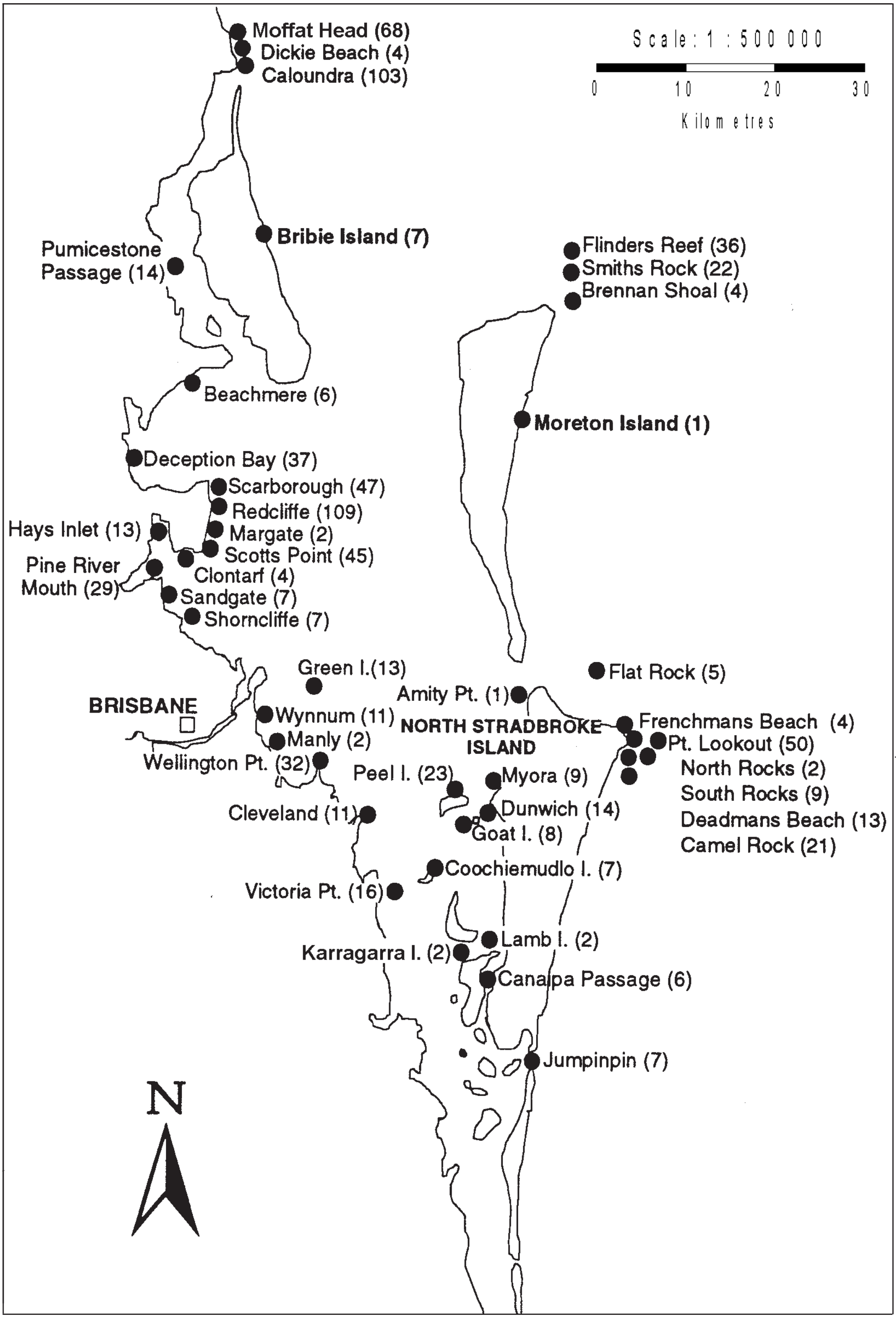

Figure 1. Macroalgal collection localities in Moreton Bay. The number in brackets after each collection locality represents the number of species recorded from the locality.

numbers recorded for Caloundra and Moffat Head and for Redcliffe and Scarborough. Similar species richness would be expected from these locality pairs which have similar habitats and are in close geographical proximity, but this requires further investigation.

Distribution patterns in the Bay

It is apparent from the limited data available that species exhibit different distribution patterns within Moreton Bay. Some species (e.g. *Halimeda discoidea*, *Ulva lactuca*, *Laurencia filiformis*, *Champia parvula*, *Padina gymnospora*, *Zonaria diesingiana*) are widely distributed, and found on rough water coasts as well as in sheltered areas. *Caulerpa peltata, Valoniopsis pachynema, Endarachne binghamiae*, *Dilophus marginatus*, *Plocamium hamatum* and *Laurencia brongniartii* tend to be restricted to open coasts, while other species (*Acetabularia calyculus*, *Sporochnus comosus*, *Petalonia fascia*, *Gelidium allanii*, *Myriogramme bombayensis*) are only found in the sheltered areas in the Bay. Further studies may detect more restricted distribution patterns within the sheltered areas as some species (*Spatoglossum macrodontum*) appear more common in the northern Bay while others (*Stypopodium flabelliforme, Sporochnus comosus*) have been found only in the southern Bay.

Habitat specificity

Ecological studies in other regions of the world have identified various macroalgal species that tend to occur together in response to similar habitat requirements. These assemblages are usually named according to the dominant species. It is possible to refer to some assemblages which occur in Moreton Bay, although many more will be identified.

Cribb (1979) described the macroalgae found in the *Avicennia* forests of Moreton Bay. The *Bostrychia*-*Caloglossa* assemblage grows attached to trunks and modified roots of mangroves worldwide and is characterised in Moreton Bay by the presence of the red algal genera *Bostrychia*, *Caloglossa* and *Catenella*. These genera are largely restricted in their ecological distribution to mangrove forests.

Macroalgae usually colonise firm substrates but many species of the green algal order Caulerpales grow on unconsolidated substrates. Species of *Caulerpa* and *Udotea* are found on the sand/mud substrates of Moreton Bay, also occurring in seagrass meadows. Algal epiphytes normally occur on seagrass leaves, but the presence on these algae in Moreton Bay has not been documented.

Biogeography

The macroalgae of Moreton Bay comprise three biogeographic groups (Table 2). Tropical/subtropical species predominate, some having their southernmost distribution limit in the Bay (for example, *Udotea argentea*, *Hormophysa cuneiformis*, *Cystoseira trinodis, Dictyopteris australis, Tricleocarpa cylindrica*), while others extend further south to the northern (*Halimeda discoidea*) or the southern New South Wales coast (*Acetabularia calyculus*, *Padina gymnospora*, *Spatoglossum macrodontum*). Some temperate species (*Scytosiphon simplicissimus*, *Sporochnus comosus*, *Petalonia fascia*, *Bellotia eriophorum*) range into subtropical southern Queensland from southern Australian coasts, although the Bay is not the northernmost limit of most of

Table 2. Biogeographic affinities of the macroalgal species from Moreton Bay.

Biogeographic group	% composition of the flora
Tropical/subtropical	64.0
Temperate	15.2
Cosmopolitan	20.8

these species. Cosmopolitan species (*Ulva* spp., *Enteromorpha* spp., *Colpomenia sinuosa*) are also found in Moreton Bay.

Further research

Data from the existing herbarium collection have clearly established that there is high macroalgal species richness and that macroalgae occur at many localities in Moreton Bay. However, it is obvious that the herbarium collection does not give an accurate representation of macroalgal species richness or distribution patterns in Moreton Bay. The collection is not comprehensive in geographical coverage as large areas have not been sampled, particularly offshore subtidal areas which are accessed only by boat and SCUBA. Furthermore, sampling has been haphazard with localities either well or poorly collected. Presence/absence data for species at each locality sampled have not been recorded. Finally, collections may be biased, often reflecting the interests of the collector or the tendency to preferentially collect larger or easily identified species.

The macroalgal communities of Moreton Bay should be surveyed to determine more accurately the level of species diversity and to document macroalgal distribution patterns. Furthermore, ecological data on the availability of suitable substrate, relative abundance of species, the composition of macroalgal assemblages and patterns of abundance in both environmental and geographic space should be collected with the aim of detecting any changes in macroalgal communities and then determining whether the changes are caused by natural or anthropogenic disturbance. The ecological data should be stored in a database and should complement the taxonomic data on the HERBRECS database. Conservation and management strategies for the marine environment must be based on detailed and accurate taxonomic, ecological and biogeographic information.

Acknowledgements

I would like to thank Dr G.P. Guymer, Chief Botanist, Queensland Herbarium, for access to the HERBRECS database, and Mr Paul Robins, Queensland Herbarium, for running the database search.

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Appendix 1: List of macroalgae recorded from Moreton Bay based on specimens lodged at Queensland Herbarium.

Chlorophyta

Acetabularia calyculus Lamouroux

Avrainvillea erecta (Berkeley) Gepp & Gepp

Blidingia minima (Naegeli *ex* Kützing) Kylin

Boodleopsis pusilla (Collins) Taylor, Joly & Bernatowicz

Bryopsis indica Gepp & Gepp

Bryopsis lubrica Cribb

Caulerpa ambigua Okamura

Caulerpa brachypus Harvey

Caulerpa cupressoides (Vahl) C. Agardh

Caulerpa fastigata Montagne

Caulerpa lentillifera J. Agardh

Caulerpa peltata Lamouroux

Caulerpa racemosa (Forsskål) J. Agardh

Caulerpa taxifolia (Vahl) C. Agardh

Caulerpa webbiana Montagne

Chaetomorpha aerea (Dillwyn) Kützing

Chaetomorpha antennina (Bory) Kützing

Chaetomorpha exposita (Børgesen) Dawson

Chaetomorpha linum (O.F. Müller) Kützing

Chlorodesmis caespitosa J. Agardh

Chlorodesmis major Zanardini

Cladophora catenata (Linnaeus) Kützing

Cladophora bombayensis Børgesen

Cladophora perpusilla Skottsberg & Levring

Cladophora pellucida (Hudson) Kützing

Cladophora prolifera (Roth) Kützing

Cladophora socialis Kützing

Cladophora vagabunda (Linnaeus) van den Hoek

Cladophorella calcicola F.E. Fritsch

Cladophoropsis gracillima Dawson

Cladophoropsis herpestica (Montagne) Howe

Cladophoropsis membrancea (C. Agardh) Børgesen

Cladophoropsis sundanensis Reinbold

Codium globosum Lucas

Codium lucasii Setchell

Codium platyclados Jones & Kraft

Codium spongiosum Harvey

Enteromorpha clathrata (Roth) Greville

Enteromorpha flexuosa (Wulfen) J. Agardh

Enteromorpha intestinalis (Linnaeus) Nees

Enteromorpha linza (Linnaeus) J. Agardh

Enteromorpha prolifera (O.F. Müller) J. Agardh

Halimeda discoidea Decaisne

Microdictyon umbilcatum (Velley) Zanardini

Ostroebium quekettii Bornet & Flahaut

Phaeophila dendroides (P. Crouan & H. Crouan) Batters

Pilinia novae-zelandiae (V. Chapman) Papenfuss & Fan *ex* Papenfuss

Pseudoendoclonium submarinum Wille

Pseudopringsheimia sp.

Rhipiliopsis echinocaulos (Cribb) Farghaly

Rhizoclonium capillare Kützing

Rhizoclonium implexum (Dillwyn) Kützing

Rhizoclonium riparum (Roth) Harvey

Siphonocladus tropicus (Crouan) J. Agardh

Spongocladia vaucheriaeformis Areschoug

Struvea anastomosans (Harvey) Piccone & Grunow *ex* Piccone

Udotea argentea Zanardini

Ulvella sp.

Ulva lactuca Linnaeus

Ulva rigida C. Agardh

Ulvaria oxysperma (Kützing) Bliding

Valoniopsis pachynema (Martens) Børgesen

Wittrockiella salina Chapman

Phaeophyta

Bachelotia antillarum (Grunow) Gerloff

Bellotia eriophorum Harvey

Chnoospora minima (Hering) Papenfuss

Colpomenia peregrina (Sauvageau) Hamel

Colpomenia sinuosa (Mertens *ex* Roth) Derbès & Solier

Cystoseira trinodis (Forsskål) C. Agardh

Dictyopteris acrostichoides (J. Agardh) Børgesen

Dictyopteris australis (Sonder) Askenasy

Dictyopteris crassinervia (Zanardini) Schmidt

Dictyopteris repens (Okamura) Børgesen

Dictyota acutiloba J. Agardh

Dictyota bartayresiana Lamouroux

Dictyota dichotoma (*Hudson*) *Lamouroux*

Dictyota dichotoma var. *intricata* (C. Agardh) Greville

Dictyota pardalis forma *pseudohamata* Cribb

Dilophus intermedius (Zanardini) Allender & Kraft

Dilophus marginatus J. Agardh

Ecklonia radiata (C. Agardh) J. Agardh

Ectocarpus siliculosus (Dillwyn) Lyngbye

Endarachne binghamiae J. Agardh

Feldmannia indica (Sonder) Womersley & Bailey

Feldmannia irregularis (Kützing) Hamel

Hincksia mitchelliae (Harvey) Silva

Hormophysa cuneiformis (Gmelin) Silva

Hydroclathrus clathratus (C. Agardh) Howe

Lobophora variegata (Lamouroux) Womersley

Mesospora sp.

Nemacystus decipiens (Suringar) Kuckuck

Padina australis Hauck

Padina gymnospora (Kützing) Sonder

Padina tenuis Bory

Petalonia fascia (Müller) Kuntze

Ralfsia sp.

Rosenvingea intricata (J. Agardh) Børgesen

Rosenvingea orientalis (J. Agardh) Børgesen

Sargassum amalie Grunow

Sargassum crassifolium J. Agardh

Sargassum filifolium var. *aciculare* Grunow

Sargassum flavicans (Mertens) C. Agardh

Sargassum lophocarpum J. Agardh

Sargassum parvifolium (Turner) C. Agardh

Sagassum piliferum (Turner) C. Agardh

Sargassum polycystum C. Agardh

Sargassum spinifex C. Agardh

Scytosiphon simplicissimus (Clemente) Cremades

Spatoglossum macrodontum J. Agardh

Sphacelaria rigidula Kützing

Sporochnus comosus C. Agardh

Stypopodium flabelliforme Weber-van Bosse

Tomaculopsis herbertiana Cribb

Zonaria diesingiana J. Agardh

Rhodophyta

Acanthophora dendroides Harvey

Acanthophora muscoides (Linnaeus) Bory

Acanthophora spicifera (Vahl) Børgesen

Acrochaetium pulvinatum Levring

Acrosorium venulosum (Zanardini) Kylin

Aglaothamnion byssoides (Arnott *ex* Harvey L'Hardy-Halos & Rueness

Amphiroa anceps (Lamarck) Decaisne

Amphiroa ephedraea (Lamarck) Decaisne

Amphiroa fragilissima (Linnaeus) Lamouroux

Anotrichum tenue (C. Agardh) Nägeli

Asparagopsis taxiformis (Delile) Trevisan

Audouinella microscopica (Nägeli in Kützing) Woelkerling

Audouinella saviana (Meneghini) Woelkerling

Audouinella tuticorinense (Børgesen) Garbary

Bangia atropurpurea (Roth) C. Agardh

Bostrychia moritiziana (Sonder *ex* Kützing) J. Agardh

Bostrychia radicans (Montagne) Montagne

Bostrychia simpliciuscula Harvey *ex* J. Agardh

Bostrychia tenella subsp. *flagellifera* R.J. King & Puttock

Bostrychia tenuissima R.J. King & Puttock

Botryocladia leptopoda (J. Agardh) Kylin

Callithamnion corymbosum var. *australis* Askenasy

Callophyllis sp.

Caloglossa leprieurii (Montagne) J. Agardh

Caloglossa ogasawaraensis Okamura

Catenella nipae Zanardini

Catenella subumbellata Tseng

Caulacanthus okamurai Yamada

Caulcanthus ustulatus (Turner) Kützing

Centroceras clavulatum (C. Agardh) Montagne

Ceramium flaccidium (Harvey *ex* Kützing) Ardissone

Ceramium mazatlanense Dawson

Ceramium tenuissimum (Roth) Areschoug

Chamaebotrys boergesenii (Weber-van Bosse) Huisman

Champia compressa Harvey

Champia parvula (C. Agardh) Harvey

Chondria armata (Kützing) Okamura

Chondria minutula Weber-van Bosse

Chondria rainfordii Lucas

Chondria succentula (J. Agardh) Falkenberg

Choreonema thuretii (Bornet in Thuret & Bornet) Schmitz

Corallina berteri Montagne in Harvey

Corallina officinalis Linnaeus

Crouania sp.

Cryptonemia sp.

Dasya flagellifera Børgesen

Dasya pilosa (Weber-van Bosse) Millar

Dasya stanleyi (Weber-van Bosse) Millar

Dasyclonium flaccidium (Harvey) Kylin

Dasyclonium incisium (J. Agardh) Kylin

Delisea pulchra (Greville) Montagne

Dictyurus purpurascens Børgesen

Digenea simplex (Wulfen) C. Agardh

Enantiocladia robinsonii (J. Agardh) Falkenberg

Erythrotrichia carnea (Dillwyn) J. Agardh

Euchema sp.

Euptilota articulata (J. Agardh) Schmitz

Fernandosiphonia nana Millar

Galaxaura marginata (Ellis & Solander) Lamouroux

Galaxaura obtusata (Ellis & Solander) Lamouroux

Galaxaura rugosa (Ellis & Solander) Lamouroux

Gelidiella acerosa (Forsskål) Feldmann & Hamel

Gelidiopsis variabilis (J. Agardh) Schmitz

Gelidium allani Lindauer

Gelidium crinale (Turner) Lamouroux

Gelidium pusillum (Stackhouse) LeJolis

Gigartina acicularis (Wulfen) Lamouroux

Gracilaria edulis (S.G. Gmelin) P.C. Silva

Gracilaria textorii (Suringar) DeToni

Gracilaria verrucosa (Hudson) Papenfuss

Grateloupia subsimplex Levring

Griffithsia sp.

Gymnogongrus sp.

Haliptilon roseum (Lamarck) Garbary & Johansen

Haloplegma duperreyi Montagne

Halymenia durvillaei Bory

Halymenia floresia (Clemente) C. Agardh

Haraldiophyllum sinuosum (Lucas) Millar

Herpopteros zonaricola Okamura

Herposiphonia secunda (C. Agardh) Ambronn

Herposiphonia subdisticha Okamura

Heterosiphonia crispella (C. Agardh) Wynne

Heterosiphonia pulchra (Okamura) Falkenberg

Heterosiphonia multiceps (Harvey) Falkenberg

Helminthocladia australis Harvey

Hydrolithon sp.

Hypnea cenomyce J. Agardh

Hypnea cervicornis J. Agardh

Hypnea chariodes Lamouroux

Hypnea cornuta Lamouroux *ex* J. Agardh

Hypnea musciformis (Wulfen) Lamouroux

Hypnea pannosa J. Agardh

Hypnea spinella (C. Agardh) Kützing

Hypoglossum harveyanum (J. Agardh) Womersley & Shepley

Hypoglossum heterocystideum (J. Agardh) J. Agardh

Jania adhaerens Lamouroux

Jania verrucosa Lamouroux

Laurencia brongniartii J. Agardh

Laurencia cartilaginea Yamada

Laurencia cruciata Harvey

Laurencia filiformis (C. Agardh) Montagne

Laurencia gracilis Hooker & Harvey

Laurencia moretonensis Cribb

Laurencia nifida sensu Cribb

Laurencia obtusa (Hudson) Lamouroux

Laurencia papillosa (C. Agardh) Greville

Laurencia pygmaea Weber-van Bosse

Laurencia pedicularioides var. *queenslandica* Cribb

Laurencia rigida J. Agardh

Laurencia tenera Tseng

Laurencia venusta Yamada

Leveillea jungermannioides (Hering & Martens) Harvey

Liagora ceranoides Lamouroux

Liagora robusta Yamada

Lithothamnion simulans Foslie

Lithothamnion sp.

Lomentaria sp.

Lophocladia sp.

Lophosiphonia prostrata (Harvey) Falkenberg

Lophosiphonia subadunca (Kützing) Falkenberg

Lophothalia harveyi (Kützing) Schmitz

Martensia fragilis Harvey

Mastophora affinis Foslie

Melobesia sp.

Melanamansia daemelii (Sonder) Norris

Melanamansia dietrichiana (Grunow) Norris

Melanamansia glomerata (C. Agardh) Norris

Monosporus australis (Harvey) J. Agardh

Murrayella periclados (C. Agardh) Schmitz

Myriogramme bombayensis Børgesen

Myriogramme pulchellum (Harvey) Millar

Nitophyllum crispum Hooker & Harvey

Peyssonnelia sp.

Platysiphonia delicata (Clemente) Cremades

Plocamium hamatum J. Agardh

Polysiphonia blandii Harvey

Polysiphonia denudata (Dillwyn) Greville ex Harvey

Polysiphonia infestans Harvey

Polysiphonia macrocarpa Harvey

Polysiphonia mollis Hooker & Harvey

Polysiphonia opaca (C. Agardh) Zanardini

Polysiphonia platycarpa Børgesen

Polysiphonia sertularioides (Grateloup) J. Agardh

Polysiphonia scopularum Harvey

Porphyra denticulata Levring

Porphyridium purpureum (Bory) Drew & Ross

Portieria hornemannii (Lyngbye) P.C. Silva

Pterocladia capillacea (S.G. Gmelin) Bornet

Pterosiphonia pennata (Roth) Falkenberg

Rhodymenia leptophylla J. Agardh

Sacronema filiforme (Sonder) Kylin

Scinaia tsinglanensis Tseng

Solieria robusta (Greville) Kylin

Spyridia filamentosa (Wulfen) Harvey

Stictosiphonia kelanensis (Grunow *ex* Post) R.J. King & C. Puttock

Symphyocladia marchantioides (Harvey) Falkenberg

Taenioma perpusillum (J. Agardh) J. Agardh

Tolypiocladia glomerulata (C. Agardh) Schmitz

Tricleocarpa cylindrica (Ellis & Solander) Huisman & Borowitzka

Valeriemaya geminata Millar & Wynne

Wrangelia sp.

Aspects of the Winter Phytoplankton Community of Moreton Bay

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Abstract

Knowledge of species composition and distribution of phytoplankton communities in coastal waters is essential for research on nutrient and trophic dynamics as well as monitoring for potential harmful species. Previous data on phytoplankton species present in Moreton Bay are dated, scant and focused on large 'net' phytoplankton. Phytoplankton from whole water samples and tows (20 µm net) throughout Moreton Bay were identified and documented from May to September, 1996. Diatom communities present in the central and northern Bay during a large phytoplankton bloom in May were dominated by common, estuarine centric species (e.g. *Guinardia flaccida*, *Rhizosolenia setigera*, *Skeletonema costatum* and *Thalassiosira* spp.) and the pelagic pennate diatom *Pseudonitzschia* sp. In southern Moreton Bay, phytoplankton communities were dominated by benthic diatom species (e.g. *Melosira sulcata*, *Bacillaria paxillifer* and *Nitzschia* spp.). Dinoflagellate species, whilst rarely dominant within the Bay, were ubiquitous and diverse. Although large thecate species previously described from the Bay were found (e.g. *Prorocentrum micans* and *Protoperidinium depressum*), small (< 25 µm) previously unrecorded species (e.g. *Scrippsiella trochoidea*, *Prorocentrum minimum*, undescribed *Gymnodinium* sp.) were often present. The large, predacious dinoflagellate *Noctiluca scintillans* was present in > 95% of tows over the sampling period in central and northern Moreton Bay, often with identifiable ingested diatom, dinoflagellate and silicaflagellate cells as well as pollen grains. Potential toxic dinoflagellates identified from Moreton Bay include *Dinophysis caudata*, circumstantially linked with diarrhetic shellfish poisoning, and *Ostreopsis* sp. and *Gambierdiscus toxicus*, species responsible for ciguatera poisoning. *Dinophysis caudata*, present in the northern Bay at low concentrations during the entire study, apparently represents a stable, endemic population as this species was noted in the Bay in the late 1950s. In 1996, the winter phytoplankton community of Moreton Bay thus consisted of a diverse estuarine assemblage dominated by large diatoms, which reflected benthic influences in the southern Bay and oceanic influences in the northern Bay.

Introduction

Phytoplankton (*phyto*, plant: *planktos*, wandering) are a ubiquitous, trophically cosmopolitan group within the marine environment whose distribution is often related to physical and chemical parameters. They are a taxonomically diverse group, comprising 13 classes of both prokaryotes and eukaryotes which range in size from picoplankton (< 2 µm) to large diatoms (up to 2 µm) and colonial cyanobacteria (> 2 µm). They are important in trophodynamics, both as the base of classical grazing-based marine food chains as well as heterotrophic grazers, and also play a significant role in nutrient dynamics and water quality. Phytoplankton are also a major component of the microbial loop (Azam *et al.,* 1983) in which dissolved organic material derived from phytoplankton supports bacterial production prior to transfer to classical food webs via microzooplankton grazing (Hagström *et al.,* 1988; Baretta-Bekker *et al.,* 1994).

Phytoplankton communities are comprised of both a pelagic and a benthic component. Marine pelagic phytoplankton are dominated globally by diatoms and dinoflagellates, of which there are approximately 10 000-100 000 species (Hasle & Syvertsen, 1996) and 2 000 (Steidinger & Tangen, 1996) species, respectively. Benthic phytoplankton communities are comprised of the microphytobenthos as well as species that are epiphytic on marine macrophytes. Within Moreton Bay, pelagic phytoplankton fix 4.3 x 10^5 carbon/day, accounting for 65% of primary production by marine plants in the Bay (Abal *et al.,* this volume). The contribution of

In: Tibbetts, I.R. Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 291-300.

microphytobenthos to primary production in Moreton Bay is unknown, but within Port Phillip Bay the micophytobenthos accounts for >50% of total phytoplankton primary production and is a significant factor in nutrient cycles (Harris *et al.,* 1996).

Despite their ecological importance, little is known of the taxonomic composition and community structure of phytoplankton within Moreton Bay. Although E. F. Wood included Moreton Bay in his classic studies of Australian phytoplankton (Wood, 1954; 1964a; 1964b; 1964c; 1964d), he provided few specific details of collection (e.g. season, location, abundance) and examined 'net' phytoplankton, those species which were retained on silk nets and preserved well (e.g. large diatoms and thecate dinoflagellates). Within Australian subtropical waters, picoplankton and unarmoured dinoflagellates have also been shown to be regionally important, but no data exist on these within Moreton Bay. This current study examines the phytoplankton community composition of Moreton Bay from May to December of 1996 to provide a baseline for future studies.

Methodology

Phytoplankton within Moreton Bay were sampled at 15 stations at weekly to bi-weekly intervals from mid-May to September, 1996. This period included a large, Baywide phytoplankton bloom which followed a one-in-twenty year flood event in Moreton Bay and its catchment. These sites encompassed Moreton Bay from the mouths of the Caboolture, Brisbane and Logan Rivers to the eastern Bay and from the Logan River to the entrance of the northern Bay (Heil *et al.,* Chapter 8, this volume). At each site, samples of surface water (150 mL) were collected by hand and a phytoplankton tow was conducted (two minute tow with a 20 µm mesh net from a boat at idling speed). Both whole water and tow samples were preserved on site with Lugols solution (Tomas, 1996). All samples were stored in darkness until analysed.

Phytoplankton were identified by morphological characteristics using an Olympus CH-2 microscope. Cells were counted with either a Palmer-Maloney nanoplankton counting chamber or a Sedgewick-Rafter counting chamber depending upon sample concentration. Samples with low cell concentrations (< 200 cells/mL) were concentrated prior to sampling by a modification of the Utermohl (1958) method. A 50 mL sample was placed in a centrifuge tube and allowed to settle for 24 h. Forty-five mL was then aspirated off the sample and the concentrated sample was counted as previously described. Where possible, >400 cells were counted to provide 10% accuracy at the 95% confidence level (Guillard, 1973).

Results

One hundred and forty-five species of phytoplankton were identified in samples taken from Moreton Bay from mid-May to early September, 1996 (Table 1). Diatoms were the most diverse phytoplankton class present, with 81 species identified. Dinoflagellates comprised the second most diverse class, with 54 species present. Of these, 31% were non-thecate species. Cryptophyta, Chrysophyta and Prasinophyta were present in limited numbers throughout the sampling period. The cyanobacteria *Trichodesmium erythraeum* was present from one sample in the northern Bay.

A brief description of the taxonomy and ecology of phytoplankton species which appear to be important in the ecology of Moreton Bay in winter of 1996 follows. Species were selected based upon their (1) numerical dominance, (2) role in trophodynamics, and/or (3) involvement in potential public health issues.

Table 1. Phytoplankton species identified from Moreton Bay from May to September, 1996.

Bacillariophyceae
Amphora javonica
Amphiprora sp.
Asterionellopsis glacialis
Bacillaria paxillifera
Bacteriastrum delicatulum
Bacteriastrum furcatum
Cerataulina dentata
Cerataulina pelagica
Chaetoceros alboranii
Chaetoceros ceratosporus
Chaetoceros compressus
Chaetoceros constrictus
Chaetoceros curvisetus
Chaetoceros danicus
Chaeatoceros debilis
Chaetoceros decipiens
Chaetoceros didymus
Chaetoceros eibenii
Chaetoceros lauderi
Chaetoceros lorenzianus
Chaetoceros peruvianus
Chaetoceros radicans
Chaetoceros socialis
Chaetoceros sp. 'b'
Coscinodiscus centralis
Coscinodiscus granii
Coscinodiscus sp.
Cyclotella sp.
Cylindrotheca closterium
Dactyliosolen blavyanus
Dactyliosolen fragellissima
Detonula pumila
Ditylum brightwellii
Eucampia cornuta
Eucampia zodiacus
Fragilariopsis sp.
Grammatophora marina
Guinardia delicatula
Guinardia flaccida
Guinardia fragilissimus
Guinardia striata
Gyrosigma sp.
Heliotheca tamesis
Hemiaulus hauckii
Hemiaulus indicus
Histionesis sp.
Leptocylindrus minimus
Leptocylindrus danicus
Lichmophora abbreviata
Melosira nummuloides
Melosira sulcata
Meuniera membranacea
Navicula bullata
Navicula granulata
Nitzschia longissima
Nitzschia reversa
Nitzschia seriata
Nitzschia sp.
Odontella mobiliensis
Paralia sp.
Pleurosigma angulatum
Pleurosigma directum
Porosira sp.
Proboscia alata
Pseudonitzschia sp.
Rhizosolenia imbricata
Rhizosolenia robusta
Rhizosolenia setigera
Rhizosolenia stolterfothii
Skeletonema costatum
Striatella unipunctata
Thalassiosira allenii
Thalassiosira curviseriata
Thalassiosira mala
Thalassiosira pseudonana
Thalassiosira rotula
Thalassionema frauenfeldii
Thalassionema nitzschioides
Toxarium hennedyanum
Triceratium favus

Dinophyceae
Alexandrium minutum
Alexandrium sp.
Amphidinium acutissimum
Amphidinium caterae
Amphidinium flagellans
Amphidium klebsii
Cochlodinium sp.
Ceratium furca
Ceratium fusus
Ceratium pentagonum
Ceratium trichoceros
Ceratium tripos
Dinophysis accuminata
Dinophysis caudata
Dinophysis diegensis
Dinophysis tripos
Gambierdiscus toxicus
Gonyaulax digitale
Gonyaulax polygramma
Gonyaulax spinifera
Gymnodinium allophron
Gymnodinium dentatum
Gymnodinium galeatum
Gymnodinium guttiforme
Gymnodinium parvum
Gymnodinium pellucidum
Gymnodinium sanguineum
Gymnodinium simplex
Gymnodinium sp.
Gyrodinium spirale
Gyrodinium nasutum
Gyrodinium sp.
Katodinium rotundatum
Noctiluca scintillans
Ostreopsis lenticularis
Ostreopsis siamensis
Oxytoxum longiceps
Peridinium hirobis
Peridinium minusculum
Podolampus elegans
Polykrikos kofoidii
Polykrikos schwartzii
Prorocentrum compressum
Prorocentrum gracile
Prorocentrum lima
Prorocentrum micans
Prorocentrum minimum
Prorocentrum scutellum
Protoperidinium depressum
Protoperidinium divergens
Protoperidinium steinii
Pyrophacus horologium
Pyrophacus steinii
Scrippsiella troichoidea

Cryptophyceae
Chroomonas sp.
Rhinomonas sp.

Chrysophyceae
Dictyocha fibula
Dictyocha speculum

Cyanophyceae
Trichodesmium erythraeum

Ebriideae
Ebridia tripartitia

Rhaphidophyceae
Heterosigma sp.

Euglenophyceae
Eutreptiella sp.

Prasinophyceae
Pyramimonas adriatica
Pedinomonas sp.

Dinophysis spp. (Dinophyceae)

Species of the dinoflagellate genus *Dinophysis* are responsible for diarrhetic shellfish poisoning (DSP), a human gastro-intestinal disorder caused by the consumption of toxin-laden shellfish containing polyether-type toxins derived from ingested cells. *Dinophysis* species have been directly linked with toxic DSP outbreaks in diverse regions of the world (Japan (Yasumoto *et al.,* 1979); Norway (Sechet *et al.,* 1990; Dahl *et al.,* 1995); France (Belin, 1993); Spain (Blanco *et al.,* 1995) and Ireland (Jackson & Silke, 1995)). The presence of this genus in Moreton Bay is of concern because: (1) *Dinophysis* has exhibited a global geographical increase in distribution over the last decade (Bravo *et al.,* 1995), (2) *Dinophysis* species can be toxic at extremely low water column concentrations (e.g. <200 cells/L, Lassus *et al.,* 1985) and (3) blooms of *Dinophysis* have been circumstantially linked with the input of sewage-derived nutrients (Santhanam & Srinivasan, 1996).

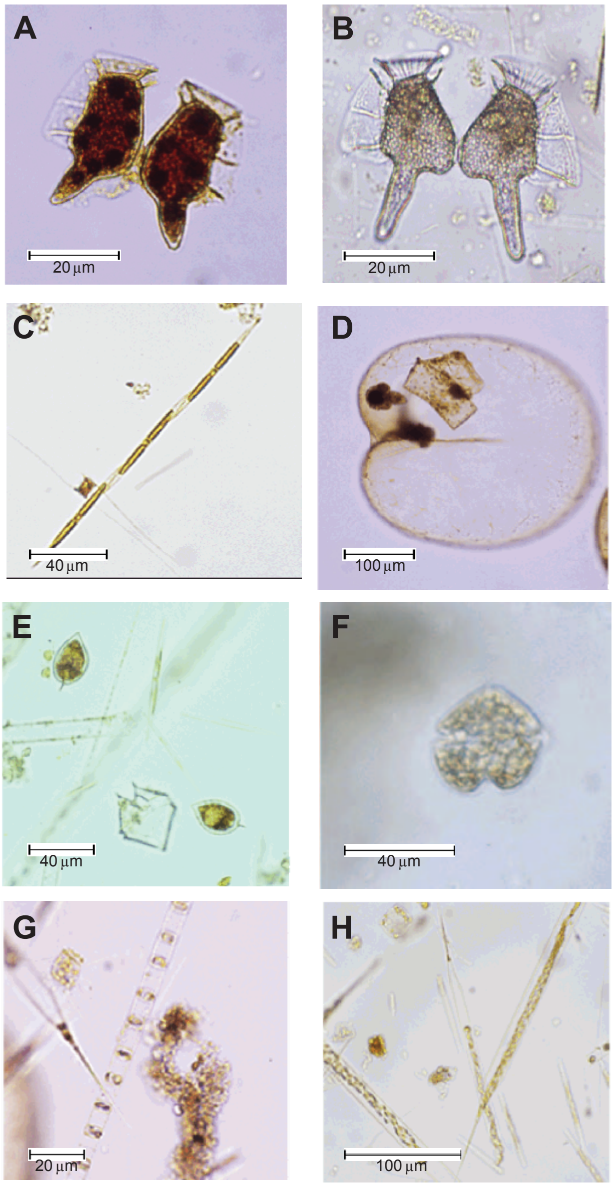
Dinophysis spp. are widely distributed in tropical, subtropical and warm temperate seas. They exhibit a cosmopolitan distribution within Australian waters (Hallegraeff & Lucas, 1988) where they are generally considered to be estuarine species (Wood, 1964f). Within Moreton Bay, *Dinophysis caudata* Saville-Kent was the numerically dominant species during the study period. Both *Dinophysis acuminata* Claparede & Lachmann and *Dinophysis diegensis* Kofoid were also present. Wood (1954; 1964f) described distinct regional morphotypes of *D. caudata* in Australian waters. The taxonomy of *Dinophysis* is based upon thecal morphology (e.g. size and shape of the plates and sulcal lists) and is currently considered inadequate due to the large degree of morphological variability present in the genus (Solumn, 1962; Hansen, 1993; Bravo *et al.,* 1995). The variant of *D. caudata* observed in Moreton Bay (Figure 1A) was morphologically distinct from specimens observed north of Moreton Bay at the mouth of the Maroochy River, at sites within the Great Barrier Reef (Heron Island (Figure 1B), and Low Isles; unpublished data). The Moreton Bay morphotype displayed a variable shape with two or more chloroplasts and weak aureolation on the thecal surface and corresponded to the 'angular' Australian form described by Wood (1954). The reef type of *D. caudata* corresponds to the 'rounded' form which Wood (1954) described as characteristic of the Coral Sea. This morphotype is characterised by heavy plates, prominent aureolation, a rigid structure and the lack of chloroplasts.

Dinophysis spp. in Moreton Bay were found primarily within the northern Bay, in the region from the mouth of the Caboolture River northward. Highest concentrations were observed at Shark Spit, on the western side of Moreton Island, although cells of *D. caudata* were occasionally observed at Peel Island. No cells were observed in phytoplankton tows south of Bramble Bay. Wood (1964f) described *D. caudata* as endemic to Moreton Bay. Its presence more than 30 years later is indicative of its persistence within Moreton Bay, despite possible changes in water quality over time. Its restriction to sites within the northern Bay, especially the northeastern section, suggests, however, that it prefers higher salinity (> 35), oligotrophic Bay waters.

Noctiluca scintillans **(Macartney) Ehrenberg (Dinophyceae)**

Noctiluca scintillans is a neritic, cosmopolitan dinoflagellate which is morphologically, taxonomically and nutritionally distinctive. It is large (200-600 µm); single cells are visible to the naked eye, and can be bioluminescent or non-bioluminescent. Only during gametogenesis does *N. scintillans* exhibit a shape characteristic of gymnodiniod dinoflagellates: vegetative cells are generally balloon-shaped with a single, large tentacle used to obtain prey. Exclusively polyphagous, this species is a voracious predator of diatoms and dinoflagellates as well as fish, zooplankton eggs and larvae (Kimor, 1979; Daan, 1987) and has been strongly implicated in the decline of coastal fisheries in India (Bhimachar & George, 1950) and Indonesia (Adnan, 1985). Often occurring in quantities sufficient to discolour water, this species has also been reported to cause fish kills due to localised oxygen depletion (Aiyer, 1936) as well as ammonia toxicity (Adnan, 1989; Devassay, 1989).

In Australia, *N. scintillans* is common in east coast estuaries (Wood, 1964) and in coastal waters in the vicinity of river mouths (Hallegraeff, 1993). Hallegraeff (1993) reported that *N.*



A.

Dinophysis caudata from Moreton Bay ('angular' or 'bay' type).

B.

Dinophysis caudata from Heron Island ('rounded' or 'reef' type).

C.

Pseudonitzschia sp.

D.

Noctiluca scintillans with ingested diatom cell.

E.

Prorocentrum micans.

F.

Gymnodinium sp. from Bramble Bay.

G.

Skeletonema costatum.

H.

Rhizosolenia setigera.

Figure 1. Light micrographs of important Moreton Bay phytoplankton.

scintillans is responsible for red tide blooms in Lake Macquarie, New South Wales. Although Wood (1964f) reported *N. scintillans* present in 25% of tows from Moreton Bay, he did not consider it to be an important species in Australian waters.

During the present study, the highest concentrations of *N. scintillans* (Figure 1C, D) were consistently observed in northern Moreton Bay, usually at the mouths of the Brisbane and Caboolture Rivers. Noctiluca populations within Moreton Bay were bioluminescent but did not possess an endosymbiont. Highest *N. scintillans* concentrations coincided with high concentrations of diatoms and dinoflagellates, although we observed a surface slick comprised entirely of *N. scintillans* ingesting *Pinus radiata* pollen in northeastern Moreton

Bay. Its presence in > 95% of phytoplankton tows during the sampling period, frequently with identifiable ingested phytoplankton cells, suggests that this species is an important grazer of phytoplankton within the Bay and may, when present in large concentrations, have a significant negative impact on zooplankton and fish populations via grazing of egg and larval stages.

Prorocentrum micans Ehrenberg (Dinophyceae)

The dinoflagellate *Prorocentrum micans* is a common, neritic bloom-forming species with a world wide distribution. It is considered the most globally abundant species within this genus (Yoo & Lee, 1986) and is characterised by an extremely variable cell shape such that local forms are often described as new species (Dodge, 1975). Wood (1954) described *P. micans* as exhibiting a highly variable morphology in Australian waters. This species was reported as continually present at Port Hacking, but especially abundant in January, April and September (Hallegraeff & Reid, 1986).

Prorocentrum micans (Figure 1E) was the third most abundant dinoflagellate found during this survey. It was present in relatively high concentrations (> 2.3 x 10^4 cells/L) at the mouth of the Brisbane and Caboolture Rivers and within Deception and Bramble Bays. Wood (1964f) recorded Throughout the year; as either dominant or subdominant species. This species often occurs in eutrophic areas, characterised by high concentrations of organic nutrients, and has been shown to utilise both inorganic and organic nutrients (Mahoney & McLaughlin, 1977). Its distribution within Moreton Bay suggests that nutrient input from rivers is an important determinant in its distribution within the Bay.

Gymnodinium sp. (Dinophyceae)

A previously undescribed species of dinoflagellate, *Gymnodinium* sp. (Figure 1F), was the most numerically abundant dinoflagellate observed in Moreton Bay during the survey period. The non-thecate genus *Gymnodinium* is differentiated from the similar genus *Gyrodinium* solely on the basis of cingulum (girdle) displacement; displacement of *Gymnodinium cingulum* is < 1/5 the cell length whereas that of *Gyrodinium* is > 1/5. It has been suggested, however, that cingulum displacement is not a conservative feature (Kimball & Wood, 1965) and due to preservation effects (e.g. stressed and preserved cells readily change their morphology, Haywood *et al.,* 1996), this genus is easily confused and misidentified.

Gymnodinium sp. from Moreton Bay was 35 μm in length, dorsoventrally flattened with a distinct apical groove and epitheca and hypotheca of equal lengths. No ventral concavity was observed and nuclear placement could not be determined from preserved samples. This species was easily distinguished in preserved material due to its sulcus, which was extremely excavated toward the antapex of the cell, and its general shape.

This species reached concentrations of greater than 2.6 x 10^5 cells/L in Moreton Bay during the survey period and was found primarily on the western side of the mid and northern Bay. *Gymnodinium breve* 'look-alikes' have been described from New Zealand (Haywood *et al.,* 1996), where a toxic event occurred in 1993 which was probably due to *Gymnodinium* (MacKenzie *et al.,* 1995). Although *Gymnodinium* sp. from Moreton Bay has morphological similarities to *Gymnodinium breve* (e.g. shape, antapical excavation of sulcus), differences exist, and identification from preserved material is inconclusive.

Skeletonema costatum (Greville) Cleve (Bacillariophyceae)

Skeletonema costatum is a common, estuarine and neritic diatom with a global distribution (Hasle, 1973). Although considerable genetic variability exists between clones of this species on a seasonal basis in temperate waters (Gallagher, 1980), phenotypic expression of this variability is highly conservative, making *S. costatum* one of the most easily recognised species of marine phytoplankton. A tropical *Skeletonema* species, *S. tropicum* (Hulburt &

Guillard, 1968) exists, but is easily distinguishable from *S. costatum* by its greater number of chloroplasts and large diameter (Hasle & Syvertsen, 1996). *Skeletonema pseudocostatum* has been described from Australian waters (Medlin *et al.,* 1991) but may not be an actual species (J. Gallagher, pers. comm.).

A common species of temperate Australian estuaries (Wood, 1964a; 1964b; 1964e; Hallegraeff & Jeffrey, 1984; Hallegraeff & Reid, 1986), *S. costatum* is one of the dominant diatom species in the September spring bloom in Port Hacking (Jeffrey & Carpenter, 1974; Hallegraeff & Reid, 1986) but is also a component of the brief, episodic diatom blooms which characterise this region (Hallegraeff & Jeffrey, 1993). Wood (1964b) makes no mention of *S. costatum* in his list of diatom species from Moreton Bay. Its presence at other Australian sites (e.g. Port Hacking, Port Philip) in Wood's study suggests that the absence of *S. costatum* from Moreton Bay was not due to sampling protocol, but either the sample timing and location, or an actual absence from the phytoplankton community. *Skeletonema costatum* (Figure 1G) was the third most numerically abundant diatom during the sampling period, with concentrations up to 1.0 x 10^5 cells/L. The presence of this species in the central and southern Bay during the diatom bloom in May probably reflects its relatively rapid growth rate and ability to utilise low light levels (Smayda, 1973; Cosper, 1982).

Rhizosolenia setigera Brightwell (Bacillariophyceae)

Rhizosolenia setigera is a common coastal phytoplankton species of North America (Cupp, 1971; Marshall & Cohn, 1987), Europe (Hendley 1964) and Australia (Wood, 1964a–f). It probably has a ubiquitous coastal distribution as it has been described as tolerant of a wide range of temperatures and salinities (Wood, 1964a–f). It has been reported as a species that is continually present in coastal waters off Sydney (Hallegraeff & Reid, 1986) but was rare in a warm core eddy of the East Australian Current (Jeffrey & Hallegraeff, 1980).

Wood (1954) found *R. setigera* in 55% of Moreton Bay samples, suggesting that it is a relatively consistent and common component of Moreton Bay phytoplankton. *Rhizosolenia setigera* (Figure 1H) was the dominant diatom species in Moreton Bay over the survey period, reaching concentrations >4.5 x 10^2cells/L. Highest concentrations were observed in the mid-region of Moreton Bay, associated with the Brisbane River plume.

Discussion

Jeffrey & Hallegraeff (1990) characterised three unique phytoplankton assemblages within Australian waters: a temperate neritic community, a tropical neritic community and a tropical oceanic community. Based on their map of the location of these assemblages, Moreton Bay phytoplankton communities should exhibit a tropical oceanic phytoplankton community similar to that of the Coral Sea, with a diverse dinoflagellate assemblage, symbiotic associations and a high percentage of nanoplankton and picoplankton. During the current study however, phytoplankton assemblages of Moreton Bay were more similar to those of the temperate neritic community of New South Wales. Chain-forming diatoms (e.g. *S. costatum*, *R. setigera*) were common, and picoplankton and nanoplankton comprised a low percentage of the community numerical abundance.

As this study was undertaken during a period in which Moreton Bay experienced a one-in-twenty year flood event, it is unknown whether this community is typical of the winter phytoplankton of the Bay. Systematic information on phytoplankton assemblages in Moreton Bay is restricted to Wood (1954; 1964a–f) or to species of economic or ecological concern (e.g. parasitic (Hudson & Shields, 1994) and ciguatera (Gillespie *et al.,* 1985) dinoflagellates). Sampling site locations are not specified in Wood's surveys and neither seasonal abundance nor concentration data were presented. Wood concentrated on large 'net' diatoms and dinoflagellate species which survived the harsh preservation techniques of the period. Currently there is no baseline knowledge of a phytoplankton taxa of Moreton Bay, or of the contribution of nanoplankton (2-20 μm size fraction) and non-thecate dinoflagellates to this community. The importance of a non-thecate dinoflagellate, *Gymnodinium* sp., which is yet to be described, during the period of study further underscores this lack of knowledge of phytoplankton species which are difficult to preserve.

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Factors Limiting Phytoplankton Biomass in the Brisbane River and Moreton Bay

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Abstract

Increasing eutrophication of coastal marine environments has led to the development of nutrient sampling programs to monitor water quality. Various shortcomings of chemical analyses common in the majority of these sampling programs have identified the need to develop biological indicators (bioindicators) that can be used to detect the source, fate and impact of nutrients within eutrophic systems. Using phytoplankton bioassays, we tested the accepted model that phosphorus (P) limits phytoplankton biomass in freshwater and nitrogen (N) is limiting in coastal marine waters. The study was conducted along a salinity gradient from Lake Wivenhoe (0 ppt), through the nutrient rich, highly turbid Brisbane River to Moreton Bay (35 ppt). Water samples from seven sites along the transect were spiked to make treatments with the following nutrient concentrations: 30 μM NH_4^+; 200 μM NO_3^-; 20 μM PO_4^{3-}; 66 μM SiO_3^{2+}; all nutrients combined and an unspiked control. Chlorophyll *a* fluorescence was measured daily for 7 d in each treatment. Nutrient limitation was inferred if there was an increase in chlorophyll *a* fluorescence in nutrient spiked samples relative to the control. Light limitation was inferred from an increase in chlorophyll *a* fluorescence in controls. Phytoplankton bioassay results indicate that phytoplankton biomass was limited by: N and P in the freshwater sites; N in the upper reaches of the Brisbane River; light in the middle reaches; and N and silica (Si) in the lower reaches. Within Moreton Bay, phytoplankton populations exhibited no response to nutrient addition. These results suggest that the generalised models of nutrient limitation may not be applicable to all regions. In particular, P may not limit phytoplankton at any salinity regime within the Moreton region.

Introduction

The Brisbane River is the largest tidal estuary flowing into Moreton Bay, and is characterised by high turbidity. It receives large inputs of nutrients from both point and non-point sources such as terrestrial runoff, sewage treatment plants and release from resuspended sediments (Moss, 1990). To assess the role of nutrients and suspended solids in eutrophication, traditional water quality analyses involved periodic sampling of parameters such as dissolved inorganic nitrogen (DIN) and phosphorus (DIP), chlorophyll *a*, and total suspended solids (TSS). These techniques are limited as they only provide an instantaneous measurement at the time of water collection whereas large fluctuations in the concentrations of dissolved nutrients can occur on short time scales in estuaries (Wheeler & Björnsäter, 1992; Valiela, 1995). Additionally, these analyses do not directly assess the impact of eutrophication on marine life in the system (Lyngby, 1990). Bioassays can be used to investigate the nutrient responses of the phytoplankton community, thereby providing information on the history of nutrient availability at a site.

Some bioassay studies supply all but one nutrient to each treatment containing an axenic culture of phytoplankton (Smayda, 1974; Hitchcock & Smayda, 1977). If the particular treatment shows lower growth rates than the treatment with all nutrients added, then that nutrient is considered to be limiting. The technique used in the present study has been employed widely (Valiela, 1995) and uses ambient phytoplankton populations spiked with single nutrients. Rapid increases (before the other ambient nutrients have been assimilated) in phytoplankton biomass in such treatments indicate that the nutrient was limiting for the ambient phytoplankton community.

In Moreton Bay, phytoplankton bioassay techniques have been used to assess both short term

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 301-308.

(~15 hr) physiological responses to nutrient enrichment and long term (up to 7 d) responses in biomass to nutrient enrichment. Short term bioassays measure CO_2 uptake via ^{14}C after 15 h incubations in artificially increased nutrient concentrations (O'Donohue & Dennison, 1997). Long term bioassays examine changes in algal biomass with nutrient additions (O'Donohue *et al*., this volume). Changes are measured as *in vivo* fluorescence, which can be directly correlated to chlorophyll *a* concentration. Interpretations of the nutrient(s) limiting to phytoplankton can be made based on how rapidly the population increases and to which nutrient(s) they respond. In some cases the population may be adapted to oligotrophic conditions and may not be capable of rapid assimilation of the nutrients. In a system where nutrients are available, but increases in biomass are limited by the absence of one particular nutrient (most commonly nitrogen), then supplying this nutrient will result in increases in biomass. When the phytoplankton community is not limited by either N or P alone; the addition of a combination of N and P may produce a marked response. Light limitation may be inferred from increases in biomass in the controls (no nutrient), after suspended solids settle out of suspension.

This investigation was conducted to identify factors limiting phytoplankton biomass in the Brisbane River estuary and Moreton Bay, and to define more accurately where efforts should be directed to best monitor water quality in the region. Ultimately, this information will benefit decision making on a number of management issues, particularly nutrient removal strategies.

Materials and Methods

Seven sites were selected along a transect from Lake Wivenhoe, along the Brisbane River and into Moreton Bay (Figure 1). The transect spans the full salinity range from freshwater (0), through the tidal reaches of the river, to full salinity (35) seawater. Sites will be referred to throughout the text as distance in kilometres upstream (negative) or distance downstream (positive) from the mouth of the Brisbane River.

Water quality

Chlorophyll *a* was determined by filtering a known volume of water sample through Whatman GF/F filters, which were immediately frozen. Acetone extraction and calculation of chlorophyll *a* concentration was performed using the methods of Clesceri *et al*. (1989) and Parsons *et al*. (1989).

The water collected from filtering for chlorophyll *a* analysis was transferred into 120 mL polycarbonate containers and immediately frozen. NH_4^+ and NO_3^-/NO_2^{2-} were determined within two weeks of sampling using the methods of Parsons *et al*. (1989).

The concentration of total suspended solids (TSS) was determined by filtering a known volume of water onto a pre-weighed and pre-combusted (110ºC; 24 h) Whatman GF/C glass fibre filter. The filter was then oven dried at 60ºC for 24 h and TSS calculated by comparing the initial and final weights.

Secchi depth was determined by lowering a 30 cm diameter secchi disk (black and white alternating quarters) through the water column until it was no longer possible to distinguish between the black and white sections.

Bioassays

Phytoplankton bioassays were conducted with ambient phytoplankton assemblages collected from seven sites in the Brisbane River and Moreton Bay (Figure 1). One 30 L drum of water

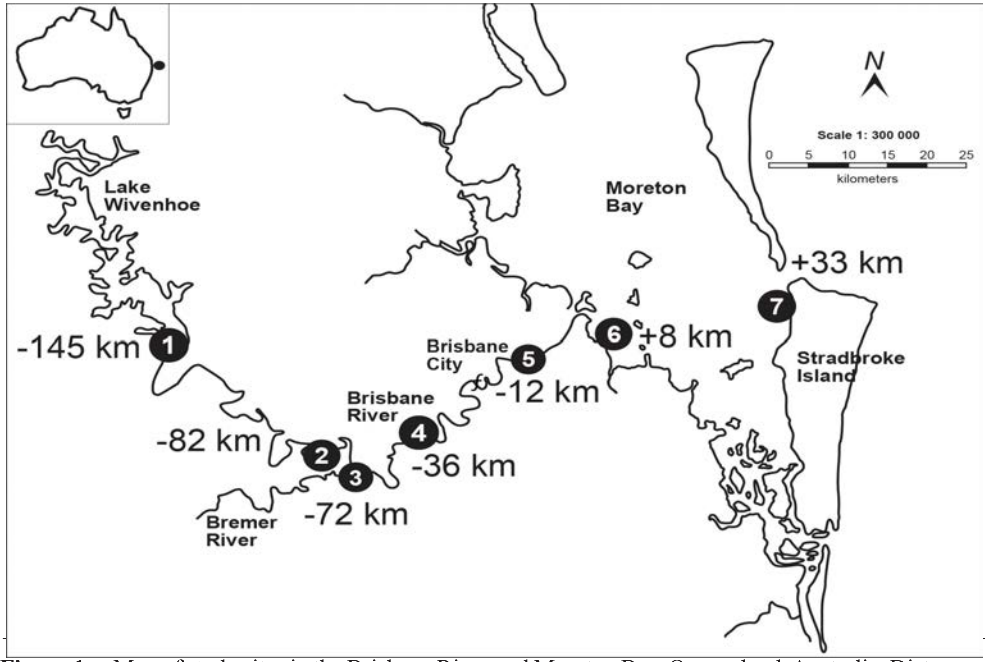


Figure 1. Map of study sites in the Brisbane River and Moreton Bay, Queensland, Australia. Distances are relative to the mouth of the river (negative upriver from the mouth and positive into Moreton Bay). Site 1 – Lake Wivenhoe (-145 km); Site 2 – Karana Downs (-82 km); Site 3 – Bremer River Junction (-72 km); Site 4 – Long Pocket (-36 km); Site 5 – Gateway Bridge (-12 km); Site 6 – South of Fisherman's Island (+8 km); Site 7 – Myora (+33 km).

was collected from each site, kept cool and shaded, and returned to an outdoor incubation facility. Four litres of water from each site was filtered through a 200 µm mesh (to screen out the larger zooplankton grazers) into sealed transparent 6 L plastic containers and placed in incubation tanks filled with water (2 m diameter, 0.5 m deep). Temperature was maintained at ±2°C of the ambient found at each site by flowing water through the tanks and light levels were maintained at 50% of incident irradiance with neutral density screening. For each site there were six bioassay containers, each with a different nutrient treatment. Samples were spiked to make the following concentrations: NO_3^- (200 µM); NH_4^+ (30 µM); PO_4^{3-}(20 µM); SiO_3^{2+} (66 µM); all nutrients at those concentrations (+All); and a control (no nutrient addition). The concentrations were chosen to ensure saturation by each particular nutrient based on the highest ambient concentrations at the study sites. *In vivo* fluorescence was measured for all treatments daily for 7 d, using a Turner Designs Fluorometer.

The potential of light and nutrients to stimulate significant increases in phytoplankton biomass (blooms) in the bioassays was investigated. The nutrient control treatments functioned as light response treatments because sedimentation of suspended solids in the samples increased light availability above ambient levels. Light stimulated bloom potential was calculated as the difference between initial and maximum *in vivo* fluorescence values in the control water sample over 7 d. Nutrient stimulated bloom potential was calculated as the difference between the +All nutrients treatment and the control.

Results

Water quality

The seven study sites occur along a salinity gradient from freshwater at -145 km to full strength seawater in the Bay sites. Water column NH_4^+ concentration ranged from <1.5 µM at +33 km to a peak of 11 µM at -12 km. NO_3^- ranged from <4 µM at +33 km to 112 µM at -36 km, near the middle of the river. TSS concentration ranged from 4 mg/L at +33 km to 23 mg/L at -36 km and the chlorophyll *a* concentration from 0.5 µg/L at +33 km to 12 µg/L at -72 km (Figure 2).

Bioassays

Freshwater sites (-145 km and -82 km) demonstrated responses in phytoplankton biomass in the +All treatments, with little or no difference in the controls and other nutrient additions. At the estuarine sites (-36 km, -12 km, +8 km), phytoplankton biomass increased in all treatments, and the control. At -72 km, the response in the control was not as great as that in the treatments. The response of phytoplankton at +8 km was primarily to nitrogen (NH_4^+ and NO_3^-) and then to silica (SiO_3). Populations at the oceanic site (+33 km) showed no almost no response (Figure 3).

Discussion

Nutrient (source and concentration) and light availability (from secchi depth measurements) vary considerably along the gradient of the Brisbane River and Moreton Bay (Figure 2). The peak in NH_4^+ at the -12 km site may be due to a fertiliser plant, located at -7 km (Moss, 1990), and the Luggage Point sewage treatment plant near the river mouth (0 km) (Moss *et al.*, 1992). The relatively high concentrations of NO_3^- (compared to NH_4^+) at most of the mid and upper reaches of the river may result from non-point sources, enhanced nitrification and preferential uptake of NH_4^+ by phytoplankton (Lipschultz *et al.*, 1986).

The response of the bioassays (Figure 3) to the +All nutrients treatment at the -145 km and -82 km sites indicates limitation by more than one nutrient. At the -72 km and -36 km sites, there was no increase in biomass above the control, indicating a response to increased light due to sedimentation of particles. At the -12 km site there was also a light response, but N and Si stimulated biomass above the control. The +8 km site had almost no response in the control, but strong responses to N and Si treatments. This indicates that light is no longer a factor controlling biomass in this region of the estuary and is consistent with the higher light availability (secchi disk readings) at this site. The +33 km site showed almost no response except for the +All treatment on the seventh day. The responses in the +All treatments at day 7 may be due to bottle effects, as all the other sites responded within the first 4 days. This response suggests that the phytoplankton community within the Brisbane River is better adapted to respond rapidly to high nutrient availability compared with those in the low nutrient waters of eastern Moreton Bay.

There was an inverse relationship ($r^2 = 0.93$) between nutrient concentration and light availability (as total suspended solids) along the study transect. At some sites phytoplankton biomass was light limited (tidal estuary), and at others nutrient limited (freshwater and marine). Light limitation was observed when phytoplankton biomass from the highly turbid mid-river sites increased in controls after the suspended solids in the sample had settled out. This response indicates that once light limitation was removed, controls had sufficient ambient nutrient concentrations to grow as well as the treatments, to which nutrients had been added.

Bloom potentials, calculated as the difference in maximum biomass (as *in vivo* fluorescence)

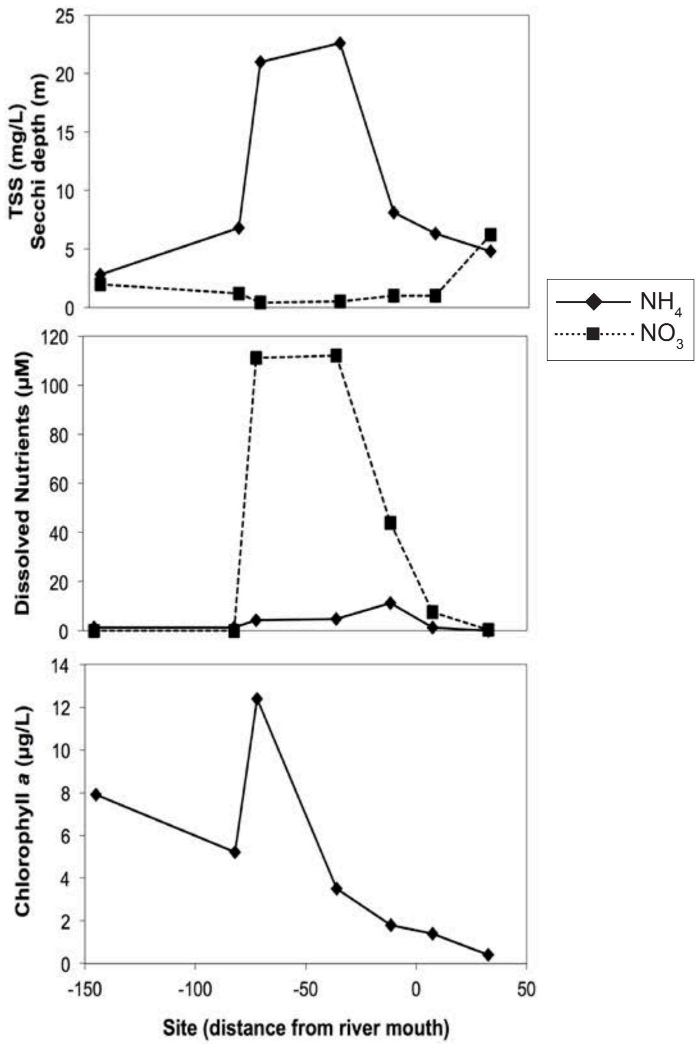


Figure 2. Water quality parameters at the seven sites along the Brisbane River/Moreton Bay study transect from Lake Wivenhoe (-145 km) to Myora in eastern Moreton Bay (+33 km).

between the treatment and control, represent the relative increase in biomass given saturating light or nutrient conditions. The upriver sites show the greatest nutrient-induced bloom potential due to relatively high light availability coupled with a environment containing relatively high concentrations of nutrients. The highly turbid midriver sites had the highest light-induced bloom potential (Figure 4).

Our new model (Figure 5) describes N and P limitation in freshwater sites, N limitation in

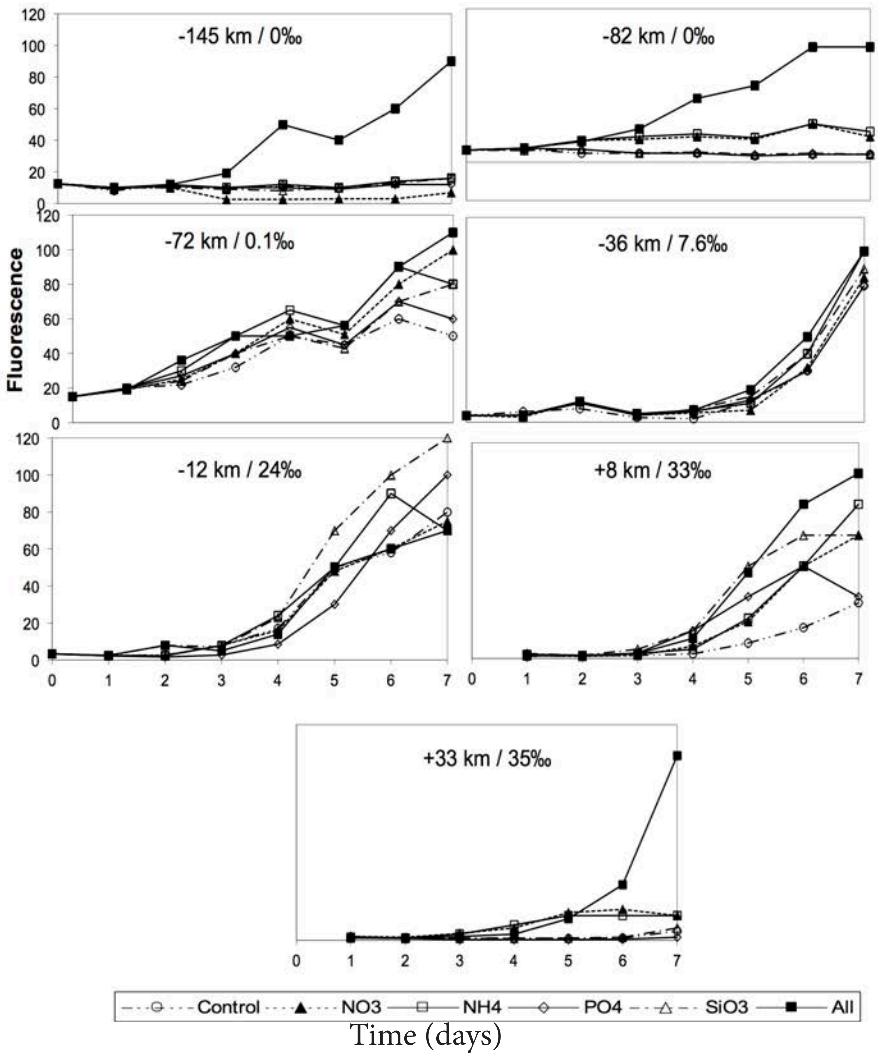


Figure 3. Phytoplankton bioassay responses at the seven sites along the Brisbane River/Moreton Bay study transect. Salinity at each site is given in parts per thousand.

the upper reaches of the river, light limitation in the middle reaches, and N and Si limitation in the lower reaches and the Bay. In contradiction to the widely accepted nutrient limitation model, we found no P limitation of phytoplankton biomass in samples from the freshwater sites. This trend has been observed elsewhere in southeast Queensland, in the Maroochy and Tweed Rivers. This departure from the accepted worldwide trend may be explained by a number of factors including large inputs from fertilisation, nutrient content of Australian soils and from

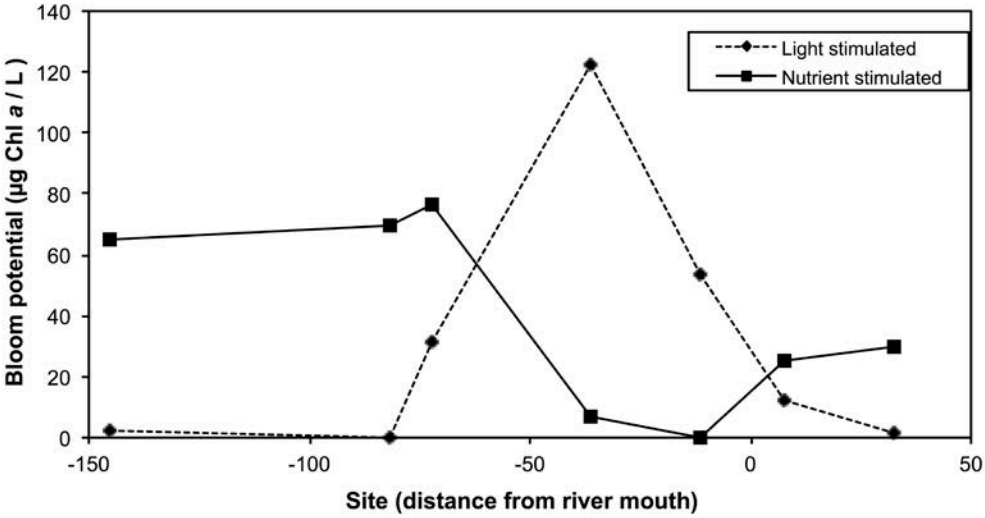
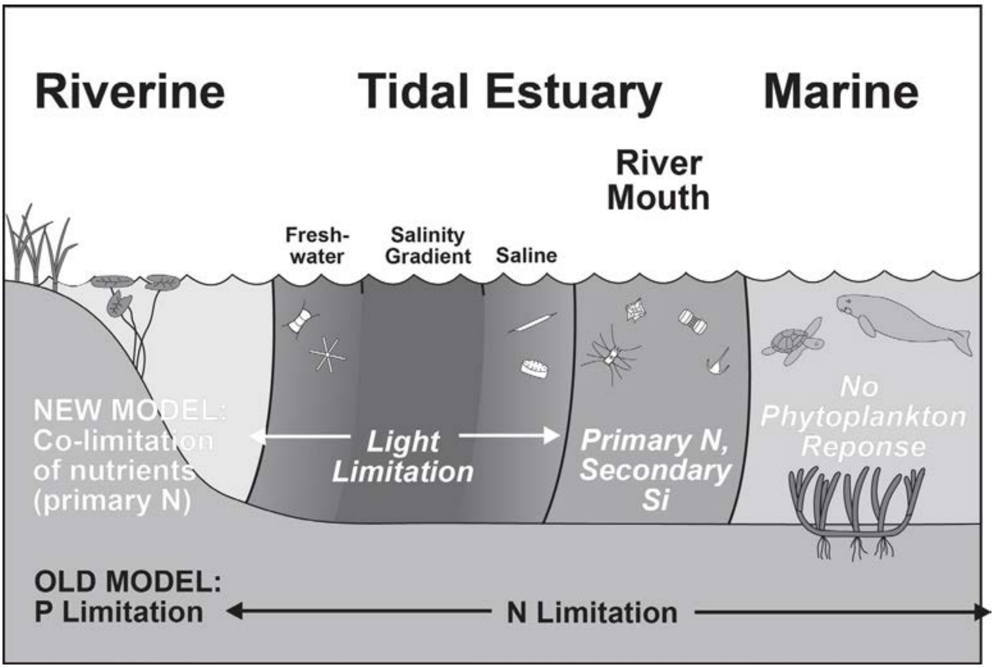


Figure 4. Light stimulated and nutrient stimulated bloom potential along the Brisbane River/Moreton Bay study transect.



(Graphics–Karen Holloway)

Figure 5. New and old models of factors limiting phytoplankton biomass in the Brisbane River and Moreton Bay system.

the high loading of suspended solids to which P binds.

These results demonstrate the need to develop a better understanding of the interactions of light and nutrients as factors limiting phytoplankton biomass in the Brisbane River estuary and Moreton Bay. Phytoplankton bioassays indicate that the established model for limiting factors may not apply to this region, and that such assays, in conjunction with traditional water quality measurements, provide key information not available from traditional water quality monitoring programs.

Acknowledgements

We would like to thank the Marine Botany practical class (BT 320) for sample collection and analysis. Andrew Moss (Queensland Department of Environment) and James McEwan (Brisbane River Moreton Bay Wastewater Management Group) provided assistance with interpretation of results.

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Baseline Monitoring of the Phytoplankton and Phytoplankton Bloom Associations in the Southern Moreton Bay Catchment

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Abstract

A baseline monitoring program was initiated in June 1995 in order to document the phytoplankton communities in five estuarine creeks that discharge into Moreton Bay, southeast Queensland. This program is specifically aimed at identifying the taxa present by assessing different fixation and processing protocols for the identification of phytoplankton, monitoring their seasonal changes in abundance and species diversity, and their response to periodic nutrient enrichment.

Water samples were collected by the Queensland Department of Environment on a routine basis from specific sites along Tingalpa Creek, Tarradarrapin Creek, Hilliards Creek, Coolnwynpin Creek and at the mouth of Weinam Creek.

All samples which were initially fixed in glutaraldehyde (final solution 3%) contained well-preserved phytoplankton. Those samples which were initially fixed in Lugols fixative contained well-preserved diatoms, however, some dinoflagellate species were poorly preserved. Subsequent processing for scanning electron microscopy proved necessary for the identification of some dinoflagellates and small diatom species which were beyond the resolution of the light microscope.

With the exception of Tarradarrapin Creek, all creeks support a wide range of phytoplankton flora. Tarradarrapin Creek is unique in that over 18 months it consistently yielded very few phytoplankton, both in number and species diversity. Samples collected from the lower tidal reach of Tingalpa Creek are dominated by diatoms with periodic blooms of *Thalassiosira rotula*, *Skeletonema costatum* and *Pleurosigma* spp. Minute centric diatoms (cf. *Cyclotella* spp.) and picoplankton blooms commonly occur in the upper reaches of Tingalpa Creek, close to Tingalpa Reservoir. A dinoflagellate bloom of *Protoperidinium nudum* has also been identified in the estuarine reaches of Tingalpa Creek. Low numbers of cyanobacteria, *Microcystis*, *Oscillatoria* and *Spirulina* have been identified in all except Tarradarrapin Creek.

Introduction

With the exception of the plankton studies commissioned by the Queensland Department of Environment (QDoE) in Tingalpa Creek (Bowles, 1991-1995) no studies have provided baseline data on the phytoplankton species composition of waterways that drain into the enclosed embayment of southern Moreton Bay. Consequently, little is known of the phytoplankton composition of these waterways and whether or not red tide organisms, notably dinoflagellates, are responsible for the periodic fish kills reported in Moreton Bay (e.g. on 17 March 1996 in Tingalpa Creek at site 0.0 of the present study approximately 200 dead silvermoon catfish were reported to John Fraser of the QDoE. G. Miller, pers. comm.)

This paper reports on the progress of a baseline monitoring program for phytoplankton in Moreton Bay and in the Moreton Bay catchment, which was initiated in June 1995 between The Centre for Microscopy and Microanalysis (The University of Queensland), QDoE (Pollution Management Branch) and Redland Shire Council. In particular, this paper illustrates, by way of light and electron microscopy, the phytoplankton common to the estuarine creeks of southeast

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Brisbane. Quantitative species counts and statistical analyses have not been conducted to date.

The research reported here forms a component of a study which aims to:

- document phytoplankton community structure in Moreton Bay and the Moreton Bay catchment;
- compile a reference collection, taxonomic descriptions, and light and electron photomicrographs of the phytoplankton in the nearshore marine environment of southeast Queensland north to Hervey Bay; and
- develop suitable fixation and processing protocols for the identification of phytoplankton, particularly naked dinoflagellates, using light microscopy and scanning and transmission electron microscopy.

Methodology

Geographical setting

Tingalpa Creek (Figure 1) is a small tidal estuary approximately 24 km long in Redland Shire, southeast Queensland. Below the Leslie Harrison Dam (Redland Shire potable water supply), the creek is estuarine with occasional freshwater releases from the dam. There are four licensed point source discharges into Tingalpa Creek, these being Thorneside sewage treatment plant, Capalaba water treatment plant, Capalaba sewage treatment plant and Edgell's lagoon (Edgell/Birds Eye food processing plant). The estuarine part of the creek drains mostly urban catchment with some industry, flower and turf farms and the Howeston Golf Course. The confluence of Coolnwynpin Creek and Tingalpa Creek occurs at approximately AMTD 8.4 (Adopted Middle Thread Distance – distance from the mouth of the creek in kilometres) along Tingalpa Creek. One licensed point source discharge occurs along Coolnwynpin Creek at approximately AMTD 0.2 from the Capalaba sewage treatment plant.

Hilliards and Tarradarrapin Creeks both drain northwards through urban and agricultural catchment; the former into Waterloo Bay and the latter to the coast just south of Wellington Point. Cleveland sewage treatment plant discharges into Hilliards Creek between AMTD 11 and AMTD 9.8. Weinam Creek drains into the southern part of Redland Bay.

Sample collection

Water samples were collected by QDoE from the AMTD sites along Tingalpa Creek, Tarradarrapin Creek, Hilliards Creek, Coolnwynpin Creek, and at the mouth of Weinam Creek (Figure 1). Samples were obtained monthly from all creeks and from Tingalpa Creek weekly for one month every six months. Water samples were collected in 250 mL or 500 mL bottles and were maintained between at approx. 15°C prior to preparation for analysis. The contents of each sample were concentrated within 24 h of collection to 5 mL by centrifugation at 2300 rpm for 15 min. and then fixed with glutaraldehyde (3% final solution). Some samples were provided already preserved in Lugols fixative. (Although not reported here, data for other water quality variables were collected simultaneously (for future comparison) with the phytoplankton samples.) All samples were collected at midstream collection points generally on an outgoing tide. Measurements of pH, temperature, conductivity, turbidity and dissolved oxygen were carried out at all sites at depths of 0.2 m and at 1 m intervals to the bottom by QDoE for routine monitoring purposes. Likewise, triplicate samples were also collected for laboratory analysis of turbidity, suspended solids, biological oxygen demand, phosphorous

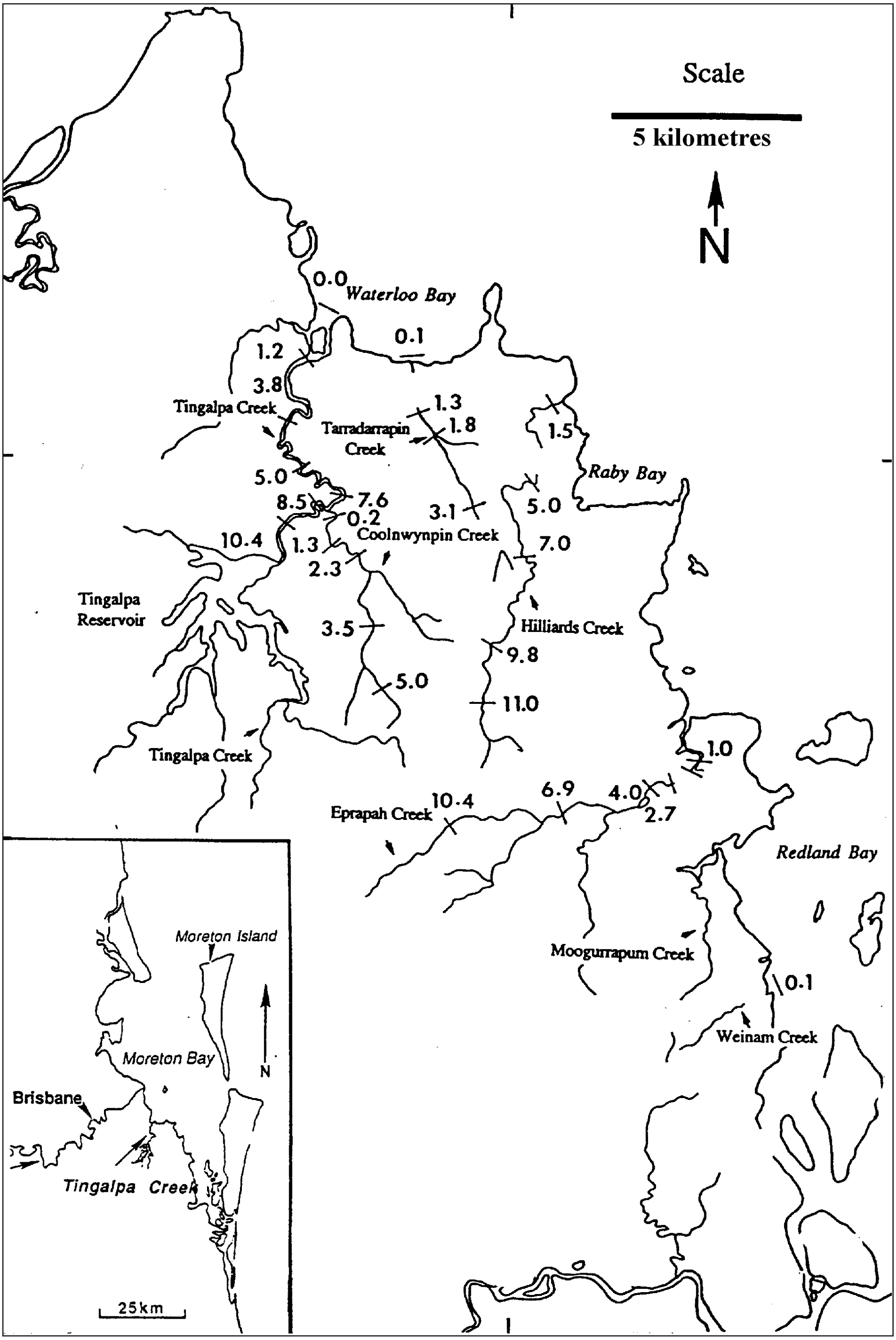


Figure 1. Map of southern Moreton Bay catchment showing the location of Tingalpa, Coolnwynpin, Tarradarrapin, Hilliards and Weinam Creeks and the location of AMTD sample collection points.

(total), phosphorous (dissolved reactive), nitrogen (oxidised), nitrogen (ammonia), nitrogen (organic) and chlorophyll *a* (see Semple *et al.*, this volume).

Sample preparation

Light microscopy

Samples for light microscopy were prepared using standard palynological techniques (Brasier, 1980) by dispersion of 0.25 mL in gelatine containing phenol. Slides were examined using an Olympus BH2 optical microscope equipped with Nomarski and phase contrast filters.

Scanning electron microscopy (SEM)

Method A – Total phytoplankton

A small amount of concentrated sample (0.25 mL) was pipetted onto plastic coverslips coated with poly-L-lysine and allowed to settle overnight. Samples were washed in a series of three 30 min. buffer washes (0.66 M) sodium cacodylate solution containing 0.15M sucrose, pH 7.2, (4^oC). These were post fixed in a mixture of 1% osmium tetroxide in freshwater or filtered sea water for 90 mins. The samples were washed twice again in buffer at 4^oC for 30 mins each, allowed to reach room temperature and then washed in high purity water (four changes of 30 mins each). Samples were dehydrated through a series of methanol or acetone solutions (15, 20, 30, 40, 50, 60, 70, 80, 90, and 100%) for 10 mins each at room temperature, transferred to fresh 100% methanol and dried in a Polaron Critical Point Dryer.

Samples were mounted on SEM stubs using colloidal silver paint, coated with platinum and imaged using a JEOL 890F field emission scanning electron microscope operating between 20-25 kV.

Method B – Diatoms

Unfixed samples were carefully filtered onto 0.25 mm millipore filters and rinsed with distilled water in order to remove excess salt. The filters with attached diatoms were mounted on SEM stubs using colloidal silver paint, coated with platinum and imaged using a JEOL 890F field emission scanning electron microscope operating between 20-25 kV.

Results

Preservation of samples

Water samples which were initially fixed in 3% glutaraldehyde contained well-preserved phytoplankton and nanoplankton. Those samples which were initially fixed in Lugols contained less well-preserved dinoflagellates (Figure 2a) when compared with well preserved specimens (Figure 2b).

Processing for SEM was necessary for the identification of some dinoflagellates and small diatom species. Samples which were filtered gently onto millipore filters and left to air dry prior to coating with platinum showed considerable distortion of some diatom frustules (Figure 2c) when compared with well preserved specimens (Figure 2d).

Common phytoplankton bloom associations

Tingalpa, Coolnwynpin and Hilliards Creeks, in the southern Moreton Bay catchment, supported phytoplankton populations dominated by centric and pennate diatoms. One essentially monospecific dinoflagellate bloom (> 20 000 cells/mL (Bowles, 1995) and periodic

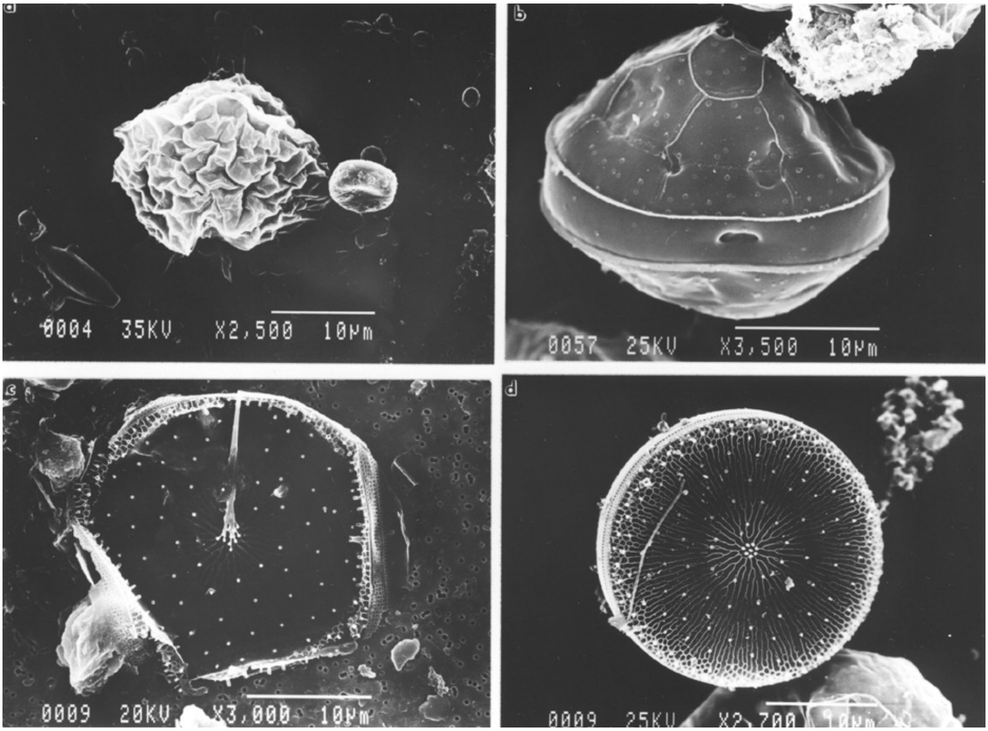


Figure 2. Scanning electron photomicrographs showing differences in sample preservation of phytoplankton due to poor fixation and dehydration. Figure 2a shows a poorly preserved *Protoperidinium nudum* and is compared with a well preserved specimen (Figure 2b). Figure 2c shows a poorly preserved specimen of *Thalassiosira rotula* due to filtration and air drying on a millipore filter, and is compared with a well preserved specimen (Figure 2d) prepared by careful dehydration and critical point drying for SEM.

intense picoplankton blooms (10^9 cells/mL) were identified in Tingalpa Creek. Samples from the mouth of Weinam Creek were dominated by small dinoflagellates. In strong contrast to the other four creeks, however, Tarradarrapin Creek yielded very low numbers (< 25 cells/mL) of phytoplankton throughout the 1995-1996 period.

Phytoplankton, phytoplankton blooms and phytoplankton bloom associations

Relative abundances of phytoplankton were classified according to the following definitions:

- Low, less than 5% of an assemblage < 25 cells/mL
- Abundant, more than 25% of an assemblage > 100 cells/mL
- Blooms, more than 50% of an assemblage > 20 000 cells/mL
- Intense blooms, more than 75% of an assemblage, 10^9 cells/mL

Diatom bloom associations

Eight putative phytoplankton bloom associations were detected (Table 1). Five occurred in marine and brackish waters and three in brackish and freshwater environments.

Marine/brackish environments

Association 1. Near monospecific blooms of *Thalassiosira rotula* (Figures 3a; b) which are associated with low numbers of large centric diatoms, *Coscinodiscus* spp.

Association 2. Blooms dominated by *Skeletonema costatum* (Figures 3c; d) in association with low numbers of *Pleurosigma* spp. (Figures 3e; f).

Association 3. Blooms dominated by *Pleurosigma* spp. (Figures 3e; f).

Association 4. Mixed blooms of centric and pennate diatoms dominated by *S. costatum* (Figures 3c; d) and *Pleurosigma* spp. (Figures 3e; f).

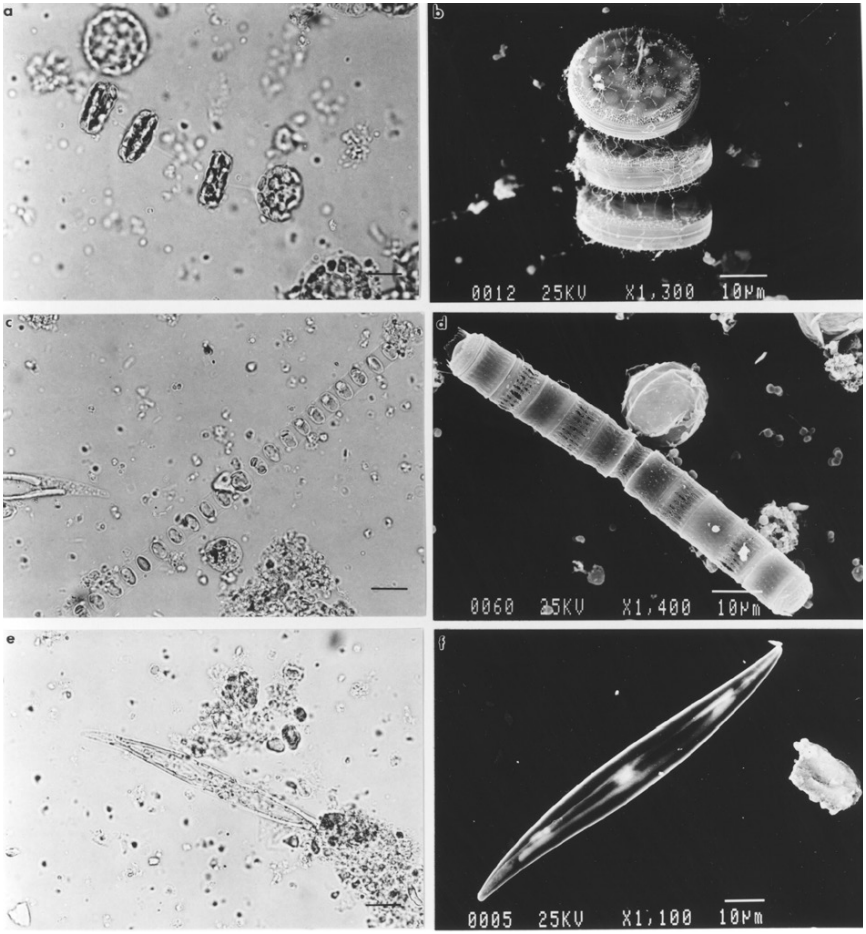


Figure 3. Bright field light micrographs (left) and scanning electron micrographs (right) of common diatoms preserved with glutaraldehyde: **a,b:** *Thalassiosira rotula* characteristic of association 1, from Tingalpa Creek, AMTD site 1.2, 4/7/95. **c,d:** *Skeletonema costatum* characteristic of associations 2, 3 & 4, from Tingalpa Creek, AMTD site 2.8, 17/1/96, **e,f:** *Pleurosigma* spp. characteristic of associations 3 & 4, from Tingalpa Creek, AMTD site 1.2, 15/5/96 site. Note: The identification of *S. costatum* is based on SEM examination of the connecting processes which are trough-shaped with a longitudinal split facing the valve margin. Scale bar on light photomicrographs = 10 µm.

Table 1. Phytoplankton bloom events from some creeks in the southern Moreton Bay catchment during the study period.

Association	Predominant bloom species	Location	Site
Association 1	*Thalassiosira rotula*	Tingalpa Creek	
		4/7/95	1.2
		14/7/95	2.4
Association 2	*Skeletonema costatum*	Tingalpa Creek	
		5/12/95	5.9
		5/12/95	8.5
		17/1/96	5.9
Association 3	*Pleurosigma* spp.	Tingalpa Creek	
		17/1/96	1.2
		15/5/96	1.2
Association 4	*Skeletonema costatum*	Tingalpa Creek	
	Pleurosigma spp.	17/1/96	3.8
		19/2/96	5.9
Association 5	*Skeletonema costatum*	Tingalpa Creek	
	Suriella spp.	19/2/96	7.6
Association 6	cf. *Cyclotella* sp.	Tingalpa Creek	
	Picoplankton	27/6/95	5.9, 7.6, 8.5, 9.6
		14/7/95	8.5, 9.6, 10.4
Association 7	Picoplankton	Tingalpa Creek	
	Cryptomonads	14/7/95	11.0
		17/1/96	3.8
		1/7/96	8.5
		Coolnwynpin Creek	
		22/4/96	0.2
Association 8	*Protoperidinium nudum*	Tingalpa Creek	
		14/7/95	5.9
		17/1/96	10.4
		12/6/96	10.4
		18/6/96	10.4
		1/7/96	11.0
		Hilliards Creek	
		22/2/96	5.0
		Coolnwynpin Creek	
		22/4/96	3.5

Association 5. Blooms dominated by centric (*S. costatum*) diatoms (Figures 3c; d), and pennate (*Surirella* spp.) diatoms (Figure 4a).

Brackish/freshwater environments

Association 6. cf. *Cyclotella* sp. (Figures 4c; d) associated with intense picoplankton blooms (Figures 4c; d).

Association 7. Intense picoplankton in association with abundant crytomonads (Figures 4e; f).

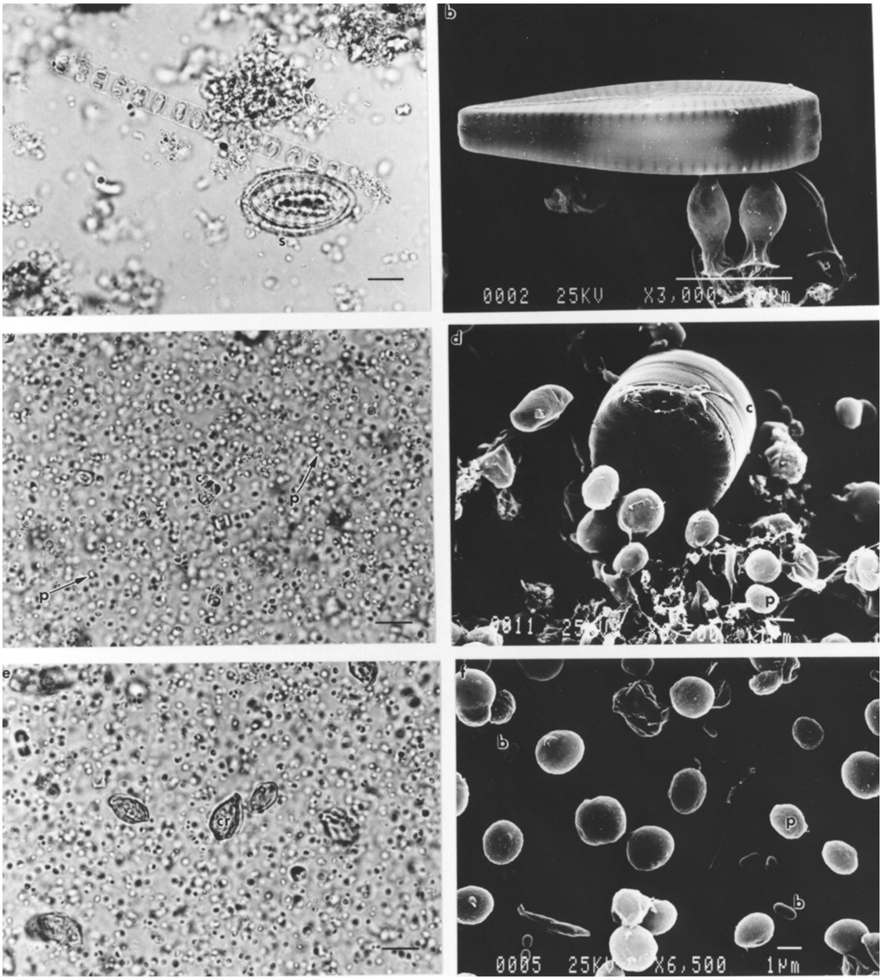


Figure 4. Bright field light micrographs (left) and scanning electron micrographs (right) of common phytoplankton preserved with glutaraldehyde. (a) *Skeletonema costatum* and *Surirella* spp., characteristic of phytoplankton associations, from Tingalpa Creek, AMTD site 5.9, 19/2/96; (b) *Gomphonema* spp.; (c,d) cf. *Cyclotella* sp.; (c) and picoplankton (p) characteristic of association 6, from Tingalpa Creek AMTD site 9.6, 4/7/95. (e,f) picoplankton (p), cryptomonads (cr) and bacterial cells (b) characteristic of association 7, from Tingalpa Creek, AMTD site 11.0, 14/7/95. Scale bars on light micrographs = 10 µm.

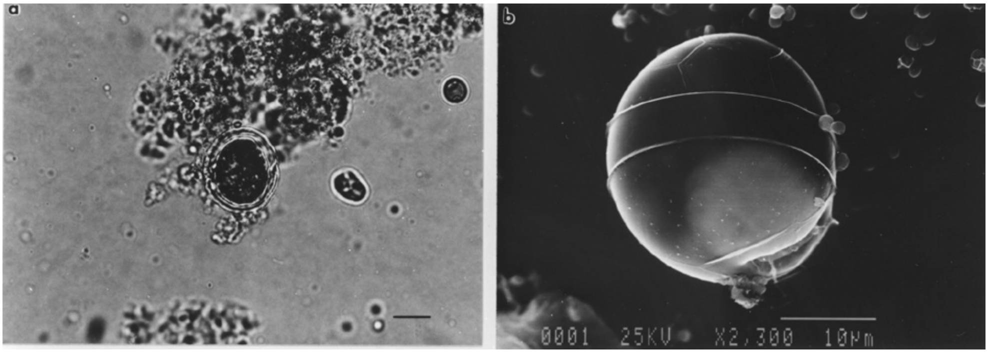
Dinoflagellate blooms

Marine and brackish environments

Association 8. Bloom of *Protoperidinium nudum* (Figures 5a; b).

Cyanobacteria and Green Algae

Edgell's lagoon supports a phytoplankton community dominated by cyanobacteria and green algae (Figure 6). The most common cyanobacteria include *Microcystis* spp. (Figures 6a; b); *Oscillatoria* spp.; *Spirulina* spp. (Figure 6c); and picoplankton. Common green algae include *Pediastrum* spp. (Figures 6e) and *Scenedesmus* spp. (Figure 6f).



Discussion and Conclusions

The study has shown that in order to evaluate phytoplankton community structure and species diversity, it is important to assess the most suitable methods of preserving these microorganisms. Furthermore, it is clear that SEM is an important aid for the identification of those smaller phytoplankton species that cannot be easily resolved by light microscopy.

Several distinct phytoplankton bloom associations were identified in the studied creeks. Regular monitoring over the next few years will enable a baseline data set to be established for future reference and will also determine the seasonal changes in phytoplankton abundance and species diversity. The consistently low yield of phytoplankton species from Tarradarrapin Creek throughout the 1995-1996 period requires further investigation in order to determine why such low numbers of phytoplankton have been recorded in this waterway. Whilst toxic cyanobacteria in particular *Microcystis* are abundant in Edgell's lagoon discharge they do not generally survive the saline conditions of Tingalpa Creek and have not been reported in high enough numbers to pose a threat to livestock and human health to date.

The occurrence of the *P. nudum* bloom is of particular interest because this dinoflagellate has not been previously reported in this region during the preceding five years of seasonal monitoring in Tingalpa Creek (Bowles, 1995; initially reported as *Gymnodinium* sp.). The *P. nudum* bloom followed a large bloom of *Thalassiosira rotula* (4/7/97 and 14/7/97) which suggests that *P. nudum* moved into Tingalpa Creek to feed on *T. rotula*. Other protoperidinioid dinoflagellates are known to selectively prey on colonial diatoms (Taylor, 1987).

Acknowledgements

Michelle Hennessey and Pauline Semple from the Queensland Department of Environment are thanked for supplying water samples for this project. Jane Davissen and Rob Gould are thanked for their help with sample preparation for optical microscopy and SEM. Julian Baker and Gavin Miller kindly reviewed this manuscript. Logistical support from the Centre for Microscopy and Microanalysis, The University of Queensland is appreciated.

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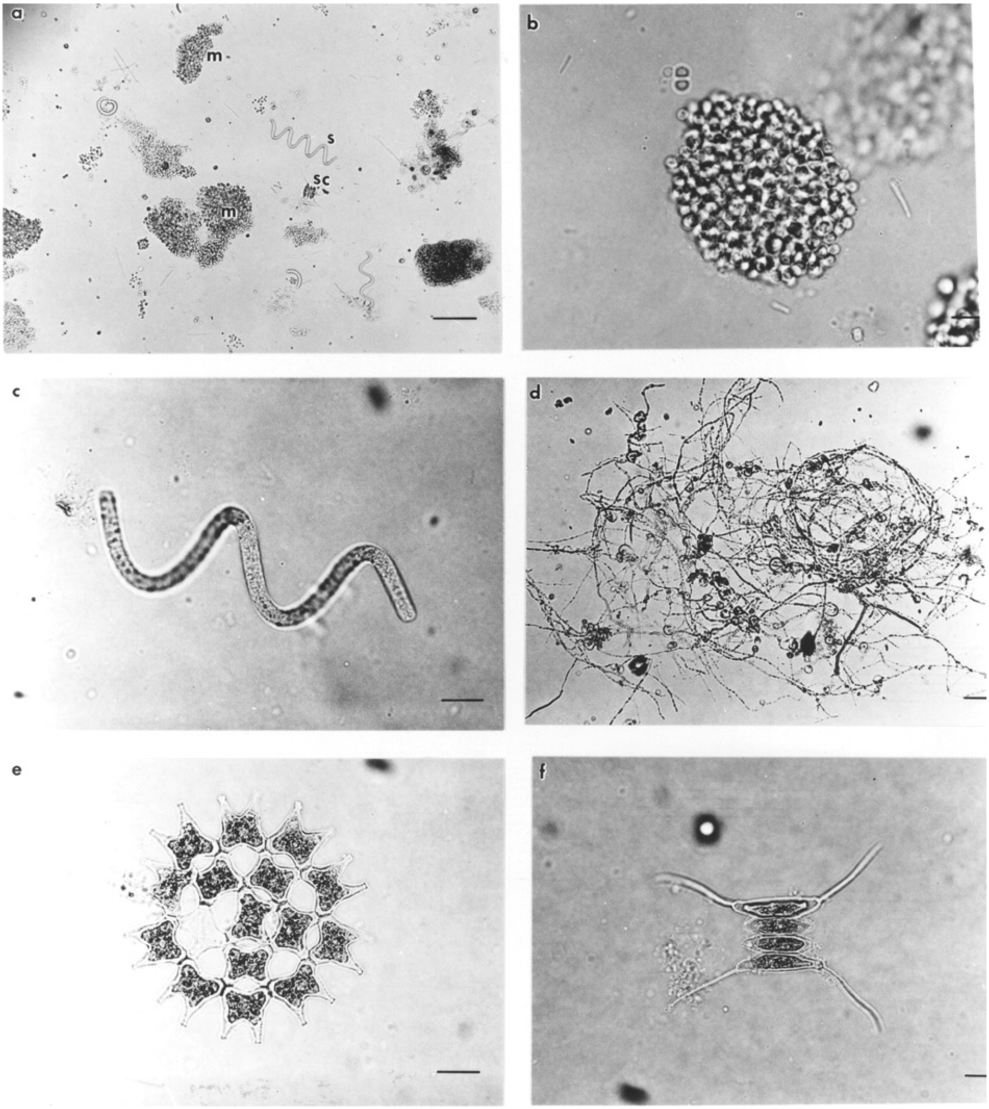

Figure 6. Light photomicrographs of blue green and green algae preserved with glutaraldehyde. (**a**) *Microcystis* spp. (m), *Spirulina* spp. (s) and *Scenedesmus* spp. (sc), scale bar = 5 µm. (**b**) *Microcystis* spp., scale bar = 10 µm. (**c**) *Spirulina* spp., scale bar = 10 µm. (**d**) Filamentous blue green algae from Edgell's lagoon 12/6/96, scale bar = 10 µm. (**e**) *Pediastrum* spp., scale bar = 10 µm. (f) *Scenedesmus* spp., scale bar = 10 µm.

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A Novel Dinoflagellate Bloom in Southeast Queensland: The Importance of Scanning Electron Microscopy for Characterisation and Identification

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Abstract

In order to assess whether point source discharges impact on the biota of Tingalpa Creek, Redland Shire Council, in conjunction with the Queensland Department of Environment, have carried out detailed water quality monitoring surveys in Tingalpa Creek since 1991. In June-July 1995 a dinoflagellate bloom occurred for the first time in the tidal reaches of Tingalpa Creek.

Of particular importance was the identification of the small dinoflagellate responsible for this bloom. Initial observations using optical microscopy revealed similarities between the dinoflagellate in Tingalpa Creek and the known harmful red tide dinoflagellates, *Alexandrium* and *Scrippsiella*. Differentiation of the Tingalpa Creek dinoflagellate from other small photosynthetic and heterotrophic dinoflagellates using optical microscopy was difficult due to their small size, extremely delicate theca and pronounced cell contents which obscure the thecal sutures. The dinoflagellate was therefore identified using scanning electron microscopy as *Protoperidinum nudum* Meunier 1919 (Balech, 1974) based on its small size, protoperidinioid tabulation, presence of three cingular plates and surface ornamentation. Hence this study shows the importance of scanning electron microscopy in the identification of small dinoflagellate species.

Introduction

Systematic baseline monitoring programmes for phytoplankton identification in southeast Queensland are scarce. Little is known of the phytoplankton ecology in this region of Australia mainly because Queensland has not suffered from harmful algal blooms similar to those recorded in the marine environments of Tasmania, South Australia and Victoria (Hallegraeff, 1993). Red tide organisms other than *Trichodesmium* spp. have also not been previously reported in Queensland. However, anecdotal evidence indicates that there may have been an increase in the number of algal blooms in nearshore marine environments in southeast Queensland. The outbreak of a cyanobacterium in Moreton Bay (causing contact dermatitis in swimmers) in March 1995 (J. Phillips, 1995; Queensland Herbarium, pers. comm.), and outbreaks of *Cochlodinium cf helix* in Bramble Bay (A. Moss, 1994, Queensland Department of Environment (QDoE) pers. comm.) may indicate that with the increasing environmental impacts on nearshore marine environments (e.g. eutrophication, overfishing, aquaculture etc.) red tides could occur in Queensland given the right conditions.

Because of the detrimental environmental and economic impacts of red tides in marine and estuarine environments (McLean, 1989; Smayda, 1989; Worth *et al.*, 1975; Anderson & White, 1992; Hallegraeff, 1993), a baseline monitoring programme has been established in conjunction with QDoE in order to document the current phytoplankton ecology of southeast Queensland. This paper describes a dinoflagellate bloom which was identified in water samples collected during a routine water quality monitoring programme by QDoE in Tingalpa Creek, Brisbane. The specific aim of this paper is to describe and illustrate the dinoflagellate responsible for this bloom due to its morphological similarity to other known noxious and toxic dinoflagellates.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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Tingalpa Creek is a small tidal estuary approximately 24 km long in Redland Shire, southeast Queensland which drains into the enclosed embayment of Moreton Bay (Figures 1 & 2). Downstream from the Leslie Harrison Dam (Redland Shire potable water supply) the creek is estuarine with occasional freshwater releases from the dam. Four licensed point source discharges into Tingalpa Creek occur along its length, namely Thorneside sewage treatment plant, Capalaba water treatment plant, Capalaba sewage treatment plant and Edgell's lagoon (Edgell/Birds Eye food processing plant). The estuarine part of the creek drains mostly urban catchment from the Howeston Golf Course, some industry, and flower and turf farms.

In order to assess whether these point source discharges impact on the biota of Tingalpa Creek, Redland Shire Council, in conjunction with QDoE, have carried out detailed water quality monitoring surveys in Tingalpa Creek since 1991. During one four week survey in June-July 1995 a dinoflagellate bloom was identified in the creek. The dinoflagellate responsible for the bloom was present in the plankton in low numbers (1 per mL) during the first and second week of the survey, but reached concentrations of 20 000 cells per mL at AMTD (Adopted Middle Thread Distance – the distance from the mouth of the creek in kilometres) 5.9 by the fourth week (Bowles, 1995). No dinoflagellate blooms had been reported during the previous five years of monitoring in Tingalpa Creek.

Examination of the Tingalpa Creek dinoflagellate using optical microscopy showed this organism to have characteristics of four genera, one heterotrophic *(Protoperidinium* Bergh 1881), and three autotrophic (*Alexandrium* Halim 1960, *Scrippsiella* Balech 1959 ex Loeblich III 1965, and *Ensiculifera* Paulsen 1905). These genera are characterised by their small size

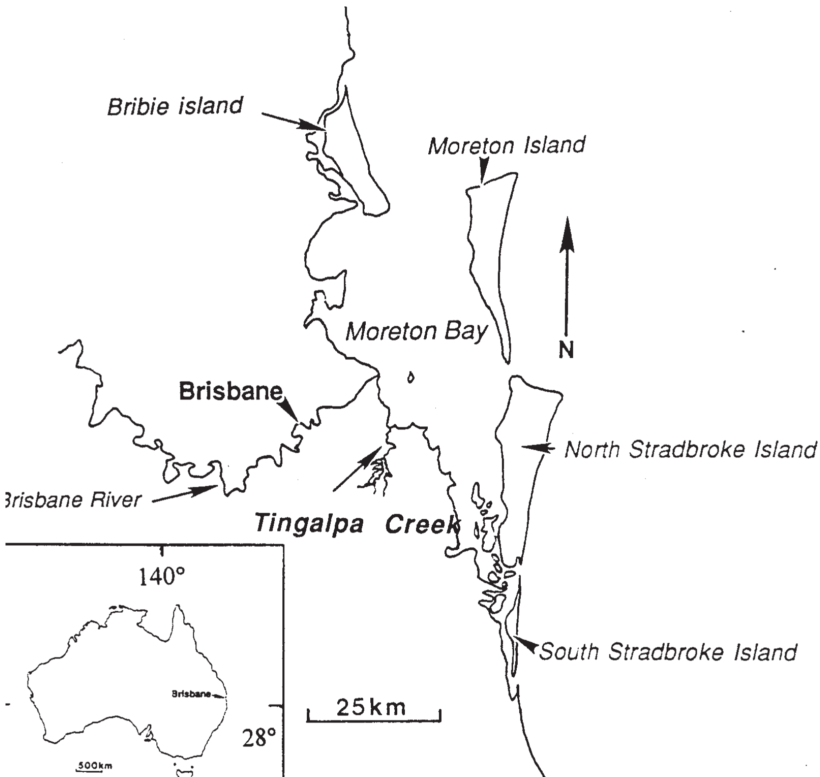

Figure 1. Location map of Tingalpa Creek in southeast Queensland.

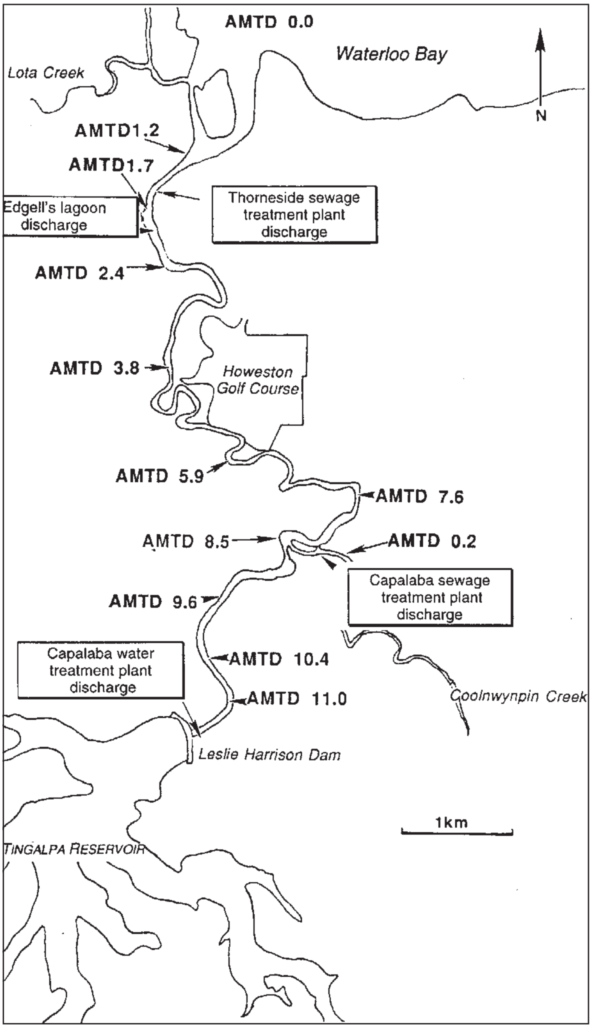

Figure 2. Location map of Queensland Department of Environment sampling sites along Tingalpa Creek.

and extremely delicate theca (Balech, 1974; Dodge, 1981). Given that some *Alexandrium* species cause paralytic shellfish poisoning which is toxic to humans, other mammals, birds and fish, it was important to identify the dinoflagellate responsible for the bloom reported in Tingalpa Creek. Additionally, several *Scrippsiella* species have been implicated in causing fish kills during intense blooms through the generation of anoxic conditions (Hallegraeff, 1991; Larsen *et al*., 1995).

Identification of dinoflagellates to genus and species level cannot always be based on the mode of nutrition given that some species of dinoflagellates can be heterotrophic and autotrophic under different conditions (Wood, 1954). The identification of thecate dinoflagellates requires the determination of the complete plate pattern for critical taxonomy, which can be difficult to perform using optical microscopy on small cells with cell contents that obscure the thecal sutures (Taylor, 1978). Therefore, the identification of small dinoflagellates with pronounced cell contents can, in some instances, be best determined by complete examination of the thecal plate arrangement using scanning electron microscopy (SEM).

Methodology

Collection of samples

Water samples were collected once a week over the four week period 20/6/95-12/7/95. Samples were supplied from three of the samplings on the days 27/6/95, 04/7/95 and 12/7/95. Samples were collected from 11 AMTD sites along the creek (Figure 2) in 500 mL bottles and were maintained at a temperature below 15°C prior to preparation for analysis. The contents of each sample (500 mL) was concentrated within 24 h of collection to 5 mL by centrifugation at 2 300 rpm for 15 min. and then fixed with glutaraldehyde (3% final solution).

Sample preparation for light microscopy

Samples for light microscopy were prepared using standard palynological techniques by dispersion of 0.25 mL in gelatine containing phenol and were examined using an Olympus BH2 optical microscope equipped with Nomarski and phase contrast filters.

Sample preparation for SEM

A small amount of concentrated glutaraldehyde fixed sample (0.25 mL) was pipetted onto plastic coverslips coated with poly-L-lysine and allowed to settle overnight. Samples were washed in a series of three 30 min buffer washes (0.66 M) sodium cacodylate containing 0.15 M sucrose, pH 7.2 at 4°C. These were post-fixed in a mixture of 1% osmium tetroxide in filtered sea water for 90 mins. The samples were washed twice again in buffer at 4°C for 30 mins each, allowed to reach room temperature and then washed in Elgastat high purity water (four changes 30 mins each). The samples were dehydrated through a series of methanol solutions (15, 20, 30, 40, 50, 60, 70, 80, 90, and 100%) for 10 mins each at room temperature, transferred to fresh 100% methanol and dried in a Polaron Critical Point Dryer.

Samples were mounted on SEM stubs using colloidal silver paint, coated with gold in a Biorad S52 sputter coater and imaged using a JEOL 890F Field Emission Scanning Electron Microscope operating between 20 to 25 kV.

Results

Examination of SEM samples showed highly variable preservation of cells, ranging from well preserved specimens to total thecal collapse. Well preserved specimens revealed a thecal plate arrangement characteristic of *Protoperidinium* Bergh 1881. The dinoflagellate was identified as *Protoperidinium nudum* Meunier 1919 (Balech, 1974). *P. nudum* is a small (approx. 25 µm x 25 µm) rounded, *Protoperidinium* species with very delicate thecal plates and no apical or antapical horns. Those recorded in Tingalpa Creek in June-July 1995, however, were smaller than previously described species (20 µm x 23 µm) and showed intraspecific variation ranging from specimens with rounded hypotheca and no antapical horns to specimens with distinct small hollow antapical horns (Figures 3 & 4).

Description of Protoperidinium nudum *Meunier 1919 (Balech 1974) from Tingalpa Creek, Brisbane*

Genus Protoperidinium: Species *P. nudum*

Dimensions: average thecate cell length 20-23 µm, breadth 20-23 µm (10 specimens measured).

P. nudum is a small orthoperidinioid dinoflagellate with extremely delicate thecal plates. The theca is circular to slightly rhombic in dorsoventral view (Figure 3a; b) and circular in apical view (Figure 3c). The theca may show some anterior/posterior compression. The hypotheca is rounded but in some specimens small hollow antapical horns are developed (Figure 3a).

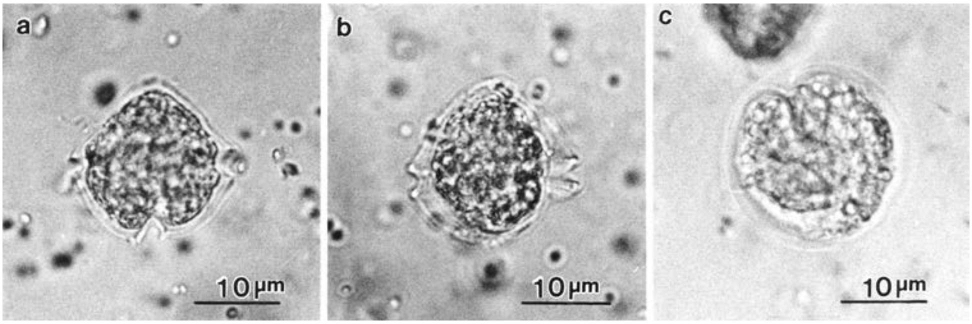

Figure 3. Optical photomicrographs of *Protoperidinum nudum* showing presence (**3a**) and absence (**3b**) of antapical horns and circular apical view (**3c**). Scale bar = 10 µm.

Tabulation: The thecal plate pattern is Po, $4'$, 3a, $7''$, 3c, $5'''$, $2''''$. Plate 1' is four sided (ortho) (Figure 4a). Plate $3'$ is four sided (ortho), plate 2a is six sided (hexa) (Figure 4b). Plate sutures are simple raised ridges (Figure 4c) with or without distinct intercalary bands (Figures 4a; b; e). The intercalary bands show vertical striae in some specimens.

Ornamentation: The thecal plates are smooth to very finely granular with no patterning other than irregularly scattered, trichocyst pores. These pores form a continuous ring approximately 1 µm apart immediately above and below the cingulum in the precingular and postcingular plates. (Figure 4f).

Cingulum: The cingulum is circular, slightly excavated with fine lists approximately 1 µm wide (Figures 4e; f) and not offset at the sulcus (Figures 4a; d).

Sulcus: The sulcus is deeply excavated, broadening posteriorly and does not protrude into the epitheca (Figures 4a; b). The sulcus is bordered by fine lists (Figures 4c; d).

Apical pore plate: The apical pore plate is tube-shaped.

Discussion and Conclusions

Due to the very small size of this dinoflagellate, determination of the tabulation scheme was achieved using SEM, and hence shows the importance of SEM for the identification of small dinoflagellate species. *P. nudum* differs from *Scrippsiella* spp. and *Ensiculifera* spp. in the cingular and sulcal regions only. In *Protoperidinium* there are only three cingular plates and therefore no plate sutures are visible on the cingulum in dorsal view. *Scrippsiella* possesses six cingular plates and four sulcal plates and therefore two plate sutures are visible in the cingular region in dorsal view. *Ensiculifera* possesses five cingular and five sulcal plates, with only one central suture visible on the dorsal surface of the cingulum. *Alexandrium* has an entirely different tabulation pattern to the above genera with no intercalary plates. The tabulation scheme of *Alexandrium* is Po $4'$, 0a, $6''$, 6c, 7-8s, $5'''$, $2''''$. The major differences in thecal plate arrangement of these four genera are shown in Figure 5.

The dinoflagellate, *P. nudum,* responsible for the dinoflagellate bloom in Tingalpa Creek in June-July 1995 is not a known noxious or toxic species. However, the nuisance potential of *P. nudum* blooms in Tingalpa Creek, and the enclosed embayment of Moreton Bay has not been determined to date and requires further investigation. Given the fact that novel phytoplankton blooms of species not previously recorded in an area in bloom proportions are indicative of the early signs of eutrophication (Hallegraeff, 1996), continued seasonal monitoring in Tingalpa Creek with parallel water quality data are warranted.

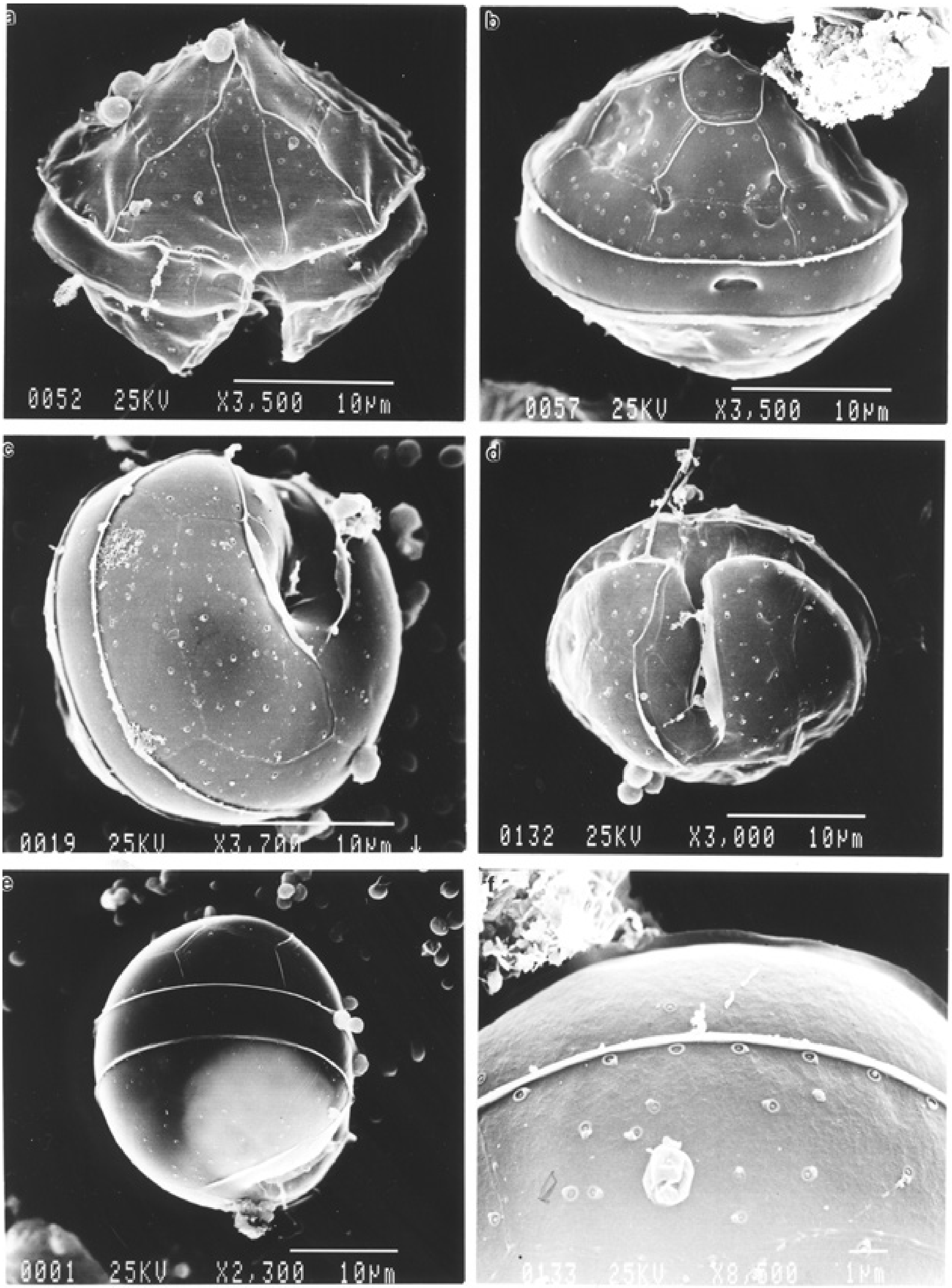

Figure 4. Scanning electron micrographs of *Protoperidinum nudum* in ventral view showing variation in shape and in preservation. (**4a**) shows a moderately well-preserved conical specimen in ventral view with the characteristic 4-sided ortho plate 1'. (**4b**) shows a larger conical specimen in dorsal view with clearly defined plates and intercalary bands. (**4c**) shows antapical view. (**4d**) shows ventral view. (**4e**) shows a well-preserved rounded specimen in side view with clearly defined plates with intercalary bands. (**4f**) shows the trichocyst pores encircling the cingulum.

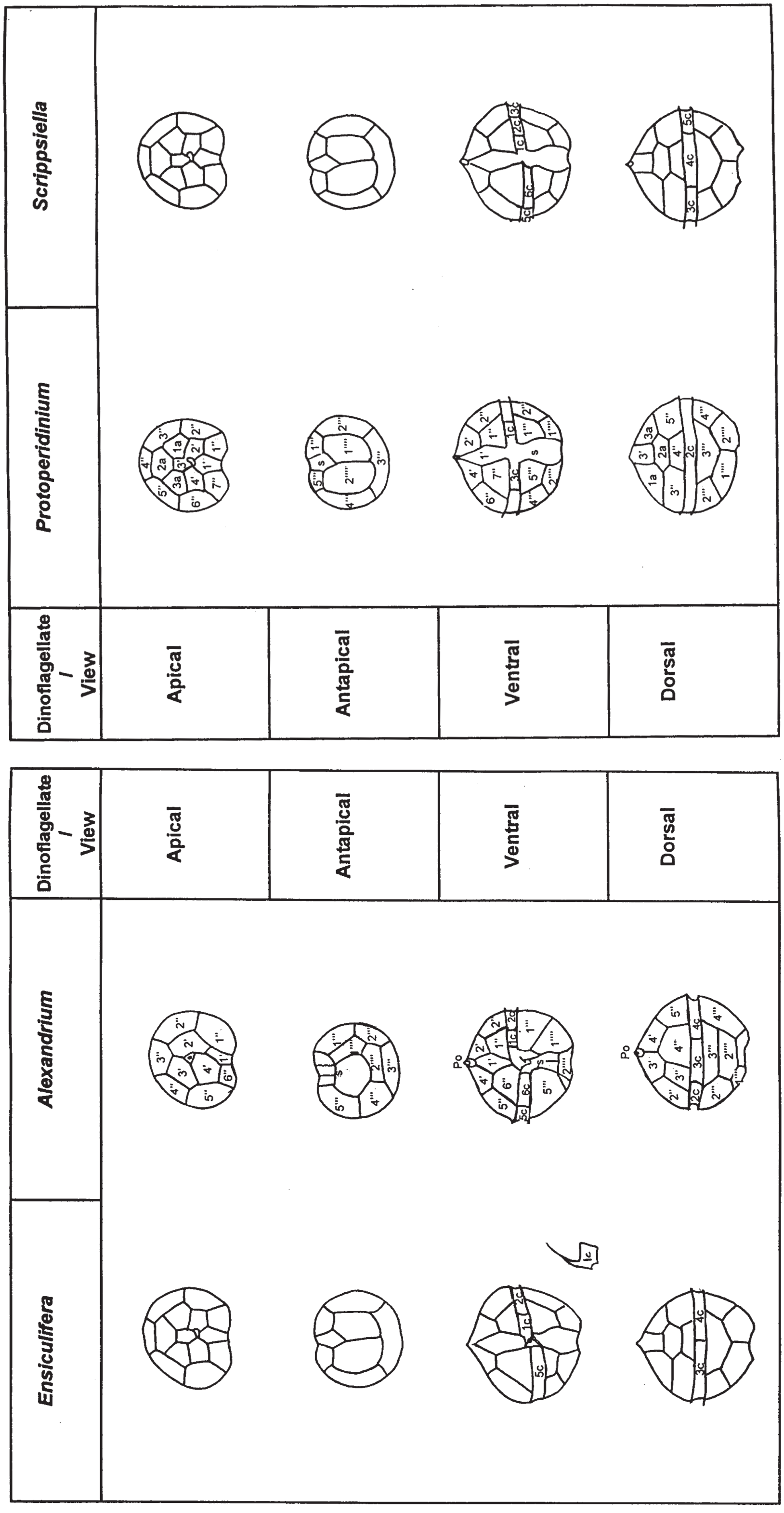

Figure 5. Comparison of the general tabulation patterns of *Protoperidinium*, *Scrippsiella*, *Ensiculifera* and *Alexandrium*.

Acknowledgements

Ms Michelle Hennessey and Mrs Pauline Semple from the Queensland Department of Environment and Heritage are thanked for supplying water samples for this project. Ms Ruth Grinnan, Mrs Anya Yago, Ms Jane Davissen and Mr Rob Gould are thanked for their help with sample preparation for optical microscopy and SEM. Dr Julian Baker and Mr Gavin Miller kindly reviewed this paper. Logistical support from the Centre for Microscopy and Microanalysis, The University of Queensland is greatly appreciated.

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Are Expanding Populations of the Tropical Green Alga *Caulerpa taxifolia* a Potential Threat for Moreton Bay?

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In 1984 a low temperature resistant and vigorous strain of *C. taxifolia* was introduced into the Mediterranean Sea and has since proliferated along the coasts of France, Spain and Italy (Meinesz *et al.*, 1993). From a single 1 m² patch observed off Monaco in 1984, *C. taxifolia* spread rapidly to occupy an area of > 3 000 ha by 1994. Such an unprecedented increase in abundance has never been described for this species in tropical endemic regions. In fact, *C. taxifolia* has shown a surprising ability to colonise diverse habitats within the Mediterranean Sea, including all types of substrata, all water depths from zero to greater than 90 m, both sheltered and exposed environments and either polluted or clean waters. *C. taxifolia* produces toxic mono- and sesquiterpenoids such as Caulerpenyne and other potentially toxic terpenes. These secondary metabolites act as antifouling substances for the plant and possess a wide range of toxicological activities. Crude extracts of *C. taxifolia* show antibacterial, antiviral, cytotoxic and antimitotic action spectra, which effectively prevent grazing by a wide range of macrozoobenthic organisms and demersal fish. Invasion of *C. taxifolia* leads to a drastic impoverishment of algal and seagrass communities and causes substantial changes in the fauna associated with the original vegetation. In general, a decline in bio- and ecodiversity can be noted in Mediterranean littoral ecosystems invaded by *C. taxifolia* (Boudouresque *et al.*, 1996).

C. taxifolia is distributed world-wide in tropical seas but is absent in adjacent temperate areas where the winter temperature drops below 20°C (Meinesz *et al.*, 1994). In Australia, *C. taxifolia* has been recorded from the north and northeast coast and the Great Barrier Reef region. Since 1933, records are available on its occurrence at Lord Howe Island and in 1946 the southern limit of its repartition was recorded to be ~400 km south of the Great Barrier Reef, in Moreton Bay (Cribb, 1958). Winter sea surface temperatures in Moreton Bay range from 16.0°C ± 0.25 (SE) in the western Bay to 17.3°C ± 0.16 (SE) in the eastern Bay with a minimum of 14.7°C in August (Preen, 1992). These winter temperatures are the lowest known for a natural population of *C. taxifolia* and are comparable to the winter temperatures found in the Mediterranean Sea. The observations of *C. taxifolia* in Moreton Bay since 1946 have been sporadic and no data are available on distribution and abundance of *C. taxifolia* from these early records. In 1990 it was observed for the first time that the Moreton Bay population of *C. taxifolia* was rapidly expanding and competing with indigenous seagrass communities (Preen, 1992).

This paper represents the first record of an expanding dense population of *C. taxifolia* located at One Mile Jetty, Dunwich, North Stradbroke Island (27° 29' 35'' S; 153° 24' 16'' E). The *C. taxifolia* meadow was situated at the southern edge of the Harbour Channel, 100 m from the shoreline. It extended over a distance of 30 m from 0.5-2.5 m depths at low tide. *C. taxifolia* was growing within a monospecific stand of the seagrass *Halophila spinulosa*. The total area covered by *C. taxifolia* was approximately 500 m^2. In dense patches of *C. taxifolia* the number of primary fronds reached 4 183 ± 1 370 phylloids per m^2 (± S.D.). The mean phylloid length

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 327-328.

(± S.D.) was 6.0 ± 3.09 cm (with maximal length of 18.0 cm). The leaf area index (LAI in m^2) was 3.7 ± 0.73 and biomass values reached 86 ± 7.2 g dw/m^2. The biomass of *C. taxifolia* in Moreton Bay was 50 times higher than in typical tropical regions (Garrigue, 1994), but was as high as in the Mediterranean Sea (Pillen, 1995), where densely growing populations reach upper biomass values of 480 to 700 g dw/m^2 (Meinesz *et al*., 1995). Additionally, growth form and abundance of the stock investigated were similar to those characteristic of rapidly proliferating *C. taxifolia* in the Mediterranean Sea. Our data and the observation of invasion and overgrowth of seagrass beds by *C. taxifolia*, recently confirmed at several sites in Moreton Bay (Dennison & Horrocks, pers. comm.), suggest that *C. taxifolia* from Moreton Bay possesses a competitive potential and might become a threat to seagrass ecosystems. It is, however, not known if the observed and measured expansion of *C. taxifolia* are due to natural cycles or responses to anthropogenic disturbances and further research is required.

Seagrass beds in Moreton Bay are of great importance for coastal ecosystems serving as nursery, refuge and food supply for a wide range of organisms. In addition, the dugong (*Dugong dugon*) is dependent on seagrass, which is the main item in its diet. Observations suggest that dugongs strictly avoid seagrass areas in which *C. taxifolia* is found (Preen, 1992). This implies that *D. dugon* would be directly affected by further expansion of *C. taxifolia* in the shallow waters of Moreton Bay.

Overall, field observations and measurements of a single *C. taxifolia* meadow in Moreton Bay indicated that this alga has the potential to increase its extent at the expense of seagrass beds. Further proliferation of *C. taxifolia* could therefore affect Moreton Bay fisheries, dugong and sea turtle populations. The reasons for the recent expansion of *C. taxifolia* (e.g. natural cycles or response to anthropogenic-mediated disturbance) are still unknown, thus we strongly recommend that *C. taxifolia* be monitored closely.

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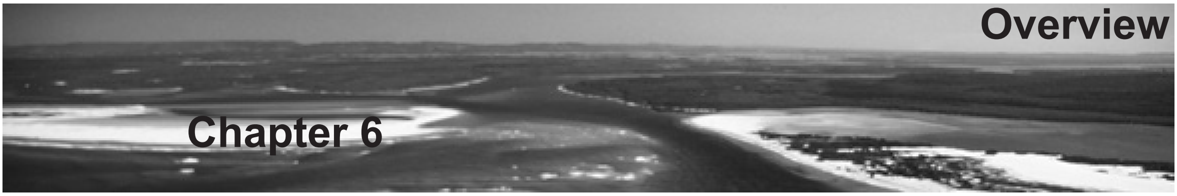

Overview

Chapter 6

Marine Fauna

The following chapter covers key areas of the marine fauna of Moreton Bay, and focuses on interactions with the habitats discussed in the previous section. While the division of ecological papers into two sections is somewhat arbitrary, essentially the previous section was about primary producers whilst this section is about consumers of that production.

The four review papers cover the following topics: biodiversity and biogeography, zooplankton, benthic invertebrates, and nekton (fish and prawns). Authors were requested to summarise what is known about their topic, including interactions with habitat and any effects riverine inputs might have on the animals, and to state research priorities.

Our level of knowledge differs markedly from group to group. Information about economically important fish and crustacean species is generally quite detailed, and usually if the organism is of economic significance or has a high public profile then its basic biology is known. On the other hand, even the identification and taxonomy of most less conspicuous faunal groups, let alone their biology and ecology, remains in a limited state. Ecological research into some major groups of crustaceans and into the very small invertebrates, for example meiofauna, is being hampered because workers have no published guides with which to identify them.

The paper reviewing our knowledge of zooplankton covers invertebrates of all sizes that spend time drifting in the sea. This includes not only the animals that undergo their entire life cycle in the water column, but also larval stages of fish and crustaceans, the adults of which are discussed in the other papers. The nekton review is about fish and prawns, and is overtly aimed at reviewing ecological works, without concentrating on fisheries *per se*. The discussion of benthic invertebrates covers invertebrates living in or on the sediment. Some of these invertebrates emerge into the water column at some stage, so there is overlap with both the zooplankton and nekton. Crabs are considered in the paper on benthos, while prawns are covered under nekton. The next paper is about both the biodiversity and biogeography of fish and invertebrates.

The overall picture is one of scientists having a good grasp of pattern (where various types of animals occur) but very little idea about process (what makes the pattern, i.e. the dynamics). In general our understanding of underlying ecology of animals seems less advanced than that of the ecology of plants, particularly the links between nutrients, sediments, and seagrass growth.

Some topics are not covered specifically in these review papers. The fisheries of Moreton Bay have a very high public profile and contribute a disproportionately high fraction of Queensland's total marine harvest (around 50% of the commercial catch of several major species including whiting, bream, mullet, tailor, flathead, garfish, squid and sand crabs): these have been well reviewed elsewhere (Quinn 1993; Williams 1992) and are mentioned here only in an ecological context. Moreton Bay is also an extremely important region for birds. A considerable amount is known about the seabirds and the migratory wading birds that use the Bay (e.g. Blaber & Wassenberg, 1989; Driscoll, 1993), but mention is made here only of their role as predators of benthic invertebrates. Humpback whales enter Moreton Bay occasionally, and spotting them from Point Lookout on North Stradbroke Island as they migrate between the Southern

Ocean and north Queensland waters has become a very popular pastime. They are presumed not to be of major importance in the ecology of the Bay, however, and are also not discussed in detail in this volume. Although each paper covers the effects of human activities on the faunal group under review, the effects of some activities are wide-ranging and, given the extent to which they trouble the minds of managers, might not get appropriate coverage. An example is the modification of intertidal wetland habitats (mangroves and saltmarshes) aimed at controlling insect pests.

Many exciting topics are covered in the non-review papers and, together with the review papers, give a wide coverage of the ecology of the Bay fauna. Topics include two papers on cumaceans, and, on vertebrates: fish larvae of Pumicestone Passage, fish that live in rock pools, a re-examination of the importance of saltmarshes as fish habitat, diet of juvenile whiting (they do eat small greasyback prawns, posing an interesting dilemma for any attempt to artificially increase whiting stocks by seeding), measurement of dispersal of pipefish using otolith microchemistry, ecology of hardyheads, small fish that feed in the water column, humpback dolphin biology and the bottlenose dolphins of Tangalooma on Moreton Island.

Two key issues remain unresolved, and reflect the issues of the greater Bay area, viz.: are fish stocks declining and what proportion of a fish stock should be allocated to commercial rather than recreational fishers? In the closing session of the conference from which this book originated, it was clear that most members of the public present considered that fish were harder to catch now than they were decades ago, and are probably smaller too. This contrasts with the evidence that very large amounts of fish, crustaceans and squid are taken commercially and recreationally from the Bay. The suggestion made at the conference by Col Limpus that the very high numbers of large grazing vertebrates (turtles and dugongs) might indicate a highly productive Bay, is also germane. This brings us to consider just what we hope to achieve in managing human activities in the Moreton Bay catchment. Assuming the Bay has suffered over recent decades from increased eutrophication (an oversupply of nutrients), would a reduction in nutrient input offer us a Bay of plenty or, in fact, a more pristine Bay with less harvestable stock?

The shift towards considering not only the Bay but also its catchment as an integrated system, presents us with opportunities to focus attention on the effects on fauna of changes in the quality or quantity of water arriving from rivers into the Bay. There is little mention of such effects in this volume – an indication of lack of study on these topics. Whilst many effects of riverine input on fauna might be expected to act indirectly via habitat changes, research is also needed into direct links between river quality and Bay fauna, given that many species spend part of their life in both rivers and the Bay.

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Patterns of Biodiversity in Marine Invertebrate and Fish Communities of Moreton Bay

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Abstract

Moreton Bay supports a high faunal diversity of enormous taxonomic and biogeographic interest. It is a meeting point of southern (Peronian) and northern (Solanderian) faunas, and represents a peculiar mix of both temperate and tropical species. Over 3 000 species are recorded from the Bay, although no formal checklist has yet been prepared. To look for patterns in biodiversity, Queensland Museum collection records of fish and invertebrates were collated using five minute squares of latitude and longitude. As a generality there are two major biodiverse regions – an inshore estuarine dominated region, and an eastern marine dominated region. High species numbers occur around the mouth of the Brisbane River and along the western shores of the Bay and adjacent waters, gradually diminishing to the north and east. This is indicative of an important estuarine influence from the many rivers and creeks that feed into the Bay. The clean labile sands of the northern channels are extremely species poor. A second and higher centre of diversity occurs around the northern end of Stradbroke Island, and includes Myora, Peel, Bird and Goat Islands. This region has well developed coral reefs and a mix of consolidated hard and muddy-sand bottoms. There is a shift in the composition of this fauna to include a large number of cleaner water, reefal species. Sponges appear to have restricted local distributions and a very high species endemicity within the Bay. Between adjacent sites there is about 20% overlap in species, but only 6% or less of the Bay species are shared with the outside reefs.

Introduction

Moreton Bay has been a rich source of marine biological resources since pre-European times although over recent decades it has come under increasing environmental pressure with the rapid growth of greater Brisbane. Williams (1992) estimated about 1 000 full-time fishers utilise the Moreton Bay region, taking about 10% by value of the catch of the entire east coast of Australia. He also found approximately 200 prawn trawlers operate in the area taking 10% of the Queensland trawl catch, and 41% by weight of the total sea-food production from Moreton Bay. Although the Bay is suffering sustained pressure, it remains a remarkable system with consistent high diversity and biomass of marine fauna. Conservation of biodiversity is obviously a major priority for the continued healthy functioning of communities, but marine biodiversity issues have not received the attention currently given to terrestrial systems, perhaps because they are less easily studied, impacts are less conspicuous, and taxonomic difficulties are immense. This present study examines the marine invertebrate and fish resources of the Bay, as currently known, including patterns of biodiversity, biogeographical relationships, and implications for potential management strategies.

In this paper, the terms "diversity" and "biodiversity" refer only to species richness, and are not used in an ecological sense which typically encompasses an abundance measure of species populations to derive a synthetic diversity measure (see Clifford & Stephenson, 1975).

Both the faunal composition and the underlying ecological processes within Moreton Bay have received considerable scientific attention. Extensive studies have been made of the soft

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 331-346.

bottom macrobenthos of Moreton Bay (e.g. Stephenson *et al.,* 1970; 1974; 1976; 1977; 1978; 1979; Raphael, 1974; Stephenson & Cook, 1977; 1979; Poiner, 1979; Stephenson, 1980a; b; c; Stephenson & Sadacharan, 1983). Trawl studies were conducted by Jones (1973), Stephenson & Dredge (1976), Quinn (1978), Burgess (1980), Stephenson & Burgess (1980) and Stephenson *et al.* (1982a; b). Investigations on the dredged macrobenthos near the mouth of the Brisbane River were reported by Hailstone (1972; 1976), Boesch (1975); and Park (1979). Campbell *et al*. (1974) sampled the fauna of nine estuaries in southeastern Queensland, most of which feed into Moreton Bay, and Stephenson & Campbell (1977), Campbell *et al.* (1977a) and Davie (1986) studied the sublittoral fauna of Serpentine and Jacksons Creeks. Young & Wadley (1979) examined the distribution of shallow water epibenthic macrofauna in the Bay. Intertidal faunal studies were conducted by Vohra (1965) and Catterall & Poiner (1984) on seagrass communities of North Stradbroke Island; Campbell *et al.* (1977b) on the mangrove communities of the Serpentine and Jackson's Creeks; and Stejskal (1984) and Stejskal & Chamberlain (1984) surveyed the intertidal macrofauna of Bramble Bay.

There are many gaps in our knowledge of the taxonomic inventory of the Bay. Some of the more significant studies are listed below, but also see Davie (1990) for a checklist and bibliography of the fauna of the Brisbane River estuary which shares many marine species in common with the Bay.

Annelida: Russell (1962); Rullier (1965); Hutchings (1977).

Ascidians: Kott (1972; 1985; 1990; 1992).

Brachiopods: Emig (1979).

Cnidaria: Pennycuik (1959); Tixier-Durivault (1966); Bayer (1981).

Corals: Wells (1955); Veron (1974); Veron & Pichon (1976; 1977; 1979; 1982); Wallace (1978); Lovell (1989); Fisheries Research Consultants (1993).

Crustaceans: copepods (Thwin, 1972; Greenwood, 1974; 1976; 1977; 1978; 1979, cumaceans (Tafe & Greenwood, 1996a; b), stomatopods (Stephenson, 1952; 1953a; b; 1960; Stephenson & McNeill, 1955), portunid crabs (Stephenson & Hudson, 1957; Stephenson, Hudson & Campbell, 1957; Stephenson & Campbell, 1959; 1960; Stephenson, 1961; Rees & Stephenson, 1966), penaeid prawns (Racek, 1955; Dall, 1957; Racek & Dall, 1965), crabs (Campbell & Stephenson, 1970). Wadley (1978) prepared a checklist of 40 species of epibenthic shrimps from shallow water and intertidal soft substrates of Moreton Bay; and provided an illustrated key to carideans and sergestids. While this is a useful compilation, it is not a complete guide to the fauna.

Sponges: Hooper (1991; 1996), Hooper & Lévi (1993a; b; 1994).

Methods

Queensland Museum databases were interrogated for all records of marine invertebrates and fishes collected from the greater Moreton Bay area, defined here as the area contained by 26°55'-28°00'S, and 153°00'-153°30'E, and additional five-minute squares encompassing Flat Rock to the east and Caloundra to the north.

Museum collections have been accrued over many decades and comprise of: (1) incidental collections; (2) Museum field work; (3) survey collections from research and environmental assessment projects. Database records linked to preserved specimens ensure that identifications are verifiable, and enable us to provide an objective scientific record of the fauna.

Interrogation of the computer databases provided a total number of taxa for the Bay. Many older records that were simply recorded as "Moreton Bay" were given a generic point source which enabled them to be excluded from the geographical analysis, but they were included in the total species count.

Sampling effort has an obvious influence on the number of species found. A rough measure of sampling effort for crustaceans and fish was established by calculating the number of separate collecting dates i.e. visits, within each sampling square. While this does not indicate the intensity of sampling for any one visit, it does reflect the overall attention that has been paid, over time, to a particular area.

Results and Discussion

Biodiversity patterns

Faunal richness and composition

A complete faunal list for the Bay has not yet been compiled, and there are still enormous taxonomic problems within many groups, with many new taxa still to be described. According to Queensland Museum invertebrate databases over 3 000 species of free-living marine macro-invertebrates occur in Moreton Bay (Table 1). Estimates of mollusc species have been obtained from an unpublished list of gastropods being compiled by Mr Terry Carlis of the Malacological Association of Australia (Queensland Branch), and from Lamprell (1990; 1998, pers. comm.) for bivalves. Conspicuously missing is any estimate of the poorly known interstitial fauna (but see Faubel *et al.,* 1994). Similarly, the parasite communities have not been included.

The estimate of fish species richness is accurate as it is based on a checklist soon to be published (Johnson, unpubl. data). However, species counts for the invertebrate phyla may

Table 1. Species counts of marine fish and invertebrates for Moreton Bay.

Fish	713	Crustacea	852	Mollusca	1325	Annelidata	279
Porifera	148	Cnidaria	240	Bryozoa	57	Chordata	92
Phoronida	4	Brachiopoda	4	Echinodermata	224		
Total (invertebrate species)	**3 225**						

be overestimated as identifications are in many instances unverified, and the same taxon may occur in the collection under several synonymous names. Also different survey collections may overlap in the species recorded but use different names. To counter this potential overestimation is the fact that there are literature records for which no Museum voucher exists, and these have not been included in the totals for the present estimates.

In general, there is a high correlation between sampling effort and high species diversities. This is explainable in two ways: (1) the richest areas with the most diverse habitat types were probably preferentially targeted by collectors looking specifically for high return on their collecting effort; and (2) the more visits to a particular area will increase the probability of encountering rare species, and seasonal variation in species composition. To temper this correlation, however, are the results of the systematic dredge survey of the macrobenthos of the Bay by Stephenson *et al.* (1970) who found roughly similar patterns of biodiversity to those reported here.

Area patterns and community composition

The major pattern to emerge shows high species numbers occurring around the mouth of the Brisbane River and along the western shores of the Bay, gradually diminishing to the north and east. This corresponds to the "Seasonal Estuary" of Dennison *et al.* (this volume). There is also a marine dominated zone comprising two distinct centres of high diversity – one around the northern end of Stradbroke Island, including Peel, Bird and Goat Islands and Myora, where there is a shift to consolidated bottoms and reefal species; and a second area around Middle Banks and Tangalooma. The clean labile sands of the northern and eastern openings are extremely species poor.

A similar conclusion was reached by Stephenson *et al.* (1970) who collected 355 species from 400 dredged stations. Richest sites occurred near the edges of the Bay which they attributed to favourable hydrographic-sedimentary conditions and/or to the location of these sites on the periphery of areas worked by prawn trawlers.

Similarly, in a two-year study of sand and seagrass communities of the Sholl Bank, northeastern Moreton Bay, Poiner (1979) found that the biota of the sand communities was relatively depauperate, in a continual state of flux, and showed no evidence of a stable climax community. In contrast, seagrass communities were species-rich and relatively stable. Winter-wave stress (produced by persistent winterly trade winds) was suggested as an important disrupting factor, particularly at the shallow sites.

Inshore communities

High species numbers occur around the mouth of the Brisbane River and along the western shores of the Bay and adjacent waters, gradually diminishing to the north and east. This is indicative of an important estuarine influence from the four major rivers and numerous creeks that feed into the Bay – presumably providing an overlay of nutrient rich sediments and a variety of sediment particle sizes which would support a variety of feeding types. Annual or unpredictable flood events would also add an element of instability to the fauna of this region. Regular nutrient enrichment combined with low level disturbance could stimulate recruitment and encourage high diversity.

Sediment characteristics have long been recognised as presenting a complex of limiting values influencing the distribution of benthic fauna (e.g. Kay & Knights, 1975; Coleman & Cuff, 1980). Coarser sediments with a greater range of grain sizes should create a large number of potential niches (Gray, 1974). Stephens (1992) calculated that the Brisbane River supplies mud to Moreton Bay at a minimum rate of 175 000 tonnes/yr. Ruello (1973) noted the importance of seasonal freshwater runoff through mangrove and reed swamps into the Hunter River estuary where the enormous quantities of nutrients and plant detritus carried by the runoff played a particularly important role in the nutrition of juvenile school prawns (*Metapenaeus macleayi*). Similarly, Copeland (1966) observed increased productivity in estuarine and inshore oceanic waters as a result of the influx of vitamins and nutrients following heavy rains, and he emphasised the adverse effects of drought and decreased river flow on estuarine ecology.

Reef communities

The region around the northern end of Stradbroke Island, including Peel, Bird, and Goat Islands and Myora, has well developed fringing coral reefs and a mix of consolidated hard and muddy-sand substrates. There is a shift in the composition of this fauna to include a large number of cleaner water reefal species. Recruitment to these areas is apparently sustained by the influx of clean coastal waters through the passage between Moreton and Stadbroke Islands, and directed south by the Rainbow Channel. Many species found here are considered southern outliers of a tropical Great Barrier Reef assemblage.

The small reef at Myora is particularly diverse with high numbers of molluscs and crustaceans. The Myora coral community occurs as a shallow (1-3 m) narrow reef, with six coral species (Lovell, 1989). It is unusual within Moreton Bay in having large plates of *Acropora digitifera* (Dana). These grow attached to loose rubble on a muddy-sand substrate, which means they are easily overturned to collect the associated fauna. This small reef has a high diversity of invertebrates, including species which are more typically expected on outside reefs, for example the rare shrimp, *Phyllognathia ceratophthalma* Balss (see Davie, 1992), and coral associated crabs such as a unique record of *Cryptochirus coralliodytes* in the Queensland Museum collection, and several species of trapeziids. The position of the reef just to the side of the main channel leading in and out of the Bay means recruitment from open coastal waters, and the warm southerly flowing Eastern Australian Current can be expected.

This central-eastern region also shows a marked resistance to flood dilution and sedimentation. Stephenson (1968), reporting on the effects of a flood in March 1963, stated

> *Three areas in the region investigated had reasonably saline bottom waters with high water salinities of 33 p.p.t. or greater at roughly the period of greatest flood effects ... the northern opening (South Passage) maintained tolerably high salinities for approx. 10 miles southwards but for a much smaller distance (approx. 3 miles) to the west. The more southern opening at Jumpinpin permitted only a slight intrusion of more saline waters. The third area lies to the NNW of Peel Island.*

Patterson & Witt (1992) have modelled the hydraulic processes in the Bay. Their maps of tidal current vectors clearly show the interaction of incoming northern and eastern tidal flood currents causing an anticlockwise pooling of seawater to the NNW of Peel Island. On the ebb tide this is maintained but with a clockwise current flow. In the words of Stephenson (1968)

> *This area and that to the south-east of it, are of great faunistic interest. They contain the greatest variety of live corals in Moreton Bay and the greatest densities of corals, which form an ill-developed fringing reef. ... It seems likely that the special occurrence of corals at Peel Island is due to their normally escaping flood outflows from the two rivers. To the north of Peel Island occur some of the richest dredging grounds in Moreton Bay. Several of the more interesting species otherwise only occur in the Bay near its north-eastern entrance ... including the cephalochordate* Branchiostoma moretonensis *Kelly ... the solitary coral* Cycloseris cyclolites *(Lamarck) ... and the very large arenaceous foramniferan* Discobotellina biperforata *Collins [see Stephenson & Rees, 1965a; b].*

Patterns for different ecological faunal groupings

Underlying the generalised faunal pattern are some important variations shown by particular groups of taxa which comprise broad ecological units, e.g. a predominantly sessile encrusting assemblage (cnidarians, sponges, bryozoa, tunicates etc.); a mobile benthic infaunal and epifaunal group (crustaceans, annelids, echinoderms etc.); and a nektonic group (fish).

The centre of diversity of sessile species is strongly centred in the marine zone around the reefs adjacent to North Stradbroke where there is less chance of being smothered by seasonal sediment loads from the rivers (Figure 1). Conversely, the typically infaunal or mobile epibenthic species of crustaceans, echinoderms and annelids have a strong presence in the inshore and riverine sediments, and also around Middle Banks and Tangalooma (Figure 2). Fish distribution is similar in showing an estuarine/marine dichotomy, but of particular interest is the very high diversity on the inshore of Moreton Island, which coincides with the wrecks and artificial reefs at Tangalooma, Cowan, and Bulwer (Figure 3). Another fish diversity hotspot, not obvious for the invertebrates, is centered on the South Passage around Amity Point.

Seasonal patterns

High rainfall with concurrent reduced salinity has been proposed as a causal factor of faunal seasonality in a number of local studies, i.e. Park (1979) working on macrobenthos at the mouth of the Brisbane River; Vohra (1965) and Stejskal (1984) working on intertidal fauna at Victoria Point and Cribb Island respectively; and Young & Wadley (1979) studying epibenthic fauna in Moreton Bay. The Brisbane River has an average annual streamflow of about 1.35 million megalitres (Cossins, 1990), but this is not constant with very low flow periods interspersed with short flood events (Odd & Baxter, 1981).

Studies by Stephenson (1980a;b;c) indicate major recruitment to the Bay's benthic communities in August/September. Stephenson (1980b) found prawn catches to be significantly correlated with raised temperatures and rainfall of seven months previously. Major depletion of benthic stocks occurred in December and was attributed to increases in mobile predators and benthic-disturbers (fish and prawns) (Stephenson *et al.,* 1978; 1982). Presumably species diversity would also show such seasonal fluctuations. The occurrence and abundance of rarer species would certainly be related to periods of high and low flux levels. Predictability of benthic species' occurrence is also a complex issue. Stephenson (1980c) found that whilst some species showed annual cycles, more species showed significant long term cycles of between 2-7 years, with evidence of successional replacement of species' groups. The scale of this type of species replacement is unknown – it is probably a relatively local phenomenon, but it is also possible

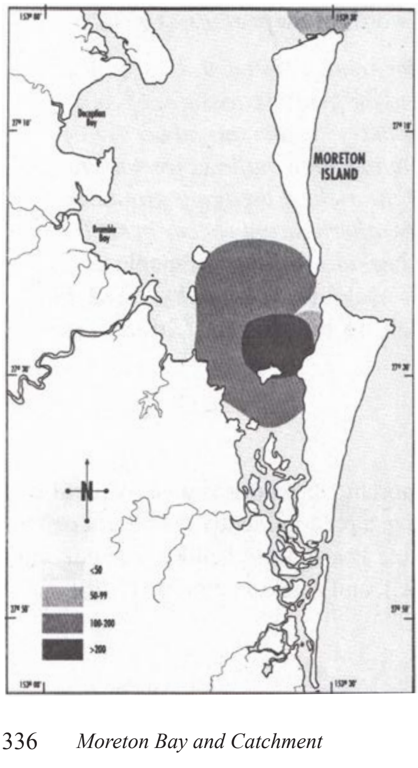

Figure 1.
Distribution of species in predominantly sessile phyla. The centre of diversity is strongly centred in the marine zone around the reefs adjacent to North Stradbroke where there is less chance of being smothered by seasonal sediment loads from the rivers. Source: Queensland Museum database.

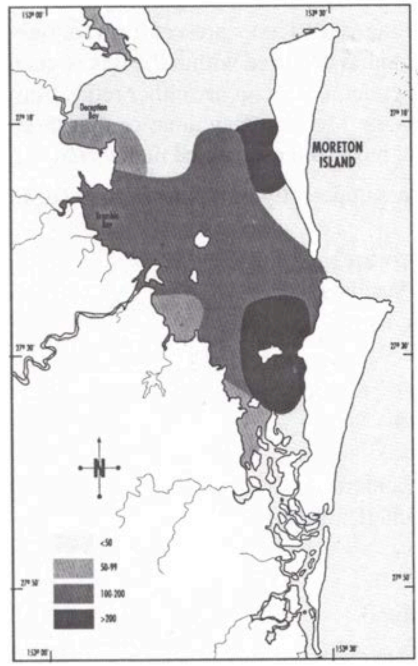

Figure 2.
Distribution of the typically infaunal or mobile epibenthic species of crustaceans, echinoderms and annelids. This group have a strong presence in the inshore and riverine sediments, and also around middle Banks and Tangalooma. Source: Queensland Museum Database.

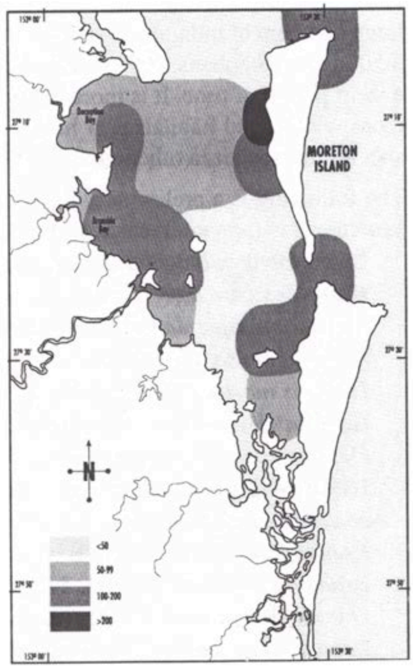

Figure 3.
Distribution patterns of fish. This shows an estuarine/marine dichotomy, but of particular interest is the very high diversity on the inshore of Moreton Island, which coincides with the wrecks and artificial reefs at Tangalooma, Cowan, and Bulwer. Another fish diversity hotspot, not obvious for the invertebrates, is centered on the South Passage around Amity Point. Source: Queensland Museum database.

that it could be of a more general nature relying on recruitment events from outside of the Bay. Probably, however, such changes in species' presence are more moderate across the Bay and the total species richness is not markedly affected.

Biogeography

Unique faunal elements

The southeast Queensland region – from Cape Byron to Fraser Island – is of enormous taxonomic and biogeographical interest. The region is a meeting point of the southern (Peronian) and northern (Solanderian) faunas, representing a peculiar mix of both temperate and tropical species (see Endean *et al.,* 1956). The Bay is unique in its geography, and this has allowed the development of a diversity of habitats all in close proximity, which in turn has allowed a remarkably diverse fauna to develop. This may help to explain why there appears to be relatively large numbers of indigenous species. As Moreton Bay has existed in its present form for only 6 500 years (Stephens, 1992) it is unlikely that speciation has occurred within the Bay in such a short period of time. It is more likely that apparently endemic species are either relics from more widespread habitats that have been restricted to the Moreton Bay area; or that these species may occur in other enclaves along the coast but have so far remained undetected.

The following is a preliminary list of 27 species that are apparently only known to occur in Moreton Bay (apparent endemics):

Listriolobus bulbocaudatus Edmonds, 1963 (Echuroidea)
Branchiostoma moretonensis Kelly, 1966 (Cephalocaudata)
Alcyonium aspiculatum Tixier-Durivault, 1966 (Cnidaria)
Anthelia densa Tixier-Durivault, 1966 (Cnidaria)
Diaseris mortoni Tenison-Woods, 1880 (Cnidaria)
Garveia clevelandensis Pennycuik, 1959 (Cnidaria)
Heteropsammia moretonensis Wells, 1955 (Cnidaria)
Heteroxenia multpinnata Tixier-Durivault, 1966 (Cnidaria)
Malacobelemnon stephensoni Tixier-Durivault, 1966 (Cnidaria)
Pseudoplumarella echidna Bayer, 1981 (Cnidaria)
Bradiella brachiata Rullier, 1965 (Annelida)
Dasybranchus lumbricoides (Rullier, 1965) (Annelida)
Leonnates stephensoni Rullier, 1965 (Annelida)
Alpheus moretensis Banner & Banner, 1981 (Crustacea: Caridea)
Alpheus stephensoni Banner & Smalley, 1969 (Crustacea: Caridea)
Ctenocheles collini Ward, 1945 (Crustacea: Thalassinidea)
Charybdis moretonensis Rees & Stephenson, 1966 (Crustacea: Brachyura)
Libystes paucidentatus Stephenson & Campbell, 1960 (Crustacea: Brachyura)
Uca longidigita (Kingsley, 1880) (Crustacea: Brachyura)
Aplidium congregatum Kott, 1992 (Ascidiacea)
Aplidium incubatum Kott, 1992 (Ascidiacea)
Molgula rima Kott, 1972 (Ascidiacea)
Perophora multistigmata Kott, 1972 (Ascidiacea)
Perophora sabulosa Kott, 1990 (Ascidiacea)
Phallusia barbarica Kott, 1985 (Ascidiacea)
Pyura navicula Kott, 1985 (Ascidiacea)
Lingula tumidula Reeve, 1841 (Brachiopoda) (not subsequently recorded)

This list will be swollen by three new species of shrimps recently discovered in Moreton Bay in the genera *Onycocaris, Periclimenes* and *Periclimenaeus* (Bruce, 1998).

The Bay as a refuge

The Bay serves as a refuge for several species from both temperate and tropical faunas, and in some cases, considerably extends their range of distribution northwards or southwards, respectively. This is particularly evident for fish, where there are 141 species at the southern limit of their range, and 24 at the northern limit (Johnson, unpubl. data). A comprehensive species inventory for the Bay is required so that distribution patterns for invertebrate groups may be thoroughly analysed. It is also possible that while many species are able to grow here as adults, the region is outside their range of tolerance for breeding, and therefore rarer species occurrences may be only serendipitous. This may be the case for *Mictyris platycheles*, a temperate soldier crab for which only a single museum record exists. Moreton Bay is also the northern limit for the temperate grapsid crab genus *Paragrapsus* (see Campbell & Griffin, 1966). Kott (1972) explained distribution patterns of the ascidian fauna in her study area south of Peel Island, by delineating three groups. These faunal groups could be extrapolated to explain Baywide distributions and recuitment patterns from outside the Bay, i.e.: (1) a large group of ascidian species which ostensibly are enlisted from other areas via their long-lived larvae, but are unable to set up breeding populations because their sparse distribution prevents effective fertilisation of gonadial products; (2) species enlisted from other areas which set up breeding populations but experience an annual mortality preventing the establishment of self-perpetuating populations; and (3) persistent, reproductively successful species, with populations present throughout the year.

There is evidence from a number of sources that the sheltered waters and unique geography of Moreton Bay allows it to act as a refuge for Indo-West Pacific species that are not known from elsewhere in Australian waters, or if so, only from other similar sheltered embayments. This is true for at least five species of ascidians (Kott, 1972), and Hooper has unpublished data on the sponge fauna that shows a large proportion of sponge species also fall into this category. Endean (1953) indicated that the Moreton Bay echinoderm fauna was very rich but isolated, and that the nearest comparable assemblage was over 320 km to the north at Port Curtis, at the southern end of the Great Barrier Reef. Similarly Campbell & Stephenson (1970) recorded eighty-four species of sublittoral crabs from Moreton Bay and their analysis of the distribution of these species suggested that the fauna showed greater affinities with the northern Indian Ocean than with the western Pacific. Large continental embayments are relatively less common in the southwestern Pacific and also, except where they occur near large population centres, their species assemblages are poorly studied.

Sponges

The sponges (Porifera) make an interesting case study. Most sponges appear to have very narrow niches and are very sensitive to minute habitat differences, water temperatures (i.e. via currents), and geomorphology. Slight changes of habitat type can often mean great differences in species composition (Hooper, 1994).

Only about 10% of the Bay's sponge species are widespread tropical or temperate species occurring commonly along the eastern Australian coast. The remainder either only occur outside of Australian waters, or are as yet undescribed. Thus a comparison of sponge species from sites within the Bay with those in outside waters shows interesting differences in community composition (see Figure 4). Between adjacent sites within the Bay there is about a 20% overlap in species, which is of the same order of magnitude as the overlap between adjacent outside reefs; however the fauna shared between Bay sites and those outside is no more than 6%.

Inside the Bay: Species from two adjacent sites are more similar in composition to each other than they are to adjacent sites outside the Bay (see Figure 4). Species from the Green Island region are depauperate and characteristic of highly silted environments; species from Peel Island are a mixture of fringing coral reef species (with the shared species with sites from outside the Bay being those common to many southern coral reefs along the coast), silty water species (similar to Green Island) and soft bottom species (mostly these are the unique ones). Within the Bay, there are more unique species (i.e. presumed indigenous species) than outside the Bay, and there is no vertical (depth) stratification of species (presumably due to the shallow depth profile existing within the Bay).

Outside the Bay: Species from adjacent sites outside the Bay (from Byron Bay north to Noosa) are more similar to the nearest neighbour site (e.g. Byron Bay-Cook Island; Cook Island-Point Lookout; Point Lookout-Cape Moreton; Cape Moreton-Mooloolaba; Mooloolaba-Noosa), and least similar to those within the Bay (see Figure 4). There is easily observable vertical stratification of species outside the Bay, with temperate species often found at the bottom and sides of 'shark gullies' (25-40 m depth), and tropical species only on the tops of the gullies

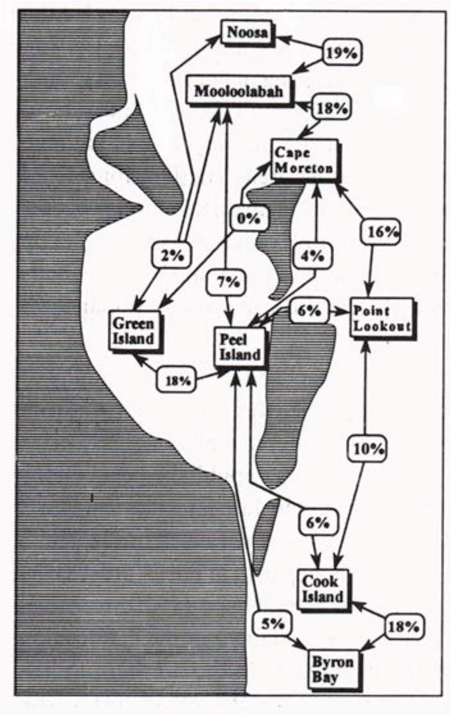

Figure 4.
A schematic representation showing percentage of sponge species shared between study sites within Moreton Bay, and those outside the Bay. Source: Queensland Museum database.

(10-20 m depth), like overlapping tongues; there are no similar habitats within the Bay, even on the artificial reefs, so maybe surface and bottom temperatures are more even (well mixed) within the Bay than outside the Bay where the cold water sits on the bottom moving north and the warm water floats over the surface southwards without extending far down.

Ascidians

Kott (1972) reported on the ascidian fauna of Moreton Bay, and in particular extensive collections made by dredging to the south of Peel Island. Because of the nature of the sampling program (i.e. principally Van Veen grab, with no collections via SCUBA), and the restricted geographic coverage, there would undoubtedly be a bias in the faunal composition, with an undersampling of hard substrate species. Nevertheless some interesting results were obtained. Of the 24 species recorded, 12 were not fixed to hard substrates but lived free on the bottom. Of this latter group, five species appear to be restricted to Moreton Bay, and of the non-endemics five are unusual in their distributions, having been previously recorded only from sheltered waters in widely separated geographic localities. The presence of non-fixed species is favoured, according to Kott, because their life histories are capable of withstanding the seasonal impact of freshwater flooding, silt deposition, and temperature fluctuations. The fixed species are all known to have a wide distribution around the Australian coast.

Alien species

Because Brisbane is a major shipping port, Moreton Bay is also a potentially important point of entry for many exotic marine invertebrates (fouling and bilge water organisms). Tracking the pattern of colonisation and dispersal of these species within and outside of the Bay is of interest to species distribution patterns and genetic flow within this system. Kott (1976) has reported on the introduction of the north Atlantic ascidian *Molgula manhattensis* (De Kay) to the Brisbane River. Two species of boring sponge, *Cliona vastifica* Hancock and *Cliona celata* Grant, which occur throughout the world as pests on oyster leases but which were previously unknown from Australia, have also recently been found in Moreton Bay (Wesche *et al.,* 1998). Their presence is attributed either to ballast water or the translocation of infected stocks.

Summary

Management strategies for maintenance of biodiversity in Moreton Bay should be aimed at two levels:

- a broad scale model recognising an inshore estuarine-dominated system; and an eastern marine system; and
- within these systems particular habitat types must be mapped and adequately monitored for changes in extent and faunal composition.

The Bay has a high faunal diversity with many species that appear to be endemic.

The Bay's wide range of habitat types and the fact that it lies on a biogeographic cusp allows it to act as a refuge for both temperate and tropical species, some of which are apparently not found in neighbouring regions. Because of this:

- biogeographically interesting species need to be recognised, and provision for their conservation made as part of habitat management plans; and
- the Bay is unusual in its potential to act as a host for alien species of both temperate and tropical origin, which could be accidentally introduced via fouling or ballast-water, and therefore at an increased risk.

The results of this study clearly demonstrate the value of lodging material in museums, and the potential for using these data to gain an understanding of distribution patterns and thus a broader ecological perspective.

Acknowledgements

We thank our colleagues at the Queensland Museum for database searches, and for providing personal knowledge of the fauna and its literature – in particular, Lester Cannon (worms), Patricia Mather (tunicates) and Jeff Johnson (fish). John Short and Libby O'Brien are especially thanked for their assistance in preparing the figures.

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Zooplankton of Moreton Bay: The Hidden Processors

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Introduction

By the accepted loose definition, plankton is composed of specialised organisms from all phyla having the one common feature: that their powers of locomotion are so poor that their distribution patterns, at least in the horizontal plane, are dictated by movements of the waters in which they float and drift. Plankton includes a complex assemblage of species of both producers (phytoplankton) and consumers (zooplankton), the composition and productivity of which is influenced largely by the available nutrient supply, solar radiation levels and spectral composition, grazing and predation relationships and, in coastal waters especially, by the distribution and spawning patterns of benthic and nektonic organisms. Being generally transparent and microscopic we are not usually conscious of their existence beneath the Bay's surface, except when swarms of larger jellyfish or large 'red tide' blooms of phytoplankton appear. But this 'hidden' planktonic system is an important link in the chain of nutrient flux and energy flow from microscopic primary producers through to higher trophic levels. It acts both as a sink for nutrients present in the water, and a primary storage system for energy incorporated mainly through photosynthesis.

Both the zooplankton and phytoplankton provide a fundamental supply of resources for higher trophic levels. In estuarine situations, zooplankton filter feeders process organic matter from both phytoplankton and detritus/bacterial producers, and themselves are consumed by a wide range of benthic and nektonic organisms. They further bridge boundaries between these latter two categories both ontogenetically and behaviourally. Much of the zooplankton is composed of the larvae (meroplankton) of benthic and nektonic animals, the survival and recruitment of which is crucial to maintenance of the adult stocks and thus biodiversity. The meroplankton, includes many mollusc, crustacean and fish species of recreational and commercial fisheries importance; their migrations both vertically in the water column, and between the sediment and water column (demersal forms), contribute to trophic links between these regions.

Through their secretions and excretions, zooplankters provide a pathway for short-term recycling of nutrients for continued primary production. Zooplankton composition is a reflection of the integration of recent conditions (scales of days to weeks) and can be useful as a biological indication of water quality, estimated from either a change in the abundance of key species, or broader species composition of the functioning of the planktonic system "community". Understanding faunal composition and dynamics is fundamental to understanding the system.

Phytoplankton is the major topic of another contribution elsewhere (Heil *et al.*, this volume), however, because of the fundamental trophic interdependence between phytoplankton and zooplankton, brief mention of it will be made here. Phytoplankton is a major producer of new organic material in all waters, especially where nutrients are not limiting and temperature and light intensities are favourable. There is a wide range of size in these primary producers, from nanoplanktonic flagellates (0.2 μm) to macroplanktonic chain-forming diatoms and cyanobacteria (several mm in length). All are capable of rapidly assimilating nutrients and increasing population biomass (especially the smaller single-celled species), and hence

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 347-364.

reflect changes in nutrient availability (other things being equal). Furthermore, different species respond to differing nutrient ratios for optimal growth rates. Therefore, both the total abundance (e.g. as reflected by chlorophyll *a* concentration) and the species composition of the assemblage present at any one time, may reflect the recent nutrient history of the waters in which they reside. In various estuarine systems, between 40% and 90% of all phytoplankton synthesis may be due to organisms in the nanoplankton range (2-20 µm) (Day *et al.,* 1989).

Removal of the primary producers from the system through grazing by zooplankters is the first step in the utilization of the energy, originally synthesised into organic compounds by the phytoplankton, by other members of the food web. Several trophic levels may exist within the zooplankton, and link through to many important estuarine carnivores, including commercially and recreationally important fish. Zooplankton grazing can regulate density, species composition and size distribution of phytoplankton populations. Again, there is a wide range of sizes in the zooplankton, which is important in considering trophic relations. Bacterial biomass may reach 40% of phytoplankton biomass (Cho & Azam, 1990), and process more than half of primary production (released as dissolved organic material) through to protozoan consumers. It is therefore important to know both the size spectrum and faunal composition of the plankton assemblage present.

Nano-zooplankters (2-20 µm) and micro-zooplankters (20-200 µm) (e.g. various protozoans, small metazoan larvae, including nauplii) are frequently much more important grazers of phytoplankton than are macrozooplankters in estuarine systems, removing from 13-88% of the phytoplankton production (mainly because of the high density of phytoplankton in the nano-size range) (Day *et al.,* 1989). Nanozooplankters can also be major consumers (40-45%) of estuarine bacterioplankton (= picoplankton; 0.2-2.0 µm) (Sheer *et al.,* 1986), and form a food source for meso- and macro-zooplankters. Most of the net-zooplankton (= mesoplankton, 200 µm-2 cm; macrozooplankton, 2-20 cm) in estuaries is herbivorous or omnivorous, but may account for considerably less than 50% of phytoplankton grazing (Day *et al.,* 1989). Additionally, copepods and other crustaceans in the zooplankton excrete NH_4^+ and dissolved organic nitrogen (DON) which can be assimilated and incorporated into new production by phytoplankters. They can also influence the rate of NH_4^+ remineralisation through sloppy feeding and by grazing on smaller micro-heterotrophs.

Early Zooplankton Studies of Relevance to Moreton Bay

Plankton in general, and zooplankton dynamics in particular, have been little studied in Australian waters. Although taxonomic studies in the region date back to 1852 (see below), there was no documented study of zooplankton in Moreton Bay prior to the 1960s. Studies which have subsequently taken place in Moreton Bay reveal a fauna of mixed oceanic, neritic and estuarine affinities, the taxonomy of which was in part documented in the earlier expeditionary studies, the reports from which are still an essential tool in any study of plankton faunistics in Moreton Bay.

Three of those early expeditions in Australian waters were particularly significant: the first serious observations were those of Dana (1852; 1855) on pelagic crustaceans collected during the U.S. Exploring Expedition of 1838-1842, then followed the “Challenger” expedition of 1873-1876, and the British Antarctic (*Terra Nova*) expedition of 1910-1913. The first comprehensive studies which included both taxonomic and seasonal components arose from the British Museum Great Barrier Reef Expedition of 1928-1929, centred on the waters around Low Isle in the northern Great Barrier Reef. Subsequent studies, mainly off south-eastern Australia, enhanced this knowledge (e.g. Dakin & Colefax, 1933; 1940; Sheard, 1949; 1965;

Kott, 1955; 1957; Tranter, 1962; and many CSIRO Division of Fisheries and Oceanography station lists and reports, see Greenwood 1973; 1976) with some taxa being studied in detail, including pelagic tunicates (Thompson, 1942; 1948; Heron, 1972a; b), chaetognath worms (Thomson, 1947), medusae (Mayer, 1910; 1915; Uchida, 1947; Kramp, 1953; 1965; 1968), trachymedusae and narcomedusae (Blackburn, 1955).

In Queensland waters the Great Barrier Reef Expedition made general seasonal studies of phytoplankton (Marshall, 1933) and zooplankton (Russell, 1934; Russell & Coleman, 1934), as well as giving detailed taxonomic and seasonal accounts of planktonic tunicates, molluscs and coelenterates (Russell & Coleman, 1935; Kramp, 1953), mysids and euphausids (Tattersall, 1936), and copepods (Farran, 1936; 1949). No other substantive taxonomic or distributional accounts of zooplankton in neritic waters of Queensland were published until the 1970s. All of the above works are of great use in studying the zooplankton of Moreton Bay, but must be supplemented with use of specialist literature from elsewhere in the Indo-Pacific region, and especially the classic works from waters to the immediate north of Australia, including for copepods those of: Dana, 1849; 1852; 1855; Brady, 1883; Cleve, 1901; Carl, 1907; Scott, 1909; With, 1918; Fruchtl, 1923; 1924; Steuer, 1926; 1932; Schmaus & Lenhoffer, 1927; Dammerman, 1929; Sewell, 1929; 1932; Delsmann, 1939; 1949; Vervoort, 1949; Chiba & Tsuruda, 1955; Wickstead, 1958; 1961; Brodsky, 1962; Fleminger, 1963; Lang, 1963.

What Do We Know About Moreton Bay Zooplankton?

Composition of the fauna

The first published studies of zooplankton in the Moreton Bay region appeared in the 1960s, with a series of subsequent papers documenting components of both the estuarine and more marine holoplankton (Table 1). Many of these papers also contain information on spatial and temporal distribution of the fauna. During the same period several papers were published describing the (meroplanktonic) larval stages of (especially) decapod crabs and shrimps that occur within the Bay system. Other studies examined the change in zooplankton composition across the estuarine axis of the Bay, the transport of oceanic forms into the Bay, and the contribution of emergent demersal (substrate-related) forms to the plankton. In addition to the published information on plankton, much resides in unpublished form in postgraduate theses and various reports to government and industry. Many of the latter reports are not available for general access. Summary listings of the known taxonomic and ecological/physiological literature on zooplankton of the Moreton Bay region are presented in Tables 1 and 2. In the following over-view paragraphs I have drawn upon information available in many but not all of these works. The ultimate sources of detailed and comprehensive information are of course the original works as listed in Tables 1 and 2.

Of the approximately 70 papers on zooplankton (Tables 1 and 2), the majority (47) have had a primary focus on documenting the fauna. Through these works we have reasonable knowledge of much of the crustacean micro- and meso-plankton (Table 1), there being 68 species of calanoid copepods, 34 species of cyclopoid copepods, 11 species of harpacticoid copepods, 67 species of cumaceans (23 from Waterloo Bay alone, Johnston, 1966), and 28 mysid species (references in Table 1 and author's unpublished records) known from Moreton Bay and its estuaries. Many of these species were new, the Bay therefore being the type locality for them. These numbers indicate a relatively high biodiversity in the fauna (see, for example, comparisons for calanoid copepods in Othman *et al*, 1990), partly due to the now broadly demonstrated overlap of tropical and temperate, estuarine and oceanic species in the region (see below, and Davie & Hooper, this volume), and partly reflecting the relatively

Table 1. Summary listing of taxonomic studies on plankton of Moreton Bay and estuaries.

Focus of Study	Region of Study	Author(s)
Medusae	Moreton Bay	Payne, 1960*; Hamond, 1971; Gorman, 1988.
Calanoid copepods	Brisbane River estuary	Bayly, 1963; 1964; 1965*; 1966; Bayly & Greenwood,1966.
	Southern Moreton Bay	Fleminger *et al.*, 1982; Greenwood, 1972; 1976; 1977; 1978a; b; c; d; 1979; 1981; Greenwood & Othman, 1979.
Cyclopoid copepods	Southern Moreton Bay	Thwin, 1972; Greenwood, 1971.
Harpacticoid copepods	Southern Moreton Bay	Thwin, 1972; Coull *et al.,* 1995.
Cumaceans	Waterloo Bay Southern Moreton Bay	Johnston, 1966; Greenwood & Johnston, 1967; Tafe, 1995*; Tafe & Greenwood, 1996a; b; 1997a; b.
Mysids	Moreton Bay and estuaries	Hodge, 1963a; Bacescu & Udrescu 1982; Greenwood & Hadley, 1982; Greenwood *et al.,* 1991; 1997; Wooldridge *et al.,* 1992.
Decapod larvae	Brisbane River estuary Moreton Bay	Greenwood *et al.,* 1976; Greenwood & Fielder, in prep. Fielder & Greenwood, 1979, 1983a; b; 1985a; b; c; 1987; Fielder *et al.,* 1975, 1979, 1984a;b;c; Greenwood & Fielder, 1979a; b; 1980; 1983a; b; 1984a;b, 1988; Shepherd, 1969.
Ichthyoplankton	Moreton Bay and estuaries	Helbig, 1969*; Pham *et al.,* this volume.

* indicates unpublished thesis

high intensity of sampling which has been undertaken here. They are also likely to be very conservative, since mesh size used in the sampling nets greatly influences estimates of both abundance and diversity (see e.g. comparative data in Greenwood, 1980). In most studies on zooplankton of Moreton Bay, mesh sizes of 195 µm or greater have been used. Such mesh greatly under-samples the smaller cyclopoid and harpacticoid copepods, and many components of the meroplankton. For example, in the temperate zone Hopkins estuary (Victoria), which has only 11 species of calanoid copepods (cf. 68 in Moreton Bay), and 11 of cyclopoids (cf. 34), Newton (1994) using plankton traps with 80 µm mesh recorded 35 species of harpacticoids (cf. 11 in Moreton Bay), and plankton numerical densities averaging 60 300/m^3 (cf. 1 700/m^3 with 195 µm mesh in Moreton Bay, Greenwood, 1980). The little which is known about the harpacticoid fauna of Moreton Bay has been derived incidentally through studies focused on other elements of the plankton (e.g. Thwin, 1972; Greenwood, 1980), or through studies on the feeding biology of juvenile fish (Coull *et al.,* 1995; Mangubhai *et al.,* this volume). There is no doubt that fine-meshed sampling would greatly increase our understanding of the diversity and abundance of zooplankton in Moreton Bay.

The meroplanktonic larval stages of 28 species of the more common brachyuran crabs (12 Portunidae, six Grapsidae, four Ocypodidae, three Mictyridae and three Xanthidae), three species of porcellanid anomuran crabs, and two species of prawns occurring in Moreton Bay have also been described, either from specimens captured in the plankton, or through the rearing of these larval stages from embryos carried by ovigerous females (Table 1).

Table 2. Summary listing of ecological and physiological studies on zooplankton of Moreton Bay and estuaries.

Field of Study	Focus of Study	Region of Study	Author(s)
Distributional & behavioural ecology	General zooplankton	Moreton Bay Raby Bay	Greenwood, 1980. King & Williamson, 1995.
	Calanoid copepods	Brisbane River estuary	Bayly, 1965a; b; Greenwood, 1980; Sheehy & Greenwood, 1989.
		Southern Moreton Bay	Greenwood, 1981; 1982a; b.
		Southern Moreton Bay	Thwin, 1972*.
	Demersal copepods	Southeastern Bay	Jacoby & Greenwood, 1989; 1991.
	Cumaceans	Moreton Bay Pumicestone Passage	Tafe & Greenwood, this volume Tafe *et al.,* this volume.
	Mysids	Estuaries	Hodge, 1963b; Greenwood *et al.,* 1997.
	Sergestid shrimp	Cabbage Tree Creek	Xiao & Greenwood, 1992.
	Penaeid/shrimp larvae/juveniles	Southern Moreton Bay Brisbane Estuary	Barber & Lee, 1975; Young & Carpenter, 1977; Aziz & Greenwood, 1982; Rothlisberg *et al.,* 1995; Thorne *et al.,* 1979.
	Brachyuran larvae	Brisbane River estuary	Greenwood *et al.,* in prep.
	Fish eggs & juveniles	Moreton Bay	Helbig, 1969*; Pham *et al.,* this volume.
Physiology & eco-toxicology	Zooplankton/*Lucifer*	Moreton Bay	Marsh, 1980.
	Crab/prawn larvae/juveniles	Moreton Bay Estuaries/lagoons	Aziz & Greenwood, 1981; Greenwood & Fielder, 1983b; Preston *et al.,* 1985.
	Mysids, copepods		Thomson & Dunstan, 1968; Brown *et al.,* 1996, this volume.

* indicates unpublished thesis.

There have been few studies of non-crustacean zooplankters in the Bay, and these are mainly in unpublished theses. Payne (1960) sampled scyphomedusae in Moreton Bay between May 1958 and February 1960, recording the presence of just five common species for which she created ‘plankton calendars’ (*Catostylus mosaicus, Cyanea capillata, Aurellia labiata, Charybdea rastoni* and *Pelagia noctiluca*) and of *Tamoya rastoni*. She commented that (loc. cit. p.68) “The medusa calendar is chiefly remarkable for the small number of species recorded....”, suggesting ...”that in a ‘normal’ summer, the number of species recorded would be much higher.” From subsequent studies this appears to be true. Hamond (1971) listed ten species of hydromedusae (“presumably originating from hydroids; not one of these hydroids has yet been described”, p.25), one scyphomedusan, and two species of ctenophores, but noted that only a few of the “immense numbers of medusae frequently encountered in Moreton Bay” were examined. Gorman (1988) added a further six species of hydromedusae to the list, plus one siphonophore and a second species of *Aurelia* (*A. coerulea*), to give a total of 16 hydromedusae, seven scyphomedusae, one siphonophore (*Muggiaea* sp.) and two ctenophores (*Pleurobrachia, Beroe*) recorded from the Bay. The present author's unpublished observations clearly indicate there are many more species of all these taxa present, including the hydromedusae of *Steenstrupia* sp. and *Sarsia* sp., the scyphozoan *Cassiopea andromeda*, several other siphonophores, and the ctenophore *Velamen* sp.

Ichythoplankton has received surprisingly little attention given its pivotal role in recruitment to stocks of commercial and recreational importance. Helbig (1969) sampled ichthyoplankton for six months at six locations extending from the southern end of Pumicestone Passage in the north, to Jumpinpin in the south. He recognised 115 different 'types' of larval fish, but was able to identify only 41 species amongst these. Rainbow Channel (Myora) was shown to have the greatest diversity of types, being an overlap region for "species" otherwise found only at Jumpinpin or Tangalooma. My associates and I have a long term (six years completed, and continuing) sampling program for macrozooplankton under way in the northern half of Pumicestone Passage. Preliminary findings on the ichthyoplankton from that sampling are given elsewhere in these proceedings (Pham *et al.,* this volume).

Distribution

General zooplankton

Accounts of general zooplankton composition and distribution come from studies initiated by Greenwood in the mid-1960s, and published in a series of papers through the 1970s and early 1980s (Tables 1, 2). The ecological components of those studies were based on samples taken with Clarke-Bumpus samplers having 195 µm mesh, from up to seven localities from the Rainbow Channel in the east, across the southern and central Bay to the Pile Light and Brisbane River mouth. Seasonal changes in total zooplankton numerical abundance were small, mean abundance being 1 691 individuals/m^3 with a more extended period of lower abundance in September/October (Greenwood, 1980). Within this relative constancy there was, however, considerable change in faunal composition. During winter, holoplanktonic forms dominated the fauna, reaching both their relative (88% of the fauna) and absolute (2 235/m^3) abundance maxima. Meroplanktonic larvae were present throughout the year, averaging 33% of the fauna, with maxima in early autumn (1 122/m^3) and spring/summer (1 481/m^3) at which time they formed 56% of the total plankton. These findings for watercolumn zooplankters differ somewhat from those noted by Jacoby & Greenwood (1989) for emergent demersal zooplankters during their study in waters off Dunwich. Twenty-five of the 35 taxa of adults taken in their study were in greater numbers in summer, as were 10 of the 14 taxa of meroplankters. The holoplankton (Greenwood, 1980) was dominated (73%) by copepods, which formed 50% of the total zooplankton, their numbers/m^3 in winter being twice those at other times. Calanoids formed 76% of all copepod numbers, with larvaceans, cladocerans and chaetognaths being the next most abundant taxa. The distributions of chaetognaths, larvaceans, salps, branchiopods and selected copepod species suggested an influx of oceanic waters into the Bay between February and August, with a possible second minor influx in late spring. The most numerous meroplankters were (in decreasing order) mollusc veligers, decapod larvae, barnacle nauplii and annelid trochophores.

Other and subsequent studies have focused on particular taxa and/or localities. In addition to giving seasonal calendars for the scyphomedusae she recorded, Payne (1960) provided the only ecological data yet available on *Catostylus mosaicus*, including relative abundance through the year, growth rate (bell diameter) and "brood" production (i.e. successive spawnings), in northern and southern parts of the Bay. She suggested that broods produced in the southern Bay region have a relatively short peak of abundance compared with those in the northern Bay, that growth rates were highest in winter when phytoplankton was abundant, and could be zero or negative in summer. She also noted that whereas the direction of movement of individuals in aggregations seemed to be random in still waters, as current velocity increased, the percentage with synchronous orientation swimming obliquely up-current increased toward 100%, a behaviour which would contribute to aggregated distribution patterns. Brief observations on aggregations and behaviour of *C. mosaicus* were also made by Gorman (1988).

Analyses of the planktonic cyclopoid (Thwin, 1972) and calanoid (Greenwood, 1982a; b) copepod fauna in Moreton Bay show that it is not composed of a series of discrete faunal units which replace each other in time and space. Rather it is a series of cline-like successions in which the extremes are recognisably different both spatially and seasonally, with intermediate situations being less distinct. Three major types of species associations are evident, their origins relating to two obvious major (hydrographic) environmental influences (see below).

In summary, calanoid and cyclopoid species distribution patterns (detailed in Greenwood, 1982a; b, and Thwin, 1972 respectively) both show that:

- The spatial changes in copepod faunal composition which are found across the east/west axis of the Bay are greater overall than are seasonal differences;
- Three main spatial faunistic zones exist – oceanic-influenced (eastern Bay), estuarine/estuarine-influenced (western Bay), and central Bay. (These three zones correspond to the three water bodies which Newell (1971) differentiated within the Bay from salinity and temperature characteristics alone);
- Within these spatial zones, seasonal influences on species composition of the zooplankton are evident, especially in the central Bay;
- Homogeneity of faunal characteristics is strongest in the more extreme situations (i.e. oceanic-influenced or estuarine-influenced regions, and summer or winter seasons);
- There are periodic injections of numerous oceanic species (see also next section) into the Bay on tidal and seasonal fluxes. (The majority of these species are infrequent in occurrence though sometimes in high abundance when present, or are significant in being present simultaneously);
- There is injection of lower estuarine species into near-estuarine (western Bay) areas, the extent of this varying with river flow and hence seasonal rains. The selective, species-filtering, influence of river dilution further modifies species composition of the western Bay fauna at these times, reducing species richness in comparison with central Bay waters. This results in a seasonally varied though less diverse presence in the estuarine-influenced western Bay, of species typically found in the central Bay assemblage.
  (Species characteristic of these latter two situations are of interest because of their fidelity to them. Such species are good indicators, in the classical sense, of particular environmental conditions. Relationships are, however, not so precise for the remaining species); and
- The calanoid fauna is dominated by a relatively small number of species (15, see Greenwood, 1982a) which occur throughout the Bay in all seasons, but whose relative abundances vary spatially and temporally. These 15 ubiquitous species were amongst the

sixteen most frequently taken, collectively accounting for 76% of all species-in-sample records, and most at some time dominated the fauna. The biology of these species, which are presumed to be of major ecological importance in the Bay, is discussed in Greenwood (1982a).

Calanoid copepods as indicator species and species-groups

Many of the less ubiquitous species show relatively high abundance and spatial restriction to particular water bodies, and hence are useful indicators of water movements. Cluster analysis of the occurrences of such species (Greenwood, 1973), and examination of the collective occurrences of 25 calanoid species which individually were too rare to be included in cluster analyses (Greenwood, 1982b), suggested two principal groups that may be diagnostic of water masses.

(1) Indicators of an offshore/oceanic water intrusion:

- Late summer/early autumn intrusion: *Acartia negligens, Acrocalanus monachus, Candacia catula, Euchaeta concinna, Eucalanus mucronatus, Euchaeta marina, Labidocera acuta, L. kroyeri, L. laevidentata, L. minuta, Rhincalanus rostrifrons,* and *Undinula vulgaris*;
- Autumn intrusion: *Candacia discaudata, Mecynocera clausii* and *Acrocalanus longicornis*; and,
- Winter intrusion: *Candacia bradyi* and *Metacalanus aurivilli*.

(2) Indicators of estuarine conditions:

- *Acartia baylyi, Gladioferens pectinatus*, and especially *Isias uncipes*.

No species were restricted to estuarine-influenced western Bay areas. The varied physiological conditions resulting from variable salinities prevailing in these regions, were probably beyond the physiological capacities of those species which are oceanic and estuarine immigrants.

Seasonality of oceanic influence

Examination of the distribution patterns of total zooplankton (Greenwood, 1980), the more frequently taken calanoids (Greenwood, 1982a), and the rarer calanoids (Greenwood, 1982b) all demonstrate that the influence of subtropical oceanic waters in coastal regions of southeast Queensland, and hence by tidal exchanges into Moreton Bay, is at its highest during the late summer to mid-winter period (February to July). Such seasonal subtropical oceanic influences were evident in all three years studied. Whilst tidal exchanges may facilitate entry of plankters from coastal waters into the Bay throughout the year (e.g. Rothlisberg *et al.,* 1995), the copepod fauna shows little evidence of incursion of subtropical oceanic waters and biota outside the February to July period.

The Moreton Bay planktonic copepod fauna is dominated (52 of the 57 species of calanoids known beyond the Bay) by species of tropical/subtropical affinity (Greenwood, 1982a). There is a southward extension in distribution of warm-water species along the coast of eastern Australia, with a progressive southward attenuation in the number of species occurring. Similar extensions of tropical planktonic forms southward along the east coast of Australia have been demonstrated for planktonic tunicates (Thompson, 1942), chaetognaths (Thomson, 1947) and holoplanktonic molluscs (Newman, 1990).

The probable role of the seasonally variable East Australian Current system (Rotschi & Lemasson, 1967) in transporting such species into coastal waters of southeastern Queensland, has been emphasised by Greenwood (op.cit.). Tropical and subtropical species are more common in waters off Moreton Bay during summer/autumn and this is reflected in increased numbers of such calanoid copepod species and individuals in Bay plankton at or soon after that time. This influence is not manifest during the late winter/early summer period because equatorial water is not fed into the system after about March, and transport velocities are then slower (Rotschi & Lemasson, op. cit.). From the adjacent coastal waters, tidal exchanges can introduce plankters into Moreton Bay. Rothlisberg *et al.* (1995) have demonstrated how a behavioural change from diel to tidal vertical migration, in combination with tidal transport through Bay entrances, is instrumental in the recruitment of penaeid prawn larvae from coastal waters into southern Moreton Bay.

Some copepod species which make only seasonal appearances in the Bay plankton may, nevertheless, be present in offshore waters for more extended periods, or throughout the year (e.g. *Mecynocera clausii* and *Labidocera kroyeri*, Greenwood, 1982a). Limited invasions of these species occur during summer/autumn. Since tidal exchange mechanisms exist which could, throughout the year, transport individuals from these populations into the Bay (Rothlisberg *et al.,* 1995), it is probable that their tolerance limits and/or trophic requirements may reduce survival in the more environmentally variable waters of a semi-enclosed Bay, other than during the more benign (warm and productive) months.

Behaviour and demersal forms

The interplay of zooplankton vertical migration patterns relative to diel, tidal and lunar cycles can be important in recruitment and maintenance of populations in bays and estuaries (e.g. Forbes & Benfield,1986; Forward, 1988; Xiao & Greenwood, 1992; Rothlisberg *et al.,* 1995; Dame & Allen, 1996; Ekman, 1996). Recruitment of commercially important penaeid prawn larvae/postlarvae into Moreton Bay has received some attention. In their studies of recruitment of *Penaeus plebejus* into the Southport Broadwater, Barber & Lee (1975) and Young & Carpenter (1977) both noted flood-tide transport to be important, with an overlying day/night effect, but data were variable. Subsequently, Rothlisberg *et al.* (1995) examined the distribution of larvae and currents in coastal waters off Moreton Bay and suggested that some larvae, undergoing a normal diel vertical migration behaviour, may be advected toward the coast by prevailing oscillatory wind-induced currents. Those postlarvae reaching shallow waters near the coastline become epibenthic and their vertical migration behaviour changes from a diel to a tidally induced rhythm. The animals then respond to (presumably) increases in hydrostatic pressure associated with the rising tide and thus achieve the previously observed pulsed recruitment into estuarine Bay waters. Vertical migrations of the copepod *Gladioferens pectinatus* have been suggested to maintain horizontal distribution of populations in the Brisbane River estuary through normal diel migrations between layers of differing water velocity (Bayly, 1965b). *Gladioferens pectinatus* in this situation was subsequently shown to have a tidal component in its migration behaviour (Sheehy & Greenwood, 1989), and it was further suggested that there is an overriding tendency for these copepods to attach (by their dorsal surfaces) to substratum-related surfaces during daytime ebb tides, thus achieving a net retention in the estuary. In studies on *Acetes sibogae* populations in Breakfast Creek (Xiao & Greenwood, 1992) it was noted that whilst a normal diel vertical migration pattern was evident, greater numbers of these shrimp were found to be high in the water column, where water velocities are greater, during flood tides than ebb tides, again achieving a net retention.

No evidence was found of utilisation by this shrimp of differing across-estuary current vectors, though this is known to be a factor in retention of some macroplankters elsewhere (Wooldridge & Erasmus, 1980), and a further study based in Breakfast Creek (author and others, in prep.) suggests it is so for populations of the mysid *Rhopalophthalmus brisbanensis*.

Comprehensive studies of demersal plankters captured in emergence and re-entry traps over coral, coral-rubble, seagrass, and mud substrata in the eastern Bay revealed complex lunar and diel patterns of emergence into the water column (Jacoby & Greenwood, 1989). Most taxa emerged at some time during or near the hours of darkness, but five different temporal emergence patterns were recognised in 34 of the 43 taxa studied. Overlying these patterns, 20 of the taxa emerged in greater numbers during new or quarter lunar phases, while 9 of the 14 meroplanktonic taxa and 10 others emerged in significantly greater numbers during full moon periods. Some zooplankters emerged at all stages of the tide, suggesting the lunar effect was due to other than tidal velocities for at least those forms. Detailed studies of the diel migrations of demersal cumaceans in similar localities in the Bay (Tafe, 1995; Tafe & Greenwood, this volume) showed that only 21 of 36 species studied migrated into the water column, that these migrations are triggered by low light intensity, the patterns being synchronised with periods of night-time slackwater, except when slackwater occurred around dusk/dawn. A bimodal or trimodal pattern of emergence was therefore dominant, based around the third lunar quarter. Three species (*Bodotria armata, Leptocuma barbarae* and *Paradiastylis mollis*) accounted for 95% of the numbers captured near the surface in autumn. Data on the temporal and spatial distribution of cumaceans along the estuarine gradient of Pumicestone passage are presented elsewhere in this volume (Tafe *et al.,* this volume).

The diversity of cumaceans emergent from seagrass habitats was marginally lower than that from silt/mud, but mean abundance was higher (Tafe, 1995) . By contrast, far greater numbers of nearly all other demersal taxa (41 of 43) (Jacoby & Greenwood,1989) were found to emerge from the more complex substrata (coral, seagrass) as opposed to coral rubble or mud, and the numbers taken per unit area in Moreton Bay were an order of magnitude greater for most taxa than those taken in a similar study in Heron Reef Lagoon (Jacoby & Greenwood, 1988). This latter finding suggests that their importance in trophic webs and nutrient flux between the sediment and water column (Bishop & Greenwood, 1994) may be greater in Moreton Bay than it is in the more fully studied coral reef environments.

Physiology and ecotoxicology

Earlier concerns about the levels of metal ions in creeks and estuaries draining into Moreton Bay led to acute toxicity studies of their impact on the survival of meroplanktonic larvae of the two commercially important crabs *Portunus pelagicus* and *P. sanguinolentus*, and of *Charybdis feriatus* (Greenwood & Fielder, 1983b). All three species were found to be sensitive to levels below those found at that time in some Moreton Bay estuaries, and estimated safe limits were well below the maximum concentrations reported for cadmium and zinc entering the Bay. It was suggested that harmful sublethal effects on larval development of the two sensitive *Portunus* species could be occurring at even lower concentration levels (Greenwood & Fielder, op. cit.). An area of recent concern in Moreton Bay has been the effect that mosquitocides sprayed along estuarine margins, can have on non-target aquatic species. Recent and on-going studies (see Brown *et al.*, 1996; this volume) utilising copepods, shrimp mysids and juvenile fish, have shown that certain of the larvicides may be highly toxic, especially to mysids, and hence have an impact on the estuarine food chain.

What Do We Need to Know?

The introduction to this paper outlined some of the interactions and processes with which zooplankters are involved. It will already be obvious to the reader from the above brief overview of our current state of knowledge, that there are many gaps and that we know very little about the interactive dynamics and feedback regulation of zooplankton/phytoplankton, and energy and nutrient fluxes through the system, or recruitment to the nekton and benthos. It would be tedious, and to some extent repetitious of that introduction, to attempt a comprehensive account of what needs to be done. Rather, in what follows I will highlight selected features which I consider to be of importance.

(1) Whilst we now have reasonable knowledge, for the southern half of the Bay system (i.e. the area approximately south of a line from the Brisbane River mouth to South Passage), of the composition and abundance of meso- and macro-zooplankters (i.e. >200 μm), our knowledge of the potentially important smaller forms is very poor. Studies are needed on the pico-/nano-/micro-zooplankton components (i.e. 0.2-200 μm), including the bacterioplankton. Studies on all components, including taxonomically difficult groups such as the amphipods, are needed in the northern half of the Bay, and other representative regions.

(2) Variations in species composition, size, biomass and production may arise not only from ambient nutrient levels and/or from cyclic influences such as changes in temperature and illumination, but also from acyclic non-seasonal influences, such as river flooding in the western Bay and oceanic intrusions in the northern and eastern Bay. The impacts of these events need to be determined.

(3) Meroplankters are the dispersal stages of benthic and pelagic species, including those of recreational and economic importance. Establishment and maintenance of the adult populations of these species is reliant upon the supply of competent larvae into regions suitable for metamorphosis and settlement. Although meroplankters are known to constitute up to 56% of the plankton, a large proportion of these forms is of unknown parentage. Much taxonomic work awaits here, as do studies on seasonality of their release, factors influencing their dispersal and survival, and mechanisms governing their recruitment to adult habitats.

(4) There are often spectacular occurrences of swarming species, in the plankton, e.g. salps, ctenophores and medusae, the impact of which is unknown. Amongst the medusae, *Catostylus mosaicus* can reach spectacular densities in the Bay, yet little is known about its biology. Being a rhizostome and hence having large numbers of microstomae, it probably consumes large amounts of both nano- and micro-zooplankton, provides habitat for juvenile fish and crustacean larvae, is a food source for some turtles and fish, contributes to the microbial system in the water column through its mucus secretions, and in the benthic system through death and decay. There is also an untapped potential for the Moreton Bay populations of *C. mosaicus* to become a small commercial fishery. Its biology in the system needs study.

To understand the processes governing plankton distribution and dynamics in a system such as Moreton Bay, not only do the general phenomena/principles mentioned in the introduction to this paper need to be considered, but there are smaller more specific trophodynamic responses which are of importance and hence should be addressed. We need to:

- Determine relationships between nutrient levels and ratios (N, P, Si), phytoplankton growth and production, and zooplankton composition and grazing, in different parts of the system;
- Determine the extent to which primary production may be sustained by nutrients regenerated or released by the actions and excretions of zooplanktonic grazers;
- Identify, quantify, and size fractionate major zooplanktonic grazers, and determine their *in situ* grazing rates on different size fractions (autotrophic and heterotrophic);
- Determine the role of demersal forms in mediating nutrient flux between the sediment and water column; and
- Determine the role of resuspended detritus as a secondary food source, either directly or through the microbial system.

The future holds many challenges for research towards a fuller understanding of the compositional and functional dynamics of the zooplankton, and its interactions with other trophic levels, of the Moreton Bay system.

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Ecology of Benthic Invertebrates in Moreton Bay

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Abstract

There is a long history of research on the ecology of benthic invertebrates in Moreton Bay and these descriptive studies set the scene for more detailed investigations on the processes structuring communities of benthic organisms, but the challenge has not been followed up with the experimental work necessary to facilitate an understanding as to whether, and how and why, many human activities affect benthic organisms within the Bay. This lack of focus on the ecology of soft-sediment benthic organisms reflects a more general pattern across Australia, where there is a distinct paucity of ecological information available for most benthic fauna in soft-sediment systems, despite these habitats comprising over 65% of our coastline.

In many ways, Moreton Bay provides an ideal locality for directing a new focus on the ecology of benthic invertebrates in estuarine systems in Australia. There is an enormous concentration of scientific expertise located in institutions such as CSIRO at Cleveland, the Southern Fisheries Centre at Deception Bay, the Queensland Museum and in the University sector, all within the immediate environs of the Moreton Bay region, which provides an exceptional opportunity to further our understanding of the interactions between a major estuarine environment and a rapidly growing conurbation, Southeast Queensland. This review describes the information currently available on the ecology of benthic invertebrates occupying soft-sedimentary habitats in the Moreton Bay region and then discusses the gaps in our knowledge with respect to specific goals identified in the management plans for the region, and more generally, as a response to particular human activities that can affect the communities of organisms occupying these environments. Some future research directions are identified for the short- and medium-term in the hope that this will provide a focus for subsequent discussion and perhaps cross-institutional collaboration.

Introduction

Moreton Bay is a large subtropical estuarine embayment on the east coast of Australia (Lat. 27° 20' S, Long. 153° 20' E), approximately triangular in shape, and enclosed by the mainland in the west and sand barrier islands (Moreton Island, North and South Stradbroke Islands) in the east (Moreton Bay Strategic Plan: Queensland Government, 1993). It has a surface area of approximately 1 500 km^2 in the open-water section to the north, and a catchment area of over 18 300 km^2. Greenwood (1993) provides a detailed summary of the main physical features of the Bay. Moreton Bay is characterised by having extensive areas of seagrass and mangroves especially in the western and southern regions, in addition to considerable expanses of saltmarsh and saltpan (Williams, 1992). A complex series of banks and channels have been formed in the tidal delta areas of North and South Passage (Stephens, 1992), and those formed through South Passage sustain large areas of seagrass beds and intertidal flats. There are also prominent areas of intertidal rocky reef, especially on North Stradbroke and Moreton Islands, and fringing coral reefs occur around many of the islands in the central Bay and along the shore of North Stradbroke Island (Greenwood, 1993).

The enormous range of different benthic habitats within Moreton Bay supports a rich and diverse invertebrate fauna (see Davie & Hooper, this volume). This paper reviews existing information on the ecology of these benthic invertebrates and provides an overview of areas

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 365-394.

for research for which there is insufficient information available to allow effective management of the Bay's coastal resources in a sustainable manner. It focuses on research which provides empirical information from field-based studies, rather than taxonomic descriptions and/or natural history papers.

Existing Information on the Ecology of Benthic Invertebrates in Moreton Bay

Fairweather (1990) reviewed 729 scientific articles published from 1980 to 1987 in 27 leading marine and ecological journals to determine how research effort was distributed among different coastal habitats around Australia. The different types of habitats considered included coral and rocky reefs, seagrasses, mangroves, sandy and muddy bottoms and saltmarshes. All of these habitats are exposed to a wide range of different potential impacts arising from human activity in the coastal zone (Resource Assessment Commission, 1993).

The results of the detailed analysis of published research showed clearly there were obvious gaps in our understanding of the functioning of communities from different habitats. Among the deficiencies in available information identified by Fairweather (1990), was the need for urgent research on saltmarshes, intertidal mudflats and sandy beaches virtually anywhere in Australia, mangroves in subtropical and temperate areas, and seagrasses in Queensland. All of these habitats are extremely well represented in Moreton Bay (Quinn, 1992), but there have been few detailed ecological studies which provide the information necessary for the sustainable management of marine resources in the Moreton Bay region. In some cases, there has been important research done in the region since Fairweather completed his review, such as the extensive work on seagrasses in Moreton Bay (Abal *et al.*, 1994; Grice *et al.*, 1996), but these studies have generally not included examination of the benthic macro- or meiofauna. There are still substantial gaps in our knowledge of the ecology of benthic invertebrates in the Moreton Bay region today.

Subtidal infaunal invertebrates

The detailed work of Stephenson, his students and colleagues established a firm database of information on the distribution and relative abundance of subtidal benthic invertebrates in the Moreton Bay region. These descriptive studies (see Table 1) provided valuable information on the diversity of taxa found in the Bay. Some of these studies were done in preparation for the impending Brisbane Airport extension: either in areas to be reclaimed as part of the construction (Serpentine Creek), adjacent areas likely to be affected by construction activities (Bramble Bay), or areas to be dredged to supply sand for landfill (Middle Banks). Other studies were aimed at providing detailed descriptions of the patterns of distribution and abundance of the epifauna and infauna at a range of spatial and temporal scales (Table 1).

These studies indicated the presence of a highly speciose fauna (e.g. Stephenson *et al.*, 1970), but also showed that the number of species varied considerably among different regions of the Bay (see also Campbell *et al.,* 1977b; Davie & Hooper, this volume). For instance, in their detailed examination of the subtidal infauna near Peel Island, Stephenson *et al.* (1974) listed 420 species collected in 0.1 m^2 van Veen grabs. Taxonomic difficulties excluded identification of some taxa (e.g. amphipod crustaceans), and also required use of a coarse sievwe (1.2 mm mesh) during initial processing of samples. It is likely that the total number of species in this relatively small area (ca. 3 km^2) would be much greater if all taxa were examined and a finer mesh-size (0.5 mm) were used. Only approximately 190 species were found, using identical methodology, in Bramble Bay (Stephenson *et al.*, 1976). Moreover, numerous sites in the Bay

Table 1. Studies which have examined the spatial and temporal dynamics of benthic macroinvertebrate assemblages in the Moreton Bay region. The list is not exhaustive and is meant as a guide into the available literature.

Reference	Study area	Focus of study	Spatial scales	Temporal scales
Davie (1986)* monthly	Serpentine Ck	Spatial & temporal patterns		> 1 km
Hailstone (1972)*	lower Brisbane R.	Patterns of distribution & abundance – descriptive	100s m	monthly
Hailstone (1976)	lower Brisbane R.	Pre-oil refinery construction	100s m	monthly
Poiner (1977)	Naval Reserve Bank	Small-scale spatial patterns	6 m	2 monthly
Poiner (1979)*	Sholl Bank	Spatial & temporal patterns	30 m	2 monthly
Poiner (1980)	Sholl Bank	Community patterns in sand & seagrass	30 m	2 monthly
Raphael (1974)*	Bramble Bay	Spatial & temporal patterns	ca. 1 km	3 monthly
Stephenson (1968)	sth Moreton Bay	Effects of flood on physical factors	> 1 km	n/a
Stephenson *et al.* (1970)	Moreton Bay[1]	Spatial patterns	1-2 kms	na
Stephenson *et al.* (1974)	Peel Is.	Seasonal & spatial patterns	10s m; 100s m; 1-2 kms	seasonal
Stephenson et al. (1976)	Bramble Bay	Pre-airport construction	1 km	seasonal; annual
Stephenson & Campell (1977)	Serpentine Ck	Pre-airport construction[2]	10-100m; 1 km	seasonal[3]
Stephenson & Cook (1977)	Bramble Bay & Middle Banks	Small-scale spatial & temporal patterns	5 m	2 monthly
Stephenson *et al*. (1977)	Bramble Bay	Effects of flooding	1 km	seasonal
Stephenson (1978)	Bramble Bay	Spatial periodicity	1 m	n/a
Stephenson *et al*. (1978)	Middle Banks	Pre-airport construction – dredging to provide sand	0.4-1 km	seasonal; annual
Stephenson (1980a)	Bramble Bay & Middle Banks	Temporal patterns	2 areas x 27 km apart	2 monthly; annual
Stephenson (1980b)	Bramble Bay & Middle Banks	Effects of abiotic & biotic factors	2 areas x 27 km apart	1-2 months
Stephenson (1980c)	Bramble Bay & Middle Banks	Temporal patterns	2 areas x 27 km apart	annual; > annual
Stephenson (1981)	Moreton Bay	Modelling of predation[4]	n/a	n/a
Stephenson & Sadacharan (1983)	Bramble Bay & Middle Banks	Small-scale spatial patterns	1 m	n/a

* denotes M.Sc. or Ph.D. thesis held in The University of Queensland library system;
n/a did not look at spatial and/or temporal scales;
1. this study examined 400 stations spread throughout the Bay region, south of Victoria Point to the northern extent of Moreton Island;
2. included the area which was reclaimed as part of the airport construction;
3. Sampling was planned to be seasonal, but difficulties prevented this on some occasions;
4. Simple population models for patchy annual predation were examined to determine whether these models could be used to account for large spatial and temporal hetergeneity in the benthic community.

were identified as being "faunistically poor" (Figure 4, Stephenson *et al.*, 1976), yet were in close proximity to other areas with greater numbers of species present. Different explanations were proposed for these patterns, including effects of physical factors (tidal scour and sediment instability) and disturbance (perturbations from prawn trawlers). Specific tests of most of these models still remain to be done. Although a close association between sediment parameters (e.g. grain size and sorting) and the structure of infaunal assemblages has often been argued as the most likely explanation for differences in the structure of benthic assemblages from place to place (e.g. Stephenson *et al.*, 1970, 1974), other equally viable explanations, such as patterns of larval supply influenced by hydrodynamics (reviewed by Butman, 1987), remain to be investigated in the Moreton Bay region.

There are some limitations in the interpretation of the data from these extensive surveys. Many of the early studies in Moreton Bay unfortunately confounded spatial and temporal variability, a result of the restrictive forms of numerical analyses available at the time, and the need to reduce multifactorial datasets to one or two dimensions. Thus, samples were pooled across times for some analyses and across sites for others (see the *Discussion* in Stephenson *et al.*, 1976). Frequently, replicate samples were also pooled to reduce the size of datasets with the result that variation at that scale could not be estimated (Andrew & Mapstone, 1987), confounding comparisons at larger scales (Hurlbert, 1984). Recent studies have demonstrated the importance of incorporating several levels of nested (hierarchical) sampling in monitoring programmes of soft-sediment infauna (e.g. Morrisey *et al.*, 1992a; b) in order to provide unconfounded comparisons at each of the spatial and temporal scales of interest. Temporal patterns were usually interpreted by Stephenson *et al.,* on the basis of seasonal or annual patterns, but these conclusions may be flawed by a lack of information about shorter-term fluctuations (Morrisey *et al.*, 1992b). It is now more widely accepted that patterns and processes can be better understood when variability is estimated at a number of spatial and temporal scales, and these scales are included within sampling designs.

Clearly though, the results of these studies suggest the potential for marked temporal fluctuations in the abundance and composition of the benthic infauna (Poiner, 1980; Stephenson, 1980a, b). As pointed out by Stephenson *et al.* (1974; 1976), this variability raises questions about the validity of environmental impact assessments based on limited temporal replication. Recently developed univariate statistical techniques (Underwood, 1991, 1992) have been developed in order to take into account such temporal (and spatial) variability at several different scales, allowing unequivocal interpretation of patterns at each of the levels of interest. These methods were developed to cope specifically with the sort of problems raised by Stephenson and his co-workers. It is now recognised that unambiguous assessments of environmental impacts can only be obtained using complex designs which partition the variation in the abundance of the biota among factors attributable to natural fluctuations in space and time versus those caused by the environmental disturbance under examination (Underwood, 1991, 1992; Keough & Mapstone, 1995). Furthermore, there are now several different multivariate statistical techniques available (e.g. Clarke, 1993) which allow probabilistic tests of hypotheses concerning the structure of assemblages, thus solving some of the problems faced by workers in the 1970s, problems which limited the capacity to make objective statements about groupings of species at different sites, or in different habitats (Stephenson *et al.*, 1974).

The regular and comprehensive sampling done by Stephenson and others during the 1970s also demonstrated fortuitously the advantages such datasets can provide, in understanding the relative importance of natural disturbances compared with anthropogenic impacts. The severe flooding followed by a cyclone in January 1974 caused dramatic reductions in the salinity of the western region of Moreton Bay, with subsequent effects on the benthic biota. These effects were only detectable because of the data which had been collected before the floods, admittedly in preparation for construction of the Airport extension, and which suggested that changes caused by the airport development were likely to be of a smaller magnitude than periodic episodes of flooding with the associated decreases in salinity and increases in turbidity.

There have also been a few studies which have examined the benthic invertebrates in the Brisbane River and its catchment. Hailstone (1976) collected 150 benthic dredge samples in the mouth of the Brisbane River in preparation for construction of the oil refineries in the Port Facility and recorded 130 species, with abundance overwhelmingly (78%) dominated by molluscs. The methodology used in this study was similar to that used by Stephenson *et al.* (1970) in the Moreton Bay region (using a benthic dredge, 1 mm sieve), with similar analyses of the data and consideration of spatial and temporal patterns. There have been no subsequent studies to determine whether construction of the port facilities caused changes to the composition of the benthic assemblages in the Brisbane River. Boesch (1977) examined the distribution of subtidal infauna along the Brisbane River in relation to gradients in salinity and concluded that classical schema of zonation developed for European estuaries were inappropriate for the Brisbane River estuarine system. He also noted the important effects of seasonal flooding on salinities and subsequent distribution of biota. The rich and diverse fauna in the catchment has attracted much attention in the past and Hailstone *et al.* (1978) list some 207 studies which examine the biology of the varied taxa. This extensive list needs updating and the creation and maintenance of a detailed bibliography covering the Moreton Bay region would serve a valuable purpose in directing research into areas and taxonomic groups where there are obvious gaps in available information.

Subtidal epifaunal invertebrates

The other major contributions towards our understanding of the ecology of subtidal benthic invertebrates have come from the work done by CSIRO and QDPI scientists, directed towards species of invertebrates of commercial importance, especially crabs and prawns. Young (1978) described in detail the distribution and abundance of the postlarval stages of penaeid prawns in relation to the physical environment throughout the Moreton Bay region, including subtidal and intertidal seagrass areas, and found that these habitats were important nursery areas for at least three species of prawns. Young & Wadley (1978) expanded this study to include another 400 species of (epi)benthic invertebrates and found that environmental influences identified as being important determinants of the distribution and abundance of the prawns were also correlated with the distribution of other components of the macrofaunal assemblage. Their sampling did not include consideration of the infauna in these areas.

Population studies

There have been detailed studies of individual species of subtidal invertebrates, in particular the commercially and recreationally important sand (*Portunus pelagicus*) and mud (*Scylla serrata*) crabs (Family Portunidae). There has also been extensive work done on various species of commercially important penaeid prawns, but these studies are discussed in detail elsewhere (Tibbetts & Connolly, this volume). The commercial sand crab fishery is Queensland's most valuable fishery (Matilda & Hill, 1981), and over 80% of this catch is derived from Moreton Bay (Quinn, 1992). This contrasts with less than 10% of the State's mud crab catch being taken from the Bay region (Quinn, 1992). Several other species of portunid crabs are also caught in the commercial pot and trap fishery in Moreton Bay, including the three-spot crab (*Portunus sanguinolentus*), the coral crab (*Charybdis cruciata*) and the rock crab (*Charybdis natator*) (Quinn, 1992).

Catch statistics for the pot/trap and trawl fishery for sand crabs have been described by Thomson (1951), Potter *et al.* (1987), Sumpton *et al.* (1989) and Weng (1992), while Smith & Sumpton (1989) have examined the catch efficiency of the traps in relation to behaviour of the sand crabs when foraging. Reproductive activity, moulting and growth information for sand crabs is provided in Sumpton *et al.,* (1989, 1994b), while incidence of parasitism has been described in Sumpton (1994) and Sumpton *et al.* (1989, 1994a). Moreover, as part of these surveys (from 1984-1986), information was obtained on the reproductive biology and growth of the rock crab, *Charybdis natator* (Sumpton, 1990a; b), and three-spot crab, *Portunus sanguinolentus* (Sumpton *et al.*, 1989), also harvested commercially in Moreton Bay. Campbell & Fielder (1986, 1988) provide data on the breeding cycles and egg development for *P. pelagicus*, *P. sanguinolentus* and *C. cruciata* (referred to as *C. feriatus* in that paper) and Fielder & Eales (1972) describes mating behaviour in *P. pelagicus*. More detailed studies on the implications of parasitic infections of sand crabs have been done by Phillips & Cannon (1978), Bishop & Cannon (1979) and Sumpton (1994).

Tagging studies (Potter *et al.*, 1987) showed that male sand crabs were generally recaught within 2 km of the release site, although the recapture rate was less than 14%. Insufficient data were obtained for female sand crabs to enable any conclusions to be drawn on their movements, including whether they make offshore migrations to spawn. Such offshore migrations by females are generally the case for portunid crabs around the world (Norse, 1977), and for estuarine *Portunus pelagicus* elsewhere in Australia (e.g. Potter *et al.*, 1983), although it has been assumed to not occur in Moreton Bay sand crabs (e.g. Weng, 1992), despite the lack of conclusive data. It also appears that female sand crabs may have different habitat requirements than males, with the females being more common on shallow banks, and males utilising deeper water (Potter *et al.*, 1987; Weng, 1992). The basis for these differences in habitat usage is unknown. Williams (1982) has described the diet of sand crabs in Moreton Bay, while Sumpton & Smith (1990) have examined the effects of temperature on feeding activity and patterns of emergence. Studies on animals discarded from prawn trawlers (by-catch) operating in Moreton Bay have indicated that the diet of sand crabs may be substantially supplemented by scavenging on this material (Wassenberg & Hill, 1987a).

Extensive work has also been done on the mud crab *Scylla serrata*. Heasman *et al.* (1985) described the reproductive cycle of mud crabs in southern Moreton Bay, based on examination of the ovaries from female crabs caught in commercial baited traps. They showed that the crabs mated throughout most of the year, with an extended spawning period from early spring through to early autumn. Peaks in recruitment occur during the warmest months of the year (Heasman & Fielder, 1977). Hill *et al.* (1982) found a clear spatial separation in the distribution of the three definable cohorts (Heasman & Fielder, 1977) of mud crab populations: adult crabs are primarily found in deeper waters, sub-adults are also found subtidally but move into the intertidal zone at high tide to feed, and juveniles are mostly resident in the intertidal areas along sheltered creeks and in mangroves. Craig (1989) described use of intertidal burrows by *Scylla serrata*, providing information on the vertical distribution of burrows in relation to tidal exposure, and utilisation of these burrows by crabs in relation to size and stage of the moult cycle. Occupation of the burrows was also examined in relation to tidal and seasonal cycles. Hyland *et al.* (1984) used tagging studies to examine the utilisation of different benthic habitats by mud crabs, in particular movements between mangroves and adjacent areas. Their results showed that post-recruitment migrations were limited, except in the case of ovigerous females which migrate offshore to spawn (Hill, 1994).

The incidence of offshore spawning in mud crabs requires that the larval stages of mud crabs are advected back into the inshore estuarine habitats where the adults are found. There is almost no information on the patterns of larval availability and subsequent recruitment into different benthic habitats within Moreton Bay (or elsewhere in Australia) for estuarine crabs. A plankton sampling programme in Pumicestone Passage from September 1980 to June 1981 captured very few mud crab megalopae, although sand crab megalopae were occasionally very abundant (Hill, 1982). Heasman & Fielder (1983) have described the complete larval development of *Scylla serrata* in the laboratory, estimating the time to reach the first crab stage at approximately 30 days, sufficient time for considerable dispersal to occur in the plankton. Williams & Hill (1982) have examined factors which affect the catchability of mud crabs in pots, and the usefulness of mark-recapture programmes for estimating population sizes.

The above discussion focused on macrofauna (animals retained on a 0.5 mm sieve). There is even less information available on the ecology of smaller benthic invertebrates, the meiofauna (animals retained on a 63-100 µm sieve). Recent work by Coull *et al.* (1995) has demonstrated the importance of meiofauna in the diets of estuarine fishes, including whiting, gobies, trumpeter and pony fish, in accordance with similar findings elsewhere in the world (e.g. Hicks & Coull, 1983). Walters & Moriarty (1993) have examined the effects of grazing by meiofauna on the dynamics of microbial communities in an intertidal seagrass bed and found large spatial and temporal variability in the trophic interactions between these groups. General descriptions of spatial and temporal variation in the structure of meiofaunal communities in Moreton Bay are lacking, in contrast to the recent extensive studies further north in tropical Queensland (e.g Alongi, 1987a; b). Meiofauna have also been found to be extremely useful in detecting the effects of pollution in the northern hemisphere (e.g. Warwick, 1988; Warwick *et al.*, 1988; 1990), but similar studies in Moreton Bay are hampered by inadequate taxonomic information and the lack of suitable keys.

Intertidal benthic invertebrates

There is less ecological information available on the ecology of intertidal invertebrates in the region, despite the prevalence of exposed mud- and sand-flats around Moreton Bay, and the relative ease of access for sampling in the intertidal zone compared with subtidal habitats. Moreover, the intertidal areas are the most accessible habitats to the general public, including fishers, bait-collectors, and bird watchers, yet we know little about the ecology of most of the invertebrates which occupy these habitats, nor how they may be affected by human activities.

As part of the studies for the Brisbane Airport extensions, Campbell *et al.* (1977a) examined the mangrove and saltmarsh areas surrounding Serpentine and Jackson's Creeks in 1972, and documented the occurrence of species in different habitats. Only qualitative information on abundances was obtained, although it was evident these areas supported large populations of some species. Samples were not processed through sieves, so it is likely that small specimens and species were missed. Stejskal & Chamberlain (1984) provided more detailed analysis of the spatial and temporal patterns of the intertidal macrobenthos in Bramble Bay in these areas. In this study, samples were processed across a 0.5 mm sieve compared with the 1.0 mm sieve which had been used for the subtidal surveys discussed above. Molluscs were the numerically dominant taxa in the samples at two of the three sites, the other being dominated by amphipod crustaceans. The samples were all characterised by large abundances of a few taxa, although a total of 143 species was identified.

Morgan & Hailstone (1986) found over 30 species of gastropods in the intertidal mangroves at Wellington Point and provided qualitative descriptions of their vertical distribution in relation to physical features of the habitats. Eertman & Hailstone (1988) described qualitatively the vertical distribution of epifauna on wooden jetty piles at Wellington Point and Dunwich. Neither of these studies included quantitative information on spatial and temporal variability in the fauna. Catterall & Poiner (1984; 1987) have described the molluscan fauna found on intertidal banks on the western shore of North Stradbroke Island, and provided estimates of population sizes in relation to susceptibility to impacts from human harvesting. Many of the larger species of molluscs found on the intertidal shores of Moreton Bay were gathered as a traditional food source by local indigenous people, although few of these species appear to be harvested today.

Hailstone & Stephenson (1961) made qualitative observations on the ecology of the common yabby, *Trypaea australiensis*, at sites near Wynnum on the western side of Moreton Bay and Dunwich on North Stradbroke Island. The yabby is exploited as bait for use by recreational fishers (see below). Their study included information on the types of habitat where yabbies are found, estimates of the number of burrows formed by each animal, and records of reproductive activity throughout the year. Vohra (1965) examined the ecology of the large mud-whelk, *Pyrazus ebeninus*, at two sites in Moreton Bay, including information on reproductive cycles, growth and physiological tolerances to salinity, temperature and desiccation.

There have been several detailed studies on intertidal crabs, especially those occupying mangroves. Snelling's (1959) study on the distribution of intertidal crabs in the Brisbane River showed a strong gradient in species richness along the river. This study also showed that the distribution of crabs varied vertically within the intertidal zone, with different species being more abundant at different heights on the shore. Moverley (1984) suggested that for some of the more common species of intertidal crabs along the Brisbane River, recruitment from the plankton may be restricted to the lower reaches of the estuary, whereas replenishment of populations further upstream was via adult immigration (Davie, 1990). If true, this has important ramifications for understanding the effects of environmental impacts in the estuarine system. The rates of recolonisation of disturbed habitats, and the effectiveness of rehabilitation programmes, would be substantially reduced if recruitment were not occurring via dispersal of planktonic propagules (Skilleter, 1995). There are no available data on rates of larval supply of crabs to different sections of the Brisbane River.

Human Impacts on Benthic Invertebrates

The aquatic habitats and associated biological assemblages in the Moreton Bay region are exposed to a wide range of human activities. McDonald & Brown (1992) provide an interesting perspective on the interactions between human activities and ecological processes in the region. They identify the fact that much of the economic activity on and around the Bay is closely dependent on the maintenance of ecological processes and, importantly, highlight the urgent need for an integrated approach to management and utilisation of all the resources, rather than a continuation of traditional, sectorially-driven decision making. The Moreton Bay Strategic Plan (Queensland Government, 1993) is an attempt to incorporate these principles into a legislative framework under the auspices of policies on ecologically sustainable development. McDonald & Brown (1992; see Table 1, p.82) identify activities which are directly or indirectly dependent on Moreton Bay, including the effects of consumptive and non-consumptive activities, and those which are associated with changes to aquatic habitats around the Bay. There have been few studies which have examined directly the environmental impacts of these activities on benthic invertebrates within the Moreton Bay region. Studies done elsewhere suggest that such research is required.

Effects of fishing: commercial

It has long been known that there are numerous impacts associated with commercial fishing, over and above those affecting the species being targeted (deGroot, 1984). Although attention in recent years has focused primarily on issues related to by-catch (review by Andrew & Pepperell, 1992), there has also been considerable concern about effects on marine benthic habitats as a result of damage caused by trawls, dredges and other fishing gear (Sainsbury *et al.,* 1993). In the Moreton Bay region, the prawn fishing industry uses beam trawls in the Brisbane River and otter trawls inside and outside the Bay, and modified otter trawls are also used in the offshore scallop fishery (Quinn, 1992). Both these gear-types have the potential to cause significant disturbance to the substrata and changes to the structure of benthic communities.

Impacts from fishing gear can be divided into five main categories.

1. Disturbance to the substratum (i.e. habitat) from the passage of gear such as otter boards, tickler chains, and beam-trawl and dredge roll-bars, across the bottom, or from the hydraulic jets used to liquefy the bottom sediments in front of scallop and clam dredges (Messieh *et al.*, 1991). This may modify features of the sediments (texture, chemistry, etc.) making them more or less suitable or attractive to the larvae of benthic organisms (Meadows & Campbell, 1972), with subsequent effects on rates of settlement and/or recruitment.
2. Disturbance to the upper-layers of sediment (up to 12 cm deep – Bergman & Hup, 1992) may expose infauna to predators and scavengers, attracted to the area after passage of the gear (e.g. Caddy, 1968). The gear may also cause damage to biotic structures which help bind the sediments, such as mats of animal tubes (Rhoads & Young, 1971; Bailey-Brock, 1984), vegetative root and rhizome material (Ginsburg & Lowenstam, 1958; Orth, 1977; Fonseca *et al.*, 1982) or microbial mats (Dade *et al.*, 1990; Madsen *et al.*, 1993), leading to destabilisation of the substratum (Grant, 1983). Destabilised sediments are more susceptible to erosion from currents, further exposing shallow-burrowing animals (Thrush *et al.*, 1996).
3. Direct mortality of epifaunal animals which are crushed or dislodged by the passage of the gear, especially fragile epibenthic fauna including soft- and hard-corals, sea-pens (Sainsbury, 1987), molluscs (Medcof & Bourne, 1964; Caddy, 1973; Rumohr & Krost, 1991), sponges (Sainsbury, 1987; Harris & Poiner, 1990; Hutchings, 1990) and echinoderms (Rumohr & Krost, 1991). Some of these organisms create substantial reefal habitat utilised by commercial species, so damage to and loss of these animals can also cause negative impacts on the fishery creating the disturbance (Sainsbury *et al.*, 1993). Moreton Bay supports a rich fauna of these organisms (Stephenson, 1970; Davie & Hooper, this volume).
4. Resuspension of sediments leading to increased turbidity (Jørgensen, 1980; Eleftheriou & Robinson, 1992; Rieman & Hoffman, 1991), decreases in O_2 content of the overlying water (Jørgensen, 1980; Rieman & Hoffman, 1991), and increased rates of benthic-pelagic nutrient fluxes which may stimulate production of phytoplankton, even leading to blooms of toxic algae (Fichez *et al.*, 1992). Decreases in water quality through increased turbidity have also been related to declining abundances of submerged aquatic vegetation, including seagrasses (Giesen *et al.*, 1991; Dennison *et al.*, 1993; Carter *et al.*, 1994). Resuspension of fine sediments can also have direct effects on assemblages of suspension-feeders, by reducing the quality of food (Arruda *et al.*, 1983) and possibly even causing smothering (Dodge & Vaisnys, 1977; Peterson, 1985; Peterson & Black,

1988). There is likely to be substantial spatial variability in these effects, given the gradients in sediment grain-size that exist across Moreton Bay (Stephenson *et al.*, 1970).

5. Resuspension of pollutants contained in sediments, leading to increased exchange between the sediments and the water of any toxic chemicals (Gabrielson & Lukatelich, 1985; Walker & O'Donnell, 1981). There may also be changes in the bioavailability of pollutants through exposure to aerobic conditions.

There is ample evidence of impacts on benthic communities from commercial fishing activities around the world (De Groot, 1984; Messieh *et al.*, 1991; Van Dolah *et al.*, 1991, Brylinski *et al.*, 1994). In Australia, several studies have examined impacts associated with the commercial fishing industry. In recent years, the CSIRO has undertaken studies in the northern Gulf of Carpentaria (e.g. Harris & Poiner, 1991) and on the North West Shelf (e.g. Sainsbury *et al.*, 1993) aimed at quantifying any effects attributable to the commercial fishing activities in those regions. Many of the techniques used in these studies were trialled in Moreton Bay (Poiner, 1993), but these detailed studies in tropical regions have not been repeated in the Bay. It is also highly doubtful that the results from such studies could, or should, be extrapolated to infer effects or lack thereof in the subtropical region of Queensland (e.g. see Underwood & Petraitis, 1993). As noted by Hutchings (1990), there are likely to be considerable differences in the dynamics of populations of invertebrates in Moreton Bay compared with tropical Australia (North West Shelf, Gulf of Carpentaria and Torres Strait) where most of the work on the effects of fishing have been done. The data available are inadequate to provide a reliable assessment of the long-term implications of any impacts in any of these areas (Hutchings, 1990), and there are no comparable studies which could be used to determine such effects in Moreton Bay.

Indirect effects on benthic communities from trawling may also arise as a consequence of changes to the structure of food webs, especially resulting from declines in populations of top predators such as finfish, crabs and prawns (Underwood, 1993). Benthic invertebrates are an important food supply for demersal species, yet our knowledge of the role of predators, including fish and crabs, in structuring assemblages of organisms in soft-sediments is far from complete (e.g. reviews by Peterson, 1979; Wilson, 1991). The long-term effects of changes to the abundance of predators, as a result of fishing, on the structure of benthic invertebrate communities are unknown. Furthermore, trawling has another direct effect on these predators: up to 33% of the diet of crabs such as *Portunus pelagicus* in Moreton Bay is derived from discards from prawn trawlers (Wassenberg & Hill, 1987a; 1989; 1990), and finfish are also prominent scavengers of this material. The extent to which this dietary bonus helps sustain populations of crabs and fish in Moreton Bay is unknown, although the implication is that populations of sand crabs (*Portunus pelagicus*) are maintained at greater levels than would otherwise occur (Wassenberg & Hill, 1987a). It has been suggested that material from by-catch makes an important contribution towards sustaining fish populations in the Baltic Sea (Rumohr & Krost, 1991), although this source of food appears to be less important for fish in Moreton Bay (Wassenberg & Hill, 1990).

Fishing may also affect the spatial heterogeneity of benthic communities, by concentrating predators into particular areas, such as the western and southern sections of Moreton Bay where trawling is more prevalent (Wassenberg & Hill, 1990). Portunid crabs are capable of making substantial migrations (e.g. Hines *et al.*, 1995), and Stephenson (1980a; c) suggested that mobile predators may contribute substantially to the large spatial patchiness of the benthic infauna. The activities of benthic infauna have a major role in determining the structure and distribution of sediments and sediment-bound nutrients, via bioturbation (e.g. Rhoads, 1967; Rhoads & Young, 1970; Brenchley, 1981) and deposit-feeding (e.g. Levinton, 1979; 1989) so conceivably changes in the structure of the benthic assemblages via localised increases in density of predators could have further indirect effects on these systems.

Effects of fishing: recreational

Australia's shallow coastal habitats are a major area for human recreation. The coast-line of Queensland, especially the south-east region, is particularly heavily used for amateur fishing, sailing, swimming and other activities. Near urban centres, very large numbers of people use the coast for family recreation. Amateur fishing has been estimated to be one of the largest industries in Queensland and yet there has been little recognition of this fact in terms of its social and economic significance (Zann, 1995).

Fishers may affect the numbers of fish directly and may also influence the abundance of plants and animals used as bait. In recent years, there has also been an increasing number of complaints that human foragers are completely denuding intertidal areas of all forms of life, not just species used as bait. Virtually no animals or plants are immune from the effects of human foraging activities (e.g. Underwood & Kennelly, 1990; Kingsford *et al.*, 1991). Animals not actually taken are often killed by the disturbance to their habitats - for example, by crowbars used to open crevices and ledges to reach hidden animals, trampling of organisms (Povey & Keough, 1991) or by the indirect effects of removing target species (Underwood, 1993).

In the Moreton Bay region, effects of recreational fishers on benthic invertebrates are primarily a result of the collection of animals for use as bait. The species targeted include yabbies (*Trypaea australiensis*), bloodworms (*Marphysa* sp.), beachworms (*Onuphis* sp.), pipis (*Plebidonax deltoides*) and soldier crabs (*Mictyris longicarpus*) (Quinn, 1992; pers. obs.). There are also at least 70 commercial operators who harvest yabbies, blood and beachworms from the Moreton Bay region. The impacts from the recreational and commercial groups are likely similar, although on a different scale, so the two are considered together.

WBM Oceanics (1993) examined baitworm harvesting near Fisherman Islands, as part of the Port of Brisbane expansion studies. Digging for the worms (*Marphysa* sp., Family Eunicidae) occurs primarily in areas with seagrass and involves disturbance and removal of extensive areas of vegetation, some of which may be later replaced. It was noted that the effects of bait digging on the intertidal assemblages are largely unknown, but could include both direct effects such as mortality of infauna and epifauna, and indirect effects including changes to the drainage of the intertidal habitats and loss of protective seagrass (WBM Oceanics, 1993). Surprisingly, given the lack of information, it was concluded that baitworm digging probably only resulted in short-term ecological impacts.

The sustainability of the fishery will depend on continued recruitment of worms into intertidal habitats, including those previously harvested, yet there are no data on patterns of recruitment of baitworms into disturbed or undisturbed areas. The potential for negative effects of seagrass loss and disturbance on recruitment of benthic infauna has been noted for other destructive methods harvesting invertebrates from vegetated habitats (Peterson *et al.*, 1987) and needs to be examined in relation to commercial and recreational harvesting for baitworms in the Bay. Hopper (1994) estimated that recovery of the seagrasses required at least 24 months, but no data were obtained on the recovery of the associated benthic assemblages.

The collection of yabbies is another intrusive practice with the potential for causing disturbance to benthic invertebrate assemblages. Yabbies are harvested using a "yabby-pump", a locally-invented tubular device that extracts a core of sediment, from which the animals are removed. There are at least four potential effects from exploitation of yabbies in the Moreton Bay region:

(1) depletion of the yabby populations themselves; (2) indirect effects on other organisms (by-catch) during the harvesting process; (3) damage to benthic habitats from the pumps; and (4) effects due to trampling of the substratum.

The potential for localised over-exploitation of yabby populations was recognised in the early 1960s (Hailstone & Stephenson, 1961), but there has been virtually no research on the sustainability of the recreational fishery. Constable (1995) used experiments in Moreton Bay to show that bait-harvesting using a yabby pump caused a localised and long-lasting effect on the substratum, reducing the number of thalassinid burrows and subsequent recruitment of *Trypaea australiensis*. Loss of, or reduction in, populations of yabbies from over-exploitation may have subsequent impacts on wading shorebirds, such as the endangered eastern curlew, *Numenius madagascariensis*, which preys on thalassinid shrimp (Constable, 1995).

Thalassinid shrimp, such as *Trypaea australiensis*, are known to play an important role in structuring soft-sediment assemblages. Their burrowing and tunnelling is a major form of bioturbation in marine sediments (Brenchley, 1981; Suchanek, 1983), contributes to constant reworking of the substratum (Posey, 1986) and aids in oxygenation and remineralisation processes (Hines & Jones, 1985; Kristensen *et al.*, 1985; Reise, 1985). The presence and activities of thalassinid shrimp have also been shown to influence significantly the abundance of meiofaunal and microbial communities (e.g. Bell & Woodin, 1984; Moriarty *et al.*, 1985; Branch & Pringle, 1987) and macrofaunal communities (Wynberg & Branch, 1994; Constable, 1995). There is a substantial by-catch generated during the harvesting of yabbies, including animals which are caught in the pump and brought to the surface of the substratum, animals which are trampled during harvesting (Wynberg & Branch, 1992; 1994; Constable, 1995), and yabbies which are either considered too small for use as bait or are left on the surface unharvested (Sturkie, 1996). Several studies have demonstrated the indirect effects on other fauna from harvesting of bait (Jackson & James, 1979; McLusky *et al.*, 1983; Wynberg & Branch, 1994; Constable, 1995).

There is little information available to assess the impacts associated with recreational or commercial harvesting of benthic invertebrates, such as beachworms and pipis, from sandy beaches along the east coast of Australia (James & Fairweather, 1995), although studies elsewhere suggest the potential for over-exploitation of these populations (e.g. Moses, 1990; Defeo & De-Alava, 1995). Many of the ecological studies that have been done on Australian sandy beaches are inadequate for the purposes of providing a quantitative assessment of the distribution and abundance of the biota or their responses to harvesting by humans (James & Fairweather, 1996).

Sand and gravel mining and dredging

There are several extractive industries which make use of the available resources within the Moreton Bay region, including sand and gravel mining for the building industry, and dredging of sub-fossil coral deposits for use in the production of cement and lime (O'Flynn & Thornton, 1990). The coral deposits around the south-western islands within the Bay (including Mud, Green and St Helena) have been extensively exploited, and most of the sand and gravel needed to support Brisbane's rapidly growing building industry has come from the Brisbane and Pine Rivers (Stephens, 1992). In Moreton Bay itself, the construction of the Brisbane Airport extensions required dredging of the Middle Banks region (Stephenson *et al.*, 1978; Poiner & Kennedy, 1984). There is also a long history, beginning in 1862, of dredging within the Brisbane and lower Bremer Rivers for navigational purposes (McLeod, 1978). Additionally, the development of the Brisbane Port facilities required extensive dredging of the mouth of the Brisbane River (Greenwood, 1993).

The effects of dredging on assemblages of benthic invertebrates within Moreton Bay and its surrounding catchments have not been addressed in any detailed manner. Several studies have provided pre-impact data from sites in the Moreton Bay region (e.g. Hailstone, 1976; Stephenson *et al.*, 1976; Stephenson & Campbell, 1977; Stephenson *et al.*, 1978) but few post-dredging studies have been done. Poiner & Kennedy (1984) took advantage of the pre-impact surveys from the Middle Banks region (Stephenson *et al.*, 1978) and re-analysed a subset of this larger dataset to provide direct comparisons, before and after, of an area subjected to extensive dredging. This study showed marked declines in numbers of species and total numbers of individuals in the areas which were dredged, but increases in these variables over the same time-period in nearby undredged areas. These undredged areas showed evidence of impacts arising from the associated sediment plume generated during dredging operations (Poiner & Kennedy, 1984), but without suitable 'control' or 'reference' sites, the magnitude of any such effects cannot be estimated.

Studies on the effects of dredging and similar extractive industries elsewhere have demonstrated clearly the potential for substantial impacts on benthic assemblages. Dredging of the substratum can expose toxic chemical and other pollutants which have accumulated in the sediments allowing these contaminants to be dispersed to other areas (Morton, 1977; Lau *et al.*, 1993). Dredging also changes the stability of the substratum (Messieh *et al.*, 1991), leading to increased rates of resuspension of fine sediments and subsequent increases in turbidity. The effects of increased turbidity have already been discussed above in relation to commercial fishing.

Resuspended sediments eventually settle back onto the substratum and may smother epifaunal and infaunal organisms. The distance over which the suspended particles are dispersed depends on prevailing hydrodynamic conditions (Dyer, 1979). Jones (1981), for example, reported substantial and wide-spread changes in the distribution of fine silt and clay in the sediments of Botany Bay (NSW) as a result of dredging, although re-analysis of her data (McGuinness, 1988) suggested these changes were not as pronounced as originally thought. Jones & Candy (1981) suggested that the differences in the composition of the fauna in dredged and undredged areas in Botany Bay were due primarily to these changes in the sediments, although few significant differences in abundance or species richness between the different areas were identified. Samples were only collected on one occasion though, and the possibility of bay-wide effects from dredging could not be assessed. This was acknowledged as a shortcoming in the study by the authors. Moreover, sediments were processed across a 1.0 mm sieve, so small species and juveniles of larger species would not have been sampled in this study.

Most of the studies which have dealt with the effects of sediment disturbance on soft-sediment communities have been done on relatively small scales (e.g. patches of defaunated sediments - Zajac & Whitlatch, 1982) and have been primarily interested in rates of recovery and the successional sequence involved in recolonisation (reviewed by Thistle, 1981), but there have been some which have dealt with the effects of smothering on infauna (Rhoads, 1974; Thistle, 1981; Wilson, 1981). Brenchley (1981) found that the impact of burial on infauna depended on the mobility of the animals (e.g. tube dwellers vs mobile burrowers) and their feeding type (suspension feeders vs deposit feeders). Similarly, Peterson (1985) found that suspension-feeding clams were more susceptible to mortality from burial than deposit-feeding clams. None of these studies specifically examined the effects of smothering from continuously deposited sediments on the survival of new recruits, the life history stage which may be most susceptible to the effects of such disturbances. Increased sedimentation as a result of dredging has been implicated in declines in the abundance of corals in Bermuda (Dodge & Vaisnys, 1977; also

Rice & Hunter, 1992).

The immediate effect of dredging is an almost total loss of benthic organisms from the dredged area, although mobile species may be able to survive entrainment (Morton, 1977). Wainwright *et al.* (1992) modelled the loss of commercially important Dungeness crabs, *Cancer magister*, due to entrainment in dredge equipment in Grays Harbour, Washington, and found some types of dredge had the potential to cause marked impacts on the fisheries. The direct impacts on populations of Dungeness crabs, and the added indirect effects due to loss of suitable habitat for recruitment, has resulted in the need for expensive mitigation programmes to be implemented (e.g. Dumbauld *et al.*, 1993).

The fauna in surrounding areas may also be affected due to slumping of material into the holes and pits created by dredging (Poiner & Kennedy, 1984; Van Dolah *et al.*, 1984). This can lead to increased rates of mortality as animals are exposed to predators and scavengers (Skilleter, 1996a). Subsequent recolonisation of the benthic habitats will depend on the nature of the substratum in the area after dredging. Removal of the upper layers of sediment can expose a different sediment-type underneath resulting in an entirely different benthic assemblage establishing itself (De Groot, 1979; Kenny & Rees, 1994). The time-scale for recolonisation would be dependent upon factors such as the season when the disturbance occurred (i.e. whether propagules are present in the water – e.g. Kennelly, 1987), and local hydrodynamics (whether larvae can settle successfully into the new habitats – e.g. Butman, 1987).

Dredging has also been implicated in substantive impacts on estuarine seagrass beds (Giesen *et al.*, 1990; Larkum & West, 1990; Onuf, 1994). Given the close associations between seagrasses and the associated fauna (e.g. Bell & Westoby, 1986; Bell & Pollard, 1989), it is likely that loss of the vegetated habitats would have large-scale effects on benthic invertebrate assemblages, although this has rarely been investigated experimentally (but see Peterson *et al.*, 1987). Declines in the extent of seagrasses in Moreton Bay have been associated with declining water quality (Abal *et al.*, this volume), but impacts on the benthic invertebrates as a response to these changes in abundance of the seagrasses have not been examined.

Loss and modification of aquatic habitats

The rapid growth of Brisbane since the 1950s (e.g. Fisher, 1992), has placed enormous pressure on the land around the Bay for development of industries, marinas and urban settlements. This growth has been accompanied by substantial losses of critical estuarine wetlands which have been reclaimed to allow construction and development (Greenwood, 1993). Greenwood & Morton (1992) estimated that 65% of all canal developments in Australia were in Moreton Bay, frequently in sections of estuaries previously dominated by seagrasses, mangroves and saltmarshes. By 1987, over 20% of the total area of mangroves around Moreton Bay had been lost to development (Greenwood, 1993). The implications for such widespread modifications to the natural habitats in the Bay region on assemblages of benthic invertebrates (and other fauna) are poorly understood, but such problems have been highlighted as a key issue for achieving sustainable growth in Australia.

In recent years, Australian policy on management of natural resources has shifted, in line with global trends, to a focus on the principles of ecologically sustainable development (e.g. Commonwealth of Australia, 1992). Australia has adopted, along with the other guiding principles, the concept of "maintenance of ecological systems and the protection of biodiversity" (Commonwealth of Australia, 1990). Any programme of effective fisheries management intending to meet the goals of ecologically sustainable development must provide an accord between numerous external influences including urban and rural sources of pollution, needs for oil, gas and mineral exploration and mining in the coastal and offshore environment, and the increasing needs of the tourism industry (Gilmour & Connor, 1991). For fisheries, it has been recognised that sustainability be interpreted in a broad context, applying not only to operations within the commercial sector, but also to subsistence and recreational activities, and it must incorporate the protection and conservation of the ecosystems in which the fisheries are found. The Fisheries ESD Working Group considered the incorporation of an ecosystem-based framework as an essential component of each of the other ESD principles (Commonwealth of Australia, 1991).

To achieve this goal, fisheries agencies have increasingly concentrated on the requirement to protect and preserve aquatic habitats considered to be critical in supporting long-term productivity of particular fisheries (Poiner *et al.*, 1993). In coastal and estuarine environments, the importance of particular habitats such as mangroves and seagrasses as nursery areas for commercially and recreationally important species of fish and crustaceans has long been accepted (reviewed by Bell & Pollard, 1989). This paradigm, with the associated attempts at economic evaluation (Ruitenbeek, 1994), has been the basis for providing legislative protection of these habitats against continued loss from development-related activities (Adam, 1992; Skilleter, 1996b), and has provided impetus for attempts at rehabilitation of damaged areas (Blair, 1991; Moy & Levin, 1991; Pollard & Hannan, 1994).

The implicit assumption behind present conservation measures is that all areas of habitat are equally suitable and valuable as nurseries and therefore deserve equal protection. This assumption is evident in studies which have examined a limited number of habitats and/or sites, found them to have a role as a nursery habitat, and then extrapolated to a call for general protection of that class of habitat (e.g. Bell *et al.*, 1984; Middleton *et al.*, 1984; Laegdsgaard & Johnson, 1995). While this may be an appropriate conclusion to draw within a framework invoking the precautionary principle for environmental protection (Commonwealth of Australia, 1992), it is unsatisfactory from a rigorous scientific standing (Peterson, 1993) and from a responsible management perspective.

In fact, there are substantial gaps in our knowledge of how and why different estuarine habitats are utilised and how varying habitat quality affects processes such as recruitment for many (if not most) species of invertebrates. The assumption that all sites containing a particular type of habitat are of equal value as a nursery, and therefore of equal value for conservation, is not supported by the available evidence. For example, it has become increasingly apparent in recent years that some sites within an estuary receive consistently greater numbers of recruits than other sites, despite there being few obvious physical differences, other than actual position, between these places (e.g. McNeill *et al.*, 1992). The fact that these sites receive greater than average recruitment is of little use in ascribing the relative worth of these sites because there is insufficient information about subsequent survival of the animals once they reach a particular habitat (but see Peterson & Summerson, 1992; Eggleston & Armstrong, 1995).

Extensive research has now demonstrated the close relationship between the distribution of commercially important groups such as prawns and the distribution of specific features of the environment, such as seagrasses and mangroves (e.g. Staples *et al.*, 1985; Bell & Pollard, 1989; Coles *et al.*, 1993; Halliday, 1995), yet the specific nature of the relationships is for the most part unclear (Poiner *et al.*, 1993). For example, loss of seagrass due to natural physical disturbances such as cyclones, or unexplained "die-back" can sometimes lead to localised decreases in prawn-catch (Poiner *et al.*, 1993), but in other cases is associated with significant increases in the abundance of some species of prawn (Halliday, 1995). Poiner *et al.* (1993) made the valid point that knowledge of the factors which make habitats suitable for the organisms utilising them is as important as information on which of the available habitats are being used. This can only be achieved with new research aimed at understanding the specific nature of how and why certain features and characteristics of habitats are important to benthic organisms.

It is also important to remember that estuarine systems exist as a mosaic of habitats, including saltmarshes, mangroves, seagrasses and unvegetated substrata, not as isolated patches separated from others. Many of the species utilising these habitats are highly mobile and could easily move between multiple habitats regularly or during the course of their life cycle. There is little information about the linkages between habitats, or how the proximity of one habitat to another affects processes occurring in either habitat (Irlandi, 1994). Studies on estuarine habitats must examine these components simultaneously, rather than only examining one or other of them in a system comprised of many linked elements (e.g. Peterson, 1990). Until there is a greater understanding of the nature of such linkages between habitats, it is premature and ill-considered to pass judgements that suggest certain types of habitat are "beneficial to an estuary" or would "benefit the ecology of [an estuary]" (Morton, 1993) purely on the basis of an expected enhancement to commercial and recreational fisheries. Increases in abundance, for instance the enhancement of abundances of fish around artificial reefs (e.g. Bohnsack & Sutherland, 1985; Ambrose & Swarbrick, 1989) still comprise an environmental impact because the structure of the aquatic assemblages have been modified compared with reference areas (Underwood, 1991; 1992).

These gaps in the current understanding of the functioning of estuarine systems are even more prevalent for the non-commercial species of invertebrates, the greater proportion of which are dependent on the estuary for some or all of their life cycle. The benthic habitats within the Moreton Bay region are well known for their role in supporting numerous commercial fisheries (Blaber & Blaber, 1980), including fisheries for penaeid prawns, portunid crabs and various finfish, in addition to smaller fisheries for other groups such as oysters, scallops, and squid (Quinn, 1992). Many species of finfish (e.g. Blaber & Blaber, 1980; Morton, 1990; Morton *et al.*, 1987), crabs (Hill, 1976; Williams, 1982) and prawns (Ruello, 1973; Stephenson, 1980; Robertson, 1988; Wassenberg & Hill, 1987b) utilise benthic macroinvertebrates as an important component of their diet. The consequences for Moreton Bay fisheries which might arise as a result of impacts affecting the benthic communities are essentially unknown (Hutchings, 1990), but are likely to be marked. Hutchings (1990) reports findings from work in seagrass beds in temperate Western Australia which showed the almost complete loss of benthic infauna as a consequence of the decline in seagrass biomass. Presumably, such changes in the benthic community would cascade through to higher trophic levels, but detailed research is needed before such effects could be predicted or managed (Hutchings, 1990).

Effects of pollution

Moss *et al.* (1992) identified the major sources of pollution into Moreton Bay, separating them into shoreline (point-source) discharges and diffuse (catchment-based) inputs. A large number of sewage treatment plants (STPs) discharge secondarily treated sewage into the Brisbane River and Moreton Bay. Apart from the organic component, the effluent also contains increased concentrations of nutrients (nitrogen and phosphorous), heavy metals and pesticides (Moss *et al.*, 1992). Brisbane River also receives effluent from two oil refineries (Mackey *et al.*, 1992).

Mackey *et al.* (1992) have shown there are elevated concentrations of several heavy metals in mangrove sediments near the mouth of the Brisbane River, but there are no data on which to determine the effects on the biological assemblages (Mackey & Hodgkinson, 1995). Elevated concentrations of zinc, copper and cadmium have been detected in the tissues of bivalves from the mouth of the river, although concentrations were not sufficiently high to be of concern to human health (Wallace & Moss, 1978). Concentrations of heavy metals sufficient to cause damage to aquatic systems have, however, been recorded in the Logan River (Moss *et al.*, 1992), although the nature of any effects on communities of benthic invertebrates were not indicated. Despite the potential for substantial effects of heavy metal pollutants on benthic communities in the Moreton Bay region, no detailed studies have been done to determine whether such impacts have occurred, and if so, to quantify their magnitude. Morrisey *et al.* (1996) have shown experimentally in Botany Bay (NSW), that increased concentrations of copper in sediments affects benthic invertebrates at the population and community level, although the magnitude of the impacts varied spatially and among different taxa.

Petroleum hydrocarbons have also been detected in the sediments and water at the mouth of the Brisbane River (Connell, 1974), although treatment of sewage from the Luggage Point treatment works is believed to have reduced the incidence of this contamination (Moss *et al.*, 1992). High concentrations of polyaromatic hydrocarbons (PAHs) are also known to occur in riverine sediments (Connell & Bycroft, 1990). Again, there have been no detailed studies to determine whether these contaminants have had an effect on benthic communities, despite the fact that petroleum hydrocarbons are known to affect a wide range of marine organisms (Miller, 1982), especially crustaceans and bivalves (Axiak & George, 1987; Axiak *et al.*, 1988; Miller, 1982). McGuinness (1990) has shown that weathered fuel oil can have short-term impacts on mangrove gastropods, although there were few residual effects. Similarly, Clarke & Ward (1994) showed that weathered petroleum hydrocarbons caused high rates of initial mortality of gastropods in saltmarsh habitats, although plots were rapidly recolonised. No similar studies have been done in saltmarsh or mangrove habitats in the Moreton Bay region.

The overall impact of petroleum hydrocarbons in aquatic environments may depend, to some extent, on the degree of microbial activity and subsequent breakdown of these products (Leahy & Colwell, 1990). The activity of micro-organisms represents one of the primary mechanisms by which petroleum and other hydrocarbon pollutants can be eliminated from the environment. In coastal environments, there is a wide range of microhabitats where microbial activity takes place (DeLaune *et al.*, 1990), and factors such as oxygen content, pH, temperature, salinity and nutrient status can vary considerably on small spatial scales. All these factors affect rates of microbial activity, so the considerable variation in these factors makes it extremely difficult to predict the rates of microbial breakdown of petroleum pollutants in different places and hence the long-term impacts on the surrounding biota based on studies done elsewhere.

Future Work

It is clearly evident that there are major gaps in our knowledge of the ecology of benthic invertebrates in the Moreton Bay region. Given the range of potential environmental disturbances that could impact on the benthic assemblages, there is an urgent need for focused ecological research aimed at understanding the ecological processes which determine the patterns of distribution, abundance and diversity of this biota. Sustainable management of coastal resources must be based on a fundamental awareness of these ecological processes and how they are affected by human activities. New sampling programmes for subtidal and intertidal benthic invertebrates in Moreton Bay are required, revisiting some of the original sites (or at least in their vicinity) sampled in the 1970s, and utilising more detailed hierarchical sampling programmes, to determine whether the conclusions drawn previously with respect to spatial and temporal patterns are supported by more rigorous analyses. These new studies would also provide an essential database which could be used to design appropriate sampling strategies to detect environmental impacts which may occur in the region. Because the developments of the airport and the Port of Brisbane occurred subsequent to the studies of Stephenson and others, it is unrealistic to expect that data from these studies would provide any reliable picture of the present distribution and abundance of benthic invertebrates in the Moreton Bay region.

There is also a need for experimental studies investigating the effects of specific forms of anthropogenic disturbance in the Bay, particularly impacts from recreational and commercial fishing and the effects of a multitude of pollutants entering the catchment. Detailed information on the responses of populations and assemblages to these environmental perturbations will provide conservation and management agencies with a much better scientific and ecological basis to make the necessary decisions to preserve the unique habitats and their occupants of the Moreton Bay region for future generations.

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The Nekton of Moreton Bay

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Abstract

Nekton are a colourful and conspicuous component of the fauna of Moreton Bay, and form the basis of important recreational and commercial fisheries. This paper is about nekton (fish, prawns and cephalopods) distributional patterns and ecological processes, and human impacts on them.

At a broad scale, the nektonic fauna of the western sector of the Bay, where there is high turbidity and variable salinity, differs from that of the eastern sector which is more oceanic and has clearer water. The relation of nekton distribution to soft sediment habitats such as seagrass, mangroves and saltmarsh creeks is known from numerous surveys. Hard substratum habitats, such as rocky and coral reefs, and pelagic waters, have been sampled far less frequently, and only very general data about species occurrence have been recorded. By determining patterns in nekton distribution, inferences have been made, especially relating to important economic species, about ecological processes such as spawning, settlement, recruitment, dispersal, migration, predation, competition, growth, mortality and energy transfer. Manipulative experimentation to clarify such processes has been undertaken only very rarely in Moreton Bay. Severe methodological constraints limit the strength of conclusions that can be made about nekton patterns and processes. More care is needed in the selection and description of sampling processes.

Recorded effects of human impacts on nekton of Moreton Bay include: changes in fish abundance due to fishing, changes in assemblage composition due to habitat modifications such as the development of canal estates, and tainting of fish by pollutants. Likely impacts on nekton of other activities such as dredging, changes in catchment land-use and collecting for the aquarium trade are undocumented.

Introduction

Undoubtedly the nekton comprise the most colourful, commonly sought after marine resource in Moreton Bay. They provide a vital economic resource for the commercial and recreational fishing industries (Quinn, 1993c), an aesthetic resource for the tourism and diving industries, and fascinating subject matter for scientists and students. They provide food for charismatic megafauna (e.g. dolphins), possibly regulate community structure in the plankton and benthos, and are intimately linked with many other ecological processes within the Bay.

Intimate knowledge of natural cycles of migration is held by Bay-side aboriginal communities, for whom marine products are of considerable social and economic value (Walters, 1986). Early chroniclers of the region's history provided accounts of the abundance of nekton resources of the Bay and its rivers, and even then made reference to changes in the structure of populations and communities of nekton as a result of fishing (Welsby, 1905).

Recent examination of catch records indicate that a decline in catch per unit effort (CPUE) and fish size has occurred over the past fifty years (Thwaites & Williams, 1994). There is little doubt that loss of habitat is a contributing factor in the modification of nekton community structure. Young (1978) and others have indicated the importance of littoral areas, particularly mangroves and seagrasses to the health of estuarine nekton. Urbanisation, construction of canal estates and their aesthetically enhanced foreshores, the extension of Brisbane Airport, and now the massive redevelopment of the Port of Brisbane have all added to the assault on the nekton through eutrophication, pollution and fishing to change the character of the nekton communities of Moreton Bay.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 395-420.

What are nekton?

In discussing the nekton of Moreton Bay and catchment, we are first faced with the problem of achieving a workable definition of nekton to avoid areas of overlap or gaps between this and other reviews. Nekton (from the Greek *nekton*, neuter of *nektos*, meaning swimming) have been variously defined as:

1. "Pelagic animals, such as adult squids, fish and mammals, that are active swimmers to the extent that they can determine their position in the ocean by swimming" (Summerhayes & Thorpe, 1996);
2. "...compris[ing] the more powerful swimming animals, vertebrates and cephalopods, which are capable of travelling from one place to another independently of the flow of water" (Tait, 1968);
3. "... hav[ing] sufficient powers of locomotion to make their way against currents" (Marshall, 1979); and
4. "Animals, such as fishes, which can maintain their position and move against local currents..." (Parsons *et al*., 1984).

Many animals either fall into or out of these definitions as a function of daily cycles in activity, ontogeny (e.g. changes in body size or the development of swimming appendages with growth), reproductive cycles (e.g. spawning migrations), or under the influence of particular stimuli (e.g. fright). Many, such as fish, prawns and crabs have a planktonic larval dispersal stage, whilst others may dwell in burrows for much of their life and can be considered part of the benthos (e.g. alpheid shrimps; blind gobies, *Trypauchen*). Most adult prawns (Penaeidae) burrow in the sediment during the day (i.e. are benthic) but also are capable of swimming against moderate currents and of very fast swimming during escape responses (i.e. may be considered nekton). Some fish are amphibious (e.g. mudskippers), and some terrestrial vertebrates enter estuarine and coastal waters, are capable of swimming underwater against currents and so could be considered nektonic (e.g. varanid lizards, crocodiles, water fowl, seabirds).

In this paper nekton are defined as: aquatic animals which usually dwell beneath the water's surface, and which for all or a part of their life, are capable of either maintaining or modifying their position in the water column with respect to the substrate, against a horizontal current of at least 0.1 m/s. The assignment of that particular current velocity is purely arbitrary and knowledge of the locomotory abilities of various taxa in the Moreton Bay marine communities is almost nil. Assignment of various taxa to the 'nekton' is based merely on the general knowledge of the authors on the relative locomotory capacities of different groups.

Accepting the above, the following groups are included in this account: juvenile and adult fish, juvenile and adult prawns, and cephalopods. Organisms which fall within the definition but are excluded from consideration because information on them occurs elsewhere are: swimming crabs, reptiles, birds and mammals. Organisms that fall outside of the definition but might be considered nekton by some, because of their taxonomic affinities include: larval fish, prawns and crabs, and fish or crustaceans that spend their lives in burrows.

Pattern, process and human impacts

Fisheries in the Bay have been well described (Williams, 1980; Stephenson & Williams, 1981; Pollock & Williams, 1983a; b; Warburton & Blaber, 1992), therefore this paper centres on the ecology of nekton. The three main foci of this review are: (1) a review of the extent of biological data available on Moreton Bay nekton; (2) an examination of survey work for indications of consistent patterns and processes within the nekton communities of Moreton

Bay; and, (3) an assessment of our current understanding of human impacts on the nekton. A knowledge of ecological processes which drive the observed community patterns and mediate the relationships between nekton and other groups (e.g. plankton and benthos) is particularly important. The present status of knowledge of the taxonomy and biogeography of the nekton is described in Davie & Hooper (this volume).

Patterns in Nekton Communities in Moreton Bay

Most detailed knowledge of the biology of Moreton Bay nekton has been found while addressing management questions flowing from the high economic worth of prawns and fish. Commercially and recreationally important species have received most research effort, with relatively little effort expended on species of little economic, but possibly major ecological importance (e.g. planktivores). There has also been a bias toward the investigation of the biology and autecology of the adult phase of nekton with relatively little knowledge sought about larvae and juveniles. Summaries of the biology of economically important species are given in Table 1 and by Quinn (1993c).

The nekton communities of Moreton Bay can be broadly divided into three: those on sedimentary habitats, such as sand or mud intertidal and subtidal areas; those over hard substrates, such as coral and rocky reefs and shores; and, those in the water column with no known specific substrate requirement. To each habitat group various subjective divisions can be applied, and structurally similar habitats at different locations within the Bay and catchment may be subject to different hydrodynamic and water quality conditions, resulting in site-specific characteristics within the nekton population.

Spatial patterns

Nekton communities in Moreton Bay may be divided into several different spatial entities. A survey by Young & Wadley (1979) of epibenthic fauna (containing many nektonic taxa) from shallow areas of Moreton Bay detected two main faunal assemblages: one to the east of line between Toorbul Point and Dunwich which they termed “more marine”, and the other to the west which they described as “less marine”. Burgess (1980) investigated demersal assemblages in the Bay and determined three species groupings characteristic of the western, central and eastern Bay. Stephenson & Dredge (1976) found the greatest numbers of fishes at stations in the upper estuary. In contrast, Quinn (1980) found greatest abundances in the lower estuary. These differences may be a result of the different years in which the studies were conducted and the different gear used in each. Weng (1990) investigated five areas, all less than 4 m depth, recording a total of 86 species, with 29 of the most abundant species classified as estuarine. He found that sites near a river mouth (Caboolture River) and a sewage outfall (Luggage Point) had the highest numerical abundance and lowest species richness, and were faunistically very similar. This finding is not surprising, as the sewage outfall is also at the mouth of a river. Weng also found distinctive assemblages at a sand drift site (south Bribie Island), seagrass site (Toorbul) and mangrove site (Deception Bay). Warburton & Blaber (1992) studied shallow water (5-8 m) assemblages by beam trawling seven sites in western Moreton Bay and found that vegetated sites supported greater abundance in four of the five most abundant species. The pattern of high abundance at vegetated sites found by Warburton & Blaber (1992) is in agreement with other studies of the Bay (Young & Wadley, 1979; Blaber & Blaber, 1980; Courtney *et al*., 1995; and see Hill & Wassenberg, 1993).

Table 1. Nekton for which there is some biological or ecological knowledge. Economic status is given as CF = commercial fishery; MCF = major commercial fishery, MRF = major recreational fishery, RF = recreational fishery; Trophic level is given as DET = detritivore, HE = herbivore, MIC = microcarnivore, MAC = macrocarnivore; Adult habitat is given as B = Bay, C = coastal waters; E = estuaries, and R = rivers; Spawning habitat is given as C = coastal waters, E = estuary mouth, P = pelagic, SB = sand bars, SG = seagrass beds.
Note: There is insufficient knowledge of larval biology and status of the fishery for most species.

Species	**Common name**	**Economic status**	**Trophic level**	**Adult habitat**	**Spawning habitat**	**Nursery habitat**
Mugil cephalus	striped mullet	MCF	HE, DET	R, E, C	C	EM
Acanthopagrus australis	yellowfin bream	MCF, MRF	MAC	R, E, C	SB	SG
Sillago ciliata	summer whiting	MCF, MRF	MIC	E, C	SB	EM, CM
S. maculata	winter whiting	CF, RF	MIC	E	E?	EM
S. analis	golden-lined whiting	CF, MRF	MIC	E	E?	
S. bassensis	diver whiting	CF, MRF	MIC	C	SB?	
Platycephalus fuscus	mud flathead	CF, MRF	MAC	E	SB?	
P. arenarius	sand flathead	CF, RF	MAC	C	SB?	
Hyporhamphus, Hemiramphus and *Arrhamphus*	garfish	CF, RF	HE	R, E, C	SG	
Pomatomus saltatrix	tailor	CF, MRF	MAC	C	C	
Pagrus auratus	snapper					
Metapenaeus bennettae	greasyback prawn	RF, MRF	DET	E	E	
M. macleayi	school prawn	RF, MCF	DET	E	E/C	SG
Penaeus plebejus	eastern King prawn	RF, MCF	DET	B	C	SG
P. esculentus	tiger prawn	RF, MCF	HE	C	C	SG
Cephalopods	squid	RF, CF	MAC	B,C	C-P	?

Structure and bottom topography are also important. Dredge & Young (1977), in a study of Middle Banks (north eastern Moreton Bay), found only low numbers and diversity of fishes from the highly mobile sand banks. They did note however, that prawns were in higher abundance in gutters and that fishes were more common around topographic features such as rocky ledges. This finding highlights our lack of knowledge of fine-scale patterns in demersal assemblages, a problem that has been overcome in other systems (e.g. Jervis Bay) by the application of video transect techniques. However, predictions of community structure cannot be made merely on the presence or absence of structure. For example, Young & Wadley (1979) noted that their data supported Young's (1978) finding that the distribution of four species of juvenile prawn was mainly driven by distribution of temperature and salinity in the Bay. Furthermore, the spatial distribution of the epibenthic fauna (including nekton) is complex. Young & Wadley (1979) wrote:

> "In view of the many inter-correlated features of the littoral and infralittoral environment, it was difficult to partition the contrasting features responsible for the observed geographical differences... ...it was apparent that the degree of marine conditions had an overriding influence upon the composition of the fauna". They went on, "within each marine-influenced area significant differences were apparent between fauna of shallowest and deepest stations, and the fauna from the same seagrass community was usually only similar if the stations were sited in areas of similar salinity and temperature conditions...(and)... In areas of less marine influence (the fauna) was usually undifferentiable between stations of different depths or the presence or absence of seagrasses".

From this they deduced that density-independent factors (degree of marine influence on temperature and salinity) may control population densities by regulating density-dependent effects (e.g. the utilization of seagrasses).

Temporal patterns

A frequently reported feature of Morton Bay nekton assemblages is that species richness and abundance are at a maximum in summer and a minimum in winter (Quinn, 1980; Young & Wadley, 1979; Blaber & Blaber, 1980; Burgess, 1980; Morton *et al.,* 1988). Annual patterns within the Bay appear to involve a large influx of fish in the spring and summer, with numbers and diversity at a maximum in most habitats studied (Young & Wadley, 1979; Blaber & Blaber, 1980; Tibbetts, 1991; Warburton & Blaber, 1992). Most species breed during the warmer months, with only a few showing increase, and presumed breeding activity, in the cooler months. Quinn's (1980) analysis of fishes in Serpentine Creek found that species richness and abundance peaked in April and May. He found significant correlations between abundance and both temperature and salinity. Jones (1986) found *Sepia esculenta* (cuttlefish) (September - January most abundant) and *Loligo etheridgi* (squid) (January - June most abundant) to dominate the cephalod component of the community sampled by trawls. Of the penaeids in his samples, he found abundance peaks for *Penaeus esculentus* (tiger prawn) in June, *P. plebejus* (eastern king prawn) from November to January and *Metapenaeus endeavouri* (endeavour prawn) from March to April. Stephenson *et al.* (1982b) reported that few of the 60 species analysed had spring maxima, most had summer maxima. They found annual cycles were more pronounced at the more marine sites. Stephenson (1980) and Stephenson *et al.* (1982b) suggested that based on earlier studies of patterns of abundance in the benthos (Skilleter, this volume), this pattern of abundance in the nekton agreed with the hypothesis that the benthos is largely controlled by the nekto-benthos (*viz.* epibenthos) and weather. However, there is the converse view that patterns of production in the benthos control the diversity and abundance of nekton. For example, Jones (1986) considered that seasonal patterns of depauperation in Moreton Bay may be a function of nutrient supply during flood events. Jones compared Moreton Bay with the situation in the Gulf of Mexico (similar latitude) and found that there was no winter depauperation, rather different species dominate in different seasons. A third alternative explaining the seasonal decrease in the nekton has been suggested by Warburton & Blaber (1992), who found strong seasonality of *Pelates quadrilineatus* (trumpeter perch) and *Leiognathus mortonensis* (ponyfish) but that maximal abundances occurred at different seasons than in other studies (e.g. Young & Wadley, 1979; Blaber & Blaber, 1980; Young, 1981). They went on to note that Blaber & Wassenberg (1989) had found that piscivorous birds showed little seasonal variation in consumption of *P. quadrilineatus*. From these data Warburton & Blaber (1992) proposed that the differences they found for both *P. quadrilineatus* and *L. mortonensis* reflect variation in recruitment timing and the location and selectivity of sampling. Indeed, Jones (1986) had suggested that "the dynamic nature of the nektobenthic community is demonstrated by the significant interactions between time and space". Young & Wadley (1979) found that only in two of nine locations did temporal change in species composition exceed inter-station differences. A consequence of their finding is that areas of superficially similar habitat (e.g. *Avicennia marina* mangrove forest) may support different communities and act as nurseries to a different extent. Thus to call all mangrove systems important is not enough, either for fisheries biologists or ecologists. Hydrographic conditions may influence larval fish supply or food availability. Furthermore, annual variation in nutrient supply as proposed by Jones (1986), may contradict the model proposed by Brewer *et al.,* (1995) that food is not limiting in tropical systems, and that in the absence of adequate food supply, organisms either die or move to more productive habitats.

Moreton Bay as a nursery for nekton

It is widely accepted that the shallow, sheltered, turbid waters of Moreton Bay provide an important nursery habitat for juvenile nekton (Blaber & Blaber, 1980; Laegdsgaard & Johnson, 1995). Weng (1990) reported that shallow sites in the western Bay, whilst important as nursery sites for estuarine species, were of relatively little importance for the juveniles of marine species. Laegdsgaard & Johnson (1995) studied two mangrove-lined shores in western Moreton Bay using seine nets and small set nets with small mesh size to estimate the relative importance of seagrass, mangrove and mud habitats as nursery areas for juvenile fishes. They found the seagrass community to be distinct from both mud and mangrove habitats, and that seagrass was of relatively little value as a nursery habitat for economically valuable species. Whilst the fish communities of adjacent mangrove and mud habitats were similar, the mangrove forest typically supported smaller or younger fishes in greater abundance. Laegdsgaard & Johnson (1995) hypothesised that mudflats represent transition zones between juvenile and adult habitats. During their study they found seven of ten economically important species in greatest numbers within mangrove forests. Summarising their findings, they wrote: "Clearly mangrove sites in Moreton Bay play a more important role and have greater potential as nursery habitats (for juvenile fish) than do adjacent habitats".

Conversely, prawns seem to prefer seagrass as a nursery habitat. Young (1978) found that *P. plejebus, P. esculentus*, *M. bennettae* (greasyback prawn) and *M. macleayi* (school prawn) use all available littoral areas of Moreton Bay, and are particularly found in seagrassed littoral areas in preference to infralittoral areas. Young (1978) summarised his data by stating that juvenile *P. plejebus* and *M. bennettae* settle everywhere in shallow areas of Moreton Bay but those in seagrass areas survive better. However, Dall (1958) found that *M. mastersi* (=*bennettae*) postlarvae in the Brisbane River prefer warm sheltered areas with algal cover, indicating that macroalgae may also be important.

While we accept that in general, mangrove forests are important for juvenile fishes and that seagrasses are important for juvenile crustaceans, and given our reservations about the notion of equal value of similar habitats as nursery areas, studies are urgently required into the variation in quality of nursery habitats around Moreton Bay. Furthermore, there is little understanding of synergies that might result from the presence or absence of certain combinations of nursery habitats. For example, while it is recognised that mangroves are an important source of food and shelter for juvenile fishes, unless juveniles spend the time during which mangroves are emmersed (more than half of every day) in tide pools (Crowley & Tibbetts, 1995), they must migrate relatively large distances between infralittoral (low tide) habitats and their feeding (mangrove) habitat. In doing so they expose themselves to predation. Certain combinations of transitional nursery habitats (i.e. those habitats they encounter on their tidal migration) may increase feeding opportunities and reduce predation rate. Thus, determining the relative contribution of superficially similar nursery habitats to fisheries production (for example) may be far more complex than originally thought. To say merely that mangroves are important nurseries is no longer sufficient.

Soft bottom communities

Most investigations of nekton assemblages in Moreton Bay have been carried out on the demersal (epibenthic) fauna of sedimentary shores and subtidal habitats. The faunas of these habitats grade into one another and there is evidence that nekton assemblages are strongly influenced by temperature and salinity regimes. Nevertheless, certain species seem to consistently dominate particular habitats and, with the exception of surf beach and surf bar communities, they are generally well known.

Relatively few studies have examined the fish communities of salt marsh areas. Morton *et al.,* (1988) found that the fish fauna of saltmarsh pools in southwestern Moreton Bay is dominated by *Gambusia affinis* (mosquito fish), *Pseudomugil signifer* (blue eye) and gobiids (gobies). At high tide such areas are visited by commercially important species such as mullet and bream (Morton *et al*., 1987)

Mugilids (mullet) and ariids+plotosids (catfish) seem to be a ubiquitous component of estuarine fish assemblages (Thomson, 1957; Pollard & Hannan, 1994). Most other species found in the riverine reaches of estuaries appear to be transients. In a study of fish community structure along an estuarine gradient in the Albert River, south western Moreton Bay, Thompson (1957) recorded 20 species. Four of these were freshwater with the remainder being estuarine. Mugilids and *Craterocephalus stercusmuscarum* (fly-speckled hardyhead) were the only permanent residents. The most complete report on the distribution of fishes in the estuarine portion of the Brisbane River is given by Mackay & Johnson (1990). They also provide details of species recorded near the river mouth and freshwater reaches.

Weng (1990) reported that the community at a shallow riverine site (Caboolture River) was dominated by *Gerres ovatus* (=*subfasciatus*) (silver biddy), *Eurythsmus lepturus* (long-tailed catfish), *Harengula* (= *Herklotsichthys) castelnaui* (southern herring) and *Sillago maculata* (trumpeter whiting). Species which only occurred at this site included *E. lepturus*, *Siphamia rosiegaster* (pink-breasted siphon fish), *S. multifasciata* (siphon fish) and *Johnius australis* (little jewfish). In areas outside of rivers, but still under estuarine conditions in the western Bay, demersal fish assemblages are dominated in terms of abundance by *Centropogon marmoratus* (fortesque), *Priacanthus macracanthus* (red bigeye), *Paramonocanthus otisensis* (dusky leatherjacket), *L. mortonensis* (pony fish), *Monocanthus chinensis* (fan belly leatherjacket), *P. quadrilineatus* and *Apogon fasciatus* (striped cardinal fish) (Stephenson *et al.,* 1982b; Warburton & Blaber, 1992). *A. fasciatus*, whilst of no economic importance, may be of ecological importance as it was 2.9-9.6 times more abundant than any other species. *Sillago bassensis* (school whiting), *Sillago robusta* (stout whiting) and *Hyperlophus vittatus* (sandy sprat) only occur at the most marine site (Weng, 1990).

Hard bottom communities

To say that the nekton faunas of hard bottom substrates and pelagic communities are less well known would be a gross understatement. There are no published reports on the community structure of either hard bottom or pelagic nekton for Moreton Bay. When compared with soft bottom habitats, rocky substrates are subject to a level of human activity which exceeds what we would expect for areas of such minor geographic area and fisheries significance. Recreational fishers recognise reefs as important to the macrocarnivorous fishes they target. Recreational divers seek diverse and visually spectacular communities that abound on rocky and coral reefs. Moreover, commercial aquarium fish collectors focus most of their efforts on such assemblages.

The structure of assemblages of nekton associated with coral reefs is affected by water quality, location with respect to larval supply, and habitat complexity. Coral reefs within Moreton Bay support species of each of the dominant coral reef fish families: Acanthuridae (surgeon fish), Scaridae (parrot fish), Pomacentridae (damsel fish), Pomacanthidae (angel fish) and Chaetodontidae (butterfly fish). The diversity of particular assemblages appears to be driven by interaction amongst the above factors. For example, the reef to the north west of Peel Island is dominated by massive corals (see Johnson & Neil, this volume), while Myora reef is dominated by branching corals. The former, being fairly devoid of complex shelter, is home to a much lower diversity of fishes than occurs at Myora (Tibbetts, pers. obs.). Flinders Reef, just outside Moreton Bay, supports a diverse assemblage of coral reef species intermixed with a few temperate fish taxa (e.g. *Girella tricuspidata,* luderick) (Tibbetts, pers. obs).

We could find no formal publications on the nekton of rocky subtidal reefs or intertidal areas of rocky shores, with the exception of Tibbetts *et al*. (this volume) which gives data on the distribution of fish species and dietary preferences for pool habitats on rocky shores at Dunwich, Manly, Redcliffe, Caloundra and Point Lookout. The rocky intertidal fish community is dominated by blenniids (blennies) and gobiids (gobies) on sheltered shores, and by blenniids and tripterygiids (triplefins) on exposed shores. Blennies dominate the biomass on these shores and, being grazing omnivores, may play a role in modifying benthic community structure, particularly that of algae.

Processes in Nekton Communities

The processes (or dynamics) underlying the patterns described above are not particularly well understood. In the following paragraphs, evidence for the various processes likely to be important in determining the distribution and abundance of nekton species will be discussed.

Note, however, that:

(1) evidence comes in the main from studies of economically important species rather than ecologically important species; and,

(2) processes are often inferred from descriptions of patterns (survey results), but the strongest evidence comes from manipulative experiments, which have been undertaken very rarely in Moreton Bay.

Spawning

Fisheries scientists like to identify the spawning grounds of economically important species so that both the spawning adult fish stock and the habitat itself can be protected. The two methods of determining spawning grounds are:

(1) to determine the location of adults when they are ripe with eggs, and

(2) to locate larvae (often planktonic) and use water flow data (e.g. seasonal and tidal flow patterns) to establish from where the larvae originated.

Three patterns of spawning have thus been described for nekton species from Moreton Bay. Some commercial and recreational fish species spawn on sandbars at the entrances to the Bay. It is well established that *Acanthopagrus australis* (yellowfin bream), for example, spawn on sandbars (Pollock, 1984; 1985), and it is thought that *Sillago ciliata* (sand whiting) spawn on sandbars at Southport, Jumpinpin, north of Moreton Island and the northern entrance to Pumicestone Passage (Morton, 1985; Quinn, 1993c). Other species, such as *Platycephalus fuscus* (dusky flathead) (Quinn, 1993c) and *Metapenaeus bennettae* (greasyback prawns) (Dall, 1958) spawn within or alongside estuaries, and either spend their life resident in the estuary or elsewhere in the Bay. Still other species live in estuaries of the Bay for part of their life but spawn outside the Bay. *Pomatomus saltatrix* (tailor), for example, join a spawning run north along the Queensland coast, with the major spawning area being Fraser Island (Bade, 1977; Morton *et al*., 1993). Zeller *et al.* (1996) proposed that the strong southward current (East Australian Current) drives highly buoyant eggs and surface-dwelling larvae southward from spawning grounds near Indian Head on Fraser Island. *Mugil cephalus* (mullet) also leave the estuaries, in which they spend much of their life, to make a spawning run along sand beaches in and around the Bay (Thomson, 1954), and it is at this time that the bulk of the commercial catch is made. Not all adults make the spawning run, as some remain in estuaries (Kesteven, 1953). *Penaeus plebejus* (king prawn) and *P. esculentus* (brown tiger prawn) move to oceanic waters to spawn, although some *P. esculentus* adults are thought to remain in the Bay and spawn (Young & Carpenter, 1977). Nektonic species that spawn outside the Bay have larvae that disperse, probably with the assistance of wind-driven and tidal currents, back to the Bay. The juveniles then take up residence in the shallow, sheltered habitats of the Bay.

Settlement and recruitment

With regard to nekton, settlement is defined to be when dispersive larval stages, often planktonic, take up a benthic existence (i.e. become more closely associated with the seabed and its habitats). Recruitment is recorded when individuals appear in a population under study. The term recruitment is somewhat subjective in that a fisheries scientist or manager might consider recruitment to be either the age or size at which nekton are first susceptible to fishing gear, whereas an ecologist might count recruits in the same species as post-settlement individuals in a "nursery" habitat. Measuring this latter form of recruitment is considered to be very important by some scientists because it seems that, in some cases at least, the level of recruitment explains very well the abundance of adults. The degree of importance of recruitment in structuring nekton communities is one of the most hotly debated topics in marine ecology. The link between recruitment levels and abundance of adults has not yet been made in Moreton Bay.

The timing of settlement is known for many nektonic species in Moreton Bay, even for species that have no direct economic importance. Several habitat specific studies (e.g. Watabe, 1993; Laegdsgaard & Johnson, 1995; in mangroves; Young & Carpenter, 1977; Blaber & Blaber, 1980; in riverine, seagrass and sand habitats; Morton *et al*., 1987, in saltmarsh) have shown that the majority of fish and crustacean species appear in shallow waters as early juveniles in the summer months, with notable exceptions being *M. cephalus*, *A. australis* and *P. plebejus*, which arrive in winter. Juvenile *P. saltatrix* recruit only in March in Moreton Bay, and in a distinct pulse (Blaber & Blaber, 1980). They then appear to inhabit chiefly shallow waters over tidal flats adjacent to mangrove forests in western Moreton Bay (Blaber & Blaber, 1980; Morton, 1990).

The data on the abundances of juveniles measured over several years have not been collected and there is, therefore, no chance of linking recruitment rates with adult abundance as has been done elsewhere. Also lacking are efforts to determine whether recruitment levels of any species are influenced by the quantity or quality of water from the catchment. *Penaeus merguiensis* (banana prawn) show a greatly increased recruitment in the Northern Prawn Fishery in the Gulf of Carpentaria in years of high rainfall (Pownall, 1994), but whether *P. merguiensis* or other estuarine species, such as *Metapenaeus bennettae,* are similarly affected in Moreton Bay is not known.

The most well established model explaining the distribution and abundance of nekton in estuarine habitats is based on studies of seagrass meadows in Botany Bay, NSW (reviewed in Bell & Pollard, 1989). According to the model, habitat selection is important during or immediately following settlement. The term 'habitat selection' has a specific meaning here, being the detection and selection of above-ground structure. The larvae of many species, when ready to settle, select vegetated over unvegetated habitat. This would explain the greater abundance of nekton in vegetated areas. Alternative explanations, such as movement to find prey items (which are more plentiful in seagrass meadows than adjacent unvegetated patches (Howard *et al*., 1989), have not been discounted. The importance of food availability in directly structuring nekton communities has recently been highlighted by experiments in other Australian estuaries (Connolly, 1994; Jenkins *et al*., 1996).

The model also contends that early juveniles will move *within* a meadow to select denser vegetation, but will tend not to cross bare patches to reach other meadows. In one of the few manipulative field experiments conducted in Moreton Bay, Hopkins (1996) examined the preferences of newly settled juveniles by placing artificial seagrass units of differing densities in shallow unvegetated and seagrass habitats. It was found that some species were more abundant in the denser habitat. More interesting was the result that for most species, abundances were no greater in habitat (Artificial Seagrass Units) left in place for three days than in habitat left in place for only one day.

Other workers have surveyed different habitats in a bid to determine the preferences of newly arrived juveniles. Recruitment of tiger and greasyback prawns is heaviest into seagrass meadows, whereas king prawns tend to be more common in unvegetated areas (Young & Carpenter, 1977; Searle, 1991). However, the importance of seagrass *per se* is not known as the distribution of young prawns is also related to salinity and depth gradients, which tend to be correlated with seagrass presence and type (Young & Carpenter, 1977). The distribution of juveniles of a wider range of nekton, including prawns, actually seems to be determined primarily by the degree of marine influence (salinity and temperature), rather than vegetation (Young & Wadley, 1979). Laegdsgaard (1996) sampled mangrove, mudflat and seagrass at two locations in the Bay, and concluded that newly settling fish are more common in the mangroves, and that they then move to mudflats and eventually seagrass as they grow older.

Dispersal and migration

Dispersal is the movement of egg, larval, or early juvenile stages. Movement of older juveniles and adults is called migration. Measuring dispersal directly is very difficult, but can be inferred where there is reliable information about spawning areas and the time and place of arrival of juveniles (e.g. Pollock, 1982a; 1983). The spawning runs by adults of several economically important fish species have been described above. Several other species of fish (Blaber & Blaber, 1980) and prawns (Young & Carpenter, 1977) move from shallow, sheltered estuarine habitats to deeper water as they grow larger. Warburton & Blaber (1992) also found some evidence of ontogenetic migration (i.e. a change in distribution associated with increase in size), as adults of *Pelates quadrilineatus*, *Leiognathus* spp., *Paramonocanthus otisensis* and *Monocanthus chinensis* occur in deeper water (Blaber & Blaber, 1980), whilst *Lovamia fasciata* (= *Apogon fasciatus*) is largely confined to relatively shallow water (5-9m, Rainer, 1984). The implications of dispersal for transfer of energy (as fish biomass) from estuaries to deeper waters are obvious but have not yet been studied. Thomson (1957) describes the penetration of estuarine species of fish such as *M. cephalus*, *Hyporhamphus regularis (= H. r. ardelio)* (river garfish) and even *Galeolamna ahenea (=Carcharinus leucas)* (bull sharks) into the freshwater reaches of the Logan and Albert rivers. Morton *et al*. (1987) showed by fin-clipping that individual bream (*A. australis*) and toadfish (*Torquigener* spp.) returned several times over a seven day period to the same small tidal creek supplying a saltmarsh flat. Other species, including mullet, did not reappear.

The measurement of movement by tagging is effective on larger fish species taken by recreational or commercial fishers since there is a reasonable chance of getting tags returned. Moreover, the attachment of tags does not significantly increase mortality. Morton *et al.* (1993) tagged 2 173 juvenile *P. saltatrix*. They found that of the 237 fish recaptured (11%) over 30 months, most had moved short distances from the release point, suggesting that estuaries such as Moreton Bay act as nursery areas for tailor prior to their movement to open surf beaches as adult fish. Pollock (1981) in a study of *G. tricuspidata*, found no fish less than 3 years old and hypothesised that adult fish move to Moreton Bay from the south. *Penaeus plebejus* settle in seagrass and are found in higher numbers in seagrass than in sand or mud and undergo a slow migration to deeper water (Young, 1975). It has been proposed that these prawns migrate on darks (i.e. last to 1st quarter phases) (Racek, 1957).

As yet, there is no standard method for following the movement of fish and prawns that are either too small or fragile to tag and/or are not likely to be recaptured once marked because they are rarely collected. Little work has been done in Moreton Bay on species such as this, yet if we are inclined to restore the nekton fauna of degraded habitats (e.g. seagrass beds), it is important to know the likelihood of recolonisation of such areas. A range of new methods including genetic (Richardson *et al*., 1986) and otolith microchemistry (Connolly, this volume), techniques for measuring movement and even dispersal of larvae are being developed with the potential to be used in Moreton Bay.

Predation and competition

Although the majority of the species of fish in Moreton Bay is carnivorous, only a few eat other fish, most take invertebrate prey (Blaber & Blaber, 1980). It has been shown that juvenile *Sillago maculata* (trumpeter whiting) include juvenile *M. bennettae* in their diet (Mangubhai *et al.*, this volume). Whilst the presence of fish or prawns in diets of other animals provides evidence of direct predation, there has been no attempt to quantify the importance of predation on populations of these prey species in Moreton Bay. Predation rates on juvenile sillaginids have been tested in a tethering experiment, which demonstrated that predation was more severe in habitat without vegetative structure (Laegdsgaard, 1996).

Suggestions of interspecific competition for resources within the nekton of the Bay are based only on weak circumstantial evidence. Weng (1986), for example, showed that in Moreton Bay, the larger whiting species (sand whiting *Sillago ciliata* and golden-lined whiting *S. analis*) occurred in the same places as *Acanthopagrus australis* (yellowfin bream), but that the smaller trumpeter whiting (*S. maculata*) did not. Given that all of these species have a similar diet, Weng (1986) suggested that the smaller species could not compete with bream. It has been well demonstrated that manipulative experiments are the soundest method of providing convincing evidence of competition for a resource (Underwood, 1986), but such experiments have not been attempted within Moreton Bay.

Growth, diet and mortality

Growth rates are known for some economically important species: *Penaeus esculentus* (tiger prawn) (O'Brien, 1994), *A. australis* (Pollock, 1982b), *Girella tricuspidata* (Pollock, 1981), and *S. ciliata* (Weng, 1988). Natural mortality rates (i.e. excluding fishing mortality) have been estimated for *P. esculentus* (O'Brien, 1994) and in the laboratory for *S. ciliata* (Weng, 1988). The diets of many species have been observed through examination of stomach contents (e.g. Moriarty, 1976; Blaber & Blaber, 1980).

The diets of animals often change as they grow, in response to changes in their size relative to prey size. The planktonic larvae stages of fish feed on plankton, the early juvenile stages of many estuarine species are likely to feed exclusively on very small, bottom dwelling organisms, collectively known as meiobenthos (Coull *et al.,* 1995), whilst late juveniles and adults feed on a more specialised diet. For example, Sumpton & Greenwood (1990), in a study of the feeding ecology of fish in the Logan-Albert estuary system, found that *Johniops* (=*Johnius*) *vogleri* (little jewfish) and *Polynemus multiradiatus* (putty-nosed perch) are initially plankton feeders, and that they both undergo a dietary shift as they grow, eating first copepods, then mysids and later *Acetes* (an abundant sergestid shrimp).

In upper estuarine and saltmarsh assemblages, terrestrial foods are often of greatest importance (Morton *et al.,* 1988; Sumpton & Greenwood, 1990). Fish assemblages in lower estuarine and open Bay waters are dominated by carnivores which feed on bentho-pelagic prey such as polychaetes, amphipods, decapods, mysids and copepods (Walkden-Brown, 1968; Sumpton & Greenwood, 1990; Warburton & Blaber, 1992). Feeding in such communities occurs mainly by day (Warburton & Blaber, 1992), however there is some evidence of temporal segregation in feeding periodicity (Walkden-Brown, 1968). Whilst authors often ascribe particular prey preferences to their subjects, there is evidence of trophic plasticity. For example, Warburton & Blaber (1992) found evidence of diet switching in *P. quadrilineatus* and suggested that this may be indicative of an adaptive response to changes in local prey availability. Trophic plasticity has been confirmed in estuarine fishes as a result of changes in prey availability following flood events. In a study of the feeding ecology of fish in the Logan-Albert estuary system, Sumpton & Greenwood (1990) found that both *Arius graeffei* (fork-tailed catfish) and *J. vogleri* shifted from a diet dominated by mysids to one dominated by copepods. They suggested also that flood-induced mortality in benthic species resulted in the loss of polychaetes from the diets of benthic carnivores. However, it is likely that major prey switching occurs only in response to an extreme event. Brewer *et al.* (1995) found that in most fishes, related species and species from similar habitats, ate similar prey. They hypothesised that competition may be low in tropical waters and thus predators can be selective. Whilst competition may be low in tropical systems, the winter decrease in species richness and abundance, which is a characteristic of Moreton Bay's subtropical nekton assemblages, may be a function of seasonal changes in benthic productivity and diversity. For example, Walkden-Brown (1968) found that the abundance and diversity of prey in fish diets decreased in the winter months. It remains to be resolved whether density-dependent effects (e.g. predator vs prey abundance) cause the decrease in species richness and abundance of the benthos and nekton in winter months, or whether density-independent factors (e.g. salinity and temperature) are the cause.

Few studies of Moreton Bay have attempted to quantify parameters that may be applied to ecological models of energy flow involving the nekton. Moriarty (1976), using measurements of muramic acid, determined that bacteria comprised 15-30% of organic carbon in the stomach of *M. cephalus* feeding on seagrass flats and that diatoms contributed 20-30%. Observations by Hollaway & Tibbetts (in prep.) indicate that mullet graze on intertidal diatom and meiobenthic assemblages in the Brisbane River and leave characteristic feeding scars which may enable their grazing impact to be quantified. Moriarty (1976) estimated that mullet ingest 50 g dry/sediment/day corresponding to a 1.5 g C/day for sediment in a stomach containing 3% C.

Wassenberg & Hill (1987a) investigated the natural diet of the tiger prawns *Penaeus esculentus* and *P. semisulcatus* in Moreton Bay and found that adults ate a range of benthos including, bivalves, gastropods and crustaceans, and that meiobenthos were only important in the diet of animals taken from seagrass beds. Moriarty (1977) found that diet depended on location. He also determined that *P. plebejus*, *M. bennettae* and *P. esculentus* fed on crustaceans, molluscs and polychaetes and that detritus was important in the diet of *M. bennettae*. Moriarty (1976) found that for the detritivore *M. bennettae*, bacteria comprised 20-35% of the proventriculus contents, which suggests that they select organic matter rich in bacteria. Wassenberg (1990) found that juvenile *P. esculentus* ate vegetable as well as animal matter, particularly seagrass seeds when in season.

Certain members of the nekton may cause an acceleration of nutrient cycling within the Bay. For example garfish, (Hemiramphidae) which feed on floating seagrass (Tibbetts, 1991), cause the sedimentation of seagrasses that would otherwise be lost to the seagrass habitats either through export to the ocean or flotsam accretion on shores. The fine trituration of seagrasses that pass through the alimentary system of such fishes render them more susceptible to rapid breakdown by saprophytes.

Morton (1990) challenged the paradigm that mangrove communities are dominated by planktivores (Blaber, 1980; Robertson & Duke, 1987) and that nutrient export is mainly through detritus export pathways. Morton suggested a model in which leaf litter is consumed by sesarmid crabs (which are in turn consumed by lower carnivores), as a more accurate model of mangrove production export than others which suggest litter export drives mangrove contribution to estuarine systems (c.f. salt marsh in temperate areas). He noted, however, that he had only sampled a single area with limited temporal replication. Whilst Morton's block net (take all) approach has yet to be repeated and his model confirmed for other mangrove habitats in Moreton Bay, his findings illustrate the extent of our uncertainty about basic ecological processes in important ecological systems in Moreton Bay. Furthermore, they demonstrate that present ecological models may more closely reflect the selectivity of our sampling gear rather than the ecology of a particular community.

Methodological Constraints on our Understanding of Pattern and Process in the Nekton

Our review of studies on the nekton of Moreton Bay has revealed the use of a wide range of methodologies (Table 2). It is clear from broad brush comparisons between some of these studies (e.g. Morton *et al.,* 1987 *vs* Morton *et al.,* 1988 and Stephenson & Dredge, 1976 vs Quinn, 1980), that the community structure determined for a particular habitat is greatly influenced by the type of gear used to sample that "community". Morton (1990) suggested that most previous studies of mangrove communities had used techniques (such as beam trawling and seining close to mangroves), that are unlikely to take fishes of direct commercial value, especially large, mobile individuals. In effect, large, mobile species that may be important to community processes may be eliminated from consideration by the use of inappropriate gear. Stephenson & Dredge (1976) and Blaber & Blaber (1980) used techniques that are likely to provide a more accurate indication of the true abundance and composition of the fish community of mangroves (Morton, 1990). Moreover, few studies attempt to estimate gear efficiency or differences in susceptibility among community members. Morton (1990) measured the catch efficiency of his block net by releasing marked fish into the sampled area. He found that demersal species such as *S. analis* are more likely to escape from the block net (only 66% recaptured), than pelagic species such as *Tylosurus macleayanus* (longtom), *Gerres ovatus* (= *subfasciatus*) (silverbiddy), *Mugil georgii* (fantail mullet), *Arrhamphus sclerolepis* (snub-nosed gar) and *Harengula (=Herklotichthys) abbreviata* (southern herring) which were all recaptured with a greater than 90% success rate. Weng (1983) noted that juvenile whiting are able to bury themselves in sand and would thus avoid capture by haul nets. Morton's (1990) data, together with Weng's observation, indicate that the contribution by some demersal species to nekton assemblages may have been greatly underestimated and that efforts should be made to calibrate the efficiency of nets against the catchability of the nekton. Allen *et al.* (1992) collected haul net (seine) samples to make comparisons between the fauna captured in the first and subsequent samples and also between seine samples and samples taken with rotenone (an ichthyocide) following seining. They recommended that in any temporal study using seine nets, collection efficiency should be measured once a season. Coles' (1979) findings lend support to the recommendations made by the previous (American) study. He found that movement by prawns in and out of the sediment influenced their susceptibility to capture and that burying behaviour changed with both season and time of day. Jones (1986), investigating the spatial and temporal patterns in nekto-benthic invertebrates (i.e. crabs, prawns and cephalopods), warned that while the differences between the abundance and species richness of nocturnal and diurnal samples are often explained by reference to supposed net avoidance during the day, divers in his study did not observe net avoidance and that even fast swimming squid were caught. He concluded that: "...nocturnal sampling is essential both for adequate description of the community and the estimation of abundance of individual species. This is because the daytime community is usually a depauperate version of the night community" (Jones, 1986; p.145). Moreover, our understanding of community processes may be influenced by sampling methodology. For example, Morton (1990) found that in terms of the herbivores, *G. tricuspidata* (luderick) dominated community biomass sampled by block net and planktivores/microcarnivores dominated biomass of seine net samples. Whilst these techniques were employed in adjacent (but dissimilar) habitats, the finding illustrates the point that sampling technique strongly influences ecological intepretation of the functioning of Moreton Bay's nekton communities.

A problem related to that of escape from the sampling device by speed, stealth or guile, is that of nekton escaping from the device because they are too small to be caught. To overcome

Table 2. A summary of the different sampling gear used to investigate fish assemblages in Moreton Bay indicating the strength of the influence of gear type and dimension on our understanding of community structure. Under gear dimensions, SM refers to the stretched mesh measurement of net mesh.

Habitat	Site	Gear	Gear Dimensions	Notes	Author(s)
Saltmarsh	Coomera	Block net	12 mm SM	19 species, 11 of economic value, dominated by tetraodontids, bream and mugilids	Morton *et al*., 1988
Saltmarsh	Coomera	Dip nets	0.5x 0.4 m x 2 mm	8 species, 4 of economic value, dominated by *Gambusia affinis, Pseudomugil signifer* and gobiids	Morton *et al*., 1987
Saltmarsh/ Mangrove	Eden Island	Pop nets	25 m^2 x 1 mm	19 species dominated by the planktivore *Ambassis marianus*	Moussalli & Connolly, this volume
Mangrove	Lota	Block net	240 m x 18 mm SM	42 species, dominated by *Mugil georgii* in numbers (30%) and *G. tricuspidata* in biomass (72%)	Morton, 1990
Adjacent to Mangroves	Lota	Seine net	70 m x 18 mm SM	30 species, dominated by planktivores/microcarnivores	Morton, 1990
Adjacent to Mangroves	Lota	Gill net	30 x 4 m 150 mm SM	12 species, dominated by *Mugil cephalus* and *G. tricuspidata*	Morton, 1990
Mangrove	Fisherman Island & Deception Bay	Trap (Block) net	8 x 2 m x 1 mm mesh	Total of 53 species (all three habitats), only 4 species exclusive to seagrass, 27 species exclusive to mangrove/mud habitats	Laegdsgaard & Johnson, 1995

this problem Young & Wadley (1979) excluded animals less than 6 mm or greater than 10 cm from analysis to reduce the bias of net avoidance or escape on the data. Unfortunately, few others have followed their example.

Another problem arises from the difficulty of repeating a previously applied methodology, either through the use of relative measures (e.g. Laegdsgaard & Johnson (1995) hauled their seine nets for 20 *paces*) or lack of information (e.g. Weng (1990) makes no mention of the *width* of his beam trawl). A sufficient description of the sampling method is one that would allow a subsequent investigation to repeat exactly the work which is described. Without some adherence to these basic principles of scientific reporting, we will miss opportunities for making long term comparisons.

Techniques vary in the quality of their quantitative data. For example, density estimates derived for pop nets (i.e. buoyant, anchored wall nets, released by a trigger or timer, that surround nekton that have moved into the netted area) are likely to be accurate (disregarding possibly serious disturbance effects) because they sample a set area. Seine nets provide reasonable density estimates, but are susceptible to variations in the pattern of deployment and net ballooning (being lifted from the seabed) in moderate currents. Gill net catches must be indexed to the sizes of mesh used and the time the nets are allowed to fish, and block nets by their catchment area (i.e. the area of habitat covered by water which drains through the net) (see Morton, 1990). Furthermore, whilst fish abundance is usually described in terms of density (numbers per m^2), they may actually occupy all levels of the water column, and in some habitats (e.g. pelagic) concentration (numbers per m^3) may be a more appropriate measure.

Replication is another important factor in gaining useful quantitative data for temporal and spatial nekton studies. Studies conducted prior to the advent of personal computers suffered from an inability to treat large data matrices. Accordingly, there was little use in taking (or using) adequate levels of replication. For example, Young & Wadley (1979) sampled each of nine stations in triplicate at night. Constrained by the length of time it took to sort and identify the epibenthos (P. Young, pers. comm.), with the exception of November, they selected one replicate at random from each triplet for identification and counting. They found that they could have described the same spatial pattern from a single month of intensive (*viz.* replicated) sampling as they determined from their extensive 12 month survey that lacked replication. However, given the heterogenous nature of the Bay, the number of samples required for accurate estimates of particular community parameters may be large. Stephenson *et al.* (1982a), in an investigation of demersal trawl catches to evaluate the effects of different sampling regimes, reported little success and suggested that this was due to random variation in the data. In a follow up study, Stephenson *et al.* (1982b) found great variation in catches between two years sampled and, considering this to be due to climatic differences between the two years, they wrote: "A very prolonged study would have been necessary before the effects of weather could have been quantified, and a basis found for assessing potential changes due to man". It is obvious that there are major obstacles to gathering meaningful estimates of community structure and flux. However, certain techniques make adequate replication problematic. For example, Morton (1990) found significantly fewer species and individuals in block net samples taken in the same area 24 h later. The extent of depauperation of fauna in terms of both numerical abundance and species richness led him to exclude those 'replicates' from his analysis.

Human Impacts on Nekton Communities

Commercial and recreational fishing

Detailed discussion of the management of Moreton Bay fisheries is beyond the scope of this paper. The reader is directed to recent reviews of fisheries productivity (Quinn, 1993b) and the role of the Queensland Department of Primary Industries in the management of fish stocks and fishing (Quinn, 1993a), as well as a detailed report on the status of Moreton Bay fisheries by Quinn (1993c). In general, he regarded the nekton stocks of Moreton Bay as stable. However, recent work by Thwaites (1996) indicates cause for concern. Both catch per unit effort (equated with a decrease in population size) and mean capture size were found to have decreased for the four species of estuarine and coastal fish examined: *Pomatomus saltatrix* (tailor), *Platycephalus fuscus* (dusky flathead), *Acanthopagrus australis* (bream) and *Sillago ciliata* (sand whiting). To what extent these decreases related to increases in recreational fishing pressure is not known. Thwaites (1996) suggests that the change is due to combination of increase in fishing effort, improvements in fishing technology and habitat modification.

Wassenberg & Hill (1990) estimated that approximately 3000 t of bycatch from trawlers enters Moreton Bay annually. Only 3% of this floats. Of fish that sink, about half are consumed by birds and dolphins. Sand crabs are dominant scavengers of fish that reach the bottom due to their speed and capacity to break up material. Fish are important scavengers of discards during the day. Blaber & Wassenberg (1989) examined the feeding ecology of birds feeding on trawler discard and suggested that *Phalacocorax varius* (pied cormorant), *P. melanoleucos* (little pied cormorant) and *Sterna bergii* (crested tern) are primarily dependent on trawler bycatch and that the population of 350 *P. varius* consumed approximately 13.7% of the total bycatch. Wassenberg & Hill (1987b), in a study of *Portunus pelagicus* (sand crab), found that about a third of its diet comprises trawl bycatch. They noted that *P. pelagicus* was the most common scavenger of baits set to simulate trawl discards. Liggins *et al.* (1996) conducted observer-based surveys of bycatch from prawn trawling in Botany Bay and Port Jackson, New South Wales. They found that bycatch included large numbers of small, recreationally important fin fish and estimated that a mean annual bycatch of 1.52 $\pm$ 0.2 million fish was taken from Botany Bay and 219 000 $\pm$ 23 000 from Port Jackson. They recommended urgent consideration of strategies to reduce bycatch.

Dredge (1974) found from a creel survey that Serpentine Creek was subjected to between 170 and 270 fisher hrs/ha/year and that on a world scale this represented very high angling pressure. Indeed, West & Gordon (1994) stated: "Although data on recreational fishing throughout Australia are limited, anglers now appear to be the dominant harvesters of seasonal estuarine fish species." This finding has been confirmed for Moreton Bay (Quinn, 1993c).

Habitat modification

All losses of mangrove and seagrass habitat must be considered to have a direct and deleterious effect on the nekton stocks in Moreton Bay (Young, 1978). In Young's (1978) words: "destruction of littoral areas in Moreton Bay will be accompanied by a corresponding decrease in the commercial prawn fishery. This decrease will be magnified where seagrasses are included in the destruction". Several studies have recently highlighted the importance of these areas within the Bay as nursery areas for juveniles (Blaber & Blaber, 1980; Warburton & Blaber, 1992) and feeding grounds for adult fish (e.g. Brewer & Warburton, 1992). Morton (1989) in an investigation of the influence of habitat modification on estuarine fish communities, found that modified estuaries generally had fewer macrobenthic carnivores (fish that tend to have greater economic value) and more planktivores and microcarnivores than did a nearby unmodified estuary. Pollard & Hannan (1994) investigated the effects of flood mitigation structures, such as flood gates and modified channels, on fish habitat in the Clarence River (northern NSW). They found that flood gates resulted in the loss of upstream mangroves and impeded fish from previously accessible nurseries and feeding habitats.

Other forms of habitat modification likely to influence nekton assemblages, yet about which we know little, include: bait digging, dredging, trawling, land reclamation, sedimentation and anchor damage. Little work has been done in Moreton Bay on the direct and indirect effects of dredging. This is particularly surprising considering the extent of dredge spoil dumping that occurs in the Bay and the continued dredging of the Brisbane River for gravel, sand and to maintain navigation channels. The suspension of sediments and release of contaminants locked in sediments are two principal effects of such activity, but as yet, the scale of their impact on nekton and upon the organisms upon which they feed is not known.

The main forms of habitat creation in Moreton Bay are the creation of artificial reefs, jetties and moorings and the development of canal estates (see Morton, 1992; Williamson *et al*., 1994). Little information is available on the impacts on nekton of the former, however, it is likely that nekton community structure will be locally modified and this may have consequences for the communities upon which they feed.

Mining activities such as those that were until recently undertaken in the extraction of limestone from reef areas within Moreton Bay (ceased in 1997), and sand and gravel from the beds of rivers (ceased in 1998), including the Brisbane River, are likely to have had a serious effect on nekton communities, however, these effects are yet to be quantified and are unlikely to ever be known.

Pollution

Connell (1974) studied a kerosene-like taint present in the flesh of mullet. He found that the probable cause was a mixture of chemicals that resembled kerosene and that these were present also in the sediment in the gut contents and in sediments in the Brisbane River. Connell *et al*. (1975) found the same mixture to be responsible for a taint in *Mylio* (= *Acanthopagrus*) *australis* (bream). Shaw & Connell (1982) looked at factors influencing PCB accumulation in food chains in the Brisbane River. They found particularly high levels in pelican (8.2 mg/kg), gull (2.6 mg/kg) and catfish (2.1 mg/kg).

Sammut *et al.* (1996) studied the Richmond River in northern New South Wales and found that acidification is an estuarine flood plain process that occurs in Holocene sulfidic sediments laid down in vegetated, low energy tidal environments (e.g. mangroves). Sulfates from seawater are changed to sulfides by microbial reduction (frequently in the form of iron pyrite). Sulfuric acid is generated by these sediments when they are oxidised by exposure to the air. One flood event caused a 90 km stretch of the Richmond River to become acidified. Sulfuric acid is produced in these sediments at a rate of 300 kg/ha/yr. Sammut *et al.* (1996) have suggested a range of potential long term effects of estuarine acidification including: disturbance to fish reproduction and recruitment, loss of food resources, acid barriers to migration, reduction in species diversity and long term habitat degradation. Sub-lethal effects on fish include red spot disease, which has been observed in the Brisbane River (I. Tibbetts, pers. obs.). Studies are currently underway on the effects of acidification in the Pimpama River which drains into southern Moreton Bay (A. Porter, pers. comm.).

The influence of waste water (e.g. nutrient input from Luggage Point) upon the nekton of Moreton Bay and catchment is likely to be a function of the differential ability of different species and life history stages to withstand reduced water quality, the impact on the abundance of their food and the disturbance to aquatic vegetation that provides various nursery habitats.

Fossicking

The impacts of fossicking (i.e. the unstructured recreational disturbance or removal of organisms or habitat), have not been assessed for the region. Fossicking is an exclusively intertidal phenomenon and is generally restricted to accessible areas of shore, achieving its maximum intensity in the summer months and during holiday periods. It is likely that this activity has some influence through disturbance or removal of organisms but whether such disturbances are any greater than those caused by natural events (e.g. storms) is unknown.

Cage-based aquaculture

Cage culture of marine fishes is gaining popularity throughout the world. In Australia, current practice includes the culture of salmon and tuna in large enclosures located in sheltered embayments. Currently there are no such systems in Moreton Bay. Studies in other areas have indicated that such systems can induce localised eutrophication resulting from uneaten food and faeces which sediment below the cages and greatly increases sediment nutrient levels. Other effects include the accidental enmeshing of large nekton in predator exclusion mesh which is hung around the cages. Organisms such as dolphins and sharks have either been injured or killed in such events. A third effect stems from the influence of the cage structure as a fish attracting device. Fish gain shelter from the structure as well as food in the form of attached epibionts, food not eaten by the cultured organisms and feces. This presumably leads to a marked change in the fish assemblage in the area of the cage culture system, and this may influence the composition of benthic communities.

Terrestrial-based aquaculture

Throughout the Indo-Pacific, the provision of space for aquaculture operations through land reclamation and the subsequent discharge of eutrophic water have been highlighted as the two principal causes of destruction of large areas of saltmarsh, mangrove, and seagrass habitat. In Moreton Bay such activity continues, albeit on a small scale, however, land use and the quality of any discharge from aquaculture systems are under strict conditions of monitoring and control. Since the influence of aquaculture systems on the nekton is mainly via either habitat loss or changes in water quality, see our comments on these issues above.

Restocking

While to our knowledge, no restocking of wild populations of estuarine or marine species has been attempted in Moreton Bay, the Maroochy River estuary to the north of Caloundra continues to be restocked from supplies of juvenile estuarine species reared at Bribie Island Aquaculture Research Centre (BIARC). Approximately 14 000 juvenile *Sillago ciliata* (sand whiting) and 76 400 *Platycephalus fuscus* (dusky flathead) fingerlings between 30 and 60 mm TL have been released with the intention of supplementing wild stocks depleted by a massive fish kill in the river in the early 1990s (P. Palmer, pers. comm.). To gauge the impact of the release on the wild population of juveniles and the effectiveness of the restocking program, attempts were made to mark the fish with oxy-tetracycline prior to their release. However, only 10-20% of whiting and flathead retained a recognisable mark. Whilst a scale pattern recognition system is able to discriminate between wild and hatchery reared whiting with an 80-90% accuracy, this technique has failed with flathead scales and the use of genetic markers is being investigated. The supplementation of wild fisheries with hatchery-reared juveniles is a regular feature of freshwater fisheries, and much work needs to be done on the effectiveness of such programs on marine and estuarine species and on any ecological impacts of such releases. Efforts to replace depleted stocks arise from the economic importance of fish such as whiting, flathead and bream. Their role as principal targets for recreational anglers generates activity in local retail and accommodation industries.

Aquarium fish collection

The continued collection of fish by private or commercial aquarists is a major cause for concern. Collections are made, usually by suspended nets operated by divers, from subtidal rocky and coral reefs, as these areas are inhabited by the smaller, more colourful fishes that are targeted by such operations. In the absence of knowledge about settlement, recruitment, survivorship, age and the resilience of communities to the selective removal of particular community members, it is uncertain whether the efforts of aquarium fish collectors have the capacity to alter long term community structure or render certain members of such communities locally extinct. Given the small area and degree of hydrographic and geographic isolation of rocky and coral reef communities in Moreton Bay, it is likely that serious anthropogenic perturbations will first be detected in such areas. On the positive side, the knowledge and experience of aquarium fish collectors will undoubtedly contribute to research and monitoring programs. Their knowledge of the care of marine aquarium fish may benefit efforts to culture high value (economic or ecological) species to obviate the necessity for their collection from the wild.

Communities and Species at Risk

Apart from the aforementioned threat to the communities of rocky and coral reefs, and the possibility (which increases to likelihood and then certainty as the time period under consideration is extended) of a major pollution event affecting nekton residents in intertidal areas of Moreton Bay, there are no specific threatened species or communities of which we are aware. This statement should be placed in the context of the precision and completeness of our taxonomic knowledge and understanding of community patterns. There may be species under threat, but for the moment they are unknown.

Generally, the larger the individual, the rarer it is in the environment. The laws of thermodynamics and trophodynamic theory dictate that the more steps up the food web (i.e. the higher the trophic level) the smaller the biomass that can be sustained for a given level of productivity. High order carnivores such as sharks and large predatory fishes are thus more susceptible to shifts in the trophodynamics of systems such as Moreton Bay. Moreover, they are the favoured target of anglers (a level of attention which may be disproportionate to their abundance in the environment) and vigilantes who perceive them as a threat to humans. Selective fishing/killing may have a serious effect on populations of such organisms and local extinction is presumably more likely in this group. Foreshadowing or determining local extinction among highly mobile animals is difficult because the effective surveying of individuals or populations is complex, time consuming and expensive. Recently developed techniques, such as the use of radio telemetry and satellite tracking, may provide information about the distribution, migration and population structure of these large members of the nekton.

Anecdotal evidence indicates that major shifts may have occurred in the structure of nekton populations in the region. For example, Welsby (1905) reported that school jew (*Johnius sp.*) were once abundant in the Brisbane River and were a major target for anglers. They are now only present in low densities and mullet and catfish dominate the system. Such phase shifts may either be a natural phenomenon driven by climatic or environmental processes, that bring about a change from one quasi-equilibral community structure to another, or they may be driven by anthropogenic influences. Such dramatic shifts in community structure have been labeled catastrophic events.

Summary and Conclusions

Despite more than 150 years of investigation of the nekton of Moreton Bay, and major advances in our understanding of the system, there remains much to learn of their biology, their interactions with the plankton and benthos, the spatial and temporal patterns within nekton communities and the processes that drive these patterns. The economic worth of the Moreton Bay is a function of its nekton communities. This has led to valuable, albeit patchy, research particularly on the biology of species targeted by recreational and commercial fishers. While it has been recognised for some time that an understanding of processes affecting nekton communities is pivotal to our capacity to manage them and their environment, it is only now that information on processes, derived from supposedly analogous systems, are being dispensed with and serious attempts at determining processes for Moreton Bay are being made.

Of great concern is the finding that of four fish species, for which there are long term catch per unit effort and catch size data, all showed a significant decrease in either catch per unit effort or size (Thwaites, 1996). Such decreases in abundance and mean size must influence structure and processes among benthic and other communities and add a further dimension to models which describe the nature and pattern of change in Moreton Bay communities.

Differentiating between changes in community structure arising from natural fluctuations in environmental factors and those arising from anthropogenic alteration of habitat quality and fishing pressure, is difficult with the general lack of information available on the nekton resources of the region. The lack of detailed studies and monitoring means that we will never know the full extent of historical changes in the Bay's nekton communities, and without a more systematic approach to gathering community data we are unlikely to detect changes in the nekton, irrespective of whether they be natural or anthropogenic. Walters (1986) inferred, from analysis of middens on North Stradbroke Island, that during the last millennium fish were caught throughout the year. Prior to this there were peaks and troughs in abundance which were indicative of either migratory behaviour among Bay-side human communities or a lifestyle less dependent on fish. Our present day community has a reliance upon the natural productivity of nekton in Moreton Bay, but how much longer will we be able to rely on its resources without a detailed knowledge of its communities and ecological processes combined with the political will to protect them?

Acknowledgements

We are indebted to Ms Fiona McKnight for assistance with locating literature, Ms Elizabeth Cotterell and Mr Hing Ang for their valuable comments on the draft manuscript, Mrs Alice O'Neil, CSIRO Fisheries Librarian for hunting up additional references, and to two reviewers who in their own right have made major contributions to our knowledge of Moreton Bay, Dr Steve Blaber and Dr Peter Young. Without their help this project would not have been possible.

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Cumacea (Crustacea: Peracarida) of Moreton Bay: A Preliminary Account of Studies on their Taxonomy, Distribution and Diel Behaviour

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Abstract

Sixty-seven species of cumaceans were recorded in samples from 40 sites in Moreton Bay between March 1989 and March 1993. Thirty-six of those species are new, 18 from the *Bodotriidae*, seven from the *Diastylidae*, six from the *Gynodiastylidae*, four from the *Nannastacidae* and one from the *Leuconidae*. In a further three species only one sex was previously known. Forty-seven of the 67 species are new records for Moreton Bay.

Introduction

Cumaceans form an integral part of the marine food chain, being important dietary components of fish such as cardinal fish *Apogon fasciatus*, fan-bellied leatherjackets *Monacanthus chinensis*, bullrout *Centropogon marmoratus* and striped trumpeters *Pelates quadrilineatus* (Tafe, 1995). These four species are amongst the five most abundant fish species in shallow areas of the western part of Moreton Bay, and all share a benthopelagic feeding habit (Warburton & Blaber, 1992). Cumaceans are also common dietary components of hardyheads *Atherinomorus ogilbyi* (Cotterell & Tibbetts, this volume).

Wassenberg & Hill (1987) and Wassenberg (pers. comm.) showed that the diets of two commercially important prawn species in Moreton Bay, *Penaeus esculentus* and *P. semisulcatus*, included cumaceans amongst a range of meiofaunal species. Dall (pers. comm.) found small numbers of cumaceans in juvenile *P. esculentus* sampled in spring from Toondah Harbour, Moreton Bay.

Cumaceans are typically found in, on or near the substrate in estuarine, neritic and oceanic waters. They are therefore regarded as demersal zooplankton (Jacoby & Greenwood, 1988), meiofauna (Higgins & Thiel, 1988; Watling, 1988) or cryptofauna (Peyrot-Clausade, 1980; Preston & Doherty, 1994). Cumaceans occur at all oceanic depths, but most occur on the continental shelves with the smaller, meiofaunal species being taken in very shallow coastal waters (Watling, 1988).

Methods

Taxonomy

Sledge-net samplers and plankton nets were chiefly employed to collect cumaceans at 40 sites in Moreton Bay for a taxonomic study (Figure 1). Re-entry trays and light-traps were also used to supplement the sampling programme. Sampling sites were chosen on the basis of substrate type, salinity range, spatial distribution and accessibility. Sites were selected in shallow areas with a thick mud bottom, in channels between intertidal sand flats, in protected bays with both sand and seagrass substrates, and in rivers and tributaries.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 421-428.

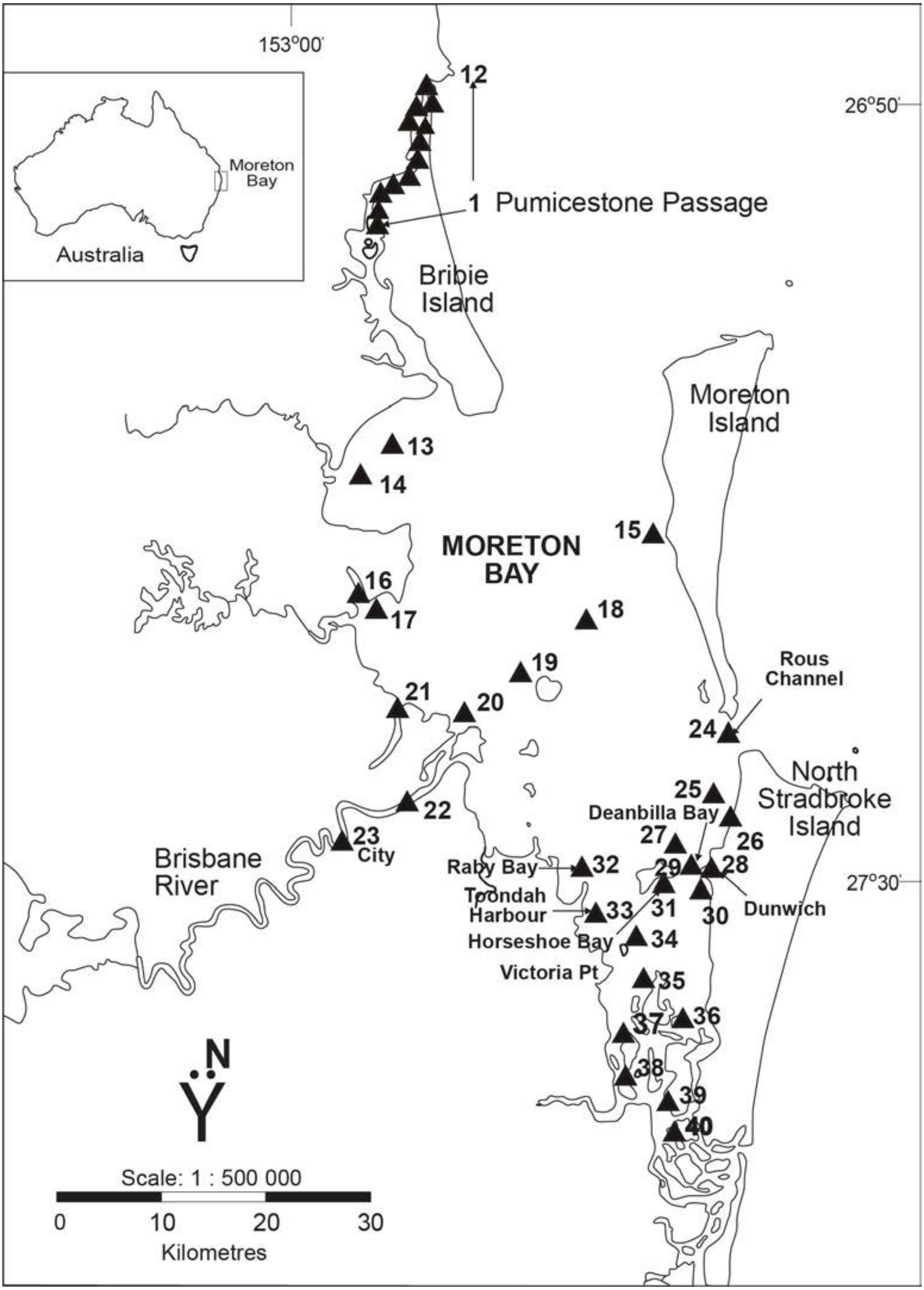

Figure 1. Map of Moreton Bay showing sampling locations.

Distribution patterns

Spatial and temporal distributions of the dominant species of cumaceans were investigated along an estuarine gradient in northern Moreton Bay (Figure 2). Twelve sampling sites in Pumicestone Passage, between the northern tip of Bribie Island and the mouth of Coochin Creek, were sampled using a sledge-net sampler after dusk at lunar fortnightly intervals over a one year period.

Diel rhythms

Diel vertical migration field studies were conducted over sand and seagrass substrates in marine (33.1 - 36.5 PSU) environments. A twin mouth sledge-net sampler was used for sampling two epibenthic levels, and a plankton net for horizontal surface samples. Samples were taken at two hour intervals over a 24 h period in all four seasons of the year. The number and species composition of cumaceans per sample were used to gauge movements of each species through the water column.

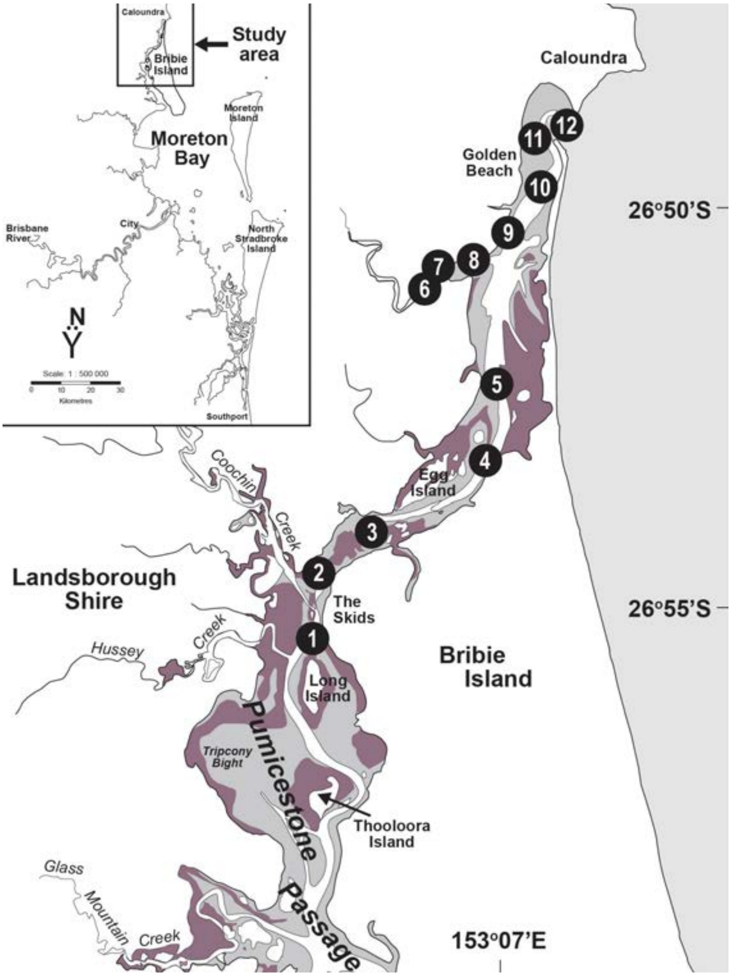

Figure 2. Map of part of Moreton Bay showing sampling locations in upper Pumicestone Passage.

Results

Taxonomy

Sixty-seven species of cumaceans were recorded in samples from the 40 sites in Moreton Bay (Table 1), thirty-six of which were new. Descriptions of fourteen of the new species have so far been published (Tafe & Greenwood, 1996a; b). Descriptions of the remaining 22 new species are being prepared for publication. Information on capture locations and relative abundance of all species are given in Tafe (1995). Taxonomic identification keys have been developed to include all species now known from the Moreton Bay region (Tafe, 1995; Tafe & Greenwood, 1996b). Scanning electron micrographs and/or scale drawings are provided for all described species in Tafe & Greenwood (1996a; b).

The life history stages of *Cyclaspis ornosculpta* Tafe & Greenwood, showing progressive development of sexual dimorphism, have been described and figured for both sexes from juvenile to mature adult (Tafe, 1995). Tafe *et al*. (this volume) details the faunal composition, abundance and distribution of cumaceans in this study site.

Distribution patterns

Twenty-two species of cumaceans were found to inhabit the estuarine system, six of which accounted for 99% of total numbers taken. Salinity varied from brackish (Site 1: minimum salinity 0.9 PSU), in creeks and tributaries, to fully marine at the mouth (Site 12: maximum salinity 35.4 PSU).

Explanations for differences in species distribution patterns of the six dominant species were sought through correlation with site, water temperature, salinity, and substrate type. The most commonly occurring species, *Gephyrocuma repandum* Hale, accounted for 48% of the total number. This species predominated in fully marine conditions over sandy substrates but was never found in brackish waters over muddy substrates. In the latter areas *Cyclaspis ornosculpta* Tafe & Greenwood, *Paradiastylis mollis* Hale and *Picrocuma crudgingtoni* Tafe & Greenwood co-dominated.

Cluster analysis suggested that the 12 sampling sites formed three lineages at the 0.8 dissimilarity level (Bray-Curtis index), representing topographically coherent groups based on species content. Examination of the cumacean species characteristic of these three lineages revealed that two had similar (estuarine) species assemblages, while the third had a very different (marine) species assemblage. A comparison of the cumacean fauna of the two estuarine lineages with that of the marine lineage showed that the latter had highest cumacean numbers but lowest species diversity (Shannon index), due to the predominance of *Gephyrocuma repandum* Hale.

Analysis of covariance (ANCOVA) was used to evaluate the relative influences of site (geographical location), water temperature, salinity and interaction between temperature and salinity, on spatial and temporal distributions of the six dominant cumacean species. The results indicated that the distributions of five of the species, namely *Gephyrocuma repandum* Hale, *Paradiastylis mollis* Hale, *Picrocuma crudgingtoni* Tafe & Greenwood, *Cyclaspis cretata* Hale and *Dimorphostylis* sp.nov.3, were most correlated with site, whilst that of the remaining species, *Cyclaspis ornosculpta* Tafe & Greenwood, was most correlated with temperature.

Table 1. List of species of Cumacea collected from Moreton Bay between March 1989 and March 1993.

Family BODOTRIIDAE

1 **Bodotria maculosa* Hale
2 **Bodotria armata* Tafe & Greenwood
3 **Bodotria* sp. nov. 1
4 *Cyclaspis cretata* Hale
5 *Cyclaspis fulgida* Hale
6 **Cyclaspis strigilis* Hale
7 **Cyclaspis usitata* Hale
8 **Cyclaspis cooki* Tafe & Greenwood
9 **Cyclaspis tranteri* Tafe & Greenwood
10 **Cyclaspis ornosculpta* Tafe & Greenwood
11 **Cyclaspis andersoni* Tafe & Greenwood
12 **Cyclaspis alveosculpta* Tafe & Greenwood
13 **Cyclaspis chaunosculpta* Tafe & Greenwood
14 **Cyclaspis* sp. nov. 1
15 **Cyclaspis agrenosculpta* Tafe & Greenwood
16 **Cyclaspis sp.* nov. 2
17 **Cyclaspis daviei* Tafe & Greenwood
18 **Cyclaspis sallai* Tafe & Greenwood
19 *Eocuma agrion* Zimmer
20 *Gephyrocuma repandum* Hale
21 **Gephyrocuma* sp. nov. 1
22 **Glyphocuma dentatum* Hale
23 *Glyphocuma halei* nov. Greenwood & Johnston
24 **Leptocuma barbarae* ♀afe & Greenwood
25 **Leptocuma kennedyi* Tafe & Greenwood
26 *Picrocuma poecilotum* Hale
27 **Picrocuma crudgingtoni* Tafe & Greenwood
28 *Pomacuma australiae* Zimmer
29 **Pomacuma* sp.nov. 1

Family DIASTYLIDAE

30 **Anchistylis* waitei Hale
31 **Anchistylis* sp. nov. 1
32 *Dimorphostylis australis* Foxon
33 **Dimorphostylis* sp. nov. 1
34 **Dimorphostylis* sp. nov. 2
35 **Dimorphostylis* sp. nov. 3
36 **Dimorphostylis* sp. nov. 4
37 *Paradiastylis mollis* Hale nov.
38 **Paradiastylis* sp. nov. 1♂
39 **Paradiastylis* sp. nov. 2

Family GYNODIASTYLIDAE

40 **Dicoides* sp. nov. 1
41 *Gynodiastylis attenuata* Hale
42 **Gynodiastylis inepta* Hale
43 *Gynodiastylis lata* Hale
44 **Gynodiastylis mutabilis* Hale
45 **Gynodiastylis polita* Hale
46 **Gynodiastylis truncatifrons* Hale
47 **Gynodiastylis* sp. nov.1
48 **Gynodiastylis* sp. nov. 2
49 **Gynodiastylis* sp. nov. 3
50 **Gynodiastylis* sp. nov. 4
51 **Gynodiastylis* sp. nov. 5
52 **Sheardia antennata* Hale

Family NANNASTACIDAE

53 *Campylaspis latidactyla* Hale
54 *Campylaspis minor* Hale
55 *Campylaspis triplicata* Hale nov.
56 **Campylaspis* sp.nov.1 ♀
57 **Campylaspis* sp.nov. 2
58 *Cumella hispida* Calman
59 *Cumella munroi* Hale
60 **Cumella turgidula* Hale
61 *Nannastacus inflatus* Hale
62 *Nannastacus nasutus* Zimmer
63 *Nannastacus johnstoni* Hale
64 **Nannastacus* sp. nov. 1
65 *Schizotrema aculeatum* Hale
66 **Schizotrema nudum* Tafe & Greenwood

Family LEUCONIDAE

67* *Leucon* sp. nov. 1

* indicates new record for Moreton Bay.

Diel rhythms

Diel patterns of vertical movement of cumaceans into the water column were examined at two locations in the southeast region of Moreton Bay. These sites were surveyed seasonally with respect to distributions of cumaceans at various depth levels over 24 h periods. The surveys were carried out over sand (Horseshoe Bay) and seagrass (Deanbilla Bay) substrates, to investigate the effects of substrate type on the cumacean fauna and its diel vertical migration patterns (Figure 3). The type of substrate was found to influence the timing and extent of migration, by affecting the intensity of light penetration to the sea floor. Diel changes in species distribution and abundance at three levels in the water column, were found to be correlated with changes in ambient light levels, tidal cycle and endogenous rhythm.

The results of the field surveys indicate that elements of both the Preferendum and Rate of Change hypotheses (Forward, 1988) are applicable to light responses in cumaceans. Individuals respond to sustained changes in light intensity near the lower threshold for photoresponsiveness, but do not respond to light changes outside those levels. Such levels are not absolute but are modified daily by light/dark adaptation and by endogenous rhythm. Dark adapted individuals

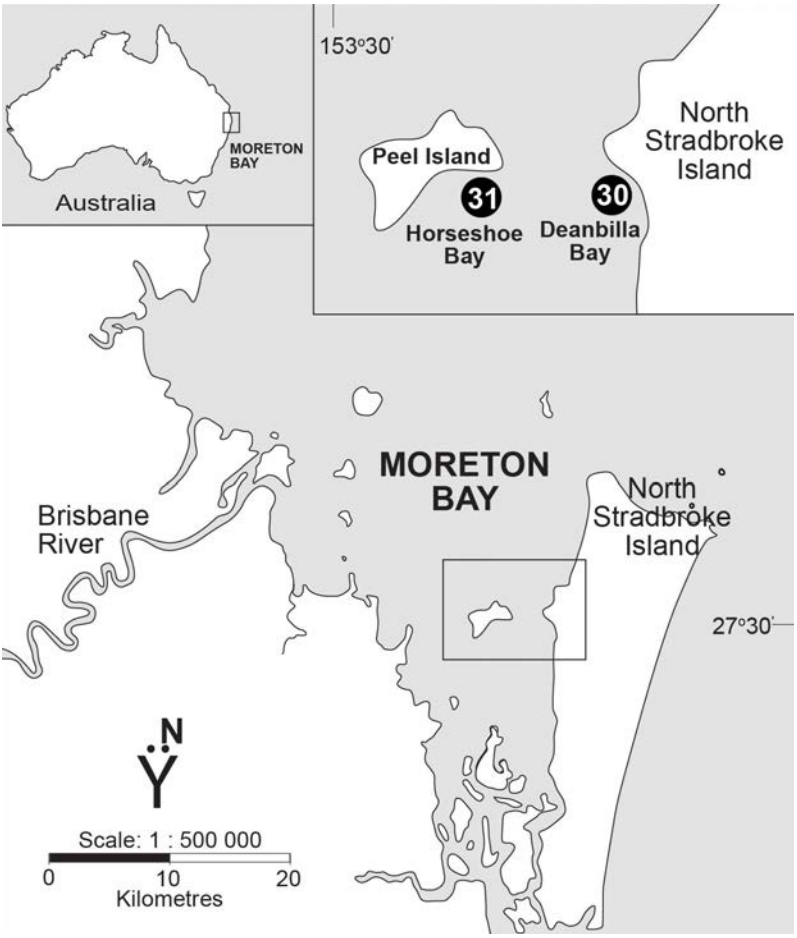

Figure 3. Map of Moreton Bay showing sampling locations in Horseshoe Bay, Peel Island and Deanbilla Bay, Stradbroke Island.

respond photonegatively to changing light regimes in the vicinity of 0.1 $\mu mol \times s^{-1} \times m^{-2}$ while light adapted individuals respond photopositively in the vicinity of 1.0 $\mu mol \times s^{-1} \times m^{-2}$. The magnitude of the response depends on the magnitude of the light change at threshold level, the size of the population, and the timing of the change relative to endogenous rhythm and tidal cycle.

Conclusion

Taxonomy

The taxonomic element of the present research represents the most comprehensive study of cumaceans undertaken in Queensland to date. Fourteen new species of cumaceans have been published and a further twenty-two new species have yet to be described from the Moreton Bay study.

Distribution patterns

Spatial and temporal distribution patterns of the six dominant species of cumaceans have been investigated along an estuarine gradient in northern Moreton Bay. However, the study was limited to one year's duration, and spatial and temporal diversities of cumacean species between successive years were not investigated.

Diel rhythms

The results of the diel vertical migration study in Horseshoe Bay showed that 21 of the 36 recorded species exhibited vertical migratory behaviour to surface waters and three of these, namely *Bodotria armata* Tafe & Greenwood, *Leptocuma barbarae* Tafe & Greenwood, and *Paradiastylis mollis* Hale, were responsible for over 95% of surface numbers in autumn (Tafe, 1995). These migrations were synchronised within species and virtually restricted to adult and sub-adult males. While adult females and juveniles also became active in the water column at the same time, they remained in the epibenthic layer (within 0.4 m above substratum), along with most of the males. Despite the fact that the number of cumaceans migrating to the surface was proportionately small, approximately one-fifth of all emergent males of selected species did undertake such migrations. The diel vertical migration phenomenon cannot be ignored, and clearly must have significance for population dynamics and dispersal of such species. These migrations may also influence the surface activity of a range of other marine fauna.

Acknowledgements

We gratefully acknolwedge Karen Holloway for improving the quality of figures and maps. The research was carried out with financial assistance from the Department of Zoology, The University of Queensland, the Queensland Museum and Reckitt & Colman Pty Ltd.

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Faunal Composition, Abundance and Distribution of Cumaceans (*Crustacea: Peracarida*) along the Estuarine Gradient of Pumicestone Passage

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Abstract

Spatial and temporal distributions of the dominant species of cumaceans were investigated along an estuarine gradient in northern Moreton Bay. Twenty-two species of cumaceans were found to inhabit the estuarine system, six of which accounted for 99% of total numbers taken. Salinity varied from brackish to fully marine. The most commonly occurring species, *Gephyrocuma repandum* Hale, accounted for 46% of the total number. This species predominated in fully marine conditions over sandy substrates. In brackish waters over muddy substrates *Cyclaspis ornosculpta* Tafe & Greenwood, *Paradiastylis mollis* Hale and *Picrocuma crudgingtoni* Tafe & Greenwood co-dominated.

Introduction

Cumaceans occur at all oceanic depths but most occur on the continental shelves with the smaller, meiofaunal species being taken in very shallow coastal waters (Watling, 1988).

Most studies on cumaceans have been taxonomic, with approximately 800 species being known worldwide (Vassilenko, 1989). Non-taxonomic studies on the group did not commence until about 1910, by which time approximately 150 species were known (Dartnall, 1981).

In Australia early taxonomic work was carried out by H.M. Hale between 1928 and 1953. He described a large number of new species, mainly from southern coasts, and estimated that at least 160 species inhabit Australian coastal waters. As Hale (1953) noted, very little is known of the Cumacea of the northern coastal region of Australia. The situation has not changed much since that time, the only subsequent taxonomic studies in Australia being those of Johnston (1966), Greenwood & Johnston (1967), Watling (1991a; b) and Bacescu (1990; 1991; 1992b).

Investigations of spatial and temporal distributions of macrobenthos in Moreton Bay, have previously been carried out by Stephenson *et al.* (1970; 1974; 1976; 1977; 1978). Taxonomic and distributional investigations of plankton in Moreton Bay have been carried out by Greenwood, (1971; 1972; 1973; 1976; 1977; 1978a; b; c; d; 1979; 1980; 1981; 1982a; b) and Greenwood *et al.* (1991; this volume). A comprehensive listing of zooplankton studies is provided in Greenwood (this volume). The present study examines spatial and temporal distributions of cumaceans along an estuarine gradient in the northern region of Moreton Bay. It is the first such study on cumaceans to be carried out in Queensland.

Pumicestone Passage (Figure 1) was selected for this study because it is an extensive and relatively pristine mangrove-fringed subtropical estuarine system, with considerable structural diversity. It has a salinity gradient which varies from almost zero (site 1, Figure 1) at certain times of the year, to almost fully marine (site 12, Figure 1). The sediment type varies from silt, in the upper reaches of some creeks and tributaries, to sand at the northern entrance. The

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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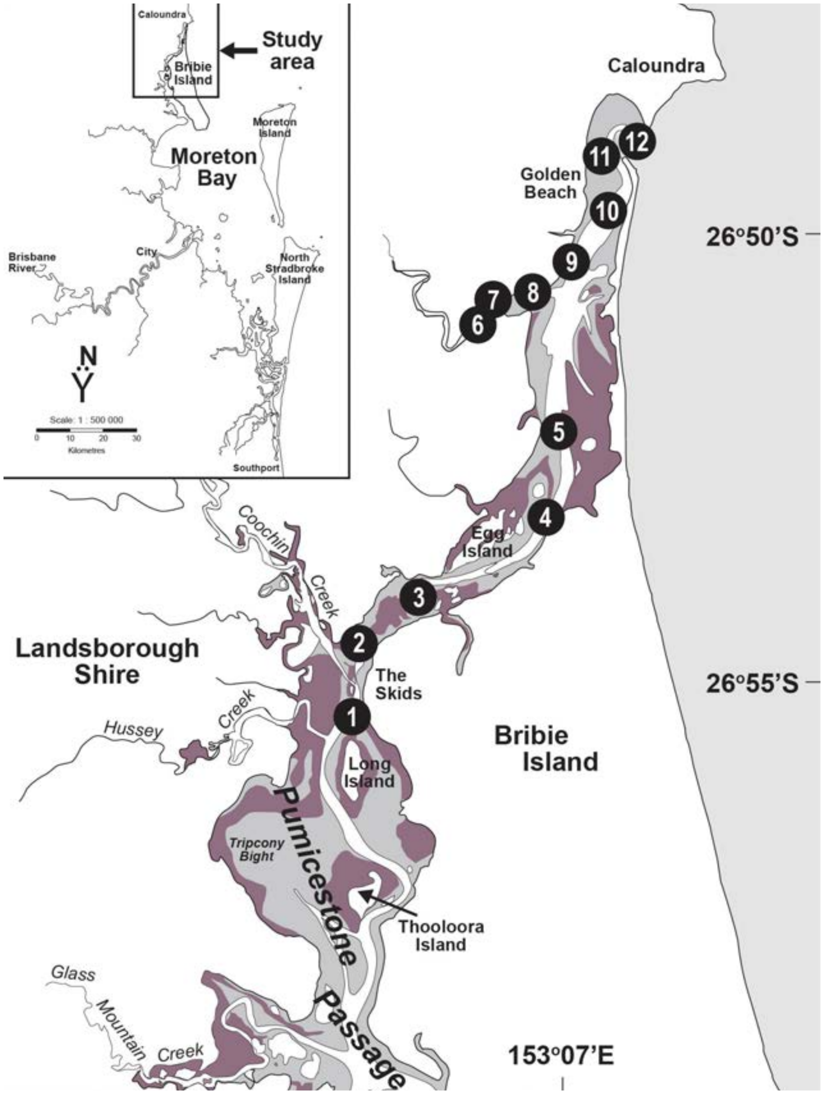

Figure 1. Map of part of Moreton Bay showing sampling locations in upper Pumicestone Passage.

estuary is a major nursery environment for at least 60 species of fish (Pham *et al.,* this volume), many of which may be expected to utilise cumaceans in their diets. Tafe & Greenwood (this volume) recorded four out of the five most abundant fish species in western Moreton Bay as consuming cumaceans. Juvenile fish are major predators of meiofauna and many pass through an "obligatory" meiobenthic feeding stage (Castel, 1992; Coull, 1988; 1990; Coull *et al.,* 1995; de Morais & Bodiou, 1984).

This study forms part of a broader investigation by Greenwood & Greenwood (see abstract of Greenwood, this volume) on the macrozooplankton/micronekton of Pumicestone Passage. Much of the present study was carried out as part of a doctoral thesis (Tafe, 1995).

Procedure

Twelve sampling sites in upper Pumicestone Passage (26°50’S, 153°7’E) (Figure 1) were sampled at lunar, fortnightly intervals over a 12 month period, using a single mouth sledge net sampler. In total 312 zooplankton samples were collected between January 1990 and March 1991 from the twelve sampling sites. The sampler was equipped with a flowmeter (Oceanics Model 2030) to indicate volume filtered per sample, and all hauls were taken just above the substrate at a towing speed of 1-2 knots. The sampler (mouth size: 50 x 30 cm; mesh size: 500 µm) collected epibenthic organisms between 5 and 35 cm above the substrate. All sampling was carried out after sunset from a four metre motorised vessel. Near-surface and near-bottom water temperatures and salinities were recorded for each site at the time of sampling. Samples were fixed in 4% buffered formalin and returned to the laboratory. All cumaceans were removed from samples, except where numbers exceeded 500 individuals per sample. In such cases (4% of samples) a Folsom splitter was used to obtain a subsample, from which cumaceans were removed. Individuals from each sample or subsample were sorted into species and enumerated.

Terms used in the results section to refer to cumacean numbers taken at various sites are: rare (isolated captures), uncommon (< 1 individual × m^{-3}), common (1-10 individuals × m^3), and very common (>10 individuals × m^{-3}).

Results

Twenty-two species of cumaceans were identified from the zooplankton samples. Ten of the 22 species are new, and 12 are new records for Moreton Bay (Table 1). Published descriptions have since been given for five of the new species (Tafe & Greenwood, 1996a). Only one or two specimens of each of *Eocuma agrion*, *Campylaspis latidactyla* and *Gephyrocuma* sp. nov. 1, were taken. Therefore these species are not included in further analyses, though their species names are included in a rank order of recorded cumacean species (Table 1). Males and females of *Cyclaspis ornosculpta* were at first thought to belong to two separate species. Subsequent examination of developmental stages of both sexes indicated sexual dimorphism within a single species. Temporal patterns of distribution of the sexes support the view that they belong to one species (Figure 2C). A new cumacean species, *Schizotrema nudum* (Tafe & Greenwood, 1996b), which was found in some parts of Moreton Bay, was not encountered in Pumicestone Passage.

Temporal patterns

Five species accounted for 97% of the total number of cumaceans taken (Table 1). Consequently the seasonal abundance pattern of all cumaceans combined, mirrors that of the first five species (Figure 2A). Autumn and winter cumacean abundances in the seasonal pattern were largely due to increased numbers of *Gephyrocuma repandum* (Figure 2B). The summer maximum was largely due to increased numbers of *Cyclaspis ornosculpta* at that time (Figure 2C).

The most commonly occurring species, *Gephyrocuma repandum*, accounted for 46% of the total number of cumaceans taken (Table 1). In winter, when it was most abundant, it accounted for 83% (Table 2). Overall abundance of this species decreased during spring and summer (Figure 3), however, it was common or very common at the three northern most sites throughout the year (Figure 4A). Another bodotriid, *Cyclaspis ornosculpta*, was common in spring and summer but numbers decreased markedly during autumn and winter (Figure 4B). In summer this species accounted for 59% of total cumacean numbers, whilst in winter it was responsible for less than 1% of numbers (Table 2). *Cyclaspis ornosculpta* was not present at the seven southern most sites during autumn and most of winter, 97% of its annual catch being taken during the six months of spring and summer. *Paradiastylis mollis* also exhibited a high

Table 1. Abundance rank order of cumacean species taken from Pumicestone Passage. A = percentage abundance over sampling period; B = number of sampling sites at which each species was taken; C = species group**.

Species		A	B	**C
1	*Gephyrocuma repandum* Hale	46.18%	6	M
*2	*Cyclaspis ornosculpta* Tafe & Greenwood	27.83%	12	U
3	*Paradiastylis mollis* Hale	11.51%	11	U
*4	*Picrocuma crudgingtoni* Tafe & Greenwood	8.62%	12	U
5	*Cyclaspis cretata* Hale	2.82%	11	U
*6	*Dimorphostylis* sp. nov. 3	2.18%	9	E
7	*Dimorphostylis australis* Foxon	0.50%	3	M
*8	*Anchistylis* sp. nov. 1 +	0.15%	3	M
9	*Gynodiastylis lata* Hale	0.07%	5	M
*10	*Dimorphostylis* sp. nov. 4	0.04%	3	M
11	*Campylaspis triplicata* Hale	0.03%	4	M
*12	*Cyclaspic cooki* Tafe & Greenwood	0.02%	1	M
*13	*Cyclaspis alveosculpta* Tafe & Greenwood	0.02%	2	M
14	*Cumella hispida* Calman	0.01%	1	M
15	*Schizotrema aculeatum* Hale	0.01%	1	M
*16	*Anchistylis waitei* Hale	0.01%	1	M
*17	*Cyclaspis daviei* Tafe & Greenwood	<0.01	1	M
*18	*Gynodiastylis* sp. nov. 1	<0.01	1	M
*19	*Gynodiastylis truncatifrons* Hale	<0.01	1	M
20	*Eocuma agrion* Zimmer	<0.01	1	M
21	*Campylaspis latidactyla* Hale	<0.01	1	M
*22	*Gephyrocuma* sp. nov. 1 +	<0.01	1	M
		Total 100.00%		

* Indicates new record for Moreton Bay;
\+ Only known from Pumicestone Passage;
** Species group: U = ubiquitous (present at 11 of 12 sampling sites);
M = marine (at least 90% of numbers taken at 3 northern sites);
E = estuarine (at least 90% of numbers taken at 9 southern sites).

degree of seasonality, being absent from sites 1-8 during autumn and most of winter (Figure 4C). The seasonal abundance of *Paradiastylis mollis* reached a maximum in summer, when it accounted for 19% of cumacean numbers (Figure 3). In autumn and winter combined the species accounted for only 2% of cumacean numbers (Table 2). Ninety-two percent of the annual catch of *Paradiastylis mollis* was taken in spring and summer, with 68% in summer alone (Figure 3). *Picrocuma crudgingtoni*, the fourth most abundant species, was present at a number of sites throughout the year (Figure 4D). It accounted for at least 10% of cumacean numbers in each of the seasons, autumn, winter and spring, and 4% in summer (Table 2). *Cyclaspis cretata*, the fifth most abundant species, occurred at the three northern sites throughout the year, but only sporadically at the other sites (Figure 4E). It was most common in summer, accounting for 3.5% of all cumaceans in this season (Table 2).

Dimorphostylis sp. nov. 3 exhibited almost the opposite trend in seasonal abundance to *P. mollis*, being most abundant at sites 1-9 during winter and spring (Figure 4F). Eighty-three percent of the annual catch of this species was taken in winter and spring, and only 6% in summer. The numbers taken in spring, when it was most abundant, represented 5.8% of all cumaceans in this season (Table 2).

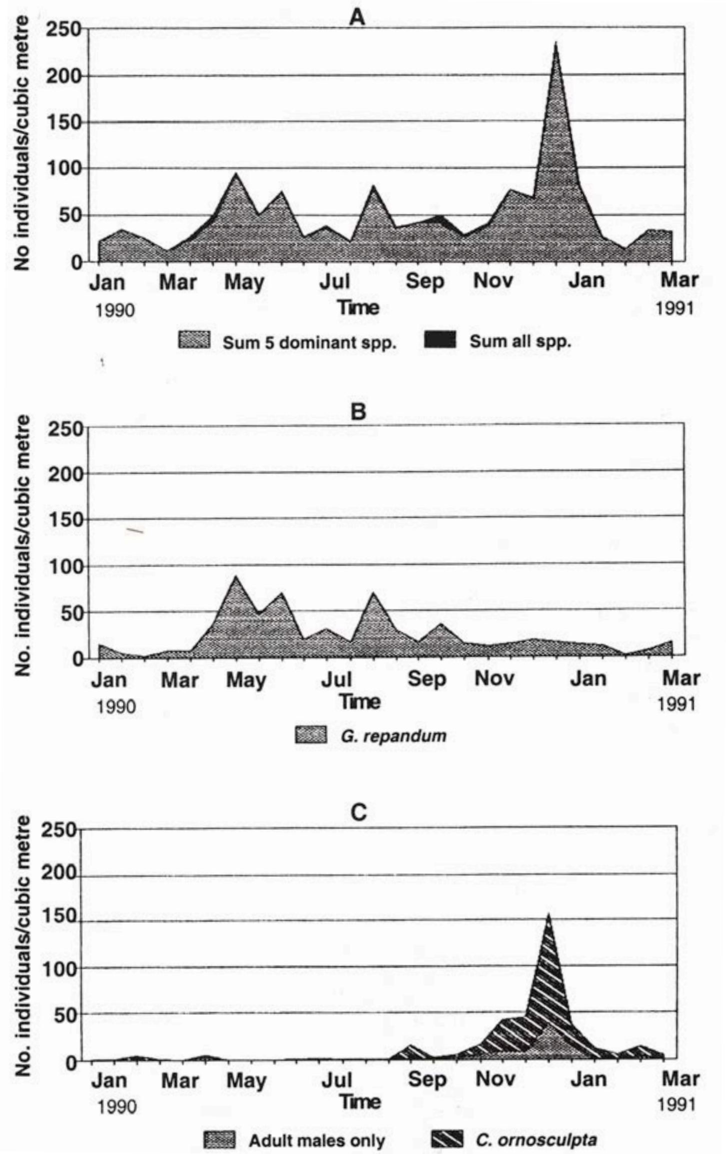

Figure 2. Abundance of cumaceans in Pumicestone Passage between January 1990 and March 1991. **A** – abundance of the five dominant species, and all species; **B** – abundance of *Gephyrocuma repandum*; **C** – abundance of *Cyclaspis ornosculpta*; adult males only, and all individuals.

Spatial patterns – Species groups

Three species groups were identifiable from spatial patterns of distribution (Table 1). Group U (ubiquitous) included species present at no less than 11 of the 12 sampling sites. Four of the five most common species are classified as ubiquitous (group U) species. Group M (marine) species: at least 90% of their numbers were taken at the three most northerly sites (which are typically marine). *Gephyrocuma repandum*, the most abundant species captured, was only taken at these sites. Seventeen of the 22 species recorded in Table 1 are classified into group M.

Table 2. Percentage abundance of cumacean species in each season. Species are listed in the same order as in Table 1.

	Species	Autumn (%)	Winter (%)	Spring (%)	Summer (%)
1	*Gephyrocuma repandum* Hale	79.20	83.56	37.65	14.21
2	*Cyclaspis ornosculpta* Tafe & Greenwood	1.75	1.00	33.99	58.79
3	*Paradiastylis mollis* Hale	0.92	1.15	6.82	18.88
4	*Picrocuma crudgingtoni* Tafe & Greenwood	10.10	10.35	14.35	3.95
5	*Cyclaspis cretata* Hale	3.05	0.45	1.05	3.46
6	*Dimorphostylis* sp. nov. 3	1.13	3.24	5.75	0.63
7	*Dimorphostylis australis* Foxon	2.86	0.00	0.00	0.00
8	*Anchistylis* sp. nov. 1 +	0.49	0.15	0.15	0.00
9	*Gynodiastylis lata* Hale	0.23	0.07	0.03	0.01
10	*Dimorphostylis* sp. nov. 4	0.04	0.00	0.11	0.04
11	*Campylaspis triplicata* Hale	0.13	0.03	0.00	0.00
12	*Cyclaspic cooki* Tafe & Greenwood	0.11	0.00	0.00	0.01
13	*Cyclaspis alveosculpta* Tafe & Greenwood	0.00	0.04	0.08	0.00
14	*Cumella hispida* Calman	0.00	0.00	0.00	0.02
15	*Schizotrema aculeatum* Hale	0.00	0.03	0.00	0.00
16	*Anchistylis waitei* Hale	0.00	0.03	0.00	0.00
17	*Cyclaspis daviei* Tafe & Greenwood	0.00	0.00	0.02	0.00
18	*Gynodiastylis* sp. nov. 1	0.00	0.00	0.00	0.00
19	*Gynodiastylis truncati frons* Hale	0.00	0.00	0.02	0.00
20	*Eocuma agrion* Zimmer	0.00	0.00	0.00	0.01
21	*Campylaspis latidactyla* Hale	0.01	0.00	0.00	0.00
22	*Gephyrocuma* sp. nov. 1 +	0.01	0.00	0.00	0.00
	Total	100.02	100.09	100.01	100.01

+ Only known from Pumicestone Passage.

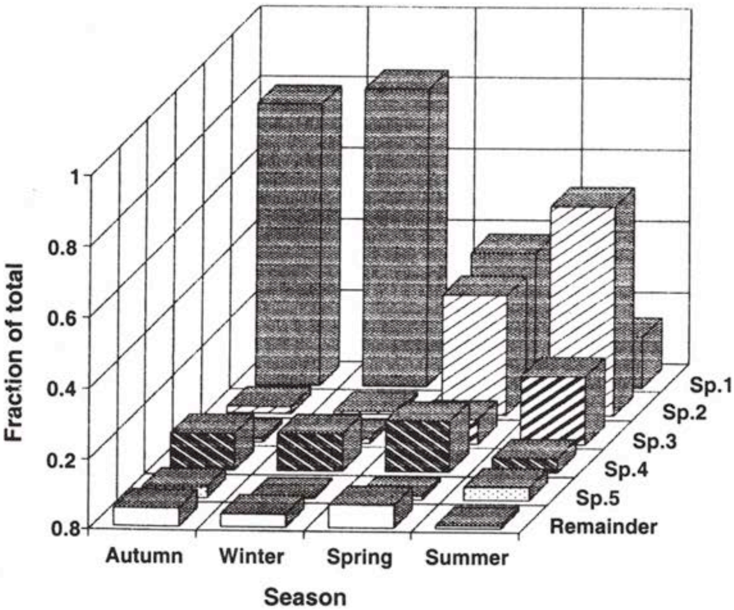

Figure 3. Species fractions of seasonal abundance in Pumicestone Passage between March 1990 and February 1991. **Sp.1** – *Gephyrocuma repandum*; **Sp.2** – *Cyclaspis ornosculpta*; **Sp.3** – *Paradiastylis mollis*; **Sp 4** – *Picrocuma crudgingtoni*; **Sp.5** – *Cyclaspis cretata*.

Group E: species for which at least 90% of numbers were taken at the nine most southerly sites, where conditions were estuarine. In the present analysis only *Dimorphostylis* sp. nov. 3 is classified as an "estuarine" (group E) species.

Discussion

Gephyrocuma repandum is the only commonly found species which was restricted throughout the year to the northern region of Pumicestone Passage (Figure 4A). Most of the other commonly found species were more widespread in their distributions, as shown by their abundance patterns (Figures 4B-4F).

The ubiquitous group of species included *Cyclaspis ornosculpta*, *Paradiastylis mollis*, *Picrocuma crudgingtoni* and *Cyclaspis cretata*. The one estuarine species, *Dimorphostylis* sp. nov. 3, was commonly found in the upper reaches of the estuary but never at the mouth.

Fluctuations in abundance

Fluctuations in the abundance of benthic organisms, including cumaceans, in Moreton Bay are postulated by Stephenson (1980a) to be strongly influenced by predation pressure and abiotic factors, particularly those related to variations in rainfall. Periods of heavy rainfall are related to increased recruitment rates of some prawn and fish species, which in turn are related to depletion rates in macrobenthic species. He suggested that macrobenthic species with strong cycles in abundance appeared to be under most control by organisms which preyed upon them or disturbed their habitat. Four species of cumaceans from the family Bodotriidae were among the cyclic species recorded by him. All were recorded over sandy substrates, as they were in the present study. Both studies indicated that their geographical distributions within Moreton Bay were centred around shallow, marine habitats. Salinity appeared to be a major abiotic factor influencing spatial and temporal distributions of macrobenthos in Moreton Bay (Stephenson, 1980a; b). In the present study spatial and temporal distribution patterns of the six most common species appeared to be influenced by seasonal fluctuations in salinity and temperature. The distribution patterns of a large number of macrobenthos are directly related to fluctuations in salinity (Stephenson, 1980a). A comparative analysis of zooplankton from different salinities along a river system showed that a much richer, more diverse fauna existed at higher salinities (Tafe, 1990). This conclusion is supported by the results of a number of other studies (Wickstead, 1961; 1962; Okera, 1974; Tafe & Griffiths, 1983). In the present study two species of cumaceans, *Picrocuma crudgingtoni* and *Dimorphostylis* sp. nov. 3, were taken at salinities below 18. These were the only two cumacean species encountered in Pumicestone Passage which appeared to be able to tolerate low salinity (<20) for sustained periods of time. *Picrocuma crudgingtoni,* as well as 20 other species, was also taken at a salinity of 35.

Acknowledgements

We gratefully acknowledge Karen Holloway for improving the quality of figures and maps. The research was carried out with financial assistance from the Department of Zoology, The University of Queensland, the Queensland Museum and Reckitt & Colman Pty Ltd.

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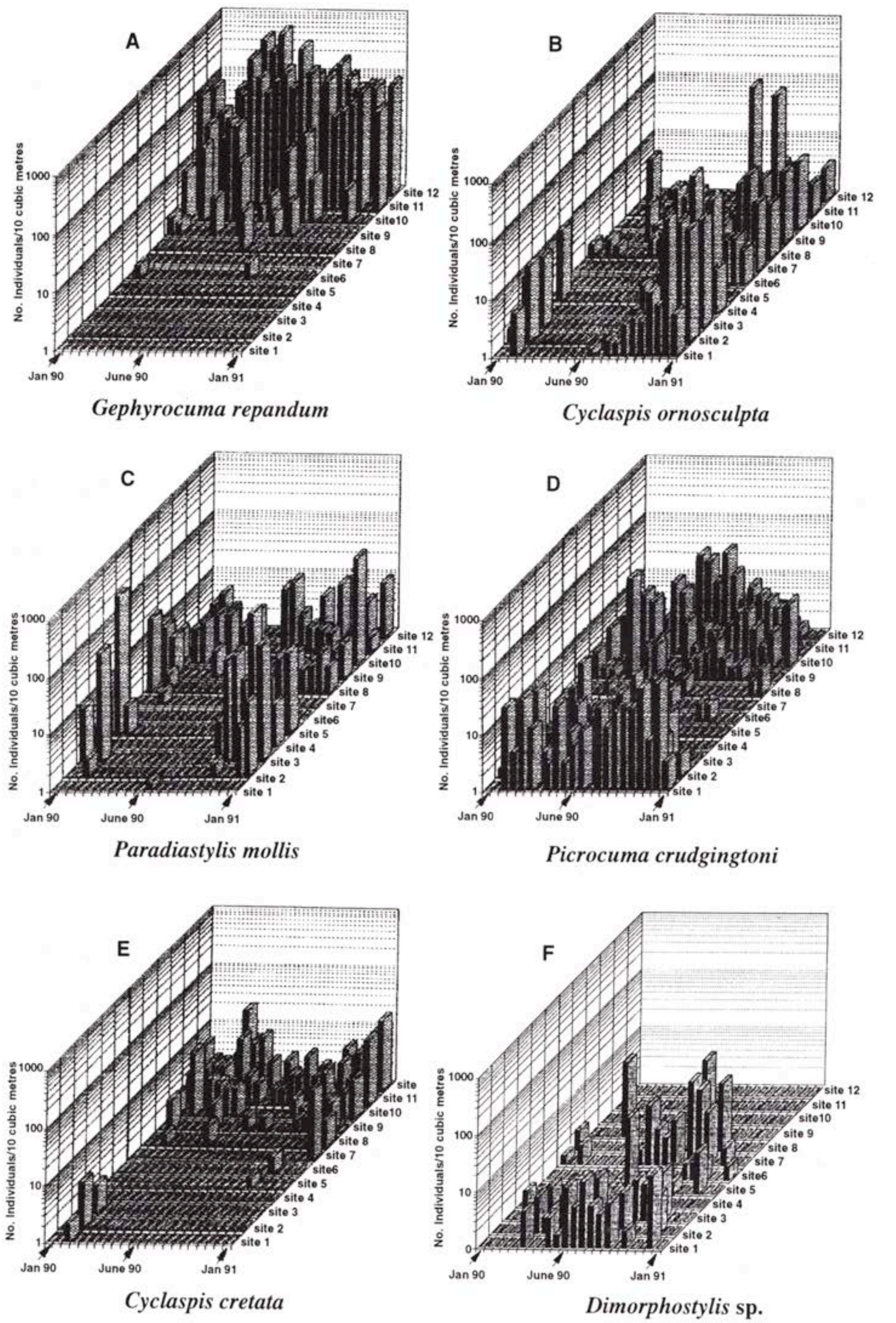

Figure 4. Abundance of each of the six dominant cumacean species at twelve sampling sites in Pumicestone passage between January 1990 and January 1991.

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Fish Larvae of Pumicestone Passage, Moreton Bay

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Abstract

A total of 33 916 fish larvae from 37 families and 71 species was caught along the axis of the northern half of the Pumicestone Passage, an estuarine system in the northern portion of Moreton Bay, southeast Queensland. Gobiidae were the most numerically abundant (59%), followed by Clupeidae (11%), Engraulidae (8%), Ambassidae (7.5%), Sillaginidae (5%) and Gerreidae (4%) with 31 families making up the remaining 5%. Various species of gobies had a restricted distribution within Pumicestone Passage indicating they may be residents, whereas clupeids, engraulids and sillaginids were found at all sampling sites. Fish larvae were most abundant during summer (between October and February). The numerical dominance of gobies in this system is similar to the situation found in Botany Bay (New South Wales) and Hopkins Estuary (South Australia), but contrasts with that in Swan Estuary, Wilson Inlet (Western Australia) and Port Philip Bay (Victoria) where engraulids and clupeids are the most abundant groups. Fish larvae were more abundant in mid-regions of the estuary, whereas species diversity was highest in areas close to the estuary mouth.

Introduction

From early this century, it has been recognised that biological and ecological knowledge of the early life history of fish is necessary for the reliable prediction of adult fish populations (Hempel, 1979). Most early works on fish larvae centred on economically important species, however, in the last few decades there has been a realisation that non-commercial species are ecologically important, and there has been an enhanced interest in the larval biology of such species (Miskiewicz, 1987; Gaughan *et al.*, 1990; Steffe, 1991; Neira *et al.*, 1992; Yoklavich *et al.*, 1992; Harris & Cyrus, 1995).

The present study was undertaken in Pumicestone Passage, an extensive mangrove-lined double-ended estuarine system in northern Moreton Bay. Although 12% of the commercial and 35% of the recreational fisheries catch from Queensland waters is from Moreton Bay (Quinn, 1992), there have been only three studies of larval fish in the region. Tosh (1902) studied the morphology of *Sillago ciliata* (as *S. bassensis*). He subsequently provided a species list of common fish larvae found in the Bay (Tosh, 1903). Helbig (1969) provided data on the distribution of larval fish, mostly at family level, at three sites over a 15 month period. Helbig's study was qualitative and yielded little ecological information because many of his taxa were multi-specific. Despite the presumed importance of Pumicestone Passage as a fish nursery (Quinn, 1992), and hence as a source for fish recruitment into the Moreton Bay fishery, there have been no previous studies on the dynamics of fish larvae in that system.

The aims of this study were to determine the species composition and abundance of fish larvae in the northern half of Pumicestone Passage, and to investigate spatial (along the estuarine axis, and between sub-surface and near-bottom water layers) and temporal patterns in composition and abundance of fish larvae.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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Materials and Methods

Sampling site locations and characteristics

All biological samples for the study of the horizontal distribution of fish larvae were collected in mid channel from ten sites along the axis of the northern half of Pumicestone Passage (Figure1). These sampling sites were chosen because of their geographically distinctive conditions and to span a range from near-oceanic to upper-estuarine conditions. Water depth at these sampling sites ranged from 1.5 m (sites 1, 2) to 6 m (sites 4, 5) with a mean depth of 4 m. Maximum tidal current velocities ranged from slow moving (0.5 m/s at sites 1 & 2) to rapidly flowing (2 m/s at sites 9 & 10). Substrata varied from sand at the tidal node (sites 1 & 2), through mud (site 3), muddy-sand (sites 4 & 5) and sandy-mud (sites 6 & 7), to sand again at the estuary mouth (sites 8, 9 & 10) (Tafe, 1995). Surface and bottom water temperature and salinity were measured at each of the ten sites using the Stansens YSI model 33 S-C-T, prior to biological samplings.

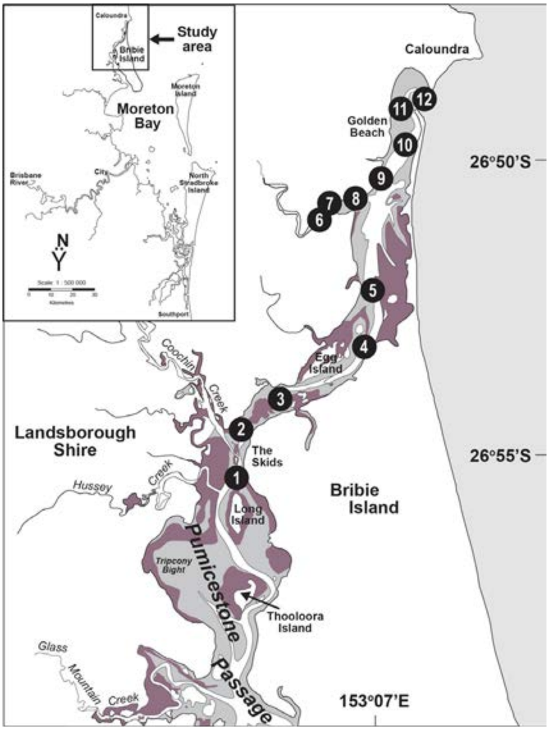

Figure 1. The northern half of Pumicestone Passage showing sampling sites 1-10.

Horizontal distribution

Fish larvae were collected between April 1990 and March 1992 from the near-bottom water layer at each of the sampling sites using a 2 m long sledge-mounted net with 500 µm mesh and mouth dimensions 0.5 m x 0.3 m. A General Oceanics Digital flow meter was fitted in the centre of the mouth of the net to allow calculation of the amount of water filtered. Samples were taken at lunar fortnightly intervals in the first twelve months, and at lunar monthly intervals in the following 12 months, so that similar tidal and diel conditions existed at each sampling occasion. All samples were collected after dusk, and within approximately 2 h of local slack high water. The net was towed for 10 mins at a speed of 1 m/s.

Vertical distribution

Samples were collected sequentially from the sub-surface and near-bottom water at lunar monthly intervals in the period from January to December 1995. Sub-surface samples were collected using a push net system (Pham & Greenwood, 1998), and near-bottom samples with a sledge net. All nets were designed to have no structures in front of the net mouth, were made of 500 mm mesh, were 2 m long and had a rectangular mouth measuring 1 m x 0.6 m with a centrally mounted General Oceanics Digital flow meter. The surface push nets collected samples in front of the boat wash; the bottom net was towed approximately 30 m behind the boat to minimise propeller influences. Duration of towing was 5 mins at a speed of approximately 1 m/s. At the end of the deployment, plankton was immediately preserved in a mixture of sea water and 10% formalin. In the laboratory, larval fish were removed from the rest of the zooplankton, identified to the lowest possible taxon based on morphological characters given in various sources (Leis & Rennis, 1983; Moser *et al*. 1984; Okiyama, 1988; Leis & Trnski, 1989), and enumerated.

Results

Water temperature and salinity

There was little variation between sampling sites in water temperature during the study period. The mean monthly water temperature ranged from 16ºC in July/August (winter) to 30ºC in December/January (summer). Temperatures of surface and bottom waters were similar throughout except in November 1995 when surface waters were 1.3^{o}C cooler.

Salinity varied little between sampling sites during low rainfall periods (Figure 2A; B) (range approximately 2 ppt), but in April 1990, December 1991 and February 1995 following periods of prolonged rainfall, salinities throughout the system were reduced, and the range between up-estuary (site 1) and down-estuary (site 10) locations was 13-15 ppt. Differences between surface and bottom water salinities were only evident following periods of heavy rain (Figure 3A; B).

Species composition of fish larvae

The larvae of 71 species from 37 families were recorded from bottom waters and the larvae of 61 species belonging to 24 families from surface waters (Table 1). Both in terms of the number of species (Table 1) and numbers of individuals captured (Figure 4), most larvae belonged to the Gobiidae, Clupeidae, Engraulidae, Ambassidae, Gerreidae, Sillaginidae and Platycephalidae. The remainder occurred in small numbers and infrequently and are categorised elsewhere as 'Others' (Table 2).

Patterns of temporal and spatial distributions and species diversities

Temporal variation patterns

The mean abundance of fish larvae in the Pumicestone Passage was lowest in July/August (ca. 30 larvae/100m^3) and increased to a maximum in November/December 1990 (ca. 800 larvae/100m^3), and in March 1995 (ca. 500 larvae/100m^3) (Figures 5A and 5B). Overall, fish larvae in Pumicestone Passage were in greatest abundance during December/January (Figure 6), their numbers declining to near zero during May-September. Numbers increased rapidly during October-December. Species diversity was highest during the warm water period of the year (Figure 7), coinciding with the period of greatest abundance of larval fish.

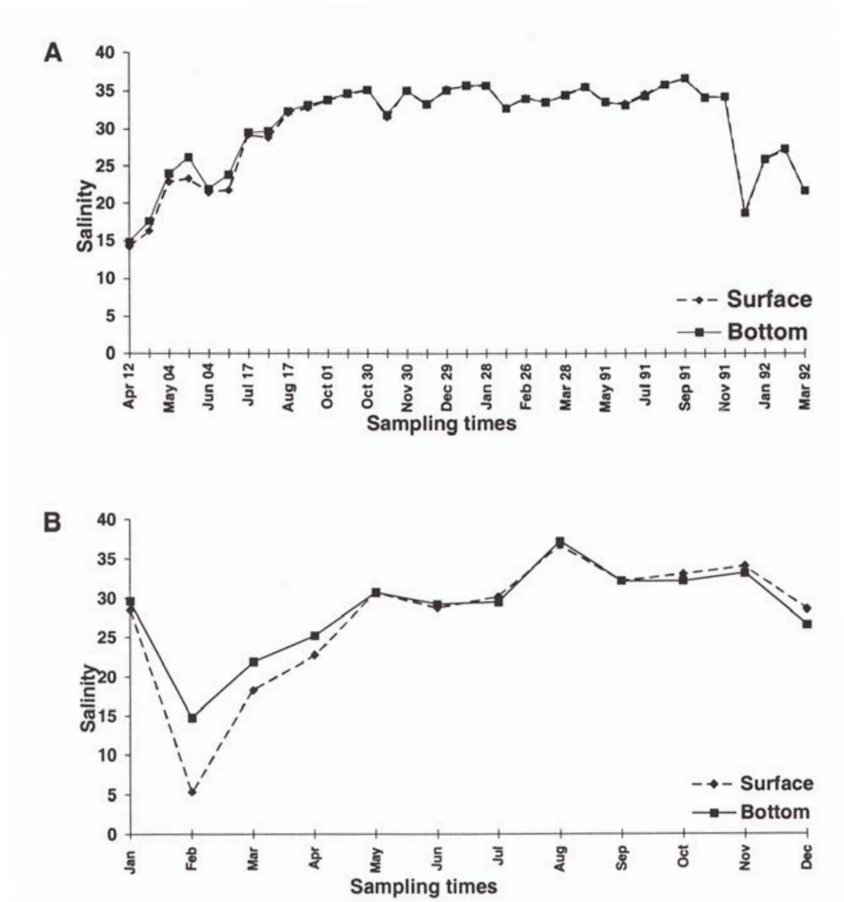

Figure 2. Mean surface and bottom salinities across all sampling sites in Pumicestone Passage from: **A.** April 1990 to March 1991; **B.** January to December 1995.

Spatial variation patterns

Along the estuarine axis

Gobies were the dominant group and contributed most to the abundance of larval fish within these sampling sites (Figure 4), even though different species of goby occurred at different locations (Table 2). Higher concentrations of fish larvae were recorded mainly in the mid-stream region of sites 3, 4 and 5 (Figure 8). Diversity was lowest at site 2 and greatest at site 8, showing a trend of increasing diversity towards the sea entrance (Figure 9).

Vertically in the water column

While the taxonomic composition and numerical abundance of the fish larvae differed between sites, overall greater species richness and abundance were found in near-bottom samples (60% of the total catch) than in the sub-surface samples (40% of the total catch) (Figure 4). Species diversity was higher in the near bottom water layers (Figure 7).

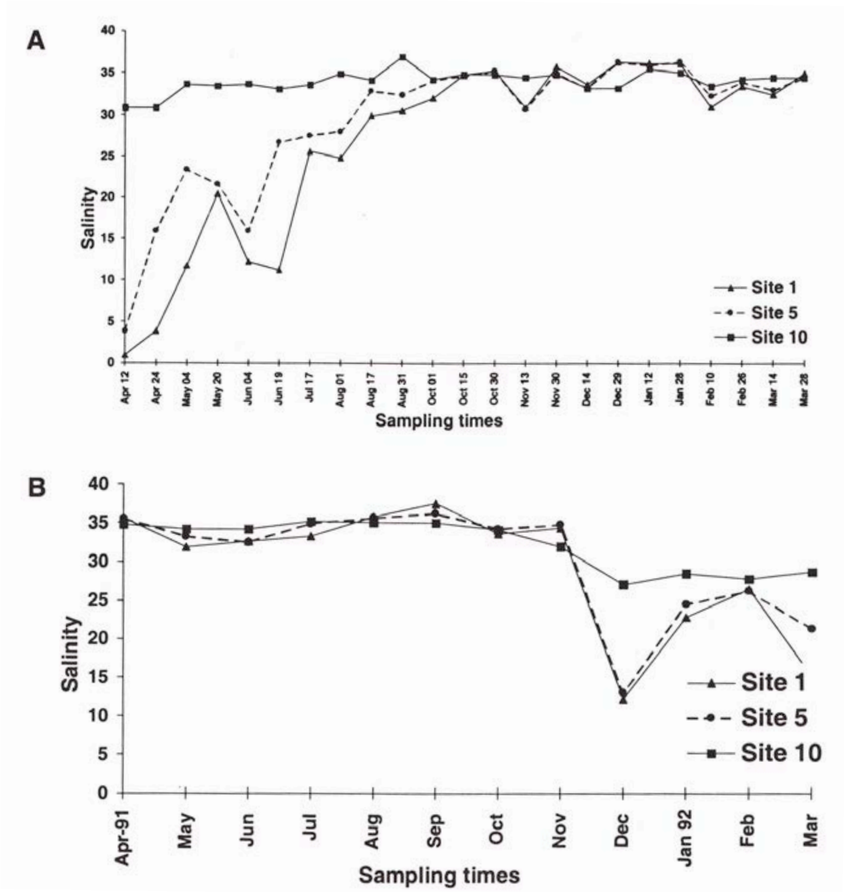

Figure 3. Variations of salinities at site 1 (upper estuary), site 5 (mid-estuary) and site 10 (estuary mouth) during the period from: **A.** April 1990 to March 1991; **B.** April 1991 to March 1992 (monthly sampling).

Discussion

Larval fish species composition

The mix of estuarine and marine species found in the larval fish assemblage of Pumicestone Passage is characteristic of marine embayments and estuarine systems elsewhere in the world (USA – Pearcy & Myers, 1974; Houde & Alpern-Lovdal, 1984; Bourne & Govoni, 1988; Yoklavich *et al*., 1992; Japan – Yamashita & Aoyama, 1984; Taiwan – Tzeng & Wang, 1992; South Africa – Melville-Smith & Baird, 1980; Melville-Smith, 1981; Beckley, 1984; 1986; Whitfield, 1989; Harrison & Whitfield, 1990; Harris & Cyrus, 1995; New Zealand – Roper, 1986), and in Australia (Western Australia – Gaughan *et al*., 1990; Neira *et al*., 1992; Victoria – Jenkins, 1986; New South Wales – Miskiewciz, 1987; Steffe & Pease, 1988; Steffe, 1991; Queensland – Morton, 1989).

The larval fish assemblage in Pumicestone Passage is characterised by only a few very abundant species. This is a common characteristic of ichthyofauna in estuarine systems. In this study, larvae of the Gobiidae dominated, being about 60% of the total catch. This dominance by Gobiidae larvae has been widely observed in similar studies elsewhere (Melville-Smith &

Table 1. Larval fish species recorded in Pumicestone Passage.

Family	Near-bottom samples	Sub-surface samples	Common names
Gobiidae	*Gobiopterus semivestita*	*Gobiopterus semivestita*	Goby
	Pseudogobius olorum	*Pseudogobius olorum*	Goby
	Favonigobius exquisitus	*Favonigobius exquisitus*	Goby
	Unidentified goby	Unidentified goby	Goby
Clupeidae	*Hyperlophus translucidus*	*Hyperlophus translucidus*	Herring
	Sardinops neopilchardus	N / A	Pilchard
Engraulidae	*Engraulis australis*	*Engraulis australis*	Australian anchovy
	Stolephorus carpentariae	*Stolephorus carpentariae*	Carpentaria anchovy
	Stolephorus indicus	N / A	Indian anchovy
	Stolephorus devisi	N / A	Devis anchovy
Ambassidae	*Ambassis jacksoniensis*	*Ambassis jacksoniensis*	Glassfish
	Ambassis marianus	*Ambassis marianus*	Glassfish
Gerreidae	*Gerres subfasciatus*	*Gerres subfasciatus*	Siver biddies
Sillaginidae	N / A	*Sillago analis*	Goldenlined whiting
	Sillago ciliata	*Sillago ciliata*	Sand whiting
	Sillago maculata	N / A	Winter whiting
Platycephalidae	*Platycephalus fuscus*	*Platycephalus fuscus*	Flathead
	Platycephalus speculator	N / A	Flathead
Hemiramphidae	N / A	*Hemiramphus* sp.	Garfish
	Hyporhamphus sp.	*Hyporhamphus* sp.	Garfish
	Arrhamphus sp.	*Arrhamphus* sp.	Garfish
Mugillidae	*Mugil cephalus*	*Mugil cephalus*	Sea mullet
	Liza argentea	*Liza argentea*	Mangrove mullet
Atherinidae	*Atherinomorus ogilbyi*	*Atherinomorus ogilbyi*	Hardyhead
Leiognathidae	*Leiognathus* sp.	*Leiognathus* sp.	Ponyfish
Bothidae	Unidentified	Unidentified	Left-handed flounder
Callyonimidae	Unidentified	Unidentified	Dragonets
Carangidae	Unidentified	Unidentified	Trevally
Congridae	Unidentified	Unidentified	Eel
Monacanthidae	Unidentified	Unidentified	Leatherjacket
Monodactylidae	Unidentified	Unidentified	Silverbatfish
Pleuronectidae	Unidentified	Unidentified	Right-handed flounder
Scorpaenidae	Unidentified	Unidentified	Scorpionfish
Syngnathidae	Unidentified	Unidentified	Pipefish, seahorse
Tetraodontidae	Unidentified	Unidentified	Puffer fish
Apogonidae	*Siphamia roseigaster*	N / A	Siphon fish
Belonidae	Unidentified	N / A	Needlefish
Blenniidae	*Omobranchus* sp.	N / A	Blenny
Elopidae	Unidentified	N / A	Giant herring
Paralichthyidae	*Pseudorhombus* sp.	N / A	Flounder
Pegasidae	*Pegasus* sp.	N / A	Dragonfish
Polynemidae	*Polydactylus microstoma*	N / A	Tasselfish
Pomadasidae	Unidentified	N / A	
Scatophagidae	*Selenotoca multifasciatus*	N / A	Butterfish
Siganidae	*Siganus* sp.	N / A	Rabbitfish
Sparidae	*Acanthopagrus australis*	N / A	Black sea bream
Synodontidae	Unidentified	N / A	Lizard fish
Terapontidae	*Pelates* sp.	N / A	Trumpeter perch

Near-bottom samples = samples taken just above the substrate;
sub-surface samples = samples taken just below the water surface;
N/A = Not available.

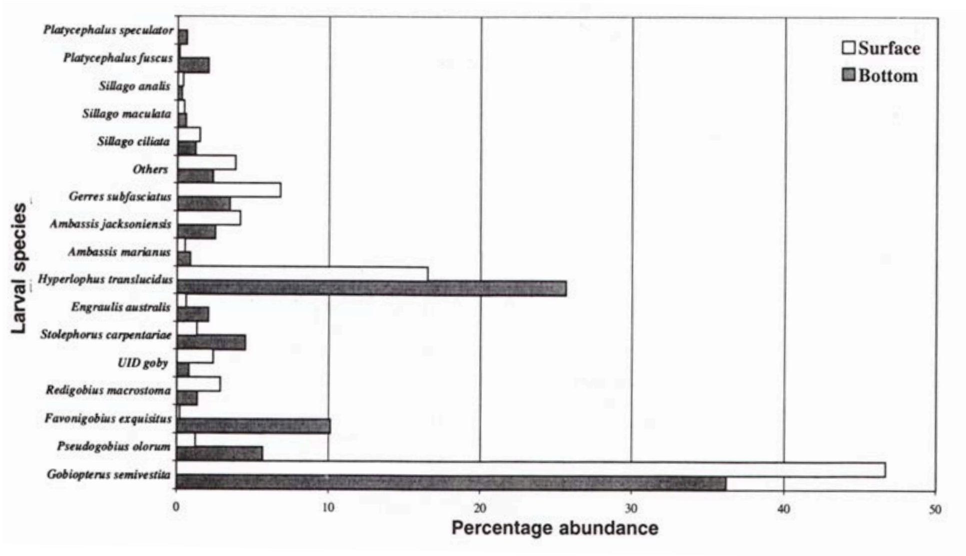

Figure 4. Faunal composition of fish larvae in sub-surface and near-bottom waters of Pumicestone Passage.

Table 2. Temporal and spatial distributions of fish larvae in Pumicestone Passage.

Species	Temporal distribution	Spatial distribution
Gobiopterus semivestita	October - January	Sites 4, 5
Pseudogobius olorum	October - March	Sites 1, 2, 3
Favonigobius exquisitus	October - March	Sites 7, 8, 9
Redigobius macrostoma	October - March	Sites 3, 4, 5
Engraulis australis	November - March	Sites 3, 4, 5, 6, 7
Stolephorus carpentariae	June - August	Sites 2, 3
Stolephorus indicus	June - August	Sites 2, 3
Stolephorus devisi	February - April	Sites 3, 4
Hyperlophus translucidus	October - March	Sites 3, 4, 5, 9, 10
Sardinops neopilchardus	December - February	Sites 7, 8
Ambassis jacksoniensis	November - March	Sites 6, 7, 8
Ambassis marianus	November - March	Sites 4, 5, 6, 7
Gerres subfasciatus	October - February	Sites 3, 4, 5, 6, 7
Sillago analis	October - December	Sites 6, 7, 8
Sillago ciliata	December - February	Sites 6, 7, 8
Sillago maculata	September - November	Sites 3, 4, 5
Platycephalus fuscus	January - April	Sites 3, 4, 5
Platycephalus speculator	January - March	Sites 1, 2, 3
Others	January - December	All sites

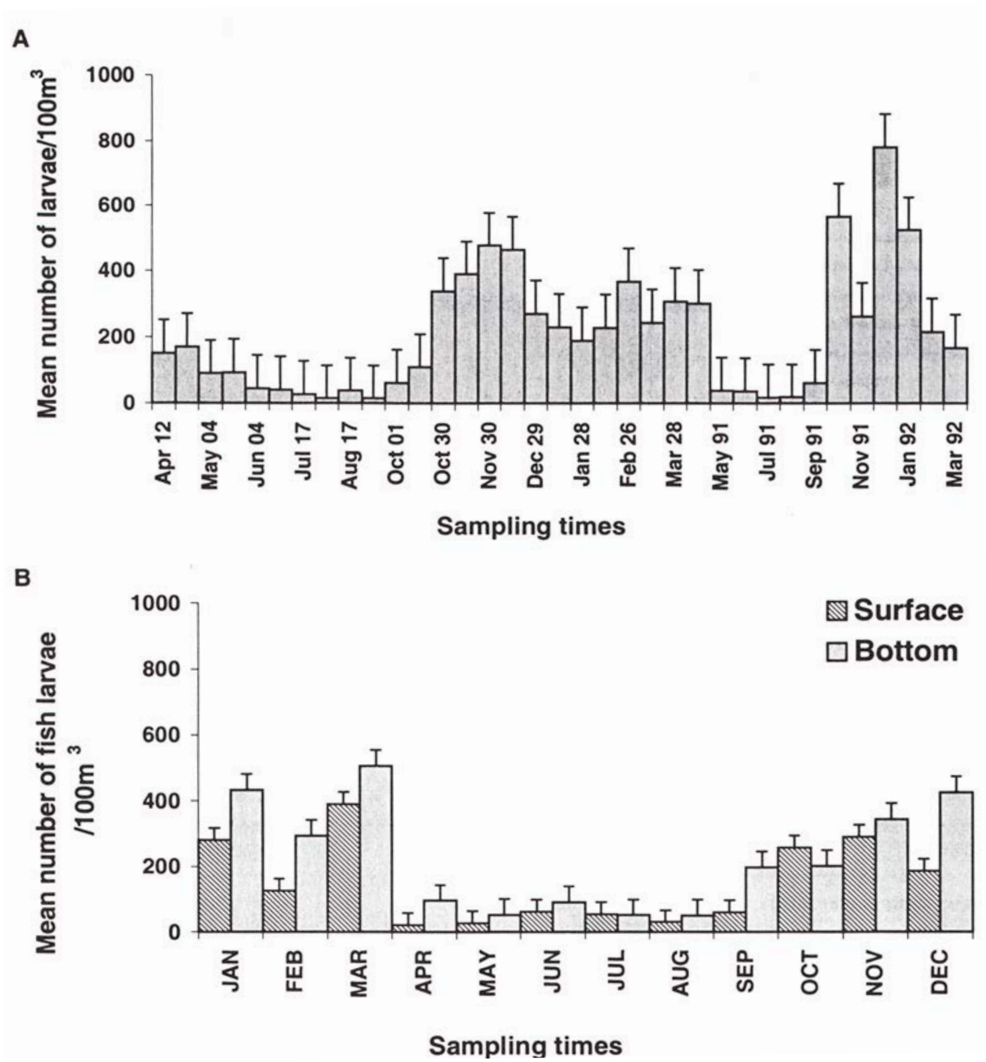

Figure 5. Mean monthly concentrations of fish larvae (number per 100 m^3) in Pumicestone Passage from: **A.** April 1990 to March 1991 (fortnightly samplings) and from April 1991 to March 1992 (monthly samplings); **B.** from January to December 1995.

Baird, 1980; Yamashita & Aoyama, 1984; Beckley, 1986; Jenkins, 1986; Roper, 1986; Steffe, 1991; Neira *et al*., 1992; Harris & Cyrus, 1995; Newton, 1996). Clupeidae, the second most abundant family in this study, have been found to be the dominant group in a number of studies (Pearcy & Myers, 1974; Houde & Lovdal, 1984; Beckley, 1984; Miskiewicz, 1987; Harrison & Whitfield, 1990; Gaughan *et al*., 1990; Tzeng & Wang, 1992).

In general, Gobiidae, Clupeidae and Engraulidae were, in decreasing order, the three most abundant families in Pumicestone Passage. Collectively, they constituted more than 87% of the total number of fish larvae caught. This finding parallels the situation found in the Swan estuary (Neira *et al*., 1992).

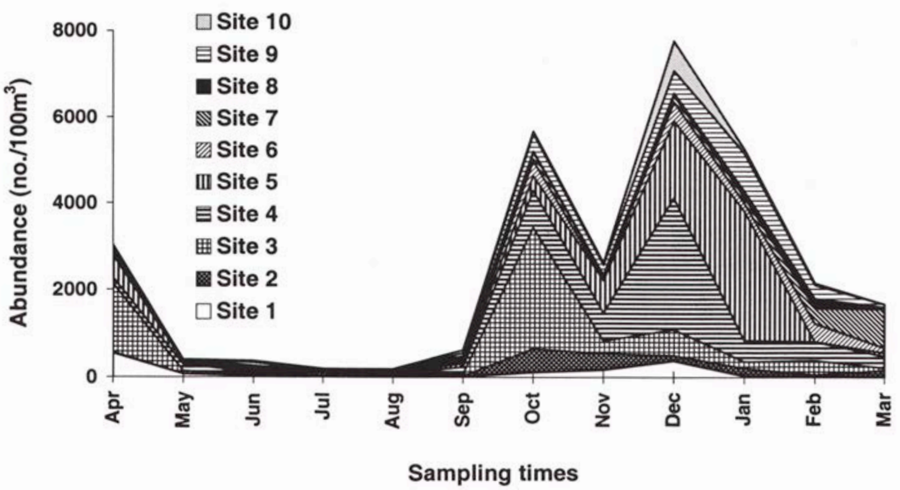

Figure 6. Temporal distribution in abundance of fish larvae in Pumicestone Passage from April 1991 to March 1992.

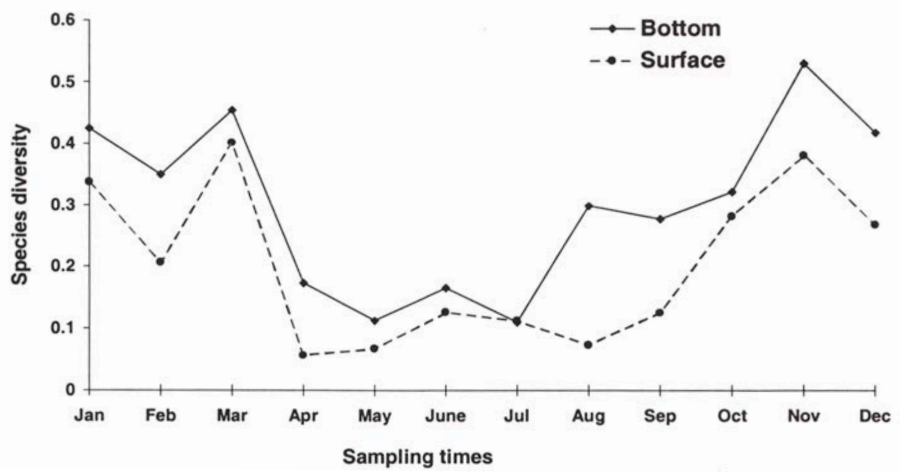

Figure 7. Temporal pattern in species diversity of fish larvae in Pumicestone Passage (Shannon-Wiener diversity (H')) from January to December 1995.

Temporal and spatial distribution patterns of fish larvae

The greater abundance of fish larvae during the warm water period from late spring to early winter is a situation similar to that found in other parts of Australia (Port Philip Bay – Jenkins, 1986; Botany Bay – Steffe & Pease, 1988; Swan estuary – Gaughan *et al*., 1990; Neira *et al*., 1992) and overseas (Swartkops estuary – Melville-Smith & Baird, 1980; Algoa Bay – Beckley, 1986; Whangateau Harbour – Roper, 1986). The greatest abundances of fish larvae overall, and of Gobiidae, Clupeidae and Engraulidae individually, were found in the mid-estuary area, indicating its probable significance as a nursery area. Species richness was highest at sampling sites close to the northern sea entrance. This is due to the presence of a number of marine species in the ichthyoplankton of this region. These presumably enter the system soon after

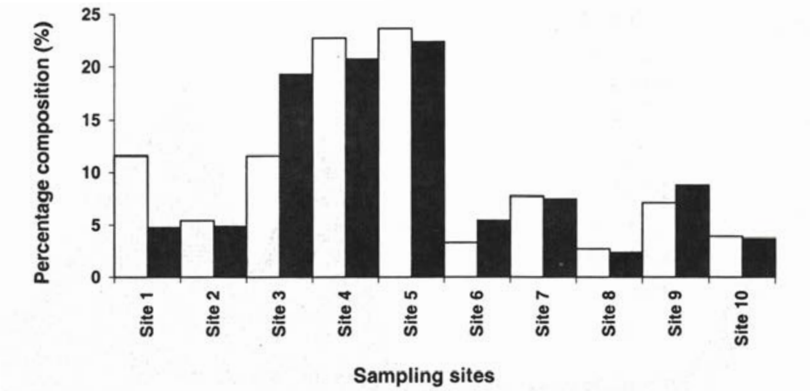

Figure 8. Proportion of total fish larvae captured at each site from April 1990 to March 1992. (Clear block = period from April 1990 to March 1991; dark block = period from April 1991 to March 1992).

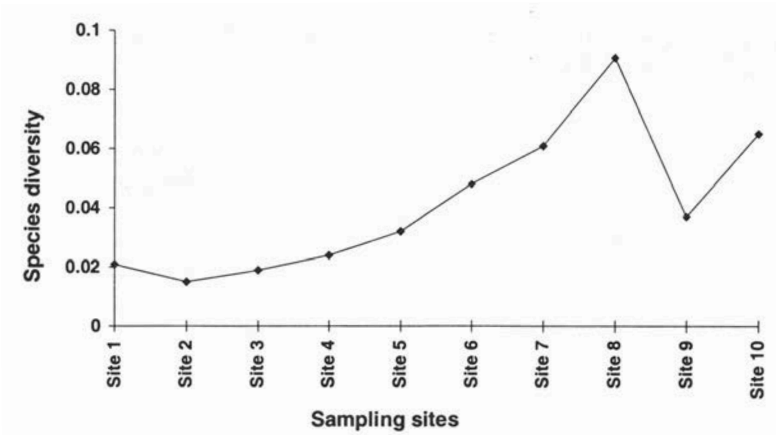

Figure 9. Spatial pattern of fish larvae diversity in Pumicestone Passage (Shannon-Wiener diversity (H')) from April 1990 to March 1992.

hatching outside the estuary, being transported by the strong tidal flow. A similar phenomenon has been reported in the Nornalup-Walpole estuary (Neira *et al*., 1992) and Swartvlei estuary (Whitfield, 1989).

The variety in structure of the substrata of the Pumicestone Passage, and variability in tidal velocities seem to play an important role in the distribution of fish larvae within the system. For instance, *Pseudogobius olorum* occurred mostly at sampling site 3 in the upper estuary region; *Gobiopterus semivestita* which spawns both inside and outside estuaries (Miskiewicz, 1987) was found more at sites 4 & 5 in the upper-mid estuary and *Favonigobius exquisitus* was recorded at sites 7, 8 & 9 close to the northern sea entrance. *Sillago ciliata* was more abundant over the sandy-mud substrata (sites 6 & 7) of the lower-mid estuary whereas *S. maculata* and *Platycephalus* spp. were found mostly over the muddy substrata (sites 3, 4 & 5) of the lower-mid estuary. The occurrence of large numbers of Clupeidae and Engraulidae in the midstream area (sites 4, 5, 6 & 7) supports the hypothesis that these larvae were transported into the system by the strong tidal flows around the sea entrance (sites 8, 9 & 10). Conditions of slow tidal flow in this mid-estuary region may prevent larvae from being advected from the system by tidal currents. In addition, at all sampling sites during the sampling of vertical distribution, fish larvae were found to be numerically more abundant in the bottom water layers than in the sub-surface waters. This pattern has been reported elsewhere (Steffe & Pease, 1988; Steffe, 1991; Harrison & Whitfield, 1990; Harris & Cyrus, 1995).

Acknowledgement

We would like to thank Drs Kevin Warburton, Ian Tibbetts and two anonymous reviewers for helpful comments on drafts of the manuscript.

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Intertidal Fish Communities of Rocky Shores in Southeast Queensland

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Abstract

An investigation of the structure and diet of fish communities occurring in rocky intertidal pools (0.1-1.0 m^3) in southeast Queensland in April and August 1995 was carried out by Rotenone sampling. Pools supported two distinct communities of fish comprising mainly blennies and gobies. Sheltered rocky shores within Moreton Bay were dominated by *Omobranchus rotundiceps rotundiceps*, whereas exposed rocky shores were dominated by *Istiblennius meleagris*. Numbers of species and individuals were significantly correlated with pool area, and biomass with pool volume. Shore level influences number of species and number of individuals in both density and concentration. Five species examined for food habits were determined to be either omnivores or carnivores. No herbivores were encountered, however, the omnivorous blennies *O. r. rotundiceps, O. punctatus* and *I. meleagris* fed mainly on plant material. *Bathygobius fuscus* and *Enneapterygius atrogulare* fed exclusively on animal prey. The five species examined fed during the day, although their nocturnal feeding habits were not determined in the present study. From patterns of gut fullness, carnivores were inferred to feed mainly while pools were emersed and omnivores while pools were immersed.

Introduction

Studies on tidepool fish communities have been undertaken throughout the world, including California (Yoshiyama, 1981), South Africa (Bennett & Griffiths, 1984; Beckley, 1985; Prochazka & Griffiths, 1992) and central Chile (Varas & Ojeda, 1990). Although Gibson (1982) stated that studies of intertidal fish communities were lacking in Australia, there is now some information on intertidal fish communities of soft sediment shores (Blaber *et al*., 1989), but little on intertidal fish communities of rocky shores.

Habitat preferences may influence the distribution of these fishes across the intertidal zone, and as a result of individual responses to topographic versus physiological limitations, resident species are more likely to show vertical rather than horizontal zonation. Gibson (1972) found vertical zonation of species on the Atlantic coast of France, and also differences in the distributions of some species between exposed and sheltered areas. Bennett & Griffiths (1984) found vertical zonation of intertidal fishes in South Africa and suggested that it was a result of factors, such as rock cover, pool size (pool area and pool volume) and the height of the pools above low water level of spring tides.

Intertidal areas can function as nursery grounds. The juveniles of species which occur subtidally and breed in the intertidal areas may find food and protection from predation by remaining in tidepools during low tides (Joseph, 1973; Beckley, 1985). The role of soft sediment intertidal pools as nurseries has received limited attention except in studies such as that by Gibson (1973) who claimed that pools on sandy shores could function as nursery grounds for juvenile plaice (*Pleuronectes platessa*). Several studies have found that rocky tidepools act as nursery grounds for subtidal fishes (Beckley, 1985; Moring, 1986; Varas & Ojeda, 1990), however, Bennett (1987) found no species that utilised rock pools as nursery grounds in rock tidepools at Koppie Alleen in South Africa.

Management and conservation of intertidal fishes of Australian rocky shores has received little

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.D. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 451-466.

attention, presumably because intertidal pools are thought to be less important as nurseries for fishes of commercial and recreational importance than other intertidal habitats. In Australia, the conservation status of endemic pool fish species, such as *Omobranchus rotundiceps rotundiceps* (Springer & Gomon, 1975) and *Istiblennius meleagris* (Springer & Williams, 1994) is yet to be assessed. The objective of this study is to describe the distribution and vertical zonation of intertidal fish communities of rocky shores in southeast Queensland.

Methods

Study area

Five rocky shore areas (Figure 1) were selected in southeast Queensland and these shores were divided into two types: open rocky shores at Caloundra and Point Lookout, exposed to moderate to strong wave action from the South Pacific Ocean; and rocky shores at Redcliffe, Manly and Dunwich sheltered by Moreton and North Stradbroke Islands.

Each rocky shore was sampled twice, in April (autumn) and August (winter) 1995. The autumn/winter sampling period was chosen to reduce the effects of the reported influx of seasonally transient species following the summer reproductive maximum (Bennett, 1987).

Three rock pools at each shore level (high, mid and low) were randomly chosen and examined on spring tides. The length, width and maximum depth of each pool was measured using a metre rule. Only pools with a volume between 0.1 and 1 m^3 were selected to allow more exact comparisons between different shores.

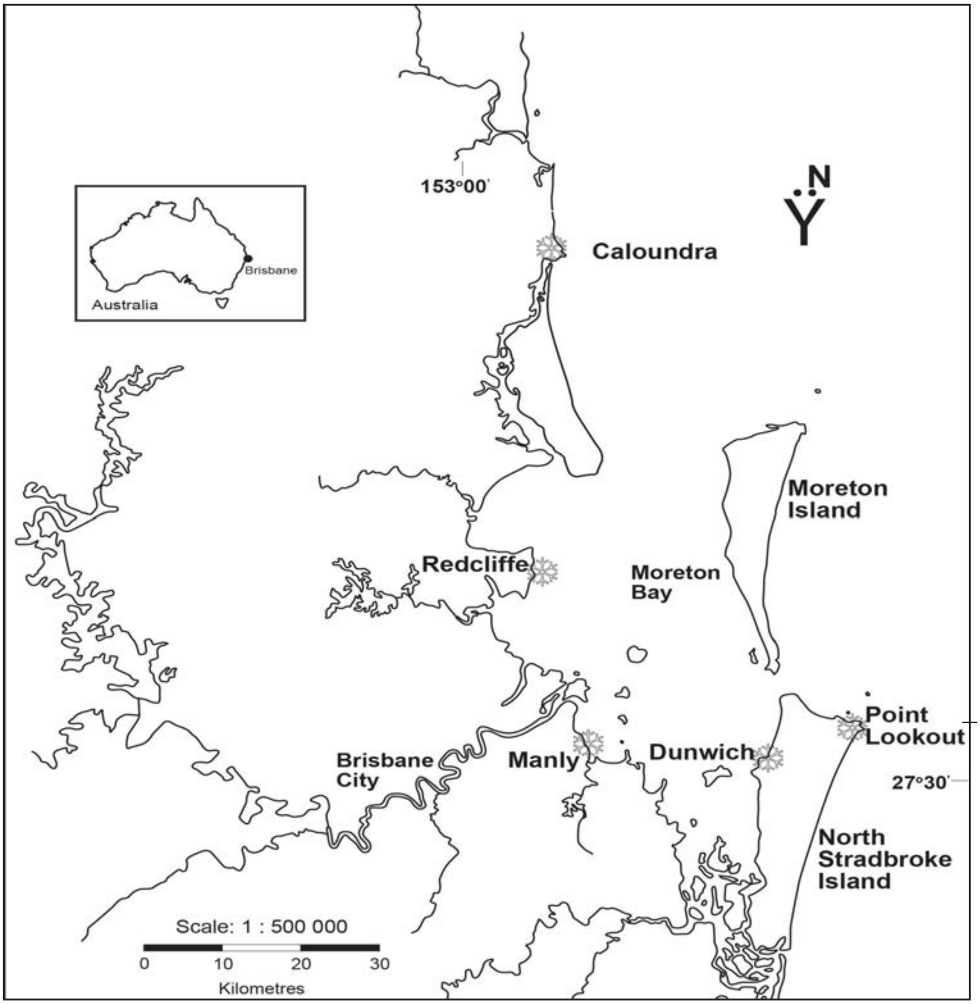
Environmental variables

The percentage cover of algae, sand and small rocks in pools were recorded to describe the micro-habitat of the pools. Tide levels of pools were determined following observations of sea levels over a tidal cycle and the physical environment of the pools was recorded at the time of sampling. Temperature was measured (± 1°C) using either a temperature meter or a standard mercury thermometer. Salinity (± 0.1 ppt) was measured using either a salinometer or an optical refractometer and pH (± 0.1) using a pH meter.

Collecting methods

Fishes were collected from pools using the ichthyocide rotenone. A suspension of rotenone was made by mixing approximately 50 mL of rotenone and 10 mL of detergent with 1 L of sea water in a two litre plastic bottle. The solution was applied on the windward side of the pool to allow advection of the poison through pool. After 15 min. fish were collected with hand nets.

Fishes collected were fixed in 30 % formalin immediately after capture and then preserved in 70 % ethyl alcohol after 24 h. Fishes were counted, their standard length (± 0.5 mm) was measured using vernier calipers and their blotted wet weight measured using an electronic balance (± 0.01 g). Fishes were identified to genus according to keys by Randall *et al*. (1990), Kuiter (1993) and Nelson (1994). The following keys were used to determine species: Springer (1972) for the tribe Omobranchini, Springer & Gomon (1975) for the genus *Omobranchus*, Springer & Spreitzer (1978) for tribe Salariini, Springer & Williams (1994) for the genus *Istiblennius*, Fricke & Randall (1992) for the family Tripterygiidae and Masuda *et al*. (1984) for the family Gobiidae.



Diet analysis

For the five numerically dominant fish species, the guts were removed and placed on a petri dish. In blennies and gobies, the first bend of the gut was used to identify the boundary between the fore and midgut (Grossman *et al*., 1980), whilst the whole gut of the triplefin fish, *Enneapterygius atrogulare,* was divided into fore- and hind-gut because of the shorter length of the gut. The fullness of foregut and midgut (hindgut in *E. atrogulare*) was visually estimated using a five point scale from empty (0) to full (4) (Grossman *et al*., 1980) to allow estimation of the feeding time of the fishes examined.

Gut contents were only examined for foreguts because the contents in the foregut were expected to be more identifiable than those of the hindgut. For identification and quantification of gut contents, food items were removed onto a Sedgewick-Rafter counting cell and examined with a binocular dissecting microscope. Percentage volume of algae versus animal material in the foregut was recorded to determine whether the fishes were carnivores (80-100% animal prey), herbivores (0-20%) or omnivores (20-80%). Animal items in the foregut were identified to the lowest taxon possible and their numbers recorded. Total volume of each prey group was estimated from coverage of grid squares (1 square ~1 mm^3) on the counting cell. The wet-weight of the total foregut contents of each fish was measured (± 0.001 g).

Statistical methods

To describe intertidal fish communities (and dietary diversity), species diversity was computed according to the following diversity indices (Odum, 1971):

(i) Margalef's species richness index: $\underline{d} = (\underline{S}-1) / \ln (\underline{N})$

(ii) Shannon-Wiener overall index: $\underline{H}¢ = - S(ni/\underline{N}) \ln (ni/\underline{N})$

(iii) Pielou's evenness index: $\underline{e} = \underline{H}¢/ \ln \underline{S}$

where N=total number of individuals, *ni*=number of individuals of each species, and S=number of species. Diversity indices are useful for comparing population structures in different locales (Moring, 1986). The Shannon-Wiener diversity index is sensitive to the distribution of individuals by species and species richness emphasises the number of species (Moring, 1986). The number of species, number of individuals and biomass were used as independent variables, and pool area and pool volume were used as dependent variables in regression analyses.

Gibson (1982) discussed the difficulty involved in expressing densities of intertidal fishes on rocky shores because densities in tidepools at low tide were likely to be higher than those at high tide. Thus, in order to allow comparison of the data of the present study with those from other studies, two methods of describing the spatial relationships of fish were chosen: density (value/m^2) and concentration (value/m^3). Although concentration (m^{-3}) of either number of species, number of individuals or biomass may be a useful measure for describing communities of fishes which remain in pools at high tide (Gibson, 1982), the estimates of density and concentration used in the present study are only valid for periods when pools are emersed.

The effects and interaction of two levels, SITE (Redcliffe, Manly, Dunwich, Point Lookout and Caloundra) and TIDE (high, mid and low), were examined by analysis of variance (ANOVA). One-way ANOVAs were used to test differences between the number of species, number of individuals and biomass within either SITE or TIDE. A crossed design two way ANOVA was applied to test the differences resulting from both SITE and TIDE. Regression analysis was used to test whether linear relationships existed between each of the dependent variables (number of species, number of individuals and biomass) and the independent variables (pool area and pool volume). Cluster analysis was also applied to estimate dissimilarities of the sampling sites as well as the trophic status of the five species examined, using Bray-Curtis dissimilarity, UPGMA (the Unweighted Pair-Groups Method using Arithmetric Averages) and the cophenetic correlation (Gauch, 1982).

Results

Twenty species in eight families and 886 individual fishes were collected from the five locations during the total sampling period (Table 1). In terms of numerical abundance and biomass, the two most dominant species were the blennies, *Omobranchus rotundiceps rotundiceps* and *Istiblennius meleagris*. The principal families of the fishes collected were Blenniidae, containing four genera, six species and contributing 60% of the numerical abundance and

82% of biomass, and Gobiidae, containing six genera, eight species and contributing 36% of the numerical abundance and 15% of biomass.

Istiblennius meleagris, Parablennius intermedius, Enneapterygius atrogulare, and one species each from the families Pomacentridae, Clinidae, Clupeiidae and Labridae occurred on exposed shores, but did not occur at either Redcliffe, Manly or Dunwich. Nine species, *Omobranchus anolius, Parenchelyurus hepburni, Pseudogobius* sp. 2*, Favonigobius exquisitus, Amoya frenatus, Amoya* sp. 4*, Mugilogobius* cf *platystoma, Pandaka lidwilli* and *Gerres subfasciatus*, that occurred on sheltered shores, did not occur at either Point Lookout or Caloundra.

In terms of numerical abundance, *O. r. rotundiceps* dominated the community at Redcliffe and Dunwich, whereas at Manly *Pseudogobius* sp. 2 dominated numerical abundance. On exposed rocky shores, *I. meleagris* dominated the community at Point Lookout and Caloundra, followed by *Bathygobius fuscus*.

In terms of total biomass, the blenny *O. r. rotundiceps* was the most abundant species at Manly and Dunwich. *O. punctatus* dominated community biomass at Redcliffe followed by *O. r. rotundiceps. I. meleagris* dominated biomass at Point Lookout and Caloundra.

In terms of pool area, there was an average of 2.8 ± 1.8 species, an average numerical density of 12.2 ± 10.7 fishes/m^2 and an average biomass of 8.8 ± 13.2 g/m^2 among all shores sampled during the study (Table 2). Caloundra had the highest numerical density of species, number of individuals and biomass. Manly had the second highest species density. The open rocky shores at Point Lookout and Caloundra had an average biomass density of 18.3 ± 28.8 g/m^2, seven times that of the highest biomass density recorded for any of the sheltered rocky shores (2.5 ± 3.0 g/m^2). This result was mainly due to the large contribution to community biomass by the blenny *I. meleagris.* In addition, the means of the dependent variables had high values of standard deviation with regard to pool area. In particular, the standard deviation of biomass at Dunwich, Point Lookout and Caloundra exceeded the mean biomass.

In terms of pool volume (Table 2), Manly had the highest mean number of species, followed by Dunwich. The highest mean number of individuals occurred at Dunwich and the smallest at Point Lookout. Caloundra had the highest mean biomass although mean number of species and mean number of individuals were relatively low. The mean number of species and mean number of individuals at Point Lookout were the lowest but the mean biomass was relatively high.

Although Caloundra had the largest mean number of species, Manly had the most diverse fish community among the five locations (Table 3), which is probably due to the large number of goby species collected there (Table 1). Dunwich had the least diverse community, with only five species, of which two accounted for 93.1% numerical abundance. The lowest value of evenness among communities at the five locations was also found at Dunwich.

Regressions of number of species, number of individuals of species and biomass versus pool volume and area showed a very significant linear relationship between pool volume, and number of species and number of individuals. Pool area influenced number of species, number of individuals and biomass.

There were no significant differences in mean biomass concentration (g/m^3) between the sampling sites and between tide levels, but there was a significant difference in mean biomass density (g/m^2) between sites (Table 4). Mean numbers of species and individuals per pool area and volume differed significantly between sampling sites and tide levels. The mean concentration of number of species and individuals was significantly different between sites and tide levels, whereas mean density of number of species and individuals was only significantly different between tide levels. Moreover, there was a significant interaction between sites and

Table 1. Composition of intertidal fish communities in five localities in southeastern Queensland. n, total number of individuals of each species; n%, percentage of each species to total numbers at each site; B, total biomass (g) of each species at each site; B%, percentage of each species to total biomass at each site. Numbers of pools sampled were 16 (Redcliffe), 18 (Manly), 18 (Dunwich), 15 (Point Lookout) and 15 (Caloundra).

Species name	Redcliffe				Manly				Dunwich				Pt. Lookout				Caloundra				Overall			
	n	n%	B	B%	n	n%	B	B%	n	n%	B	B%	n	n%	B	B%	n	n%	B	B%	n	n%	B	B%
Blenniidae																								
Omobranchus rotundiceps rotundiceps	95	0.51	14.00	0.37	9	0.16	8.05	0.29	137	0.69	27.77	0.46	-	-	-	-	6	0.3	10.81	0.04	277	0.31	60.59	0.19
O. punctatus	74	0.40	20.33	0.54	12	0.05	2.51	0.09	-	-	-	-	-	-	-	-	5	0.03	5.66	0.02	91	0.10	28.51	0.06
O. anolius	7	0.04	0.85	0.02	1	*	0.28	0.01	4	0.02	0.48	0.01	-	-	-	-	-	-	-	-	12	0.01	10.61	*
Istiblennius meleagris	-	-	-	-	-	-	-	-	-	-	-	-	37	0.66	132.27	0.95	106	0.55	192.56	0.78	143	0.16	324.83	0.63
Parablennius intermedius	-	-	-	-	-	-	-	-	-	-	-	-	7	0.13	0.88	0.01	11	0.06	8.49	0.03	18	0.02	9.37	0.018
Parenchelyurus hepburni	1	0.01	0.40	0.01	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	*	0.39	*
Tripterygiidae																								
Enneapterygius atrogulare	-	-	-	-	-	-	-	-	-	-	-	-	3	0.05	0.58	*	27	0.14	10.45	0.04	30	0.03	11.03	0.02
Gobiidae																								
Bathygobius fuscus	7	0.04	2.22	0.06	-	-	-	-	52	0.26	28.56	0.47	7	0.13	4.58	0.03	48	0.25	19.31	0.08	112	0.13	54.86	0.11
Bathygobius cocosensis	-	-	-	-	-	-	-	-	1	0.01	3.23	0.05	-	-	-	-	-	-	-	-	1	*	3.23	0.01
Pseudogobius sp. 2	1	0.01	0.15	*	21	0.48	-	-	-	-	-	-	-	-	-	-	-	-	-	-	122	0.14	6.77	0.01
Favonigobius exquisitus	-	-	-	-	48	0.19	60.62	0.23	-	-	-	-	-	-	-	-	-	-	-	-	52	0.06	5.86	0.01
Amoya frenatus	-	-	-	-	3	0.01	2.84	0.10	-	-	-	-	-	-	-	-	-	-	-	-	3	*	2.84	0.01
Amoya sp. 4	-	-	-	-	16	0.06	1.88	0.07	-	-	-	-	-	-	-	-	-	-	-	-	16	0.02	1.88	*
Mugilobius cf platystoma	1	0.01	0.10	*	3	0.01	0.24	0.01	-	-	-	-	-	-	-	-	-	-	-	-	4	0.01	0.34	*
Pandaka lidwilli	-	-	-	-	5	0.02	0.14	0.01	-	-	-	-	-	-	-	-	-	-	-	-	5	0.01	0.14	*
Gerreiidae																								
Gerres subfasciatus	-	-	-	-	4	0.02	0.14	0.01	-	-	-	-	-	-	-	-	-	-	-	-	4	0.01	0.14	*
Pomacentridae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	0.01	0.46	*	1	*	0.46	*
Clinidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	0.01	0.23	*	1	*	0.23	*
Clupeiidae	-	-	-	-	-	-	-	-	-	-	-	-	1	0.02	0.67	0.01	-	-	-	-	1	*	0.67	*
Labridae	-	-	-	-	-	-	-	-	-	-	-	-	1	0.02	0.61	*	-	-	-	-	1	*	0.61	*
Total	**186**				**252**				**198**				**56**				**194**				**886**			
Total number of species	**7**				**10**				**4**				**6**				**8**				**20**			

Table 2. Mean number of species, individuals and biomass per m^2 (A) and per m^3 (V) in the five locations. m, mean and S.D., standard deviation.

	Mean No. of Species				Mean No. of Individuals				Mean Biomass			
	A		V		A		V		A		V	
	m	S.D.	m	S.D.	m	S.D.	m	S.D.	m	S.D.	m	S.D.
Redcliffe	2.4	1.7	20.0	19.3	12.5	10.4	96.7	76.6	2.6	2.7	18.9	16.6
Manly	3.5	1.5	56.2	24.7	12.3	9.0	192.8	106.3	1.4	1.2	22.2	15.5
Dunwich	1.9	1.2	29.8	29.2	10.8	9.5	237.3	410.6	3.6	4.1	55.0	62.6
Pt Lookout	2.4	2.2	10.5	10.1	6.2	5.7	26.2	25.9	14.3	25.9	50.1	85.2
Caloundra	3.7	2.3	16.1	10.0	19.2	19.1	71.6	57.5	22.3	31.9	82.6	93.6
Overall	**2.8**	**1.8**	**26.5**	**18.7**	**12.2**	**10.7**	**124.9**	**135.4**	**8.8**	**13.2**	**45.85**	**54.7**

Table 3. Diversity indices of intertidal pool fish communities at the five locations in southeast Queensland during April-August 1995. Species richness is represented by Margalef's species richness, diversity by the Shannon-Wiener overall index, and evenness by Pielou's evenness index.

Diversity indices	Redcliffe	Manly	Dunwich	Pt. Lookout	Caloundra	Overall
Species richness	1.15	1.63	0.57	1.24	1.33	2.8
Diversity index	1.04	1.55	0.71	1.10	1.37	2.06
Evenness	0.53	0.67	0.51	0.61	0.66	0.69

tide levels in mean concentration of individuals.

Cluster analysis showed that intertidal fish communities of exposed rocky shores (Point Lookout and Caloundra) differed from those of sheltered rocky shores (Redcliffe, Manly and Dunwich) in both species composition and biomass (Figures 2 and 3). Manly clustered separately from Redcliffe and Dunwich, probably because its pools contained a higher number of species and individuals of gobies. Communities on the upper shore at Point Lookout were different from other pools on the open rocky shores, exhibiting more than 80% dissimilarity.

The mean temperatures of pools on sheltered shores (Redcliffe, Manly and Dunwich) were generally higher than those on exposed rocky shore (Point Lookout and Caloundra) (Table 5). The mean values of salinity and pH on sheltered shores were generally lower than those on exposed shores.

Rock pools on the sheltered shores (Redcliffe, Manly and Dunwich) were mainly covered by sand and mud, whereas those on exposed shores (Point Lookout and Caloundra) contained more algae (Table 6). Rock pools were chiefly covered by mud (66.9%) at Manly and sand at Redcliffe (29.7%) and Dunwich (37.2%). Algal cover in pools reached 33.7% at Point Lookout and 21.7% at Caloundra.

Table 4. The F values of two way ANOVA of sites and tide levels for species, individuals and biomass. Asterisks represent: *, 0.01<p<0.05; **, 0.001<p<0.01; ***, p<0.001 where A area (m^2), V volume (m^3).

	Sites		Tide level		Interactions	
	A	V	A	V	A	V
Number of species	2.685*	17.88***	6.01**	4.84**	0.576	1.38
Number of Individuals	2.321	5.38***	8.614***	6.33**	0.579	3.15**
Biomass	3.273*	2.20	1.035	1.42	0.970	0.65

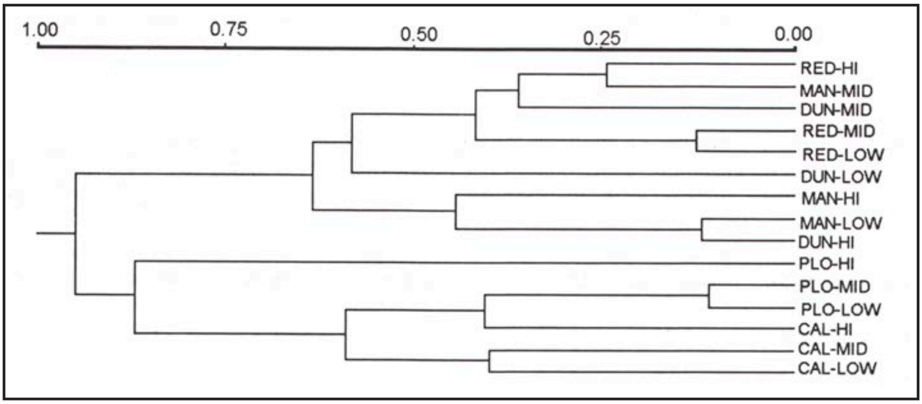


Figure 2. Dendrogram of Bray-Curtis dissimilarity of the species composition of number of individuals per species of intertidal fish communities in southeast Queensland, where the first three letters of the code represent the site (RED-Redcliffe, DUN-Dunwich, PLO-Pt Lookout and CAL-Caloundra) and the remaining letters shore level (HI-upper shore, MID-mid shore and LOW-lower shore).

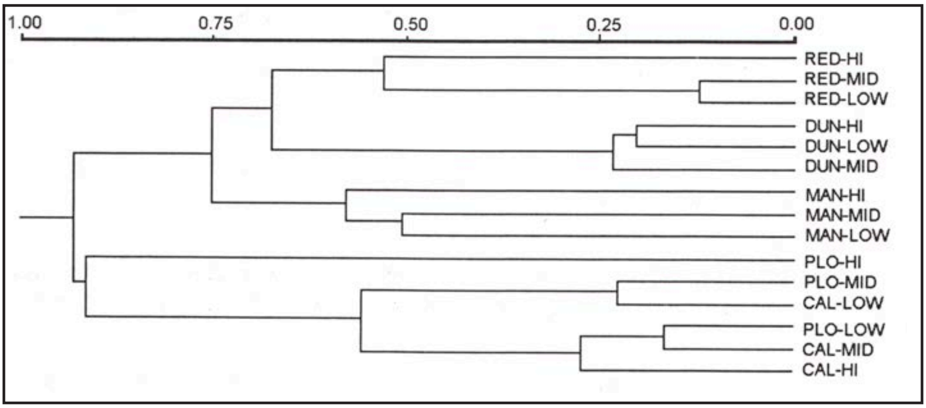


Figure 3. Dendrogram of Bray-Curtis dissimilarity of the biomass of the intertidal fish communities in southeast Queensland, where the first three letters of the code represent the site (RED-Redcliffe, DUN-Dunwich, PLO-Pt Lookout and CAL-Caloundra) and the remaining letters shore level (HI-upper shore, MID-mid shore and LOW-lower shore).

Diet

The five dominant rocky shore fish species examined for gut contents, *O. r. rotundiceps*, *O. punctatus, I. meleagris, B. fuscus* and *E. atrogulare,* can be divided into two feeding types (Table 7 and Figure 4). The blennies, *O. r. rotundiceps*, *O. punctatus* and *I. meleagris,* were omnivores with plant material being the most abundant food type (74.2%, 65.6% and 62.7% of percentage volume, respectively). *B. fuscus* and *E. atrogulare* were carnivores, each having 100% animal material in their guts. In terms of percentage volume of animal prey items in the omnivores, amphipods were the most abundant (13.9%) in *O. r. rotundiceps*, whereas foraminiferans were the most abundant animal prey of *I. meleagris* (7.4%) and *O. punctatus* (29.6%). Polychaetes were the most important prey of the two carnivores, comprising 37.5% and 46.5%, respectively.

The two carnivores had high values of trophic diversity which meant that they fed on a wide variety of taxa (Table 7). The blennies had low values of dietary diversity caused by the high proportion of plant material in their diet.

The mean foregut fullness of the blennies, *O. r. rotundiceps*, *O. punctatus* and *I. meleagris*, was less than that of the midgut (Table 7). Conversely, *B. fuscus* and *E. atrogulare* had a mean foregut fullness that exceeded those of the midgut. More than half of the total number of guts of *I. meleagris* examined were empty (Table 7).

Table 5. Temperature °C, salinity ppt and pH in pools sampled at each site. m, mean and S.D., standard deviation.

	Temperature			Salinity			pH		
	range	m	S.D.	range	m	S.D.	range	m	S.D.
Redcliffe	17-23	19.4	1.9	26-40	31.5	5.9	3.7-7.8	5.4	1.9
Manly	12-29	19.1	6.5	30-38	34.2	2.3	7.6-9.1	8.4	0.5
Dunwich	15-24	19.5	3.5	24-39	32.1	4.8	8.2-8.9	8.7	0.2
Pt. Lookout	13-18	15.9	1.4	33-42	37.8	2.9	8.7-9.8	9.2	0.3
Caloundra	16-19	17.9	0.7	34-44	38.6	3.6	8.9-9.6	9.2	0.2

Table 6. Description of percentage covers by pool microhabitats of sand, mud, rocks algae or bare rock. Where %A represents average percentage cover.

	Temperature			Salinity			pH		
	range	m	S.D.	range	m	S.D.	range	m	S.D.
Redcliffe	17-23	19.4	1.9	26-40	31.5	5.9	3.7-7.8	5.4	1.9
Manly	12-29	19.1	6.5	30-38	34.2	2.3	7.6-9.1	8.4	0.5
Dunwich	15-24	19.5	3.5	24-39	32.1	4.8	8.2-8.9	8.7	0.2
Pt. Lookout	13-18	15.9	1.4	33-42	37.8	2.9	8.7-9.8	9.2	0.3
Caloundra	16-19	17.9	0.7	34-44	38.6	3.6	8.9-9.6	9.2	0.2

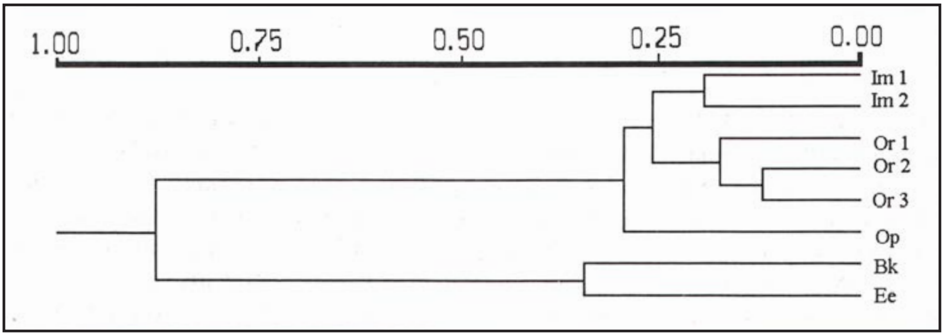


Figure 4. Dendrogram of Bray-Curtis dissimilarity of the gut contents of five species of intertidal fishes (r=0.990). Im1 and Im2, represent *I. meleagris* <40 mm and >40 mm, respectively; Or1, Or2 and Or3, *O. r. rotundiceps* <30 mm, 30 mm<L<40 mm and >40 mm, respectively; Op, *O. punctatus*; Bk, *B. kreffti*; and Ee, *E. elegans*.

Table 7. The foregut contents of the five species (*Omobranchus rotundiceps rotundiceps, Omobranchus punctatus, Istiblennius meleagris, Bathygobius fuscus* and *Enneapterygius atrogulare*) examined. %N - percentage of total number of each prey item; %V = percentage of total volume of each prey item; NI = not quantifiable in terms of numbers; = value of Shannon-Wiener diversity index for the diets; Fullness F = mean fullness of foregut; Fullness M = mean fullness of midgut on a 5 point scale of gut fullness - 0 = empty : 5 = full. * represents hind-gut of *Enneapterygius atrogulare*.

Species	*O. r. rotundiceps*		*O. punctatus*		*I. meleagris*		*B. fuscus*		*E. atrogulare*	
	%N	%V	%N	%V	%N	%V	%N	%V	%N	%V
Plant material	NI	74.2	NI	55.6	NI	62.7	-	-	-	-
Amphipods	29.6	1.0	6.9	0.2	57.1	13.7	68.4	9.7	43.0	22.3
Foraminifera	30.6	7.4	41.4	29.6	25.0	10.8	-	-	0.4	0
Gastropods	7.1	1.3	-	-	3.6	1.9	0.5	0.4	0.2	0.5
Gastropod eggs	4.1	3.2	3.4	5.0	-	-	0.5	4.7	-	-
Ostracods	16.3	4.2	27.6	0.8	1.8	0	6.8	3.7	20.1	7.4
Bivalves	1.0	0.0	3.4	0.4	1.8	0	-	-	-	-
Polychaetes	6.1	5.0	3.4	2.8	1.8	2.7	12.1	37.5	18.9	46.5
Isopods	-	-	-	-	-	-	0.5	0.8	3.5	6.5
Copepods	-	-	-	-	-	-	0.5	0.1	4.2	1.5
Decapods	2.0	0.1	-	-	-	-	3.7	10.2	8.6	10.7
Crabs	-	-	-	-	-	-	1.6	12.5	-	-
Crustacean remains		-	-	-	8.9	5.3	-	-	-	-
Cirripede larvae	2.0	1.7	13.8	4.6	-	-	-	-	0.4	2.3
Insects	0.1	0.3	-	-	-	-	4.7	6.4	-	-
Fish	-	-	-	-	-	-	0.5	5.0	-	-
Others	-	-	-	-	-	-	-	-	0.8	2.4
Empty guts	3		5		53		10		0	
Total	**72**		**25**		**90**		**30**		**20**	
H¢	1.0		1.15		1.14		1.75		1.16	
Fullness F	2.0		2.6		1.6		2.2		2.2	
Fullness M	2.4		2.9		2.5		2.1		1.8*	

Discussion

Community structure

The fish communities of intertidal rocky shores in southeast Queensland are dominated by blennies, whereas in South Africa they are dominated by clinids (Bennett & Griffiths, 1984; Bennett, 1987) and in California by cottids (Yoshiyama, 1981; Moring, 1986). There also were differences in community structure between exposed and sheltered rocky shores: the intertidal-fish communities on sheltered rocky shores were characterised by the dominance in numerical and biomass abundance of *Omobranchus rotundiceps rotundiceps*. This species is reported to inhabit shallow marine and brackish waters (Springer & Gomon, 1975) and its dominance in the sites within Moreton Bay may be a result of decreased salinity caused by outflow from the Brisbane River, although six specimens were taken from Caloundra.

The intertidal fish communities of exposed rocky shores were dominated by *Istiblennius meleagris* in terms of both total number of individuals and total biomass. This species, known only from Australia (Springer & Williams, 1994), inhabits exposed rocky shores, often tidepools (Allen & Swainston, 1988), and its range extends from tropical marine waters to Moreton Bay (Grant, 1993). During this study, some *I. meleagris* were observed outside of pools during low tides. Future studies of intertidal fish communities should also attempt to include estimates of individuals which may hide outside pools in crevices or under boulders, either to shelter during low tide or to avoid hypoxic condition in pools (Gibson, 1993).

Pseudogobius sp. 2 was the third most dominant species in terms of the total number of individuals; 99 % of the individuals captured during the study were taken at Manly. Manly is in the least saline region in the present study and lies to the west of the axis which Young (1978) used to divide Moreton Bay into less saline and more saline areas. *Pseudogobius sp.* 2 may therefore favour areas of lower salinity.

Bathygobius fuscus was the fourth and the third most abundant species in terms of the total number of individuals and total biomass, respectively. This species is known to inhabit shallow beach, rock and reef tide pools (Allen & Swainston, 1988). Eighty nine percent of *B. fuscus* captured during this study occurred at Dunwich and Caloundra.

O. punctatus may inhabit either fully marine waters or estuaries (Springer & Gomon, 1975; Allen & Swainston, 1988). This species was not collected either at Dunwich or at Point Lookout, but did occur at all other sites. Further sampling is required to confirm the full range of habitats occupied by *O. punctatus*, which is known to have a world-wide distribution in tropical waters (Springer & Gomon, 1975).

Only 12 *Omobranchus anolius* were caught. Previous studies have suggested that this species inhabits estuaries and/or tidal flats, especially where shellfish beds occur (Thomson & Bennett, 1953; Springer & Gomon, 1975; Randall *et al*., 1990; Kuiter, 1993; Grant, 1993).

The tripterygiid, *Enneapterygius atrogulare,* prefers exposed rather than sheltered rocky intertidal areas. While little is known of this species' ecology, other *Enneapterygius* species studied in Japan (Ohta & Nakazono, 1988) and in Indo-West Pacific regions (Fricke & Randall, 1992) show a habitat preference for rocky shores.

Juveniles of *Gerres subfasciatus* inhabit shallow waters in Moreton Bay (Weng, 1990). *G. subfasciatus* is an especially 'estuarine dependent' species (Blaber *et al*., 1989), and adults occur in the deeper waters of Moreton Bay (Blaber & Blaber, 1980) as planktivores or microcarnivores. The occurrence of this species in pools investigated in our study is probably due to a few individuals which recruited within tidal pools as opposed to their usual estuarine habitat. A similar occurrence was found in South Africa for the sparid, *Rhabdosargus holubi,* which recruits predominantly into estuaries but is found occasionally in tidepools (Beckley, 1985).

The overall mean density of individuals (12.2/m^2) in the present study is greater than those recorded in South Africa at Groenriver (5.49/m^2), Cape St Martin (8.11/m^2) (Prochazka & Griffiths, 1992), Cape Peninsula (7.42/m^2) (Bennett & Griffiths, 1984), and in studies cited by Gibson (1982). The intertidal rockpool fishes in the present study were highly aggregated at low tide. For example, *I. meleagris* has been reported to occur in small groups of up to twenty individuals in close proximity, presumably to reduce water loss (Grant, 1993). As a consequence of this grouping behaviour, the mean number of individuals (m^2 and m^3) at Point Lookout and Caloundra, communities dominated by *I. meleagris*, were higher than that at other places in this and previous studies.

The overall mean biomass (8.8 g/m^2) and the largest mean biomass density (22.3 g/m^2 at Caloundra) in this study are lower than those in South Africa at Groenriver (35.16 g/m^2), Cape St Martin (30.45 g/m^2) (Prochazka & Griffiths, 1992) and Cape Peninsula (49.60 g/m^2) (Bennett & Griffiths, 1984). This lower biomass density probably results from the existence of large number of small fish in pools in the present study. For instance, although Manly had more than 10 individuals /m^2, it had the lowest mean biomass (1.4 g/m^2) because of the large number of small gobies that occurred there. Conversely, Point Lookout had the lowest mean number of individuals (6.2/m^2) but had the second highest mean biomass (14.3 g/m^2), caused by small numbers of large *I. meleagris* which dominated the community.

Because of the limited use of estimates of numerical concentration, it is difficult to compare data from other intertidal fish communities with the present study. In addition, measures of fish community structure for both pool area and volume had high values of standard error. While sampling a greater number of pools would likely have reduced the standard error, sampling the number of rock pools required for accurate estimation of community structure would have seriously depleted the intertidal communities of the habitats we investigated.

Manly had the highest species richness and Shannon-Wiener diversity probably resulting from the large number of goby species (6 out of total 10 species at Manly), while other sites were dominated by two or three blenny species. The overall average species diversity (2.8) of rockpool fish communities in southeast Queensland was higher than that for intertidal fish communities in South Africa (0.10-2.96; Prochazka & Griffiths, 1992) and California (1.21; Moring, 1986), however, diversity was lower than that of Port Elizabeth in South Africa (4.36; Prochazka & Griffiths, 1992). Mean species richness (2.1) and evenness (0.688) were lower than that of Port Elizabeth (4.36 and 0.75, respectively) (Beckley, 1985).

Pool area showed a significant linear relationship with number of species, number of individuals, and biomass, however, only biomass was significantly related to pool volume. This result is similar to those in other studies. Prochazka & Griffiths (1992) claimed that biomass and number of individuals are associated with pool area, and number of individuals is associated with pool volume at Groenriver, South Africa. However, only number of individuals is correlated to both pool area and pool volume at Cape St Martin (South Africa), and Bennett & Griffiths (1983) stated that pool size (both area and volume) was positively correlated with the number of species, numerical abundance, and biomass. Thus, pool area and pool volume on rocky shores appear to be factors that influence community structure in a complex manner which involves other local environmental variables, such as percentage of rock and algal cover (Bennett & Griffiths, 1984).

The number of species and number of individuals were influenced more by shore level than they were by site, however, biomass was influenced more by site than by shore level. This result probably arises either from species selecting pools with prevailing physical conditions (Yoshiyama, 1981; Gibson, 1993) that they are able to tolerate, or from the differential survival of species due to variations in abiotic and biotic pressures among pools within a habitat.

Bennett & Griffiths (1984) stated that maximum densities of fish (and species) occurred higher on exposed shores than on sheltered shores in South Africa. This contradicts the findings of the present study in which maximum densities occur on the lower shore, in agreement with Gibson (1982). From the cluster analysis, high shore pools at the most exposed shore, Point Lookout, were very different from pools at other shore levels and sites in terms of both species composition and biomass. This was caused by three out of five pools lacking fish and the low number (2) of individuals in pools with fish. The community characteristics determined in the preent tudy are true only for the autumn and winter months surveyed and might change with the summer influx of transient species. The number of species in tidepools on South African shores decreases through the winter months but increases during the summer months, due to the presence of rare species (Beckley, 1985) or subtidal species (Bennett, 1987). Moring (1986) reported the occurrence of seasonal changes in the contribution made by the juveniles of subtidal species to the intertidal community, with an increase in late spring and summer.

The lower pool water temperature at Point Lookout and Caloundra probably resulted from evaporative cooling caused by wind associated with low air temperatures and humidities in the late winter. The microhabitats of the pools during this study seems to show clear differences in abiotic and biotic conditions. The bottoms of pools on sheltered shores are covered by sand, mud and rocks. In particular, the pools at Manly consisted of a high percentage of mud which was similar in composition to nearby sediment. Conversely, the pools on exposed shores tended to be covered by algae which provide food for some intertidal fishes. More cover, especially rock cover, seems to confer an increase in habitat diversity and thus species diversity (Bennett & Griffiths, 1984), however, the relationship between the various microhabitats and the dependent variables was not analysed because only percentage estimates of cover were collected. In future studies, a scale for estimating microhabitat cover could be valuable.

Diet

The blennies *O. r. rotundiceps, O. punctatus* and *I. meleagris* are omnivores, and *B. fuscus* (Gobiidae) and *E. atrogulare* (Tripterygiidae) are carnivores. In terms of the percentage volumetric contribution of animal prey to its diet, *O.r. rotundiceps* seemed to feed mainly on foraminiferans, but gastropods, polychaetes and ostracods were also found to be frequent prey items. The animal prey of *O. punctatus* mainly comprised foraminiferans. Seasonal changes of diet composition were not examined in this study, but Ismail & Clayton (1990) working on the rocky intertidal zone in Kuwait claimed that *O. punctatus* is an opportunistic feeder whose prey composition and percentage of animal prey in the diet are a function of prey density. While we did not examine trophic shifts in the prey composition of *I. meleagris*, Milton (1983) found that the importance of algae (Rhodophyceae, *Ulva* and *Enteromorpha*) increased with maturity in this species.

The three blennies, *O. r. rotundiceps, O. punctatus* and *I. meleagris*, have lower dietary diversities than either *B. fuscus* or *E. atrogulare*, and appear to be generalist grazers that feed mainly on plant materials, rather than specialists that feed on discrete prey items (Kortrschal & Thomson, 1986). Nursall (1981) suggested that, in order to reduce exposure to predators, feeding time should be minimised. The limited food selection, apparent in blennies, may minimise foraging time, and thus reduce their exposure to predators.

All the blennies examined had lower foregut fullness than midgut fullness. Conversely, the goby and tripterygiid species had higher foregut fullness values than midgut. Because the mean foregut fullness of *I. meleagris* and *O. r. rotundiceps* was lower than that of the midgut, the two species may be feeding at a lower rate when confined to pools. Thus, these species might feed continuously during day. *B. fuscus* and *E. atrogulare* appear to feed more at low tide, however, this observation is possibly a consequence of the rapidity and degree to which animal prey tend to be digested.

The five species examined are likely to feed during the day. Gibson (1982) stated that intertidal fishes are typically visual feeders, so they may be limited to feeding during the day. As further evidence, blennioids collected in the afternoon in the Gulf of California always had fuller guts than those collected in the morning (Kortrschal & Thomson, 1986). In coral reef systems the rise in afternoon foraging has been related to the increase in organic matter and decrease in crude-fibre levels in the turf algae that makes up the bulk of the diet of grazers (Polunin & Klumpp, 1989). This may also be true of blennies grazing algae on Australian rocky shores. Since the intertidal zone is exposed periodically and can be used by the public both for recreational and commercial purposes, and contains species of uncertain conservation status, the management of this habitat should be addressed.

Acknowledgements

The present study was funded by a Postgraduate Research Grant from the Department of Zoology, The University of Queensland, for which we are grateful. We thank Dr Helen Larson (Museum and Art Gallery of Northern Territory) and Brendan Crowley for identification of the gobies and tripterygiids sampled. We also thank Terry Dyer for technical support and Joanne Ford for discussions on the ecology of intertidal fishes.

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The Hardyhead *Atherinomorus ogilbyi* Whitley 1930 in Moreton Bay

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Abstract

Coastal planktivorous fish play important roles in processing pelagic productivity and mediating the survival of the planktonic dispersal phases of benthic organisms. In a seagrass fish assemblage in eastern Moreton Bay, the hardyhead *Atherinomorus ogilbyi* dominated both the biomass and numerical abundance of planktivorous fishes (Tibbetts, 1991). Hardyheads ranked third in both numerical and biomass abundance in samples made with a large seine net (40 m ∗ 2 m ∗ 12.5 mm stretched mesh). Dietary analysis conducted on hardyheads obtained from Moreton Bay indicates that contrary to previous published data on atherinid diets, *A. ogilbyi* feeds over the entire diel cycle. Nocturnal diet is dominated by crab zoea and the diurnal diet by crustacean remains and copepods. Polychaetes, tanaids and amphipods were found in animals captured diurnally, indicating that *A. ogilbyi* may be able to feed demersally, although this capacity has not previously been demonstrated for atherinids. This work is currently being extended to determine a predictive model for the impact of hardyheads on plankton communities in the Bay. Further investigation of this commercially and ecologically important planktivore is recommended.

Introduction

Planktivores, and particularly planktivorous fish, have an important influence on many marine and freshwater communities, both in their impact on their prey and their role as a food source for larger predators. Many large, economically-valuable fish depend upon smaller fish such as atherinids for nutrition (Prince *et al.,* 1982). Atherinids are preyed upon by coral trout, kingfish, tailor, tuna and trevally (Grant, 1987). They often form dense schools containing several thousand fish, which swim at or near the surface during the day and disperse at night. Even though they are sometimes used as food and bait, they are of little direct commercial value, which probably explains the lack of information on atherinid biology in Australia.

Prince *et al.* (1982) and Potter *et al.* (1986) have studied the life cycle and distribution of atherinids in southern Australia and Ivantsoff & Crowley (1991) conducted electrophoretic and osteological studies on *Atherinomorus* spp. *A. ogilbyi* is locally abundant in Moreton Bay, and in studies where appropriate gear (seine nets) has been used they dominate the planktivore biomass in nekton communities (Blaber & Blaber, 1980). Timing of feeding and strategies of prey selection remain unexplored. One of the aims of the current project is to investigate the ecological impact of *A. ogilbyi* by investigating the diet.

Materials and Methods

Sampling was conducted in two shallow areas of seagrass off Dunwich, North Stradbroke Island in Moreton Bay, on two days and two nights in May 1996. The first area consisted of a sandy beach exposed at low tide and sloping to broad shallow mixed beds of *Zostera capricorni* and *Halophila ovalis*. The second area was north of the first site, and comprised extensive intertidal mudflats adjacent to the One Mile jetty. Sparse beds of *Z. capricorni* were present, with some individual *H. ovalis* scattered throughout.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 467-470.

Fish were caught on the ebbing tide using a 50 m beach seine with 12 mm stretched mesh. All specimens (105 total) were placed immediately on ice, then frozen. Fish were weighed and measured (standard length ± 1 mm) with a ruler, and the alimentary tract removed. Due to the absence of a stomach (the lack of which is characteristic of the Atherinidae), the contents of the anterior intestinal swelling, or food reservoir, were examined. Prey found were identified to lowest possible taxon under dissecting microscope. The total volume contributed by each prey category was estimated using graph paper.

Results and Discussion

Atherinomorus ogilbyi is present in the Dunwich area throughout the year (Tibbetts, 1991) and dominates the biomass and numbers of planktivores. Abundance data obtained via seine netting indicate that *A. ogilbyi* is ranked third in total abundance and biomass (Tibbetts, 1991). The diets of fish captured nocturnally were dominated by crab zoae, both in terms of abundance and gut volume (Figure 1). Unidentified crustaceans were secondary in dominance, at approximately half the percentage abundance and volume of the crab zoae. Copepods only made up 5.3% of the abundance and 0.61% of the volume at night. Polychaetes, though they contributed only 5.3% to the numerical abundance, made up over 12% of the gut volume of feeding fish. Though the daytime abundance of prey items was dominated by copepods (53.9%), they only contributed to 13.3% of the gut volume (Figure 2). Unidentified crustaceans made up the majority (45.4%) of the gut volume, followed by amphipods (12.7%). Crab zoae contributed an insignificant proportion to both abundance and volume during the day (2.2% and 1.6% respectively).

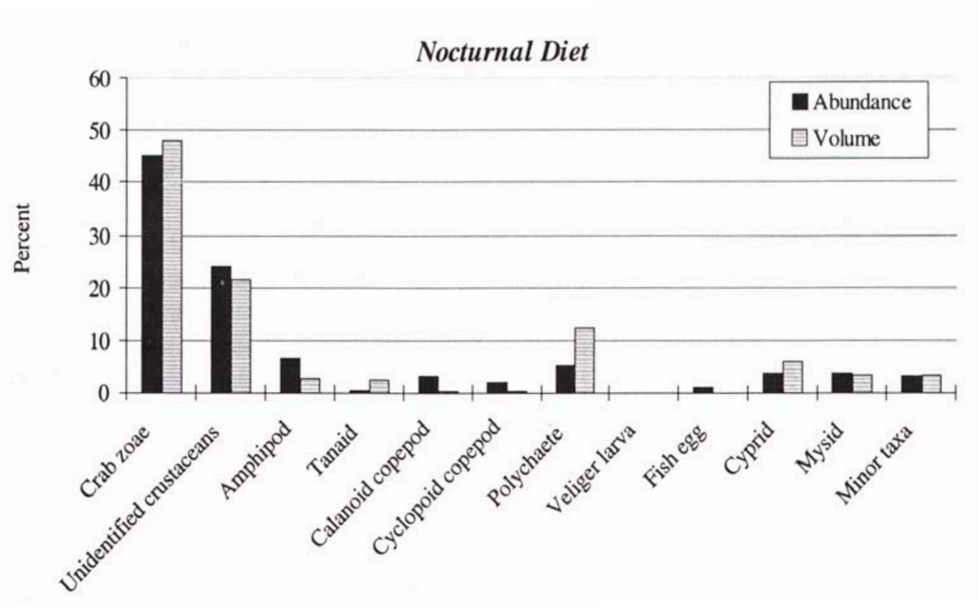

Figure 1. Percentage numerical abundance and gut volume contributed by prey items from nocturnal catches of *Atherinomorus ogilbyi* captured from Moreton Bay. Minor taxa are those prey items with only one occurrence.

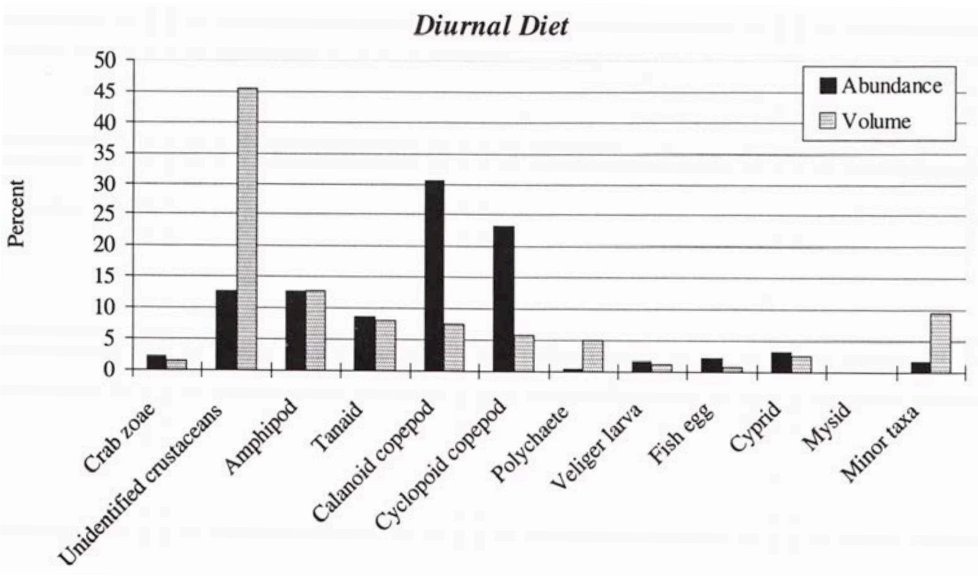

Figure 2. Percentage numerical abundance and gut volume contributed by prey items from diurnal catches of *Atherinomorus ogilbyi* captured from Moreton Bay. Minor taxa are those prey items with only one occurrence.

Analysis of the gut contents of *A. ogilbyi* revealed that while this species does feed diurnally, it is primarily a nocturnal feeder. Larval atherinids off the coast of California were found to have fuller guts during the day (Kauffman *et al.*, 1981), and *A. lacunosus (= Pranesus pinguis)* in Africa fed exclusively at night (Harman *et al.*, 1982). There is also a shift in diet between night and day. This may be due in part to differences in relative availability of various prey (although no data were obtained), as many crustaceans undergo diurnal vertical migration (e.g. mysids are more common in the surface layers at night and in the benthos during the day), but perhaps also to prey selectivity by the fish. Prey selectivity may explain the high frequency of crab zoae in fish caught at night, compared with a low frequency in fish caught during the day. Higher numbers of crab larvae are released into the plankton at night as a result of adults spawning after dark, but these organisms do not undergo vertical migration and therefore should be present in the plankton all the time (Greenwood, pers. comm.). Copepods formed the majority of the diet in diurnal samples, even though the species found (the calanoids *Eucalanus subcrassus* and *Pseudodiapthnus mertoni*, and the cyclopoids *Acrocalanus* and *Oithona*) are equally available at night (Greenwood, pers. comm.). Though far greater numbers of crab larvae were found in the guts of night-sampled fish, none of these fish fed exclusively on zoae. Unfortunately there is little information regarding diel feeding patterns in Australian atherinids, so it is not possible to compare or contrast the results of this study with those of others.

Ivantsoff and Crowley (1991) and Prince *et al.* (1982) reported a similar diet for *A. ogilbyi*, as have numerous authors for other atherinids. It has also been indicated that *A. ogilbyi* is seasonally opportunistic (Ivantsoff & Crowley, 1991), and that it may feed demersally (Prince *et al.*, 1982). These findings are reflected in the present study – while most of the prey items found are planktonic; polychaetes, amphipods (night), cyprids, mysids and the calanoid copepod *Pseudodiapthnus* are benthic (Greenwood, pers. comm.).

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Fish Use of the Inundated Waters of a Subtropical Saltmarsh – Mangrove Complex in Southeast Queensland

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Until recently saltmarshes were considered muddy wastelands fit only for filling, reclamation or for receiving dumped rubbish, and in parts of Australia as much as half of saltmarsh areas has been destroyed. Saltmarshes now are considered to be valuable coastal habitat important both in terms of their role in filtering surface overflow prior to it entering the sea and in their contribution to coastal productivity (Connolly & Bass, 1996). Part of the rationale for encouraging the conservation of saltmarshes has been their importance as fish habitat, especially for juveniles of economically important species.

There are very few studies of fish communities from saltmarshes in Australia. We cannot extrapolate directly from fish studies in northern hemisphere marshes because those marshes occur lower in the intertidal zone, in an equivalent position to mangroves in Australia (Adam, 1990). The only sampling of fish that occur naturally over the marsh flat at high tide in Australia has been in temperate waters, and those preliminary data based on sampling over three winter months in one area found low densities (0.04 fish/m^2) (Connolly *et al.,* 1997). Studies of subtropical fish communities from Australian saltmarshes have previously been conducted on Coomera Island in southern Moreton Bay (Morton *et al.,* 1987; Morton *et al.,* 1988). Fish were sampled by netting tidal creeks draining saltmarsh and semi-permanent pools on the marsh flat, and not by netting on the marshflat at inundation. The current study extends previous work showing the importance of mangroves as habitat for fish by sampling simultaneously a mosaic of saltmarsh - mangrove patches at Eden Island in southern Moreton Bay.

Despite the interest in saltmarshes and mangroves as fish habitat, no routine sampling method exists. In the present study, a 5x5 m buoyant pop net (1 mm mesh) with remote release was used to ensnare fish over a fixed area (Connolly, 1994). Fish were then retrieved from a collection pit once the tide retreated. This method was chosen because there is little in the way of above-ground structure prior to release. It can be used to sample quantitatively in habitats that include erect vegetation such as saltmarsh grasses and mangrove trees. An experiment to determine rates of recapture once fish are ensnared in the pop net showed that, on average, 93% of individuals of the most common midwater species (*Ambassis jacksoniensis*) were recaptured and that 82% of a common benthic species (*Mugilogobius stigmaticus*) were recaptured.

Fish were sampled at high tide, at night over three winter months (June - August 1996) in three habitats: upper saltmarsh, lower saltmarsh and mangroves. Mangroves typically occur lower in the intertidal zone than saltmarsh habitat, so fish were also sampled before and after high tide in mangroves when the water height was the same as that in the saltmarsh at high tide. Sampling at this lower water height in mangroves was typically done about one hour either side of slack high tide. A total of 126 net releases were made, with between 9 and 15 releases in each habitat at each sampling period.

A total of over 3 000 fish of 19 species were captured. Total fish abundance (all species combined) and species richness increased over the three months of sampling. The perchlet,

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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Ambassis jacksoniensis, was by far the most common species, constituting approximately 70% of total abundance, and occurring in 80% of samples. Fish densities (all species combined) were significantly higher in mangroves than either saltmarsh habitat at all sampling periods, and lower saltmarsh tended to have more fish than upper saltmarsh (though this trend was not always significant). Densities ranged from 0.1 to 0.8 fish/m^2 in saltmarsh and from 0.8 to 3.8 fish/m^2 in mangroves. Fish densities (all species combined, and all species excluding perchlet, tested separately) in mangroves at high tide (deep water) did not differ significantly from densities in mangroves sampled before or after high tide (shallow water). This is an important point, offering a first step in separating the importance of habitat to fish compared with the importance of water depth alone. We conclude that the difference in fish densities between saltmarsh and mangroves was not simply because mangroves have greater water depth at high tide. The differences between mangrove and saltmarsh fish assemblages are best explained by viewing the saltmarsh fish assemblage as a subset of the suite of species found in mangroves.

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Meiofaunal Selectivity in the Diet of Juvenile Whiting, (*Sillago maculata* Quoy and Gaimard), from Moreton Bay

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Meiobenthic organisms are an important component of food chains leading to higher trophic levels (Coull, 1988). Many juvenile fish pass through an 'obligatory' meiobenthic feeding stage (de Morais & Bodiou, 1984; Coull, 1990; Castel, 1992).

The present study examined the availability, diversity and abundance of meiofauna present in sediments from the foreshores of Moreton Bay (Wellington Point); the contribution of meiofauna to the diet of juvenile and post-juvenile *Sillago maculata* ('trumpeter' or the 'winter' whiting), and evidence for an ontogenetic change from a meiofaunal to a macrofaunal diet. *Sillago maculata* is both abundant and important as a recreational and commercial fish in Moreton Bay.

In March 1996, *S. maculata* juveniles were taken in seine net (10 m x 1.5 m; 2-3 mm mesh size) samples at low, mid and high tide over a 24 h period from shallow subtidal waters adjacent to Wellington Point, Moreton Bay (27º28'S, 153º15'E). Gut contents were examined to determine dietary components. Nine sediment samples were taken at low tide during the day from the exposed muddy substratum in the lower third of the intertidal strand following ebb tide, using a 2 cm diameter corer. Samples were preserved and stained with Rose Bengal formalin solution. The sediment from each core was washed successively through 250 µm, 90 µm and 63 µm sieves to remove debris and to reduce the volume of sediment for examination. The material retained on the 90 µm and 63 µm sieves was placed in Bogorov trays and examined for organisms under a dissecting microscope. Meiofaunal taxa were identified to the lowest taxonomic level practicable, and their numerical abundances recorded. Most taxa other than harpacticoid copepods could not be identified below family level.

The mean abundance of natural meiofauna was 208 animals per cm^2 of surface (S.D. ± 25.0), the dominant taxon in all samples being nematodes, followed by harpacticoids. Eighty-five percent of the *S. maculata* dissected contained food items in their guts, with harpacticoids being the most frequent and important prey item and the three species *Brianola sp. nov.*, *Coullana sp. nov.* and a member of Canuellidae (*nov. gen. et nov. sp.*) (B. Coull, pers. comm.) being most abundant. Macrofaunal crustaceans or their appendages were found in larger fish, but infrequently.

Fish of 30 mm fork length (FL) or less contained predominantly harpacticoid meiofauna, and some glaucothoe though these were small (near 1 mm). Fish of 50 mm FL or more had a predominance of macrofauna in the gut contents, including prawns and fish. Although harpacticoids were still present in fairly high numbers, they contributed little biomass to the diet. Fish between 30-50 mm FL had a mixture of meiofaunal and macrofaunal food items in the gut.

The presence of large numbers of meiobenthic fauna in the guts of juvenile and post-juvenile *S. maculata*, especially in fish of < 30 mm FL, is indicative of an 'obligatory' meiobenthic

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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feeding stage. The finding that meiobenthic crustaceans are a major (95%) component of the diet of juvenile (< 30 mm FL) *S. maculata*, is in agreement with other reports for juvenile fish (Gee, 1987).

Selective feeding was evident: harpacticoids were ingested in higher frequencies that would be expected through non-selective feeding, whereas nematodes were ingested in lower than expected frequencies. Harpacticoid intake was almost exclusively restricted to three species, especially *Brianola* sp. which is the largest of the three and has an epibenthic lifestyle. Coull *et al*. (1995) similarly found this species to be the most favoured food item of juvenile *S. maculata*, but not in fish > 43 mm FL. An ontogenetic change of diet from meiofauna to larger prey items occurs between 30-50 mm body length, and appears to be gradual in *S. maculata*.

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Dispersal of Double-ended Pipefish (*Syngnathoides biaculeatus*): Measurement Using Otolith Microchemistry

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The double-ended pipefish (*Syngnathoides biaculeatus*) is in the diverse and intriguing family Syngnathidae, which also includes seahorses and seadragons. The species is used commercially in Australia and throughout east Asia in traditional Chinese medicine. As with all species in this family the male pipefish raises the eggs, which in this case are glued onto the underside of its tail, and releases juveniles directly. Since there is no pelagic larval stage, and the adults are very weak swimmers, pipefish are thought to have limited dispersal. Ultimate threats to conservation of this species are from overharvesting and degradation of the seagrass habitat with which they are associated. The most pressing need is to know about patterns of movement, especially with a view to estimating possible rates of recolonisation after local depletion. The aim of this project is to determine how far double-ended pipefish move from the time just after hatching to adulthood, with a spatial resolution of several kilometres.

Otoliths are calcium carbonate structures in the head of fish that grow in size as the fish grows. It is well known that growth rings in the otolith can be used to age fish, but the chemistry of the otolith is also thought to reflect that of the water in which the fish lives. New technology enabling precise measurement of a wide range of elements from minute amounts of material means that otoliths can be used to give an “elemental fingerprint” reflecting where a fish has lived. We are using Laser Ablation Inductively-Coupled Plasma Mass Spectroscopy (LA-ICPMS) which permits chemical analysis of different parts of even the tiny otoliths (about 0.5 mm in adults) of pipefish. Composition of otolith nuclei (that part formed when juvenile) in adult fish can be compared with composition of otoliths from juveniles of the same cohort collected earlier. As long as it is possible to discriminate amongst juveniles from different locations (i.e. chemical composition of otoliths differs amongst locations more than within), then it is possible to determine from where adult pipefish derived.

Initially we are collecting juvenile pipefish from five locations, each separated by approximately 15 km, stretching throughout Moreton Bay. The different locations have been chosen to maximise the chance that water chemistry will differ among them. The otoliths are mounted in resin and carefully ground to expose their midplane. Initial chemical analysis indicates that the elements Li, Na, Mg, Ca, Mn, Cu, Sr, Ba, Pb, and U are likely to be most useful in distinguishing locations from which fish came. Later we will return to collect adults from the same locations to test whether they are from the same cohort. The chemical composition of otolith nuclei from adults will be compared with that of otoliths from juvenile fish. Differences in chemistry between otoliths of juveniles and otolith nuclei of adults collected in the same location will imply some degree of dispersal. Furthermore, if the chemistry of otolith nuclei of adults matches the chemistry of otoliths of juveniles from other locations, that will be taken as a strong indication of movement between those locations from the juvenile to adult stage.

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For fish, such as the double-ended pipefish, that are too small or fragile to tag, otolith elemental fingerprinting seems to offer a chance to answer carefully framed questions about movement. Rather than measuring dispersal as genetic techniques do, as average rates of dispersal over long periods and many generations, otolith microchemistry can glean information about movement over the life of an individual. This project is funded by SeaWorld Research and Rescue Foundation.

Distribution and Conservation of Delphinids in Moreton Bay

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Abstract

Moreton Bay is habitat for two species of inshore delphinid, the Indo-Pacific humpback dolphin and the bottlenose dolphin. The Bay is also used for purposes including commercial and recreational fishing and recreational boating, and fish habitat has been modified for urban and industrial developments. Boat-based searches for dolphins were conducted to estimate the relative abundance of the two species in different parts of the Bay, as a starting point in assessing the likelihood of anthropogenic impacts on population viability. The results indicate that humpback dolphins are more abundant than bottlenose dolphins in the western Bay, where human activity is concentrated, while the reverse is true in the eastern Bay. It is argued that the "rare" conservation status and particular habitat requirements of humpback dolphins warrants an increased conservation effort. A strategy is suggested which includes community education, community-based research and additional regulation, to restrain human activities around dolphins and minimise incidental capture in nets.

Introduction

Two species of inshore delphinid, the Indo-Pacific hump-backed dolphin (*Sousa chinensis*, Osbeck, 1765) or humpback dolphin (Ross *et al.,* 1994), and the inshore or *aduncus* form of the bottlenose dolphin (*Tursiops truncatus*, Gervais, 1855) are found in the Moreton Bay region. They are easily distinguished in the wild as the humpback dolphin has a shorter dorsal fin and is light grey in colour on its dorsal surface whilst the bottlenose dolphin has a taller dorsal fin and is dark grey (Figure 1). The range of both species extends from southern Africa along the continental coastlines of the Indian Ocean, through southeast Asia to the South China Sea (Ross & Cockroft, 1990; Ross *et al.,* 1994). In Australian waters inshore bottlenose dolphins are present around the entire coastline whilst regular sightings of humpback dolphins have been reported in tropical and subtropical coastal waters as far south as Coral Bay (lat. 23.2°S) on the Ningaloo coast in the west and the Tweed River (lat. 28.2°S) in the east. A third species of inshore dolphin found in Queensland, the Irrawaddy dolphin (*Orcaella brevirostris*), has been sighted in the Moreton Bay region as recently as June 1997 (Queensland Museum records), but

Figure 1. Photo of a humpback dolphin (left) and a bottlenose dolphin (right) in Moreton Bay.

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sightings are so rare that the region is not considered to be part of the species' current range, which on the east coast of Australia extends as far south as the Gladstone area (lat. 24°S).

Humpback dolphins are listed as "rare" and bottlenose dolphins as "common" in the Queensland Nature Conservation (Wildlife) Regulation, 1994. Their status is based on the relative sighting rate of the two species in aerial searches of the Queensland coastline (Queensland Department of Environment, pers. comm.). In a study conducted for the Australian Nature Conservation Agency, Bannister *et al.* (1996) classified humpback dolphins in Australian waters as "insufficiently known", with the proviso that they may be "endangered" or "vulnerable" due to their inshore habitat and the likelihood of negative anthropogenic impacts. Bottlenose dolphins were classified as "no category assigned" due to insufficient information but with the impression that their conservation status is secure.

The habitat of the two species appears to overlap considerably but differences have been detected. While the humpback dolphin is sighted in estuaries and near coastal areas (Lear & Bryden, 1980; Corkeron, 1990; Ross *et al.,* 1994; Perrin *et al.,* 1996), the bottlenose dolphin in the Moreton Bay region has been sighted in the Bay and along the ocean coastline (Lear & Bryden, 1980; Corkeron, 1990). Both species feed on pelagic shoaling fish including mullet (*Mugil* spp.) and tailor (*Pomatomus saltatrix*) (Lear & Bryden, 1980; Ross, 1984), and cephalopods (Ross, 1984; Cockroft, & Ross, 1990; Barros & Cockroft, 1991). In contrast to humpback dolphins, bottlenose dolphins feed on species associated with reefs and sandy bottoms (Barros & Cockroft, 1991).

There is some information available about the population structure of both species. Local populations of humpback dolphins have been identified in southeast Asia, as has morphological variation among disjunct populations (Perrin *et al.,* 1996). Discrete populations of inshore bottlenose dolphins have been described in the Gulf of Mexico (Shane *et al.,* 1986), the Pacific Coast of the USA (Hanson & Defran, 1993) and Shark Bay in Western Australia (Smolker *et al.,* 1992). The data suggest that populations range over distances in the order of tens rather than hundreds of kilometres along coastlines. Genetic analysis of inshore bottlenose dolphin population structure (Hale *et al.,* 1997; Gratten & Hale, 1997) supports a model of discrete local populations based on female lineages, with the conclusion that males are mobile while females exhibit natal philopatry. The range of genetically distinct populations of inshore bottlenose dolphins on the east coast of Australia is about 120 km (Gratten & Hale, 1997). Preliminary results indicate that the same model holds for humpback dolphins (Hale, unpublished).

The distribution of bottlenose and humpback dolphins in the Moreton Bay region was first studied using aerial survey by Lear & Bryden (1980) who conducted a single survey (Table 1), and then by Lanyon & Morrice (1997), who conducted six surveys in 1996 (Table 1). Seventy percent of the dolphins they sighted were in the central and eastern Bay, 25% in the western Bay and 5% in the southern Bay (Figure 2; Table 1). Aerial surveys can be used to derive estimates of dolphin numbers and distribution but suffer from problems of under-counting (Cockroft *et al.,* 1992), especially in the turbid waters of the western and southern Bay, in contrast to the clear waters of the central and eastern Bay. The concentration of suspended solids, a good indicator of turbidity, at the mouth of the Brisbane River (Cox, this volume) and nearby (Figure 2) is about 50 mg/L and drops gradually to less than 5 mg/L in the eastern Bay (O'Donohue, pers. comm.).

The primary habitat of humpback dolphins has been described as the shallow (less than 20 m) turbid waters near the mangrove and mudbank areas of estuaries, including the tidal reaches of rivers (Peddemors & Thompson, 1994; Ross *et al.,* 1994; Perrin *et al.,* 1996) rather than clear, deeper waters. This type of habitat is found primarily in the western and southern parts of the Bay, the Brisbane River and Pumicestone Passage (Figure 2), where human activity is concentrated.

Moreton Bay is utilised for purposes including commercial and recreational fishing, recreational boating, waste water discharge and shipping activities. Significant fisheries habitat has been lost due to urban and industrial development such as canal estates, tourist resorts and port facilities. Possible negative impacts on dolphin populations are from incidental capture in set gillnets and shark meshing nets, scarcity of prey items through over-fishing and loss of fish habitat, and disturbance, including strikes, by high speed boats (Hale, 1997). Shark meshing nets and drum lines are located along the nearby Sunshine and Gold Coasts and around the northeast corner of North Stradbroke Island.

The aim of the present study was to investigate the distribution and relative abundance of humpback and bottlenose dolphins in Moreton Bay. Data were obtained during a boat-based study of population size, habitat use and behavioural ecology of humpback dolphins in the region and are analysed with data from previous boat-based (Corkeron, 1990) and aerial survey (Lear & Bryden, 1980; Lanyon & Morrice, 1997) studies. The external pathology of 14 humpback and bottlenose dolphin carcases obtained from the Moreton Bay region over four years is also reported. The results are discussed with regard to likely anthropogenic impacts on dolphin populations and the roles of government regulation and community-based conservation in minimising these impacts.

Methods

Boat-based searches for dolphins on Moreton Bay were conducted on 51 days between April 1995 and December 1997, from a 5 m inflatable boat or a 6 m runabout travelling at 15-20 km/h, with three observers. Searches were carried out in calm weather, when the sea-state was no greater than 3 on the Beaufort Scale (Queensland Transport, 1997). In these conditions there is a good chance of sighting either species, as humpback dolphins are more difficult to spot than bottlenose dolphins when the water is choppy, due to their low profile on the surface (Figure 1).

The search effort was concentrated in the western and central parts of the Bay and Brisbane River. There were three days of searches in the southern Bay and six in the eastern Bay (Figure 2). Search routes were planned to minimise the chance of finding the same dolphins more than once per day.

When dolphins were sighted the search was interrupted and observations made for at least 30 minutes within which time the number of individuals in the pod (or group) was estimated. On those occasions where humpback and bottlenose dolphins were sighted together, they were recorded as separate pods. The geographic position of pods was recorded on tape from a GPS or by compass reference to land and sea marks, and later entered into the GIS package ARCVIEW, from which the map shown in Figure 2 was produced. The significance of the differences in the proportion of sightings in different parts of the Bay was tested using a two-sided t-test.

Dolphin carcases were examined for external pathology, including the presence of fresh or healed marks attributable to nets, shark attack or strikes by objects including propellers, before an internal autopsy was performed.

Table 1. Proportion of humpback dolphin (*S. chinensis*) sightings in different parts of Moreton Bay. For a description of sectors refer to Figure 2.

Sector of *S.chinensis*	Total of both	Number of species	Proportion *S.chinensis*
Boat-based surveys			
Brisbane River	95	89	0.94
Western Bay	376	278	0.74
Central Bay	316	99	0.31
Eastern Bay	23	13	0.70
Southern Bay	41	21	0.51
Western Bay (Corkeron, 1990)	30	9	0.30
Central Bay (Corkeron, 1990)	87	18	0.21
Eastern Bay (Corkeron, 1990)	248	12	0.05
Aerial surveys			
Western Bay (Lanyon & Morrice, 1997)	81	27	0.33
Central + East (Lanyon & Morrice, 1997)	228	34	0.15
Southern Bay (Lanyon & Morrice, 1997)	13	0	0.00
Western Bay (Lear & Bryden, 1980)	17	0	0.00
Central + East (Lear & Bryden, 1980)	174	8	0.05

Results and Discussion

Distribution of sightings and population size

A total of 500 humpback and 391 bottlenose dolphin sightings were made in Moreton Bay, including the lower reaches of the Brisbane River and Pumicestone Passage. As well, a few sightings were made along the ocean coast between Point Lookout (lat. 27.4^{o}S) and Hastings Point (lat. 28.3^{o}S), and in the Gold Coast Broadwater. The positions of 140 pods of humpback dolphins and 54 pods of bottlenose dolphins sighted in Moreton Bay during searches conducted over 51 days are shown in Figure 2. Sightings of dolphins in the eastern and southern Bay were considered too few for analysis and these sectors, shown in Figure 2, were not analysed separately.

The proportion of humpback dolphin sightings in the different sectors of the Bay are shown in Table 1, as are the proportional sightings calculated from previous boat-based work for the central and eastern Bay (Corkeron, 1990; Figures 2 & 3) and from aerial surveys (Lear, & Bryden, 1980; Lanyon & Morrice, 1997).

In the present study more humpback than bottlenose dolphins were sighted in the Brisbane River (0.94, n=95; Table 1) and western Bay (0.74, n=376; Table 1) than in the central Bay (0.31, n=316; Table 1), with the differences in proportions being highly significant ($p < 0.001$; 2-sided t-test). Our search effort in the eastern Bay was light, but the data of Corkeron (1990) indicate that an even smaller proportion (0.05, n=248) of humpback dolphin sightings is to be expected in this area, which is significantly less than the proportion found in the central Bay in either that or the present study ($p < 0.001$, Table 1).

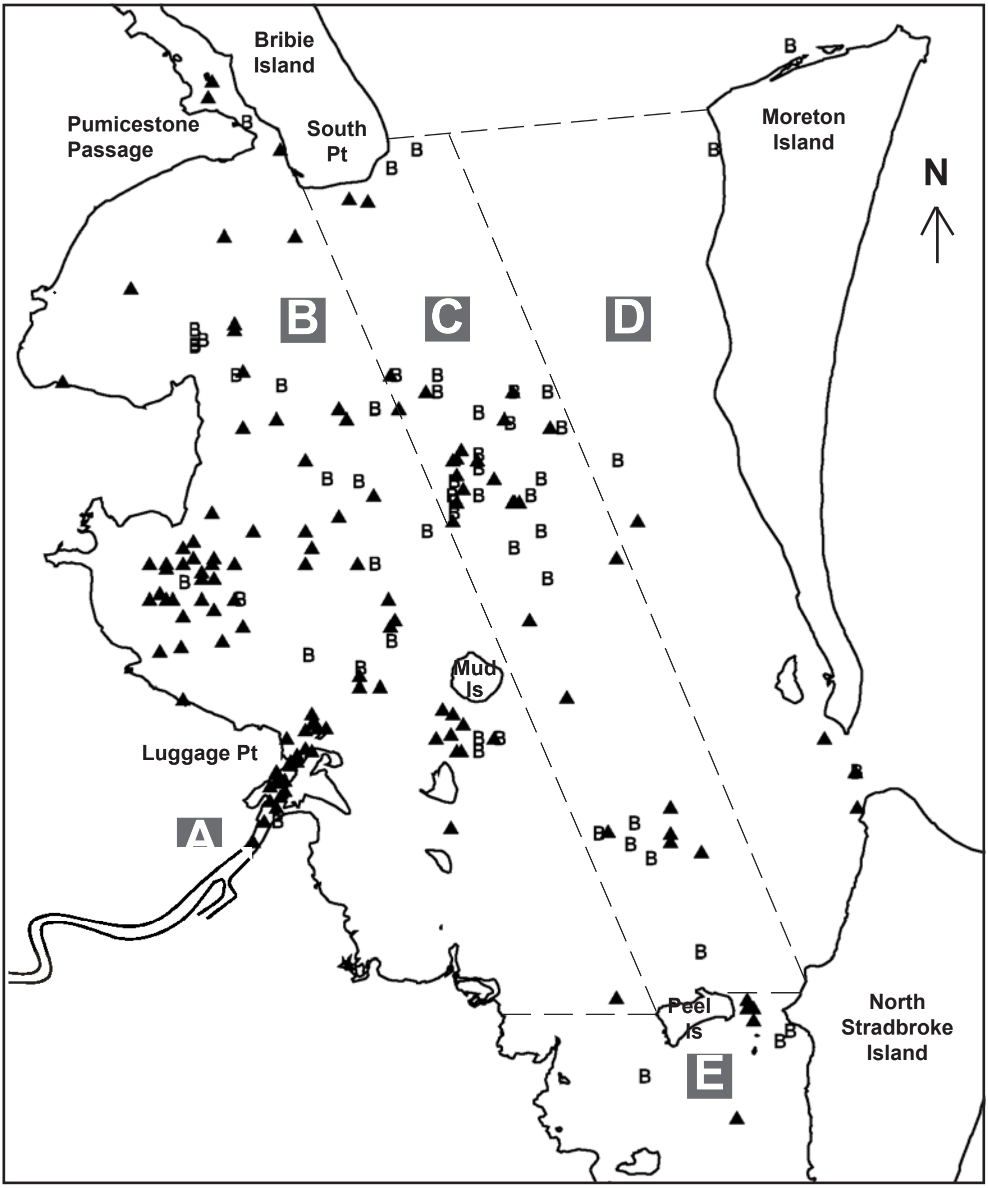

Figure 2. Map of the study area showing sightings of humpback dolphins (triangles) and bottlenose dolphins (B) in the present study. The Bay has been divided into five sectors for the analysis of sightings: **A**, Brisbane River, west of its mouth at Luggage Point; **B**, western Bay, east of the Brisbane River mouth and west of a line joining South Point on Bribie Island with the western side of Peel Island; **C**, central Bay, between the eastern and western areas; **D**, eastern Bay, east of a line running parallel to the eastern edge of 'B' and 7 km offshore from the south-east coast of Moreton Island; **E**, southern Bay, south of the northern coast of Peel Island.

The proportion of sightings made in the central Bay in the present study and in that of Corkeron (1990) (Table 1) were not significantly different (0.31, n=316 versus 0.21, n=87; $p > .05$). The studies were conducted ten years apart and the finding suggests that the distribution of the two species throughout the Bay is consistent over long periods. The proportion of humpback dolphin sightings in the central and eastern Bay from the boat-based searches was not significantly different to the proportion seen in the aerial surveys of Lanyon & Morrice (1997) (0.20, n=651 versus 0.15, n=228, $p > .05$; Table 1).

The proportion of humpback dolphin sightings in the western Bay in the present study was significantly greater than the proportion seen by Lanyon & Morrice (1997) (0.74, n=376 versus 0.33, n=81; $p<.05$, Table 1), even if all unidentified dolphin sightings in their surveys (8 of 89) are scored as humpback dolphins. The reason for this difference could be that humpback dolphins are likely to be more difficult to spot in the turbid waters of the western Bay from the air than are bottlenose dolphins, due to their light grey colour (Figure 1) and smaller pod size (3.6 versus 7.2 in this study).

The data of Lanyon & Morrice (1997) suggests that bottlenose dolphins are about four times more common (194 versus 54) in the central and eastern Bay than in the western Bay while the distribution of humpback dolphins is more even (Table 1). The total number of humpback dolphins using the Bay is likely to be only a fraction of the total number of bottlenose. The aerial surveys of Lanyon & Morrice (1997) identified 383 humpback or bottlenose dolphins, of which 16% were humpback, whilst that of Lear & Bryden (1980) identified 199 dolphins of which 4% were humpback. Corkeron (1990) identified, from photographs taken primarily in the eastern Bay, 384 individual dolphins, 13% of which were humpback. Our preliminary results (unpublished data) indicate that there are about 100 humpback dolphins using the Bay. Thus the number of humpback dolphins using Moreton Bay is likely to be 15-20% of the number of bottlenose dolphins.

The results of the present study indicate a difference in the distribution of the two dolphin species throughout the Bay, whereby humpback dolphins are more likely to be found in the shallow, turbid waters of the western Bay than are bottlenose dolphins, which are more likely to be found in the clearer waters of the eastern Bay. Data for the southern Bay (Table 1) suggest that relatively few dolphins are to be found there. However, the turbid water may have resulted in under-counting in the aerial surveys, and therefore additional boat-based searches in the southern Bay are needed.

The western Bay and lower reaches of the Brisbane River have been identified in this study as key habitat for humpback dolphins. This finding is consistent with a model of habitat use that includes the turbid water regions of estuaries. Humpback dolphins were observed in the western Bay in shallow inshore areas, along mangrove fringes, near sandbanks and mudbanks. It is here that set gillnets and highspeed boats are common and may present a problem for dolphins. Under existing regulations, gillnets may be set in both inshore (tidal) and offshore areas of Moreton Bay with the exception of Pumicestone Passage, the Brisbane River and the Bramble Bay foreshore. The western Bay is also utilised for recreational boating by highspeed vessels, including jetskis.

Conclusions from genetic analysis, supported by behavioural data, support a model of discrete, geographically localised populations of inshore bottlenose and humpback dolphins that are based on female lineages (Hale *et al*., 1997; Gratten & Hale, 1997). Moreton Bay is likely to support discrete populations of these species. Localised anthropogenic impacts, from boat traffic, gillnetting and shark meshing, are more likely to impact adversely on a localised population than on a single but extensive panmictic population distributed along the Queensland coast.

Carcass recovery

Of the 14 humpback or bottlenose dolphin carcases recovered by us from the Moreton Bay region since mid-1994, 13 were in a condition suitable for post-mortem analysis. Of these, one humpback and one bottlenose dolphin were recovered from shark-meshing nets whilst one bottlenose dolphin was removed dead from a drum line hook. Another humpback dolphin was recovered entangled in a section of gillnet. The three recovered from nets showed signs of injury from the nets which suggested that they had become entangled when alive. The other nine carcases recovered in good condition did not show any external marks from nets, evidence of strikes by objects including propellers, or shark attack. However, photographs of humpback dolphins from the region reveal that a proportion of adults carry scars from propeller strikes, shark bites and/or net entanglement (Hale, unpublished).

Incidental mortality of cetaceans in nets in Queensland is a concern. Inshore dolphins are drowned in the Queensland shark meshing nets each year (500 between 1968 and 1988, see Paterson, 1990) and in set gillnets in Queensland waters (Anderson, 1995; Qld Dept Primary Industries, pers. comm.). The problem of incidental catch in nets is also present in the Moreton Bay region, as the postmortem findings illustrate. In some areas of Queensland shark nets have been replaced with baited hooks on drum lines, but these may also result in a bycatch of cetaceans as they do for loggerhead turtles (*Caretta caretta*). The results of recent research (Kraus *et al.,* 1997; Stone *et al.,* 1997) suggest that acoustic devices (pingers) may be effective in keeping dolphins away from set nets and baited hooks.

Government regulations

The Queensland Government has attempted to establish, through legislation, a framework to promote the conservation of dolphins. The Nature Conservation (whales and dolphins) Conservation Plan (1997) for Queensland contains regulations aimed at restricting access to dolphins. Specifically, under Regulation 6.2, jetskis may not be driven closer than 300 m to a dolphin while Regs 6.3 and 6.4 prohibit a person from remaining in or entering the water within 100 m of a dolphin and Reg. 7.1 prohibits a person from intentionally feeding or touching a dolphin. While these regulations address some of the problems that may arise from human interactions with dolphins, they do not address problems from interaction with highspeed propeller-driven vessels.

We have preliminary evidence that dolphins will move away from shallow water areas when a vessel approaches at high speed (greater than 30 kph) and avoid entering areas when highspeed boat traffic is frequent. It is likely that the operation of highspeed vessels, including jetskis, has a negative impact on habitat use by dolphins. Legislation has been enacted in New Zealand that restricts the movement of vessels in the vicinity of dolphins. Thus the Marine Mammal Protection Regulations of 1992 provide guidance on the direction and speed of vessels such that they should:

- maintain a "no wake" speed within 300 m of a dolphin and if moving away from a dolphin not exceed a speed of 10 kn within 300 m;
- not undertake any sudden or repeated change in speed or direction within 300 m of a dolphin;
- not separate the members of any pod; and
- not disrupt the normal movement of dolphins.

The recent guidelines for management of the Moreton Bay Marine Park, under the Queensland Marine Parks (Moreton Bay) Zoning Plan, 1997, place restrictions on the use of vessels in Conservation Park Zones, Buffer Zones and Marine Park Zones such that they must be driven 'off the plane' in these zones except in marked navigation channels. This step has been taken to protect the dugong (*Dugon dugong*) and green turtle (*Chelonia mydas*) which feed on seagrass meadows at high tide. These measures will help protect dolphins in some parts of the Bay but do not afford any protection in the western Bay nor in other areas of Queensland. Regulations can make an important contribution to conservation by providing clear guidelines for the community as well as the power for enforcement if necessary.

Parts of the Bay are utilised for commercial gillnetting. The current regulations on the use of these nets require that they may be up to 600 m in length and can be left unattended (Queensland Fisheries Regulation, 1995). Modification to these regulations is required such that gillnets must be attended within 100 m, in line with current regulations for parts of Hervey and Shoalwater Bays (Queensland Fisheries Regulation, 1995). Recommendations aimed at protecting dugong populations in Queensland have been made recently (Anon., 1997); that bottom-set gillnets in Queensland must be attended by at least two fishers, utilise anchors to increase the tension in the mesh and be used only between sunrise and sunset. A recent move to have gillnetting listed as a key threatening process (Male, 1998) has failed. However, it has been recommended (Male, 1998) that "... effective regulations should be in place to prevent cetaceans being adversely affected". In view of this the Queensland Commercial Fishermen's Organisation has decided to introduce attendance requirements for the use of offshore gillnets throughout Queensland (QCFO, pers. comm.).

Community-based conservation

Public education, stewardship and research are important components of any conservation program. This mix of approaches is used with recreational fishers towards achieving conservation of fisheries that are heavily utilised in Australia (McPhee & Hale, 1995). The Queensland Commercial Fishermen's Organisation has implemented an awareness course (QCFO, pers. comm.) to educate fishers about the life histories of rare and endangered species and ways in which their incidental capture can be avoided. Trials aiming to minimise bycatch in nets with acoustic deterrent devices are an opportunity for commercial fishers to be involved in the monitoring of interactions between dolphins and gillnets.

The particular habitat requirements of humpback and other inshore dolphins include areas of the Bay utilised heavily by humans. Increased public awareness could have conservation benefits and be achieved at relatively little cost (Hale, 1997). It could include provisions to:

- educate the community towards compliance with conservation regulations as an alternative to enforcement, and involve the commercial and recreational fishing and recreational boating community in species monitoring;
- support community-based research underpinned by scientific methods, to achieve better understanding of habitat preferences and population dynamics as well as improved community knowledge leading to stewardship;
- regulate the speed and distance of approach of vessels in the vicinity of dolphins in line with New Zealand regulations;
- minimise incidental capture in gillnets through education, attendance rules and involvement of commercial fishers in species monitoring;
- replace shark nets with baited hooks;

- trial acoustic devices for deterring dolphins from set nets; and
- conserve fish habitat, including salt-marsh and mangrove ecosystems.

The risks to inshore dolphin populations from human activities needs to be addressed, but the detailed knowledge of population processes necessary to model their viability is not available, and populations that are now under threat may have disappeared by the time it is. Management plans provide a framework within which education, regulation, monitoring and research programs can be coordinated and given appropriate emphasis in order to achieve conservation objectives, such as minimising human-induced mortality. At present no such management plans have been enacted for dolphins in Australia.

Acknowledgements

The financial support of Environment Australia and in-kind support of the Port of Brisbane Corporation is gratefully acknowledged. Thanks to Janet Lanyon for access to, and discussion of, her aerial survey data. Many thanks also to those who provided assistance in the field, including Bronwyn Cameron, Cassandra Wittwer, Christine Fury, Dani Tikel, Deborah Jordan, Del Garbett, Eric Van der Duyes, Jacob Gratten, Jeffrey Erbacher, Justin Jordan, Leigh Slater, Michael Cunningham, Michelle Cronin, Peter Garbett, Simon Allen, Sonya Clegg, Tim Jessup, Tom Hale, Tony Turbot and Wave O'Connor. Thanks to Tony Pople and Daryl McPhee, who reviewed the manuscript.

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Wild Dolphin Provisioning at Tangalooma, Moreton Island: An Evaluation

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Abstract

Wild bottlenose dolphins (*Tursiops truncatus*) have been provisioned at Tangalooma, Moreton Island since 1992. This paper provides a brief summary of the history of human – dolphin interactions in Moreton Bay and an overview of the management regime established by the Tangalooma Resort. The outcomes of the Tangalooma provisioning program are contrasted with those reported from a similar program at Monkey Mia in Western Australia.

Introduction

Wild bottlenose dolphins (*Tursiops truncatus*) have been hand-fed at Monkey Mia, Shark Bay, Western Australia for more than 30 years (Connor & Smolker, 1985). Habituation of these dolphins has occurred progressively since the 1950s (Ross & Cockroft, 1990). More recently, in 1992 (Orams, 1995; 1996), a dolphin hand-feeding (provisioning) program commenced at Tangalooma Resort, Moreton Island, Australia (Figure 1). Opinions vary as to whether the Tangalooma program is ethical and/or should be allowed to continue. This diversity of opinion on the ethics and outcomes of the provisioning occurs within a more wide-ranging debate on the issue of human interactions with both captive and free-ranging dolphins throughout the world (Capaldo, 1989; Iannuzzi & Rowan, 1991; Frohoff & Packard, 1995).

The negative outcomes of the Monkey Mia program (e.g. relatively high calf mortality, Connor *et al.,* 1992; Richards, 1993) are often seen as evidence of the intractability of managing wild dolphin provisioning programs, resulting in criticism and adverse publicity (e.g. van Tiggelen, 1995). In this paper, the wild dolphin provisioning program at Tangalooma is described and its outcomes contrasted with those reported from Monkey Mia.

Features of the Tangalooma program which differ from Monkey Mia include dolphins spending only a limited time at the beach (< one hour daily, one feeding session only per day, with provisioning time restricted to 20-30 min), provisioning only at night, no touching and no swimming with the dolphins permitted (Anon., 1994; Hassard, pers. comm.; pers. obs.). Environmental conditions at Tangalooma (e.g. excellent circulation of oceanic waters) result in relatively low pollution risks to the dolphins compared with most mainland beaches adjacent to semi-enclosed waters.

Human – Dolphin Interactions in Moreton Bay

Interactions between humans and dolphins have a long history, both globally (Orams, 1997) and in the Moreton Bay area. Eye-witness accounts of cooperative hunting between bottlenose dolphins and Aboriginals living on offshore islands in Moreton Bay were given by several writers in the last century (e.g. Backhouse, 1843; MacGillivary, 1852; Fairholme, 1856; Campbell, 1875; Russell, 1888; Petrie, 1904, reviewed in Longman, 1926; Bryden, 1978; Hall,

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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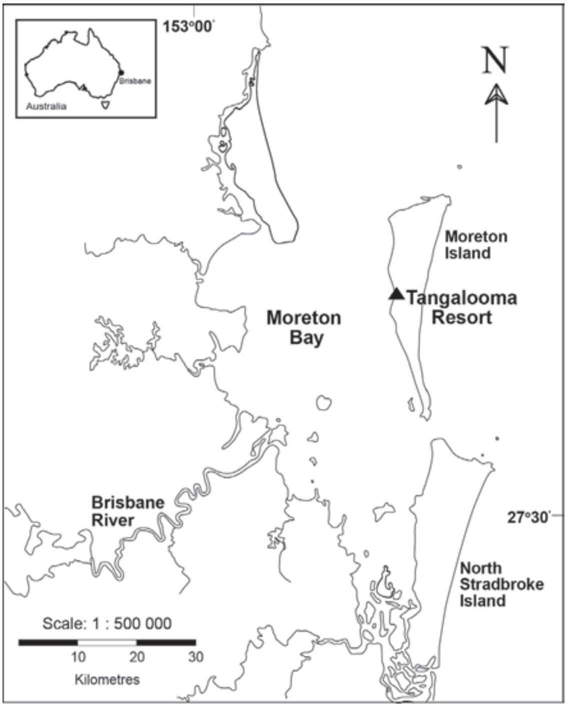

Figure 1.
Map of Moreton Bay, Queensland, Australia showing Tangalooma Resort.

1984). These accounts relate to fishing for mullet and tailor in shallow waters on the coast of Moreton Island (Petrie, 1904) and on Stradbroke Island north of Dunwich (Hall, 1984). The interaction at Amity Point is described by Fairholme (1856):

> ".. On seeing a shoal ..[of mullet].. several of the men run down, and with their spears make a peculiar splashing in the water.. they ..[the dolphins].. at once come in towards the shore, driving the mullet before them. As they near the edge, a number of the blacks with spears and hand-nets .. dash into the water. The porpoises [viz. dolphins] being outside the shoal, numbers of fish are secured before they can break away... So fearless are ..[the dolphins].. that they will take a fish from the end of a spear when held to them...".

Petrie (1904) reported that tailor fishing on Moreton Island was assisted by dolphins, apparently summoned by

> ".. jobbing with their spears into the sand under the water, making a queer noise, also beating the water with spears...", "..driving the fish towards land. When they came near, the blacks would run out into the surf, and with their spears would job down here and there at the fish, often getting two on one spear, they were so plentiful... The porpoises would actually be swimming in and out amongst all this, apparently quite unafraid of the darkies. Indeed, they seemed all to be on quite good terms, and I have more than once seen a blackfellow hold out a fish on a spear to a porpoise, and the creature take and eat it"

These historical accounts also indicate that the human - dolphin relationship was deeper than simply cooperative fishing. For example, Petrie (1904) said that on Moreton Island:

> "one old porpoise was well known and spoken of fondly. He had a .. stick of some sort stuck in his back.. and by this he was recognised...I have seen this creature take a fish from a spear, and the white men working on the island told me they often saw him

> knocking about with the blacks. At all times porpoise would be spoken of with affection by these blacks.. who said they never failed when called to drive in fish to them".

A similar relationship is reported from Amity Point, with "one old fellow, ..[identified by].. a large patch of barnacles or some fungus on his head, .. as tame – with those blacks – as a pussy cat,..[with]..a name which they believed he knew and answered to" (Russell, 1888). "The blacks will even pretend to own particular porpoises, and nothing will offend them more than to attempt to injure one of their porpoises" (Campbell, 1875). Welsby (1917) reports that "..at Amity Point porpoises were so tame as to allow themselves to be handled by the blacks in the shallower waters..".

Reports from Fraser Island and the Gold Coast area (e.g. Curtis, 1838; Gresty, 1947; Alexander, 1971) suggest that Aboriginal human - dolphin fishing cooperatives are not unique to Moreton and Stradbroke Islands and were probably quite widespread in southeast Queensland. Human - dolphin fishing cooperatives have also been reported from several other parts of the world including Africa (Busnel, 1973), South America (Lamb, 1954) and India (Lockley, 1979).

The human - dolphin interaction has been continued in Moreton Bay in more recent times with dolphins utilising trawler by-catch (Corkeron *et al.,* 1990). Again, this practice is not unique to the Bay and has been reported from North America (Leatherwood, 1975; Fertl, 1994), northern Australia (Hill & Wassenberg, 1990) and South Australia (Allen, 1996). Human - dolphin interactions at Tangalooma commenced with dolphins observed from the resort jetty and being thrown fish. Subsequently, the resort funded a research project with the objective of establishing a provisioning program. This involved attempts to attract dolphins into the resort area by feeding them by-catch from a trawler, feeding them from a small boat, and feeding them from the jetty at night (Green & Corkeron, 1991). Although these attempts were unsuccessful, it was concluded that, with more effort, dolphin feeding could be established (Green & Corkeron, 1991).

The human - dolphin interaction at Tangalooma should be considered in the context of the historical and contemporary examples of these associations, both locally and elsewhere, and of the diversity of types of human - dolphin interactions.

Regulatory context of the Tangalooma dolphin provisioning program

In Queensland the conservation and management of cetaceans and management of human - cetacean interactions are regulated under the Nature Conservation Act 1992 and the subordinate legislation of the Nature Conservation Regulation 1992, the Nature Conservation (Wildlife) Regulation 1994 and the Nature Conservation (Whales and Dolphins) Conservation Plan 1997 (Queensland Department of Environment (DoE), 1997).

Dolphin provisioning at Tangalooma is carried out under permit from the Queensland Department of Environment. The Department intends that there be no expansion of wild dolphin provisioning in Queensland and, under the proximity restrictions of the Conservation Plan, there will be no 'swim with dolphins' programs (DoE, 1997), however, human - dolphin interactions in breach of these regulations occur in several locations in southeast Queensland. Provisioning of Indo-Pacific humpback dolphins (a mother and calf) occurs at Tin Can Bay in the absence of either a permit or satisfactory management of the provisioning (Garbett & Garbett, 1995). Specific problems associated with this activity include poor control of fish quality and the condition of fish containers, large numbers of people in the water with the dolphins (as many as 43 at one time have been observed), people increasing their risk of injury by moving into deeper water to get closer to the dolphins, poor control of food quantity (feeding dolphins until satiation), feeding the calf, touching the dolphins and attempts to ride on the calf by holding onto its dorsal fin (Garbett & Garbett, 1995; 1996).

In Moreton Bay there are anecdotal reports of boat operators attracting dolphins to their boats and feeding them at locations in both the eastern and western Bay. Reports to date suggest that the dolphins provisioned at Tangalooma are not involved in these interactions. Casual boat-based provisioning may be detrimental to dolphins, particularly as it is difficult to monitor and control, and is an activity which may arise in the vicinity of provisioning programs. For example, Wilson (1994) observed that "...the frequency of dolphins begging from fishing boats and stealing baits from fishing lines and crab nets increased markedly following the introduction of regular provisioning..." at Bunbury in southwest Western Australia. Boat-based provisioning is beyond the scope of the management of provisioning at Tangalooma and beyond the scope of this discussion. Nevertheless, the existence of these casual dolphin provisioning activities in southeast Queensland highlights the need for implementation, enforcement and public education in respect of the Conservation Plan.

The dolphins of Tangalooma

A brief description of each of the dolphins provisioned at Tangalooma, largely based on Orams (1996), is provided here (summarised in Table 1):

The first dolphin to be provisioned at Tangalooma was *Beauty*, an adult female. *Beauty* took fish thrown to her from the jetty in March 1992 and was taking hand-held fish in April 1992. *Beauty* was the only dolphin accepting hand-held fish until August 1992, although several other dolphins attended the provisioning without participating. At the time hand-feeding commenced, *Beauty* was accompanied by a calf (*Tinkerbell*). *Beauty* ceased participation in the provisioning in December 1995 and is believed to have died (further discussed below).

Beauty's calf at the time provisioning commenced was *Tinkerbell*, a female now (early 1998) estimated to be about seven years of age. In early 1992, *Tinkerbell* would occasionally take fish thrown to her, but generally played with them, rather than eating them. *Tinkerbell* accepted hand-feeding in October 1992.

Shadow was born to *Beauty* in October 1994. Although still very young (about 14 months), *Shadow* was included in the provisioning program following Beauty's death.

An adult female, *Bess*, attended the provisioning with her calf *Rani* from June 1992, but did not accept hand-feeding until December 1992. In January 1997, *Bess* gave birth to another calf, *Nari*.

Bobo is a male estimated to be about ten years old. Bobo's familial relationships are uncertain, Orams (1996) suggesting that he is a son of *Beauty*. *Bobo* attended provisioning sessions from early 1992 and participated in them from October 1992.

Rani is a female calf of *Bess* with an estimated age of six. After six months attendance, *Rani* commenced hand-feeding in January 1993.

Nari was born to *Bess* in late January 1997. Although *Nari* attends the provisioning sessions with *Bess*, resort staff do not permit provisioning of *Nari*.

Fred, an adult male, has attended and participated in provisioning since February 1993. Unlike the other dolphins, he underwent no lengthy acclimatisation period, accepting hand-held fish soon after his first arrival in February 1993. It is not known whether *Fred* is related to any of the other provisioned dolphins.

Nick is estimated to be eight years old. His maternal lineage is uncertain because he was weaned at the time he commenced attendance at the provisioning (May 1994). *Nick* is an irregular and often indifferent participant.

In June, 1993, *Echo* arrived at Tangalooma and commenced hand-feeding about one month later. *Echo* arrived unaccompanied at an estimated age of one year and it is therefore assumed that *Echo* is an orphan. *Echo* is now about five years old.

Other dolphins have attended the provisioning on one or just a few occasions, but no longer do so. Over 300 individual bottlenose dolphins have been identified in Moreton Bay (Preen *et al.,* 1992) which suggests that the nine dolphins attending the provisioning at Tangalooma represent less than 3% of the population.

Table 1. Characteristics of the dolphins provisioned at Tangalooma, Moreton Island (in order of first attendance).

Dolphin	Sex	Est. age (January 1998)	Familial relationships	First attendance	First acceptance of fish
Beauty	F	Adult (Deceased)	Mother of *Bobo, Shadow* and *Tinkerbell*	early 1992	April 1992
Tinkerbell	F	7	Daughter of *Beauty*	early 1992	October 1992
Bess	F	Adult	Mother of *Rani* & *Nari*	June 1992	December 1992
Bobo	M	10	Son of *Beauty*	July 1992	October 1992
Rani	F	6	Daughter of *Bess*	July 1992	January 1993
Fred	M	Adult	Unknown	February 1993	February 1993
Echo	M	5	Unknown	June 1993	July 1993
Nick	M	8	Unknown, believed orphaned	May 1994	September 1994
Shadow	F?	3	Daughter of *Beauty*	October 1994	January 1996
Nari	M?	1	Son of *Bess*	January 1997	Not provisioned

Management of Dolphin Provisioning at Tangalooma

The Tangalooma program consists of two components.

(i) **The Dolphin Education Centre** (now known as the Marine Education and Research Centre) was established by the resort with the assistance of Mark Orams. This facility, open to the public daily, comprises a small library of publications relevant to marine mammals, displays about marine mammals in general and about the "Tangalooma Dolphins" in particular, various brochures, petitions etc. relevant to marine mammals and their conservation, a 30 seat theatrette, children's activities room, offices, and toilet facilities. People intending to feed the dolphins must book at the Dolphin Education Centre on the afternoon preceding the night on which they wish to participate in the provisioning. They are issued with one provisioning token per person, without which they will not be permitted to participate. Orams (1996) has shown that this education program significantly improved the experience of participants in the dolphin provisioning, as well as resulting in more positive environmental attitudes.

(ii) **Dolphin provisioning** (described by Orams, 1994; 1995; 1996) occurs at a dedicated area of the beach adjacent to the resort jetty. This area is marked by buoys. Landward and seaward signs state that the area is off-limits to activities such as swimming, fishing and boating at all times. Participants are required to be at the jetty 30 min. prior to the scheduled feed time, normally at sunset. Participants are given a briefing which explains:

- how to conduct themselves around the dolphins;

- what to expect of the dolphins;
- the need to disinfect their hands prior to the provisioning (disinfectant is provided for the purpose);
- a prohibition on provisioning the dolphins if the participants are suffering from colds or flu;
- prohibition on insect repellants and suntan lotions;
- prohibition on smoking in the provisioning area;
- the need to remove any sharp hand jewellery, etc. to avoid any injury to the dolphins;
- the prohibition on touching, stroking, or patting the dolphins; and
- the reasons for the short duration of their time in the water.

Following the briefing, participants are formed into several queues, the number depending on both the number of people and the number of dolphins participating on the night. From each queue, groups of participants (generally two), accompanied by a trained resort staff member, walk into the water (between knee and thigh deep) holding the fish provided. At a signal from the staff members, participants place the fish (generally two, offered one at a time) below the water surface in front of the dolphin. After a brief interval (ca. 30 sec.) all participants and staff leave the water. This procedure is repeated until all of those holding feeding tokens have fed a dolphin. The number of participants varies, generally in the range 80-100 in summer, and 20-80 in winter. Staff members are present to assist and advise the participants, and to ensure that no breaches of the provisioning guidelines occur.

Resort staff keep records of the dolphin provisioning, including the number and identity of dolphins present, the quantity of fish fed, the arrival and departure times of the dolphins as well as a video record of the provisioning sessions. Researchers from the University of Queensland also monitor specific aspects of the program.

Planning for management of the program at Tangalooma was carried out with the benefit of information from similar programs elsewhere, and was intended to minimise adverse effects on the dolphins. A full description is given in Orams (1994; 1995; 1996).

Contrasting Dolphin Provisioning at Monkey Mia with Tangalooma

Wilson (1994) identified the following areas of concern regarding the provisioning of wild dolphins at Monkey Mia:

- high infant mortality;
- low juvenile (post-weaning) survival; and
- changes in behaviour resulting from provisioning.

The main points of Wilson's (1994) report are outlined below, and are then evaluated in the context of the Tangalooma program. Whilst Wilson's (1994) findings should be treated with some caution as they were not formally published, they do provide a good summary of the relevant issues.

High infant mortality

Calf survival at Tangalooma is difficult to assess because only two have been born to provisioned dolphins since provisioning commenced. However, given the importance of calf health and survival, its discussion is warranted here. Wilson's (1994) report suggested that high mortality of calves born to provisioned dolphins at Monkey Mia may have been due to:

- prolonged exposure to polluted near-shore waters;
- exposure to human pathogens;
- provisioning distracting the mothers and offspring from attending to potential threats, especially shark attack;
- concentration of fish offal in the area which may have attracted sharks;
- provisioned dolphins accepting poor quality food items (or non-food items) from boats, causing illness; and
- nutritionally inappropriate provisions.

Prolonged exposure to polluted near-shore waters

The disappearance of seven provisioned dolphins, including three calves, at Monkey Mia during a period of 18 days in early 1989 has been attributed to pollution from sewage contamination at Monkey Mia beach (Wilson, 1994). The calves are assumed to have died because they were still nursing from their mothers at the time of their disappearance and the adults were presumed dead because they were never again seen by the researchers who were involved in ongoing studies of the Monkey Mia dolphins (Wilson, 1994). Wilson (1994) suggests that emigration is an unlikely explanation of these disappearances because it is an uncommon occurrence in dolphin communities. For example, Wells & Scott (1990) report a mean annual immigration rate of 0.021 and a maximum emigration estimate of 0.029 for a population of 156 dolphins in Sarasota Bay, Florida.

Dolphins at Tangalooma are likely to be at much lower risk from pollution than those at Monkey Mia or even those in western Moreton Bay where dolphins are commonly seen (Corkeron, 1990; Preen *et al.,* 1992). This is due to the physical conditions at Tangalooma where, by comparison with western Moreton Bay, there is: better water circulation (Milford & Church, 1977; Patterson & Witt, 1992); oceanic water (Islam *et al.* 1994; 1995); and little risk of pollution from sewage, industrial and port discharges, and urban runoff, all of which are concentrated in western Moreton Bay (Moss *et al.,* 1992). Tangalooma sewage is secondary-treated and effluent is used for irrigation on land, and not discharged to the Bay. The limited time dolphins spend in the provisioning area (generally < one hour per day) also decreases their risk of exposure to pollution. This is in marked contrast to the situation at Monkey Mia where dolphins are provisioned up to three times daily over a five hour period between 8 a.m. and 1 p.m. (Wilson, 1996).

Pathogen induced disease

Wilson (1994) also suggested that infectious disease may have caused the 1989 mortality event at Monkey Mia, and that contact with humans at the beach may have been implicated, because no comparable mortality occurred in nearby non-provisioned dolphins.

The risk to dolphins from pathogen-induced disease is likely to be far lower at Tangalooma than at Monkey Mia for several reasons:

- close proximity between any individual human and the dolphins is of very limited duration;
- physical contact between humans and dolphins, initiated by humans, is prohibited;
- provisioning by people with colds and influenza is discouraged;

- disinfection of hands is required prior to provisioning; and
- the oceanic waters at Tangalooma are not conducive to pathogen survival and transmission.

Observations by resort staff and independent observers suggest that most people readily accept the limitations imposed in the best interest of the dolphins.

High predation

Bottlenose dolphins typically increase group cohesion in the presence of predators (e.g. sharks; Shane *et al.,* 1986). Mothers and calves may maintain close association until the calf reaches three to six years of age (Navarro, 1990; Wells, 1991) and mothers have been reported to become aggressive toward calves that stray either too far or for too long (Chirighin, 1987). Consequently, dolphin calves are probably afforded some protection from predation through close association with their mother and with other adults in their group. Wilson (1994) suggests that the vulnerability of calves to shark attack is increased if they stray from their group and are not attended to by the adults, which was apparently the situation when one of the Monkey Mia calves was killed by a shark. According to witnesses, the lone calf was approximately 70 m from the beach where its mother and other adult females were interacting with people (Wilson, 1994).

The risk of predation at Tangalooma is likely to be much lower than at Monkey Mia. Reasons for this include:

- dolphins spending less than one hour daily in the provisioning area, rather than "hanging around" during the day as occurred at Monkey Mia;
- during provisioning, the dolphins spend less than half of the time actually being fed. For the rest of the time they interact or swim with each other. Observations of a calf born in October 1994 (*Shadow*) indicate that, during the time other dolphins were feeding, *Shadow* generally remained within approximately 20 m of the rest of the group. Another calf, born in January 1997 (*Nari*), exhibits similar behaviour; and
- some protection may be given to dolphins in the provisioning area by a shallow bank offshore from the provisioning area (the risk of shark attack, and the need for vigilance against attack, increases in deeper waters (Johnson & Norris, 1986)), although attacks on marine mammals, notably seals, do occur in shallow water (Corkeron, 1997; see also Wood *et al.*, 1970).

Inadequate nutrition

Wilson (1994) points out that no information is available on which fish species dolphins select under differing conditions or on the relative nutritional values of fish species eaten. He suggests it is possible that the fish given to Monkey Mia dolphins was nutritionally inadequate or inappropriate and that accepting a significant proportion of their food from hand-outs may result in malnutrition, leaving the dolphins vulnerable to disease and predation.

No data are yet available from Tangalooma regarding the nutritional status of the fish used in provisioning, although the choice of species used was based on the apparent dietary preference of the dolphins. Comparisons of the diet of the Tangalooma dolphins with that of dolphins elsewhere is made difficult by the marked spatial and temporal variability in both the quantity of prey consumed (Ross & Cockroft, 1990) and the relative importance of the various prey species in the bottlenose dolphin diet (Cockroft & Ross, 1990).

The provisioning regime adhered to at Tangalooma means that most dolphins consistently receive one-third or less of their estimated average daily food requirement (Orams, 1996) and the absence of the dolphins from the beach, except during the regulated provisioning periods, eliminates the problem of casual provisioning at the resort. Some of the potential for malnutrition at Monkey Mia is associated with *ad hoc* provisioning of dolphins from boats. There is likely to be a lower risk of this at Tangalooma because the relatively exposed waters of Moreton Bay limit access. Furthermore, this type of provisioning is prohibited under the Conservation Plan. Following the recommendations of Wilson's (1994) report, a regulation under the Wildlife Conservation Act (Western Australia) was introduced prohibiting dolphin provisioning in the Shark Bay Marine Park, with the exception of the designated area. Wilson (1996) suggests that this has virtually eliminated boat-based feeding.

Deliberate boat-based provisioning of dolphins by recreational boaters also occurs in Moreton Bay, and it is important that this practice cease as the freshness and nutritional suitability of fish supplied in this way is uncertain and would be difficult to regulate.

In Moreton Bay, dolphins are provisioned inadvertently from bycatch generated by commercial fishing operations (Corkeron *et al.,* 1990). Possibly, this source of food partially compensates for loss of food resources as a result of commercial and recreational fishing (Corkeron, 1990) which, collectively, remove between 1 300 t (data from QFMA, 1996) and 2 000 t (Quinn, 1993) of finfish from Moreton Bay annually. The use of trawler bycatch by dolphins in Moreton Bay suggests that the fishery is removing food which is attractive to the dolphins. This utilisation of trawler bycatch may also have disadvantages for the dolphins, given the attraction of sharks to the trawler bycatch and the association between the seasonal peak in trawler operations and in shark wounds on dolphins (Corkeron *et al.,* 1987; 1990; Preen *et al.,* 1992).

An indication of the probable nutritional adequacy of the fish provided at Tangalooma, in combination with other food sources, is suggested by two characteristics of the group. Firstly, calves born to and reared by provisioned mothers have survived to date. For example, *Tinkerbell* (calf of *Beauty*) is estimated to have been two years old at the time *Beauty* commenced her participation in the provisioning program. Similarly, *Shadow*, a calf of *Beauty*, was born in October 1994 while *Beauty* was a participant in the provisioning. *Nari*, born to *Bess* in January 1997, has survived with no apparent signs of ill health. More compelling evidence of nutritional adequacy is the survival of orphaned dolphins. *Echo*, now approximately five years old, arrived at Tangalooma unaccompanied, at an estimated age of one year, suggesting he was an orphan. *Shadow* was orphaned at 14 months of age by Beauty's death in December 1995, and has also survived with no apparent signs of ill health.

One of the most telling criticisms of dolphin provisioning at Monkey Mia is the observation that survivorship of calves of provisioned dolphins is significantly lower than that for calves of non-provisioned dolphins. Wilson (1994) cites data which indicate that survivorship of calves of provisioned dolphins was < 30 % in the period from 1975, and < 20 % since 1986, compared with one estimate (from another location) for natural populations of 80% survivorship. Similarly, survival of calves through the first year of life, over the 1985-1993 period at Monkey Mia was 67 % for calves of non-provisioned dolphins, and 36 % for calves of provisioned dolphins.

Only two calves have been born since the commencement of the Tangalooma program – *Shadow*, now over three years, and *Nari*, now over one year old. One calf, *Tinkerbell*, has survived to seven years, despite being involved with the provisioning program for five of those years. Another apparently orphaned calf, *Echo*, has survived to about five years with no maternal support and despite four years participation in provisioning and *Rani*, now about six years old, has been a participant in the provisioning at Tangalooma for five of those years.

Whilst a conclusive statement regarding the long term effect of provisioning at Tangalooma on calf mortality is not yet possible, all of the calves associated with the provisioning have survived to date.

Low juvenile (post-weaning) survival

Wilson (1994) states that:

> "there are few data on survival of juvenile dolphins after weaning but available information suggests that there may be a problem here also ... Four juveniles born to provisioned Monkey Mia females since research began have survived past weaning (that is about two years of age). Two of these were provisioned but died within two years of weaning. The two that still survive have never accepted handouts and, since leaving their mothers, have never been regular beach visitors."

The survival of all four juveniles, *Tinkerbell*, *Rani*, *Echo*, and *Shadow*, at Tangalooma shows that post-weaning mortality has not occurred at Tangalooma. Again, this may, at least in part, be attributed to the limited duration of provisioning times, restrictions on the amount of food given and the other management provisions applied.

Changes in behaviour resulting from provisioning

Wilson (1994) states that:

> "Researchers ... have documented significant differences in the behaviour of provisioned dolphins compared with non-provisioned dolphins at Monkey Mia. Their study included data on 32 infants, nine of which were born to provisioned females. They showed that infants of provisioned females spend less time in contact with their mothers overall. This appears to be due to the fact that provisioned infants spend less time in contact with their mothers when near the provisioning area than when they are away from the beach."

It seems that the short duration of provisioning at Tangalooma minimises the likelihood of significant alterations to behaviour. For example, a dolphin attending all feeds for the maximum duration would spend > 95 % of its time elsewhere. A dolphin attending 80% of feeds and arriving only as provisioning commenced would spend > 98 % of its time elsewhere. It follows that the provisioned dolphins spend the majority of their time away from the provisioning area, in "natural" conditions. However, insufficient observational data are yet available to determine whether, when they are away from the provisioning area, their behaviour differs significantly from that of other dolphins in Moreton Bay.

In the case of the mother-calf pair currently attending the provisioning (*Bess* and *Nari*), although they spend little time in contact during the provisioning period, these periods are of short duration. They generally maintain close association both before and after the provisioning, and in intervals during provisioning when groups of people are not in the water. At 11 months of age (observations over 14 nights during December 1997 - January 1998), *Nari* was alone (defined as > 3 m from any other dolphin) 29 % of the time, with *Bess* or in groups including *Bess* 47 % of the time, and in other groups without *Bess* for 24 % of the time. *Nari* was alone outside the illuminated provisioning area 1.2 % of the time (Takei & Neil, 1998). The net effect of this pattern is that mother - calf separation, as a consequence of provisioning, is limited in total to about 15 minutes daily, in 15-20 short increments.

It should be noted that behavioural comparisons between the Tangalooma dolphins and those from other locations are difficult as any differences which do occur may be due to geographical differences rather than the effects of provisioning.

The Illness and Death of *Beauty*

Given that *Beauty* developed an illness and died while a participant in the provisioning program, some discussion of the cause of death is warranted. The following comments on Beauty's illness are from Wendy Blanshard, the Sea World veterinarian who monitored *Beauty* during the latter half of her illness. There is no definitive diagnosis of the illness which consisted of a destructive lesion on the rostrum. The most likely cause is considered to be neoplasia (a tumour), although the other possibility, osteomyelitis (an infection of the bone), cannot be ruled out. It is likely that these illnesses occur in nature, although there are insufficient data to determine their natural rates of occurrence.

Neoplasms in marine mammals were considered a rare or uncommon event and it was speculated that marine mammals may have had some resistance to neoplasm development. More careful observation on larger numbers of animals has revealed a larger number of neoplasms than previously thought (Howard *et al.,* 1983). However, Geraci *et al.* (1987) caution that many of the reported neoplasms are not well documented. Of 41 confirmable tumours on cetaceans, Geraci *et al.* (1987) report that most were from either the gastro-intestinal tract (31%) or the skin (24%). Although there are numerous factors which may induce tumours (e.g. hormones, viruses, congenital defects, and physical and chemical agents) and cetacean tumours have been linked to environmental pollutants (e.g. Martineau *et al.,* 1985; Bossart *et al.,* 1997). "...so little is known about cetacean tumours that it hardly seems necessary to propose causes, however tempting..." (Geraci *et al*., 1987). Bossart *et al.* (1997) suggest that the presence of tumours may predispose affected cetaceans to other terminal events, such as net entanglement and shark attack.

Osteomyelitis in a dolphin maxillary bone could be caused by a puncture wound in the mouth from a fish spine. However, such injuries are likely to be a normal hazard of eating spiny fish. Blanshard (pers. comm.) suggests that it is most likely that Beauty's condition was a random accident of nature and that there is no direct evidence to link it with participation in the provisioning. Rather, provisioning allowed the observation of a condition which is likely to occur naturally. *Beauty* ceased attendance at the provisioning in December 1995, although her fourteen month old calf *Shadow* continued to attend. It is therefore assumed that *Beauty* died at this time. In the absence of a body, no necropsy was conducted.

Conclusion

In general, the Tangalooma program appears to comply with the management recommendations of Wilson (1994) and, to date, appears to have avoided many of the problems reported at Monkey Mia.

This outcome appears to be largely related to two main factors. Firstly, the physical setting assists in minimising adverse impacts on the dolphins as a result of both high water quality and limited public access. Secondly, the management regime of fixed feeding times of short duration, food rationing and no physical contact with humans limits the risks associated with dependency, malnutrition, predation and pollution. Given that provisioning at Tangalooma commenced only five years ago (cf. 30 years at Monkey Mia) chronic problems associated with provisioning may not yet be apparent, thus continued monitoring and assessment of the program is essential.

Acknowledgements

We wish to acknowledge the support of The University of Queensland and of Tangalooma Moreton Island Resort, and Dr Peter Corkeron, Dr Ian Tibbetts and an anonymous referee for their helpful and constructive comments on the draft.

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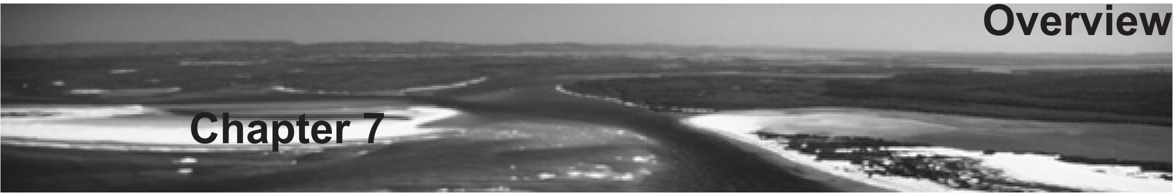

Chapter 7

Corals

This chapter deals with the corals of Moreton Bay. Corals are animals, but their requirement for light (for photosynthesis by symbiotic algae within their tissues) and reliance upon substrate and space make them behave in an ecological sense more like plants than animals. They also hold a fairly special place in our superficial estimates of health of the Bay. For example, if we know corals are present in the Bay then its condition can not be considered poor, and if corals start to die then perhaps we should become alarmed for the health of the Bay. Coral communities also tend to be particularly rich in fish and invertebrates which creates interest for visitors and thus contributes to the economy of the Bay.

The coral communities of Moreton Bay are of interest to scientists because they are quite distinct from coral assemblages that occur to the north and the south. They are definitely living with extreme conditions, including flooding, water quality gradients and water temperature, as the first paper points out, and these contribute to the distribution of particular corals within the Bay. Corals are often said to be very sensitive to lowered water quality, however, the faviids (members of the brain coral group) are able to withstand conditions that would kill most other corals. They occur at sites in the western Bay even fairly close to river mouths.

Both Peter Johnson & David Neil, and Peter Harrison and his coauthors point out the importance of short term events such as floods on the distribution of corals and also the importance of long term changes in sea level, rainfall and water temperature that have modified the structure of the Bay’s coral assemblages over millennia. Floods in particular can have major impacts on corals, with a flood in 1996 leading to quite widespread coral injury and death. The flood plume damages corals by reducing light penetration, increasing sediment deposition and lowering salinity. The communities, however, do seem to recover to their former richness, although this can take many years.

A reduction in the extent of the flood plume, created by the damping of peak flows via stream control at large dams in the catchment (e.g. Wivenhoe, Somerset and North Pine), has been a positive influence on the corals of the Bay. Negative influences, however, include an increase in sediment load, due to poor land care practices, and elevated nutrient and toxicant levels in the flood plume. At present these influences seem to be in balance with the coral communities of the Bay being fairly stable in both their distribution and diversity.

Our level of knowledge about the corals of Moreton Bay is insufficient. We do not know the source of larvae for coral recruitment. The reefs may either be self-seeding, with larvae coming from other corals within the Bay, or they may be derived from corals outside of the Bay. We lack knowledge also of ecological processes that may be important in the maintenance of coral communities within the Bay. For example, corals of the Great Barrier Reef are supported in their ascendency over benthic algae, which grow much faster than corals, by the activity of grazing fish and invertebrates which reduce the size of most plants to a fine fuzz that covers all bare hard substrate of the reef. We do not know if the same process is important in Moreton Bay coral communities. If grazers do allow corals to thrive at the expense of macroalgae then we need to know about the distribution and population characteristics of the grazers, particularly if they are vulnerable to decreased water quality or fishing pressure. Such knowledge is vital for the effective management of our coral resources and assessment of their sustainability.

Fortunately the crown-of-thorns seastar (*Acanthaster plancii*) has not been the villain in Moreton Bay as it has been in the central and northern Great Barrier Reef, however, that is not to say that coral eating organisms are not present within the coral communities of Moreton Bay, or that they may not become a problem should conditions become favourable for them. Ida Fellegara's paper reports on a survey of the snail, *Drupella*. *Drupella* are coral eaters and have formed outbreaks (population explosions) on Ningaloo Reef in Western Australia which have led to substantial coral damage. Fellegara reports, however, that the populations of this snail on Moreton Bay corals currently do not seem to be cause for concern. It is of course wise to maintain observation of any naturally occurring animal (or plant - see Overview, Chapter 5) that has the potential to outbreak and cause widespread damage.

Clearly, the corals of Moreton Bay are both diverse and vulnerable. As Peter Harrison *et al.* point out, at least two reefs in the Moreton Bay region are ecologically significant and require both protection and management.

Ian R Tibbetts

The Corals of Moreton Bay: Living with Extremes

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Abstract

This paper presents an overview of the Holocene development of reef communities in Moreton Bay, and of the physical environment and characteristics of the modern reef communities. The review provides an observational basis for interpreting relationships between the reef communities and physical conditions in Moreton Bay. We suggest that three major aspects of the physical environment have influenced reef communities in Moreton Bay. These are: flooding, water quality gradients and water temperature.

Coral cover and colony size classes can be altered by coral mortality resulting from intermittent flooding in the Bay. Sites close to the mainland may have almost all corals destroyed during a major flood event, while minor floods may selectively remove certain taxa. Sites further from the mainland are less frequently and severely influenced by floods.

Physical gradients of water quality and flushing across Moreton Bay appear to play an important role in determining the relative abundance of various hard substrate benthic taxa. This includes the spatial patterns of coral dominance, with faviids prevalent in the west and acroporids in the east. Water temperature is likely to play an important role in determining which species can colonise the Bay. Many east Australian coral species commonly occurring at the same latitude as Moreton Bay may be excluded from the Bay by temperature extremes which result from restricted thermal buffering of the shallow reefs. Other physical characteristics may also influence the Bay's reef communities, particularly at spatial and temporal scales finer than those considered in this paper. Characteristics of the coral communities of Moreton Bay are discussed in the context of anthropogenic influences on the reef communities and resultant management issues.

Introduction

Moreton Bay is one of the largest shallow estuarine bays in Australia. Its sub-tropical location leads to a flora and fauna that include a wide variety of both temperate and tropical species (Davie & Hooper, 1993). The diversity of environments within the Bay has led to the co-occurrence of a wide range of estuarine and oceanic species. Remarkably high diversities of echinoderms (Endean, 1953), molluscs (Harrison *et al*., 1995) and other invertebrates, including some significant endemic invertebrates (Davie & Hooper, 1993) exist within the Bay.

Forty-four species of living corals have been identified from Moreton Bay's natural reef communities. Of these, 22 species also occur in the subfossil assemblage along with 4 species that are no longer found in the Bay (species lists in Lovell, 1989 and Harrison & Veron, 1995b).

Early reports of scleractinian corals in Moreton Bay include those of Stutchbury (1854) who reported living coral reefs at Peel Island, of Saville-Kent (1893), who observed living corals at Mud Island as well as those of Bates (1898), who reported observations of living corals at Mud, St Helena, Green and King Islands. More recent observations (Wells, 1955; Slack-Smith, 1960; Lovell, 1975; 1989; Harrison *et al*., 1991; 1995) have included living corals around Peel Island, Goat Island and Myora (North Stradbroke Island), with depauperate coral growth also occurring in patches along the mainland coast (e.g. Wellington Point, Cleveland Point and Point Halloran) and around some of the southern Bay islands (e.g. Cassim, Sandy and Coochiemudlo Islands, pers. obs.; Figure 1).

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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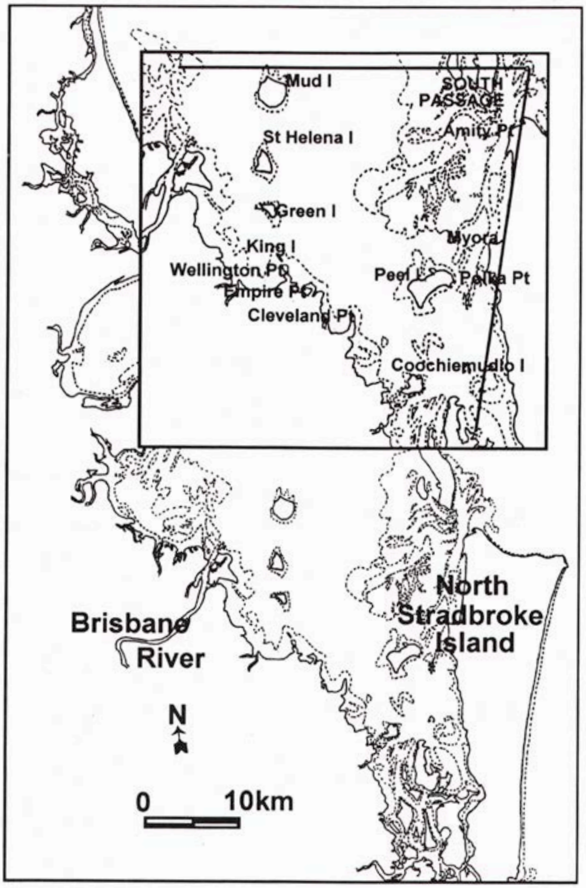

Figure 1. Map of Moreton Bay showing reef locations discussed in the text.

Veron (1995a) estimated that 70% of all central Indo-Pacific coral reefs have been significantly degraded. This degradation is concentrated in fringing and nearshore reefs that are easily accessible to user groups (such as tourists and fishers) as well as being most susceptible to pollution from settlements in coastal catchments. Moreton Bay reefs fit this description, being located adjacent to catchments that include significant areas of agricultural land use and are currently undergoing some of the fastest population growth in Australia (Skinner *et al*., this volume). While reefs within Moreton Bay are generally not seen as a significant tourist destination (in contrast to nearby Flinders Reef), they are heavily utilised by commercial and recreational fisheries, with recreational boating (including live-aboard vessels) also being a significant activity in the area (Robson, 1993). Each of the natural reef communities displays a distinctive combination of environmental characteristics as a result of gradients through Moreton Bay created by: terrestrial influences on water quality predominantly from the mainland, tidal influences from the ocean entrances, differing exposure to wind waves, and variations in sediment and substrate characteristics.

Most investigations of corals and reef communities in Moreton Bay have focused on relationships between spatial patterns within the coral assemblage and spatial and temporal variation in environmental conditions (e.g. Slack-Smith, 1960; Lovell, 1975; 1989). This paper follows a similar approach. Using information available in the literature and some previously unpublished data, this paper provides an overview of the coral communities in Moreton Bay, focusing on: changes in the Moreton Bay environment that parallel those determined in Holocene reef assemblages; physical characteristics of reef sites in the Bay; characteristics of the modern Moreton Bay reef communities; relationships between the reef communities and the physical environment; and anthropogenic influences on the reef communities. From these reviews new analysis and inferences are used to extend our current understanding of corals in Moreton Bay and their relationship to the environments in which they occur.

The Environmental Setting

The prevailing oceanic current system, the East Australian current, carries warm tropical water from north to south outside of Moreton Bay (Veron, 1995b). The East Australian Current allows the transport of plankton, including the larvae of corals and other invertebrates from the Great Barrier Reef to sites including Moreton Bay, Flinders Reef, the Solitary Islands, Elizabeth and Middleton Reefs and Lord Howe Island (Veron, 1995b).

Currents within Moreton Bay are dominated by tidal inflow and outflow through the ocean entrances with variable effects from wind forcing (seasonally) and flows from the catchment (especially during floods). The Northern Entrance (between Bribie and Moreton Islands) and South Passage (between Moreton and North Stradbroke Islands) strongly influence water movement around the reef sites. The largest tidal influence is through the Northern Entrance, which creates a strong north-south current adjacent to Moreton Island and dominates tidal movements south as far as Mud Island. The area influenced by the Northern Entrance is largely devoid of corals, having extensive areas of unsuitable substrate (mobile sands), while hard substrates near the mainland are influenced by runoff from the Brisbane and Pine Rivers, which tends to move north along the coast. South Passage is the entrance closest to reef sites in Moreton Bay. Tidal exchange through South Passage generates strong currents through the Rainbow and Rous Channels that dominate water movement around reef sites such as Myora, Goat Island and Peel Island. Currents at reef sites in the western Bay, such as Green Island and Wellington Point, are much weaker than those in the eastern Bay. The Jumpinpin Bar and Gold Coast Seaway entrances have little influence on tidal exchange around the reef sites (e.g. Patterson & Witt, 1992).

The strongest winds that influence the Bay tend to occur during summer, often associated with storms or cyclones. Summer sea breezes (northeast to southeasterlies) also tend to be stronger than those occurring at other times of year. Winter winds, usually westerlies, are generally much weaker than the summer winds and commonly lead to lower wave energies around the reef areas. Each of the reef sites is exposed differently to these wind conditions, with the dune barrier islands (Moreton and North Stradbroke Islands) sheltering sites in the eastern Bay from the sea breezes while the mainland protects western Bay sites from westerly winds.

While most of the reef sites consist of a veneer of living corals growing on unconsolidated Holocene carbonate deposits (discussed below), at a coarse scale the composition of non-reef sediments varies across the Bay. Western Bay sediments are predominantly terrestrial mud that is prone to wave resuspension. Such sediments have been deposited at reefs around Cleveland, Wellington Point and Green Island. Around Peel Island, in the central Bay, the sediment consists of muddy sands, less influenced by terrestrial inputs and hence containing less fine material. Sites such as Myora and Goat Island, in the eastern Bay, have received relatively little sediment input from the mainland. Coarse quartzose sands deposited through the South Passage tidal delta dominate these areas (Stephens, 1992). There are exceptions to this pattern, with fine sediments occurring in the eastern Bay at sites such as the Lazaret Gutter (Peel Island) and Polka Point (North Stradbroke Island; pers. obs.).

Holocene Reef Development

Coral assemblages have probably occurred in Moreton Bay during most inter-glacial sea level highstands during the Quaternary period, however, little evidence remains of their extent and characteristics. Pickett *et al*. (1984, 1985) describe a coral assemblage from Amity Point that occurred during the last interglacial period (c. 105 thousand years ago (ka), revised to c. 135 ka (Pickett *et al*., 1989)). Being located at an entrance to the Bay, however, this site and its coral assemblage cannot be regarded as analogous to Holocene reefs within Moreton Bay.

Phase one – rising sea level, warm waters, reef initiation, rapid reef growth

Based on the most recent comprehensive compilation of sea level data for the GBR region (Larcombe *et al*., 1995), it can be inferred that antecedent platforms in the depth range -15 to -5.5 m AHD (Figure 2) would have been submerged by sea levels rising at c. 28 mm/yr. Platforms shallower than 5.5 m would have been submerged at the much lower rate of about 2.8 mm/yr from about 8 to 5.5 ka, after which sea level was apparently stable for several millenia (Figure 2). Rapid sea level rise in conditions marginal to coral growth probably inhibited reef development on platforms deeper than 5-6 m. The decline in the rate of sea level rise after 7.8 ka, accompanied by improving environmental conditions (particularly warmer temperatures), facilitated reef growth on shallow (< 5-6 m) platforms. It was also from about 8 ka that Moreton Bay began to take the form of an embayment rather than an estuary (Evans, 1990), becoming a marine system rather than a riverine one, thus meeting an essential precondition for coral colonisation.

Colonisation of suitable substrate in Moreton Bay probably occurred as the Bay filled during the postglacial marine transgression. At Empire Point, initial colonisation occurred at 7.5 ka, at a depth of 7 m below present mean sea level (Flood, 1978). By analogy with sites elsewhere (e.g. the Great Barrier Reef (Hopley *et al*., 1983; Johnson & Risk, 1987)), water quality in the Bay at this time is likely to have been quite poor as a result of reworking of soils of the drowning coastal plain. As sea level continued to rise, winnowing and removal of fine sediments and

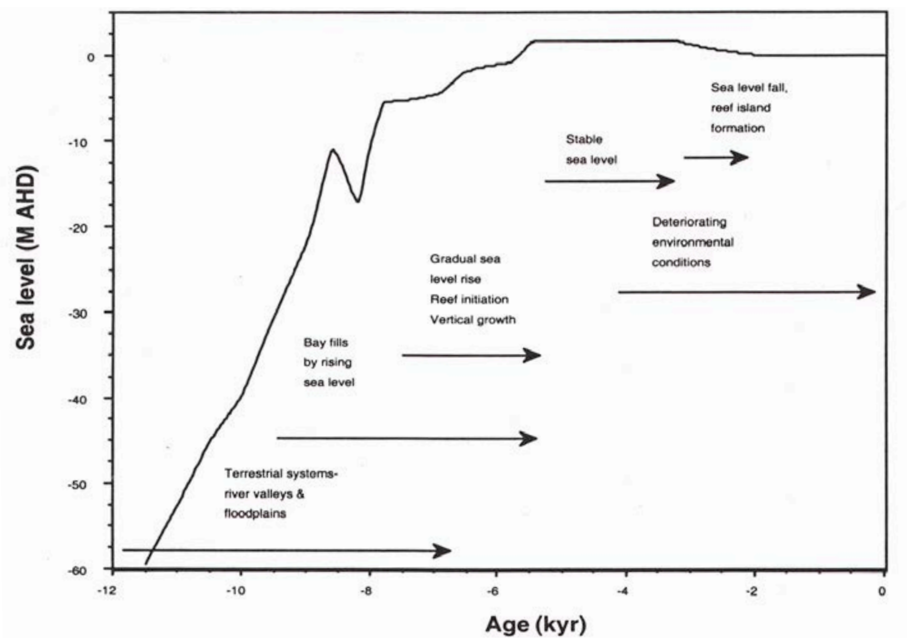

Figure 2. Late Holocene sea level change.

increasing water depth is likely to have progressively improved water quality .

Although there are few dates and limited stratigraphic data for reefs in Moreton Bay, some generalisations can be made regarding the sequence of reef development. Fringing reefs and coral communities developed around the margins of Mud, St Helena, Green, Peel, Goat and the northern coasts of Coochiemudlo, Macleay and Lamb Islands (Jones, 1978; Stephens, 1992). The reef/ island complexes of Green and Mud Islands may be more correctly classified as "low wooded" (Neil, 1993; *sensu* Stoddart, 1965) and "mangrove" (Allingham & Neil, 1995; *sensu* Stoddart & Steers, 1977) islands respectively, rather than as fringing reefs. Although little is known of the detailed stratigraphy of the reefs, data from Mud Island indicate that, at the position of the present inner reef flat, reef development occurred on a platform of about 6 m depth (Richards, 1932) and reached the level of the present intertidal at about 6 ka (Marshall, 1975).

In the western Bay, mainland fringing reefs dominated by *Acropora* spp. developed, particularly between Cleveland Point and King Island (Flood, 1978). At Empire Point the reef attained a thickness of up to 5 m at the landward margin where colonisation occurred close to the time of sea level stabilisation. Coral growth was apparently unable to keep pace with sea level rise at the seaward margin of the reef. The development of this extensive mainland fringing reef is significant, given that there are no true mainland fringing reefs extant south of Cape Conway (20°30' S; Hopley, 1982).

Phase two – Stabilisation of sea level at c. 6 ka set coral growth upper limit

At this time, flooding from the catchment is likely to have been less variable with fewer extreme events, and sediment and nutrient concentrations in river waters lower than at the time of European settlement. In the Bay, conditions are likely to have been more "oceanic", with deeper, warmer, less turbid waters that experienced less variable temperatures and better tidal flushing (Neil, this volume). Such conditions represent a period, at about 6.5-5 ka, which was optimum for coral growth in Moreton Bay. By comparison with the GBR at the same time, poor water quality and cool temperatures in Moreton Bay would have resulted in lower coral growth rates. Consequently, reefs that had lagged behind sea level rise were probably unable to catch up, thus maintaining the seaward sloping morphology that characterised their development. The physical conditions at this time led to a coral assemblage of relatively high diversity, dominance by acroporid corals and a reef area of much greater extent than at present. Lovell (1989) reported reef development and coral deposits in numerous areas where they are now scarce or absent, including Coochiemudlo, Macleay, Perulpa and Lamb Islands and the Pelican Banks.

Sea level may have remained stable from the mid Holocene highstand until about 3.7 ka (Beaman *et al*., 1994), although there are alternative interpretations (e.g. sea level falling smoothly from 6 ka to present, e.g. Chappell, 1983; Lambeck & Nakada, 1990). Hekel *et al*. (1979) point out that the most recent date for the *Acropora*-dominated facies in the Bay is 3 260 ± 260 BP (at Peel Island; Rubin & Alexander, 1958) suggesting that *Acropora* dominance of the coral assemblages was maintained until at least that time. Persistence of the *Acropora* facies is consistent with the results of Beaman *et al*. (1994) and with inferences regarding sea levels in Moreton Bay during the present highstand (e.g. Flood, 1984; Kelley & Baker, 1984, Hacker & Ward, 1985).

Sea level stability should not be taken to imply stability in other aspects of the physical environment. Some deterioration in the climate is likely to have commenced prior to 3 ka, as evidenced by the palynological data for eastern Australia, and catchment conditions and water quality are likely to have deteriorated as a result (Neil, this volume). The interplay of various forcing functions, thresholds and lags means that not all physical changes and ecological responses are necessarily in phase.

Phase three – Global climatic changes

Global climatic changes in the late Holocene led to deterioration of environmental conditions in Moreton Bay, both directly and indirectly, as a result of significant changes in the catchment (Neil, 1998). Both Stutchbury (1854) and Saville-Kent (1893) noted the difference between modern Moreton Bay coral communities and those of the past. Their observations indicate that not all of the deterioration in these communities can be attributed to land use intensification following European settlement, which largely postdates these reports. Evidence for the changes includes the shift from *Acropora* spp. to faviid corals and a marked decline in the hard coral diversity of the Bay (Wells, 1955; Lovell, 1975, 1989). Recent surveys (e.g. Harrison *et al.*, 1991) indicate higher diversity at present than in the sub fossil assemblage, although this is likely to be, at least in part, due to the greater taxonomic difficulties and lesser search effort applied to the subfossil skeletal material. Hekel *et al.* (1979) suggest that the shift in coral composition occurred between 3.2 ka (the date of the Peel Island *Acropora* facies) and the time of European settlement (the observations of Stutchbury, 1854). Similarly, at Empire Point, the once prolific fringing reef was buried by up to 0.5 m of terrigenous and organic sediments and colonised by mangroves and seagrasses (Flood, 1978).

Similar late Holocene changes in coral communities occurred elsewhere in the Indo-Pacific region (Veron, 1992; 1995b; Cabioch, *et al.*, 1995; Kennedy & Woodroffe, 1997). At Tateyama in Japan (Veron, 1992; 1995b), where a tectonic uplift event preserved the coral assemblage of 6-5 ka, 72 hermatypic coral species were recorded compared with only 35 recorded there this century. Citing the specific example of Huon Peninsula (Chappell, 1974), Veron (1995b) notes that, although other mid- to late Holocene reefs have been uplifted, none reveal changes to coral species composition during the late Holocene. The contrast between environmental conditions at these two sites suggests an explanation of the marked change at Tateyama and the lack of it at Huon. Huon Peninsula is an equatorial site with a small, limestone dominated hinterland, resulting in few stressors (e.g. low temperature, terrigenous runoff and sedimentation) for corals. Conversely, Tateyama is "a place where physical-environmental constraints are critical" (Veron, 1995b). It is at sites at the environmental extremes for reef growth, such as Moreton Bay, that environmental changes are likely to have strongly influenced reef development, composition and diversity.

At Tateyama the decline in diversity was attributed to a decrease in mean annual temperature of less than 2^{o}C. At fringing reef sites in New Caledonia, a change from branching (*Acropora*) facies to massive (*Porites*) facies during the late Holocene is attributed to changing wave energy and terrigenous sediment inputs (Cabioch *et al.*, 1995). Changes in the Moreton Bay coral assemblage appear to reflect elements of the relationship between environmental pressures and coral community response at both Tateyama and of New Caledonia.

The pattern of reef growth at Empire Point (Flood, 1978) and the absence of an *Acropora* reef slope community on modern reefs in Moreton Bay suggest that vertical, rather than lateral, reef accretion, and transitions forced by environmental change rather than ecological disturbance are likely to have been the major influences on the structure and composition of Moreton Bay reefs (see Chappell *et al.*, 1983). The persistent colonisation of disturbed sites in the Bay by massive faviids, rather than the faster growing acroporids (Johnson & Neil, Chapter 7 this volume), also supports the conclusion that environmental change, rather than transitions forced by ecological disturbance or reef zonation changes, is the best explanation of Holocene changes in coral community characteristics in the Bay.

The late Holocene changes in Moreton Bay coral communities are particularly interesting, given that: the late Holocene change at nearshore reef sites was generally from *Acropora* to *Porites*, even as far south as Lord Howe Island (Kennedy & Woodroffe, 1997); *Porites* (four species (Wells, 1955; Lovell, 1989)) occurs at Flinders Reef; an assemblage co-dominated by *Acropora* and *Porites* occurred at Amity Point during the last interglacial (Pickett *et al.*, 1984; 1985; 1989) and also at a mainland site 5 km inland of the present Evans Head (Pickett, 1981), about 200 km south of Moreton Bay; *Porites* does not occur in either sub-fossil or modern coral assemblages within Moreton Bay; and the Moreton Bay phase change involves *Acropora* » *Favia*. These patterns suggest that there is an unusual, if not unique, coral community in Moreton Bay, in terms of its composition and association with the physical environment, and that water quality and temperature are unlikely to be the only factors influencing the species assemblage in the Bay. If this were the case, *Porites* should have occurred at some point during the late Holocene.

In summary, coral communities probably started to colonise antecedent platforms in Moreton Bay from about 8 ka, at which time water quality would have been poor during reworking of soil mantles. Water quality and environmental conditions are likely to have improved to their most suitable for coral growth during the mid Holocene climatic optimum. Reef growth was predominantly by vertical accretion, keeping pace with sea level rise at the landward margins and lagging behind in deeper water. Since the climatic optimum a deterioration of conditions resulted in a marked shift in the characteristics of Moreton Bay coral communities, a pattern similar to that in some marginal reef environments elsewhere. Falling sea level is also likely to have been a trigger for sediment mobilisation and onshore transport to form biogenic reef islands and beach deposits in the Bay.

The Physical Environment

Terrestrial Inputs

Terrestrial input to Moreton Bay reefs is predominantly a result of high flow stream discharges (floods) originating from the mainland, and includes fine suspended sediments and low salinity water. Floods in the area are generally caused by storms associated with tropical cyclones (Cossins, 1990). The Brisbane River catchment, the largest source of freshwater input to the Bay, is often the focus of discussions about flooding in the Bay. The importance of additional sources of terrestrial input to Moreton Bay reefs, includes smaller rivers and streams originating from mainland and island catchments, as well as runoff from rainfall in intertidal and adjacent supratidal areas adjacent to the reef sites has yet to be studied (Johnson & Neil, Chapter 7 this volume). Most studies of living reefs in Moreton Bay have stressed the importance of floods as the predominant structuring force (e.g. Slack-Smith, 1960; Lovell, 1975; 1989) despite the absence of data about other processes in the reef communities.

Observations made by Stephenson *et al*. (1968) were used to explain patterns of coral mortality observed by Lovell (1975) in the absence of a significant data set pertaining to a major flood. Depth profiles of salinities measured throughout an area including the Brisbane River mouth, Mud, St Helena, Green, Peel, Coochiemudlo and North Stradbroke Islands during a minor flood in 1963 (Stephenson *et al*. 1968) provided important data on the physical characteristics of flooding in the Bay, but no observations of coral morbidity or mortality at the same time. The greatest reduction in salinities occurred in shallow waters in western areas of the Bay. Higher salinities were maintained in deeper waters where limited mixing impeded their reduction, and near the ocean entrances (i.e. in eastern areas of the Bay) where the tidal exchange limited the reduction of salinity. These patterns are likely to vary in intensity, extent and duration depending on the nature of the rainfall event driving them (Johnson & Neil, Chapter 7 this volume), particularly as the 1963 flood was relatively minor compared with those of 1956 and 1974, both of which caused coral mortality in the Bay.

Sediment dynamics

Islam *et al*. (1995) used Landsat TM (Thematic Mapper) imagery to investigate Baywide spatial patterns of water quality. The study showed a strong gradient of decreasing suspended sediment concentrations and turbidities between the Brisbane River and South Passage. While a number of studies have examined suspended sediment concentrations and/or water clarity in open waters (e.g. Dennison *et al*., 1993; Islam *et al*., 1995), few data have been collected pertaining to shallow waters, particularly in reefal areas. A brief summary of the methods and results of a study investigating spatial and temporal patterns of sedimentation at the reef sites follows.

Sediment samples were collected using 50 mm diameter, 500 mm long cylindrical sedimentation traps (aspect ratio of 10, recommended by Bloesch & Burns, 1980). Replicate pairs of sediment traps were deployed on reef flats at Green Island, Peel Island, Goat Island and Myora with samples collected at approximately monthly intervals between June 1993 and July 1994. Sediment traps were positioned vertically on the bottom with the opening at approximately the height of the upper surface of living corals at the site (50-70 cm above the seabed). Sedimentation rates and sediment settling rates were calculated for each sample. Total dry weight of each sample was measured to allow the calculation of sedimentation rates. The settling rate of sediment samples collected from the traps was tested by completely resuspending a subsample in a 100 mL beaker to give a 200 ntu (nephelometric turbidity units) starting turbidity (measured using an ANALITE high sensitivity nephelometer) and recording turbidities every fifteen seconds for thirty minutes after the magnetic stirrer maintaining the suspension was switched off. No floods occurred during the sampling period so sedimentation in the traps can be attributed to bottom resuspension rather than supply from runoff.

Two main patterns are evident in the sedimentation data. Both the magnitude and the seasonality of sedimentation rates decrease with distance from the mainland (Figure 3). The summer maximum sedimentation rate (57.1 mg/cm^2/day) at the site closest to the mainland, Green Island, exceeded those at all other sites and times, while the winter minimum (5.1 mg/cm^2/day) at Green Island was comparable with those at Peel Island (2.0 mg/cm^2/day), Goat Island (6.1 mg/cm^2/day) and Myora (5.9 mg/cm^2/day). In contrast, Myora, the site furthest from the mainland, had consistently low sedimentation rates with relatively little difference between the summer maximum (16.1 mg/cm^2/day) and winter minimum (5.9 mg/cm^2/day) during the 1993/94 sampling period.

Sedimentation rates show a positive relationship to 3 pm wind energies (calculated as the sum of cubed wind velocities measured from all directions) recorded at the Brisbane International Airport (Figure 4). At Peel Island, the only site where fetch is relatively similar in most directions, the relationship is quite strong. Green Island, Goat Island and Myora (all of which have uneven fetches in different directions) also show positive (though much weaker) relationships between sedimentation rates and 3 pm wind energies. The positive relationships between sedimentation rates and wind energy demonstrates the importance of wave resuspension and the reworking of fine components of the sediments at these sites.

Sediment settling rates also show a strong spatial pattern (Figure 5). Settling rates of sediments collected from sites closer to the mainland were slower than those from sites in the east of the Bay. This is likely to be a result of the presence of fine terrigenous sediments that become less prevalent further from the mainland.

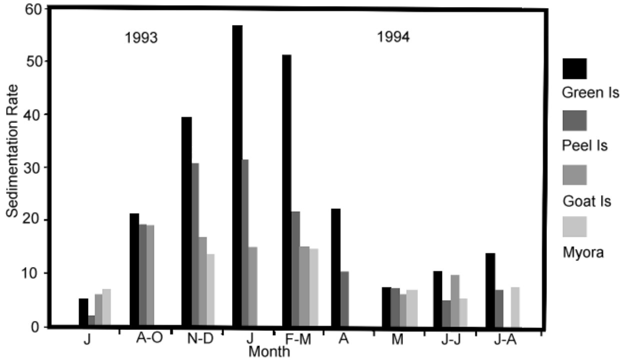

Figure 3. Seasonal sedimentation (mg/cm²/day) at four Moreton Bay reef sites.

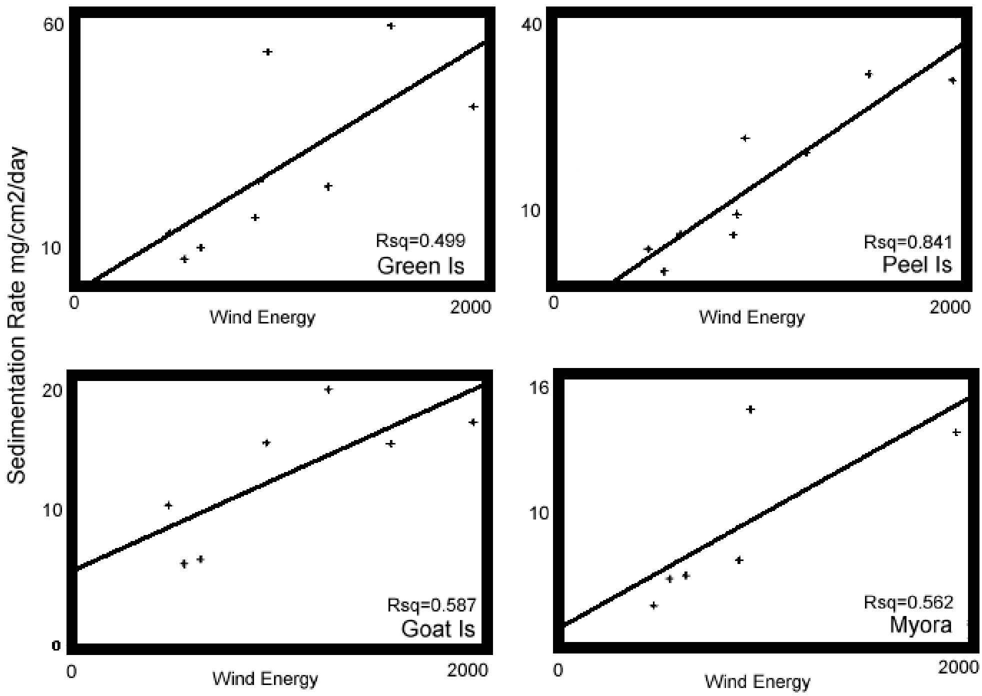

Figure 4(A-D). Relationships between wind energy and sedimentation rates at Moreton Bay reef sites.

Sediment concentrations in the water column were not measured directly, however, estimates of the mean sediment concentration at each site can be calculated as the product of sedimentation rates (Figure 6), which are proportional to the quantity of suspended sediments passing over a trap (Bloesch & Burns, 1980)), and average flow rates, which determine the volume of water sampled by the trap. Average flow rates were calculated from hydraulic process model data provided by WBM Oceanics (discussed below). While there are likely to be errors associated with the estimates, the magnitude of the differences between sites suggests a seasonal and spatial variation of suspended sediment concentration consistent with, but much stronger than, that based on sedimentation rates.

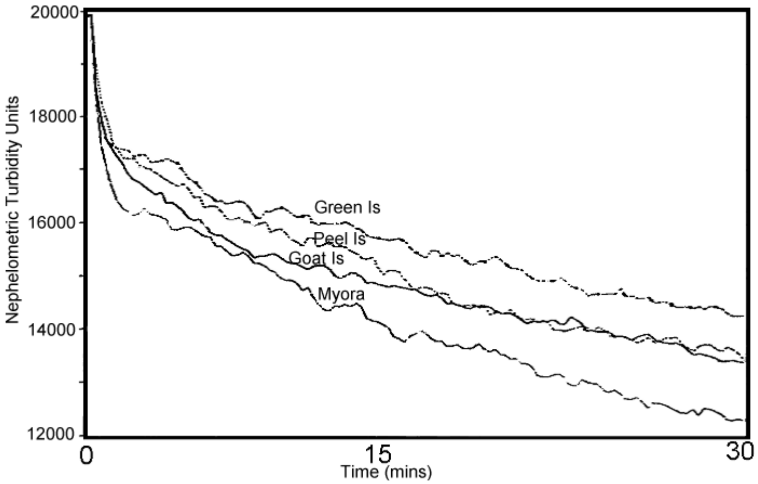

Figure 5. Mean sediment settling rates at Moreton Bay reef sites.

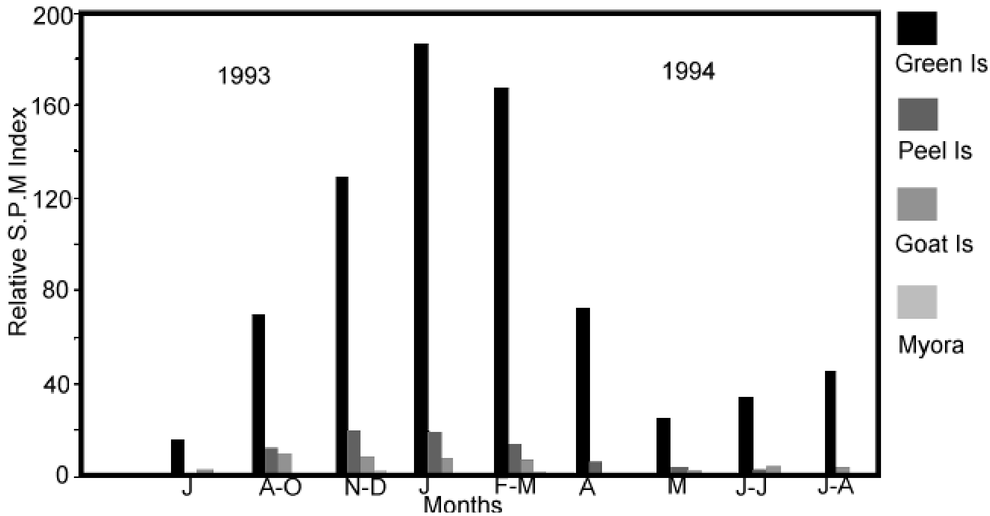

Figure 6. Relative suspended particle matter (SPM) at four Moreton Bay reef sites.

Both the quantity and composition of suspended sediments and sedimentation at reefs in Moreton Bay vary with distance from the mainland. This inshore to offshore (west to east) gradient is likely to be related to the increasing tidal flows, decreasing exposure to the predominant winds (southeast to northeasterly) and resultant waves, the supply of terrigenous sediments from the mainland and the presence of substrates derived from aeolian and marine sands in the eastern Bay.

Tidal flows

Estimates of the tidal currents at the reef sites studied were obtained using the hydrological model of Patterson & Witt (1992). Data obtained from the model (which uses a 500 m x

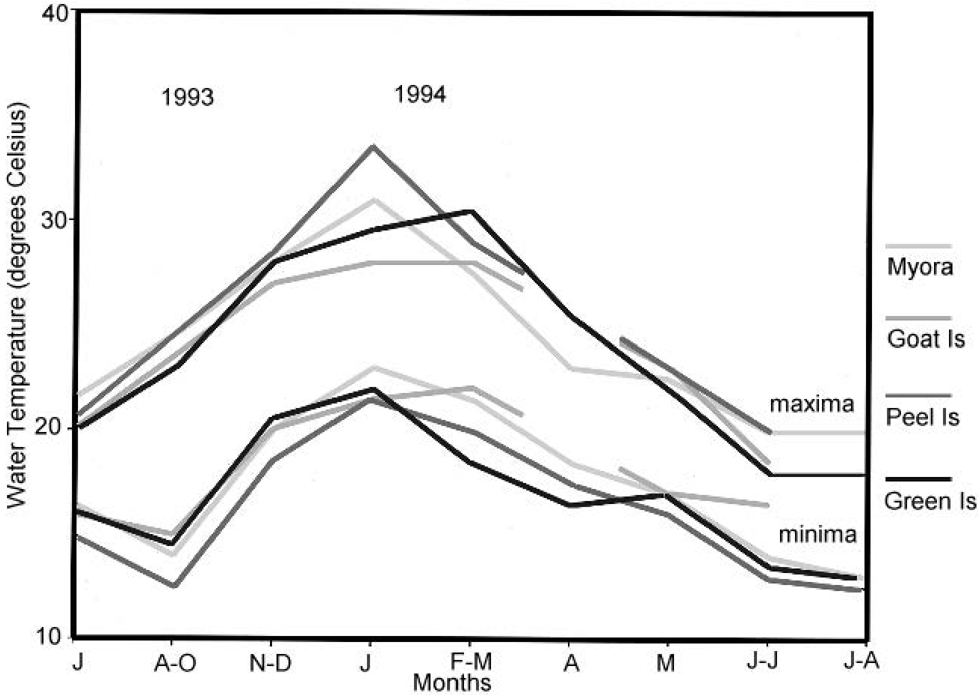

Figure 7. Minimum and maximum temperatures at four Moreton Bay reef sites.

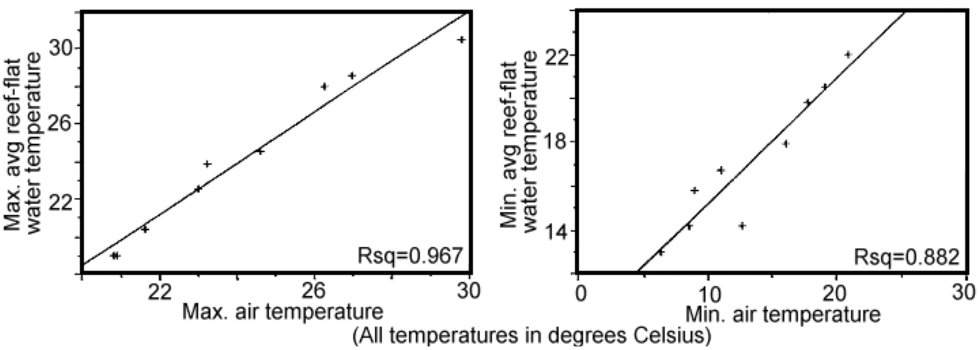

Figure 8. Correlations between air and water temperatures for Moreton Bay reef sites.

500 m grid) are limited in the extent to which they can describe flows over a narrow reef flat, however, the general pattern of increasing tidal flows with proximity to the ocean entrance (South Passage) is clear. The cross-Bay variation in tidal movement is likely to have a strong influence on sediment dynamics and terrestrial inputs at each of the reef sites as well as directly influencing the benthos.

The model (Patterson & Witt, 1992) predicts strongest tidal influence through the Northern Entrance, generating north-south flows that dominate tidal exchange as far south as Mud Island. Most of the other reef sites are dominated by tidal exchange through South Passage, which is strongest in the east (near the Passage). The stronger tidal flows of the eastern Bay are responsible for greater flushing which contributes to better water quality in the eastern Bay.

Temperatures

Temperature patterns relevant to reef sites in Moreton Bay were investigated using: sea surface temperatures for Moreton Bay and similar latitudes outside the Bay from the published literature; air temperature records from the Brisbane International Airport (taken daily at 3 pm, Australian Bureau of Meteorology data); and observations made at reef sites in the Bay (Johnson & Neil, unpublished data). To obtain the latter, minimum/maximum thermometers in housings (which excluded sunlight and allowed water movement) were deployed on reef flats at Green Island, Peel Island, Goat Island and Myora and recordings made at approximately monthly intervals between June 1993 and July 1994. The thermometers were positioned on the bottom (approximately two metres below mean sea level, at the depth of coral growth) at each of the sites.

Sea surface temperatures of waters outside Moreton Bay are strongly influenced by the East Australian Current which flows south from the tropics, giving temperatures that are generally between 18°C and 26°C (Lough, 1994; Table 1). The annual variation in surface water temperatures from Moreton Bay is approximately 16°C to 28°C (Hedley, 1915; Greenwood, 1963). The annual range of temperatures recorded for reef sites within the Bay was approximately 12.5°C to 32°C during the June 1993 – July 1994 study period (Figure 7). Reef flat temperatures are likely to be most influenced by air temperatures, having a small volume of water per unit area (especially at low tide) and limited exchange with water from deeper areas. Surface (open) water data for Moreton Bay are less influenced by air temperature, as they are recorded in areas that are deeper and exchange more readily with the open ocean. Sea surface temperatures from outside Moreton Bay are probably least affected by air temperatures since the volume of water per unit area is far greater, as is the potential for exchange with deeper waters. The difference in sea surface temperatures between Bay waters and oceanic waters has been demonstrated by Steele & Kuhl (1993), who showed that a tidal plume of warm ocean water (1-5°C warmer than Bay water) extended up to 5 km inside South Passage on flood tides during winter.

Table 1. Sea surface temperature ranges of Moreton Bay and surrounding waters ([1]Johnson & Neil, unpublished data;[2]Hedley, 1915; Greenwood, 1963; [3]Lough, 1994).

All of the reef sites studied in Moreton Bay had a similar temperature regime with no detectable spatial variation. This suggests that the tidal thermal plume detected by Steele & Kuhl (1993)

Table 1. Sea surface temperature ranges of Moreton Bay and surrounding waters ([1]Johnson & Neil, unpublished data;[2]Hedley, 1915; Greenwood, 1963; [3]Lough, 1994).

TEMPERATURE (°C)	*Moreton Bay Reefs*[1]	*Moreton Bay Surface Waters*[2]	*Adjacent Sea Surface*[3]
Minimum	12.5	16	18
Maximum	32	28	26

has little or no effect on the temperature range at the reef sites studied. Given the limited tidal circulation in shallow waters covering the reef flat and the limited incursion of oceanic waters into the Bay (Islam, 1998) this should be expected.

Simple linear regressions between the average minimum or maximum reef flat temperature (averaged between the four sites) and the average minimum or maximum air temperature (averaged between the days of each sampling period) indicate that a strong relationship exists between air temperatures and water temperatures on the reef flat (Figure 8). This suggests that reef flat temperatures are strongly influenced by minimum ($r^2 = 0.88$) and maximum ($r^2 = 0.97$)

air temperatures.

Reef Assemblages

Fine scale patterns

Moreton Bay reefs are generally ‘patchy’, often interspersed with seagrasses or soft substrate communities (Lovell, 1975; Harrison *et al*., 1991; Johnson & Neil, 1993). Zonation patterns across reef flats occur within Moreton Bay (Munro, 1940; Slack-Smith, 1960; Harrison *et al*., 1991) with hard corals being present subtidally and occasionally in the lower intertidal zone. The seaward extent of the coral communities is generally delimited by the outer edge of the hard substrate (Harrison *et al*., 1991). Most corals in Moreton Bay occur in depths of < 3 m below mean low tide, with some patch reefs extending to a depth of almost 5 m (Lovell, 1975). Moreton Bay reef flats support not only the coral assemblages which are the focus of this paper, but also seagrass, algal and soft substrate habitats with associated benthic communities (Harrison *et al*., 1991; Johnson & Neil, 1993).

Baywide patterns

Descriptions of Moreton Bay reef communities by Wells (1955), Slack-Smith (1960), Lovell (1975, 1989) and Harrison *et al*. (1991, 1995) can be used to outline Baywide patterns within the Moreton Bay coral assemblage. Some caution must be used when interpreting these observations since most of these studies focused largely on areas of prolific coral growth, which may only represent part of the reef community at any given location.

Modern corals are distributed within a roughly triangular area extending from Mud Island in the northwest and Point Halloran in the southwest to Myora (North Stradbroke Island) in the east. This area includes reef communities on the mainland coast (Wellington Point/ King Island, Empire Point, Cleveland Point and Point Halloran), the central Bay Islands (Mud, St Helena, Green, Peel and Goat), the northernmost of the southern Bay islands (Coochiemudlo, Cassim, Sandy, Macleay, and Lamb) and North Stradbroke Island (Polka Point, the Parrot Hole and Myora Reef). For the purposes of this paper this area will be referred to as the ‘coral quadrant’.

Within the Moreton Bay assemblage the most obvious patterns occur from southwest to northeast across the coral quadrant (between the turbid mainland waters and the South Passage oceanic influence). These patterns include a reduction in the dominance of *Favia speciosa* from southwest to northeast, with Myora in the northeast being dominated by *Acropora digitifera*. Also occurring from southwest to northeast is a pattern of increasing coral cover, with many western Bay sites (e.g. Cleveland Point, Point Halloran) being represented by small, sparsely distributed coral colonies. Maximum hard coral cover occurs at Peel Is, Goat Is and Myora (Harrison *et al*., 1991; 1995, pers. obs.). Sites further south (e.g. Point Halloran, Coochiemudlo, Sandy, Cassim, and Macleay Is) display similar spatial variation from south to north, with the southern sites having low coral cover and being dominated by faviid corals, although these areas have received much less attention in the literature. The cross-Bay patterns evident in the coral assemblage are also reflected in other benthic communities at reef sites. Both macroalgae and bivalves cover much more of the substrate in the western Bay than in the east. *Alcyonium* sp. is the dominant soft coral in the western Bay while *Xenia* sp. and *Sarcophyton* sp. tend to occur more often at eastern Bay sites.

The extent of reef communities also varies through the Bay, with the most extensive communities occurring around the central Bay Islands (especially Peel Is). It is likely that the area occupied by reef communities at both the western and eastern margins of Moreton Bay are limited by sediment accumulation (Stephens, 1992). The introduction of artificial hard substrates in the eastern Bay, such as the Amity Point rock wall (North Stradbroke Island) and the Tangalooma Wrecks (Moreton Island) has allowed coral colonisation in relatively good (near oceanic) water quality. In both situations *Pocillopora damicornis* has been found growing along with *Acropora digitifera* and other *Acropora* species (pers. obs.). The only other record of *Pocillopora damicornis* occurring in the modern assemblage was made by Harrison *et al*. (1995, this volume), who found a single "…small, recently-killed colony" at Myora Reef.

Biophysical Relationships

Disturbance events

The literature recognises floods as the most significant disturbance events that affect Moreton Bay coral communities. Slack-Smith (1960) and Lovell (1975) have suggested that Baywide patterns in coral assemblages, particularly patterns of species composition and relative abundances, occur as a result of flood related mortality. Observations of coral mortality after flooding have been inconsistent, although some patterns can be identified (e.g. Johnson & Neil, Chapter 8 this volume). Flood related coral mortality tends to be patchy, probably due to specific tide, wind and runoff conditions during the flood. Reef sites that are further from the mainland (in the eastern Bay) or are in deeper water tend to be less susceptible to the effects of flooding. During large floods (e.g. 1974 (Lovell, 1975)) almost all corals are lost from most sites near the mainland. Selective mortality of coral species may occur at sites subjected to minor flood impacts (Lovell, 1975; Johnson & Neil, Chapter 8 this volume).

Floods have the potential to create a distinct temporal cycle at disturbed sites. For example, many sites in the western Bay suffered almost total mortality after the 1974 flood. Because of this, coral abundances, cover and size classes all increased with time after the flood. Both Slack-Smith (1960) and Lovell (1975) suggested that floods are also responsible for the spatial variations in dominance and relative abundances in the Moreton Bay coral assemblage. Temporal changes in relative abundances with recovery after a disturbance (or 'succession') has been recognised in other coral reef ecosystems (e.g. Connell, 1978; Cameron & Endean, 1985). However, flooding can, at best, only offer a partial explanation of the spatial variation in Moreton Bay reef communities because the recovery response is not uniform. For example, reef sites which were devastated during the 1974 flood have recovered in different ways rather than going through similar 'successional' stages. The reef communities that developed since 1974 tend to be similar to those that existed at the same sites before the flood, leading to persistent patterns of relative abundance throughout the Bay.

In Moreton Bay it appears that coral recovery after a flood at any given site does not rely on the persistence of corals at that site (Johnson & Neil, Chapter 8 this volume). Many reef sites experienced 100% coral mortality from the 1974 flood and would be unable to recover persistence of each species was required. Many of the coral species are so few in number they are unlikely to represent a viable self-seeding population. This suggests that recovery from flooding is a function of settlement and recruitment of coral propagules supplied from less affected sites. Patterns of flood-induced mortality do not explain the persistent spatial patterns of relative abundances and spatial variation in the rates of growth and/or recruitment. This probably relates to spatial patterns in environmental conditions (discussed below) and may also be influenced by variations in the supply of larvae.

Biophysical cross-Bay gradients

Sediment quality, sedimentation rates, the seasonality of sedimentation, and flow rates all vary across the Moreton Bay coral quadrant. Clear water and high flushing rates occur close to South Passage in the east, while in the west and south near the mainland, water clarity and flushing rates are low while sedimentation rates are high. It is generally accepted that environments with limited flushing, high sedimentation rates, and high suspended sediment loads and/or turbidities are deleterious to coral growth, as is often reported for nearshore fringing reefs (Endean *et al*., 1956; Sheppard, 1982; Hopley, 1982). Several studies have implicated high sediment loads and fluxes as being responsible for the paucity of certain coral communities (e.g. Squires, 1962; Cortes & Risk, 1985; Rice & Hunter, 1992). Within reefs, zonation patterns and aspects of reef morphology vary with turbidity (Johannes, 1975; Sheppard, 1982), presence of fine sediments (Loya, 1972) and the characteristics of the sediments (Johannes, 1975). Cross-shelf variations in coral communities of the Great Barrier Reef parallel changes in sediment loads, sediment types and associated hydrodynamic conditions. Increases in the dominance of *Acropora* species (Done, 1982) and increases in the settlement and recruitment of *Acropora* larvae (Sammarco, 1983) occur with distance from the mainland (and terrigenous sediments). Examples of mechanisms that could cause the cross-shelf patterns on the GBR as well as the cross-Bay patterns in Moreton Bay include settlement and recruitment patterns (Sammarco, 1983) as well as processes within mature populations. For example, Hubbard & Pocock (1972) demonstrated that the ability of corals to remove sediments from their surfaces differs between species, and that species differ in their capacity to remove specific particle sizes.

In Moreton Bay differences occur between sites with increasing distance from the mainland, including an increase in the proportion of the substrate covered by scleractinian corals and a reduction in the dominance of *F. speciosa*, with *A. digitifera* becoming the dominant species at Myora. The differences in the community structure between coral reefs within Moreton Bay, particularly the density and relative abundances of the dominant taxa, are likely to relate to cross-Bay variation in environmental conditions in a similar way to that documented for the Great Barrier Reef and other coral reefs throughout the world. Moreton Bay is perhaps different in that these patterns occur at a much smaller scale (all of the natural reef sites occur within 30 km of each other) and there is no true “oceanic” end point to the Moreton Bay gradient, with reefs close to South Passage (e.g. Myora) subject to tidal currents much stronger than the offshore oceanic currents and generally subjected to Bay waters as well as oceanic water quality (Islam, 1998).

Summary of Reef Community Pattern and Process in Moreton Bay

This review has focused on coarse scale relationships between reef communities (emphasising scleractinian corals) and physical conditions in Moreton Bay. Biotic processes (such as competition and predation) may structure these communities at scales finer than those addressed in this paper, although there has been no attention to such processes at reefs in Moreton Bay.

In western Moreton Bay coral colonies tend to be smaller (although some large colonies do exist in these areas (Harrison *et al*., 1991)) and all coral taxa occur at low colony densities and hence, low coral cover. At western Bay reefs other invertebrates (such as soft corals, sponges and bivalve molluscs) and algae occupy substantial proportions of the substrate. In the western Bay scleractinian colony sizes probably reflect disturbance at the site (principally flooding), with most colonies having recruited since the last major disturbance (e.g. the 1974 flood).

Moreton Bay reefs exist along water quality gradients that seem to control which taxa can dominate reef sites. Even after the 1974 flood, reef communities such as those at sites such as Green and Peel Islands (Lovell, 1989; pers. obs.) recovered with similar patterns of relative dominance to those prior to the flood (Lovell, 1975). These patterns of dominance are likely to represent patterns of water quality since taxa often described as stress tolerant (e.g. faviids, Veron, 1986) tend to be more dominant in areas with higher sedimentation rates and turbidites, lower flushing rates etc., while faster growing less tolerant species (e.g. *Acropora* spp.) only become more common at sites strongly influenced by oceanic waters.

While the temperature range is extreme, it is unlikely to influence cross-Bay spatial patterns between the reef communities since all of the reef sites examined had similar temperature minima and maxima. The temperature regime is likely to influence the Moreton Bay species assemblage since the summer maximum temperatures appear to be similar to reef flat maxima in the southern Great Barrier Reef (e.g. Potts & Swart, 1984) while winter minimum temperatures drop below those of many higher latitude coral communities. The temperature range may explain why the Moreton Bay coral assemblage is so different from neighbouring reefs (e.g. Flinders Reef) and others of the central Indo-Pacific despite the oceanic influences in the eastern Bay. Alternatively, limited tidal exchange and therefore connectivity with other reef communities may explain Moreton Bay's distinct biogeography (Johnson & Neil, 1995). Limited connectivity, however, cannot fully explain the characteristics of the Moreton Bay assemblage since species have been found that recruit to the entrance sites but not the natural reef sites and many species exist in such small numbers at the reef sites that they cannot represent a self sustaining population. This includes new species reported by Harrison & Veron (1995) which have apparently colonised Moreton Bay since the 1974 flood. Such species must be the result of larval supply from outside Moreton Bay.

Anthropogenic Influences on Corals in Moreton Bay

Anthropogenic influences on Moreton Bay reefs occur directly through use of the reef sites, particularly extraction of resources, and indirectly with use of the catchments and of Moreton Bay waterways. Land-use change in the catchment can be attributed to rapid population growth in the region, while increasing use of the Bay relates to cultural changes as well as population growth, with more people having the opportunity to use the Bay for recreational activities.

Direct impacts

Corals in Moreton Bay have suffered acute localised impacts from coral extraction. During early European settlement in Brisbane many sites in western Moreton Bay were utilised for limestone extraction (Petrie, 1904). Petrie describes punts being filled with coral heads collected by hand from reef flats in the western Bay (particularly King Island). In these cases, the impacts of coral extraction are likely to have been most pronounced in the benthic communities, with the removal of living corals and suitable hard substrates from the surface of the reef flat. There are also anecdotal suggestions, from long term North Stradbroke Island residents, that similar forms of extraction may have taken place at Goat Island for early road works on North Stradbroke Island.

More recently (1937-1997) limestone extraction has intensified, with dredging of the reef flats (Allingham & Neil, 1993). Coral dredging has led to the removal of significant sections of reef flat at Empire Point, St Helena Island and Mud Island. In the worst case, at Mud Island, coral dredging has removed the majority of natural substrates for coral recruitment, and the mobile rubble substrates left behind continue to prevent benthic recruitment. Coral dredging has also caused significant bathymetric modification, leading to effectively permanent major geomorphic changes of the Mud Island cay as a result of increased wave exposure (Allingham & Neil, 1993). Coral extraction is not currently undertaken in Moreton Bay, although its effects are still evident. Most direct impacts occur as a result of fishing and recreation at the reef sites. This includes physical damage from anchors and fishing tackle (Harrison *et al*., 1991), as well as pollution associated with boating such as effluent from live-aboard vessels.

Increased aquarium collecting for commercial trade (Harrison *et al*., 1995) and for research and educational use (e.g. university classes and museum collections) has lead to physical damage (such as overturned coral colonies and breakage of branching corals) as well as removal of the target species, usually fish and invertebrates. These activities are most evident in the eastern Bay (e.g. Myora Reef), where reef fish and invertebrates are most easily collected because of their abundance, good visibility, and the ease with which coral colonies can be overturned (Davie & Hooper, 1993).

Indirect impacts

Changes to catchment inputs are the most significant anthropogenic threats to Moreton Bay's reef communities. The most obvious of these are increases in pollutants associated with agricultural and industrial growth and urbanisation of the catchments. Increased pollution in the form of nutrients, pesticides, metals, hydrocarbons and other trace contaminants (Moss *et al*., 1993) can directly effect the functioning of reef organisms if they are toxic, or may shift the ecological balance of reef communities (e.g. increased nutrient levels and or altered nutrient ratios may allow macroalgae to outcompete corals and other sessile benthic organisms). Pollution in Moreton Bay has the potential to be more concentrated than many other sections of the coastline because of the restricted tidal exchange with ocean waters. Sites in the western Bay have the weakest tidal exchange and are subject to the highest levels of pollution.

Other changes to catchment inputs, such as the restriction of flooding through channel modification and the introduction of dams (Cossins, 1990), may have more subtle effects on the Bay's coral communities. A change in the frequency of flood impacts could lead to shifts in reef community structure, although there is currently insufficient understanding of the variability of the reef communities or of the effects of flooding.

Management implications

Moreton Bay's coral reefs are biogeographically distinct from others within Australia's marine reserve system (Harrison *et al*., 1995; Johnson & Neil, 1995). They are subject to the impacts of human activities in the Bay as well as stresses and pressures from the catchment.

Coral extraction is no longer likely to be a significant threat to Moreton Bay reef communities given the current zoning plan and permit system. Efforts to rehabilitate coral dredging sites have largely been limited to the removal of industrial debris for aesthetic and navigational purposes. The seafloor habitats at Mud and St Helena Island dredge sites remain degraded with little recovery of the benthic communities. Other direct impacts are not effectively managed. Under the current Moreton Bay Marine Park zoning plan the northeastern section of the Peel Island reef is the only area of reef currently protected from all extractive activities, having been listed as a National Park Zone. Fishing and collecting are still permitted at all other reef sites within the Bay, while trawling and mining are not permitted at Green Island, Goat Island and Myora, which have been listed as Conservation Park Zones. All of the remaining reef sites have been included in Habitat Zones, which prohibit habitat destruction but allow most other activities to take place. Many of the reef sites within Moreton Bay are distinct from each other, largely as a result of the environmental gradients discussed above. Each site represents a different physical environment, a different benthic species assemblage and different ecological relationships, i.e. each of the different sites is a distinct ecological entity. Zoning should reflect these patterns if biodiversity and representativeness of Moreton Bay reefs are to be retained. Other habitat types with similar cross-Bay patterns (e.g. seagrasses, mangroves and soft substrates) would also benefit if the Moreton Bay zoning plan accommodated such spatial variation in environmental pattern and process.

The most significant attempt at managing pollutants and changes to catchment inputs is the establishment of the Brisbane River Management Group. The Brisbane River Management Group is an attempt at implementing management that crosses the political boundaries of city councils and state government jurisdictions by including representatives of local councils and various state government departments. Similarly the Brisbane River and Moreton Bay Waste Water Management Study has been initiated with the support of such agencies so that more informed decision making can be achieved. This type of initiative needs to applied to all of the catchments entering Moreton Bay in order to limit the impacts of urban and rural land uses. The success of such initiatives will depend on the direct actions taken by each of the agencies involved once recommendations have been made.

Research needs

Coral research in Moreton Bay has provided a description of the distinctiveness of Moreton Bay reefs from others of the Indo-Pacific (Wells, 1955; Harrison *et al.*, 1995) as well as from each other (Lovell, 1975; 1989, Harrison *et al.*, 1991; 1995). To understand why these reefs are so different a better understanding of biophysical processes is needed. Recruitment experiments to quantify aspects of the supply of propagules to and within the Bay could explain some aspects of the connectivity between Moreton Bay and other east Australian reefs. Investigations of the sublethal effects (such as impaired growth or reproduction) of the environmental extremes discussed previously may help to explain the relative dominance of certain taxa within the Bay (e.g. comparing faviid-dominated and acroporid-dominated reef communities). Reef and coral coring accompanied with dating could be used to provide more accurate and detailed descriptions of Holocene sea level and environmental change as well as associated biotic changes in the reef communities.

Conclusion

The Moreton Bay coral quadrant is influenced by gradients of water quality, disturbance frequency and intensity, and oceanic exchange. These gradients interact to create different physical conditions at each of the Moreton Bay reef communities, leading to a complex of different habitats and biota with many sites including species that are not found elsewhere in the Bay. Our current understanding of the Bay's reef communities does not include the mechanisms that link the physical characteristics to patterns and processes within the reef communities, although many of these show similarities to patterns and processes recognised on the Great Barrier Reef (Done, 1982).

Current management of reef communities in the Bay does not reflect the complexity within the system, with a single site zoned as a National Park reserve to represent the Moreton Bay assemblage. This fails to represent the reefal biodiversity or the range of habitats and environments found within Moreton Bay.

Acknowledgements

Funding for new research was provided by the Department of Geographical Sciences and Planning, The University of Queensland as part of a postgraduate research programme. The authors wish to thank the volunteers who assisted with the field work. We are grateful to Peter Harrison, Vicki Harriot, John Jell and Ian Tibbetts for their helpful comments on the manuscript.

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The Coral Communities of Flinders Reef and Myora Reef in the Moreton Bay Marine Park, Queensland, Australia

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Abstract

Flinders Reef and Myora Reef occur within the Moreton Bay Marine Park and contain diverse and highly significant coral reef communities. The marine benthic communities at Flinders Reef and Myora Reef were surveyed in October - November 1994 using replicated video transects to quantify the status of the coral communities present. Flinders Reef is a small 10 ha reef which lies 5.5 km to the north of Moreton Island, and supports a flourishing coral reef community dominated by hard corals, soft corals and algae. The mean cover of living corals was high, and ranged from 30-43% at the three sites surveyed. The western reef area contains a high cover of branching and staghorn *Acropora* colonies, while the southern and eastern reef areas exposed to south-easterly swell are dominated by robust massive and encrusting corals. A total of 119 coral species has now been recorded from Flinders Reef, which represents an extraordinarily high coral species richness given the small size of the reef and its isolation from the Great Barrier Reef. The coral community at Flinders Reef is also of considerable biogeographic interest because it contains a unique association of tropical and sub-tropical coral species, and many of the tropical species are at their southern-most distribution limit. Myora Reef is located on the western side of North Stradbroke Island in the central-eastern area of Moreton Bay. The Myora coral community is dominated by large *Acropora digitifera* colonies, and is the most significant living *Acropora* assemblage inside Moreton Bay. Mean coral cover was high (42%), and at least 12 other coral species were present at this site. The extensive coral communities at Flinders Reef and Myora Reef are likely to have critically important roles in providing dispersive planktonic larval recruits that are essential for maintaining other coral communities within Moreton Bay, and for re-establishing coral communities damaged or destroyed by floods. Thus both reefs support ecologically significant coral communities which are locally and regionally important, and clearly merit protection and management as Marine National Parks.

Introduction

The Moreton Bay region of southeast Queensland (Figure 1) contains both extensive subfossil coral reefs and some areas with living coral communities. The region has had a long and complex history of coral reef growth resulting from fluctuating sea levels and more recent flood events within the Bay (Lovell, 1989). The greatest coral reef development in Moreton Bay occurred during the Holocene period approximately 6 000-7 500 years ago, when sea level was a few metres higher than present mean sea level (Lovell, 1989). These subfossil coral assemblages were dominated by *Acropora* species, and formed extensive reefs throughout the Bay, including areas where living corals are now rare or absent (Lovell, 1989). In contrast, the present living coral communities in Moreton Bay are dominated by brain corals. This dramatic change in species composition and community characteristics between the subfossil and living coral assemblages in Moreton Bay is probably related to the decline in mean sea level, resulting in increased influence from flooding and sedimentation from nearby rivers. Sensitive corals such as *Acropora* species are likely to be periodically eliminated from reefs by flood disturbance, leading to increased dominance of massive brain corals which are physiologically more tolerant of flood conditions.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.* School of Marine Science, The University of Queensland, Brisbane. pp. 525-536.

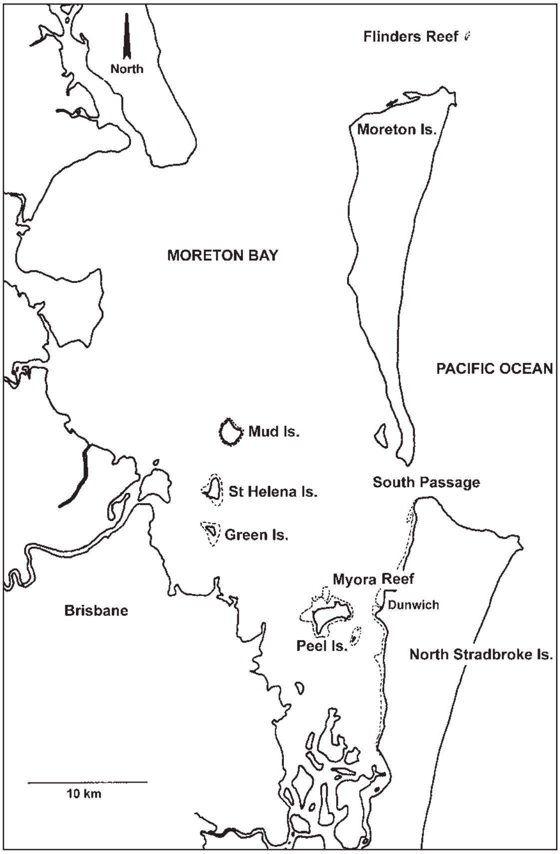

Figure 1. Map of central and northern Moreton Bay showing the position of Myora Reef and Flinders Reef.

Prior to 1974, a total of 18 coral species were recorded living within Moreton Bay, with communities dominated by *Favia speciosa* and other brain corals (Lovell, 1975; 1989). Reefs at Peel Island, eastern Green Island, and Bird and Goat Islands had relatively high coral cover and diversity, while a unique monospecific stand of *Acropora digitifera* was recorded at Myora Reef on North Stradbroke Island (Figure 1). Torrential rainfall in the Brisbane region during January - February 1974 caused the largest flood this century (Cossins, 1990), which resulted in greatly reduced salinities causing catastrophic mortality in many coral communities (Lovell, 1975). Flood effects were mainly confined to the western region of the Bay and around Mud, St Helena and Green Islands which suffered total or massive coral mortality. No flood damage was observed around Myora Reef and the northern and eastern reefs at Peel Island (Lovell, 1975; 1989). In contrast, heavy coral mortality was recorded along the eastern and northern reefs at Peel Island following the major flood in the summer of 1956 (Slack-Smith, 1959). These studies indicate that different patterns of seawater dilution occurred during the 1956 and 1974 flood events, resulting in differing patterns of coral mortality in Moreton Bay.

Surprisingly few studies have been done on coral communities in Moreton Bay since the 1974 flood. Lovell (1989) re-surveyed one site at southwest Peel Island and found that coral cover had declined from 21% prior to the flood to 1.6% in 1981, however active coral recruitment had begun shortly after the flood. Harrison *et al*. (1991) and Harrison & Veron (1993) recorded a total of 40 scleractinian species in extensive surveys of the fringing reefs around Green Island, Wellington Point - Empire Point and Peel Island. This represents a substantial increase in coral species richness compared with the 18 species previously recorded by Lovell (1975, 1989). Recent studies have shown that many ecologically important coral species in Moreton Bay are sexually reproductive during summer (Harrison, 1993; P. Harrison, unpubl. data).

Flinders Reef is a small sandstone platform situated 5.5 km north of Moreton Island (Figure 1). The reef supports a luxuriant coral community with a dense coral cover (Veron, 1986). In contrast to coral assemblages within Moreton Bay, Flinders Reef contains a highly diverse coral community dominated by *Acropora* species (Lovell, 1989). Lovell (1989) recorded 112 coral species present, and more recent records have increased this number to 118 coral species (Veron, 1993). This represents a remarkably high number of coral species given its small reef area and its southerly location (Veron, 1986).

Despite the availability of recent species lists for coral communities in Moreton Bay, there is very little quantitative information on benthic community structure at most reef sites. The aims of this study were to quantify coral cover and benthic community structure at Flinders Reef and Myora Reef, and to assess the significance of the coral and other marine communities. The survey results provide a baseline against which future changes in these communities can be evaluated. The significance of these sites locally and regionally with respect to ecology, biogeography and human use are discussed.

Methods

Flinders Reef and Myora Reef were surveyed during two field trips in October and November 1994. Flinders Reef lies approximately 5.5 km northwest of Cape Moreton to the north of Moreton Bay (Figure 1). The Myora Reef area is located on the western side of North Stradbroke Island approximately 3 km north of Dunwich in the southeastern part of Moreton Bay (Figure 1).

At Flinders Reef (26° 58.8' S; 153° 29' E), three sites were surveyed along the western, southern and eastern reef area. Heavy swell conditions prevented surveys of the northern reef area. At each site, replicated video transects were done to quantify the cover of the major benthic categories present (Carleton & Done, 1995). Within each site, a series of 30 m fibreglass tape measures was laid out in an area where the benthic community was relatively homogeneous and representative of the reef habitat. At site 1 on the western reef area, a total of seven 30 m transects were surveyed between 6-10 m depth. Four 30 m transects were surveyed at both sites 2 and 3 at depths ranging from 5-9 m. The larger number of transects at site 1 was required because the benthic community was less homogeneous than at sites 2 and 3.

A Sony 805 Hi-8 video camera installed in an Amphibicam underwater housing was used for the surveys. A diver with the video camera swam along the transect line at a constant speed, approximately 30 cm above the substratum, with the camera pointed vertically downward recording the community in the vicinity of the transect line. Swimming speed was approximately 1 m per 6 seconds, so that a 30 m line was swum in approximately 3 minutes. In addition to the video transects, the survey sites were searched for any additional coral species present that had not previously been recorded (Lovell, 1989; Veron, 1986; 1993).

The Myora Reef coral community is located near 27° 28'19" S; 153° 24'38" E on the western side of North Stradbroke Island (Figure 1). Benthic community structure within the main coral area was quantified using five 50 m video transects orientated parallel to the reef edge. The video transect method was similar to that used at Flinders Reef, except that the longer transect length required approximately 5 minutes for each transect to be recorded. The transects were located in the upper reef area at 1-3 m below datum. In addition, a search was made for corals to determine the number of scleractinian species present at the Myora Reef site.

The Hi-8 video images of the transects were transferred to Super-VHS video format for analysis. Video footage was initially viewed by playing the tape at normal speed to determine the major benthic categories present along the transects. The footage was then analysed by playing the tape for each transect at approximately one-sixth normal speed on a S-VHS player and a JVC high resolution large screen monitor. The benthic category underlying a fixed point on the screen was identified and recorded, with samples taken at approximately regular intervals throughout the transect line. Harriott *et al*. (1995) found that approximately 200 sample points per 30 m transect line and 300 sample points per 50 m transect line were the optimal sample sizes for analysis of community structure from video transects of sub-tropical benthic communities at Lord Howe Island. Therefore, at least 200 or 300 sample points were recorded for each 30 m or 50 m transect line, respectively.

The percentage cover of each benthic category was calculated for each transect, and the mean and standard deviation of percentage cover were calculated from the replicate transects at each site. Most scleractinian corals were identified to generic level, while some corals with obvious features were identified to species level (e.g. *Pocillopora* spp., *Acropora palifera, Turbinaria* spp.), and others were identified to family level or major lifeform categories. Non-scleractinians were grouped into higher taxonomic categories including sponge, soft coral, anemone, mollusc, echinoderm and ascidian. Various genera and lifeform categories of algae were recorded, as well as four categories of substratum.

Results

Flinders Reef surveys

Flinders Reef is a small sandstone platform reef. The upper reef area is approximately 200 m long and 50 m wide and is exposed from mid-tide. The reef top contains a shallow lagoon dominated by algae and oyster beds. The main subtidal reef area extends down to approximately 10-15 m depth and supports a flourishing coral community and other reef-associated biota.

Site 1 was located on the western (leeward) side of the reef. The greatest coral cover occurred between 4-12 m depth. Below 12 m, coral cover became sparse with increased areas of sand and rubble. The reef topography was generally flat with a gradual slope extending up toward the intertidal reef flat. The reef community consisted of a mixed assemblage of scleractinian corals and soft corals, with a low cover of macro-algae. Scleractinian corals dominated the community, with a mean percentage cover of 30.6% (Table 1). However, coral cover was quite variable, with values ranging from 11.7% in transect 3 up to 76.7% in the southernmost transect which extended into a large patch of staghorn coral.

A wide variety of scleractinian corals was present including *Acropora palifera, Pocillopora damicornis, Turbinaria frondens* and various encrusting corals. *Goniopora, Montipora*, *Leptoria* and *Seriatopora* colonies were recorded in transects only at this site (Table 1). A significant amount of anchor damage was observed in this area, with a number of fragmented and scarred coral colonies scattered over the reef slope.

Soft corals had a mean cover of 17.3% (Table 1) and had an inverse relationship with hard coral cover. Colonial anemones were also abundant and had a mean cover of 12.3%. A range of other invertebrate groups was present including sponges, molluscs, echinoderms and ascidians which tended to occur sporadically and contributed less than 2% cover. Algal categories had a combined mean cover of 27.7%, comprised mostly of turf algae growing on rock (Table 1). Brown and red macroalgae were present but generally contributed less than 5% cover. An average of 10.7% of the reef area was bare substrata, which consisted mainly of sand and rubble, and dead coral which had a mean cover of 1.8% (Table 1).

Site 2 was located in a small embayment on the southern side of the reef, at 8-9 m depth. The reef consisted of a gently sloping platform approximately 30 m from the reef top. The reef community was more uniform than at site 1 and was dominated by scleractinian corals, various algae, and soft corals.

The percentage cover of scleractinian corals was generally higher than at site 1, with a mean cover of 43.1% (Table 1). The coral community was dominated by corals with an encrusting or massive growth form, and there was a substantially increased cover of *Acropora palifera* compared with site 1. Fragile branching and plate corals were less abundant than at site 1 (Table 1), reflecting the more exposed aspect of this site. The occurrence of *Pocillopora verrucosa* at this site was of particular interest, as this coral species had not previously been recorded at Flinders Reef.

Soft corals were generally less abundant than at site 1, with a mean cover of 10.7% (Table 1). Other invertebrate groups contributed only 2.9% mean cover, and very little bare substrata was present in the transects. The combined algal categories had a mean cover of 42.6%, representing a substantially higher cover compared with site 1. Fine turf algae and red algae dominated these assemblages, with a variety of other algae present including *Chlorodesmis* turtle grass (Table 1).

Site 3 was located on the eastern (windward) side of the reef, between 5-9 m depth. The reef topography was more complex than at sites 1 and 2, consisting of numerous gutters up to 12 m deep and various pinnacles and bommies, some of which rose to within 3 m of the surface. The reef community was more heterogeneous than at site 2 and was dominated by scleractinian corals and algae, with a moderate cover of soft corals and other invertebrates.

Scleractinian corals were growing throughout the ridges and gutters, and had a mean percentage cover of 34% (Table 1), with a range from 28-40% cover among the four transects. Most corals exhibited a low, thickened or encrusting growth form, reflecting the relatively exposed nature of this area of Flinders Reef. The coral community was dominated by *Acropora palifera* which had a mean cover of 17.3%, and contributed between 7.6% and 26.2% cover in the four transects. Encrusting corals, *Goniastrea, Pocillopora damicornis* and digitate or corymbose *Acropora* colonies were also prevalent. Fewer categories of hard corals were present at this site than at sites 1 and 2.

Soft corals had a mean cover of 12%, while the cnidarian zoanthids and colonial anemones had a combined mean cover of 12.4% (Table 1). Algae had a mean cover value of 35.2%, with fine turf algae (15% mean cover) and red algae (9.3% mean cover) the dominant algal categories (Table 1).

Myora Reef surveys

The Myora Reef area encompasses a variety of reef habitats. A wide reef flat extends approximately 350 m from the mangrove habitat in the upper intertidal area on the shoreline of North Stradbroke Island, to the Rainbow Channel. Much of the reef flat is exposed at low tide and consists of various mixtures of mud, sand and rubble substrata with patches of algae.

The main Myora coral community begins approximately 250 m to the east of the Myora Light and extends for about 200 m to within 50 m of the Port Hand Lateral Mark. The coral

Table 1. Summary of transect data from Flinders Reef and Myora Reef. Values indicate mean percentage cover for each category at a site.

	Flinders Reef						Myora Reef	
	Site 1 (%)		Site 2 (%)		Site 3 (%)		(%)	
Category	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Acropora branching	16.5	27.41	0.9	0.72				
Acropora digitate	0.7	1.14	0.2	0.24	1.2	1.06	42.3	5.15
Acropora palifera	2.8	2.89	12.9	3.87	17.3	7.71		
Acropora plating	1.4	2.71	3.3	2.77	0.1	0.25		
Acropora corymbose	0.9	1.32	1.7	1.43	2.1	0.45		
Acanthastrea	0.2	0.44	0.7	0.83	0.6	0.44		
unspecified coral	0.1	0.00						
Encrusting coral	2.1	1.00	11.3	1.61	5.7	2.84		
Faviid			0.2	0.23	0.7	0.67	0.1	0.13
Goniastrea	0.4	0.46	5.2	1.58	2.5	1.59		
Goniopora	0.1	0.28						
Hydnophora	0.1	0.15	1.1	0.90				
Leptoria phrygia	0.1	0.00						
Montipora	0.4	0.75						
Pavona			0.4	0.55	0.4	0.51		
Pocilloporid			0.4	0.83				
Pocillopora damicornis	2.0	1.11	3.0	0.62	3.0	1.74		
P. verrucosa			0.1	0.20				
Porites	0.1	0.32	1.0	1.94	0.4	0.61		
Seriatopora	0.1	0.16						
Turbinaria frondens	2.0	1.34	0.5	1.01				
T. patula	0.8	1.71	0.1	0.22				
Algae	6.8	2.79	2.8	2.32	8.9	3.51	1.2	1.20
Brown macro-algae	1.2	1.60					14.3	3.42
Codium							<0.1	0.10
Halimeda					0.4	0.82		
Lobophora	0.3	0.55						
Red algae	3.3	3.83	16.0	6.84	9.3	8.97	<0.1	0.10
Turtle grass			0.2	0.40	0.7	0.45	0.1	0.27
Turf algae	15.2	5.41	21.9	5.20	15.0	5.29		
Ulva	0.8	0.90	1.3	1.60	0.5	0.58		
Dead coral	1.8	1.65	0.2	0.24			6.9	3.37
Bare rock	0.3	0.40						
Rubble	2.0	1.65					14.5	9.40
Sand	6.6	5.57	0.6	0.84	1.9	2.69	6.7	1.83
Colonial anemone	12.2	8.53			5.4	6.36		
Ascidian	0.1	0.16			0.1	0.25	0.1	0.23
Crinoid	0.7	0.48	1.0	0.42	0.1	0.25		
Echinoderm	0.2	0.21	0.2	0.22				
Encrusting sponge	0.2	0.47	0.9	1.36	4.1	2.21		
Hydroid							2.0	

community extends across the outer margin of the subtidal reef flat to approximately 2-3 m below datum, covering an area approximately 30-50 m in width.

The main coral community is dominated by extensive colonies of *Acropora digitifera*, with a few smaller colonies of *Acropora glauca, Favia speciosa, Cyphastrea serailia, Goniastrea australensis, Acanthastrea hillae, Goniopora* sp., *Psammocora superficialis* and *Turbinaria peltata* also present. Additional coral species observed on the reef flat in the vicinity of the transect site include *Barabattoia amicorum, Acanthastrea echinata, Acanthastrea lordhowensis*, and the skeleton of a small *Pocillopora damicornis* colony.

The reef areas adjacent to the main coral community are characterised by mixtures of sand, silt and shell rubble with patches of sparse seagrass and algae, infaunal burrows, and sporadic coral colonies. Offshore from the main coral area, the coral colonies are smaller and more isolated and the deeper reef slope grades into bare silty substratum in the Rainbow Channel where coral colonies were not observed. The video transects were recorded in the zone of maximum coral cover, from 0.5-3 m below datum.

Results from the video transects show that scleractinian corals dominate the community, with a mean cover of 42.4% (Table 1). Coral cover among the five transects was fairly consistent, ranging from 37.8% to 50.7%. *Acropora digitifera* and other digitate *Acropora* corals had a combined mean cover of 42.3%. The *A. digitifera* population consisted of numerous small colonies interspersed among very large colonies extending over several metres. Although a range of other coral species were present, only a few faviid brain coral colonies were recorded in the transects (Table 1).

Soft corals had a mean cover of 9.2%, while sponges, erect branching hydroid colonies, and ascidians had a combined mean cover of 4.8% (Table 1). Algal categories had a combined mean cover of 15.7%, with brown algae dominant. Bare substratum occupied 28.1% of the transects, with dead coral having a mean cover of 6.9% (Table 1).

Discussion

Status and significance of the Flinders Reef communities

Flinders Reef supports a flourishing reef community dominated by reef corals, soft corals and various other reef-associated biota. Coral cover is relatively high, and falls within the range of coral cover recorded in transect surveys on tropical fringing reefs from the Great Barrier Reef (e.g. Pichon & Morrissey, 1981; Bull, 1982; Bradbury *et al*., 1987a; b). The mean coral cover of 30-43% at Flinders Reef also compares favourably with results from broadscale surveys of hundreds of reefs throughout the Great Barrier Reef region where the estimated average live coral cover for the whole GBR varied from 22.5% to 31.3% between 1985 and 1992 (Moran *et al*., 1993).

The survey results show that the coral community at Flinders Reef is not uniform, but varies significantly around the reef, particularly in relation to exposure to wave action. The southern and eastern reef areas exposed to southeasterly swells are dominated by robust corals which tend to have an encrusting or massive growth form, while the more protected western reef area contains a higher cover of branching and staghorn corals which are much more susceptible to both wave and anchor damage.

One of the most remarkable features of Flinders Reef is its extremely high species richness of scleractinian corals. Previous extensive surveys by Lovell (1975; 1989) and Veron (1986; 1993) recorded 118 scleractinian species from Flinders Reef. In the present study *Pocillopora* *verrucosa* was found at the southern site, bringing the total number of coral species recorded at Flinders Reef to 119. This finding also greatly extends the biogeographic records for *P. verrucosa,* as it has not previously been recorded south of the southern Great Barrier Reef (Veron, 1993). The total of 119 coral species is extraordinarily high given the small extent (10 hectare) of Flinders Reef, its southerly location, and its relative isolation from other large reef systems likely to provide coral larval recruits.

Recent coral surveys have recorded over 40 coral species living in Moreton Bay (Harrison *et al*., 1991; Harrison & Veron, 1993; Harrison, 1996). Thus the ten hectares of Flinders Reef contain three times as many species as the whole of Moreton Bay, and substantially more species than at the Gneering Shoals, about 50 km to the north, where 77 coral species have been recorded (Harrison *et al*., 1993; Banks & Harriott, 1995). Further south, a total of 90 scleractinian species has been recorded at the Solitary Islands (Veron *et al*., 1974; Harriott *et al*., 1994), and 83 scleractinian species have been recorded at Lord Howe Island (Veron & Done, 1979; Harriott *et al*., 1995; Harrison *et al*., 1995a). Veron (1993) recorded a total of 118 coral species on Elizabeth and Middleton Reefs, each of which are approximately 2 000 ha in extent (Veron, 1986). Thus Flinders Reef has the highest number of coral species recorded on any reef system along the east coast of Australia south of the Great Barrier Reef.

Flinders Reef is not only significant for its extraordinary species richness and high coral cover, but also has regional and national biogeographic significance as it contains a unique association of tropical and sub-tropical coral species. Flinders Reef represents the southernmost recorded site along the east Australian coast for *Pocillopora verrucosa* and 35 other coral species, and many other species are near their southern latitudinal limits (Veron, 1993). Furthermore, unlike coral communities from the Great Barrier Reef which are totally dominated by tropical scleractinian species, Flinders Reef also contains a range of sub-tropical species which are rare or absent from tropical reefs. The primarily sub-tropical suite of species recorded includes: *Acropora glauca, A. palmerae* and *A. solitaryensis, Astreopora moretonensis, Acanthastrea bowerbanki, A. hillae* and *A. lordhowensis, Scolymia australis, Goniastrea australensis, Favia speciosa, Turbinaria patula, T. bifrons* and *T. radicalis* (Veron, 1986; 1993). Hence Flinders Reef represents a biogeographic overlap area, with a mixture of tropical and sub-tropical species co-existing on the one small reef system. Multi Dimensional Scaling (MDS) and cluster analyses show that the Flinders Reef sites are regionally distinct from most other reef communities along the east coast of Australia (Harrison *et al*., 1995b).

The extensive coral community at Flinders Reef is likely to play a critically important role in providing coral larval recruits to reefs within Moreton Bay. Larval dispersal, settlement and recruitment processes are essential for maintenance of coral communities (reviewed by Harrison & Wallace, 1990), and it is likely that Flinders Reef is a major source of coral larvae for re-seeding reefs in Moreton Bay following devastation by catastrophic events such as the floods that occurred in 1956 and 1974 (Slack-Smith, 1959; Lovell, 1989). In addition, Flinders Reef is also of great potential significance as a source of larvae for dispersal to southern reefs including Julian Rocks Aquatic Reserve and the Solitary Islands Marine Reserve in northern NSW (Harriott & Banks, 1995).

Status and significance of the Myora Reef communities

Myora Reef contains a very important coral community which provides habitats for a high diversity of other reef fauna (Harrison *et al*., 1995b). The relatively high coral cover (41%) is similar to that at Flinders Reef and compares well with tropical coral reef systems in the Great

Barrier Reef (e.g. Bull, 1982; Moran *et al*., 1993). The only other reef areas within Moreton Bay with relatively high coral cover are the unique faviid dominated coral communities along Lazaret Gutter and the subtidal reef patches on the northwestern area of Peel Island (Lovell, 1989; Harrison *et al*., 1991; P. Harrison & N. Holmes, unpubl. data).

Although *Acropora* species generally characterise Indo-Pacific reefs and used to dominate coral assemblages in Moreton Bay during previous periods of higher sea levels and active reef growth (Lovell, 1989), Myora Reef contains the only major stand of living *Acropora* present within Moreton Bay. Small isolated colonies of various *Acropora* species occur around Green Island and Peel Island but they are generally less than 0.5 m in diameter (Harrison *et al*., 1991; P. Harrison & N. Holmes, unpubl. data). The only other subtropical reefs which have extensive stands of *Acropora* are some areas of Flinders Reef, and Elizabeth and Middleton Reefs and Lord Howe Island, which lie more than 600 km to the southeast of Moreton Bay (Veron & Done, 1979; Veron, 1986; 1993; Harriott *et al*., 1995; Harrison *et al*., 1995a). However, different species of *Acropora* dominate these other assemblages. The Myora Reef *Acropora* assemblage is thus of regional significance, and MDS and cluster analyses show that Myora Reef is distinct from other sub-tropical reef communities along the east coast of Australia (Harrison *et al*., 1995b).

The *Acropora* assemblage at Myora is remarkable because *Acropora* species are typically highly sensitive to disturbance and flooding effects, which probably explains their rarity or absence from most other reefs in Moreton Bay. The large expanse and size of *Acropora* colonies at Myora indicates that these corals have been growing there for a long period. *Acropora digitifera* colonies were recorded at Myora by Lovell prior to 1974 and clearly survived the catastrophic floods of 1974, which destroyed many of the coral communities throughout the Bay (Lovell, 1975, 1989). Stephenson (1968) suggested that the area to the north of Peel Island which encompasses Myora Reef may be relatively insulated from flood dilution by the input and pooling of oceanic water from the South Passage between Moreton and North Stradbroke Islands. Maintenance of high salinity conditions during flood events probably accounts for the high biodiversity of the Myora Reef benthic community. Lovell (1975, 1989) recorded six other scleractinian species from Myora Reef, and this survey recorded an additional nine species present. A total of 16 coral species have now been recorded from Myora Reef, which represents 40% of the species living in Moreton Bay. One small recently-killed colony of *Pocillopora damicornis* was also present, and represents the first record of this species in the recent coral assemblages within Moreton Bay.

The large size of the table coral assemblage at Myora may help to stabilise the sediment and buffer the community from the effects of strong currents sweeping along the adjacent Rainbow Channel. The relative stability of the coral habitat has probably allowed the remarkably high diversity of many other faunal groups to become established at Myora (Davie & Hooper, 1992; Davie, 1992; Harrison *et al*., 1995b). The Myora Reef may also play an important role in providing larvae that settle in other areas of Moreton Bay. During the present survey *Acropora digitifera* coral colonies were found to be gravid, and the large size of the eggs indicates that they probably spawned in November or December 1994. Previous work has shown that a number of brain coral species in Moreton Bay contain ripe eggs and sperm after full moon periods in December and January, and *Cyphastrea serailia* colonies have been observed spawning 5-9 nights after full moon periods in December 1992 and January 1994 (Harrison, 1993; P. Harrison, unpubl. data). These observations indicate that a range of coral species in Moreton Bay are reproductively viable, and are likely to provide an important source of larval recruits for maintaining coral populations in this region.

Conservation status

On the evidence presented here, Flinders Reef is unequivocally of major biological, ecological and biogeographic significance, from local, regional and national perspectives. Given the extraordinary diversity of reef corals and other reef biota recorded at Flinders Reef it is, by any criteria, sufficiently important to be protected from adverse impacts of human origin. The reef is currently heavily utilised by recreational divers and fishers, and aquarium fish are collected commercially and recreationally. During these surveys, there were clear signs of anchor damage on the leeward side of Flinders Reef which is dominated by fragile branching corals. Declaration as a Marine National Park within the Moreton Bay Marine Park is the most appropriate and effective approach to protection of the present unique characteristics of Flinders Reef. This would enforce some management of the site and restrict potentially damaging activities. Installation of mooring buoys for boats, and restrictions on collecting activities should be given high priority.

The coral community at Myora Reef contains an unusual range of species and is regionally distinct. The associated fauna are also extremely varied and this small patch of reef appears to form a stable, permanent habitat. In view of its highly unusual characteristics and of its likely role in maintaining other communities in Moreton Bay, Myora Reef should be zoned as Marine National Park to ensure that the area's unique communities are adequately protected.

Acknowledgements

We thank Kris Kristensen for supporting this study, the Queensland Department of Environment and Heritage for funding, Gary Selwood and other boat crew members for field assistance, and Darren Richardson and Joanne Wilson for help with surveys. We also thank John Hooper, Peter Davie, Darryl Potter, Steve Van Dyck, Kevin Lamprell and Thora Whitehead for contributing unpublished information.

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The Coral-eating *Drupella* (Mollusca: Gastropoda) from Moreton Bay (Southeast Queensland)

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Abstract

Coral-eating gastropods of the genus *Drupella* (Thiele, 1925) have caused significant coral damage in Japan, the Philippines, the Marshall Islands and Australia. In some areas the damage has been of the same intensity as that caused by the crown-of-thorns starfish, *Acanthaster planci* (L.). *Drupella* is present along the Great Barrier Reef (GBR) and in Moreton Bay. A survey conducted at Myora Reef and Flat Rock (North Stadbroke Island) found three species of *Drupella*: *D. rugosa* at Myora, *D. fragum* at Flat Rock and *D. cornus* at both locations. Only a few individual of *D. cornus* were found and *D. fragum* were more abundant than *D. rugosa*. Because of the relatively high density of *D. fragum*, periodic surveys of their populations are advised.

Introduction

Drupella is widespread throughout the Indo-Pacific. Members of this genus have caused significant coral damage during outbreaks in the Philippines, the Marshall Islands, Japan and Western Australia (Ningaloo Reef). Species involved in the outbreaks were *D. rugosa* (Born, 1778), *D. cornus* (Roeding, 1798) and *D. fragum* (Blainville, 1832) with densities ranging from 175-3 000 *Drupella*/m² (Moyer *et al*., 1982; Boucher, 1986; Ayling & Ayling, 1987) killing up to 70% of total coral cover (Stoddart, 1989).

Four species of *Drupella* are present on the Great Barrier Reef (GBR): *D. cornus*, *D. fragum*, *D. rugosa* and *D. eburnea* (Kuester, 1862) (Ayling & Ayling, 1987; Page, 1987; Oxley, 1988; Cumming, 1992; Fellegara, 1996). *Drupella* spp. densities on the GBR range between 0.3-2.4 individuals/m² causing coral damage of between 0.7-49% (Ayling & Ayling, 1987; Page, 1987; Oxley, 1988; Cumming, 1992; Fellegara, 1996).

Drupella feeds mainly on fast growing corals such as *Acropora* and *Montipora* species and, to a lesser extent, poritids and pocilloporids (Turner, 1994), but a recent study conducted at Heron Island (southern GBR) has shown that *D. cornus* is a trophic generalist, feeding both on *Acropora* spp. and *Montipora* spp., whereas *D. rugosa*, *D. fragum* and *D. eburnea* feed mainly on a few species of *Acropora* (Fellegara, 1996).

Three species occur in Moreton Bay – *D. fragum* (Page, 1987), *D. cornus* (Pickett *et al*., 1984) and *D. rugosa* (J. Healy, pers. comm.). *Drupella cornus* has been found in late Pleistocene deposits. Because the coral communities of Moreton Bay contribute to its high biodiversity, and as *Drupella* is a potential threat to hard corals, the purpose of this study was to establish the distribution and abundance of *Drupella* spp. in Moreton Bay, their food preferences, and the extent to which they damage corals.

Materials and Methods

Surveys of *Drupella* were conducted at three sites within Moreton Bay (Myora Reef, Peel Island and Green Island) and one site (Flat Rock) outside the Bay. Myora Reef is situated inside Moreton Bay, close to Dunwich (North Stradbroke Island), and has a relatively high coral cover dominated by *Acropora digitifera* colonies growing on sand from 1-3 m depth. Peel and Green Islands are situated south-west from Myora Reef, and are characterised by

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 537-540.

massive coral communities at 1-3 m depth. Flat Rock lies north-east of Point Lookout (North Stradbroke Island) and has sparse digitate and massive coral colonies growing on a rocky substratum, in 8-24 m depth.

In December 1995, I carried out two 20 x 1 m belt transects on SCUBA at Myora; and an hour-long random snorkel at Peel and Green Islands. In September 1996, I conducted a random search whilst on SCUBA at Flat Rock. For each transect, I recorded the number of individuals of *Drupella*, identified and measured the coral colony upon which they were feeding and the area of their feeding scars with a tape measure and a plastic ruler (± 0.5 cm). The area of the coral colonies and feeding scars were always measured in two dimensions.

Results

At Myora Reef, 107 adult *D. rugosa* were found (Table 1). Outside the transects, two groups of *D. rugosa* with 6 and 8 individuals were observed. In each group half of the individuals were recruits (< 10 mm shell length) and juveniles (10-20 mm shell length) and half were adults (> 20 mm). The only individual of *D. cornus* found was outside the transects. All *Drupella* were found on *A. digitifera*. Coral cover was of 35% and 95% along the two transects.

At Flat Rock, five *Pocillopora damicornis* colonies and 22 *Acropora* spp. colonies were searched. Totals of 91 juvenile and 21 adult *D. fragum* were found on two *P. damicornis*, three *Acropora glauca*, one *A. cerealis* and two *Acropora* spp. (Table 1). Two adult *D. cornus* were observed feeding on *Montipora* sp. *Drupella* was not found at either Green or Peel Islands.

Table 1. *Drupella* spp. densities at Myora Reef and Flat Rock recorded surveying 20 x 1 m belt-transects. Coral damage was calculated on the size of the feeding scar with respect to the size of the coral colony and expressed in percentage.

	Site	Density/m² of transect ± SE	Density/m² of living coral ± SE	Coral damage (in % ± SE)
D. rugosa (adult)	Myora Reef	0.33 ± 0.09	79 ± 24	0.77 ± 0.25
D. cornus (adult)	Myora Reef	one individual	–	–
D. cornus (adult)	Flat Rock	two individuals	–	–
D. fragum (adult) and	Flat Rock	–	120 ± 54	43 ± 14 (both adults
D. fragum (juvenile)	Flat Rock	–	202 ± 67	juveniles)

Discussion

Three species of *Drupella* were found during this study: *D. cornus* (one specimen) and *D. rugosa* at Myora Reef, and *D. fragum* and *D. cornus* (two specimens) at Flat Rock. *Drupella rugosa* density, and the extent to which it had caused damage to corals, was slightly below the lowest figures recorded on the GBR (Ayling & Ayling, 1987; Page, 1987; Oxley, 1988, Cumming, 1992; Fellegara, 1996). *Drupella rugosa* was found exclusively on *Acropora digitifera*, but this is not surprising as *A. digitifera* is almost the only coral species present at Myora Reef.

Despite the different survey methodology, at each site the number of *Drupella* per m² of living coral was estimated in the same way and these data are comparable. *Drupella fragum* was considerably more abundant than *D. rugosa* and its percentage coral damage much higher. *Drupella fragum* density was similar to the densities recorded during the outbreaks at Ningaloo Reef (175 snails/m²) (Ayling & Ayling, 1987) and the Marshall Islands (300-500 snails/m² (Boucher, 1986); however, percentage coral damage did not exceed the maximum percentage recorded on the GBR (Ayling & Ayling, 1992; Cumming, 1992; Fellegara, 1996). As found in previous studies, *D. fragum* fed mainly on *Acropora* and secondarily on *Pocillopora*.

This study found a greater density of *D. fragum* than was recorded by Page (1987) at Flinders Reef (off Moreton Island), where he found between 6-15 individuals in a 30 minute dive.

Drupella spp. can coexist (Moyer, 1986; Cumming, 1992; Fellegara, 1996) but these results suggest that *D. rugosa* and *D. fragum* have marked ecological preferences. *Drupella rugosa* was previously found on Heron Reef flat, whereas *D. fragum* occur only on Heron Reef slope (Fellegara, 1996). Heron Reef flat and Myora Reef are similar in that each is shallow and is subject to relatively wide temperature ranges in comparison to the water temperature ranges found outside Moreton Bay and in the deepest water off Heron Reef flat (Flood, 1993; Mather & Bennett, 1993). Possibly, *D. rugosa* is more tolerant of water temperature fluctuations.

As expected, *Drupella* was absent from sites dominated by massive corals, Peel and Green Islands. These gastropods have seldom been recorded feeding on massive and large-polyped corals. However, factors other than food preferences (e.g. predation or abiotic factors) could cause their absence from these sites.

Drupella outbreaks have come to attention only very recently and nothing is known about their duration and recurrence. At Ningaloo Reef, the *Drupella* population has been increasing since the mid 1980s reaching a level which might be described as an outbreak at the beginning of the 1990s. The causes for *Drupella* outbreaks have not yet been established. Turner (1994) reviewed two models that have been proposed: first, that outbreaks are natural phenomena; and second, that anthropogenic perturbations cause outbreaks.

Natural population fluctuations occur in many marine invertebrates due to high recruitment survival (Giese & Pearse, 1977). *Drupella* outbreaks have only been reported from large reef areas, but because of their patchy distribution, even small reefs could experience an outbreak.

Anthropogenic perturbations include eutrophication and siltation from increased terrestrial run off that could result in an increased survival of *Drupella* larvae and, therefore, higher recruitment. Moreton Bay is subject to periodic flooding and this situation occurs, but, as McClanahan (1994) points out, possibly multiple factors interact to favour a population increase.

Over-fishing, and removal of *Drupella* predators, could also result in an increased population. McClanahan (1994) surveyed *D. cornus* in unfished and fished reefs. He found a negative correlation between the number of *D. cornus* and the abundance of predators such as balistids and labrids. However he stresses that further experimental studies are needed to confirm these results. These two fish families are present in Moreton Bay but there are no quantitative data on their abundance.

Outbreaks on small reefs, like those present in Moreton Bay, could also be easier to control under such situations. Fisk *et al.* (1996) report the success of manual removal of *A. planci* from small reefs. Because of the small size of *Drupella* and because of their aggregated distribution during outbreaks, the same technique could be applied in a *Drupella* outbreak to protect small reefs of particular scientific or economic importance.

In conclusion, the density and predatory activity of *D. rugosa* and *D. fragum* are within the known limits of a normal non-outbreaking population. However, because of the subtlety of *Drupella* outbreaks, periodic monitoring should be undertaken.

Acknowledgements

I wish to thank Peter Johnson and Dave Harris for their help in the field. In particular, I would like to thank Dr Ian Tibbetts for his encouragement.

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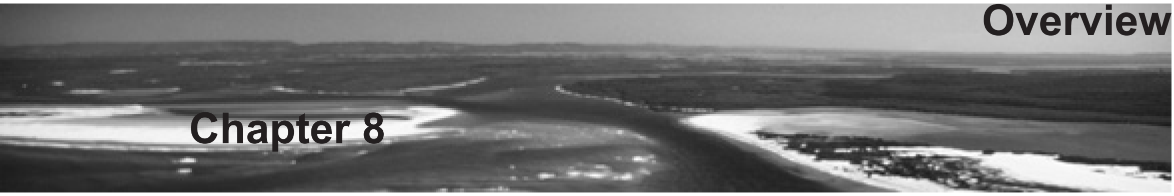

Overview

Chapter 8

Flood Effects

Brisbane and its catchment have arguably the best-documented history of flood events within Australia (Heatherwick, 1986; Bush, 1930; Australian Bureau of Meteorology, 1974). Earliest descriptive records documenting flood height in the Brisbane River date to the flood of 1825 (Henderson, 1896), although Lt John Oxley R.N., the Surveyor-General of New South Wales, commented on the marks of previous river floods (exceeding 15.2 m above the junction of the Bremer River) during his exploration of the river in 1824 (cited in McLeod, 1990). Records of flood levels within the Brisbane River have been kept since the flood of 1841.

Prior to 1900 and the intensive urban and residential development of land adjacent to the Brisbane River, flood events were a fairly frequent feature within the catchment, occurring at approximately one to eight year intervals (Figure 1). Especially severe floods occurred in 1841 and 1893. The 1893 flood, the severest in Brisbane's history, was actually comprised of three separate flood events which occurred during a 16-day period in February. During this period over 100 cm of rainfall was recorded within one month, with over 46 cm in a single day (Table 1). Sufficient suspended solids were delivered to the lower Brisbane River during these three combined events to reduce water depth at the dredged channel at the river mouth from 6.06 m to 2.61 m (McLeod, 1990).

Although minor floods have occurred in 1906 and 1931, during the 20th century the most severe flood to date within the Moreton Bay catchment was the flood of January 1974 (Heatherwick, 1986). Heavy rainfalls resulting from a monsoonal trough associated with a decaying cyclone resulted in extensive flooding within the Brisbane River (Heatherwick, 1986). Over 65 cm of rainfall was recorded within five days in metropolitan Brisbane with greater than 31 cm of rainfall in a single day, the second highest 24-hour rainfall total in Brisbane's recorded history. Damage from this event is estimated at $200 million (1974 dollars) with over 7 000 households directly affected (Heatherwick, 1986).

In May 1996, Moreton Bay and its catchment experienced a one-in-twenty year flood which resulted in significant changes in the physical dynamics, chemistry and biology of Moreton Bay and its principal rivers. As this flood event was followed by a period of several months with no significant further rainfall, the event provided a unique opportunity to assess the physical, chemical and biological impacts of a single large freshwater pulse on Moreton Bay and its catchment.

Given the major role that freshwater inflow and associated sediments have had on the system it is not surprising that flood events are of particular significance and an understanding of their effects of particular importance. The papers in this chapter describe the results from a flurry of research that followed the 1996 one-in-twenty year flood. They provide an important insight into the scale and distribution of flood effects and the need for incorporation of these considerations into management principles and practices.

Davies and Eyre present data and a generalised three step model for the changes to nutrient and suspended sediment load of the Brisbane River estuary following this flood. They reveal that sediments and incorporated nutrients lain down during non-flood years are “cleansed” from the system. Naturally, the converse of this statement is that the Bay is periodically “dirtied” by estuarine basin deposits that are reworked by floods. They report that the recovery of the estuary following a flood is relatively rapid and its nutrient loads soon become once more dominated by lateral non-point sources such as sewage treatment plants.

Andrew Moss’s paper relates changes that occurred to the Bay, his sampling stations beginning where those of Davies and Eyre left off, i.e. at the mouth of the Brisbane River. He notes that of all the factors that act to decrease both the intensity and duration of flood effects the most influential is strong tidal mixing. Nutrient loads increased rapidly during the flood and the total nutrients delivered to the Bay during the period of flood influence equated to the total annual nutrient load from point sources for a normal year, however, the expected phytoplankton bloom did not occur. Andrew is quick to warn those who may consider his datum indicative of a Bay that would suffer no ill effects from an even greater anthropogenic nutrient load, and suggests that the lack of any serious phytoplankton bloom associated with the nutrient pulse should not be taken to mean that the Bay can accommodate large increases in nutrient loads.

The clear message from these summaries of the fate of nutrients and sediments during a major flood event is that what goes in to the river will end up in the Bay – each dribble of sump oil, each patch of bare soil will eventually have an impact on the Bay. Sediment-locked nutrients and toxicants are likely to remain within the system, not flushed from it to the open sea as some may wish or expect.

The two papers that follow examine the impact of the 1996 flood on plant communities. Cynthia Heil and coauthors describe progressive changes in phytoplankton abundance and community structure that followed the flood. They demonstrate that floods can have Baywide effects, reporting an east-west gradient in abundance and a north-south gradient in community composition. Moreover, besides being illustrative of the impact of floods, the phytoplankton community plays an important role during floods, acting like a nutrient sponge, as it incorporates between 50% and 100% of the nitrogen entering the system.

Mark O’Donohue and his colleagues use a range of techniques on various marine plant groups in their study of flood effects. They not only provide a comprehensive picture of Moreton Bay as a habitat for plants during interflood and flood periods, but they also offer a fascinating set of new biological tools for the assessment of water quality (see also Udy & Dennison, Chapter 4, this volume). Of the flood signatures revealed by plants, perhaps the most startling is the reduction by 40% in depth distribution of a seagrass. Fortunately, most of the changes to plant communities following floods, including the reduction in depth distribution of the seagrass, are short-lived. Marine plant bioindicators allow a direct assessment of the impact of both natural and anthropogenic influences on the marine system, making them a research and monitoring tool of great importance.

The final paper in this chapter examines the impact of periods of reduced salinity, which is one of the features of flood events, on two species of coral that occur in Moreton Bay. Johnson and Neil use a translocation experiment, the results of which may provide a partial explanation of the differential impact of floods on corals, and thus the extent to which coral distribution in Moreton Bay is influenced by floods.

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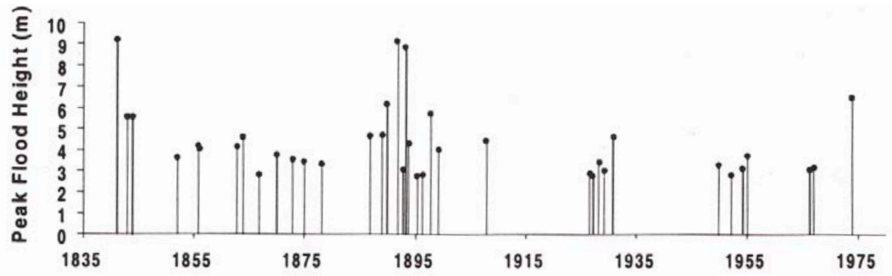

Figure 1. Historical record of the occurrence and peak height of floods at the Brisbane Port Office from 1841 to 1974. Figure was adapted from Australian Bureau of Meteorology Report (1974).

Table 1. Highest monthly rainfall at Brisbane between January 1840 and May 1996. Data taken from Bureau of Meteorology Report (1974).

Rank	Date	Rainfall Amount (mm)	Number of Rain Days in Month
1	Feb 1893	1,026	25
2	Jan 1974	872	26
3	March 1870	865	26
4	Jan 1895	704	22
5	Feb 1975	691	24
6	May 1996	661	7
7	June 1967	647	18
8	Jan 1887	593	17
9	Jan 1927	570	20
10	Feb 1954	548	21

Cynthia Heil, Andrew Moss and Ian Tibbetts

Nutrient and Suspended Sediment Input to Moreton Bay – The Role of Episodic Events and Estuarine Processes

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Abstract

Phosphorus, nitrogen, suspended sediment concentrations and salinity were examined in the Brisbane River estuary during a 20 year return-interval flood and during four sampling runs after the flood event. During the flood peak the estuarine basin flushed almost fresh to the mouth and approximately 150 000 t/day of suspended sediments, 44 t/day of phosphorus and 288 t/day of nitrogen were discharged directly into Moreton Bay. Some of this material was scoured from the bottom of the estuarine basin highlighting the importance of including estuarine processes when calculating terrestrial fluxes to Moreton Bay. This flushing event also had a "cleansing effect" on the estuarine basin, flushing the nutrient loaded pre-flood estuarine water into Moreton Bay. The estuary recovered rapidly by way of a salt-wedge penetrating landward along the channel bottom, and progressed from a small highly stratified, through a moderately stratified, to a large vertically homogeneous system. As river flow decreased and the estuary recovered, dissolved nutrient concentrations increased rapidly as the estuary once again became dominated by the lateral point-source input (e.g. sewage treatment plants).

Introduction

The hydrography, geochemistry and primary productivity of many coastal embayments is ultimately linked to freshwater, sediment and nutrient input from their catchments. In subtropical systems such as Moreton Bay much of this terrestrial loading occurs during episodic high-energy rainfall events (Douglas, 1969; Hart *et al.*, 1988; Davies, 1996; Eyre & Twigg, 1997). Estuaries may significantly modify this input as they flow towards the sea through a combination of physical, geochemical and biological processes (Eyre & McConchie, 1993). The degree to which estuaries modify the forms and quantities of riverine constituents will depend largely upon the residence time of material within the system (Balls, 1992; 1994; Eyre & Twigg, 1997). Hence, the final exports to coastal waters may not necessarily reflect the flux of materials from the catchment (Balls, 1994; Davies, 1996). This ability of estuaries to process riverborne nutrients, has resulted in them attracting considerable research attention over the past 30 years from which general conceptual models have been developed (Church, 1986; Fisher *et al.*, 1988). Most previous studies have been conducted in temperate northern American and western European estuaries. Southeastern Queensland is influenced by a different set of hydrological and climatic conditions, questioning the direct application of these overseas models.

This paper presents the preliminary findings of a study designed to quantify riverborne sediment and nutrient input to Moreton Bay from the Brisbane River catchment, and the modification of this input as it passes through the Brisbane River estuary. This study forms part of a larger study, currently being undertaken by the Centre for Coastal Management, of 12 estuaries in southeastern Queensland and northern New South Wales which aims to develop a conceptual model of nutrient behaviour in subtropical eastern Australian estuaries.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 545-552.

Study area

Moreton Bay is a large semi-enclosed coastal embayment located in southeastern Queensland. The catchment has a topographic relief of 0 to 682 m and is drained by four major rivers, the Caboolture, the Pine, the Brisbane and the Logan (Figure 1). The climate of this region is characterised by high summer rainfall with most rainfall occurring as high energy storm events and intense rainfall associated with tropical cyclones that generally form in the Gulf of Carpentaria or the Coral Sea. The lower portions of the Caboolture, Pine, Brisbane and Logan rivers are comprised of bar built estuaries which are dominated by tidal exchange except during periods of river high flow during such episodic high rainfall events.

This paper focuses on the Brisbane River, which is the largest river flowing into the Bay and has a catchment of 13 500 km^2. Both the riverine and tidal components of the hydrodynamic regime of this river have been significantly altered through the construction of instream storages, and through dredging for navigation and sand and gravel extraction. Tidal influence extends to Colleges Crossing, 86.4 km from the mouth. For the purposes of this study, however, the estuary is defined by the zone of mixing of salt and fresh water. The estuarine basin is defined as the area of tidal influence.

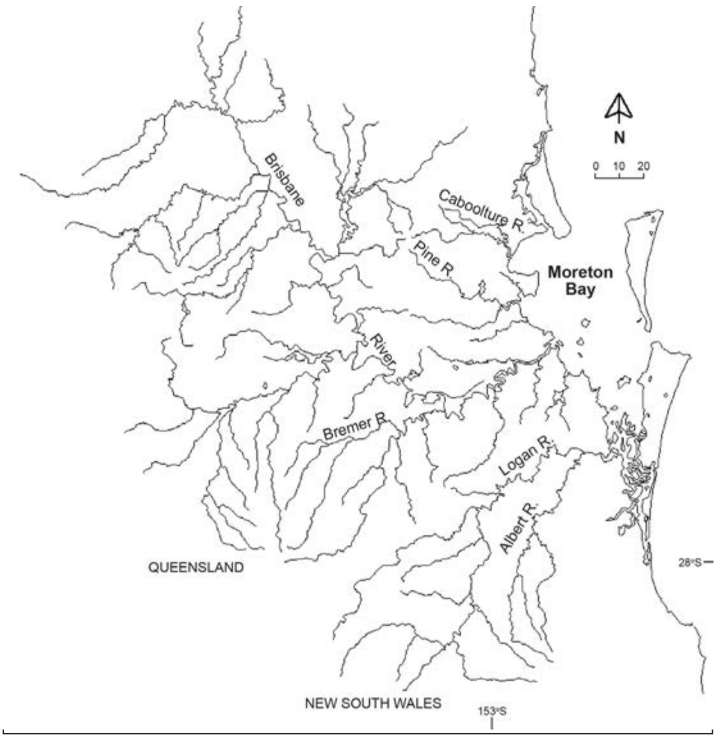

Figure 1. Moreton Bay catchment showing the major rivers.

Methods

Estuarine sampling program

The Brisbane River estuary was sampled during the peak of a 20 year return-interval flood on the 4-5 May 1996, and four sampling runs were undertaken following the flood in May, June, August and October (Eyre *et al.*, 1998). All sampling runs began one hour before high tide.

Samples were collected at intervals of approximately 2 on the Practical Salinity Scale, from seawater to freshwater along the axial salinity gradient of the estuary. Two sample collection strategies were employed depending on the salinity stratification of the water column. When vertical salinity profiles indicated the presence of a halocline, surface samples (top 30 cm) were collected using an acid-washed and sample-rinsed bucket (being careful not to collect the surface scum), and if the water column was well mixed, samples were collected from a depth of 1.5 m by hand pumping water to the surface. During peak flood flow (4-5 May), the estuarine basin was flushed almost fresh to the mouth and bad weather made sampling on Moreton Bay impracticable. Surface samples were therefore collected at approximately equal distances through the estuarine basin from the mouth to College's Crossing. Every tenth sample was collected in triplicate for quality control.

Samples for dissolved nutrients were filtered immediately through 0.45 µm cellulose acetate membrane filters (Sartorius) into acid-washed and sample-rinsed polyethylene vials. An unfiltered sample was collected in an acid-washed and sample-rinsed polyethylene vial and a 250 mL sample was also collected in a sample-rinsed polyethylene bottle for Total Suspended Sediment (TSS). All nutrient samples were frozen immediately in the dark on dry ice.

Salinity/conductivity was measured *in situ* at the point of sampling using a Horiba U-10 multiprobe. The U-10 was calibrated against 0.005, 0.05, and 0.5 M standard potassium chloride solutions. In several places, particularly in the lower estuary, water depths (up to 20 m) exceeded the maximum operational depth of the U-10. In these cases the profiles only extend to 10 m depth (Figure 2).

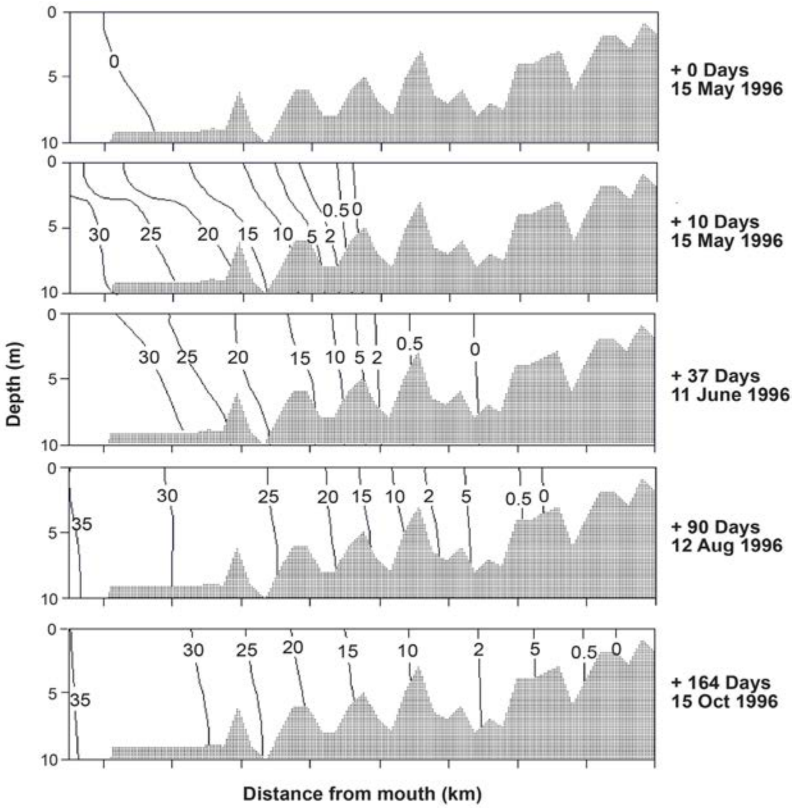

Figure 2. Salinity profiles in the Brisbane River estuary during the five sampling runs.

Analytical procedures

Nutrient forms analysed, abbreviations, analytical procedures and errors are summarised in Table 1. All nutrient analyses were carried out colourmetrically using Lachat™ Flow Injection Analysis. Approximately every tenth sample was collected and analysed in triplicate and analytical errors were determined as the average %CV of the triplicates. Analytical accuracy was checked using standard additions of certified laboratory standards in both high pure water and low nutrient seawater.

Table 1. Nutrient forms, abbreviations, analytical procedures, detection limits and analytical errors.

Nutrient Form	Abbrev-iation	Method	Source	Detection Limit	Error
Total phosphorus	TP	Persulfate digestion	Valderrama, 1981	0.10 μmol/L	3.8%
Total dissolved phosphorus	TDP	Persulfate digestion	Valderrama, 1981	0.10 μmol/L	3.5%
Dissolved inorganic phosphorus	DIP	Ascorbic Acid	Lachat, 1994	0.03 μmol/L	2.3%
Total nitrogen	TN	Persulfate digestion	Valderrama, 1981	0.70 μmol/L	4.3%
Total dissolved nitrogen	TDN	Persulfate digestion	Valderrama, 1981	0.70 μmol/L	4.1%
Nitrite	NO_2	Sulphanilamide	Lachat, 1994	0.07 μmol/L	2.8%
Nitrate	NO_3	Cadmium reduction	Lachat, 1994	0.07 μmol/L	3.6%
Ammonium	NH_4	Hypochlorite/ phenolate	Lachat, 1994	0.35 μmol/L	5.1%
Total Suspended Sediment	TSS	Gravimetric	Strickland & Parsons, 1972	0.10 μmg/L	3.0%

Results and Discussion

Physical structure of the estuary

Salinity profiles along the Brisbane River estuary during the five sampling runs (Figure 2) show a full range of mixing regimes from highly stratified to well mixed. On both 4 and 5 May the estuarine basin was flushed almost fresh to the mouth. The estuary recovered rapidly and ten days (+10) after the flood event a salt wedge had developed on the incoming tide with a pronounced halocline occurring at a depth of 3 to 4 m. This allowed salt water to intrude 34 km inland from the Brisbane River mouth. By day +37 salt water had intruded 54 km inland, the sharpness of the halocline decreased and a partly-mixed salinity regime developed. The estuary returned to its normal well-mixed state by day +99, but salt water continued to intrude landward reaching just upstream of the Bremer River junction by day +164.

Nutrient and suspended sediments in the estuary

Nutrient and suspended sediment concentrations in the Brisbane River estuarine basin during the flood peak are summarised in Figure 3. Phosphorus transport during the flood is dominated by the particulate fraction. In contrast, nitrogen transport is fairly evenly distributed between the dissolved and particulate fractions. TSS, TPN and TPP (Table 1) all show a mid-estuary maximum suggesting scouring and resuspension of bottom material during the flood. The decrease in suspended sediment concentrations toward the mouth is associated with the deposition of large quantities of material in the swing basin during the 1996 flood event which was subsequently removed by the Port of Brisbane Authority. These data highlight the importance of including estuarine processes when calculating nutrient fluxes to Moreton Bay; fluxes calculated by integrating upstream flow and concentration flood data would underestimate the actual load. In contrast to the particulate fractions, the dissolved nutrient concentrations are quite consistent through the basin suggesting a riverine source. Preliminary calculations based on concentrations measured at the mouth of the estuary multiplied by river discharges measured at upstream Department of Primary Industries' stream gauges indicate that approximately 150 000 t of suspended sediments, 44 t of phosphorus and 288 t of nitrogen were exported to Moreton Bay from the Brisbane River estuarine basin during the 24 h peak flood period.

NO_3, NH_4, NO_2, DIP and TSS (Table 1) concentrations as a function of salinity for the four

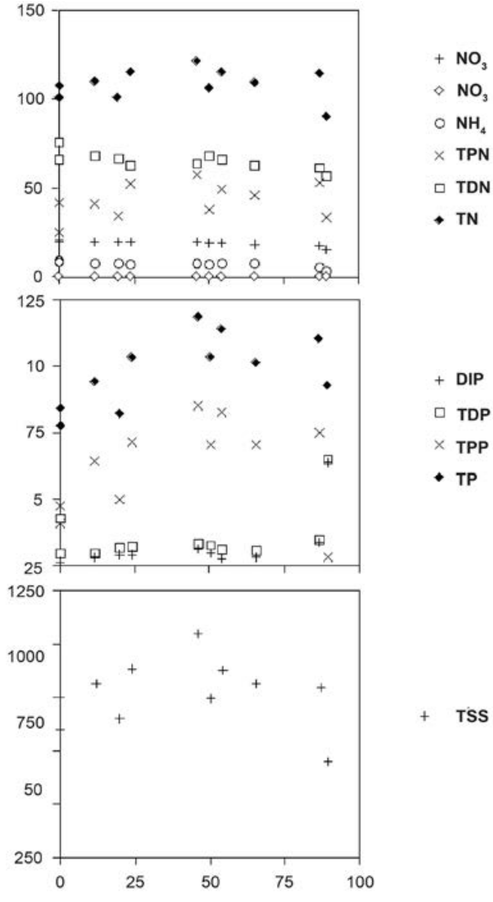

Figure 3. Nutrients and suspended sediments in the Brisbane River Estuary basin during the May 1996 flood event, 5 May 1996.

post-flood sampling runs are summarised in Figure 4. Average peak flood NO_3, NH_4, NO_2, DIP and TSS concentrations in the estuarine basin are also given at zero salinity. During the flood the estuarine basin is dominated by input from the river which has a "cleansing effect" effectively flushing the pre-flood brackish estuarine water into Moreton Bay. As river flow decreases, and the estuary recovers, dissolved nutrient concentrations increase as the estuary once again becomes dominated by the lateral point-source input (e.g. sewage treatment plants). The maximum influence of sewage treatment plant discharges can be seen during low river flow and long flushing times (i.e. sampling run 5), conditions which amplify such features. Upstream sources within the catchment appear to dominate DIP input, upper estuary sources appear to dominate NO_3 input and lower estuary sources appear to dominate NH_4 and NO_2 input. The geographical location of the maximum NO_3 and DIP concentrations could possibly be attributed to input from Oxley Sewage Treatment Plant whereas high concentrations of NO_2 and NH_4 adjacent to Luggage Point Sewage Treatment Works would indicate that this outfall contributes more reduced forms of DIN to the estuarine water column.

The estuary appears to act as a sink for TSS during the four post-flood sampling runs as

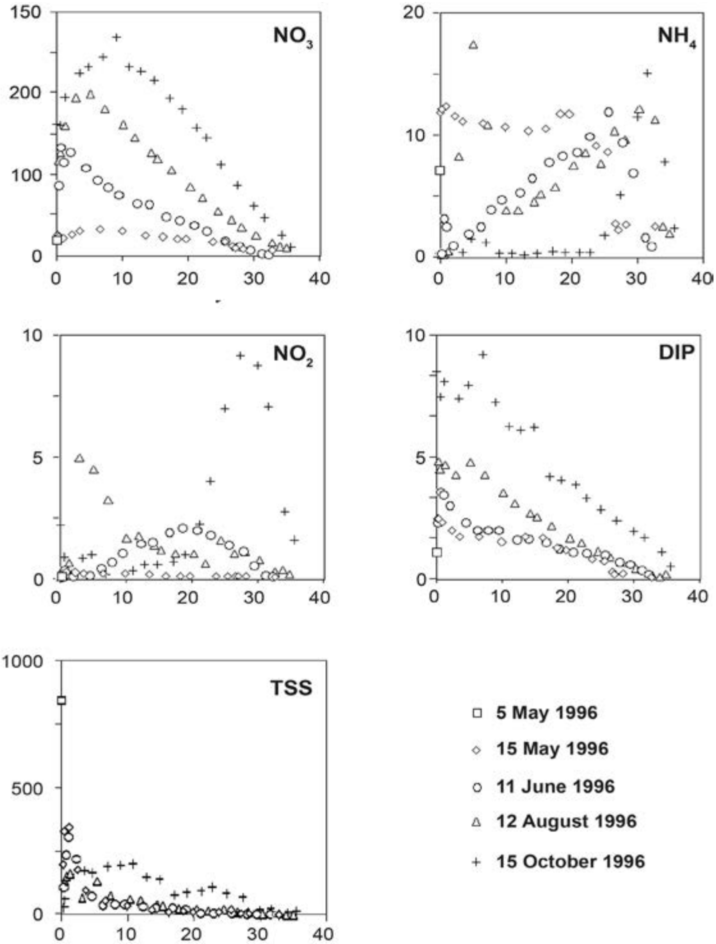

Figure 4. NO_3, NH_4, NO_2, DIP and TSS concentrations plotted as a function of salinity for the four post-flood sampling runs. Average peak flood NO_3, NH_4, NO_2, DIP and TSS concentrations in the estuarine basin are given at zero salinity.

indicated by the downward curvature of the TSS mixing plot. This suggests that sediments are deposited in the recovering estuary during low flows and subsequently scoured and transported during flood events.

A conceptual model for the Brisbane estuary

Eyre & Twigg (1997) present a three stage conceptual model for the Richmond River estuary that can be adapted to the Brisbane River Estuary.

Stage 1 represents flood conditions where the estuary is flushed fresh to the mouth and catchment material and material from within the estuarine basin is discharged directly into Moreton Bay.

Stage 2 represents the recovery stage as the estuary progresses from a highly stratified salt wedge structure, through a partially mixed system with a moderately developed two layer circulation, to a vertically homogenous system. During this stage material is probably deposited within the basin and at the mouth as the flood derived material settles from the upper layer to the lower layer and is transported landward. This process also develops a turbidity maximum at the salt/freshwater interface. Initially in the recovery stage material passes through the estuary relatively conservatively due to short flushing times, but processing of material increases as flushing times increase, as the estuary recovers. The estuary progresses from being river dominated to being dominated by lateral point source input.

Stage 3 represents normal conditions when the estuary has returned to a vertically homogenous system due to very low freshwater input. Long flushing times, in the order of months, typify this stage amplifying the influence of lateral point source input. Significant processing and transformation of material in the estuary may also occur during this stage and much of the river and point source nutrients may be transformed to other oxidation states, phases and compartments.

Acknowledgements

The authors wish to acknowledge the following organisations and individuals for their logistical support and participation in field work: Marine Botany Department, The University of Queensland, Department of Environment and Heritage, Fort Lytton staff and Richard Southon, Michelle Solomon, Sally Klein, Lenn Powell, Mark Bateman, Angus Ferguson, and Richard Roper. We also thank the Brisbane River and Moreton Bay Wastewater Management Study for permission to publish this paper.

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Impacts of the May 1996 Flood on Water Quality in Moreton Bay

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Abstract

In May 1996, much of the catchment of Moreton Bay experienced an approximately one-in-twenty year rainfall event. As a result, the Bay was subjected to a pulse loading of sediment and nutrients far in excess of normal dry weather inputs. A water monitoring program was set up to determine the impact on the Bay. Salinity values, particularly in the western Bay were considerably reduced immediately following the flood, however, salinity recovered rapidly once inflows ceased, providing evidence of the strong tidal mixing in the Bay. A large load of sediment entered the Bay during the flood. Most of this was deposited within the Bay with relatively little lost to the ocean. Some of the initial settlement occurred at locations not normally subject to fine sediment deposition, however, within a few weeks, Bay processes transported these atypical deposits to more permanent deposition sites. While much of the post-flood discolouration of the Bay was initially thought to be due to suspended particulates, measurements showed that dissolved colour (humic material) was a very significant contributor. Order of magnitude estimations indicate that nutrient loads entering the Bay during the flood were of a similar order to annual loads from point sources. Nutrient levels rose sharply following the flood but recovered quickly to background values, at most sites within three to four weeks. A distinct phytoplankton response (as measured by chlorophyll *a*) to this nutrient pulse was recorded at all sites, however, comparison with long term Bay chlorophyll *a* records showed that peak post-flood values were similar to or lower than maximum values recorded during non-flood conditions. Given the size of the nutrient pulse, this was an unexpected result.

Introduction

Moreton Bay was formed when the sea rose to its current level approximately 6 000 years ago (Stephens, 1992). Since that time, Moreton Bay has been subject to the impacts of marine processes from the east and catchment runoff and inputs from the west. These influences have formed Moreton Bay as we see it today. Marine processes have created sand deltas at the northern and eastern entrances whilst catchment input has formed sandy deltas at the mouths of rivers and deposits of fine sediment in the western part of the Bay (Stephens, 1992).

In its present form, Moreton Bay is subject to tidal exchange via its northern and eastern entrances and to freshwater inflows from rivers entering along its western edge. Tidal exchange with the ocean is the dominant hydrodynamic process in the Bay with a volume equivalent to that of the entire Bay being exchanged over only a few tidal cycles (McAlister, pers. comm.). In contrast, freshwater inflows are highly intermittent and, except during major flow events, are hydrodynamically insignificant compared to tidal influences.

For most of the time, the Bay functions as a relatively steady state marine system (McEwan *et al.*, 1998). Under this “dry weather” condition, there are no diffuse source pollutant inputs from the catchment. There are however, constant ongoing inputs of nutrient and other pollutants from point discharges, principally secondary treated sewage. These inputs occur as a result of direct discharge to the Bay or due to tidal exchange with estuaries subject to sewage discharges entering further upstream. The Bay assimilates the resultant nutrient load through various pathways and water quality in the Bay under dry weather conditions reflects an equilibrium with this constant input.

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 553-568.

At the beginning of May 1996, southeast Queensland experienced a six day period of intense rainfall, approximately equivalent to a one-in-twenty year event. This generated significant runoff and one of the largest short term inflows to Moreton Bay since the 1974 flood. There was no significant further rainfall in the two months after the event so that the inflow comprised a single discrete pulse and its impacts were not complicated by any subsequent inflows.

The May 1996 event provided a rare opportunity to study the Bay under conditions of significant freshwater inflow. More specifically, it was an opportunity to study the behaviour of the Bay under conditions quite different from its normal steady state. It was considered this would advance understanding of Bay processes and assist in the calibration of predictive models. Of particular interest was the reaction of the Bay to a large increase in nutrient supply over normal dry weather loads.

Overall, the flood study incorporated a number of interrelated measurement programs, one of which was the water quality monitoring program. Several other studies from this program are also reported in this volume. The aim of the water quality monitoring program was to record the changes in water quality in the Bay that occurred as a result of the flood. This paper presents and discusses these results.

Methods

At the time when the flood occurred, no comprehensive program for studying the impacts of a flood event on Moreton Bay was in place. Investigations of the flood were therefore planned at very short notice and with only limited resources available. With the benefit of hindsight, no doubt better or additional measurements would have been made, however, every attempt was made to coordinate measurements carried out during the flood with data collected in previous monitoring programs.

Five basic water quality parameters were sampled during the flood event.

(1) *Salinity*. Determined from conductivity and temperature measurements using Hydrolab or Grant YSI field probes. The probes were calibrated at regular intervals and precision for conductivity was ± 0.2 mS/cm and for temperature ± 0.10 ^{o}C.

(2) *Secchi depth clarity*. Standard Secchi disc, method according to the Australian Standard AS3550.7 (Standards Australia, 1993).

(3) *Total suspended solids* (TSS). Triplicate 1 L samples at each site, chilled and analysed within 48 h according to Clesceri *et al.* (1996).

(4) *Nutrients* (oxidised N [$NO_2 + NO_3$], ammonia N [NH_3], organic N, total P [TP], filterable reactive P [FRP]). Triplicate samples for soluble fractions and single samples for organic N and total P. All soluble fractions filtered in field through 0.45 µm filters and frozen. Organic N and total P were collected as unfiltered samples and stored frozen. All nutrients analysed according to Standard Methods (Clesceri *et al*., 1996). Standard deviations of the triplicate samples averaged less than 1 µg/L for NH_3, between 1 and 2 µg/L for oxidised N and less than 1 µg/L for FRP.

(5) *Chlorophyll* a. Samples filtered in field through 1.2 µm GFC filters, stored as frozen filters, extracted with acetone (24 h) followed by grinding and analysed according to Clesceri *et al.* (1996) using spectrophotometric methods. Standard deviations for triplicate chlorophyll *a* samples averaged between 0.3 and 0.4 µg/L.

Sites sampled are shown on Figures 1A (northern and central Bay) and 1B (southern Bay). Salinity was measured as a depth profile with readings at the surface (0.2 m) and at 2.0 m intervals thereafter. Water samples were all collected at 0.2 m depth. Some parameters such as salinity were measured at a wide range of sites throughout the Bay whilst parameters such as nutrients were measured at a subset of sites. Appendix A details all measurements at each site.

The main runoff period was from May 2 - May 7 (see Figure 2 which shows runoff patterns

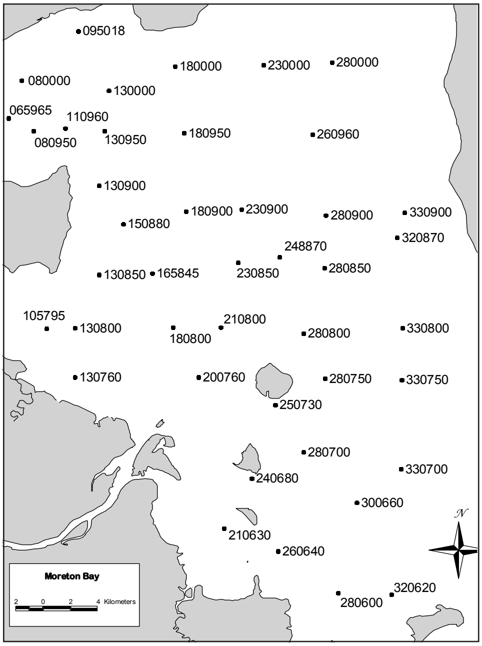


Figure 1A. Flood study sites in the northern and central Bay.

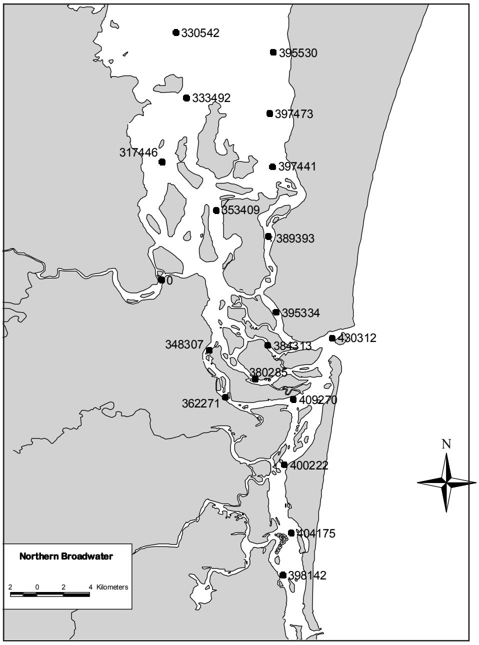


Figure 1B. Flood study sites in southern Bay.

in three sub-catchments). Initial sampling of the estuaries and a few sites just offshore from the estuary mouths occurred on May 4 (Davies & Eyre, this volume). The first comprehensive sampling in the Bay occurred on May 10, about five days after the peak of the inflows. This was followed by two further major surveys on May 24 and June 7. More limited sampling was carried out on May 31, June 21 and 28 and July 5.

Results

Timing of flood event

The rainfall event responsible for the 1996 flood event was principally concentrated over the period May 1 to May 7. Figure 2 shows flows in three streams in the Bay catchment, the Caboolture River, Lockyer Creek (a tributary of the Brisbane River) and the Logan River during the flood period. These tributaries are spread from north to south across the Bay catchment and the flow data therefore provide a reasonable representation of flood inflows to the Bay. Peak flows occurred at slightly different times in each catchment but were concentrated over the period May 2 to May 7. By May 9, flows in all streams had fallen at least an order of magnitude below peak flows and by May 12 flows had tailed off.

Hydrodynamics and salinity

Hydrodynamic aspects of the flood are discussed in more detail elsewhere in this volume and only an overview is given here.

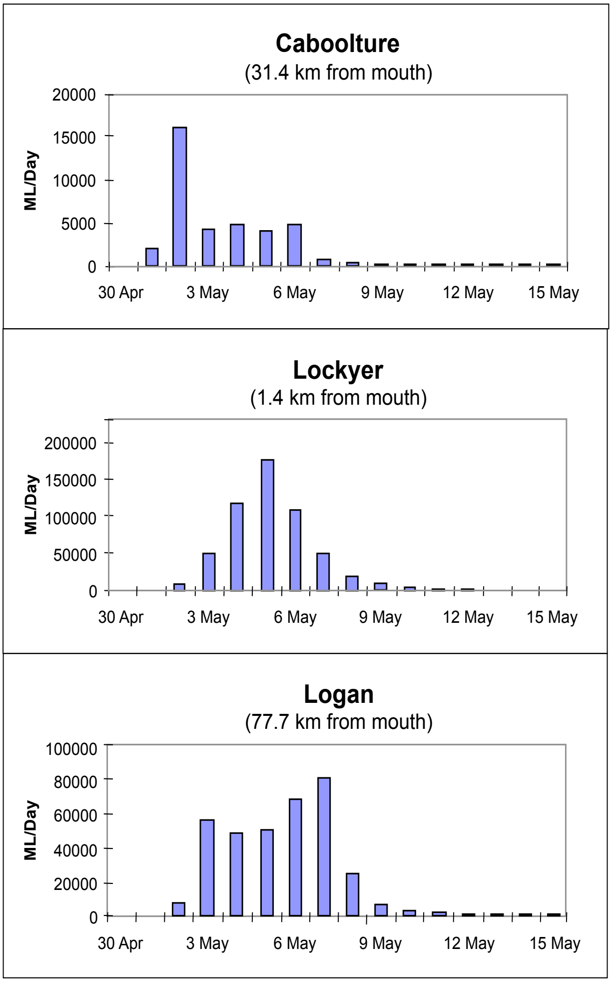


Figure 2. Stream flows in three Moreton Bay tributary streams during the period of the 1996 flood. (ML/Day = megalitres/day). (Data from Department of Natural Resources).

Flood volumes

Estimates of freshwater inflow to the Bay during the flood period are given in Table 1. These data are separated into two areal components, the main northern/central Bay area and the southern Bay from Russell Island to Jumpinpin Bar. The data are based on Brisbane River catchment estimates by Eyre (pers. comm.) and on Queensland Department of Natural Resources flow records. The table also provides data on Bay volumes and tidal prisms (McEwan, pers. comm.).

Table 1. Relative volumes (all $m^3 . 10^9$) of Moreton Bay, tidal prism and flood.

	Bay volume	Tidal prism	Flood volume
Northern Bay	11.70	1.36	1.2
Southern Bay	0.60	0.14	0.5

The relativity of flood and Bay volumes is illustrated in Figure 3. In the northern/central Bay, the combined flood inflows from the Brisbane/Cabbage Tree/Pine/Caboolture catchments were quite small (about 15%) compared to the Bay volume. Also, the inflows over the approximately 9 d flood period were about the same order of magnitude as a single tidal cycle. Thus, in hydrodynamic terms, the effect of the freshwater inflows in the northern/central Bay was quite limited.

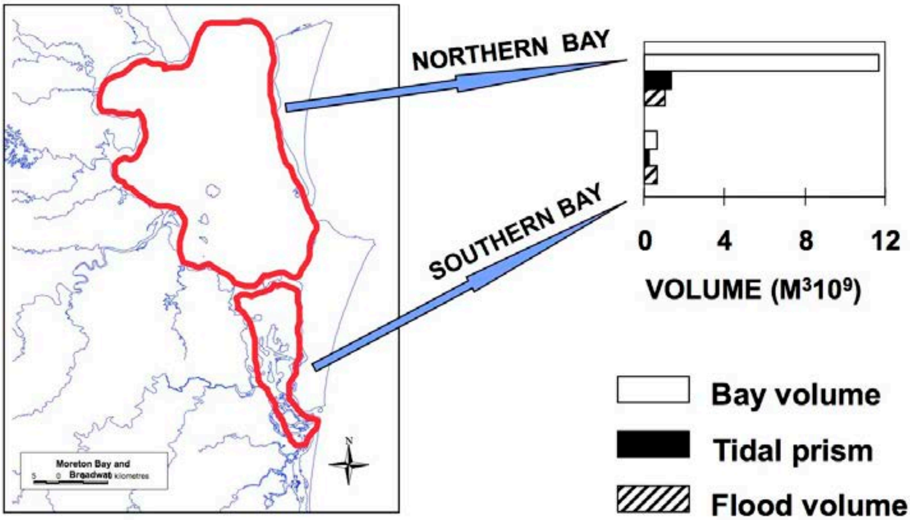


Figure 3. Comparison of relative volumes of Moreton Bay, its tidal prism and flood inflows.

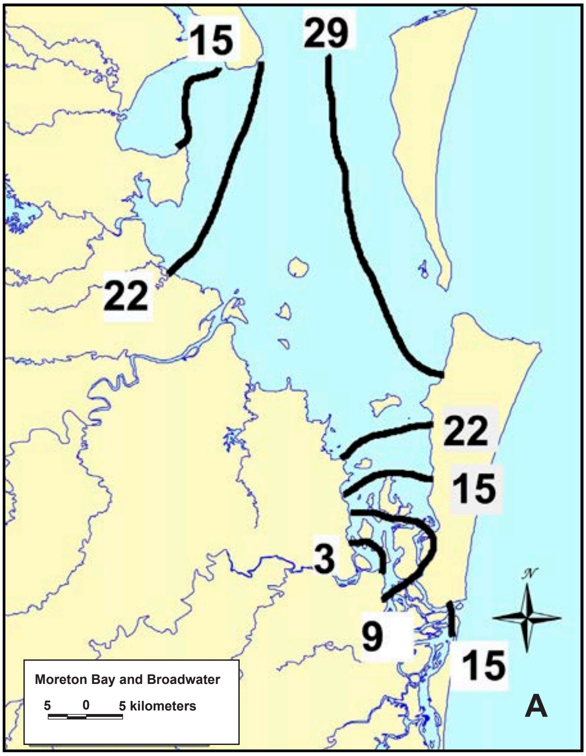
In contrast, inflows from the Logan/Albert system to the southern Bay were about the same order of magnitude as the Bay volume and larger than the daily tidal prism. Thus hydrodynamic and also other impacts of flood inflows could be expected to be much greater in this part of the Bay.

Salinity response

Measurements on May 4 at the height of the flood (Davies & Eyre, this volume) showed freshwater at the mouths of all the estuaries. The first comprehensive survey of salinity in the Bay did not occur until May 10 with subsequent surveys on May 24 and June 6. Because surveys were done at fortnightly intervals, measurements were taken at similar tidal states and are therefore comparable.

Figures 4A and 4B show surface salinity contours three days (May 10) and 17 days (May 24) after the end of the flood. On May 10, reduced salinity values were apparent in the western Bay but, as would be expected from consideration of relative flood volumes, values in much of the Bay were reduced by less than 20% compared to dry weather conditions. It is of interest that on May 10, lowest values in the northern Bay were around the mouth of the Caboolture River rather than the mouth of the Brisbane River, which would have contributed much larger inflows. This indicates that rates of tidal mixing and dispersion in the Deception Bay area are slow relative to other parts of the Bay. This has implications for issues such as effluent disposal.

Again, as would be expected from consideration of flood volumes relative to Bay dimensions, reductions in salinity values around the Logan were larger than in the northern Bay.



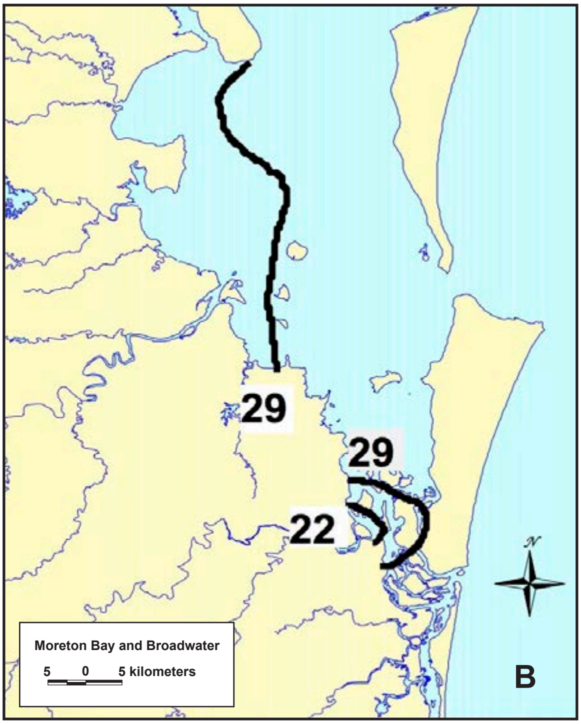


Figure 4. Surface salinity (ppt) in Moreton Bay after end of the 1996 flood: A. 3 days; B. 17 days.

By May 24, salinity values in the northern/central Bay had increased considerably indicating that a large proportion of the freshwater inflow had moved out of the Bay during the 14 d period. This rapid exchange is a good indication of the strength of tidal mixing that occurs in Moreton Bay. Concurrent large changes in salinity in the southern Bay indicate that quite rapid exchange also occurs in this part of the Bay.

On the first Bay survey on May 10, a degree of vertical stratification was evident in some western Bay sites, with surface-bottom salinity differences of up to 7 ppt. Differences were much lower at eastern Bay sites. By May 24, these differences had reduced to 2 ppt or less at virtually all sites, further evidence of strong tidal mixing in the Bay. A well mixed water column is the normal dry weather state of Moreton Bay (Department of Environment, unpublished data).

Suspended solids, colour and clarity

Suspended solids

Figure 5 shows total suspended solids (TSS) values at three representative sites in the Bay over the period of sampling. Values at the river mouths, measured by Davies & Eyre (this volume) on May 4, are also shown.

Davies & Eyre's measurements indicate, as would be expected, that at the flood peak, waters entering the Bay contained high levels of suspended solids. These ranged from 730 mg/L at the mouth of the Brisbane River to 65 mg/L at the mouth of the Caboolture River.

On May 10, the first date on which TSS measurements were taken, values at most sites in the northern/central Bay were around 20 mg/L. Higher concentrations, around 40 mg/L, occurred

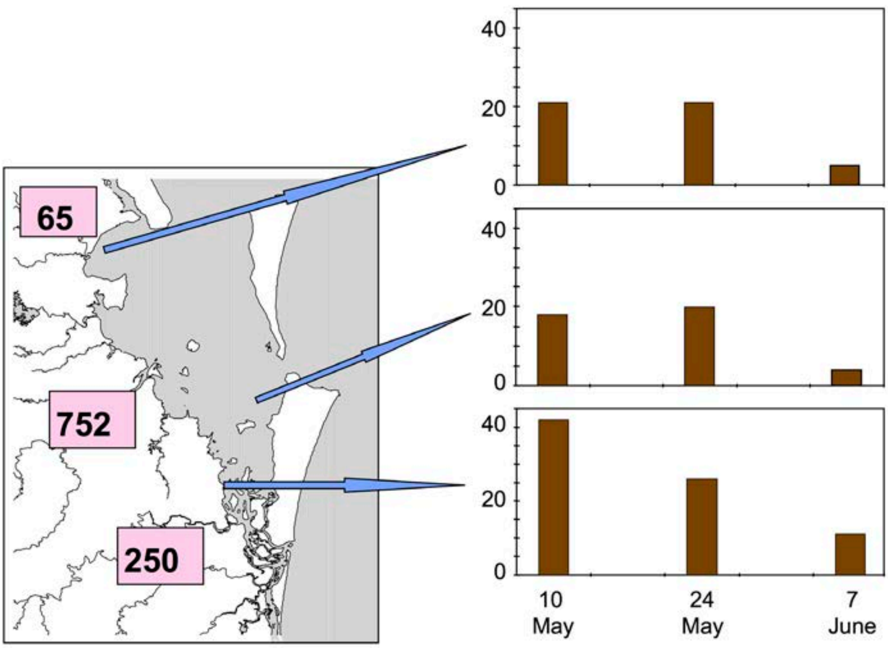


Figure 5. Time series of post-1996 flood total suspended solids (TSS) values (mg/L) at three Moreton Bay sites. Figure also shows TSS values (in boxes) at river mouths on May 4th.

at sites close to the Logan River mouth, as would be expected given the higher volumetric loadings in the southern Bay. A fortnight later, on May 24, values at most Bay sites were still around 20 mg/L. Given the considerable exchange of water that had occurred during that period (see salinity results), values might have been expected to have fallen by this time. However, gale force winds that occurred during some of the intervening period are presumed to have resulted in the remobilisation of sediments that had settled from the flood. By June 7, TSS values had returned to near background concentrations.

The reduction in TSS values from the high concentrations recorded at river mouths on May 4 to the concentrations recorded in the Bay six days later would be mediated by the processes of dilution, dispersion and settlement. The salinity data indicate significant rates of dilution and dispersion but no direct measurements of settlement were attempted. The fact that considerable amounts of fine sediments did settle out is attested by visual observations of thick deposits of fine mud on seagrass beds in the Pelican Banks area (J. Horrocks, pers. comm.). This particular deposit would have originated from the Logan River plume. It is of interest to note that many of these deposits subsequently disappeared (J. Horrocks, pers. comm.) which suggests that they were not in equilibrium with the dry weather hydrodynamic regime. The eventual destination of these fine particulates is not known but it is likely they were dispersed among the various permanent fine silt deposits within the Bay. Evidence from parallel measurements of colour and TSS (see below) suggests that most of the suspended sediment was trapped within the Bay rather than dispersed out into the ocean.

Colour

Colour measurements were not included in the main monitoring program, however, some colour data were collected as part of a post-flood seagrass monitoring exercise undertaken by the Department of Environment.

Figure 6 shows colour and TSS data collected at a seagrass monitoring site north of Peel Island. These data are representative of data collected at a number of other sites along the eastern fringe of the Bay north of Peel Island during that period.

The data show a post-flood colour peak but no measurable increase in TSS (detection limit = 5 mg/L). This suggests that most of the suspended load had already settled out before the flood

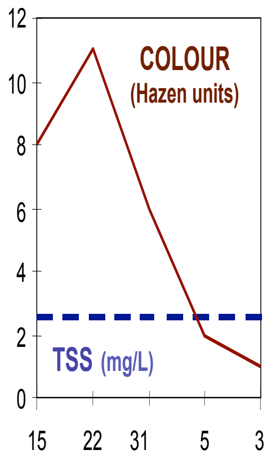


Figure 6. Post-flood colour and total suspended solids (TSS) values at a seagrass monitoring site north of Peel Island, Moreton Bay in 1996 following a one-in-twenty year flood event.

plume reached this area and only the dissolved colour component remained. Further evidence of the prior settlement of particulates is that whilst Pelican Banks were covered in a mud layer deposited by the Logan plume, sites further north (northern side of Peel Island) experienced little or no deposition (J. Horrocks, pers. comm.). A similar colour/TSS pattern was recorded at sites along Moreton Island indicating that very little sediment from the Brisbane River and other northern Bay rivers reached the eastern Bay.

These results indicate that the majority of the particulate load from the flood plumes settled out in the western and southern Bay. This is in accord with the known distribution of fine sediments in the Bay (Stephens, 1992).

Following the flood, it was directly observed by the author that virtually the whole of Moreton Bay exhibited a brown, tea-like colouration. At the time this was assumed to be due mainly to suspended particulates but, in retrospect, it appears that much of this colouration was due to dissolved colour rather than suspended material. A distinguishable flood plume was noted as far north as Mooloolaba and it now seems likely this was largely made up of dissolved colour. Increased colour and TSS led to reduced water clarity at all sites. In most cases, clarity returned to normal values within four to five weeks of the end of the flood event. In the absence of comprehensive colour data, it is not possible to state whether colour or suspended load was the main contributor to reduced clarity in the central and western Bay.

Nutrients

Nutrient loads

Dry weather point source nutrient loads to the Bay and its estuaries are estimated to be of the order of 2 000 tonnes N and 750 t P per day (SKM, 1996). Reliable estimates of loads from the flood event are not available but order of magnitude approximations can be made by multiplying flow data (obtained from the Department of Natural Resources), and total catchment flow estimates (Davies & Eyre, this volume) by data for nutrient levels near the mouths of estuaries.

Table 2 summarises dry weather loads and estimated flood loads of N and P entering the Bay. The loads are split into northern and southern Bay components. Nitrogen loads entering the Bay during the six day flood period were of the same order as annual point loads in the northern/central Bay and about four times the annual point loads in the southern Bay (Table 2). Flood loads of phosphorus were approximately one-third the annual point loads in both areas of the Bay.

Thus, over the six day flood period, the nutrient load was equivalent to about one year of dry weather load of N and four months of P. Whilst these values are subject to significant errors, it is clear that over the flood period, the Bay was subjected to nutrient loads far in excess of normal dry weather inputs.

Table 2. Estimated annual point and 1996 flood nutrient loads (tonnes N or P). Note that the flood load estimates cover a 6 d period whilst the point source estimates are annual loads.

	Point source (Yr^{-1})		Flood load	
	N	P	N	P
Northern Bay	1 800	600	1 400	200
Southern Bay	180	140	700	60

Nutrient levels in the Bay

Total inorganic nitrogen (TIN) concentrations at all sites were well above background on May 10, but by May 24 concentrations of TIN fell to near background at northern/central Bay sites (Figure 7). The reduction in TIN was slower in the southern Bay, probably due in part to the higher flood loadings in this part of the Bay.

Filterable reactive P (FRP) also fell rapidly at the northern Bay site but relatively more slowly than TIN at the central Bay site. At the southern Bay site, TIN initially fell more rapidly but both fractions reduced to baseline levels within four weeks.

In the first sampling period (May 10), TIN comprised less than 50% of total N at nearly all sites. This suggests that more than half the N load entering the Bay during the flood was in an organic form. The form of the organic N load is not known but if a significant proportion was refractory, this could partly explain the lower than expected phytoplankton response in the Bay.

In contrast, filterable P on May 10 comprised over 90% of the total P at nearly all sites. Since most of the catchment P load in flood events is in the particulate form (Cosser, 1989; Hunter,

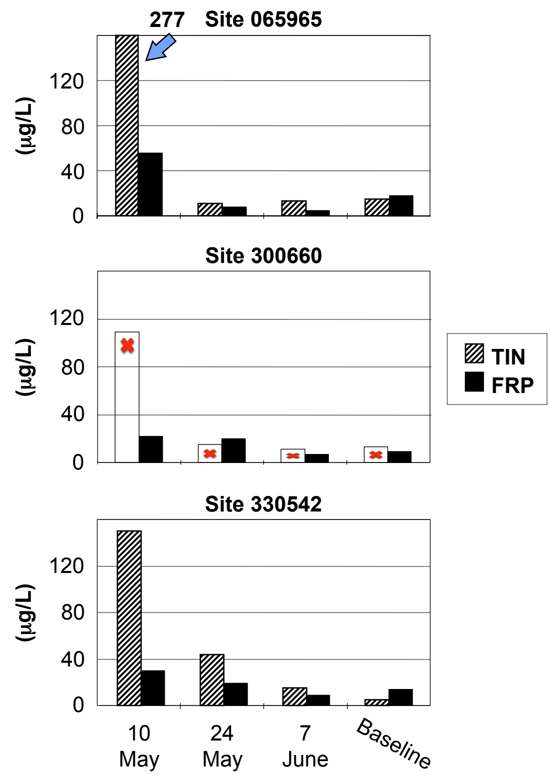


Figure 7. Post-1996 flood changes in TIN (total inorganic N) and FRP (filterable reactive P) concentrations (µg/L) at three Bay sites (see Figures 1A & 1B). Shows TIN (total inorganic N [NH^3 + Oxidised N]) and FRP (filterable reactive phosphorus) levels at three representative sites in the post flood period. The figure also includes typical dry weather (background) concentrations (Department of Environment, unpublished data) at these sites for comparison.

1996), this observation would indicate that most of the particulate P load had either settled out in the estuaries or had very rapidly settled once it entered the Bay.

Chlorophyll a

Flood response

The results clearly indicate a pulse of phytoplankton growth throughout the Bay following the flood (Figure 8). The pulse was short lived at the northern site and quite small at the central Bay site. At the southern Bay site, only three data points were available but it would appear that phytoplankton growth at this site was more sustained. This was probably a function of the higher nutrient loadings in this part of the Bay.

A long term (5 years) series of weekly chlorophyll *a* data (Department of Environment, unpubl. data) is available for two sites in the Bay, one in the eastern Bay and one in the outer western Bay at the mouth of the river shipping channel. These data allowed the chlorophyll *a* response to the flood to be assessed in relation to a long temporal record.

At the eastern Bay site, the long term data set shows a consistent seasonal pattern of mid to late summer maxima and winter minima (Figure 9). The post-flood response in May 1996 is readily apparent as a set of unseasonally elevated values. However, compared to many of the

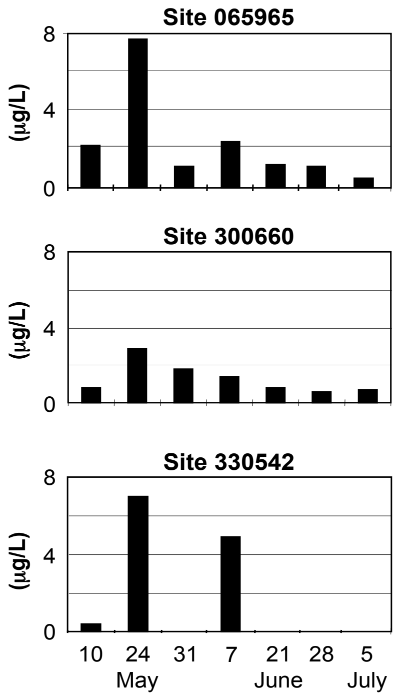


Figure 8. Post-flood chlorophyll *a* values (µg/L) at three Moreton Bay sites (see Figures 1A & B) in 1996, showing time series of chlorophyll *a* values at sites in the northern (065965), central (300660) and southern (330542) Bay. While only three sites are shown, the chlorophyll *a* patterns at these sites are representative of other parts of the Bay.

summer peak values, the magnitude of the flood response is not particularly large. In fact, some previous summer peak values exceed peak post-flood values. Given that Bay chlorophyll *a* values normally peak during summer months it is reasonable to speculate that, had the flood occurred during the optimum midsummer growth period, then the flood response may have been much larger.

At the western Bay site, chlorophyll *a* values were considerably higher than at the eastern Bay site. This would be expected given the generally higher levels of nutrient and lower rates of dispersion. Unlike the eastern Bay, no seasonal pattern was readily apparent. Chlorophyll *a* levels in the western Bay are more likely to be influenced by inshore factors such as small river inflows or wind driven re-suspension of sediments. Such factors may override the influence of seasonal factors.

Elevated chlorophyll *a* values occurred in the western Bay during the post flood period. However, as with the offshore site, comparison of the post flood maxima with the long term record indicates that the post-flood values were well within the range of previously recorded peak values.

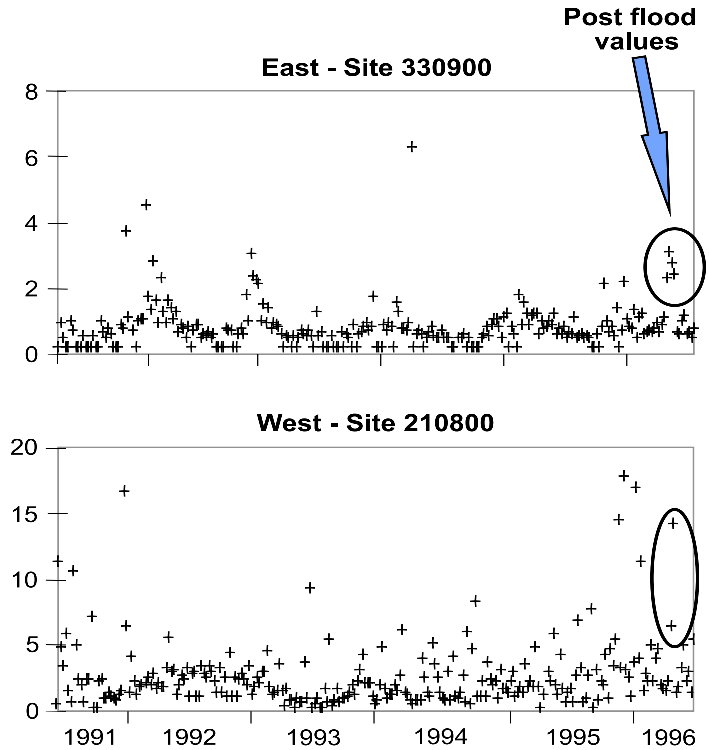


Figure 9. Long term chlorophyll *a* (µg/L) record at an eastern and western Moreton Bay site. Shows the long term data set (1992-1996) with the post-flood data points near the end of the series.

Growth limitation by light and nutrient factors

At the start of the sampling series, just after the main inflow to the Bay ceased, chlorophyll *a* and secchi depth values were low while nutrient levels were high (Figure 10). Two weeks later, chlorophyll *a* values peaked while TIN values had decreased to the very low concentrations characteristic of dry weather conditions. FRP values were similarly reduced in the northern Bay but to a lesser extent than TIN in the central Bay. From that point on, both chlorophyll *a* and nutrient concentrations declined while light penetration increased due to ongoing reductions in suspended load and colour.

The most straight forward interpretation of these results is that phytoplankton growth was stimulated by the additional nutrient load but within two weeks became limited by either N on its own or a combination of N and P. Continued improvement in light penetration failed to stimulate any significant further growth.

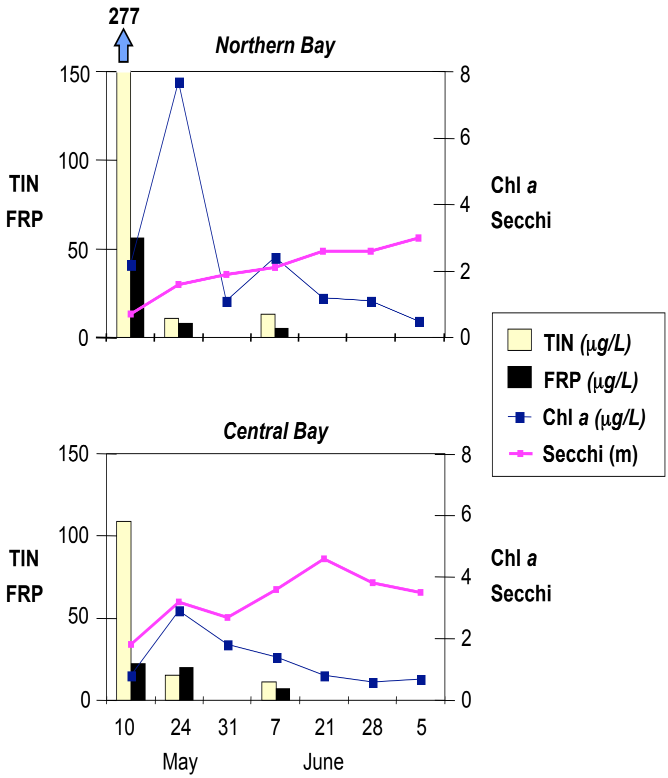


Figure 10. Comparison of time series nutrient, chlorophyll *a* and secchi values in the post-flood period in Moreton Bay peak inflows between May 2 and May 7 1996. Shows chlorophyll *a* values at two representative sites plotted alongside secchi depth values and dissolved N and P concentrations.

Discussion

The results of these studies provide information on the direct impacts of the flood on water quality in Moreton Bay. Knowledge of the reactions of the Bay to the flood also provide an opportunity to make inferences about Bay processes. Much is known about Bay processes from dry weather studies but information from wet weather studies allows models developed from the dry weather studies to be tested.

One strong inference from the salinity data is that the whole Bay is subject to powerful tidal mixing processes. This was evidenced by the rapid decline of salinity gradients after the end of the flood and the rapid breakdown of any vertical stratification. The greater persistence of salinity gradients in the Deception Bay area indicates that mixing is relatively slower in this part of the Bay.

Whilst mathematical modelling (McEwan *et al.*, 1998) has allowed estimation of the influence of floods on mixing in the Bay, data from the 1996 flood event will provide a useful means of verifying the predictions of such models.

The suspended solids and related colour data indicate that the majority of the particulate load entering the Bay during all but the very largest floods is sedimented out in the western, central and southern Bay. Evidence from the seagrass studies indicates that sediment from the Logan River settled out south of Peel Island (J. Horrocks, pers. comm.). At the same time, plumes from the Brisbane river and rivers to the north tend to head in a northeasterly direction. Thus, the area of the Bay from Mud Island south to Peel Island is little affected by flood plumes and sediment deposition. It is probably no coincidence that the most prolific coral communities within the Bay also occur in this area.

Very large nutrient loads entered the Bay during the flood period, however, water column concentrations of dissolved nutrients declined rapidly during the post-flood period. This decline is partly linked to the processes of dilution and dispersion which salinity data indicate are very significant within the Bay. Other removal mechanisms include adsorption onto suspended bottom sediments and biological assimilation. To quantify the relative importance of these processes would requires a modelling approach which is beyond the scope of this paper.

Dry weather studies of Bay water quality (Moss, 1992) show that TIN levels throughout much of the Bay are consistently close to the limits of detection. At the same time, ratios of dissolved N:P are generally around 1 (by weight). These data suggest that N is a key factor limiting phytoplankton growth in the Bay during dry weather. The post-flood pattern of nutrient levels indicates that nutrients, rather than light, were the key limiting factor. N was reduced to background levels more rapidly than P at some sites but at others both N and P were rapidly reduced. Chlorophyll *a* levels peaked soon after the flood and fell rapidly once N or both N and P had declined to dry weather values.

Light has been considered to be an intermittent limiting factor in the western embayments but the flood results suggest that it may be less significant than was previously thought. The chlorophyll *a* peak was reached in the two weeks immediately following the cessation of inflows. During this period, water clarity was still low. Improved clarity during the subsequent period had no impact on growth. It appears therefore that provided nutrients are present in excess, phytoplankton grow quite well even in a relatively poor light regime.

An unexpected finding of the flood study was that whilst the flood inflows stimulated widespread phytoplankton growth, the magnitude of this response was within the range of previously observed growth pulses (see Figure 9). Thus, although flood nutrient loads were similar in order of magnitude to total annual inflows from point sources, their impact in terms of growth response was not commensurate. The reasons for this are not well understood. The strong tidal mixing in the Bay may be one factor preventing buildup of phytoplankton biomass. The fact that the flood occurred in May, which is a time of naturally low biomass (see Figure 9) may also be a factor. Much larger phytoplankton populations may have resulted had the flood occurred in midsummer. A large proportion of the N load was in the organic form and it is possible that much of this was in a refractory form and therefore unavailable to the phytoplankton. Also, a significant fraction of the nutrient load may have settled out with the suspended particulates and thus become less readily available.

It is, however, important that the apparently small impact of the flood nutrient load should not be taken to necessarily indicate that the Bay has the capacity to assimilate large increases in point nutrient loads. Whilst this might be true of the Bay as a whole, it is probably untrue of the less well flushed western embayments. As noted above, the situation is complex and a number of factors are involved.

A final general comment on the flood study is that during such episodes, things happen very fast. During the main inflow period, significant changes in water quality occur over periods of one or two days. A lesson for any future studies is that a monitoring plan should be determined in advance so that sampling can begin immediately it is evident that a major flood event is in progress, however, it is necessary to also develop practical means of undertaking frequent measurements of key parameters so that short term variations are not lost. Above all, the aims of any future flood monitoring studies need to be clearly articulated so that data collected may be used to address the key questions.

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Appendix A – Summary of Water Quality Sampling Program

Sampling regime at each site			
1	**2**	**3**	
Northern Bay	080000	095018	065965
130000	110960	180950	
180000	150880		
230000	248870		
280000	320870		
080950	165845		
130950			
180900			
230900			
280900			
330900			
130850			
230850			
280850			
Central Bay	130800	210800	100800
180800	260640	330600	
280800		210630	
330800			
130760			
200760			
280750			
330750			
250730			
280700			
330700			
240680			
320620			
280600			
Logan-Coomera	317446	330542	
355358	333492		
Logan mouth	353409		
348307	362271		
400220	404175		
409270	430312		
395334	397473		
389393			
397441			
395530			

1. Field readings (DO, Temp, pH, Cond, Turbidity) - 10/5, 24/5, 7/6;
2. Field readings + Nutrients, Chl *a*, TSS - 10/5, 24/5, 7/6;
3. As for '2' + Chl *a* - 31/5, 14/6, 21/6, 28/6, 5/7.

Phytoplankton Community Response to a Flood Event

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Abstract

Moreton Bay experienced a Baywide phytoplankton bloom in May 1996, which followed a one-in-twenty year flood event in the Moreton Bay catchment. Sampling was conducted at 15 sites throughout Moreton Bay for a two-month period following the flood. During the event, Moreton Bay was characterised by both an east-west gradient in phytoplankton abundance associated with each of the three major rivers entering Moreton Bay and a north-south gradient in phytoplankton community composition. Tows (<20 µm) conducted one week after the cessation of rainfall at the Caboolture, Brisbane and Logan River mouths and plumes were comprised of a mixture of large diatoms and heterotrophic dinoflagellates. High concentrations (>2.9 x 10^6 cells/L, >19.0 µg chlorophyll *a*/L) of diatoms (*Rhizosolenia setigera, Pseudonitzschia* sp. and *Skeletonema costatum*) dominated the entire Bay one week later. Cell concentrations during this period were highest within each river plume, where phytoplankton biomass was maintained by nutrient input from the rivers, while decreased light penetration associated with high concentrations of suspended material limited phytoplankton abundance at the river mouths. Five weeks after cessation of rainfall cell concentrations had decreased in the northern Bay, but remained elevated in southern Moreton Bay. Although dinoflagellate concentrations increased in the northern Bay, the diatoms *R. setigera* and *Pseudonitzschia* sp. remained numerically dominant within the central and northern Bay. Species composition in the lower Bay shifted to dominance by estuarine *Chaetoceros* spp. and the benthic diatom *Baccilaria paxillifer.* By mid-June, cell concentrations within the entire Bay declined to 4.2-7.0 x 10^5 cells/L (0.9-9.4 µg chlorophyll *a*/L) and population composition shifted to a diverse assemblage of estuarine diatom species. During the entire survey period, dinoflagellate populations comprised a large percentage of the northern Moreton Bay phytoplankton community. The bloom event indicated that although non-bloom chlorophyll *a* concentrations in Moreton Bay are generally low (<0.3 µg/L), the phytoplankton community can respond to nutrient input on a Baywide scale.

Introduction

Australian estuaries are characterised by pulsed freshwater input which varies both temporally and spatially (O'Donohue & Dennison, 1997). The effects of freshwater flood events on coastal systems are two-fold: they introduce large quantities of dissolved inorganic nutrients (both nitrogen and phosphorus) and suspended material from catchments into coastal waters and also alter the physical dynamics and regime. Both effects can significantly impact phytoplankton community composition, nutrient cycling and trophic dynamics in bays and estuaries. In temperate ecosystems, periods of heavy rainfall and runoff are often correlated with increases in phytoplankton biomass and productivity in downstream and receiving waters on time scales that range from days (Garcia-Soto *et al.,* 1990) to weeks (Rudnek *et al.,* 1989) to seasons (Bennett *et al.,* 1986; Jordan *et al.,* 1991). In subtropical waters, the ecological consequences of variable freshwater input on estuarine phytoplankton communities are largely undocumented. Differences in physical parameters (e.g. daily and seasonal light and temperature regimes) and phytoplankton community structure and composition between temperate and tropical marine coastal ecosystems (e.g. the dominance of phytoplankton biomass and production by the < 0.2

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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µm picoplankton fraction (Li *et al.,* 1983; Takahashi & Bienfang, 1983; Longhurst & Hardy, 1987; Furnas & Mitchell, 1996) suggests that these systems potentially respond to significant freshwater input in different manners.

In May 1996, southeast Queensland experienced an extended period of sustained rainfall (30 April to 8 May) which resulted in a one-in-twenty year flood event in Moreton Bay and its catchment. A significant ecological consequence of this flood event was the development of a large phytoplankton bloom sufficient to discolour the water within Moreton Bay. The development, potential causes and ecological significance of this bloom are discussed in this paper.

Methodology

Nine sites were sampled within Moreton Bay by The University of Queensland (UQ) Marine Botany group (Figure 1). A site was located at the mouth of each of the three largest rivers entering Moreton Bay: the Caboolture River, the Brisbane River and the Logan River. Three sites were located within the plumes of each of the three rivers, and in addition, an oceanic water site located in the eastern Bay adjacent to each river system. Six additional sites (located within inner and outer Deception Bay, Bramble Bay and Waterloo Bay) were sampled jointly by UQ and Queensland Department of Environment. The nine sites were sampled on 20 May, 4 and 18 June and 16 September. The six additional Bay sites were sampled at weekly intervals from 31 May to 5 July.

Sampling protocol consisted of both a water quality and a phytoplankton identification component. Temperature, salinity, dissolved oxygen concentration and pH were measured on site with a Horiba Model U-10 Water Quality Checker. Light attenuation was determined by secchi depth at each site. Depth of the photic zone was calculated from secchi disk depth measurements according to Cloern (1987). Triplicate water samples were collected from 1 metre below the surface at each site with a modified Nisken bottle and immediately transferred to clean buckets. A sample from each bucket was filtered (Whatman GF/F filters) and immediately frozen for chlorophyll *a* analysis (Parsons *et al.,* 1984).

Samples were also taken at each site for enumeration of phytoplankton community composition. These consisted of a total water sample (150 mL) taken from the Nisken sample and a 2-minute phytoplankton tow. The tow was conducted at ~1 knots using a 20 µm net. Both samples were preserved on site with Lugols preservative (Tomas, 1996) and stored in a dark, cool room prior to enumeration.

Phytoplankton were identified to the species level by morphology using light microscopy according to available taxonomic guides (Dakin & Colefax, 1933; Wood, 1954; 1968; Hendley, 1964; Taylor, 1966; Cupp, 1971; Sournia *et al.,* 1979; Chang, 1983; Hallegraeff, 1984; Rines & Hargraves, 1988; Tomas, 1993; 1996). When required, samples were destained with 0.1 N sodium thiosulfate to allow identification of dinoflagellate plate tabulation. Samples with total cell concentrations less than 200 cells/mL were concentrated prior to enumeration according to a modification of the method of Utermohl (1958). Samples were gently mixed and 50 mL was decantered into a 60 mL plastic centrifuge tube. Samples were settled for 24 h and the top 45 mL was aspirated off each sample. The remaining 5 mL of concentrated sample was counted as previously described.

As the addition of excessive amounts of Lugols preservative to phytoplankton samples can result in cell disruption (Sournia, 1978; Tomas, 1996), a preservative comparison experiment was conducted during the survey of 14 June. In addition to samples preserved with Lugols preservative, duplicate samples from inner Deception and Bramble Bays were taken and preserved with 1% formalin and 5% buffered glutaraldehyde (in 0.05 M phosphate buffer (pH=7.4) with 5% sucrose) to test for the efficiency of preservation of unarmoured flagellate cells. Samples were stored and enumerated as previously described.

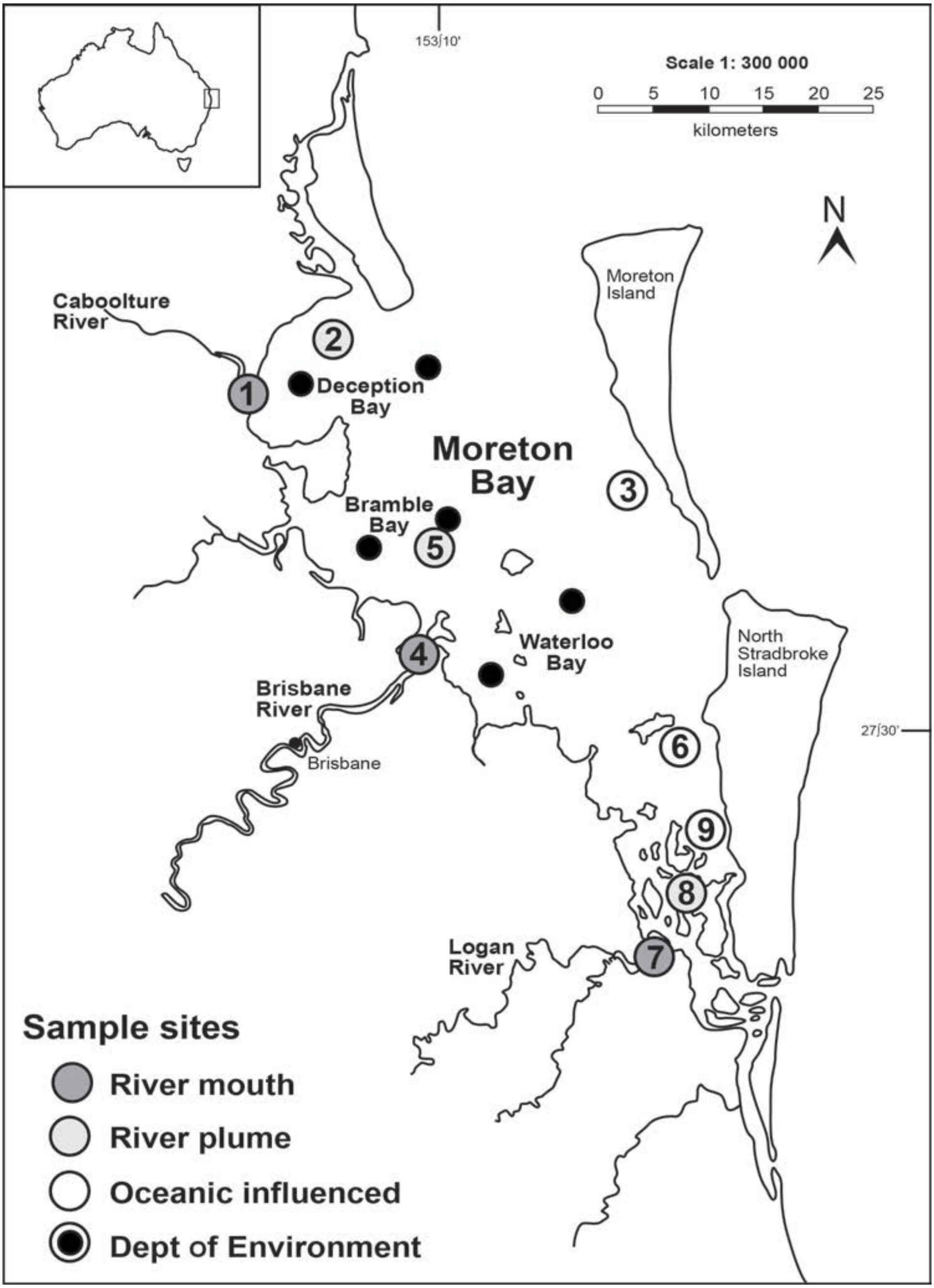

Figure 1. Location of Moreton Bay post-flood (1996) phytoplankton and water quality sampling sites.

Results

Successional changes in phytoplankton abundance and community composition during the phytoplankton bloom which occurred in Moreton Bay following the flood event were examined from two weeks after the cessation of rainfall through the decline of the bloom in late June. An additional survey, conducted 129 days after the cessation of rainfall served as a post-bloom survey for comparison purposes. Maximum cell concentrations, up to 1.1×10^7 cells/L on 25 May, occurred at the outer Bramble Bay site (Figure 2). Nearshore and southern Moreton Bay phytoplankton abundance were also elevated at this time. Restricted sampling did not allow evaluation of abundance within the northern Bay during this period. Phytoplankton communities consisted of a mixture of diatoms and dinoflagellates within Bramble Bay and a diatom dominated community within the southern Bay. Within Bramble Bay, the diatoms *Pseudonitzschia* sp. and *Skeletonema costatum* and an unknown *Gymnodinium* sp. were numerically dominant and comprised greater than 79% of phytoplankton. Southern Bay sites were dominated by *S. costatum*, with concentrations up to 1.9×10^6 cells/L.

Within one week, a Baywide decrease in cell concentration was accompanied by a shift in phytoplankton community dominance to diatoms, primarily *Rhizosolenia setigera*, *S. costatum* and *Pseudonitzschia* sp., within the northern Bay (Figure 3). The apparent community composition boundary between the northern and southern Bay, located the previous week in the central Bay, shifted northward. Populations with high concentrations of flagellates were found only in the extreme northern Bay at two sites north of the Caboolture River. During this period the highest cell concentrations consistently occurred at or adjacent to the plume sites of each of the three rivers, while lowest concentrations occurred at each river mouth (Figure 4). On a comparative basis, highest cell concentrations were associated with the Brisbane River system and lowest cell concentrations with the Caboolture River system.

An east-west gradient in theoretical photic zone depth (calculated from secchi depth) was a consistent characteristic of Moreton Bay during the bloom period. Plume sites at each of the three rivers were generally characterised by a photic zone depth approximately equal to the water column depth (Figure 5). In contrast, light attenuation at the river mouths was high, and photic zones were generally confined to the upper third of the water column during the month following the flood. A gradual increase in photic zone depth over time is evident at both the Caboolture and Logan River mouth sites. Photic zone depth at the mouth of the Brisbane River was approximately 50% of the water column for the entire survey period. Oceanic sites in eastern Moreton Bay were generally clear to the bottom, except within two to three weeks of the flood in the southern Bay.

Five weeks after the cessation of rainfall, cell concentrations decreased to less than 1.0×10^6 cells/L in the central and northern Bay (Figure 6), however, relatively high concentrations of diatoms, $> 1.0 \times 10^6$ cells/L, remained in the southern Bay in the vicinity of the Logan River plume. Although community composition within the entire Bay remained diatom dominated, flagellates comprised up to 45% of central and northern phytoplankton communities.

A north-south gradient in phytoplankton community composition in Moreton Bay was evident four months after the flood event (Figure 7). The northern Bay communities were dominated by flagellates at very low cell concentrations, $< 4.1 \times 10^4$ cells/L. Diatoms comprised the greatest proportion of central and southern Bay communities. The east-west trend in phytoplankton biomass evident during earlier stages of the bloom when highest cell concentrations were found in the river plumes (Figure 4) was not evident during this survey; cell concentration maxima were found at the mouth of the Caboolture River, in the plume of the Brisbane River and at the oceanic site associated with the Logan River.

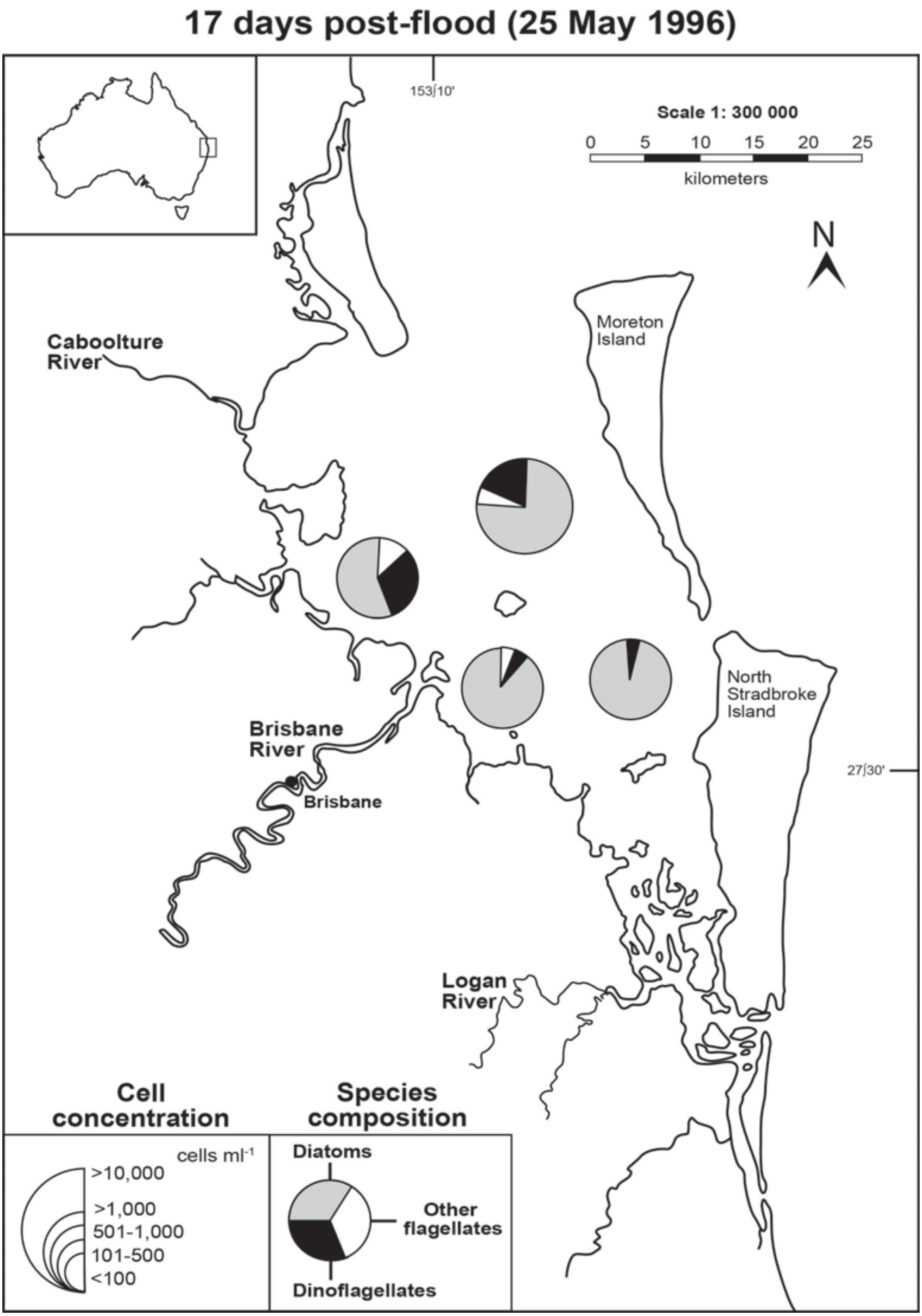

Figure 2. Phytoplankton concentration and community composition in Moreton Bay 17 days after cessation of rainfall on 8th May 1996.

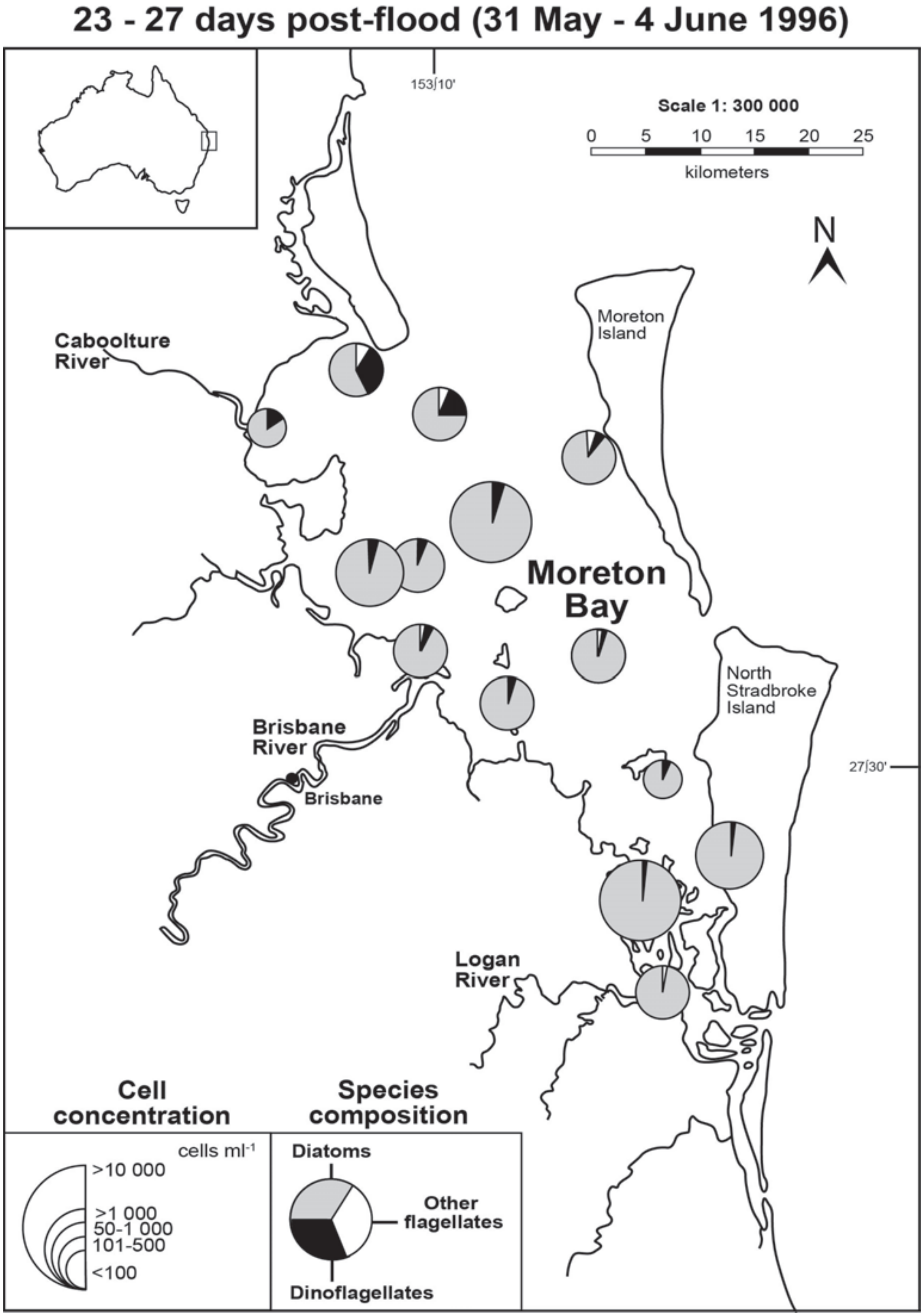

Figure 3. Phytoplankton concentration and community composition in Moreton Bay 23 - 27 days after cessation of rainfall on 8th May 1996.

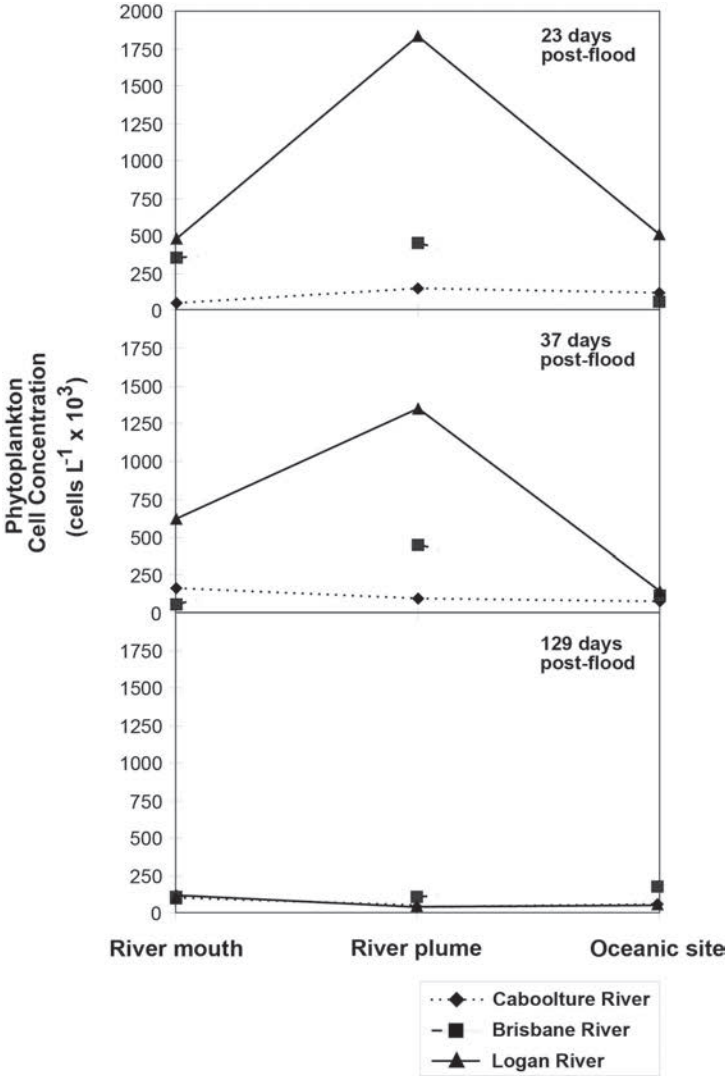

Figure 4. Total phytoplankton abundance (cells/L x 10^3) at the river, plume and oceanic influenced sites of the Caboolture, Brisbane and Logan Rivers during the course of the bloom.

No evidence was found that the use of Lugols preservative resulted in cell lysis and underestimation or misidentification of phytoplankton species present during this survey. Concentrations of dinoflagellates and other flagellates were highest in samples preserved with Lugols preservative compared with either formalin or glutaraldehyde at both of the sites examined.

Discussion

Diatom blooms in Australian coastal waters occur after events which introduce significant concentrations of inorganic nutrients to coastal waters (Jeffrey & Hallegraeff, 1990), and include nutrient enrichment from tidal upwelling (Andrews & Gentien, 1982), seaward transport of deep slope water by current and eddies (Rochford, 1975; 1984; Tranter *et al.,* 1986), interactions

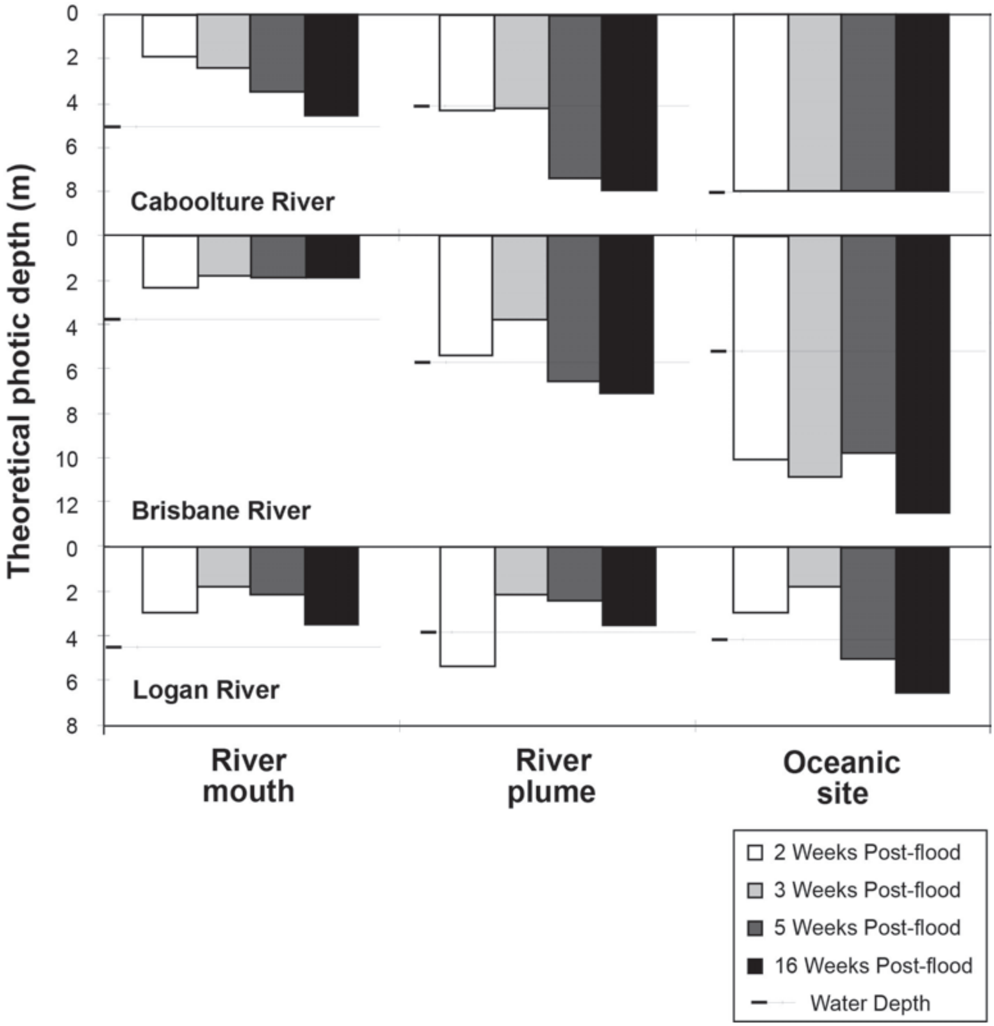

Figure 5. Theoretical photic depths at the river, plume and oceanic influenced sites of the Caboolture, Brisbane and Logan Rivers during the sampling period. At the oceanic water site associated with the Caboolture River the photic depth was equivalent to water column depth as the secchi disk was visible at the bottom. Theoretical photic depth was calculated from secchi depth measurements according to Cloern (1987).

between the sea bottom and internal waves (Holloway *et al.,* 1985), sedimentary nutrient input (Forbes, 1984) and enrichment resulting from oceanographic fronts (Harris *et al.,* 1987). Interannual variations in phytoplankton biomass have been related to annual precipitation and river discharge in North American coastal systems, e.g. San Francisco Bay (Cloern *et al.,* 1985), Chesapeake Bay (Schubel & Pritchard, 1986), Neuse River estuary (Mallin *et al.,* 1993) and the Potomac River (Bennett *et al.,* 1986), and diatom blooms have been reported to occur after anomalous rainfall and runoff events (Honjo, 1974; Garcia-Soto *et al.,* 1990; Mallin *et al.,* 1993). This is the first report of a diatom bloom in Australian coastal waters resulting from nutrient input derived from rainfall and river runoff after an anomalous flood event.

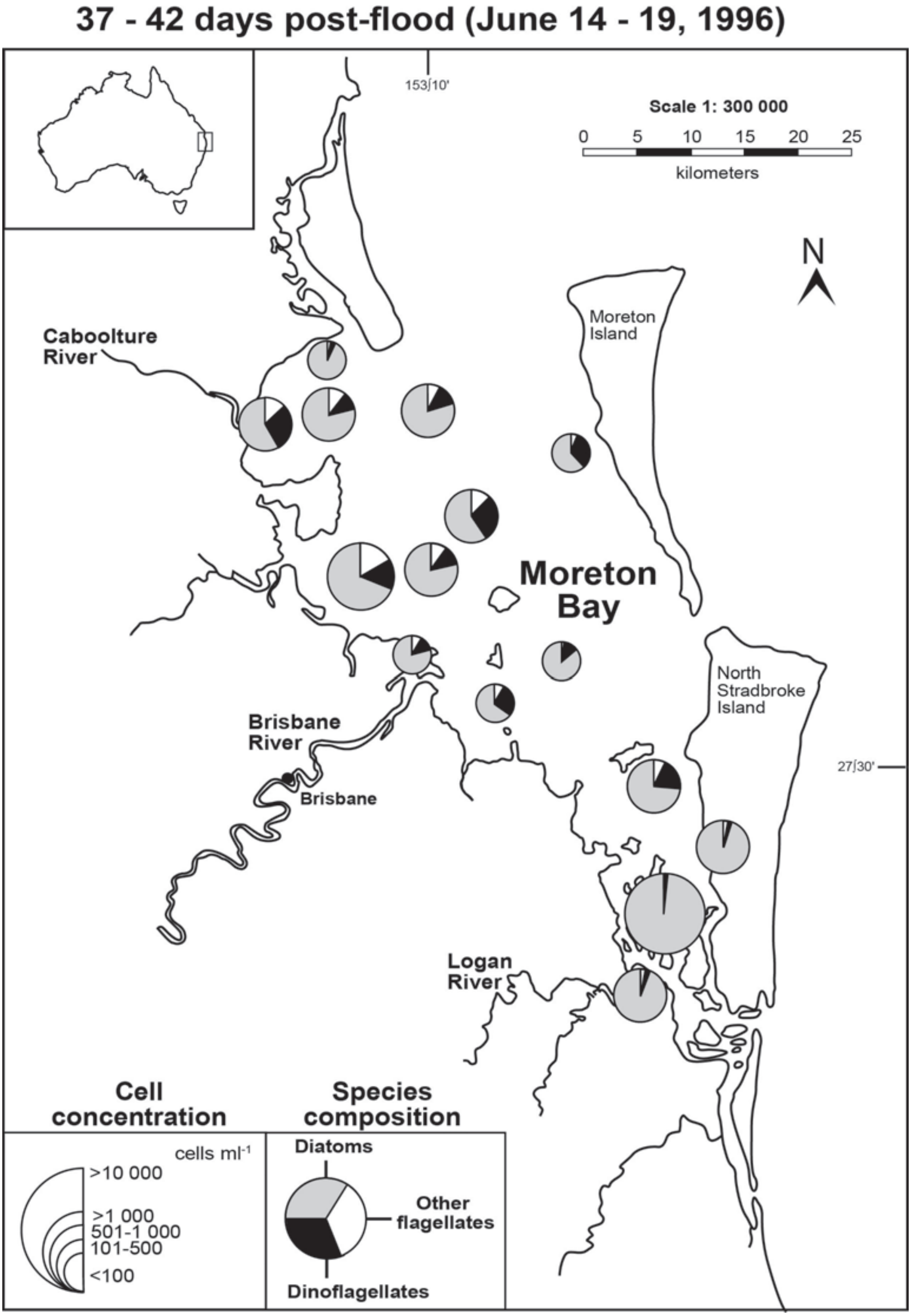

Figure 6. Phytoplankton concentration and community composition in Moreton Bay 37 - 42 days after cessation of rainfall on 8th May 1996.

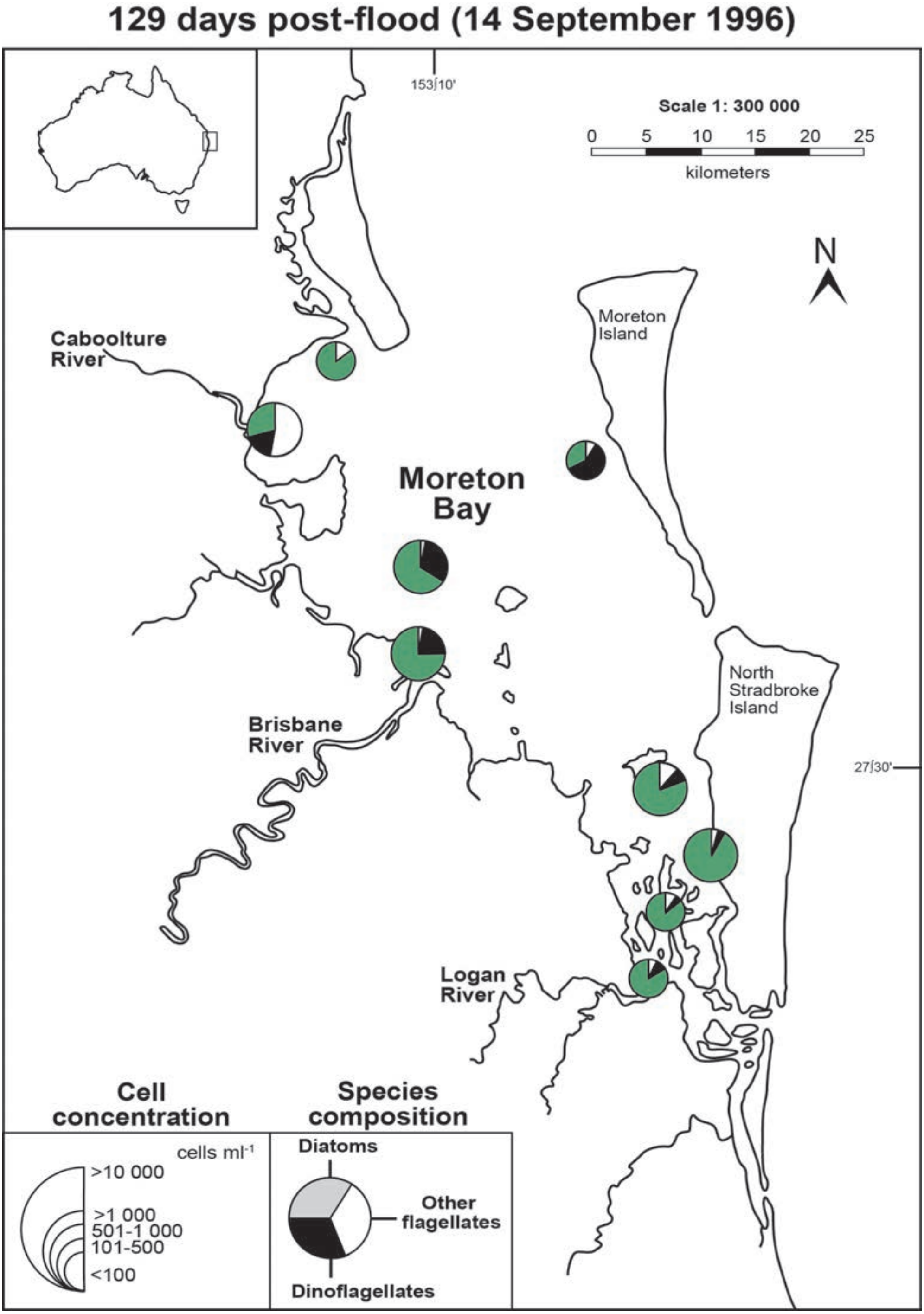

Figure 7. Phytoplankton concentration and community composition in Moreton Bay 129 days after cessation of rainfall on 8th May 1996.

The typical phytoplankton successional pattern in east Australian coastal waters includes a spring diatom bloom in October in which small chain-forming diatoms are replaced by larger centric diatoms, with all diatoms subsequently replaced with dinoflagellates (Dakin & Colefax, 1940: Jeffrey & Carpenter, 1974; Jeffrey & Hallegraeff, 1974; Hallegraeff & Reid, 1986). This pattern is typical of most coastal waters examined (Smayda, 1980). The succession of phytoplankton species during the Moreton Bay bloom differed from this pattern considerably. Large diatoms and dinoflagellates were initially co-dominant within the population, especially within the northern Bay. These populations were rapidly replaced by a mixture of large and small diatoms (*S. costatum*, *R. setigera* and *Pseudonitzschia* sp.) which dominated the bloom until replaced in late June by dinoflagellates within the northern Bay. This difference in succession of phytoplankton taxa during the course of the bloom could result from either the quantity of nutrients delivered to the Bay, the process of nutrient delivery to the Bay and/or the large suspended sediment load which accompanied nutrient input from rivers during the flood. River input delivered large quantities of inorganic nutrients rapidly to the Bay immediately after the flood event. Combined nutrient input from the Caboolture, Brisbane and Logan Rivers during a single day of the flood event included 4.42×10^4 kg of phosphorus and 2.92×10^5 kg of nitrogen (Davies & Eyre, this volume). The delivery of a large amount of inorganic nutrients into the Bay over a short time period could have eliminated competition for nutrients as a basis for phytoplankton succession during this period and allowed species with high growth rates to dominate abundance.

High concentrations of both total suspended solids (TSS) and dissolved organic carbon have been shown to reduce light penetration and prevent the utilisation of available nutrients by phytoplankton in some coastal marine systems (Randall & Day, 1981; Cloern, 1987; Doering *et al.,* 1994). Inorganic nutrient input from the three large rivers which entered the western side of the Bay were accompanied by a large input of total suspended solids (Davies & Eyre, this volume). High TSS resulted in reduced light penetration at the river mouth sites, restricting the photic zone to the top two-thirds of the water column. Photic zone depths were thus considerably shallower than water column depth at the river mouth of all three major rivers for the duration of the bloom. Reduced light penetration inhibited the ability of diatoms to utilise inorganic nutrients present at the start of the bloom at the river mouth sites, while dinoflagellate growth may not have been limited by light due to their motility and ability to vertically migrate to suitable light environments (Spector, 1984). Based on their lack of pigmentation, a significant proportion of dinoflagellates at the start of the bloom were probably heterotrophic, thus highlighting the role of light penetration at this stage of the bloom.

East-west trends in water quality parameters (Abal & Dennison, 1996) and biological indicators (O'Donohue *et al.,* 1996) are a consistent feature of Moreton Bay. The east-west trend in phytoplankton cell concentration evident throughout the bloom strongly implicates the Caboolture, Brisbane and Logan Rivers as the source of nutrients for the bloom. Cell concentrations were low at the river mouth due to light attenuation resulting from high concentrations of total suspended solids which were introduced to the Bay during the event. At the oceanic sites, TSS had decreased sufficiently so that ambient phytoplankton assemblages had sufficient light to utilise inorganic nutrients from freshwater sources. Clean water sites on the eastern side of the Bay supported low phytoplankton concentrations relative to either the river or plume sites. This suggests that either insufficient nutrients were available to allow an increase in biomass or that phytoplankton assemblages endemic to the relatively oligotrophic eastern Bay were physiologically incapable of utilising additional inorganic nutrient input.

The photic zone depth at the Caboolture and Logan River mouth sites and all three plume sites generally increased over the course of the bloom. The notable exception to this occurred at the mouth of the Brisbane River where depth of the photic zone remained relatively constant over the entire study period at approximately 50% of the water column depth. This suggests that potential light limitation of algal growth at the mouth of the Brisbane River is a constant feature of this system, even during non-bloom conditions. This area is the site of the main port of Brisbane and the Luggage Point sewage outfall as well as a centre of recreational use. Any combination of these features, in conjunction with the high levels of TSS input from the Brisbane River itself could contribute to water quality conditions that result in high light attenuation at this site.

A north-south gradient in phytoplankton community composition, with a mixture of diatoms, dinoflagellates and other flagellates in the northern Bay and an almost exclusively diatom community in the southern Bay were consistent features of Moreton Bay both during the bloom and afterwards. This gradient may be due to a variety of features, including water depth, flushing rates, residence times and the nature of riverine input into the Bay. Under favourable conditions, diatoms exhibit faster growth rates than dinoflagellates (Chan, 1978; 1980; Parsons *et al.,* 1978; Thomas *et al.,* 1978). High growth rates, in combination with the shallow, enclosed nature of southern Moreton Bay may result in sufficient available light and nutrients to allow diatoms to outcompete other phytoplankton groups within this region. The middle and northern Bay, generally deeper and characterised by increased oceanic flushing, may favour slower growing, motile dinoflagellates.

Longhurst & Hardy (1987) report that diatoms can be important in tropical estuaries, often dominating community composition (e.g. West Africa – Bainbridge, 1960; Trinidad – Bacon, 1971; Costa Rica – Hargraves & Viquez, 1985). The most frequently observed diatom genera are *Coscinodiscus*, *Chaetoceros*, *Rhizosolenia* and *Skeletonema* (Longhurst & Hardy, 1987). *S. costatum* and *R. setigera* were two of the most numerically abundant diatoms during the Moreton Bay bloom. *S. costatum* has not been previously reported from Moreton Bay (Wood, 1964) although it is common in other Australian estuaries (Wood, 1964; Hallegraeff & Reid, 1986; Hallegraeff & Jeffrey, 1993). *S. costatum* is a common, often dominant, species found after rainfall and river input in temperate coastal waters (Honjo, 1975; Sanders *et al.,* 1987; Garcia-Soto, 1990). It is able to tolerate a wide range of salinities (Marshall, 1982; Sakshaug & Olsen, 1986), adapts readily to changes in nutrient concentrations (Sanders *et al.,* 1987), and exhibits rapid growth rates (Smayda, 1973; Cosper, 1982). *R. setigera* is a common species within Moreton Bay; Wood (1964) reported it to be present in 55% of phytoplankton samples from the Bay.

Under normal flow conditions, estuaries act as filters, reducing TSS and nutrient concentrations (Schubel & Kennedy, 1984). In wet conditions, nutrient input can exceed the assimilatory capacity of the biota within an estuary, resulting in nutrient export from the system and/or the development of large phytoplankton bloom as nutrients are incorporated into phytoplankton biomass. Seagrass, which accounts for 17% of primary production in Moreton Bay (O'Donohue *et al.,* 1997), has been suggested to function in the removal of inorganic nutrients from estuaries (Short & Short, 1984). Reductions in seagrass depth range which occurred after the flood event (Horrocks, pers. comm.) do not support such a function for seagrass during the flood event. The progression and generally autotrophic community composition of the bloom which co-occurred in Moreton Bay suggests that a large portion of the nitrogen which entered The Bay during the flood converted to phytoplankton biomass during the bloom. The maximum observed chlorophyll *a* concentration during the bloom was 15 μg/L. Assuming a C:Chl-*a* ratio of 30 (Ayukai, 1992; Gallegos & Vant, 1996) and the Redfield Ratio (106:16:1, C:N:P), 63.7 mg nitrogen m^{-3} was required to support this biomass. Based upon a Moreton Bay volume of 1 400 km^2 (Stephans, 1992) and an average photic depth of 2 m, 1.5 x 10^5 kg of nitrogen was required to support phytoplankton biomass at the height of the bloom. Estimates of the combined nitrogen input from the Logan, Brisbane and Caboolture rivers during the flood event was 2.92 x 10^5 kg of nitrogen/day (Davies & Eyre, this volume). Thus, approximately 50 to 100% of the flood input of nitrogen to Moreton Bay were incorporated into phytoplankton bloom biomass. The eventual fate of the bloom (e.g. sedimentation, loss to grazing, export from Moreton Bay) is unknown.

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Assessing the Impact of a Flood Event on Moreton Bay Using Marine Plants as Bioindicators of Water Quality

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Abstract

During May 1996, the Moreton Bay catchment experienced a one-in-twenty year flood. The impact of this flood event on Moreton Bay was examined using phytoplankton productivity bioassays, macroalgal metabolic profiling, seagrass depth ranges and vegetation isotopic signatures. These bioindicators were assessed before and after the flood at three river mouth sites, three sites near the edge of the river plumes, and three sites on the eastern side of Moreton Bay. ^{15}N stable isotope analyses of mangrove and seagrass samples prior to the flood event demonstrated elevated sewage N incorporation into tissue by plants on the western side of Moreton Bay. Macroalgae deployed during summer prior to the flood had the greatest increase in tissue %N at Brisbane and Logan River mouths, whilst samples deployed on the eastern side of Moreton Bay showed the least increase in their tissue N content. Macroalgal bioindicators reflected a breakdown in ambient bioavailable nutrient gradients within the Bay one month after the flood. Similar surveys conducted four months after the flood demonstrated that ambient dissolved N and sewage N gradients were re-establishing. Phytoplankton productivity was stimulated in response to short term (15 h) nutrient additions during the dry summer months. Short term phytoplankton bioassays conducted one and four months after the May 1996 flood event showed minimal productivity stimulation at the sites on the western side of the Bay, suggesting that phytoplankton nutrient requirements were saturated in the short term. However, increases in phytoplankton biomass with long term (seven day) bioassays conducted at the same time suggest that natural phytoplankton communities still retain the potential to form blooms at the river mouths. Seagrass depth ranges were considerably reduced after the flood event with the depth penetration of *Zostera capricorni* decreasing by nearly 40% from December 1995 to June 1996. Recent surveys have shown that this reduction was only temporary.

Marine plant bioindicators reflected (a) ambient water quality gradients east/west and north/south in Moreton Bay, (b) seasonal variations in plant responses, and (c) an impact of the flood event on plant communities with a subsequent recovery, suggesting that the impact on these ecosystem components appears to be transient.

Introduction

During May 1996 the Moreton Bay catchment experienced a large flood event. The impact of this flood event on the flora of Moreton Bay was examined using marine plants as bioindicators of water quality changes. Water quality is most often defined by a combination of physical, chemical and biological variables (ANZEC, 1992; DoE, 1992). The quality of water can, in a practical sense, be defined by the impact that a given combination of these parameters has on aquatic flora and fauna. These impacts, however, are not necessarily generic, are not presently well defined within Australian systems, and will vary dependent on a number of factors including site, plant species present and season. Studying the responses of plant species or communities as a means of assessing water quality is a powerful tool, as: (a) it is impacts of water quality on aquatic biota that managers are often trying to assess; (b) it avoids the

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
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necessity of interpreting water quality values in terms of their impact on plants; and (c) plant communities will reflect integrated 'assessment' of water quality parameters that: (i) may act synergistically, where the combined impact is more severe when alone (e.g. temperature and light effects on plant respiration); (ii) may vary sporadically and outside an established sampling schedule; and (iii) may be below the detection limits of standard sampling and analysis capabilities.

The Marine Botany group at The University of Queensland have developed four approaches to using marine plants as indicators of water quality. These four methods include phytoplankton productivity bioassays, macroalgal metabolic profiling, seagrass depth ranges and vegetation isotopic signatures (Figure 1). By examining the responses of phytoplankton to artificially increased nutrient concentrations, it is possible to determine the nutrient most likely to limit growth, and by recording the degree of response, the relative degree of nutrient limitation (Fisher *et al.,* 1992, O'Donohue & Dennison, 1997). During periods of elevated nitrogen in the environment, some plants may store nutrients as tissue nitrogen, amino acids and pigments (Jones *et al.,* 1996, Lyngby *et al.,* 1990). Macroalgal tissue nutrient concentrations have been shown to reflect the bio-availability of water column nutrients (Horrocks *et al.,* 1995). The amount of light available to seagrasses significantly affects the water depth to which these submersed plants can grow, and seagrass depth penetration is a sensitive integrator of water quality parameters (Dennison *et al.,* 1993; Abal & Dennison, 1996). The delta ^{15}N ($\delta^{15}N$) of plant material is the relative abundance of heavy (15 neutron) N isotope to a lighter (14 neutron) N isotope within the plant tissue (Peterson & Fry, 1987). This $\delta^{15}N$ signature in the plant tissue can be compared to the $\delta^{15}N$ signatures of possible sources that may be supplying nitrogen to the plants (McClelland *et al.,* 1997).

Study site

Phytoplankton, macroalgae and seagrass bioindicators were used to survey water quality both before and after the flood event. Data from three sample periods are presented here. The first survey was conducted during December 1995 (summer) approximately 4.5 months before the flood. Two further surveys were conducted approximately one month (June 1996) and four months (September) after rainfall had ceased. Phytoplankton and macroalgal bioassays were employed at three river mouth sites (Caboolture, Brisbane and Logan River mouths at sites 1, 4 and 7 respectively), three sites near the edge of the river plumes (Caboolture, Brisbane and Logan plumes at sites 2, 5 and 8 respectively), and three oceanic influenced sites (sites 3, 6 and 9) on the eastern side of Moreton Bay (Figure 2). Seagrass depth ranges were determined at three sites in southern Moreton Bay (sites 6, 8, and 9), and vegetation isotopic signatures were assessed prior to the flood event at various estuarine and open water sites around Moreton Bay.

Methods

Two phytoplankton bioassays were employed during this study. Short term phytoplankton productivity bioassays used the addition of nutrients to 1 L water samples, followed by incubation of the water samples under low light for 15 h (O'Donohue & Dennison, 1997). At the end of the incubation period 100 mL subsamples were incubated with ^{14}C to measure phytoplankton productivity. Results were expressed as the increase in productivity relative to the control sample to which no nutrients were added. The long term bioassays involved incubating 4 L water samples with added nutrients, and examining the daily changes in phytoplankton biomass measured as changes in fluorescence. Results were standardised as

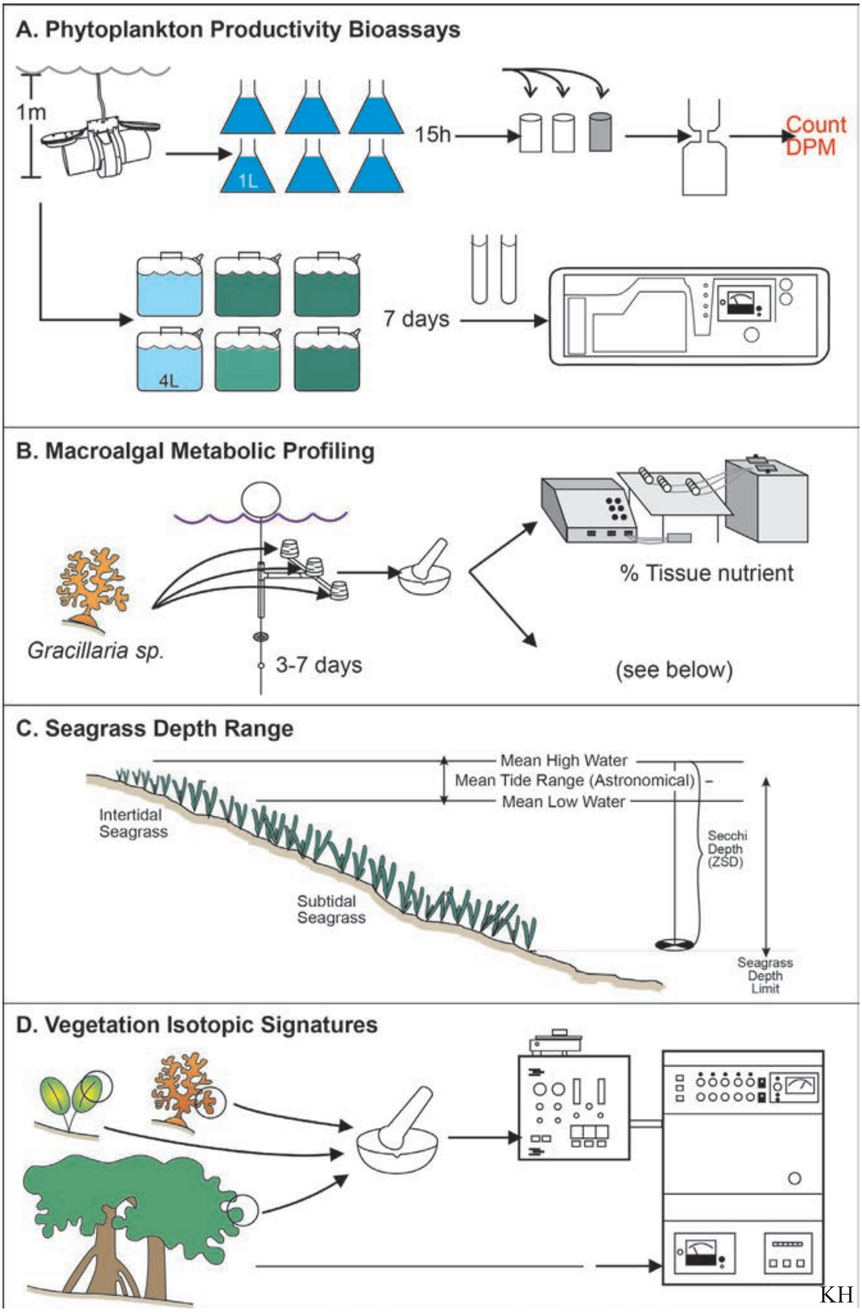
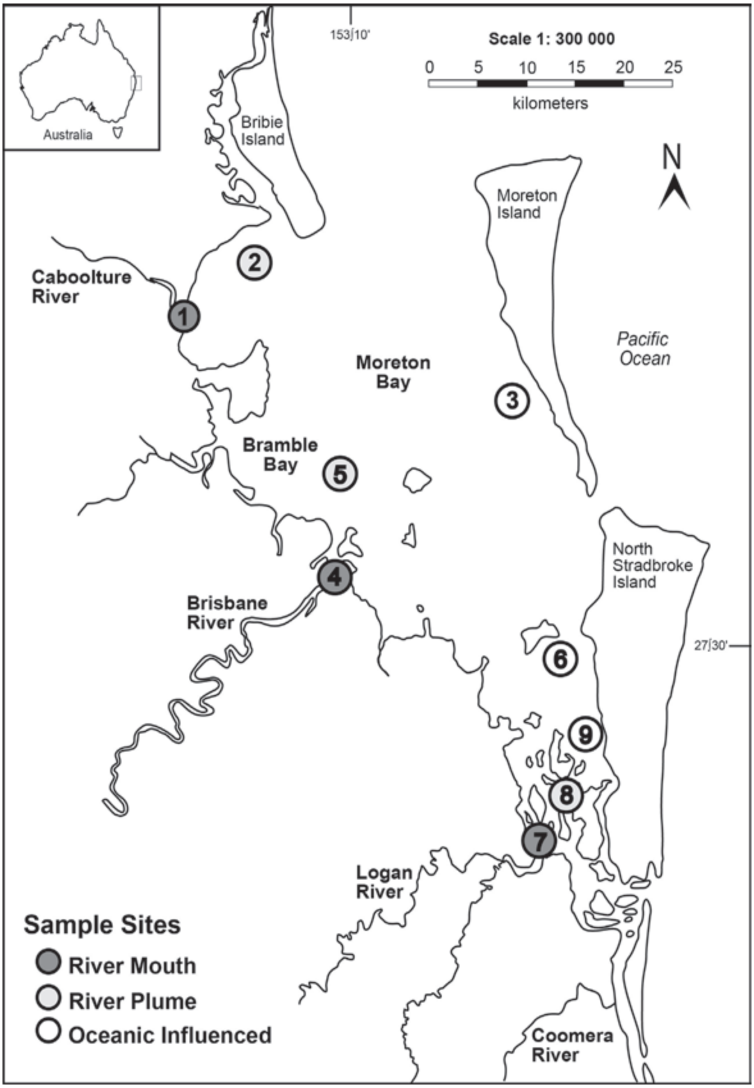


Figure 1. Four approaches using marine plant bioindicators to determine aspects of water quality. **A.** Phytoplankton responses to nutrient pulses are assessed using productivity and biomass changes in water samples augmented with elevated nutrients. **B.** Bioavailable N and sewage N are determined using nutrient depleted macroalgal samples which are deployed *in situ* for three to seven days and then analysed for tissue %N and $\delta^{15}N$ content. **C.** The affect of water quality on seagrass depth penetration is determined using a staff and level. **D.** The influence of sewage N on plant nutrients requirements is determined using stable isotope ratios of marine plants.



relative fluorescence units, which may be calibrated to quantitatively indicate changes in phytoplankton chlorophyll *a*.

In the macroalgal metabolic profiling bioassay, the red alga *Gracillaria* sp. was collected from the field and incubated in low nutrient seawater for several weeks to deplete tissue nutrient reserves. After this period, triplicate 5 g (wet weight) samples of the algae were deployed in clear plastic chambers at approximately one half of the secchi depth using a fixed base and a floating mooring buoy (Horrocks *et al.,* 1995; Jones *et al.,* 1996). Water exchange between the algae in the containers and surrounding water was achieved by drilling holes in all sides of the plastic chambers. At the end of a three day deployment, algae samples were collected and analysed for changes in tissue nitrogen content and stable isotope ratios. Tissue N and stable isotope values were expressed as a percentage of the highest recorded value.

The depth to which seagrasses grow (seagrass depth ranges) was determined using a staff, surveying level and transect line (Abal & Dennison, 1996). The upper edge of the seagrass bed (often exposed during low tides) was marked as the start point, and the point at which continuous seagrass cover disappears was determined to be the end point. The vertical distance between these two points was taken as the seagrass depth range.

Representative algal, seagrass and mangrove tissues were collected from various sites around Moreton Bay and, together with samples of *Gracillaria* deployed after the flood, analysed for stable isotope ratios. Results are presented as ppt $\delta^{15}N$ values which represent the ratio of ^{15}N to ^{14}N in the samples compared to an atmospheric standard (Peterson & Fry, 1987).

Results and Discussion

Preflood surveys of seagrass and mangrove $\delta^{15}N$ values have revealed a strong east/west and south/north gradient, with elevated values in the western and southern portions of Moreton Bay (Figure 3). Elevated values have been attributed to the increased use of sewage-derived dissolved inorganic nitrogen (DIN) by plants on the western Bay. Low $\delta^{15}N$ values on the eastern side suggest a greater reliance by the plants on biologically fixed N (N_2 fixation), a microbial process that results in little discrimination between the two N isotopes and therefore a $\delta^{15}N$ value of the fixed N close to 0. Stable isotope values provide a means of assessing the sphere of sewage influence within Moreton Bay, consequently stable isotope analyses were performed on macroalgal samples deployed around Moreton Bay after the flood.

Macroalgae deployed during the summer preceding the flood had the greatest relative tissue %N at Brisbane and Logan River mouths (Figure 4). There were smaller responses within the river plumes, while samples deployed on the eastern side of Moreton Bay showed up to 40% less tissue N content when compared to the river mouth sites. This gradient in response is consistent with the ambient nutrient gradients within central and southern Moreton Bay, where highest DIN is measured at river mouths in the western bay, and the lowest concentrations are on the eastern side of Moreton Bay (Moss *et al.*, 1992). Macroalgal responses in the northern Bay were smaller around the Caboolture River and plume, suggesting that nutrient loads from this river were not as high as those from the two southern river systems. One month after the flood event, there was no discernable gradient in tissue nutrient response observed at any of the three river transects. Similarly, $\delta^{15}N$ values of the deployed algae showed no trend, however, the highest value was recorded at the mouth of the Caboolture River, suggesting a higher proportion of sewage N at this site during the incubation period. Four months after the flood event, both tissue %N and $\delta^{15}N$ along the Logan River transect showed near pre-flood gradients in DIN and sewage N from western to eastern Moreton Bay. In the northern Bay, the highest values in tissue N were recorded at the river mouth, with low values recorded in the plume and at the oceanic influenced site. A nutrient plume from the Caboolture River was not detected in Deception Bay (Site 2). Macroalgae deployed within the Brisbane River plume contained the greatest tissue %N and $\delta^{15}N$, suggesting that the sewage nutrient discharges at the mouth of the Brisbane River were influencing plant processes. Results of samples deployed at eastern Moreton Bay (Site 6), however, were similar to site 2 and the other oceanic influenced sites (3 and 9), with low %N and $\delta^{15}N$ values indicating relatively minimal sewage nutrient impact.

Phytoplankton biomass was stimulated in response to nutrient additions in the long term phytoplankton bioassays (Figure 5A). As with the $\delta^{15}N$ and macroalgal bioassays, there was an increasing east/west gradient in phytoplankton response within Moreton Bay. The greatest biomass developed in the shortest time at the river mouth sites in the western Bay. During all surveys, phytoplankton biomass only increased in response to N (NO_3 and NH_4) and plus all

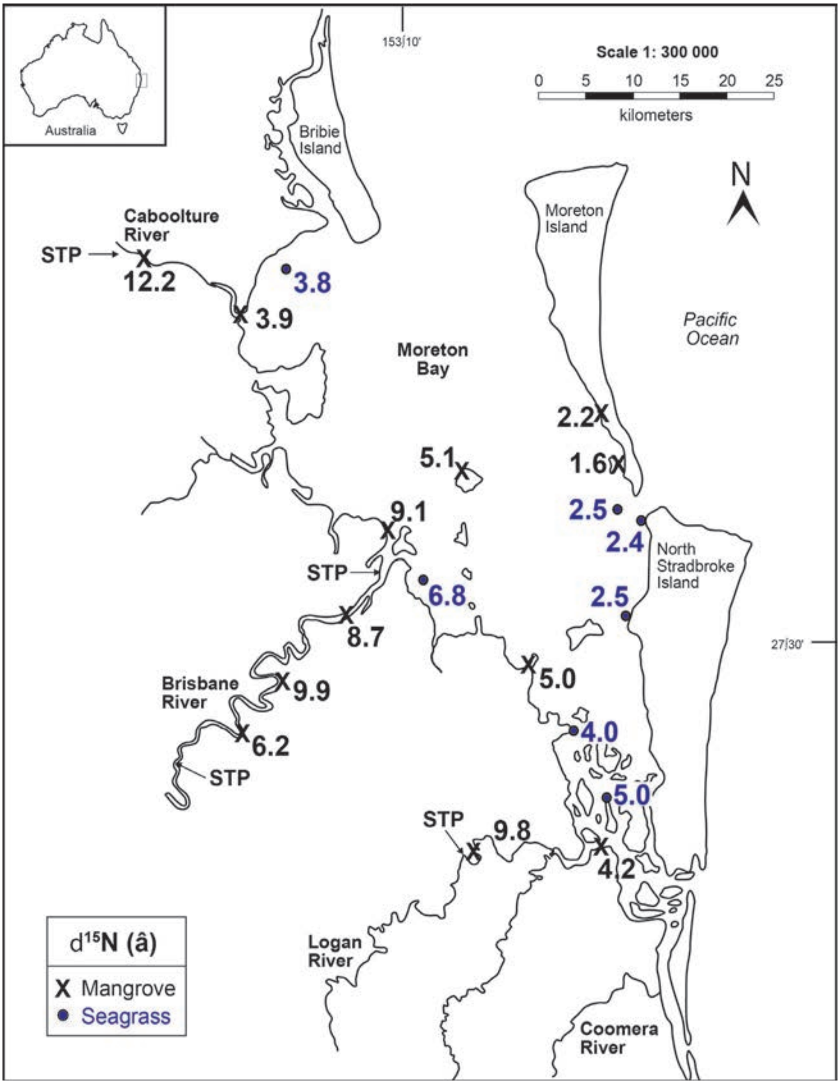


Figure 3. Map of Moreton Bay in southeast Queensland showing N stable isotope ratios ($\delta^{15}N$) of mangrove and seagrass.

additions, suggesting that P alone was not limiting increases in phytoplankton biomass. In some instances, (e.g. Caboolture River mouth), the addition of all nutrients (N+P) together gave a greater response than the addition of N alone, suggesting a synergistic affect of N and P on growth. In most cases, there was a lag period after addition of the nutrients before an increase in phytoplankton biomass was observed. This lag time can be used as an index of the capacity of phytoplankton in those regions to bloom, with shorter lag times suggestive of waters with a greater propensity for phytoplankton blooms. One month after the flood event, the biomass of phytoplankton from the river mouths increased within the first day of incubation. Initially, all treatments including the control rapidly increased phytoplankton biomass at the same rate. These responses indicate that (i) a common factor had been limiting phytoplankton growth within these sites, and (ii) this factor was not nutrient availability as all assays increased equally.

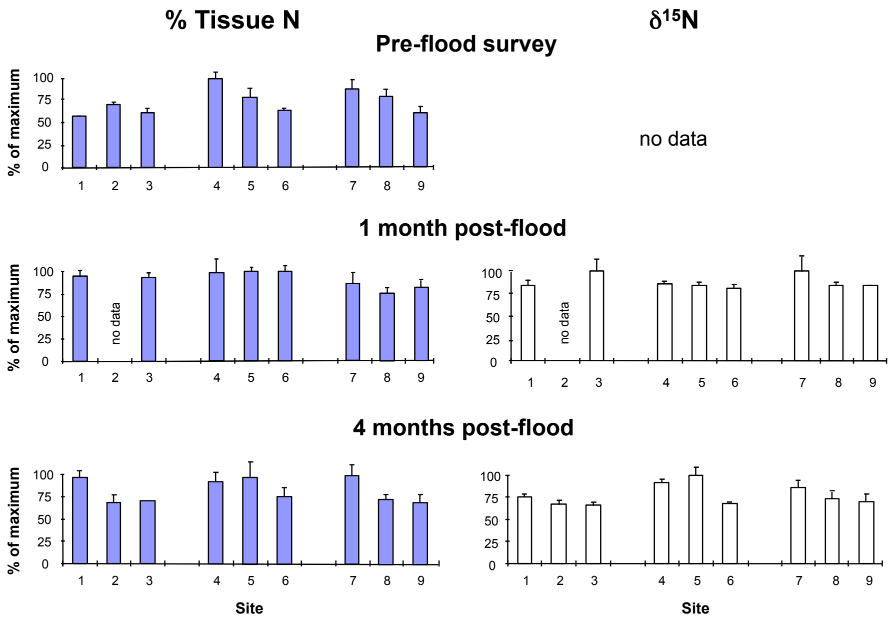
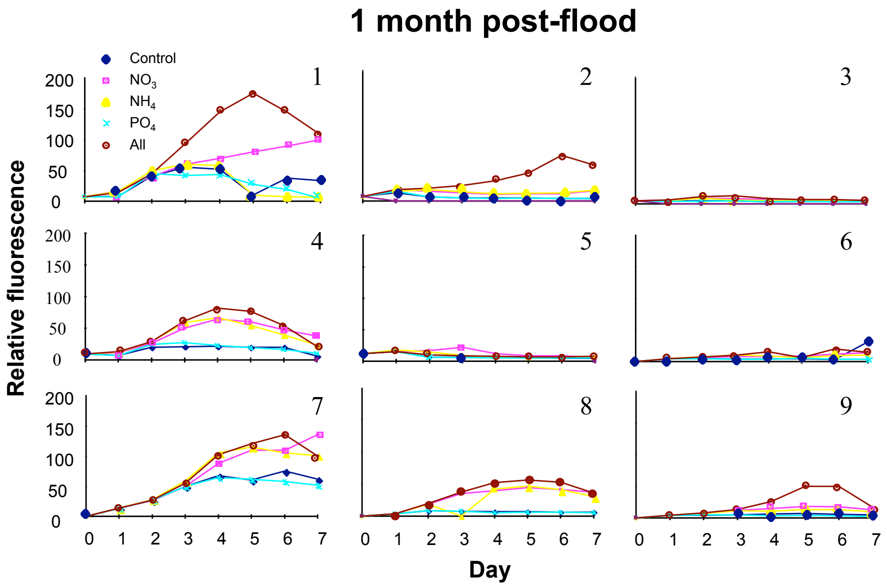


Figure 4. Tissue percent N and stable isotope ratios ($\delta^{15}N$) values of *Gracillaria* sp. deployed at nine sites within Moreton Bay. *Gracillaria* were deployed *in situ* during December 1995 (pre-flood survey), and one and four months after the May 1996 flood event. Data are presented as percentage of maximum values ±SE.

Light availability has been shown to limit summer phytoplankton productivity in the turbid Logan River estuary (O'Donohue & Dennison, 1997), and light availability may therefore limit phytoplankton productivity within other turbid rivers in Moreton Bay. At the beginning of the long term incubations, relative light availability to phytoplankton was increased as light attenuation within the incubation volume (4 L) was minimal. Therefore, at the start of the assay, light availability was optimal, and this, coupled with high riverine nutrient concentrations, resulted in increases in phytoplankton biomass. This elevated phytoplankton growth, however, rapidly consumed the available nutrients, and growth quickly became limited by nutrient availability. Hence, during the one month post-flood assay, light availability was the primary factor limiting phytoplankton biomass, however, there was only sufficient N available for approximately two days' growth at all three sites. Phytoplankton biomass production at the oceanic influenced sites was apparently not limited by N or P availability as there was no increase in biomass with added nutrients one month after the flood event. Phytoplankton responses at two of the river mouth sites varied when assayed four months after the flood event (Figure 5B). Responses at the Brisbane River mouth were similar to those seen one month after the flood, indicating that light limitation may be a constant factor in controlling phytoplankton growth at this site. There was little change in the photic depth at the Brisbane River compared to the other river mouths (Heil *et al.,* this volume), supporting the contention that phytoplankton growth is light limited at the mouth of the Brisbane River. In contrast to the Brisbane River mouth site, there was a three and four day lag in phytoplankton biomass increases at the Logan and Caboolture River mouth sites, respectively. Additionally, maximal biomass at both sites was less than that measured during the first post-flood survey. There



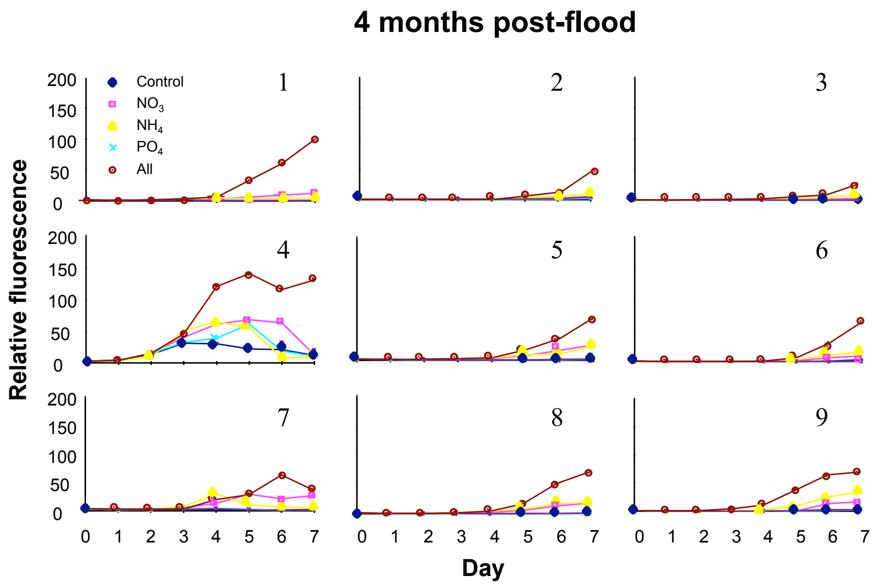


Figure 5. Changes in phytoplankton fluorescence in water samples from 9 sites (1-9) in Moreton Bay incubated with elevated nutrient concentrations. Six 4 L water samples from each site were incubated under 50% ambient light with: 200 µM nitrate (NO_3), 30 µM ammonium (NH_4), 20 µM phosphate (PO_4), a combination of all of the above (plus all), or without added nutrients (Control). Water samples were collected one (5A) and four months (5B) after the flood event, and results are in relative fluorescence units.

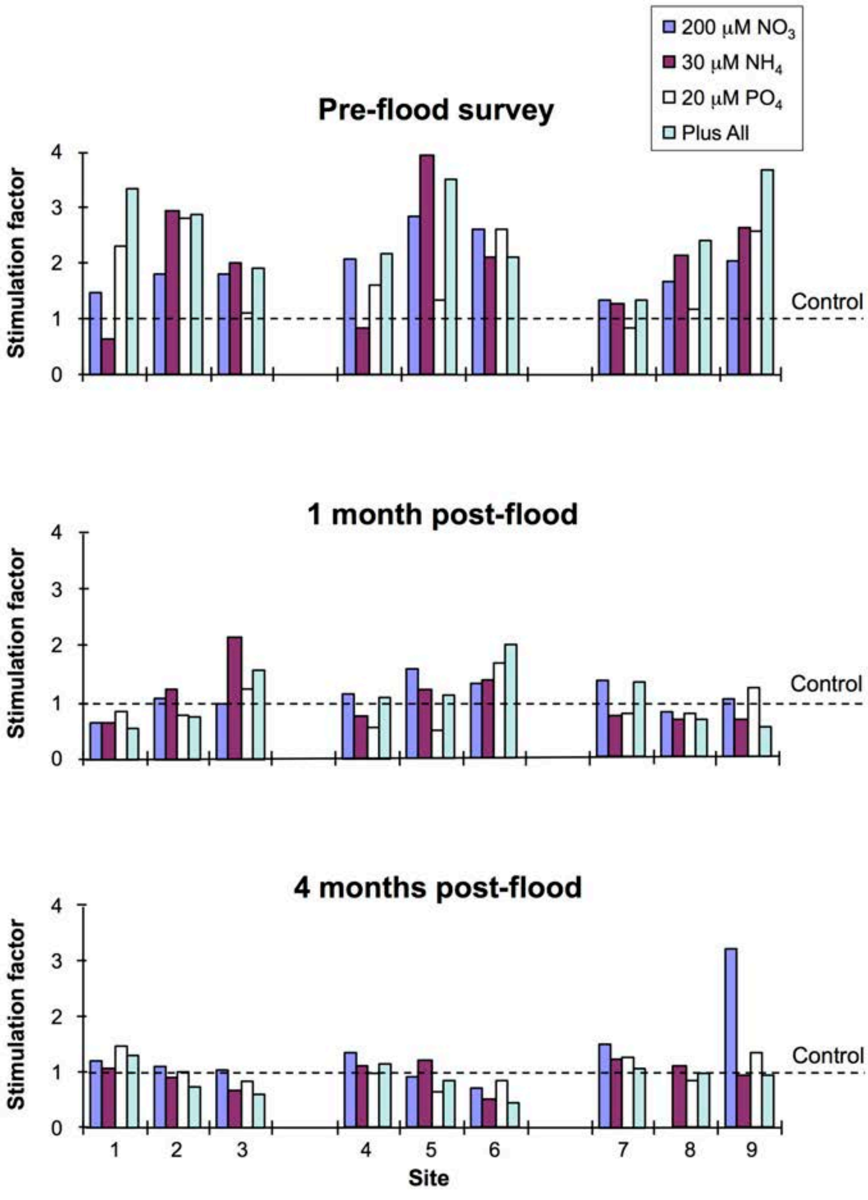


Figure 6. Response of phytoplankton from nine sites in Moreton Bay (1-9) to short term incubations with elevated nutrient concentrations. One litre water samples from each site were incubated under low light with: 200 µM nitrate (NO_3), 30 µM ammonium (NH_4), 20 µM phosphate (PO_4), a combination of all of the above (plus all), or without added nutrients (Control). Water samples were collected before (preflood) and at one and four month intervals after the flood event. Results are presented as nutrient stimulation relative to a control with no nutrients added (dashed line).

was also no increase in control biomass at these sites, consequently light availability was not limiting biomass production. Additions of nutrients to oceanic influenced sites (3 and 6) led to increases in phytoplankton biomass, however the lag period prior to biomass increases was the longest of all sites.

Phytoplankton productivity was stimulated in response to short term (15 h) nutrient additions during the dry summer months prior to the flood (Figure 6). One month after the flood event there was minimal stimulation of productivity at the river mouth sites, however, phytoplankton at oceanic influenced sites 3 and 6 responded to nutrient additions. Delivery of high concentrations of nutrients to Moreton Bay during the flood (Davies & Eyre, this volume), may have saturated phytoplankton demand at the near shore sites, and subsequently stimulated phytoplankton growth at the normally oligotrophic eastern Bay waters. The responses of sites 3 and 6 suggest that these populations, normally in oligotrophic waters, had adapted to elevated nutrient concentrations. Four months after the flood event, there was minimal phytoplankton response to nutrient additions, and samples from eastern Moreton Bay (sites 3 and 6) were demonstrating potential inhibition by the artificially elevated nutrient concentrations. Four months after the flood, during the expected recovery period, phytoplankton short term nutrient responses were still minimal compared to the summer responses, suggesting seasonal influences governing phytoplankton growth responses to nutrient pulses. O'Donohue & Dennison (1997) have previously shown a seasonal temperature regulation of phytoplankton growth, even during periods of elevated ambient nutrient concentrations.

Pre-flood seagrass depth ranges in southern Moreton Bay were measured over a two-year period prior to the 1996 flood event, and values were relatively constant at sites 21 km and 26 km from the Logan River mouth (Figure 7). Reduced water quality is likely to be responsible for an ongoing decline in seagrass depth distribution at a site 9 km from the Logan River mouth (Abal & Dennison, 1996). Seagrass depth ranges decreased at all three sites after the flood event, with the depth penetration of *Zostera capricorni* decreasing by up to 40% from

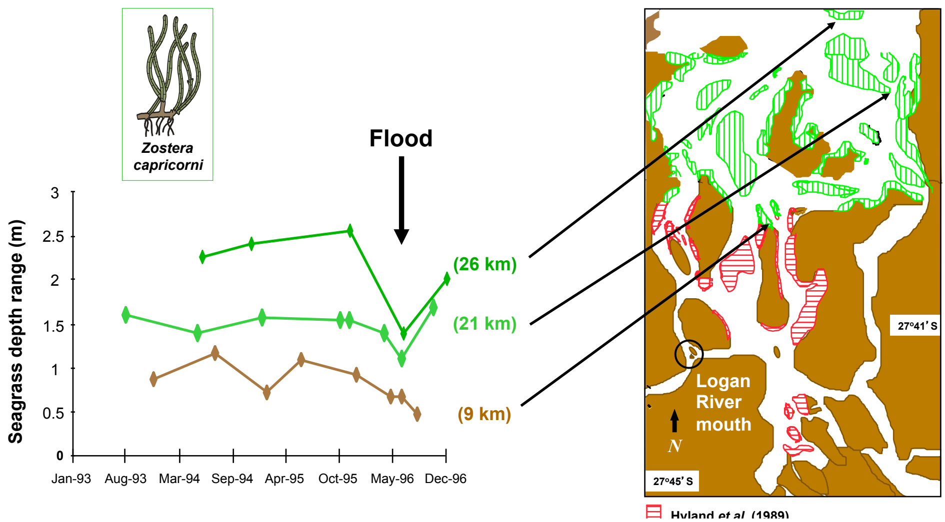


Figure 7. Changes in seagrass depth range at three sites within southern Moreton Bay. Seagrass (*Zostera capricorni*) depth range was determined over a 2.5 year period at 9 km, 21 km and 26 km from the mouth of the Logan River. The occurrence of the 1996 flood event is arrowed.

December 1995 to June 1996. The most likely cause for this decline is a reduction in water quality and specifically water transparency due to increased suspended solid loads with the flood waters (Davies & Eyre*,* this volume). Post-flood surveys have shown that this reduction was only temporary at the two sites distant from the Logan River mouth, and seagrass depth ranges appear to be returning to pre-flood values. Seagrasses nearest the Logan River mouth however, are apparently continuing to decline. This failure to recover, together with the recorded pre-flood decline in depth ranges, suggests a continuing chronic impact on seagrass distributions in this area, separate from the flood event.

Summary

Marine plant bioindicator responses suggest that prior to the flood, ambient nutrient gradients existed west to east, and south to north within Moreton Bay. Stable isotope ratios and macroalgal tissue N and $\delta^{15}N$ values support a strong gradient from west to east in biologically available nutrients in central and southern Moreton Bay. While elevated $\delta^{15}N$ values were recorded in mangrove and seagrass samples near the mouth and plumes of the Logan and Brisbane Rivers, lower values were determined from samples collected at the mouth and plume of the Caboolture River. Additionally, pre-flood macroalgal tissue %N did not exhibit a similar gradient away from the Caboolture River as for the Brisbane and Logan Rivers. That similar gradients were not observed in the north of Moreton Bay suggests that during average river flow periods, the contribution of nutrients from the Caboolture River to Moreton Bay is minimal, and this, combined with proximity to the major flushing zone through northern Moreton Bay (Patterson & Witt, 1992), minimises potential water column nutrient gradients.

The flood event appears to have significantly influenced plant processes within Moreton Bay. All bioindicators detected a flood impact, however the impact appears to have been transient. Macroalgal tissue %N and $\delta^{15}N$, and seagrass depth ranges changed, but appear to be reestablishing within four months of the flood event.

Phytoplankton bioassays determined that productivity was primarily light limited at the river mouths after the flood event, however, potential nutrient impacts on phytoplankton growth were also greatest at these sites. There was no apparent light-limitation at the Caboolture and Logan River mouths four months after the flood event. The flood event was coincident with greater phytoplankton biomass responses, but attributing these responses to the flood event is difficult without a better understanding the seasonal responses of phytoplankton within Moreton Bay. Short term phytoplankton responses also suggest that there may be seasonal variations in biotic parameters underlying the effect of the flood event. Short term nutrient stimulation of phytoplankton productivity was greater in summer prior to the flood. The smaller responses after the flood event may reflect elevated concentrations of bioavailable nutrients in the water column, however, seasonal parameters such as temperature may also modify plant responses, masking or negating the flood effect. A flood event earlier or later in the year may impact marine plants differently.

Marine plant bioindicators allow a direct assessment of the impacts of both natural and anthropogenic influences on the marine environment. Macroalgal bioindicators were used to describe a gradient in biologically available nutrients from west to east and south to north in Moreton Bay. Stable isotope ratios have determined that treated sewage contributes significant amounts of N to these nutrient pools, influencing plant nutrient requirements in western and central Moreton Bay. Additionally, phytoplankton responses after the flood event indicate phytoplankton growth was primarily limited by light availability in the highly turbid river systems on the western side of Moreton Bay. These changes in light availability also contributed to a decrease in seagrass distributions within Moreton Bay.

Finally, the post flood recovery in marine plant responses, measured using the bioindicators, has shown that although the flood did influence plant processes, this event was not immediately responsible for long term environmental degradation.

Acknowledgements

We would like to acknowledge the industrious and enthusiastic field and laboratory assistance from L. King, B. Dudley and M. McLennan. This work was partially supported by monies received from the Brisbane River and Moreton Bay Wastewater Management Study and an ARC-Industry Collaborative Grant to W. Dennison.

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Susceptibility to Flooding of Two Dominant Coral Taxa in Moreton Bay

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Abstract

Flood-induced mortality has been presumed to be the dominant characteristic of the environment influencing the distribution and community structure of corals throughout Moreton Bay. A reassessment of observations reported after the 1956 and 1974 floods, combined with new observations, indicate that mortality after flooding is unlikely to be responsible for persistent patterns of relative abundance within coral communities in the Bay. Differences in coral species composition between sites within the Bay are not consistent with the patterns of mortality reported following the 1974 flood. The potential of flooding to influence coral community structure in Moreton Bay was investigated using: (i) a comparison of observations of coral mortality following the 1996 flood with patterns of coral mortality reported after the 1956 and 1974 floods to determine patterns of flood-related disturbance; and (ii) a transplant experiment in which colonies of *Favia speciosa* (the most common faviid species in the Bay) and *Acropora digitifera* (the most common acroporid in the Bay) were moved to different locations within a small area that experience a range of lowered salinity, allowing for a comparison of the ability of each of these species to withstand varying levels of low salinity stress. These observations provide a basis for reinterpreting the relative distribution patterns of faviid and acroporid corals, as well as some general aspects of coral community structure, in Moreton Bay in the context of flood impacts.

While coral community responses to the 1956 and 1974 floods suggest that patterns of mortality resulting from flooding are unlikely to be responsible for the patterns of dominance existing in Moreton Bay (since the dominant species, *F. speciosa* was more susceptible to mortality than some of the others observed), observations following the 1996 flood show that less severe floods may selectively remove or impede certain taxa at some of the sites. Most importantly, the transplant experiment showed that acroporids appear to be more susceptible to reduced salinity (viz. flooding) than faviids.

Introduction

Differences in the community structure of the coral reefs within Moreton Bay are recognised in the scientific literature. Faviid (especially *Favia speciosa*) dominance of coral communities (at sites other than Myora, which is dominated by *Acropora digitifera*) has been documented by a number of authors (Wells, 1955; Slack-Smith, 1960; Lovell, 1975; 1989; Harrison *et al*., 1991; Harrison & Veron, 1993). Pronounced differences in community structure between coral sites in Moreton Bay are associated with increasing distance from the mainland. These differences, from west to east, include an increase in the proportion of the substrate covered by scleractinian corals and a reduction in the dominance of *F. speciosa*, with *A. digitifera* becoming the dominant species at Myora. Temporal changes in the dominant taxa during the late Holocene have also been described (Wells, 1955; Lovell, 1975; 1989) with *Acropora*-dominated reefs being replaced by the modern faviid-dominated reefs, except for the Myora reef patch, which is the only modern *Acropora*-dominated reef in the Bay. It has been suggested that both of these patterns of dominance are a response to patterns of terrestrial influence (Slack-Smith, 1960; Lovell, 1975; 1989), with a decrease in the effect of mainland runoff with distance from the western shore. Most authors who have referred to these patterns have suggested that they are primarily a result of coral mortality occurring during flooding. In this study previous observations of the effects of flooding are combined with observations made after the 1996 flood and a transplant experiment (simulating some characteristics of flooding)

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 597-604.

in order to extend current understanding of the relationship between coral community structure, the influence of catchment runoff and intrinsic site characteristics in Moreton Bay.

Methods

The May 1996 flood impact survey

Visual inspections, while snorkelling, of coral communities at Sandy Island, Wellington Point/ King Island, Green Island, Peel Island and Myora (Figure 1) were made three to four weeks after the May 1996 flood to assess the extent (spatially and taxonomically) of coral mortality and/or morbidity. Of these locations, sites at Green Island, Peel Island and Myora had been visited regularly prior to the flood (usually about once per month since June 1995).

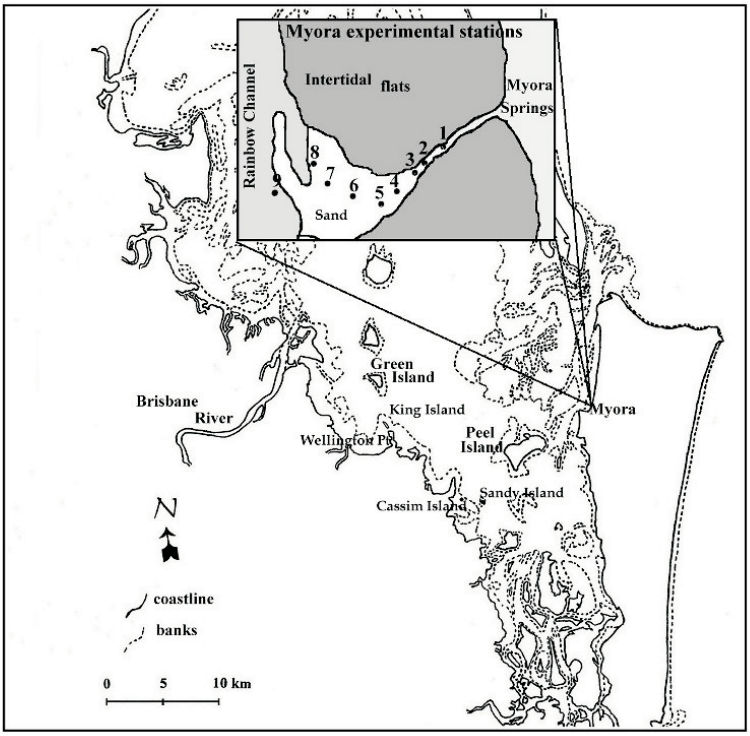

Figure 1. Map of the study area including an inset with the Myora Springs experimental sites.

Four weeks after the flood, five 30 m video transects, across the width of the reef flat, perpendicular to shore, were surveyed at Green Island. Each transect was marked with a fluorescent surveyor's rope marked at 10 cm increments and weighted to the bottom. The video camera was focused on the transect line by a snorkeller swimming slowly on the surface (1.2 m to 0.3 m from the substrate). Counts of colonies of each taxon present (faviid, *Acropora*, *Goniopora* or *Turbinaria*) were recorded and classified into one of four conditions. These were:

- Normal – tissues appeared healthy showing no visible signs of damage or bleaching.
- Bleached – tissues of the colony were alive but with a visible loss of pigmentation.
- Partial death – presence of both living and dead tissue on the colony.
- Dead – no living tissues observed on the colony, bare skeleton visible.

The Myora transplant experiment

The salinity tolerances of *Favia speciosa* and *Acropora digitifera* were investigated in the field over a nine day period during July (1996). Colonies of both species used in the experiment were collected from the Myora coral patch and transplanted. Two colonies from each species were placed in each of eight PVC trays. The trays were placed in the intertidal channel at Myora Springs (Figure 1) at increasing distances offshore in order to expose them to varying degrees of low salinity stress resulting from tidal influences on the freshwater outflow. This experiment was not designed to estimate other water quality changes associated with floods (e.g. increased turbidity and nutrient concentrations). Moreover, lowered salinities during floods are likely to be much less affected by tides and would be associated with other stressors (e.g. elevated suspended sediment loads). A control tray (to test for colony responses to transplanting) was positioned seaward of these trays in waters not subjected to salinity fluctuations during tidal oscillations. Observations of the colonies were recorded during low tide for each of the nine days of the study. The condition of each colony for each of the daily observations was described using five condition categories:

- Healthy – no discernible change in either the colour or extent of intact living tissue on the polyps.
- Pale – intact tissues retained the original hue of the colony with a noticeable reduction in the intensity.
- Bleached – colour loss from intact tissues, with tissues tending towards grey hues, as with bleaching described by Slack-Smith (1960).
- Partial polyp death – tissues of the surface of the coenosteum lost leaving bare skeleton with the tissues of the oral disk retained.
- Whole polyp death – bare skeleton of whole polyp exposed.

During three days of the study, rising tide salinities were measured opportunistically (using a Horiba multiprobe) at the time when colony condition was being recorded. Measurements were made by lowering the probe into the PVC tray next to the transplanted coral colonies and allowing sufficient time for the readings to stabilise. Data from the three days' observation were aggregated into time increments of 15 minutes duration following low tide so that the general pattern of salinity change during the rising tide could be plotted.

Results

The May 1996 Flood

Qualitative surveys

Observations of faviids at Sandy Island, Wellington Point/King Island, Peel Island and Myora revealed no visible effects from the flood. At Green Island *Acropora* colonies (not recorded in the video survey) were observed, some showed no mortality, some partial mortality and others complete mortality. One colony, >1 m in diameter, had been observed (alive) since June 1995 during regular sampling trips and was completely dead subsequent to the 1996 flood. Similar patterns were observed at Peel Island with some *Acropora* colonies showing no mortality and others showing partial mortality, although no completely dead *Acropora* colonies were observed at Peel Island. The only other site where *Acropora* was observed was the Myora reef patch where *A. digitifera* covers most of the substrate. At this site partial mortality and partial bleaching of some *Acropora* colonies was observed, although the majority of colonies were apparently unaffected by the flood.

Green Island video survey

A total of 261 coral colonies wasrecorded from the five transects surveyed on the southwestern side of Green Island, approximately 10 km from the Brisbane River mouth (245 faviids, 2 *Acropora*, 11 *Goniopora*, 3 *Turbinaria*). There was no evidence of any flood-related effects on any of the faviid colonies or on the few *Turbinaria* colonies present (Figure 2). In contrast, about half of the *Goniopora* and *Acropora* colonies present suffered flood-related morbidity (partial mortality in *Acropora*, and bleaching or partial bleaching in *Goniopora*). No whole colony mortality was observed on any of the video transects.

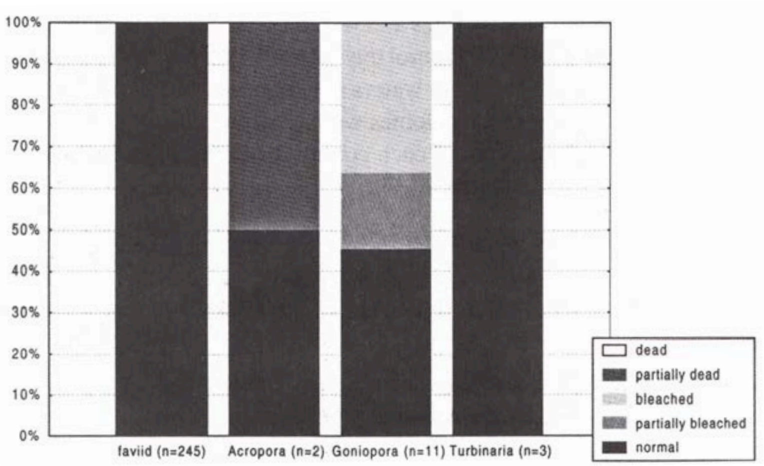

Figure 2. Percentage of colonies from Green Island affected by the May 1996 flood.

No mortality of *Goniopora* colonies was evident at any of the sites. Bleaching and partial bleaching of some of the *Goniopora* colonies was observed at Green Island, Wellington Point/ King Island and Peel Island, although not all colonies were affected. *Goniopora* colonies were also observed at Myora but none of the colonies at this site showed any effects of the flood. While the sample size is very small for all taxa except faviids, the patterns of morbidity recorded in the video transects are similar to those found in the qualitative surveys.

The Myora coral transplant experiment

The aggregated salinity measurements show the different exposure to fresh water received by transplanted colonies at each of the transplant sites. The duration of fresh water influence increased at stations shoreward of the control station with the longest exposure to lowered salinities occurring at station 1 (Figure 3). Stations 7 and 8 had reached seawater salinity (above 34 ppt) by 45 mins after low tide. In contrast, station 1 was still predominantly freshwater 2 h after low tide and only reached seawater salinity after about 2.5 h. Based on these data, estimates of the duration of depressed salinity at each of the stations show a monotonic increase in the duration of low salinities between station 8 and station 1. Station 8 was exposed to lowered salinities less than 10% of the time, and station one about 45% of the time (Figure 4). Coral morbidity patterns are described below in relation to these salinity fluctuations.

The results of the transplant experiment are consistent with the observations of mortality made after the 1996 flood. *Acropora* colonies were more susceptible to mortality induced by exposure to fresh water than were *Favia* colonies. Two main patterns were evident among the transplanted coral colonies:

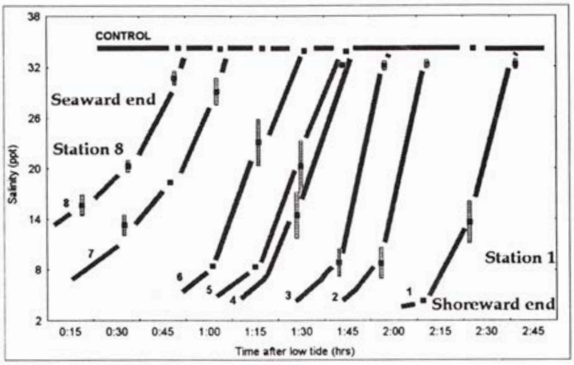

Figure 3. Salinity measurements (mean and std error) at control and test sites during the Myora transplant experiment – near seawater salinities progressively invade each of the experimental stations with the rising tide.

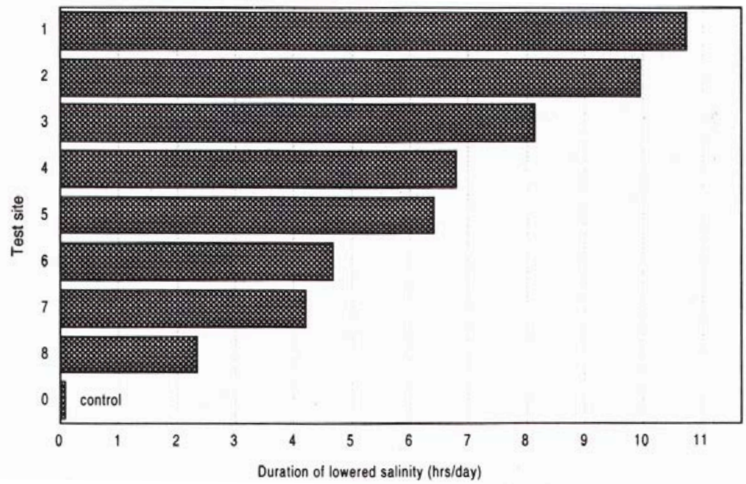

Figure 4. Estimated daily exposure of control and test sites to lowered salinities during the Myora transplant experiment – based on timing of the freshwater/seawater salinity change above.

- At each of the eight test sites at Myora Springs *Acropora* mortality occurred before any change was observed in the *Favia* colonies (Figure 5A,B).
- On each of the nine days of the study *Acropora* mortality extended to test sites seaward (i.e. with less freshwater influence) of the extent of *Favia* mortality (Figure 5A,B).

At stations 1 and 2 (with the longest exposure to lowered salinities) all *Acropora* colonies were completely dead after the first day. In contrast *Favia* colonies at the same stations showed gradual increases in morbidity over the first six days after transplantation with no whole colony deaths until the seventh day of the experiment. *Favia* colonies at stations 6, 7 and 8 appeared to remain largely unaffected by the depressed salinities for the duration of the study while *Acropora* colonies at all of the test stations were completely dead by the sixth day of the experiment. No whole colony mortality of *Favia* occurred except at test stations 1 and 2 over the last three days of the experiment. Neither morbidity nor mortality of either *Acropora* or *Favia* was observed at the control station.

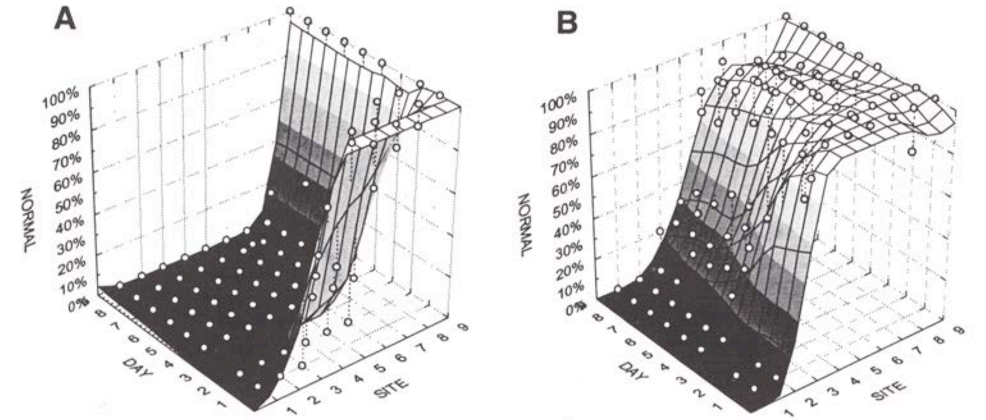

Figure 5. Colony morbidity of *Acropora* (**A**) and *Favia* (**B**) at each of the experimental stations during the salinity transplant experiment.

Favia colonies used in the transplant experiment showed greater variability in the intermediate stages between healthy colonies and dead colonies. For example, the retention of tissues around the oral area of the polyps after most other tissues had been lost was only noticeable upon close inspection. The ability of colonies to recover from these sublethal effects is likely to play an important role in determining the impact of flooding, but was beyond the scope of this project. Mortality amongst *Acropora* colonies observed in the field and during the coral transplant experiment tended to occur without other evidence of morbidity. The intermediate stages characteristic of the deaths of *Favia* colonies were either less common in *Acropora* colonies, or happened too quickly to be detected within the sampling framework.

Discussion

The results of the Myora Springs transplant experiment are not considered to be representative of the effects of flooding. Flood events are likely to be less influenced by tidal cycles and include significant variations in parameters other than salinity (e.g. turbidity and nutrients) which could also be directly or indirectly detrimental to corals. Nevertheless, both the observations made subsequent to the 1996 flood and the results of the Myora Springs transplant experiment suggest that Moreton Bay's acroporids are much more susceptible to depressed salinities than are its faviids.

Other reports describing the influence of flooding on Moreton Bay corals include those following the flood of 1956 (Slack-Smith, 1960) and 1974 (Lovell, 1975; 1989). Slack-Smith (1960) surveyed Peel Island and also reported observations from Coochiemudlo, Macleay and Cassim Islands, Raby Bay, Dunwich and Myora. The major findings of the study were that moderate and patchy coral mortality was observed throughout the central and southern sections of Moreton Bay, and that *Favia speciosa* was the only species affected by the flood, with other species (including *Goniopora* spp. and *Acropora* spp.) showing no visible responses to the flood.

Some caution must be exercised when interpreting these findings as Slack-Smith recorded corals as dead (showing "imminent death") when the tissues of the colony were a pale grey/brown colour. It is possible for corals to recover from bleaching (e.g. Buddemeier & Fautin, 1993), so mortality estimates from the 1956 flood may have been exaggerated.

Reports following the 1974 flood (Lovell, 1975; 1989) differ substantially from those of the

1956 flood in that the impact on corals at sites on or near the mainland coast ranged from total to very little mortality, with deeper sites tending to experience lower mortalities. Many species were affected by the 1974 flood, including *Favia speciosa* (*Acropora* colonies were not observed), and at one location where only partial mortality of the community occurred (the deep reef on the eastern side of Green Island) *Goniopora lobata*, *Turbinaria peltata* and *T. frondens* all suffered higher mortality rates than *Favia speciosa* (mortality rates for *Acropora* spp. were not reported).

The pattern of coral mortality observed after the 1996 flood differed greatly from those after the 1956 and 1974 floods in two main ways:

(i) observed impacts were limited to two genera, with mortality only observed in *Acropora* and bleaching only observed in *Goniopora* (although not all colonies of either genus were affected at any of the sites); and

(ii) faviids showed no response to the flood at any of the sites observed.

The taxonomic extent of flood impacts varies between the reports of Slack-Smith, Lovell, and this study. While Slack-Smith (1960) reported that *Favia speciosa* was the only species affected, and therefore was the most susceptible to flooding, reports by Lovell (1975; 1989) and this study both indicate that other species are more susceptible to the effects of floods. Observations of the taxon-specific impacts of the 1996 flood are consistent with reports following the 1974 flood (Lovell, 1975; 1989), although the 1996 flood affected fewer taxa overall. If we consider reports from these two floods as being reasonably accurate then susceptibility to flooding of the genera observed can be ranked as follows:

Most affected ⟶ **Least affected**

Acropora[2] ***Goniopora***[1,2] ***Turbinaria***[1,2] ***Favia***[1,2] ***Psammocora***[1]

[1] Lovell, 1989 ; [2] this study

The lack of any observations of mortality or morbidity of faviid corals (which are the dominant scleractinians at all of the sites except Myora) during the 1996 flood highlights the limited impact (taxonomically as well as spatially) that this flood had on corals in Moreton Bay. Despite differences in the pattern of impacts on corals following each of the floods, the spatial patterns of dominance (during recovery after floods) across Moreton Bay have been reasonably consistent (pers. obs. cf. Slack-Smith, 1960; Lovell, 1975; 1989; Harrison *et al.*, 1991). While Cassim, King and Coochiemudlo Islands, most of Green Island and west and southwest Peel Island, which suffered either complete or near complete mortality in the 1974 flood, have recovered differently, they display general patterns of increasing cover and decreasing faviid dominance with distance from the mainland similar to those reported prior to the flood (Harrison *et al.*, 1991; pers. obs.). Lovell (1989) reports strong similarities between species assemblages at southwest Peel Island prior to the 1974 flood and after seven years of recovery from complete mortality following the flood. These observations suggest that differences between sites that have occurred since the 1974 flood must be generated by different patterns of coral recruitment and/or growth. It follows that these processes must play a critical role in determining the persistent patterns of coral community structure in the Bay. Environmental characteristics that influence recruitment and/or growth in Moreton Bay need to be investigated to provide an understanding of the mechanisms that generate these persistent patterns of community structure in the absence of flooding.

It appears that the role of flood disturbance as a structuring force in Moreton Bay coral communities has been overemphasised in the literature. Spatially, marked differences in community structure occur between sites that were subjected to the same degree of flood impact (e.g. Lovell, 1975; 1989). Temporally, sites affected by flood impacts return to a similar community structure to that which existed prior to flooding. Taxonomically, the dominant species, *Favia speciosa*, is not the most resistant to flooding. Aspects of the timing, intensity and duration of a flood are likely to determine how a flood influences the biota. Substantial differences in the response of coral species and communities to the three floods investigated (1956, 1974 and 1996) suggest that flooding is unlikely to be responsible for the apparently persistent spatial patterns of coral dominance in Moreton Bay. As flood impacts have been overemphasised as a structuring force in Moreton Bay's coral communities other processes have been neglected. To extend the current understanding of pattern and processes in Moreton Bay's coral communities, alternative mechanisms influencing the reef environments must be investigated (Johnson & Neil, Chapter 7 this volume).

Acknowledgements

We wish to thank Belinda Andrell for her assistance conducting the video transects and other field work after the 1996 flood, as well as Timothy Ault and Ida Fellegara for their assistance with the salinity transplant experiment. Genevieve Quirk collected the data for the salinity transplant experiment. We are grateful to Peter Harrison, Vicki Harriot and Ian Tibbetts for their comments on the manuscript.

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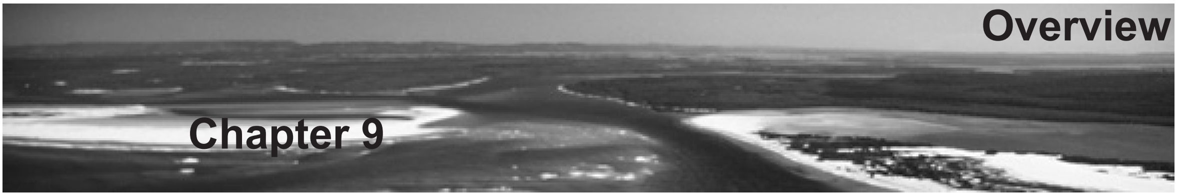

Management Options

This final chapter examines how management of the Bay and catchment has been attempted and what is required for effective management. Government, it appears, has matured in its approach to managing the system, as there are portents of a more unified and holistic management approach, incorporating elements from a diversity of scientific disciplines and stakeholder values. This improvement has been particularly rapid in the 1990s. We can only hope that this continues. It is of concern that the recent global economic downturn in 1998 may draw attention away from environmental issues and stall the development of an effective integrated management system, driven ideally by a single agency, for the Bay and its catchment.

Tony Pressland and coauthors examine management responses to pressures on the system. They provide a summary of the resource and social values of Moreton Bay and its catchment, discuss impacts upon those values and provide survey of actions that have been taken by government, industry and community in response to these impacts. They call for the allocation of major funding and other resources in a new system of management for the Bay and catchment.

The final paper in this book provides a model for the development of this new management system. Darryl Low Choy catalogues the many management initiatives that have attempted management of the Bay and catchment, illustrates their deficiencies and suggests an alternative approach. It is a fitting summary to the volume as it offers a way forward. While scientists face many exciting challenges in unravelling the complexity of the Bay and such information will be critical to our ability to manage it, in the absence of an effective and authoritative management system these efforts are likely to be in vain. Moreover, in the development of a new management program for the Bay it is vital we both comprehend and incorporate the many dimensions of change. This book has demonstrated that change is an ever-present feature of Moreton Bay and its catchment; it is hoped this knowledge will assist us to achieve our vision for a diverse, productive, healthy and beautiful system.

Ian R. Tibbetts

Responses to Pressures on the Integrity of Moreton Bay and its Catchment

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Introduction

Moreton Bay and its catchment is a unique ecosystem, with landscapes ranging from the eastern escarpments of the Great Dividing Range, through fertile alluvial freshwater floodplains to marine vistas of sand islands and mangrove-lined shores. These varying landscapes have been recognised for thousands of years for their value as sources of food and shelter, and for their spiritual significance.

Whilst Aboriginal people helped shape the mosaic of vegetation characteristics of the region, it was not until European settlement that the system began to be placed under severe pressure. In response, governments and communities have put in place procedures and programs to redress damage and plan for the future well-being of the area.

This paper briefly outlines the values which society as a whole places on Moreton Bay and its catchment. These values drive changes in management to ameliorate past actions, reduce present pressures, and paint a picture for the future. We also briefly discuss the present state of the Bay and catchment, and summarise the pressures on the system which have led to that state. The bulk of the paper, however, focuses on government, community and industry responses to the perceived state of Moreton Bay, its catchment rivers and contributing land areas.

Values

Moreton Bay is seen by many as an extremely valuable resource. It is of national and international importance, as recognised through the international *Ramsar Agreement* and the declaration of the Bay as a Marine Park by the State Government, amongst other responses. The Bay is the habitat for some significant animals including the dugong, marine turtles and rare birds (e.g. the chestnut teal), as well as valuable fisheries including fin fish and crustaceans. The extensive seagrass beds are themselves an important attribute of Moreton Bay (Queensland Department of Environment and Heritage (QDEH), 1991).

For people living in the 1990s, Moreton Bay and its catchment offer many opportunities for recreational activities including fishing, sailing, water skiing, boating, camping and just relaxing (QDEH, 1991). The area also has significant value for its economic resources, including commercial and recreational fisheries, and the sand and coral deposits sought by the construction industry (Queensland Fisheries Management Authority (QFMA), 1997; Williams, 1992). It has landscape and amenity value as well as being an icon for the residents of the capital city of Queensland. Local indigenous groups have a long history in the area – at least 20 millennia. Their values include a 'spiritual' sense of belonging and relationship to land and sea, as well as economic and potential native title interests (QDEH, 1991).

The large and rapidly-growing population base on the Brisbane – Ipswich axis will continue to value the area for all these reasons (Queensland Department of Housing, Local Government and Planning, 1995). Thus the maintenance of Moreton Bay in sound condition is a prerequisite for the maintenance of the values of the region. What then, is the current condition of Moreton Bay and its contributing catchments?

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School of Marine Science, The University of Queensland, Brisbane. pp. 607-620.

Condition of Moreton Bay and its Contributing Catchment

Recently, information to determine the state of Moreton Bay and its catchment has been collated through the *State of Brisbane River, Moreton Bay and Waterways* study. The study shows that the shallow western Moreton Bay areas in particular (such as Bramble Bay), suffer from sedimentation due to sediment movement down the rivers, while the water quality in the Bay declines markedly from the western Bay to the east (Moreton and Stradbroke Islands). Pollutants and nutrients from rivers and coastal areas move north towards Pumicestone Passage and have resulted in a significant decline in its quality over the past 20 years. These changes have led to algal blooms, often limited, but more widespread on occasions (such as that following the May 1996 flood).

There are about 14 700 hectares of marine vegetation (mangrove, saltmarsh and claypans, but excluding seagrasses and associated vegetation) between the Coomera and Caboolture Rivers. Between 1974 and 1987 over 1 200 hectares were lost, and probably at least this area again has been lost from 1987 to the present. However, sediment deposits along the rivers, as can be seen along the Brisbane River, will lead to an increase in mangroves in these areas (Queensland Department of Environment (QDoE), 1997b).

Away from the Bay, and further back into the catchments, similar changes in the state of the natural resource have been identified. Freshwater flows down the streams feeding Moreton Bay have declined markedly over the past 100 years as a result of water extraction for urban use, agricultural irrigation, industrial requirements and the construction of impoundments (QDoE, 1997a; b). Riparian zones are generally in poor condition. They are narrow to nonexistent in places, and the natural vegetation has been replaced in many areas with introduced species, many now declared as weeds under the *Rural Lands Protection Act*. Reports (Queensland Department of Natural Resources (QDNR), 1996a; b) indicate, for example, that 75% of the riparian vegetation in the Lockyer Creek and Bremer River subcatchments are in poor to very poor condition. As a result, the biodiversity of both natural plants and native animals is low and the aquatic habitat is in moderate to poor condition (QDNR, 1996a; b).

Sediment movement to Moreton Bay has occurred as a result of the collapse of stream banks due to lack of vegetation protection, extraction of sand and gravel and to the erosion of adjacent lands, particularly on the floodplain, but also on duplex soils further up the slope. Further, clearing of lands in the upland areas has resulted in some salt breakout in lower sections due to rising water tables. Salination of underground waters used for crop irrigation has also occurred. Cultivation for crops and improved pastures for grazing have provided an additional source of nutrients making their way to the waterways and eventually to Moreton Bay, particularly during storm events (QDoE, 1997b).

All these factors impact on water quality in the freshwater catchment areas. Whilst the quality of waters in the Stanley River and upper Brisbane River sub-catchments is good, below the Wivenhoe Dam it is much lower. In the more populated reaches of the Brisbane, Pine and Caboolture Rivers, the potential for a significant deterioration in condition is much greater (QDoE, 1997b).

Pressures Impacting on Moreton Bay and its Catchment

The findings of the "State of" report outlined in the previous section indicate the range of pressures which human activities have inflicted on the natural resource base of the area. These can be summarised as falling into three main categories. Firstly, there have been catchment and land use changes including urbanisation, agricultural impacts, degradation of the riparian corridor and clearing of natural vegetation. Secondly, waterway corridor engineering has impacted on the system through water extraction, water impoundment, modification to the floodplain and extraction of sand and gravel. Thirdly, there have been instream activities such as fishing, introduction of exotic species and recreation, which have also affected the catchment (QDoE, 1997b).

Urbanisation results in increased and more concentrated runoff, and an increase in contamination from nutrients and pollutants such as that from septic seepage, sewage effluent and household wastes (e.g. detergents, pesticides, etc.). Light industry and commercial operations are located along streams leading to Moreton Bay. These streams have been used to dispose of effluent containing many potential contaminants including chemicals and heavy metals.

Another effect of urbanisation is reduced access to the Bay, island foreshores and to the riparian environment of the rivers and streams feeding into the Bay. This has led to a decline in the opportunity for riverside and bayside recreation, and the loss of vista recognised as a high value attribute by residents and visitors alike. The latter also involves the loss of landscape associated with a number of the heritage class structures originally built on the River and Bay because of this amenity value (BRMG, 1996a). Shafston House, Newstead House and Yungaba are examples of those retained in better than average condition.

Much of the natural vegetation along the rivers and creeks leading to Moreton Bay has been removed and replaced by, in the main, undesirable herbaceous and woody species. These have contributed to the reduction in habitat quality of aquatic plant and animal life and to the choking of smaller streams so that little fresh water flows to the major rivers and the Bay. Most of these exotic species are tropical, not temperate species and, because they use a different photosynthetic pathway, their products are not as useful or as available for the digestion or production processes of indigenous instream fauna. The poor condition of the riparian corridor has also resulted in excessive bank erosion, the movement offsite of excess fertiliser and chemicals used in agriculture and the spread of weed species.

Clearing of vegetation has resulted in an increased opportunity for loss of top soil, and for a rise in the height of the watertable which can, and has, lead to salination in particular parts of the catchments supplying fresh water to Moreton Bay. Biodiversity has also been reduced.

Water extraction occurs along the length of the rivers and streams which feed Moreton Bay. Whilst the main use of this water has been for agriculture, the construction of impoundments for urban water use and flood mitigation (e.g. Somerset, Wivenhoe, Manchester and North Pine Dams) has resulted in changed flow regimes downstream (BRMG, 1996b; c). The impact of agricultural extraction was felt strongly during the long drought in the Lockyer Valley from 1992 to 1996. During this time, water use was severely restricted and the quality of available water was often poor. A large amount of water is also extracted for industrial uses, including for cooling in electricity production, particularly along the Bremer River, Oxley Creek and the Lower Brisbane River. A further problem is that the effluent from these uses has often been of poor quality.

Modification of the floodplains of large areas of the Moreton Bay catchment has resulted in changed water flow during peak times, and has enhanced opportunities for contamination from chemicals, fertiliser and household wastes. In particular, these changes have allowed water to be channelled and concentrated artificially, contributing to increased scouring and erosion downstream and consequently, to increased sediment loads in the water (QDoE, 1997). Changes to floodplains can also limit the interaction between them and waterways during peak flows, thus reducing the opportunity for organic energy input to the floodplains downstream (BRMG, 1996c).

Agricultural production may be associated with increased levels of soil erosion and contamination of waterways with chemicals (e.g. fertiliser, pesticides, effluent from piggeries and dairies), and agricultural irrigation removes considerable quantities of water from the system downstream (BRMG, 1996b).

Another activity which has had a marked effect on both Moreton Bay and the streams contributing to the Bay, is the extraction of sand and gravel. For example, dredging of sand and gravel in the Brisbane River over the past 100 years for both navigational and commercial purposes has resulted in the removal of the bar at the mouth of the river and the emergence of rock outcrops along the stream. Dredging activity has deepened the river by an average of 6 m in the lower 25 kms, resulting in an increased tidal prism. This has contributed to the resuspension of sediments and the "brown" colour of the water in the city reaches of the river. Removal of dredges from the river will probably have only a minor effect on this problem (QDoE, 1997a; b).

Recreational and commercial fishing has contributed to an apparent decline in fish catch in Moreton Bay and rivers, although destruction of habitats, particularly for breeding and barriers to fish migration in streams, are also significant causes (QDoE, 1997a; b).

Responses to Pressures and Current State of Moreton Bay and Waterways

The desirable outcomes of responses to maintain and improve the health of Moreton Bay and its catchment areas may include:

- reduced sediment and nutrient loads entering the western parts of the Bay, particularly those from stream bank erosion;
- enhanced water flow and condition of instream and waterway corridor habitats;
- retention of ecological integrity and return of biological processes to an appropriate system level;
- security and quality of water supply for urban, industrial and rural purposes commensurate with ecological needs;
- enhancement of the social and cultural values of Moreton Bay and its attendant streams;
- preservation of examples of past activities, buildings and landscapes;
- fostering of agricultural activities which are in the interests of all residents of southeast Queensland;
- disposal of wastewater with minimal effect on the ecology of the Bay and its rivers and streams; and
- enjoyment and recreational use of the area by all residents and visitors in a sustainable manner conducive to attaining societal needs.

The achievement of these outcomes will depend on a combined government, industry and community commitment to making the necessary changes happen. There is a perception held by many in the community that "government" – "THEY" – should do something about the health of Moreton Bay, Brisbane, Pine and Caboolture Rivers and their tributaries. However, there is also a need for individuals and communities in southeast Queensland to take responsibility for the present condition of the area, and for implementing any management options which are developed to improve it.

State and local governments can have a major impact on the use and management of waterways in the Moreton Bay catchment area. Legislative responses are the prerogative of the State, whilst many opportunities exist for Local Government plans and bylaws to be enacted to facilitate these. Furthermore, through their role and relationship with industry, both levels of government have the responsibility to ensure that appropriate planning occurs to protect the environment whilst providing for appropriate development. Figure 1 shows the 17 local governments whose jurisdiction includes the catchment of Moreton Bay.

Industry also has an interest in maintaining the health of the Bay and its river systems. These areas are important for investment, for the use of natural resources and for facilitating tourism, transport and shipping. In recent years, many industries have recognised the need to be a "team player" and commit resources to enhancing the environment of the area in which they do business. In some instances, this commitment has extended outside the usual zone of influence of the industry sector. Examples are the corporate sponsorship of Landcare and sporting activities.

Finally, the many ordinary citizens who have had a prick of conscience are taking a higher profile role in influencing others in the community, including government, to make changes to management practices. Community groups such as sporting associations, conservation groups, landcare and catchment management groups have taken a proactive approach to the needs of Moreton Bay and its catchment, and have implemented a range of actions designed to improve the state of catchments, and consequently, of Moreton Bay. A range of current strategies and initiatives developed by government, industry and the community sectors are described in the following sections.

Government responses

Several State and Local Government programs are designed specifically to foster and improve catchment management.

Integrated catchment management

The Integrated Catchment Management (ICM) Program initiated by the QDNR (formerly the Department of Primary Industries – DPI) in 1992 is designed to empower communities to address significant catchment needs in a partnership arrangement with government (QDPI, 1991). In the catchment area of Moreton Bay, one pilot study for this initiative was instigated in 1992 in the Lockyer Creek catchment. The catchment coordinating committee for the Lockyer Creek, the Lockyer Resource Management Group, has marshalled local resources, including existing groups such as the Lockyer Watershed Management Association, to develop plans and actions relating to locally determined priorities.

Since 1992, other catchment management groups have been established, mostly with funds from a range of sources including QDNR, local government and the National Landcare Program (NLP). In the Moreton Bay area, there are now groups based in the Pumicestone Passage, Pine Rivers and Bremer River areas, and in the catchments in the Brisbane area of the Oxley, Nundah and Downfall Creeks. The Pumicestone Passage group originated from a report commissioned by the Department of Environment (QDoE) in 1993 which indicated that poor water quality in the passage was due to land management practices in its catchment, and indicated some best management practices which could be implemented to reverse the trend (DEH, 1993).

Catchment groups endorsed under the QDNR catchment program are eligible for Establishment – \$500, and Operating grants – up to \$5 000/year for three years (DPI, 1994; 1995). They also become eligible for Project Grants (primarily for planning and research) and Implementation Grants (designed to implement strategies with on-ground works). The value of these is not fixed and is dependent on the issues being addressed and the availability of funds. The grants are on a cost-share basis, and are decided through the Regional Assessment Process used for grants under the National Landcare Program.

QDNR also provides support for catchment coordinating committees through Extension Officers skilled in working with community groups and with a knowledge of natural resource management. One officer, based at Ipswich, has responsibility for linking the operations of all catchment groups in southeast Queensland, and ensuring good communication between groups both regionally and statewide. Extension Officers at Ipswich have a role in providing more detailed support for catchment groups in the catchment area of the Brisbane River.

In addition, QDNR provides funds to the Lockyer Resource Management Group to assist in

Figure 1. The 17 local governments whose jurisdiction includes the catchment of Moreton Bay.

its employment of a Catchment Manager, for operational activities and for part funding of a Catchment Coordinator in the Pumicestone Passage area.

Technical backup for catchment groups for water, river and floodplain management, wastewater management, water quality, riparian zone management and forest management is available through QDNR. It also provides a scientific basis for sound catchment management by research undertaken through the Resource Sciences Centre at Indooroopilly. Information and research for fisheries habitat and fish stock issues are also provided by QDNR, and QDoE provides support in the area of water quality.

Landcare

The QDNR has been the lead agency for Landcare in Queensland since the QDNR's inception of Landcare in 1986. It administers the federal National Landcare Program (NLP) in the State, and provides direction for the activities of Extension Officers in Landcare related topics. It also provides information and undertakes research on a wide range of issues including land conservation, weed and forest management, erosion, rainfall-runoff modelling and normal property management practices and infrastructure. The QDNR provides a similar service for production oriented issues and works with QDNR in the provision of property management planning services. Landcare Extension Officers who service the needs of the Moreton Bay catchment are based at Ipswich, Brisbane, Gatton and Gympie.

The *Watch* programs

There are a number of school-based programs designed jointly by QDNR and the Education Department aimed at improving the knowledge of students in natural resource management. *Waterwatch* is a program funded partly from the Australian Nature Conservation Agency and partly through the ICM Project Grants. It is designed to allow community and school groups to study the health of waterways in an educative way so that the importance of maintaining their condition is highlighted. *Waterwatch* is often coordinated with *Saltwatch* so that salinity can also be monitored in key areas.

Pasturewatch is aimed at increasing the awareness and knowledge of youth to the importance of grass and herbaceous plants not only for animal production (milk, meat, wool), but also for protecting the soil surface from the erosive actions of raindrop impact and runoff from rainfall.

Waterwise

Waterwise is a national program administered in Queensland by QDNR. It is primarily an education program run through local government to decrease water wastage. The Brisbane City Council in particular has a comprehensive *Waterwise* program focused on the community and industry.

Resource Allocation Management Plans (RAMPS)

QDNR is developing a framework for the allocation of public resources in an equitable and sustainable manner. Water Allocation Management Plans (WAMPs) will be developed in catchments to achieve sustainable use and management of Queensland's water resources. WAMPs may be implemented in the Brisbane River catchment in the near future. A WAMP is being undertaken in the Logan River catchment to the south of the Brisbane River catchment area. A similar type of management planning framework will be developed for extractive industries.

Fisheries management plans

The Queensland Fisheries Management Authority, together with the DPI, the Queensland Commercial Fishermen's Organisation and Sunfish, the recreational fishing organisation, are developing management plans for the important fisheries in the State. These will allow the harvesting of fish to be consistent with sustainable development principles, and ensure the maintenance of fisheries habitat in a sound condition. A Moreton Bay Fisheries Management Plan is under way with the release of a discussion paper (QFMA, 1997).

Legislative responses

There is a range of new legislation which is being enacted to protect the environment and to allow the sustainable use of natural resources. The *Environment Protection Act 1995*, together with Environmental Protection Policies for *Water, Noise, Waste, Mining and Air* which are presently being developed, will allow some of the concerns caused by pressures on the system to be addressed. The proposed *Natural Resource Management Act* would provide a legislative base for sound management of water, land and forest resources and for recognition of community-led integrated catchment management.

There are a number of other existing legislative responses which impact on the sustainable management of Moreton Bay and its environs. These include the *Coastal Protection and Management Act 1995*, the *Water Resources Act 1989*, the *Local Government (Planning and Environment) Act 1990*, the *Local Government Act 1993*, the *Fisheries Act 1994*, the *Noise Abatement Act 1978*, the *Nature Conservation Act 1992*, the *Transport Operations (Passenger Transport) Act 1994* and the *Transport Operations (Marine Safety) Act 1995*.

Other responses by government include the establishment of the Brisbane River Management Group in 1993 to facilitate sustainable ecological development in the Brisbane River catchment and to develop a management plan for the Brisbane River and its tributaries. The Brisbane City Council has also convened a group to develop a wastewater management strategy for Moreton Bay and its waterways, including the Brisbane, Pine and Caboolture River systems. This study is developing a model of the water quality aspects of the area and is studying in some detail the biological processes involved.

Local Government strategies

Local Government across the Moreton Bay catchment has developed and implemented a range of strategies designed to foster improved management of waste, pollution, and nutrients. Most local government instrumentalities fund catchment management activities in their shire or city. Many provide significant funding (e.g. Ipswich, Pine Rivers), while others provide in-kind support (e.g. Gatton, Laidley).

The Brisbane City Council, as a component of its Brisbane 2011 Plan, has a Metropolitan Green Space Strategy for parks and gardens. It has also worked with the community and industry to produce documentation on a range of aspects of development including *Environmental Best Management Practice for Parks, Environmental Impact Assessment, Waterways and Wetlands and Erosion and Sediment Control*.

Brisbane City Council has also developed Local Area Plans throughout the city. These plans cover such topics as *Community Profile, Transport, Open Space/Parks/Recreation/ Environment, Landmarks, Location of industrial and commercial areas,* and *Social and Community Development Issues*. In addition, Brisbane City Council has a network of Bushcare Coordinators who work with residents to enhance the bushland areas within the boundary of Brisbane City. To help fund these initiatives, a 'green levy' is part of the rate base for residents of Brisbane City.

The Ipswich City Council is in the process of implementing an *Enviroplan* which is aimed at environmental management planning for the future. As part of this initiative, the Council is supporting the Bremer Catchment Association, a community association interested in the development and use of the natural resources of the Bremer River and its catchment. Through this initiative, $300 000 is being spent on weed control and revegetation of 16 rural and urban areas along the Bremer River and its tributaries. Rehabilitation of the riparian environment is seen as a significant step in improving the state of the Bremer River.

The Pine Rivers Council has a Green Plan which details green corridors and complements the Shire's Strategic Land Use Plan. The Council has implemented a publicly accessible geographic information system which provides residents with the opportunity of assessing proposed developments in the light of future desired planning. Pine Rivers Shire was also instrumental in initiating a catchment management program for the catchments of the North and South Pine Rivers.

Initiatives of the Caboolture Shire Council include a *Greenlink 2001 Strategy* which includes programs for greening drainage easement, an Environmental Education Centre and a waste water re-use strategy. The Council has also provided funds to assist the ICM process in the Shire and Pumicestone Passage region.

Industry responses

Industry is responding to the increasing requirements for environmentally sustainable practices by developing codes of practice and benchmarking. In Queensland, rural industries have responded well to this changing management requirements. For example, the Queensland Farmers' Federation has drafted an *Environmental Code of Practice for Agriculture* in an effort to reduce the impacts of agriculture on land degradation, water quality and noise and air pollution. Individual rural industries are also going down a similar track. For example, the Queensland Dairy Farmers' Organisation has developed a *Manual for Dairy Farm Effluent Disposal* which is directed towards best management practice for disposal of wastes from dairies. Other examples include codes of practice for piggeries, prawn farming, egg production, chicken processing and the nursery industry.

The manufacturing industry has also been active in developing codes of practice and standards over the past few years. The Stockfeed Manufacturing industry, Dry Cleaning Institute, and Plastic and Chemical Institute are some examples of industries which have responded to environmental requirements. Industry-specific codes of practice have also been developed for the areas of sugar milling, foundries and engineering, sawmilling and timber, tanning, used tyre storage, quarries and clay brick manufacturing. These industry responses are of particular significance to Moreton Bay and its catchment. On a broader scale, similar standards and codes exist at the national level.

Urban residential developers are involved in developing codes of practice and strategic plans to reduce their impact on the waterways of the Moreton Bay catchment. For example, the Forest Lake development in the southwest, between Brisbane and Ipswich, was designed in accordance with the Brisbane City Council's *Metropolitan Green Space Strategy*, and utilises best management practice provisions for parks and environmental protection for wastewater discharge.

Community responses

Communities tend to respond to issues about Moreton Bay and its catchment through the formation of action groups. For example, there is now a wide range of community groups which have applied the principles of integrated catchment management to their local area. The interesting thing about these groups in the Moreton Bay area is that they are the first catchment groups in large urban areas, and they are actively engaged in discussion on a range of issues in a variety of industries – primary and secondary, as well as in the commercial arena. This will be a good test of the urban application of the principles of integrated catchment management, which in the past have mainly been applied to rural areas.

Similarly, the concept of *Landcare* is now a well established "way of life" for many communities. A number of active community groups are addressing a range of issues including water quality, extractive industries, revegetation of degraded areas, pasture studies, weed management, erosion control, maintenance of wildlife corridors, riverine management, education, rural tourism and recreational area development. The number of members in the groups varies, but most have between 20 and 80 members. Activities are funded from local sources as well as from the Natural Landcare Program, now a component of Natural Heritage Trust.

Table 1 describes some of the major groups and summarises the ways they have applied the principles of ICM, Landcare and ESD approaches. The scope of community groups is shown on Figure 2.

Future Management Needs

To manage the Brisbane River catchment effectively, it will be necessary to develop a more comprehensive understanding of the biological processes occurring in the catchment, waterways and Bay. Many government agencies, both state and local, are currently researching a range of topics which will help fill the knowledge gaps. Tertiary institutions are also playing a role, and are involved in compiling information which will be important in making management decisions.

The completion of numerous scientific investigations and development of a mathematical model of river and bay waters over the last 18 months has contributed significantly to our understanding of the major processes operating in the Brisbane River and Moreton Bay. This information has contributed to, and now underpins, many actions and strategies as part of a Water Quality Management Strategy for the Moreton Bay catchment. Further investigations are needed to better understand the major processes operating in freshwater catchments and the links between land runoff, water quality and effects on downstream water quality in estuaries and the Bay.

Strategic planning and implementation work is also progressing across a number of themes, including catchment land use, water entitlements and flows, extractive industries, recreation, noise, culture, heritage and tourism, as well as links with coastal management, natural resources planning and ports activities.

The above work has culminated in the release of the *1998 Waterways Management Plan: a framework for the management of the waterways of the Brisbane River and Moreton Bay catchment* which outlines strategies and actions for government, industry and the community

Table 1. Community Catchment Action groups in the Brisbane River catchment area.

Organisation	Main area of interest
Catchment management groups	
Oxley Creek Catchment Association (OCCA)	Aim is to protect and enhance the land, water and vegetation in the catchment through coordinating activities and improving management. The group is funded by the BRMG and the National Landcare Program. A State of Oxley Creek Catchment Report has been prepared. The group is actively engaged in addressing the issue of extractive industry, as well as having an Education Subcommittee and a Water Cycle working Group. The membership of this group comprises 270 active members and 200 "for information" members.
Nundah & Downfall Creek Coordinating Committee	An initiative of the Brisbane City Council in coordinated and participative management of the Nundah Creek catchment. A steering committee has been formed and a draft management strategy prepared.

Lockyer Catchment Centre	Provides the 'neutral ground' in which Landcare and ICM can cooperatively function in the Lockyer. The centre acts as a main shopfront address, resource centre and project base for many ICM/Landcare projects.
Residents Action Group for the Environment (RAGE)	Community initiative aimed at educating people in the catchment about their effects on the environment. Liaison role with Brisbane City Council and its planning and environmental projects.
Norman Creek Catchment Coordinating Committee	An initiative of the Brisbane City Council in coordinated and participative management of the Norman Creek catchment. The committee was formed early in 1996. A draft management strategy has been prepared.
Wildlife Preservation Society (Qld) - Kedron Brook	Re-establishment of the natural creek functions and characteristics. This includes the removal of 'hard' drainage structures, replacement of exotic and weed species in riparian areas and re-engineering the stream channel.
Caboolture Region Integrated Catchment Management Group	Catchment management in Caboolture Shire including protection of the Pumicestone Passage. A strong group with a good working relationship with Caboolture Shire Council.
Lockyer Watershed Management Association	Formed in the early 1980s to carry out Landcare and Catchment Management activities. Have been involved in a vast range of projects and have considerable expertise and knowledge about the area. Key areas of interest are education and vegetation.
North and South Pine Rivers Integrated Catchment Association	A coordinator was appointed in October 1996. The Group is in the initial phases of strategy development, and sees the main issues for its attention as planning, extractive industry, erosion and education and advocacy.
The Pumicestone Region Catchment Coordinating Committee	This group is based at Caboolture and focuses on the streams which flow into Pumicestone Passage from the mainland and Bribie Island and streams in Caboolture Shire which flow into Deception Bay. The Group is developing a catchment management strategy based on stakeholder endorsement. Community involvement has identified the following significant issues: planning, research and development, land and water management, pollution, water quality, effluent and community education and awareness.
Bulimba Creek Protection Society Inc.	Recently formed urban catchment group concerned with bushland protection, stormwater, maintenance of habitat and riparian zone management.
Save Our Waterways Now - Enoggera Creek	Coordinates a program to replace exotic and weed species in riparian areas with natives and restore vegetation. Extensive involvement of riparian land holders.
Bremer Catchment Coordinating Association Inc.	Focuses on catchment management in Bremer River catchment and Ipswich City Council area. The initiative is based at the Council with an on-ground focus to problem solving. The committee has been operating since mid-1996 and has the development of a catchment management strategy as a priority. Major issues to be addressed include weeds, sedimentation and erosion, urban stormwater quality and amount, conflicts in future land uses, security of water supply and riparian land management.
Lockyer Resource Management Group	One of the initial five pilot study catchments of the State Government's ICM program. The committee has been active on land use, local government planning, stream management and water quality and quantity issues. It has developed specific strategies for land use, fire management, and water resource identification and management and developed a Business Plan and a Local Authority Strategic Plan. It is actively engaged in negotiations on property valuations. Collaborated with the Lockyer Watershed Management Association to establish the Lockyer Catchment Centre, home for many projects and for a number of staff including the manager, and a technical officer for ICM and Landcare.

Environmental groups

Sunfish Inc. & Blue Fin Fishing Club Inc.	Interested in many matters related to or affecting fish or marine ecology. Large membership (3 500)
Greening Australia	National organisation with local branches in the catchment, assisting communities to achieve sustainable land management through conservation and establishment of vegetation for the environmental, social and economic benefit of all Australians.
Australian Marine Conservation Society	Previously the Australian Littoral Society. Leading group for the protection and management of the River and its waterways. Scientific, educational and consultancy activity programs plus hands-on rehabilitation. International reputation in coastal environmental management.
Moreton Bay Alliance	Organised by the Australian Marine Conservation Society, it circulates information on issues related to the care and protection of the environment of Moreton Bay among environmental groups and concerned residents.
Toowoomba Regional Environment Council	Regional environment group with strong links in community and with Greening Australia.
Men of the Trees	Undertakes revegetation projects in the catchment and supports groups with advice.

Landcare Groups

Rosalie North Landcare Group	Well-established group with extensive membership. Currently involved in weed eradication program, erosion demonstration project, cabinet timber plantation demonstration plot, field trips and information dissemination.
Boonah & Disrict Landcare Group Inc.	An active group in the Boonah Shire. Main projects concern weed control, property management planning, field days, education and information dissemination.
Crows Nest Shire Landcare Group	Rural group active in promoting landcare practices. Currently encouraging the formation of subcatchment/social groups to work on local problems.
Murphy's Creek Landcare Group	Formed in 1994 over concerns with stabilisation of new roadworks. Activities have expanded to include a riparian revegetation site and erosion control works.
Brisbane Valley - Kilcoy Landcare Group	Formed in 1989 to improve landcare awareness and practice in the upper Brisbane ctchment. Area of influence includes most of the catchments of Somerset and Wivanhoe Dams. Current suite of projects include weed control, wildlife corridors, gully rehabilitation, tree thinning trials, tree planting, waterwatch and nature walks.
Atkinson's Buaraba Catchment Water Management Group	Formed in 1994 to address water issues in the Buaraba Creek and Lower Lockyer Creek. Current projects concerned with water quality, water storage and distribution, sand and gravel and habitat.
West Moreton Landcare Group Inc.	Landcare group with mixed interests, including salinity and revegetation work.

Management of specific areas of public lands

Boondall Wetlands Committee	Established by the Brisbane City Council to advise on management of the 640ha Boondall Wetlands at the mouth of Nundah Creek. Also a key school and community educational group.
Bushland Care Groups - Brisbane City Council	A program of the Brisbane City Council to receive in-kind support from the Council to make sure remaining bushland areas remain in a healthy condition. Currently there are 66 Bushland Care Groups in the city.
Chapel Hill Environment Centre	Sponsored by the Gould League, this group is active in waterwatch, community and school eduction.

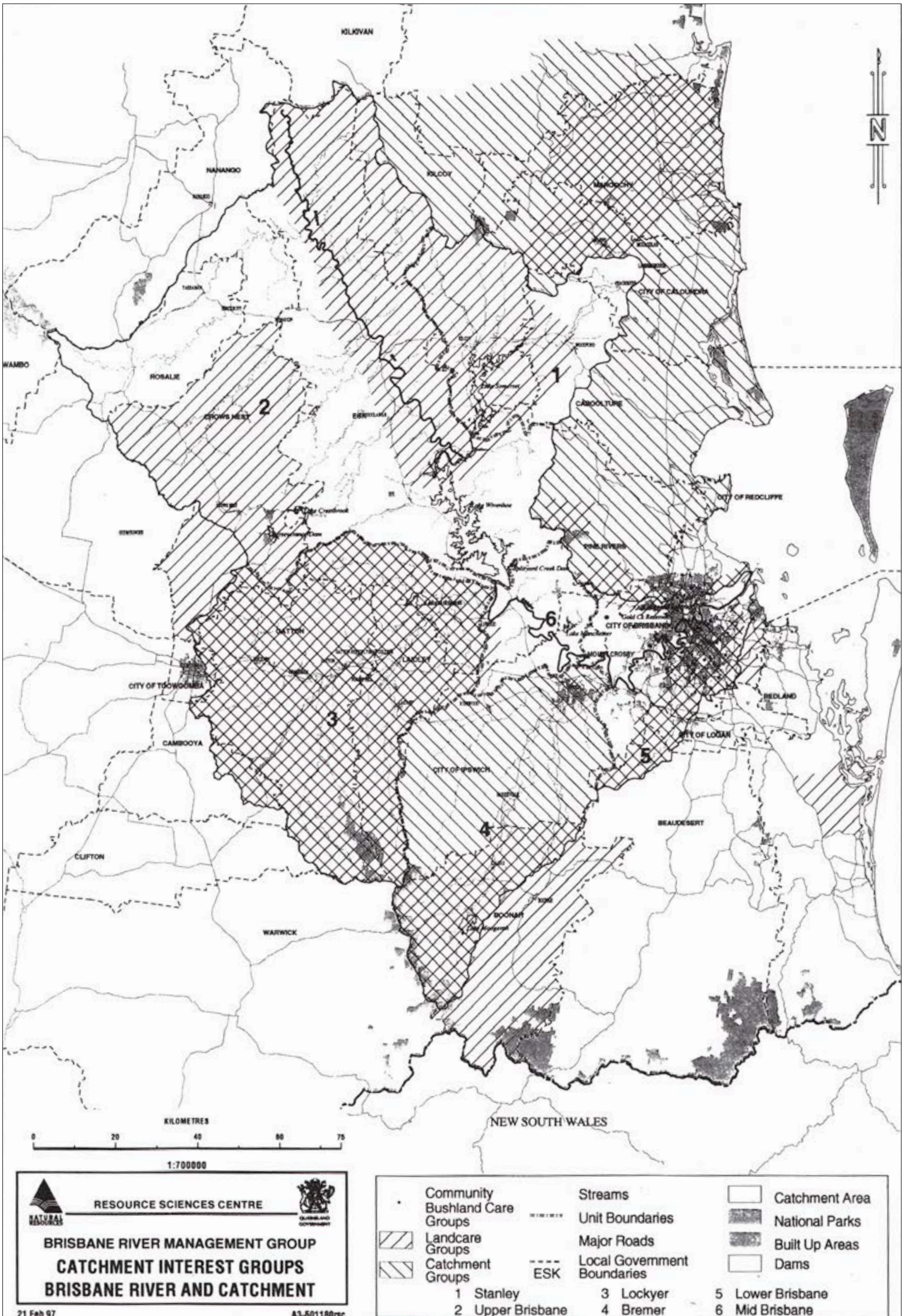

Figure 2. The scope of community groups in the Moreton Bay catchment.

to achieve the vision for healthy waterways. The vision reads: *"Moreton Bay, its land catchments and its waterways will, by 2020, be a healthy ecosystem supporting the livelihoods and lifestyles of people in southeast Queensland, and will be managed through collaboration between community, government and industry."*

The keys to achieving the vision and healthy waterways are strong community support for the change process, a well informed general public, clearly articulated strategies to achieve goals and government commitment to management needs. There is a need for some new structures and processes to coordinate, refocus and build on existing work on the important issues. To do this requires funds, expertise and commitment. Unless additional resources are available, many important new initiatives may never commence. By their very nature, some management practices which are needed will require significant resources. This applies particularly to improvements to wastewater disposal systems and rural and urban runoff. Nevertheless, commitment to such strategies is necessary for the health of Moreton Bay and its waterways.

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Towards a Comprehensive Management Plan for the Moreton Bay Subregion

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Introduction

The Moreton Bay subregion (the Bay) is of immense scientific, economic, and cultural significance for Queensland. It has provided a favoured area for settlement, a reservoir of resources and a sink for the disposal of waste. Growing concern has been expressed for the Bay in regard to the impact of past and current land and water use activities, the sustainability of current practices and the carrying capacities of various natural systems within the Bay. The requirements to safeguard the Bay's resources and to provide opportunities and options for future generations are now viewed as paramount. This intensity of focus and increase in scrutiny of Bay utilisation has come from many quarters, including many scientific and professional bodies.

The numerous scientific assessments of various components of the Moreton Bay environment, such as those exemplified by the papers in this volume, have universally noted that the major common thread linking all aspects of the Bay is the characteristic of "change". Authors from many disciplines have reiterated and reinforced the outcomes of previous scientific conferences and seminars in advocating the proper management of the various Bay resources and the "changes" that are occurring. Proper management requires a predetermined, proactive and comprehensive planning approach.

Planning is a professional activity with a major responsibility for the management of "change". Change is natural both in biophysical and socio-cultural environments, and along with human induced changes, is the concern of planning activity. Planning is the means of deriving and delivering policies and actions as part of a coordinated strategy or plan, to address a range of environmental problems and issues relevant to an agreed ***planning study area.***

In essence, the essential requirement of the contemporary planning process is to define the limits to acceptable change. It should be both transparent and participatory (i.e. provide opportunities for the active participation of all recognised stakeholders) and holistic (ranged across the biophysical and the socio-cultural environments).

The objectives of this paper are to review the condition of Moreton Bay, provide an overview of past and current planning activities, identify the planning framework in terms of the ruling dimensions and context for future planning and outline a process for future planning.

Past Scientific Interest and Concerns

The Moreton Bay subregion has been the focus of Aboriginal and European interest for centuries. Hall (1990) noted that from some 6 000 years ago, *"once sea level stabilised and the Bay was formed, people began taking more and more advantages of the estuarine and marine resources of the Bay"*. He argues that over time particular groups developed specific coastal-littoral/marine interests and that they, like their inland-terrestrial/riverine counterparts had an "*… impact on their environment ...* [that] *... may have caused profound changes to the local environment of the Moreton Region"*. This interest has continued into European times with earliest examples being associated initially with the requirement to provide safe and

In: Tibbetts, I.R., Hall, N.J. & Dennison, W.C. eds (1998) *Moreton Bay and Catchment.*
School of Marine Science, The University of Queensland, Brisbane. pp. 621-636.

convenient navigation, first to the original Moreton Bay penal settlement and then after 1842, to the newly developing Brisbane settlement. Subsequent to the appointment of the first official pilot in July 1827, the Bay was extensively surveyed in 1846 by Captain Wickham to chart navigational channels through northern Moreton Bay (Davenport, 1986). Davenport (1986) describes the continual survey work that has been needed in order to maintain safe navigation in response to the constant natural changes occurring in the Bay.

More recent interest has come largely from scientific quarters, namely in the form of a number of scientific conferences and symposia, including:

1974 – *Stradbroke Island Symposium* of the Royal Society of Queensland;

1978 – *Northern Moreton Bay Symposium* of the Royal Society of Queensland (Bailey & Stevens, 1979);

1984 – *Focus on Stradbroke Symposium* of the Royal Society of Queensland (Coleman *et al.*, 1984);

1989 – *Moreton Bay in the Balance Symposium* of the Australian Marine Science Consortium and the Australian Littoral Society (Crimp, 1992);

1993 – *Future Marine Science in Moreton Bay Conference*, School of Marine Science, The University of Queensland (Greenwood & Hall, 1993);

1996 – *Moreton Bay and Catchment Conference*, School of Marine Science, The University of Queensland (this volume)

In the first three cases, the Royal Society of Queensland sought to draw together scientific knowledge from disciplines such as geology, oceanography, anthropology, history, pedology, hydrology, botany, zoology and ecology for particular regions of the Bay that were considered to be of ecological significance and social value.

The last three cases have likewise drawn together the latest research findings, but at a time coincident with a period of major initiatives by the Queensland State Government to improve management for the Bay. This series of scientific benchmarks has provided specific guidance to the planning and management processes that have been operating during the 1990s. These works provide a valuable interface between the scientific research community and the planners and administrators, whilst also beginning to acknowledge the important role of the community in this process.

The background scientific, technical and general data available to support planning initiatives involving the Bay are quite extensive. Additional useful support references for planning include: Dowling, 1986; Oliver, 1989; Daniels, 1990; Quinn, 1992. These publications are supported by others which deal largely with the islands of the Bay, including: Gold Coast and Hinterland Wildlife Preservation Society (undated); Durbridge & Covacevich, 1981; Horton, 1983; Salter, 1983; and Rowe, 1991. A number of historical-cultural references to the Bay also have relevance and utility for future planning considerations (Ludlow, 1992; Pearn, 1993; Nolan & Longhurst, 1996). These works not only contribute to our understanding but can also enhance planning outcomes by providing important clues to such questions as Bay/island identity and character; genesis loci (special landscape qualities); the perceptions of resident and visitor and the cultural significance of the Bay.

History of Previous Planning

A number of island-focused planning and development-related studies has been undertaken. All of these investigations were concerned with a relatively narrow range of potential land uses, such as recreation, sand mining or land development, and were specific for the island concerned. None was undertaken as a serious forward planning exercise but rather as reactionary, *ad hoc* responses to conflicts in management, administration and land use. Examples include:

Moreton Island

1976 – Moreton Island Environmental Impact Study and Strategic Plan (A.A. Heath & Partners, 1976);

1976 – Proposed Management Plan for Moreton Island by the Brisbane City Council (BCC, 1976b);

1977 – Queensland Conservation Council's position on the future of Moreton Island (Claridge & Davies, 1977);

1977 – Queensland State Government's Committee of Inquiry – Future Land Use Moreton Island (Cook *et al.*, 1977);

1988 – The Lord Mayor's Statement of Principles for Moreton Island (BCC, 1988);

1989 – Statement of Intent formalising Moreton Island as a Recreation Area for management (QRAMB, 1989);

1991 – Moreton Island Recreation Management and Action Plans of the Queensland Recreation Areas management Board (Loder & Bayly, 1991a; b).

North Stradbroke Island

1983 – Consultant's study into a bridge proposal and development potential of the islands of southern Moreton Bay, including North Stradbroke Island (Cameron McNamara, 1983a);

1984 – North Stradbroke Island Development Strategy (Queensland Government, 1984).

Early planning-style studies undertaken at a macro or "regional" scale and inclusive of the Bay area, include:

1974 – Moreton Region Non-Urban Land Suitability Study (DPI, 1974);

1975 – Coastal Management Investigations (Gutteridge, Haskins & Davie, 1975);

1976 – Moreton Region Growth Strategy Investigations (COG & CC, 1976).

While these early studies noted the existence of Moreton Bay, none considered it to be of regional significance, or of sufficient importance to warrant separate and specific proposals and recommendations for its management. In fact, in all cases, the biophysical and socio-cultural role of the Bay within the region was overlooked.

Within the context of these early studies, a number of regional scale planning studies with a near whole-of-Bay focus were subsequently completed, including studies initiated by the Co-Ordinator General, Premier's Department to produce a Draft Improvement Plan for Moreton Bay (Cameron & McNamara, 1987). Such consultant reports formed the basis on which the first Strategic Plan for the Bay was derived some two years later (DEC, 1989). The objectives of this latter study as set out in the Summary Report were:

- to document existing relevant data on Moreton Bay and identify deficiencies in the existing data base;
- to identify important conservation elements and areas *(omitted from Main Report)*;
- to identify existing and potential opportunities for development of Moreton Bay;
- prepare a strategic plan indicating priorities for various uses in different parts of Moreton Bay;

- to identify necessary management functions and ways in which these might be achieved; and
- to recommend mechanisms for coordinated and rational decision making including monitoring and planning reviews to involve all agencies concerned with use, management and development of Moreton Bay resources.

This draft Strategy was never adopted. Some of the problems experienced with this initial planning initiative for the Bay included:

- planning and development proposals were inconsistent with the available scientific evidence and consequently the draft Strategy contained some technical implementation difficulties;
- the timing for the completion of the draft document was inappropriate, its release coinciding with a change of government, with the consequence that the Strategy did not have the support of the incoming administration;
- the planning process lacked adequate public consultation and consequently public support; and
- the planning study area did not include the whole Bay, nor did it include both aquatic and terrestrial areas, particularly the land-water interface zone which is subject to continual change.

In addition to the scientific conferences and symposia that were independently conducted in parallel to the planning process for the above Strategy, a number of other planning related initiatives were conducted by the Brisbane City Council (BCC). These initiatives made additional and important contributions to the broader planning process, and presented a local government point of view. They included the BCC sponsored Moreton Bay Seminar of April 1989 and Bay Search Seminar in November 1989 (BCC, 1989a; b).

The 1989 draft Strategy was subsequently re-released in September 1991 by the new State Government and, after a period of public review and consultation, a final Strategy was completed and adopted as policy in February 1993 (DEH, 1991; 1993a). This Strategy embraced a planning study area of greater geographical extent than its predecessors but again it was restricted to areas below the High Water Mark (Highest Astronomical Tide). It coincided with the boundary of the Moreton Bay Marine Park that was also declared in February, 1993 (DEH, 1993b). The stated goal of the strategy was:

> *To provide for ecologically sustainable use of Moreton Bay and to protect its natural, recreational, cultural heritage and amenity values* (DEH, 1993a).

Under the principal issue headings of: *Nature Conservation; Fishing and Collecting; Water Quality; Recreation and Tourism; Cultural Heritage; Landscape Character; Industry, Mining and Extraction; Terrestrial and Bayside Development; Transport*; and *Management*, the Strategy document contained some 72 objectives. Unfortunately it failed to include other policy or action proposals on how these objectives could be reconciled, prioritised and achieved. Likewise, responsibilities and timeframes for action and ongoing undertakings were not assigned. A draft zoning plan (DEH, 1993c) was released for comment at the end of 1993, and enacted in December 1997 (as subordinate legislation under the Marine Parks Act 1982) as the Marine Parks (Moreton Bay) Zoning Plan 1997. It was further amended in May 1998.

In the first acknowledged regional planning exercise undertaken for southeast Queensland (SEQ), the 1995 SEQ 2001 Regional Framework for Growth Management (RCC, 1995), the Bay received specific attention. Included amongst its recommended *Priority Actions* were a number of specific proposals for improved planning and management of Moreton Bay, including its Islands and the waterways associated with the Bay system. The South East Queensland Regional Framework for Growth Management 2001 (RCC, 1998) consolidates the previously described *Principles* and updates the *Priority Actions* set out in the regional planning initiatives of the SEQ 2001 project. The Bay's sensitive environmental characteristics and its special role as open space had been earlier recognised (Regional Planning Advisory Group: RPAG, 1993). Both land and water elements of the Bay were incorporated into the proposed Regional Open Space System (ROSS) for southeast Queensland (RPAG, 1993 and RCC, 1995).

Queensland has a long history of the State government delegating planning responsibility to local government. There are seven local authorities with frontages directly to the Bay or to the catchment, comprising Pine Rivers Shire, Redcliffe City, Caloundra City, Caboolture Shire, Brisbane City, Redland Shire and Gold Coast City (former Albert Shire) . The latter five also have jurisdictional planning responsibility for islands in the Bay. All have strategic plans which form part of their gazetted town planning schemes. A review of these contemporary statutory town planning schemes is provided in the following section.

The earliest example of a strategic plan for a "Bay" local authority is the Brisbane City Council's 1976 Statement of Intent – Part A (BCC, 1976a). A review of this and subsequent strategic plans of Brisbane City and the other local authorities reveals that the Bay did not receive acknowledgment as an issue for either management policy consideration or action. The earliest recognition of the Bay as a management entity was largely in terms of the linkages in recreational usage between it and the foreshore lands under Council control (see Section 11 – Recreation Zones, BCC, 1986). Redland Shire appears to be an exception, however, with its 1983 Coastal Management study that embraced sensitive environmental elements of the Bay foreshore within its boundaries (Cameron McNamara, 1983b).

In the case of other local authorities, the earliest comprehensive recognition, including the sensitive environmental nature of the Bay's ecosystem as well as the economic opportunities provided by those assets, appears to have been made only in more recent planning studies (BCC, 1990; 1991). These reports consider the Bay and Islands under separate headings, thereby acknowledging their individual importance. An example of a specific environmental management action arising from these studies follows:

> *Promote the conservation and retention of Moreton Bay and its islands in their natural and undeveloped state* (BCC, 1991).

This brief review of past planning studies suggests that there is a reasonably strong correlation between the growth in greater recognition and increased community awareness and understanding of the Bay, together with the community's maturity towards more specific environmental values. This has come at a time when the community has gained in their appreciation of the Bay's physical, social, cultural, aesthetic and spiritual role, its special contribution towards the achievement of improved "livability" objectives and the need to safeguard and maintain multiple use waterbodies of regional significance in pristine condition. If these planning studies represent the prevailing community values and attitudes, and as the community's attitudes towards the Bay mature, then it is reasonable to expect that future studies should grow in sophistication with respect to both recognition of the Bay and its environmental attributes, as well as in innovative proposals for sustainable management actions.

Most studies have had an exclusive single focus such as the land or the water component. The fact that they have not been holistic in terms of their geographic focus has been due largely to the legislative and administrative responsibilities of the agencies initiating the planning. These examples also provide evidence of past duplication (both intentional and unintentional) in the management and administration of the Bay's assets, particularly the islands. Together with the previously discussed scientific conferences and symposia, they serve to demonstrate that past interest has been spasmodic and focused in time as crisis management exercises responding to *ad hoc* proposals relating to the future of the Bay and its islands. They also demonstrate that there are conflicting and complicated issues involved and a range of different agendas on how to proceed with the forward planning that will be necessary to achieve acceptable outcomes.

Current Environmental and Planning Activities

Current planning activity in relation to the Bay occurs in response to two relevant but distinct statutory planning requirements: the Integrated Planning Act of 1997 (IPA) and the Coastal Protection and Management Act (CPMA) of 1995 as amended. The IPA came into effect in April 1998 to replace the *Local Government (Planning and Environment) Act* of 1990 and all existing local authority statutory planning schemes established under the latter legislation are still valid. However, under the IPA legislation, Councils have a five-year period within which they must derive a new set of planning instruments to replace these existing schemes.

The stated purpose of the new legislation is to achieve ecological sustainability by:

- integrating local, regional and state planning and decision making;
- managing the development process; and
- managing the effects of development on the environment.

Local authorities have the delegated statutory responsibility for the control of activities on land (i.e. above the High Water Mark of the coast and coastal waterways), which they exercise through their respective Town Planning Schemes. The CPMA requires that Regional Coastal Management Plans be completed for various coastal regions of the State under the umbrella of the State Coastal Management Plan.

The South East Queensland Regional Coastal Landscape Assessment is an associated initiative to the SEQ Regional Coastal Management Plan activities. Under the direction of the (then) Department of Environment, consultants developed a methodology for the assessment of regional coastal landscapes. This methodology has been applied to four regions, including the southeast Queensland region.

This initiative has resulted in the assessment of a range of scenic and cultural landscape values associated with the coast, and the identification of significant landscapes for management. This was accomplished through the previously mentioned regional coastal management planning process as well as the statutory land use planning process under the responsibility of the local authorities. The desired coordination between these two levels of planning, management and governance under the CMPA has yet to be developed and demonstrated.

Examples of the various types of plans, planning instruments and planning activity that currently relate to the Bay include statutory plans such as Town Planning Schemes of Local Authorities (including Strategic Plans, Planning Scheme Provisions, Zoning & Regulatory Maps, and Development Control Plans), Regional Coastal Management Plan for the SEQ Region, Non Statutory (Advisory) Plans such as Corporate Plans of Local Authorities, (Outdoor) Recreational and/or Tourism Strategies, Economic Development Strategies, General City-wide Planning Studies (e.g. Brisbane 2011) and intermediate plans (which may lead to a statutory outcome) such as various types of Wastewater Management Strategies and Resource Management Strategies.

The philosophical shift to a comprehensive environmental position towards the Bay that was first advocated in BCC (1991) has endured and has flowed through to the current City Strategy (BCC, 1996b). This position is supported by the Council's State of the Environment Report (BCC, 1996a), which devotes an entire chapter to the "Coast and Moreton Bay". A similar position has been adopted by recent initiatives of the Redland Shire Council, which has the largest frontage to the Bay as well as responsibilities for the majority of the Bay islands. Examples include their recently completed Corporate Plan (RSC, 1996a) and statutory Strategic Plan gazetted in February 1998. These works drew upon earlier non-statutory (advisory) studies such as the Council's Tourism Strategy (Coopers & Lybrand, 1994) and Open Space and Recreation Strategy (Loder & Bayly, 1992). Local authorities, however, do not have jurisdictional responsibility for the entire area.

The *"Southern Moreton Bay Islands Planning Strategy"* is currently underway under the auspices of the Departments of Local Government & Planning and Environment and the Redland Shire Council. This project seeks to define a planning and land use framework for the management of future development and conservation of Russell, Lamb, Macleay, Perulpa and Karragarra Islands (DLGP, DoE & RSC, 1996). The planning study is expected to address the controversial issues of land subdivision – past and future, development in drainage problem areas and environmentally sustainable population growth levels. Launched in September 1996, the project completed a draft planning and land use strategy in October 1998 and it is expected to be adopted by early 1999 (DHLGP, 1994; DLGP, DoE & RSC, 1996). Of particular note is the Redland Shire Council's joint initiative, the Quandamooka and North Stradbroke/ Minjerribah Planning and Management Study, which is currently a quarter completed.

The preparation of the Regional Coastal Management Plan for the southeast Queensland region is the responsibility of the Department of Environment and Heritage and is being undertaken under the umbrella of a State Coastal Management Plan that has yet to be completed and will embrace the existing Moreton Bay Strategic Plan and the Zoning Plan. This project has commenced, although its timelines have now been extended which will result in a draft report by mid 1999 (twelve months later than previously announced) and a final regional plan by mid-2000.

The stated objectives of the draft regional plan are: to ensure the management of the coastal zone is comprehensive, coordinated and effective; ensure coastal resources are available for ecological sustainable development and activity; and to conserve the natural and cultural resources of the coast. The project is to focus on the establishment of a "control district" embracing the sea-land interface. Whilst the subject area is to include Moreton Bay and the Marine Park, the landward extent has yet to be determined. This uncertainty reinforces the previously made comments in regard to the "planning framework", in particular, the "planning study area and its regional setting".

An associated study that is currently underway and has the potential to have a significant impact on the future of the Bay in both a statutory and an advisory capacity is the *Brisbane River and Moreton Bay Wastewater Management Study* (BRMBWMS). This program was commenced in January 1994 in direct response to Priority Action outcomes of the SEQ 2001 Regional Framework for Growth Management that recommended the development of regional water quality strategies and guidelines (RCC, 1995). An Interim Water Quality Management Strategy, with an emphasis on the Brisbane River system, has already arisen from the Study.

Moreton Bay Catchment Water Quality Management Strategy

With a focus on the estuarine environment, the scientific studies supporting this initiative confirm that the waterways of this region are facing real threats. The strategy outlines a range of management actions that cross institutional and geographic boundaries. Again it is noted that the only solution can be through a cooperative approach involving all levels of government, industry and the community of the Bay region (BRMBWMS, 1998a). It is proposed that the ongoing initiative of Stage 3 of this project will now embrace a wider geographic area for scientific study: to include the freshwater environment. It is also proposed to include all river systems with an influence on the water quality of the Bay, which had hitherto been excluded.

These proposed initiatives are consistent with the recommendations and priorities of the SEQ Regional Framework for Growth Management which sought the development of a Regional Water Quality Management Strategy for southeast Queensland. A wastewater management study has been completed for the southern Moreton Bay area including the river systems of the Logan, Albert and Coomera Rivers (Sinclair Knight, 1994).

The South East Queensland Outdoor Recreational Demand Study

Whilst this current study does not focus on the Bay it is applicable to the wider region. Subsequent initiatives resulting from its recommendations could provide useful data for the development of recreational management policies. The integration of these policies will be crucial to the achievement of successful management of the Bay, especially as the Bay develops further as a recreational resource and destination.

It will be particularly important to achieve sustainable outcomes whilst reconciling competing activities, perhaps none more important than that of recreational and commercial fishing (see also Moreton Bay Task Force, 1997). These recent initiatives provide significant reinforcement to the previous point concerning the contemporary focus on the Bay and the imperative to derive a sustainable management strategy that embraces a wide range of stakeholder concerns and views.

The shortcomings of past planning studies of particular note is the exclusive single focus on either the land or the water, but not both. This is largely due to the legislative and administrative responsibilities of the agencies initiating the planning activity. There is also evidence of continuing duplication in the management and administration of Bay assets, particularly the islands. Again, this highlights the ongoing, conflicting and complicated issues that are yet to be resolved. It suggests that a workable planning framework remains elusive.

The Planning Framework

The framework adopted for future planning (Low Choy, in prep.) should satisfy the following principles:

1. It should embrace a regional setting which allows the inclusion of all elements and issues of **regional significance**;
2. The scope of the study should be **comprehensive** and multidisciplinary and it should embrace the biophysical and socio-cultural elements of the marine and the terrestrial environments of the Bay;
3. Planning considerations need to be based on **scientific knowledge**;

4. The underlying planning philosophy should embrace the **environmental planning principles** of diversity, sustainable development, environmental carrying capacity, equity and the precautionary principle;
5. The planning study area should approximate a **natural area**, and be delineated on the basis of an ecosystems or biophysical approach, without regard to the existing legislative and administrative arrangements;
6. It should be a **democratic and participatory** process that facilitates the maximum involvement of all stakeholders;
7. Future planning should promote a cooperative approach that involves the community at all levels of government in **partnership** arrangements;
8. It must be capable of resolving conflicts, but more importantly **managing potential conflicts** before they arise;
9. It should be an open and **transparent** planning process that achieves and retains the confidence of all participants;
10. It should be capable of producing a viable range of **alternative options**.

The planning study area and its regional setting

The delineation of the "planning study area" should be consistent with the above principles. It should be capable of achieving consensus from the outset. This is necessary in order to focus the initial investigatory tasks on the collection of relevant environmental data in response to the predetermined planning study objectives. It is also required in order to establish a geographic framework for the policy and action outcomes of the planning process. This will ensure that subsequent policies that are derived are relevant to a specific geographic area and that the responsibilities for implementation can be identified unambiguously.

Ideally it should bear some correlation to the South East Queensland Planning Region and embrace the Moreton Biophysical Region that comprises:

- The catchments of major regional waterways associated with the Bay (i.e. the Brisbane River system, the Albert and Logan Rivers, together with the Pine and Caboolture Rivers);
- Intervening foreshore land within the limits of coastal influences but not associated with the catchments;
- The offshore islands (Bribie to South Stradbroke); and
- The Marine Environment (as delineated by the Moreton Bay Marine Park).

In view of the size of the area involved and the number and complexity of potential issues, it would be appropriate to focus the planning study towards a two-tiered structure. The primary tier would comprise the marine environment of the Bay, the offshore islands, and the immediate foreshore zone of the mainland. The associated catchments, when added to this primary study area would constitute the secondary tier and the immediate regional setting for the study. It will also be necessary to incorporate biophysical, and socio-cultural inter-regional linkages for both the marine and terrestrial environments.

Institutional arrangements

Preliminary consideration of the institutional arrangements will need to examine the history of government agency activity in Bay planning and management. It should investigate in particular, their effectiveness in the environmental planning and management fields. This review should also include qangos, including statutory authorities such as:

- Beach Protection Authority (BPA)
- Gold Coast Waterways Authority/Gold Coast Harbour Authority
- Queensland Recreation Areas Management Board (QRAMB)
- Moreton Island Planning Advisory Committee

Consideration of a suitable institutional arrangement to facilitate and oversee future planning activities for the Bay will need to take cognisance of the existing array of overlapping jurisdictional responsibilities for the coastal zone. QUT (1992) provides a useful analysis of these circumstances. The horizontal and vertical overlap and duplication between State agencies and between local authorities presents both existing and potential problems for planning undertakings that will have to be overcome before planning can commence.

Options for institutional models range from *ad hoc*, committee-type groupings, to more formal arrangements including new statutory organisations, such as the proposed Moreton Bay Commission. The previously mentioned review of government agencies and qangos can provide valuable insight into likely problems with these arrangements.

Successful planning outcomes will be improved if a workable institutional arrangement that facilitates cooperative planning and management can be achieved. The structure of the arrangements should be flexible to facilitate the public involvement/participation program and allow for the involvement of all stakeholders. This model must be capable of facilitating the planning process, including the implementation phase, whilst cutting across the existing institutional arrangements to allow an holistic approach to be adopted in both planning and management implementation.

The Principal Stakeholders

A preliminary stakeholder analysis (Low Choy, unpubl. data) has revealed the existence of the following categories of stakeholders for a Moreton Bay planning exercise (both internal and external to the Bay):

- Recreational users (e.g. recreational fishing; yachting; power boating; scuba and snorkelling; camping; bushwalking etc.)
- Conservation/Environmental Groups
- Artisans
- Historians
- National Trust
- Aboriginal and Torres Strait Islander Communities
- Trade Unions
- Scientific and Academic Community
- Schools
- Youth Groups
- Professional Groups
- Island Residents (full and part-time)
- Tourists
- Community/Resident Action Groups
- Community Service Organisations
- Commercial Fisherman
- Real Estate Agents

- Land Developers
- Miners
- Extractive Resource Users (including water)
- Tourist Operators
- Other Commercial Interests
- Marine & Terrestrial Transport Agencies (Public and private)
- Emergency Services (including Coast Guard and Police)
- Local Authorities
- State Government Agencies
- Commonwealth Government Agencies
- qangos (e.g. Port of Brisbane Corporation)

The Key Planning and Management Issues

The identification of the key planning and management issues that are relevant to the Bay is beyond the scope of this paper, however, they should be consistent with the principle of comprehensiveness and include issues from across the biophysical and socio-cultural environments. An initial grouping of issues can be sourced from the numerous environmental and planning studies that exist, but they would require confirmation through a contemporary community consultative process. The various surveys of island residents and visitors would also prove useful in this regard (see in particular: A.A. Heath, 1976; BCC, 1996a; BCC, 1996b; Coopers & Lybrand, 1994; Loder & Bayly, 1992: Pearn, 1992; QUT, 1992; and RSC, 1996a).

A Planning Process

Consideration should be given to a planning process of two phases, plus a preliminary phase that is consistent with the previously discussed principles. This would comprise a **Preliminary Phase** that should involve a scoping study to confirm the following dimensions: the planning study area; principle stakeholders; elements and issues of regional significance; key issues; and institutional arrangements for the planning activity.

Phase 1 would involve the completion of a comprehensive strategic plan for the entire planning study area. This can be accomplished as a composite compiled from past strategies including: A.A. Heath (1976); Cameron McNamara (1983b); Loder & Bayly (1991a; b; 1992); Crimp (1992); DEH (1993a, b & c); Greenwood & Hall (1993); RSC (1993; 1995; 1996a; b); Coopers & Lybrand (1994); ASC (1995); BCC (1996a; b); DLGP, DoE & RSC (1996); WBM Oceanics & SKM (1996); Brannock & Humphreys (1997a; b); BR&MBWWMS (1998a; b) and DEH (1998). This minimalist approach would require confirmation against the perceptions and attitudes of a contemporary stakeholder group.

Phase 2 planning activities should be in the form of a continuous process that includes the following stages:

- decision to plan (political);
- confirmation of the *planning study area*;
- confirmation of stakeholders;
- implementation of public involvement/participation;
- research initiatives flowing from the identified problems and issues;
- confirmation of the key issues (Delphi study);
- identification of the problems;

- formulation of the general study goals and specific, measurable objectives;
- collection and collation of environmental data (in response to planning objectives);
- completion of a strengths, weaknesses, opportunities and threats (SWOT) analysis to examine the strategic issues;
- identification and analysis of the potential constraints and opportunities;
- research and development of relevant standards;
- derivation of a statement of planning and management principles;
- prioritisation of objectives;
- projection of future scenarios;
- generation of alternative courses of action;
- evaluation of alternatives and reconciliation of conflicting objectives;
- development of a preferred plan (including policies, programs, procedures and actions);
- development of an action plan for implementation; and
- implementation, continuous monitoring and the establishment of a feedback loop into the management process.

A Way Ahead

Future planning for the Bay should occur within the context of a suitable institutional arrangement that is capable of embracing cooperatively all levels of government and other stakeholders, in appropriate partnership arrangements. These arrangements should be consistent with the requirements previously discussed. Planning can provide the required proactive framework for improvement of the deliberate environmental management of "change". It can pursue multiple objectives, be integrative and provide coordination of management actions and activities. Importantly, it will provide a choice of options for the decision-making process.

A Management Plan, not another Strategy, is required, although a comprehensive Strategy embracing both marine and terrestrial components of the Bay, will be a necessary requirement in the first instance. Hence the two phase planning process is recommended, which will allow a composite strategy to be assembled with a minimum of delay and repetition. A comprehensive outcome will be achieved by the inclusion of all issues of scientific interest, and through the bridging of both land and water issues associated with the environmentally sensitive tidal zone that is constantly undergoing change.

Most importantly, the final planning outcomes must be capable of being implemented and achieved. The study must not produce another advisory document: a fate that has typically befallen many of the previous planning exercises. The continued improvement in awareness and education of residents of the region, visitors and stakeholders of the Bay, will also be imperative to the success of the plan.

A proactive, comprehensive and forward planning approach will significantly improve the chances of attaining an acceptable management outcome with community ownership for the many conflicting Bay interests. The achievement of such an outcome, before an environmental crisis can arise, eliminates the costly *ad hoc* response of convening a special inquiry or commission – a typical reaction that has characterised the approach of many governments to environmental crises of the past.

Integrated Planning Act of 1997

A more comprehensive approach is emerging, particularly in recent attempts to understand the complex nature of the Bay and its related parts, however, this can not yet be said of the interlinkages between the professions who must provide the holistic advice to the political decision-makers. This is still very much fragmented as is the government response to the complex question of complete Bay management.

Encouraging signs, however, do exist, particularly in the form of legislation such as the Integrated Planning Act, and certain institutional arrangements such as the proposed Environmental Protection Act.

Individually, the majority of these recent initiatives show signs of embracing more holistic approaches, linking science-based foundations to proactive planning and management. Additionally, there is also a consistent theme which runs through the stated objectives of these recent initiatives, i.e. the desire to achieve effective coordination. It must also be presumed that these coordination objectives refer to the plan implementation aspects as well as to the plan making activities.

With the possible exception of the IPA intentions, which have yet to be demonstrated, let alone achieved, there is as yet no effective cooperative mechanism capable of producing a more integrated set of holistic policy initiatives, nor overseeing their effective implementation. In the quest for effective **planning for change** the avoidance of future **management with uncertainty** is perhaps the "holy grail" of planning and management of our Moreton Bay.

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Glossary

Acid sulfate soil	—	Waterlogged soil containing pyrite (FeS) which is converted to sulfuric acid (H_2SO_4) upon aeration.
Aeolian	—	Wind borne.
AHD	—	Australian Height Datum: Reference point for tide levels set at current sea level.
Anthropogenic	—	caused or produced by humans.
Ascidian	—	animals without backbones except in larval stages; sac-like body with two siphons for water exchange (in and out); a type of tunicate.
Barnacles	—	filter-feeding animals with calcareous shells that attach firmly to substrate.
Bioturbation	—	sediment reworking by animals (e.g. burrowing, boring)
Bycatch	—	incidental capture of various sea creatures during fishing/trawling that are not the target species.
Chlorophyta	—	Green algae, containing the green-colored chlorophylls *a* & *b* photosynthetic pigments.
Copepod	—	tiny swimming crustaceans that feed primarily on phytoplankton; "sea fleas".
Crustaceans	—	animals with hard shells in the Class Crustacea (e.g. shrimp, lobsters, crabs, barnacles).
Cumaceans	—	a small (5-10 mm) shrimp-like crustacean found in marine areas, especially near the seafloor in estuaries. Important as food for juvenile fish.
Cyanobacteria	—	(blue-green algae) – primitive and widespread plant life that can convert atmospheric nitrogen into plant tissue, growth often filaments.
Delta	—	region of sediment accumulation where water velocity gradients occur.
Denitrification	—	microbially mediated process in which nitrate (NO_3^-) is converted into nitrogen gas (N_2)
Detritus	—	fragments of dead plant material undergoing decay.
Diel	—	24-hour period combining diurnal (daytime) and nocturnal (night-time).
East Australian Current	—	a warm water current from N to S along the east coast of Queensland.
Environmental flow	—	the controlled release of water from dams to restore river processes.
Epibenthic	—	living on, not in, the sea bottom.
Estuarine	—	pertaining to water that is partially influenced by both marine and freshwater sources.
Exotic species	—	non-native organisms that have been imported to a region.
Food webs	—	the inter-connected relationships between organisms relating to their various food sources.
Holocene	—	last 10 000 years.
Infauna	—	animals that live in, as opposed to on, sediments, e.g. worms.
Interstitial	—	porewaters in between sediment grains.

Macroalgae — large (visible to the naked eye) marine plants that include Chlorophyta (green algae), Rhodophyta (red algae), Phaeophyta (brown algae).

Macrobenthos — large (visible to the naked eye) animals that live on the benthos (seafloor); e.g. fish, crabs, prawns.

Macrofauna — large, visible to the naked eye, animals.

Macrophytes — large (visible to the naked eye) plants; e.g. mangroves, seagrasses.

Mangrove — salt-tolerant trees that live in the inter-tidal zone.

Megalitres — one million litres (ML)

Meiofauna — minute animals that live in the sediments; (retained on 65 µm seive)

Mollusc — animals that generally live within calcareous shells (Phylum Mollusca).

Nitrogen fixation — microbially mediated process in which nitrogen gas (N_2) is converted into ammonium (NH_4^+)

Nutrient — an element used by organisms for structure and/or metabolism; for example, nitrogen (N) and phosphorus (P).

Phaeophyta — Brown algae, containing the green-colored chlorophylls *a* & *c*, and yellow-orange coloured fucoxanthin photosynthetic pigments.

Phytoplankton — minute plants that float in the water (diatoms, dinoflagellates).

Pleistocene — 2 million years ago until 10,000 years ago.

Productivity — a measure of energy flow within an ecosystem; primary productivity = rate of sunlight converted into chemical energy by primary producers (plants); secondary productivity = rate of consumption of primary producers by animals.

Quaternary — last two million years (Pleistocene and Holocene).

Ramsar — An International Convention (held in Ramsar, Iran) in which wetland areas important to migratory wading birds were identified.

Rhodophyta — Red algae, containing the green-colored chlorophyll *a* & the red and blue colored phycobilins photosynthetic pigments.

Salt-marsh — salt-tolerant grasses, sedges and rushes that live in the inter-tidal zone.

Seagrass — flowering plants that grow totally submersed in seawater.

Scleractinian — “hard” corals comprised of a cnidarian host and a dinoflagellate symbiont (zooxanthellae) which secrete calcium carbonate skeletons.

Sponge — a primitive animal with a porous structure (Phylum Porifera).

Turbidity — a characteristic of water in which suspended particles act to attenuate light.

Zooplankton — minute animals that float in the water.

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Moreton Bay and Catchment represents a timely review of our knowledge of the health of the waterways of southeast Queensland. It details the history of the Bay, present usage, estimates of population growth and information on the geology, water quality, fauna and flora. It offers a way forward in our objective for a diverse and healthy Bay as we set out on the third millennium.

While scientific in most of its structure and content, *Moreton Bay and Catchment* will be of enduring value as a resource for scientists, managers, teachers, students and conservation and community groups.

Published by:

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The University of Queensland
Brisbane Qld 4072
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