Innovative Solutions for Aquaculture Planning and Management – Project 5, Environmental Audit of Marine Aquaculture Developments in South Australia

Prepared for PIRSA Aquaculture and FRDC

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OBJECTIVES:

1. Review the current environmental status of marine aquaculture in South Australia by assessing the level and adequacy of existing information and information collection protocols in relation to environmental impacts.
2. Assess and prioritise the actual and perceived environmental impacts of marine aquaculture in South Australia using a formal risk assessment framework.
3. Investigate identified high priority environmental impact issues through targeted field based R&D, including the development and evaluation of methodologies and sustainability indicators.
4. Develop aquaculture sector-based optimal environmental monitoring programs, including identifying the parameters to be measured (environmental as well as farm management), the spatial and temporal frequency of monitoring required, and select critical decision points against which ESD performance can be measured.
5. Support PIRSA Aquaculture by providing scientific and technical input into discussion with other stakeholders during the process of negotiating and implementing the desired changes to the existing system.

NON TECHNICAL SUMMARY:

OUTCOMES ACHIEVED TO DATE

This report provides a detailed assessment of the impacts of yellowtail kingfish aquaculture in Fitzgerald Bay, and of land-based abalone around South Australia, on a range of benthic and pelagic environmental variables. The results indicate that the environmental impacts of both of these sectors are minimal as currently undertaken, and provide the basis for the development of environmental monitoring protocols for each sector. A range of potential options for each sector is outlined. PIRSA Aquaculture intend to undertake a review of current monitoring requirements based on these suggestions. Due to the low level of environmental impacts, it was not possible to identify unequivocal sustainability indicators for these sectors.

Yellowtail kingfish aquaculture in Fitzgerald Bay is currently having minimal environmental impact. While some effects of farming were detected on several sediment chemistry parameters (porewater nutrient levels, total organic carbon, and total nitrogen), there were no effects on benthic fauna that could be clearly attributed to aquaculture. Instead, there were often complex patterns of small-scale spatial variation, presumably related to natural variation in the environment. There was a relationship between seagrass biomass and proximity to leases, however, this was not reflected in the other seagrass parameters measured, and because of a lack of data from prior to the commencement of aquaculture, it is difficult to determine what the cause of this pattern might be. Analysis of the seagrass elemental composition did not show any differences, and does not appear to indicate that they are suffering from increased nutrients, which is supported by a lack of differences in epiphyte loads. In the water column, an increase in ammonia levels could be detected next to cages, but this was not reflected in phytoplankton abundance or composition. Based on these results, several options for revising the environmental monitoring program for finfish are presented, backed up by power analyses that indicate the required level of sampling to determine if an effect is occurring.

A pilot study on light availability off Cape Jaffa and Kingston SE indicates that light penetration is low, and a simple productivity model is used to show that conditions for seagrass growth in all but the shallowest areas are as marginal as, or even worse than, at the deep limit for seagrasses off the Adelaide coast. As the deep limit is defined by low light, it is clear that any reduction of light in these regions may be detrimental to the long-term survival of any seagrass beds in the region. This suggests that any aquaculture undertaken in this region would have to be conducted in such a way as to minimise light reduction to seagrasses, although it must be recognised that this conclusion is based on only a few weeks of data, and it would be useful to obtain data from other times of year to back this up. The current strategy for achieving this is to use swing moorings, which allow the cages to move with the currents, rather than keeping them fixed in place.

Surveys of land-based abalone farms were conducted at the three main farming regions of Smith Bay (Kangaroo Island), Point Boston (near Port Lincoln), and Streaky Bay (west coast). At all three regions, discharge waters contained elevated levels of dissolved nutrients that can be detected in adjacent intertidal and subtidal waters, and in the nitrogen content of subtidal seagrass (*Posidonia*). Discharge waters transform intertidal habitats into subtidal habitats that have vastly altered communities to adjacent intertidal areas. The spatial extent of this impact is restricted to the areal cover of discharge water, which is relatively small compared to the areal cover of intertidal habitat in each farming region. Due to the lack of data before farms commenced discharging, it is not possible to conclude whether any differences observed between farm and non-farm sites are due to farm discharges. Nonetheless, some site-specific and subtle differences were detected adjacent to the abalone farms that are consistent with the known

effects of increased nutrients on subtidal marine communities, viz. some evidence of a negative impact on seagrass (*Posidonia*) at Smith Bay, strong evidence of a negative impact on canopy-forming macroalgae at Point Boston, and a greater gastropod (viz. *Turbo*) density at Smith Bay and Point Boston. Importantly though, no major changes to subtidal communities have occurred due to farm discharges, with diverse communities of macroalgae, seagrasses and invertebrates still present directly adjacent to farm outfalls. The spatial extent of the apparent subtidal changes at Smith Bay and Point Boston is unknown. The apparent low level of impact in subtidal waters adjacent to farms is probably related to the relatively low nutrient concentrations and low total annual nutrient loads. Escaped abalone are having no impact on the adjacent marine environment, as they apparently do not survive once they leave the farm outfall pipes. There was no evidence of seagrass scouring around intake pipes at the two farms investigated. A number of options for a land-based abalone environmental monitoring program are proposed.

Prolonged heavy shading is known to kill *Posidonia* seagrass. A sublethal shading effect of BST long-line baskets on *Posidonia* seagrass was apparent at the three sites surveyed at South Spit, Stansbury, South Australia. A lethal shading effect of BST long-line spat trays on *Posidonia* seagrass was observed at one of the sites at South Spit, Stansbury, South Australia. Whilst the relative area and degree of shading impact on *Posidonia* meadows from BST long-line culture at South Spit is low, lethal and sublethal impacts could be reduced with the following farming practices: (1) BST long-line baskets to be kept suspended above the seabed as high as practicable, (2) Use of a rotational schedule for stocking BST long-lines with baskets, and (3) BST long-line spat trays to be moved every few months. While trampling can affect seagrass aboveground biomass and could potentially affect razorfish, there was no evidence for a trampling affect on *Posidonia* or *Pinna* at South Spit, Stansbury, South Australia. There was no evidence for an effect of chemical leaching (viz. copper chromium and arsenate, CCA) from treated posts on benthic infauna at South Spit, Stansbury, South Australia.

KEYWORDS:
Yellowtail kingfish, land-based abalone, environmental impacts of aquaculture, environmental monitoring programs, light model

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Background

The aquaculture industry in South Australia is in the process of undergoing expansion. In response to this, PIRSA Aquaculture was in the process of revising the management plans for a number of key aquaculture areas when this project was proposed in 2002. Subsequent to this, the Aquaculture Act (2005) was introduced, and PIRSA Aquaculture has been developing aquaculture zone policies for a number of key aquaculture areas to either consolidate the existing industry in the area or provide for allocation of additional area for further industry growth. As part of this revision, SARDI Aquatic Sciences undertook a series of comprehensive field surveys of potential aquaculture areas in 2002, and made assessments of the potential impacts of any aquaculture in these areas. There was a major problem in conducting these assessments, however, in that we had a very poor understanding of how aquaculture affects the environment (except for tuna farming). This lack of information also contributes to a substantial negative perception of aquaculture amongst elements of the general public, and continual challenges to any proposals for expanding aquaculture.

One of the major sectors of concern is finfish, the culture of which involves large feed inputs. Waste feed and faecal material then directly enters the environment, where it may have adverse impacts on the biota present. For example, it is well documented that finfish aquaculture in close proximity to seagrass nearly always has a negative effect on the seagrass (although in SA there are strategies in place to try and minimise this). Epifaunal and infaunal assemblages may also be affected. Indeed, infauna are regarded as one of the best groups to study in environmental monitoring programs, as they are both sensitive to elevated nutrient inputs, and ubiquitous in soft sediments. Thus, for example, the Tuna Environmental Monitoring Program provides a good example of a well designed monitoring program focussing on infauna.

Shellfish aquaculture is generally considered to be more benign, as most cultured shellfish are bivalve filter feeders that extract food from the environment.

There are thus no artificial food inputs in these systems. However, this extraction in itself could be harmful to naturally occuring filter-feeders if it induces substantial competition for food. The deposition of faeces and pseudofaeces by bivalves may also result in localised nutrient enrichment, and there will be some-scale physical disturbance from farm operations (e.g. trampling). In South Australia, there have been several small-scale studies that have examined the consequences of intertidal oyster culture for nearby seagrasses, with the tentative conclusion being that impacts are very localised (within 1-2 m of racks). The nature and extent of these impacts was still regarded as an extremely contentious issue when this project commenced, however, and have previously been the basis for frequent court action to prevent the establishment of new oyster leases. Early in the project, it became clear that these problems were becoming much less of an issue, and thus the focus on oysters was substantially reduced.

Onshore abalone farming probably lies between open-water finfish and bivalve farming, in terms of its impact on the environment. This industry relies on feed inputs, but being land-based there is considerable scope to reduce the amount of waste released to the ocean by the use of settling ponds etc. The outfall from an abalone farm may still be a substantial source of nutrients, however, with consequent adverse effects for the surrounding environment.

Given the recent expansion of aquaculture in South Australia, and its scope for growth, it is timely to take stock of the current situation with respect to the environmental performance of existing operations. This will then allow best environmental practise to be identified and scientifically defensible environmental monitoring programs to be developed and implemented, thus minimising future impacts of the aquaculture industry on the environment and improving public perception of the industry. This knowledge will also allow managers to plan with greater confidence, and give the industry greater security of access to resources, improving the likelihood of continued investment and growth.

Need

Aquaculture is a rapidly growing industry in Australia, and as such there are substantial resource allocation issues. South Australia is at the forefront of this development with a range of innovative aquaculture industries, an active group in PIRSA Aquaculture addressing policy and management issues, and another in SARDI Aquatic Sciences providing the scientific and technical background information for such matters through targeted research and development (R&D). As such, South Australia provides an ideal model for other States.

While a reasonable level of information exists and, through the Aquafin CRC, continues to grow for tuna farming, this is not the case for most of the other SA marine aquaculture industry sectors. The purpose of this application is therefore to gather and review the existing information, assess the actual and perceived environmental impacts of the key industry stakeholders, investigate through targeted

R&D key environmental impact issues, develop industry sector based environmental monitoring programs and support PIRSA Aquaculture in implementing the outcomes from this project. Apart from tuna, there is currently very little information on the environmental impacts of finfish farming in South Australian waters, especially for the rapidly growing yellowtail kingfish sector. In order to obtain public support for further development of this industry, it is essential that its environmental impacts are assessed, and strategies implemented for reducing these impacts. Similarly, for shellfish farming, at the time that this project was proposed there was still a great deal of contention about impacts on the ecosystem, particularly seagrasses, which co-occur with the largest shellfish aquaculture industry in the state – Pacific oysters. Oyster farming has become less contentious since the inception of this project, and PIRSA Aquaculture have instituted a policy of no more rack and rail systems over seagrass, reducing the potential for damage to seagrasses.

This project will therefore provide background information for improving and further developing the project "Innovative solutions for aquaculture planning and management – Project 2, Spatial impacts and carrying capacity: Further developing, refining and validating existing models of shellfish and finfish carrying capacity". Both projects will provide much of the scientific and technical data for input into the project "Innovative solutions for aquaculture planning and management – Project 1, Decision support system for aquaculture development", where "Decision support system" is defined as a computer based, integrated method for supporting management decisions. Decision support systems must incorporate rigorous and scientifically sound decision criteria and, as such they require a good understanding of the potential environmental impacts that may result from aquaculture, as well as the characteristics of existing or future farm sites and the ecosystem in which they exist.

Objectives

1. Review the current environmental status of marine aquaculture in South Australia by assessing the level and adequacy of existing information and information collection protocols in relation to environmental impacts.
2. Assess and prioritise the actual and perceived environmental impacts of marine aquaculture in South Australia using a formal risk assessment framework.
3. Investigate identified high priority environmental impact issues through targeted field based R&D, including the development and evaluation of methodologies and sustainability indicators.
4. Develop aquaculture sector-based optimal environmental monitoring programs, including identifying the parameters to be measured (environmental as well as farm management), the spatial and temporal frequency of monitoring required, and select critical decision points against which ESD performance can be measured.

5. Support PIRSA Aquaculture by providing scientific and technical input into discussion with other stakeholders during the process of negotiating and implementing the desired changes to the existing system.

Objective 1: A report on the current status of environmental information in relation to aquaculture in South Australia was prepared in 2004, covering information available up to and including 2003. This report covered the 5 main aquaculture sectors in South Australia: Tuna, Marine finfish, Marine shellfish (intertidal), Marine shellfish (subtidal) and Land-based abalone. A synopsis of this report is provided in Chapter 1.

Objective 2: A series of risk assessment workshops were conducted in late 2003 and early 2004 to canvas the views of stakeholders on the environmental risks associated with three aquaculture sectors: Marine finfish (excluding tuna), Intertidal oysters and Land-based abalone. These workshops utilised the aquaculture supplement to the National ESD framework for Australian fisheries. This approach had a number of shortcomings, identified in Chapter 1, that caused considerable delays, however, risk assessment reports for all three sectors were eventually completed and have been produced as separate reports.

Objective 3: Following on from the risk assessment reports, field investigations were conducted for all three sectors covering a range of issues that were identified as being potentially important in the risk assessment process and which were consistent with the initial scope of the project. After commencement of work on intertidal oysters, studies of this sector were discontinued due to reprioritisation by PIRSA Aquaculture.

Objective 4: A range of options for environmental monitoring programs for both the marine finfish and land-based abalone sectors are discussed in the final two chapters of this report. These chapters identify several potential critical decision points that can be used to manage the respective sectors. In addition, any additional work required to pursue some of these options is identified.

Objective 5: PIRSA Aquaculture have not yet decided what changes to make to the system, and thus this objective has not yet been addressed.

Chapter 1: Review of current environmental status of marine aquaculture in South Australia, and outcomes of risk assessments.

1.1 Introduction

To assess the environmental performance of marine aquaculture industries in South Australia, it was first necessary to document the extent and nature of existing information available on the performance of each sector, and then to conduct a formal risk assessment to identify those issues that were relevant for each sector. This information will then be used to determine the scope of field investigations required to meet Objective 3 of the project.

1.2 Review of existing information

The review of existing environmental information pertaining to the marine aquaculture industry in South Australia is presented as a separate document (Bryars 2004). This review covers the period up until June 2003, and does not include the results of any monitoring or research conducted, submitted or published after that date. The following is extracted from the executive summary of Bryars (2004).

"This report addresses Objective 1 of the Environmental Audit project by assessing the level and adequacy of existing information and information collection protocols in relation to both the environment in which aquaculture operates and the impacts of aquaculture on this environment. The specific aims of the review are to report on (1) the extent of existing information[1] (2) the value of existing information, (3) the usefulness and feasibility of collating existing information, and (4) recommendations for collection and collation of future information.

A large amount of environmental information related to the five marine aquaculture sectors of Marine Tuna, Marine Finfish, Marine Shellfish (Intertidal), Marine Shellfish (Subtidal), and 'Land Based Abalone' was found to exist. This information has been collected through numerous monitoring programs, surveys, and impact assessments, as well as research and development and various other activities. The existing information is located in a variety of sources/locations, including PIRSA Aquaculture licence files, published and unpublished reports, and government datasets.

Environmental information collected in relation to aquaculture can and has contributed greatly to the (relatively small) knowledge base on the marine

[1] The review was conducted during 2003 and includes information available at the time. It should be recognised that as the report is retrospective in nature and that advances in environmental management and changes in reporting requirements are ongoing, the issues raised in the report may no longer be significant.

environment in South Australia. Collectively, it provides a large and useful source of qualitative and quantitative environmental information on water quality parameters, oceanography, phytoplankton, bio-fouling organisms, marine fauna, sediment infauna, benthic substrates, and epibenthic flora and fauna. At present, however, this information is scattered across many different agencies and in many different formats, thus making its accessibility and interpretation difficult.

While there is a large amount of information about the environment in which aquaculture operates, relatively little definitive information exists on the actual impacts of aquaculture on the marine environment in South Australia. Some reliable data are available for the Marine Tuna and Marine Shellfish (Intertidal) sectors, however, there is a paucity of impact data for the Marine Finfish, Marine Shellfish (Subtidal), and Land Based Abalone sectors. Despite aquaculture licensees having a licence condition to submit both draft and annual environmental monitoring program reports about their operations, relatively few reports have been submitted historically[2] to PIRSA Aquaculture for sectors other than Marine Tuna.

While the sources of environmental information are widespread and the quality of the information is highly variable, some form of collation within and between sectors of the existing information would be useful and, in some cases, is necessary. For collation across information formats, it is recommended that it be done as a 'geo-meta-database' in which the source of environmental information can be searched and located using key words and/or geographical coordinates. Collation of any existing and future information should address issues of accessibility, storage, geographical referencing, compatibility, improved collection procedures, transparency, and frequency of updating, reporting, review and revision.

The assessment of environmental impacts associated with the Marine Tuna sector are adequately addressed by the Tuna Environmental Monitoring Program and ongoing investigations within the Aquafin CRC - FRDC SBT Aquaculture Subprogram. Similar structures are lacking, however, for all other marine sectors. This disparity partly reflects the greater value and size of the tuna industry compared to other sectors. Nonetheless for each of the 'non-tuna' marine sectors there is an urgent need to identify and investigate potential and actual environmental impacts (Objectives 2 and 3 of the present project). This information will enable the development of sector-based environmental monitoring programs that are based on specified known impacts with allowable levels (and trigger values) of these impacts for use in an Ecologically Sustainable Development (ESD) performance framework (Objective 4 of the present project). In this respect, it is worth noting that globally there has recently been a change in emphasis from site-based to regional ecosystem-based environmental impact assessment/monitoring of aquaculture activities and that

[2] At the time of writing, environmental monitoring reports before and up to the end of the 2002/2003 financial year had been received by PIRSA Aquaculture and were viewed by the author. At the time of printing this document, the number of environmental monitoring reports submitted to PIRSA Aquaculture for the 2003/2004 financial year had drastically increased with a 76% return across all aquaculture sectors as at 23 July 2004 (S. Madigan, PIRSA Aquaculture, personal communication).

this is an area in which there is little information for all aquaculture sectors in South Australia."

Since the publication of Bryars (2004), the *Aquaculture Regulations 2005* have been introduced. These regulations include provision for issuing expiation notices to license holders who fail to submit an environmental monitoring report, and thus it is anticipated that return rates will improve considerably. In addition, PIRSA Aquaculture have revised and streamlined the monitoring requirements and developed formal monitoring programs for a range of sectors.

1.3 Risk assessments

Subsequent to the initial review of existing information, and in consultation with PIRSA Aquaculture, the government regulator of aquaculture in South Australia, it was decided to proceed with formal risk assessments of three industry sectors: Marine finfish (other than tuna), Intertidal shellfish (effectively oysters), and Land-based abalone. The full outcomes of these workshops and subsequent literature reviews are presented in separate reports (Marine finfish – De Jong and Tanner 2006, Intertidal shellfish – Wear et al. 2004, Land-based abalone – Theil et al. 2004), and are only summarised here.

The risk assessments for each sector were conducted using the National ESD reporting framework for Australian fisheries (Fletcher et al. 2002) and the aquaculture supplement (Fletcher et al. 2003). The reports of Fletcher et al. (2002, 2003) were developed to provide a framework that could be used consistently across all fishery and aquaculture sectors in Australia. The framework is based on the Australian standards for risk management (AS/NZS 4360 1999), which is used to conduct risk assessments for a wide variety of industries. This particular framework focuses on ESD outcomes by developing operational objectives and indicators to monitor and evaluate performance of management (Cheeson et al. 2000).

In the development of the framework, all the possible environmental, social and economic issues relating to all forms of aquaculture are identified and then grouped together in the form of eight generic component trees (see Figures. 1.1-1.4):

1. The environmental effects of the whole industry.
2. Environmental effects of the industry on the catchment/region.
3. Environmental effects of the individual facilities.
4. Impacts on the indigenous community wellbeing.
5. Impacts on community wellbeing.
6. Impacts on the national socio-economic wellbeing.
7. Governance.
8. Impact of the environment on the industry.

Each issue within a tree is assigned a risk ranking using a risk analysis tool outlined in the ESD framework, which is based on the Australian standard for risk management (AS/NZS 4360 1999). To assign a level of risk to an issue, two factors must be determined – the potential consequence arising from a particular activity, and the likelihood that this consequence will occur. The combination of consequence and likelihood produces an estimate of the risk associated with a particular issue. The main aim of the risk assessment is to determine if current management is sufficient, and therefore the current management strategies need to be considered when determining the consequence and likelihood levels. Each issue is assigned a level of consequence (from negligible to catastrophic) and likelihood (from remote to likely). In assigning a likelihood level it is important to remember that an assessment is being made of the likelihood of that consequence occurring and *not* the likelihood of that particular activity occurring. The consequence and likelihood levels are determined using the tables outlined in the framework (Tables 1.1 and 1.2). The risk value and ranking for each issue are then determined using a risk matrix (Table 1.3). Each risk ranking has an associated level of management response and reporting requirements (Table 1.4).

Table 1.1. The Consequence Table for use in ecological risk assessments related to aquaculture (from Fletcher et al. 2003). While this is the table used in the workshop, participants were asked to assess the situation over the next 5 years.

Level	Descriptor
Negligible (0)	Very insignificant impacts. Unlikely to be even measurable at the scale of the stock/ecosystem/community against natural background variability.
Minor (1)	Possibly detectable but minimal impact on structure/function or dynamics.
Moderate (2)	Maximum appropriate/acceptable level of impact (e.g. full assimilation rate for nutrients).
Severe (3)	This level will result in wider and longer-term impacts now occurring (e.g. increased plankton blooms).
Major (4)	Very serious impacts now occurring with relatively long time frame likely to be needed to restore to an acceptable level.
Catastrophic (5)	Widespread and permanent/irreversible damage or loss will occur – unlikely to even be fixed (e.g. extinctions).

Table 1.2. Likelihood Definitions (from Fletcher et al. 2002).

Level	Descriptor
Remote (1)	Never heard of, but not impossible
Rare (2)	May occur in exceptional circumstances
Unlikely (3)	Uncommon, but has been known to occur elsewhere
Possible (4)	Some evidence to suggest this is possible here
Occasional (5)	May occur
Likely (6)	It is expected to occur

Table 1.3. Risk Matrix – numbers in cells indicate risk value, the colours/shades indicate risk rankings (from Fletcher et al. 2002). NB the risk level is calculated by multiplying the likelihood value by the consequence value.

		Consequence					
Likelihood		Negligible	**Minor**	**Moderate**	**Severe**	**Major**	**Catastrophic**
		0	1	2	3	4	5
Remote	1	0	1	2	3	4	5
Rare	2	0	2	4	6	8	10
Unlikely	3	0	3	6	9	12	15
Possible	4	0	4	8	12	16	20
Occasional	5	0	5	10	15	20	25
Likely	6	0	6	12	18	24	30

Table 1.4. Suggested risk rankings and outcomes (amended from Fletcher et al. 2002).

Risk Rankings	Risk Values	Explanation and Likely Management Response	Likely Reporting Requirements
Negligible	0	Nil	Short justification only
Low	1 – 6	No specific additional management is needed, but low level monitoring of the issue may be required. Any current management should continue, as the risk ranking is based on the current management in place.	Full justification needed
Moderate	7 – 12	Additional information may be needed or the issue may require monitoring. No immediate management is required, but the issue should be the subject of continuous improvement with the aim of achieving a low risk ranking in the future.	Full performance report
High	13 – 18	Possible increases to management activities in addition to those already being applied. Needs to be monitored and any information deficiencies should be addressed.	Full performance report
Extreme	> 19	Increases in management activities in addition to those already being applied are strongly recommended.	Full performance report

In order to successfully undertake an environmental risk assessment, all of the perceived environmental issues need to be identified. Identification of all issues can only be achieved when opinions and thoughts are obtained from a number of stakeholders/stakeholder groups. Workshops have been widely recognised as one of the most efficient ways to gather all of the information required for a formal risk assessment.

Only the four component trees that related to environmental issues (i.e., trees 1-3 and part of tree 8) were addressed in the workshops (Figures 1.1-1.4). During the workshops, each of the four generic component trees was modified to produce trees specific to the sector under consideration. This process involved either deleting or adding issues. Each issue was then discussed in terms of current knowledge and management and assigned a risk ranking based on the perceived risk associated with that particular issue. Participants were asked to score the consequence and likelihood on the basis of what they expected over the next five years, not just on the current situation.

The focus of the workshop was to evaluate all perceived environmental risks of the aquaculture sector being considered, rather than just known risks, because there is very little documented information available on this aspect for South Australia.

While the National ESD reporting framework for Australian fisheries was used to perform the risk assessment, it was not the aim of the study to produce an ESD performance report for any sector. Rather the workshops were aimed at determining the perceived environmental risks associated with each aquaculture sector as a precursor to conducting field assessments of the environmental impacts and developing environmental monitoring programs.

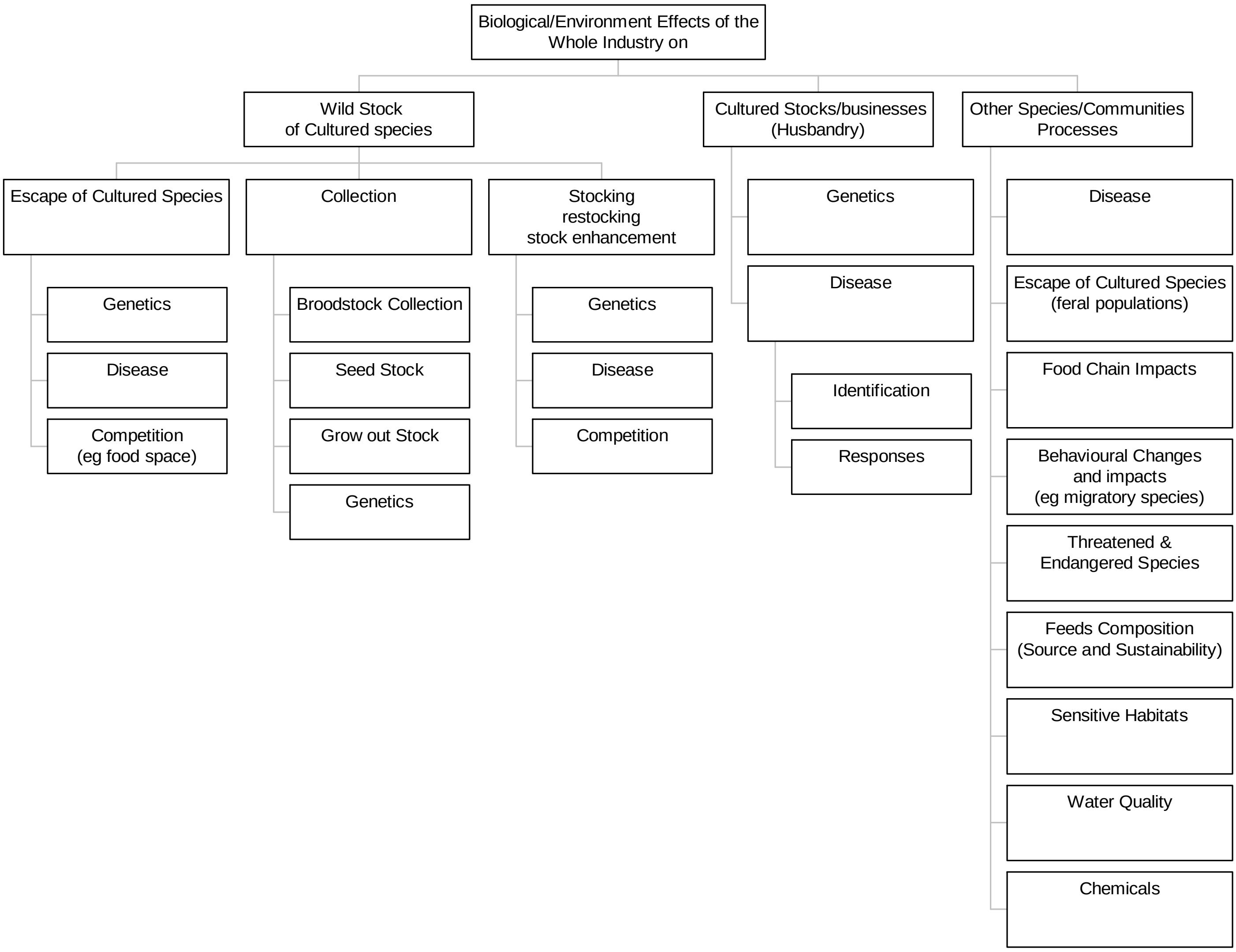

Figure 1.1. Component Tree 1: Biological/Environmental effects of the whole intertidal shellfish aquaculture industry (modified from Fletcher et al. 2003).

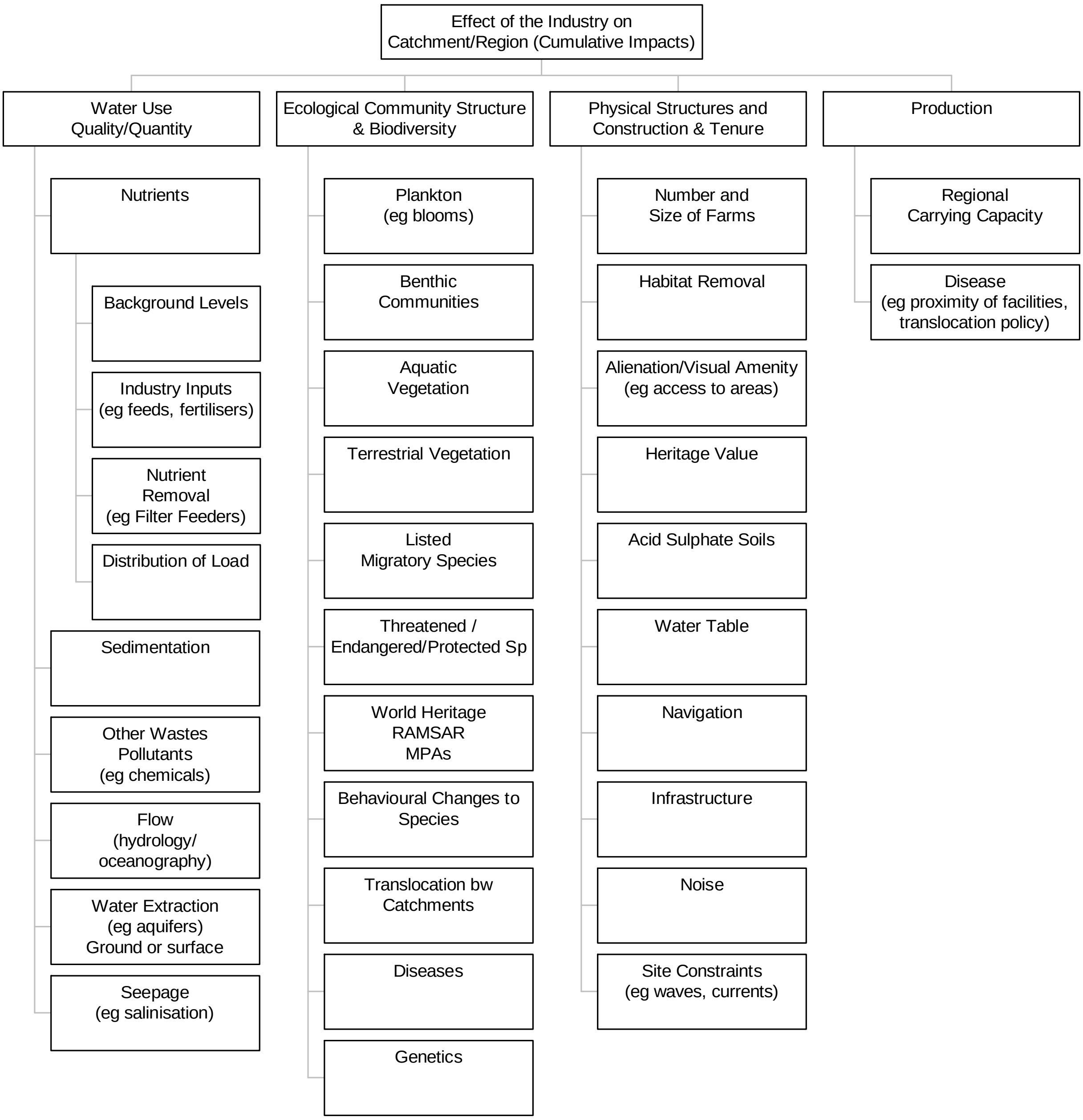

Figure 1.2. Component Tree 2: Impact of intertidal shellfish aquaculture on the catchment/region (modified from Fletcher et al. 2003).

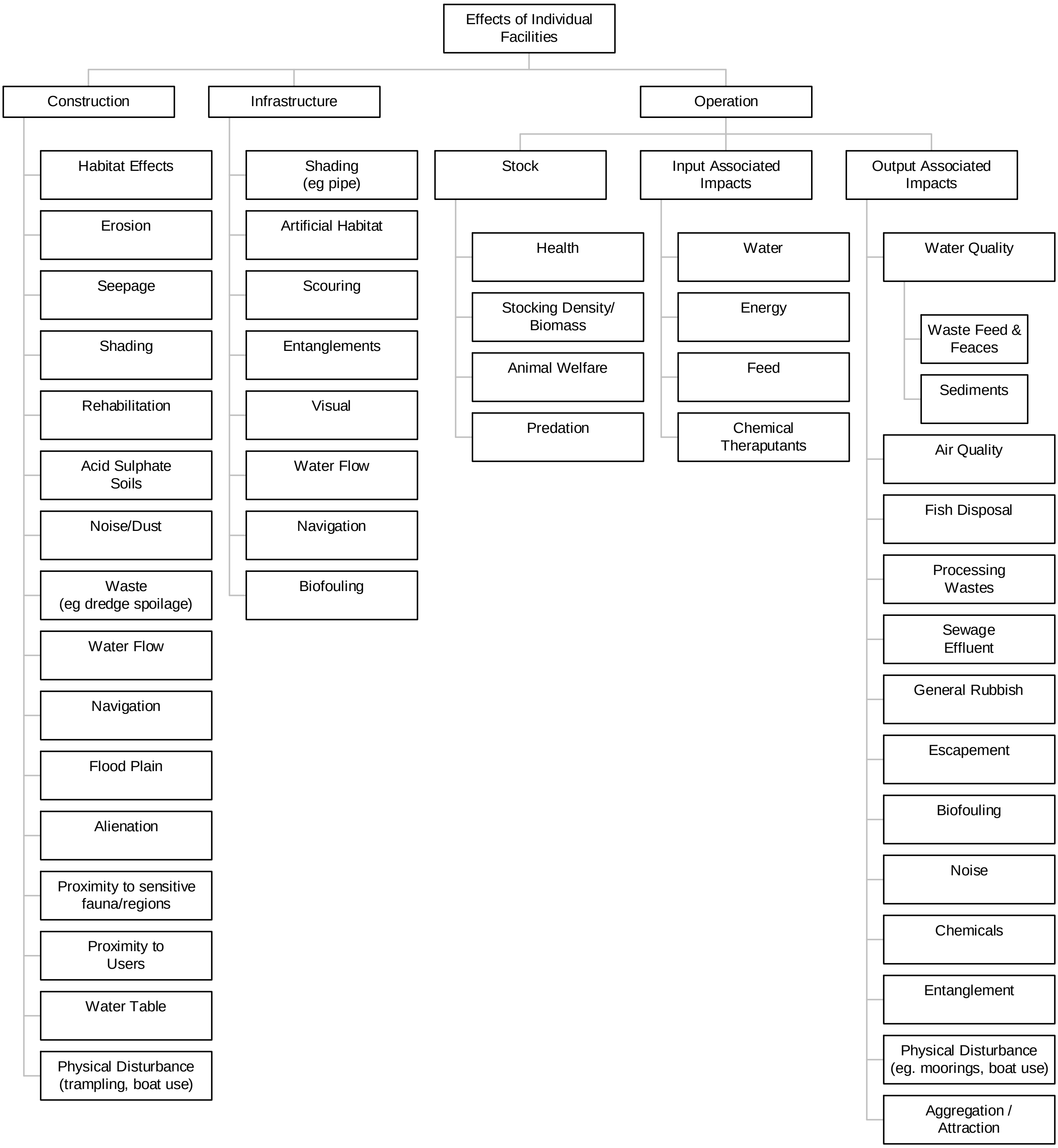

Figure 1.3. Component Tree 3: Impacts of individual intertidal shellfish aquaculture facilities on the environment (modified from Fletcher et al. 2003).

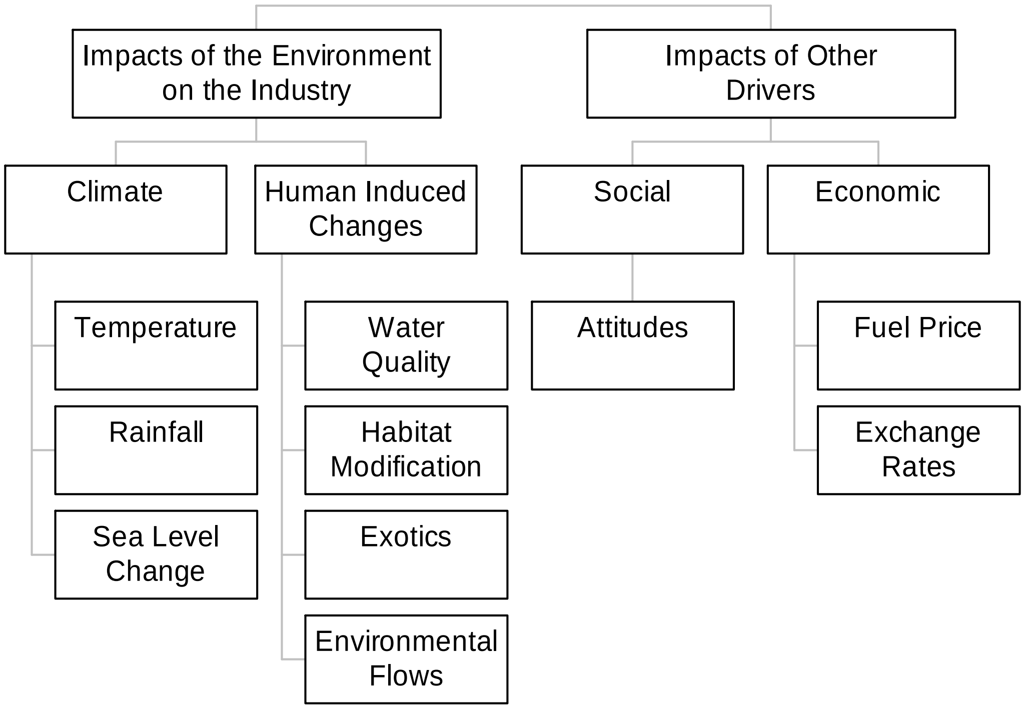

Figure 1.4. Component Tree 8: External impacts on the intertidal shellfish aquaculture industry (modified from Fletcher et al. 2003).

Following the workshops a number of steps were undertaken to complete the risk assessment and finalise this report:

1. A summary of the minutes and outcomes of the workshop were sent out during early 2004 to attendees (and invitees who were unable to attend) for any further comments.
2. Due to a lack of industry attendees at the oyster and land-based abalone workshops, additional meetings were held at Port Lincoln on the 23rd April 2004 to obtain industry input for those issues ranked during the workshop as moderate or higher.
3. A brief literature review was conducted on all issues ranked as moderate or higher. Of particular interest for the review was any information available from South Australia, although for a number of issues, none was found. Where there was little or no information on an issue in South Australia a broader literature search was conducted to find any relevant information either in Australia or worldwide. While the suggested outcomes for the determined risk rankings in Table 1.4 indicate that a full performance report is required for any issue determined to be of a moderate risk or higher, such a report was beyond the scope of the present report. Furthermore, it is the responsibility of the relevant management authorities to write full performance reports.
4. The risk rankings were re-assessed by the authors based on both the comments made during the two workshops and also the results of the literature review.
5. A draft final report was sent out to workshop attendees and invitees for final comment.

The marine finfish risk assessment workshop was held on September 23-24, 2003, with Dr Rick Fletcher acting as facilitator. This was the first attempt at such a risk assessment using the newly developed aquaculture supplement to the National ESD reporting framework for Australian fisheries (Fletcher et al. 2002, 2003). As such, a number of shortcomings were identified. The two major issues with the process were confusion between actual and perceived risk, and a lack of knowledge about many issues on behalf of the workshop attendees that meant that many of the risks identified were perceived rather than actual. As a consequence of this, industry expressed dissatisfaction with the process, and subsequently industry representatives refused to attend the oyster and land-based abalone workshops. In response to this, there was a change of emphasis in how the workshops and their results were presented, to stress that they dealt with perceptions, and follow-up workshops were conducted with industry representatives in Port Lincoln as mentioned above. To further address this issue, subsequent to the workshops a literature review was conducted on all the major issues identified, and risks were reassessed after this. All workshop invitees were then invited to comment on the revised risk rankings. Ideally, this literature review would have been conducted prior to the workshop, and the results presented as each issue discussed, so that workshop participants had the latest information available from which to draw conclusions about risks, however, this would have been impractical, requiring a great deal of extra work both in preparing for the workshop, and in the workshop itself.

Extracts from the executive summary and conclusions of each risk assessment report are provided below. These extracts were current at the time of publication of the original reports, but have not been updated to reflect changes that have occurred since.

1.3.1 Marine finfish risk assessment

The following is extracted from de Jong & Tanner (2004).

The value of marine finfish aquaculture production in South Australia has risen from $87 million in 1997/98 to $261 million in 2001/02. At present most finfish aquaculture occurs in Spencer Gulf at Boston Bay and Tumby Bay near Port Lincoln, as well as Arno Bay, Franklin Harbour and Fitzgerald Bay. There are also a few farms around Cape Jaffa and Rivoli Bay on the southeast coast. In 2003/2004 the main species being cultured was southern bluefin tuna off the Port Lincoln coast. Yellowtail kingfish in Spencer Gulf and salmonids (Atlantic salmon and rainbow trout) on the southeast coast were also farmed to a lesser extent. This report does not cover southern bluefin tuna, which are the subject of considerable research as part of the Aquafin CRC, but does cover the other finfish species being farmed at present. The industry looks set for substantial expansion in the near future, and its ecologically sustainable development (ESD) is a vital factor for ensuring long-term viability.

Twenty-one out of the 50 issues discussed were given a 'moderate' risk ranking in the workshop. A number of these issues are not discussed in this report as they are not considered environmental impacts. The others were the subject of a literature review, after which the risk was reassessed for some (Table 1.5).

For many of the issues raised, it is clear that there is insufficient information available to make a definitive estimate of the risk. There are also several instances, such as impacts on terrestrial vegetation, where it is clear that existing management regimes and requirements are not properly enforced, leading to potential problems. These particular issues are generally issues that relate to a wide variety of industries, not just aquaculture, and the shortfall is in the development planning process.

A number of knowledge gaps regarding the impacts of marine finfish aquaculture in South Australia were identified throughout the report. A number of these are currently being, or are soon to be, investigated and these are listed below. A greater number of issues require further research and are also listed. A few required monitoring and management programs have been identified as well.

Table 1.5. List of environmental issues from Component Trees 1 (Whole of Industry) and 2 (Catchment/Region) that were given a moderate risk ranking during the marine finfish workshop. For issues from Component Tree 2 – Catchment/ Region, the values given here are for Fitzgerald Bay, which was the main focus during the workshop.

Issue	Component tree	Consequence	Likelihood	Risk Ranking	Authors ranking
Effects of **competition** for food with **escaped cultured species** on the wild stock of that species	Whole of Industry	3	4	12	Low – Mod
Impacts of marine finfish aquaculture on the **food chain (i**ncluding **predation and competition)**	Whole of Industry	3	4	12	Low – Mod
Effects of **disease** (including parasites) from the **escape of cultured species** on the wild stock of that species	Whole of Industry	3	3	9	Mod
Effects of **disease** in the cultured stocks on other species and community processes	Whole of Industry	3	4	12	Mod
The effects of **translocations** of stock	Catchment / region	4	1-2	4-8	Low
Impacts of **feed composition,** including source and sustainability	Whole of Industry	4	3	12	Low
Effects of **industry inputs (e.g. feeds)** on the catchment/ region	Catchment / region	2	6	12	Low – Mod
Effects of marine finfish aquaculture on **phytoplankton**	Catchment / region	3	4	12	Low – Mod
Regional carrying capacity	Catchment / region	2	4	8	Mod
Behavioural changes and other impacts on migratory species (e.g. birds and whales) due to marine finfish aquaculture	Whole of Industry	3	3-4	9-12	Mod
Effects of marine finfish aquaculture on **dolphins** (threatened, endangered, and protected Species)	Catchment / region	3	4	12	Low
Effects of marine finfish aquaculture on **sharks** (threatened, endangered, and protected Species)	Catchment / region	2	4	8	Mod
Effects of marine finfish aquaculture on **terrestrial vegetation**	Catchment / region	3	4	12	Low
The effects of infrastructure on **Heritage areas**.	Catchment / region	3	4	12	Low
Effects of aquaculture on **World Heritage areas, RAMSAR and MPA's** – Cuttlefish Closure zone near Fitzgerald Bay	Catchment / region	3	4	12	Low – Mod

Current research

- FRDC proposal by SARDI to investigate the impacts of escaped yellowtail kingfish on other species and the food chain

- Identification and quantification of parasites of cultured, wild and escaped yellowtail kingfish and transmission between the stocks by Adelaide University
- Continuing research into fish meal replacement
- Movement and assimilation of pellet feed in the environment to be undertaken by SARDI Aquatic Sciences in 2004/05
- Validation of carrying capacity models by SARDI Aquatic Sciences
- South Australian Museum and Maquarie University will begin monitoring of the impacts of dolphin entanglement and habitat loss
- Development of techniques for monitoring the impacts of marine finfish aquaculture on catchments or regions by SARDI Aquatic Sciences
- Development of models specific to South Australia to simulate and estimate dispersion and deposition of wastes in the environment by SARDI Aquatic Sciences
- CSIRO are investigating the behaviour, ecology and population dynamics of great white sharks

Research required

- Behaviour, reproductive success and longevity of escaped yellowtail kingfish. Do they form self sustaining populations or integrate with wild populations (which may increase chances of disease transmission)?
- Impact of yellowtail kingfish escapes on competition for food with wild stock
- Impacts on the food chain from increases in silver gull populations due to marine finfish aquaculture (is being studied for tuna by Flinders University)
- Impacts on the ecology, population dynamics and reproductive success of wild fish that scavenge on uneaten food underneath the cages and their impact on the food chain (there is a current honours project at SARDI Aquatic Sciences and Adelaide University looking at aggregations around yellowtail kingfish cages at Fitzgerald Bay)
- Disease transmission (other than parasites) between wild and cultured fish.
- Can disease transfer occur through the cages? i.e. can only escaped fish transmit disease to the wild stock?
- Impacts of the baitfish fisheries on the food chain
- Investigate the impacts of wastes from marine finfish aquaculture on the environment using improved experimental design and develop appropriate monitoring techniques (is being done by SARDI Aquatic Sciences for tuna)
- Does marine finfish aquaculture affect the composition and abundance of phytoplankton communities and does the industry directly cause harmful algal blooms?
- More information on the behaviour, ecology and population dynamics of dolphins is needed in order to determine the impacts of marine finfish aquaculture
- Impacts of marine finfish aquaculture on the cuttlefish closure zone near Fitzgerald Bay

Monitoring and Management required

- Monitoring of the frequency, intensity and composition of algal blooms
- Monitoring of the interactions with migratory species (birds and whales).
- Monitoring and reporting of interactions with sharks (especially great white sharks)
- Improved management of impacts on terrestrial vegetation and heritage areas

Very little is known about the impacts of marine finfish aquaculture in South Australia. A few issues, such as the impacts of waste on water quality, benthic organisms and sediments, have been well studied in other parts of the world. Although we can draw on the experiences of other countries it is important to also investigate actual impacts in South Australia because the impacts of aquaculture can vary greatly between countries and regions due to physical, chemical and biological differences. This requirement is accentuated as South Australia leads Australia in temperate finfish aquaculture production, so the potential for adverse environmental impacts is greatest if we fall behind in research.

Overall, the results of this risk assessment workshop are relatively favourable to the industry. No issues were identified as being high or extreme risks, with the highest ranking being moderate. This ranking means that the issue either needs further research to determine the true risk, and/or that management needs to be improved. Such improvements should occur over a timeframe of 5-10 years, so as to bring the risk down to low.

1.3.2 Land-based abalone risk assessment

The following is extracted from Theil et al. (2004).

The value of abalone aquaculture production in South Australia has risen from $856,000 in 1998/99 to just under $2 million ($1,901,000) in 2001/02. Currently, there are nine abalone farms in South Australia; six in Louth Bay, two in Smith Bay on Kangaroo Island and one in Streaky Bay. Of the six abalone species known to inhabit South Australian waters, two are farmed commercially; these are the blacklip abalone, *Haliotis rubra* and the greenlip abalone *H. laevigata*. Many farms are still experimenting with methods and equipment in order to find the conditions to maximise production and minimise costs. With the industry looking for maximum production in the near future, ecologically sustainable development (ESD) is a vital factor for ensuring long-term viability. To further promote, expand, and ensure ESD of the abalone aquaculture industry there is an urgent need to assess the perceived risks of associated environmental impacts.

Sixty two of the 78 issues discussed (excluding tree 8) were given a negligible or low ranking. The remaining 16 issues (Table 1.6) were given a moderate or moderate to high risk ranking. No issues were rated as an extreme risk. The moderate and moderate to high risk issues fall into four broad groups:

1. Effect of escapes or disease (including parasites)

2. Habitat alteration/Loss
3. The effect of water quality discharged from land-based abalone aquaculture on the marine environment.
4. Impact of land-based abalone aquaculture on the attraction/aggregation of species.

After reviewing the literature, most of the issues identified were given a low risk ranking. The only issues given a moderate rating were impacts on sensitive habitats erosion and water quality-nutrients. All 3 issues are of concern at the individual facility level, rather than at regional or whole of industry levels. The impact on sensitive habitats is based on previous problems related to a farm that is no longer operational, and on the potential for future developments to cause problems if sensitive habitats are not given adequate consideration in the planning process. The impact of erosion relates to beach erosion around pipes, which has reportedly occurred at several facilities. The later is based on evidence of eutrophication found by the EPA in 2001. A number of construction related issues were also ranked as moderate during the workshop, but are not discussed further here as they are not specific to aquaculture, and apply to all coastal developments.

Currently, there are no research programs underway addressing the potential environmental impacts of abalone aquaculture, although PIRSA Aquaculture are in the process of developing protocols to address issues related to disease. The most important research needs are related to determining the effects of nutrient discharges on the environment, although the results of the 2003 EMP's should be assessed before the exact needs are determined. Other lower priority areas of research include:

- Effects of abalone farms on aggregation of birds
- Potential for disease transmission from cultured stock to wild stock
- The effects of settlement ponds on water quality
- Seasonal variations in quality and quantity of nutrient and solid discharges

Tighter management and/or reporting protocols may also be useful to reduce or better assess the consequences of several issues:

- Development of best practise construction protocols to minimise terrestrial impacts, including restoration of surrounding area after construction
- Protocols to maximise chances of site rehabilitation after production ceases
- Reporting of chemical use (type, amount, duration of use, discharge rates during use) as part of EMP process.

Table 1.6. List of environmental issues from Component Trees 1 (Whole of Industry), 2 (Catchment/Region) and, 3 (Individual Facilities) that were given a moderate or moderate to high risk ranking during the land-based abalone workshop.

Issue	Component tree	Consequence	Likelihood	Risk Ranking	Authors ranking
Effects of **disease** (including parasites) from the **escape of cultured species** on the wild stock of that species	Whole of Industry	3	4	12	Low
Effects of **disease** in the cultured stocks on other species	Whole of Industry	3	3	9	Low
The effects of **translocations, escapements** etc. of stock	Catchment / region	3	3	9	Low
Effects of **disease** transmission (proximity of facilities, translocation policy)	Catchment / region	3	4	12	Low
Effects of land-based abalone aquaculture on **terrestrial vegetation**	Individual facilities	3/4	6	12/18	
Impact of construction of land-based abalone aquaculture on **erosion**	Individual facilities	2/3	4	8/12	Mod
Rehabilitation of site	Individual facilities	3	4/5	12/15	
Impact of construction **noise** and **dust**	Individual facilities	2	4	8	
Impact of wastes **produced during construction**	Individual facilities	2	4	8	
Behavioural changes and other impacts on **migratory species** (e.g. birds) due to land-based abalone aquaculture	Individual facilities	2	4	8	Low
The impact of land-based abalone aquaculture on **sensitive habitat**	Individual facilities	2	3/4	6/8	Mod
The effect of **water quality (**e.g. **nutrients)** on the marine environment	Individual facilities	2	5	10	Mod
The effect of **water quality** (e.g. **feed and faeces**) on the marine environment	Individual facilities	2	5	10	Low
The effect of **water quality** (e.g. **chemicals**) on the marine environment	Individual facilities	2	4	8	Low
Effects of **disease** (including parasites) from the **escape of cultured species** on the wild stock of that species	Individual facilities	3	4	12	Low
Impact of land-based abalone aquaculture on the **attraction/aggregation** of species	Individual facilities	2/3 1	6 6	12/18 6	Low

1.3.3 Intertidal shellfish (oyster) risk assessment

The following is extracted from Wear et al. (2004).

The value of oyster aquaculture production in South Australia has risen from $5,489,000 in 1998/99 to just over $16 million ($16,118,000) in 2002/03. Currently, there are ten oyster growing areas along the South Australian coastline, with farms situated on the West Coast, Spencer Gulf, Gulf St Vincent and Kangaroo Island. While several species of shellfish are licensed for cultivation in the Marine Shellfish (Intertidal) sector, the Pacific oyster (*Crassostrea gigas*) is the main species farmed.

With the industry looking for maximum production in the near future, ecologically sustainable development (ESD) is a vital factor for ensuring long-term viability.

Forty one of the 51 issues discussed at the workshop were given a negligible or low ranking. The remaining 10 issues were given a moderate ranking (Table 1.7). No issues were rated a high or extreme risk. The moderate issues fall into the following groups:

1. Effects of diseases and parasites in the cultured stocks on other species
2. Risks associated with the introduction of invasive/exotic species
3. Effects of nutrient removal on other filter feeders
4. Impact of intertidal shellfish aquaculture on terrestrial vegetation
5. Risks associated with the formation of feral populations
6. Impact of physical disturbance of the environment
7. Effects of shading on the environment

Table 1.7. List of environmental issues from Component Trees 1 (Whole of Industry), 2 (Catchment/Region) and, 3 (Individual Facilities) that were given a moderate or moderate to high risk ranking during the intertidal shellfish (oyster) workshop.

Issue	Component tree	Consequence	Likelihood	Risk Ranking	Authors ranking
Effects of **disease** in the cultured stocks on other **species**	Whole of Industry	3	4	12	Low - Moderate
Risks associated with the introduction of **invasive species**	Catchment / region	4	2	8	Moderate
Effects of **nutrient removal** on other filter feeders	Catchment / region	2	4	8	Moderate
Impact of intertidal shellfish aquaculture on **terrestrial vegetation**	Catchment / region	3	4	12	Low
The effects **diseases** as a result of **translocations, escapements** etc. of stock	Catchment / region	3	4	12	Moderate
Risks associated with introduction of **exotic species**	Catchment / region	5	2	10	Moderate
Risks associated with the formation of **feral populations**	Catchment / region	4	2	8	Low-Moderate
Impact of **physical disturbance** of the environment	Individual facilities	2	4	4/8	Low-Moderate
Effects of **shading** on the environment	Individual facilities	2	6	12	Low-Moderate
Impact of **BST long-lines** and **shading** on the environment	Individual facilities	2	4	8	Low-Moderate

While there are numerous national and international research programs addressing the potential environmental impacts of oyster aquaculture, research in South Australia is limited. Results from the workshop indicate that the most important research needs are to determine:

- the food requirements of oysters and the potential for competition with native filter feeders

- the effects of physical disturbance and shading

Other lower priority areas of research include determining:
- the effects of diseases and parasites in the cultured stocks on other species,
- the potential for the introduction of invasive/exotic species
- the potential for establishment of feral populations and their associated impacts.

Tighter management and/or reporting protocols may also be useful to reduce or better assess:
- the consequences associated with damage to dune vegetation
- coastal erosion stemming from inappropriate access to lease areas.

It should be noted that these issues are not restricted to aquaculture, and to be fully effective any controls should be placed on all relevant users of the coastal environment.

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Chapter 2: Investigations into the environmental impacts of yellowtail kingfish aquaculture in South Australia.

Jason Tanner

2.1 Introduction

The farm gate value of aquaculture production in South Australia has risen from \$87 million in 1997/98 to \$273 million in 2002/03, although it then declined to \$187.8 million in 2004/05, the latest year for which complete figures are available (Knight et al. 2003, 2005; Econsearch, 2006). Of this, \$140 million in 2004/05 was due to tuna, with the remaining \$47.8 million being predominantly oysters (\$21.2 million), abalone (\$5.3 million) and 'other' species (\$17.6 million). This last category includes various finfish species, most notably yellowtail kingfish (YTK - *Seriola lalandi*) with ~ 2,000 tonnes produced (Chambers and Ernst 2005). At present most sea-cage finfish aquaculture occurs in Spencer Gulf around the Port Lincoln region in Tumby Bay and near Boston Island, as well as Arno Bay, Franklin Harbour and Fitzgerald Bay. There are also a few farms around Cape Jaffa and Rivoli Bay on the southeast coast. As at June 2006, there are 25 finfish (excluding tuna) aquaculture licences in South Australia (PIRSA 2006), although not all of them are in operation at the one time, and there are a number of new applications under consideration. The finfish species that are currently licensed for marine sea-cage aquaculture in South Australia are yellowtail kingfish, Atlantic salmon *Salmo salar*, rainbow (ocean) trout *Oncorhynchus mykiss*, snapper *Pagrus auratus*, southern bluefin tuna *Thunnus maccoyii* and mulloway *Argytosomus japonicus*. In 2005/2006 the main species being cultured was southern bluefin tuna off the Port Lincoln coast. Yellowtail kingfish in Spencer Gulf are currently the dominant species being cultured after tuna, and there is some intermittent farming of salmonids (Atlantic salmon and rainbow trout) on the southeast coast, with the remaining species only being cultured on a minor basis. As tuna are the subject of extensive research as part of the Aquafin CRC, and because yellowtail kingfish are the dominant finfish species being farmed after tuna, this chapter deals solely with this latter species.

Marine finfish aquaculture in South Australia involves the "grow out" of fish in offshore sea cages (also known as pontoons). In the case of southern bluefin tuna the fish are sourced from the wild to a strict quota, while all the other cultured finfish species are sourced from hatcheries either in South Australia (yellowtail kingfish), Victoria or Tasmania (Atlantic salmon and ocean trout). The fish are kept in the sea cages for anywhere between 1-3 years depending on the species. The fish are fed either on baitfish such as Australian sardines (the main feed for tuna), or manufactured pellets (the main feed for all other species), until they grow to a marketable size and are then harvested.

Yellowtail kingfish are primarily farmed in Fitzgerald Bay, in northern Spencer Gulf, and Arno and Boston Bays, further south. Current production is slightly over 2,000 tonnes per annum (Chambers and Ernst, 2005). Typically, ~8,000 fish are stocked into nursery cages in about October. These are kept for ~2 years, during which time they undergo two gradings for size, at which time they are transferred between cages. Harvest size is ~3.5 kg. A typical cage is 25 m in diameter, and 6 m deep. Standard husbandry procedures include the changing of nets on the sea cages and freshwater bathing of fish to remove ectoparasites. The nets on the sea cages are changed every couple of months because they become clogged with fouling organisms that can reduce the amount of water flow and affect waste build up often clog them. The animals in the fouling communities can also be vectors for parasites that may infect the cultured fish (Tan et al. 2002), and detritus on the nets can harbour parasites. The fouled nets are taken to shore to be cleaned. The *Aquaculture Regulations 2005* contain provisions relating to the use of chemicals for therapeutic or prophylactic purposes or as an antifoulant in the course of aquaculture carried out under an aquaculture license. The licence holder must adhere to these regulations. The area in which the sea cages are kept must undergo a fallowing period as stipulated in the licence. Fallowing means that once the stocked sea cages have been removed from an area no more sea cages can be placed in that same area for 2 years to allow the site time to recover before being used again.

As part of the risk assessment discussed in Chapter 1 (see also de Jong and Tanner 2004), a two-day workshop was conducted to assess the potential environmental risks associated with marine sea-cage aquaculture of finfish in South Australia. While the workshop was intended to cover finfish broadly, it was decided that due to the dominance of yellowtail kingfish, and the restricted time available, to focus primarily on this species. Southern bluefin tuna were not considered in this workshop, as they were the focus of an earlier workshop, reported in Theil et al. (2004). A total of 50 issues were discussed in the workshop, covering both the whole of industry and catchment levels. Due to time constraints, and the range of sites at which finfish aquaculture is conducted, issues at the site level were not discussed. Twenty-one issues were identified in the workshop as having a moderate or higher risk associated with them, although six of these were not environmental issues per se, and so were not considered further. Examples of these include the impacts of aquaculture on navigation, and impacts of disease on the cultured stock. The other 15 issues were considered in a brief literature review, and then reassessed based on the findings of this review. For many of the issues raised, it was clear that there is insufficient information available to make a definitive estimate of the risk. There were also several instances, such as impacts on terrestrial vegetation, where it was clear that existing management regimes and requirements are not properly enforced, leading to potential problems. These particular issues were generally issues that related to a wide variety of industries, not just aquaculture, and the shortfall was identified to be in the development planning process.

Of the 15 issues assessed in the literature review and given revised rankings, four were scored as low-moderate, and a further five as moderate. The six remaining issues were scored as low risks. It should be noted that issues receiving a moderate risk ranking are not considered to require urgent attention, but rather, industry should work towards reducing these risks over a time frame on the order of five years.

Issues considered to be low-moderate risks were:

- Impacts on the food chain (including predation and competition)
- Effects of industry inputs (e.g. feeds) on the catchment/region
- Effects on phytoplankton
- Effects on the cuttlefish closure zone near Fitzgerald Bay

Issues considered to be a moderate risk were:

- Effects of disease from the escape of cultured species on wild stocks of that species
- Effects of disease in the cultured stock on other species and community processes
- Regional carrying capacity
- Behavioural changes and other impacts on migratory species
- Effects on sharks

This chapter considers the field investigations that have been conducted to determine if a number of the risks identified above are currently causing environmental problems. These issues were selected after consultation with PIRSA Aquaculture, assessment of other research projects underway, and consideration of the original project scope. Regional carrying capacity is not considered here, as it is the subject of a complimentary project (FRDC 2003/222: Innovative solutions for aquaculture: spatial impacts and carrying capacity – further developing, refining and validating existing models of environmental effects of finfish farming). In addition, effects of disease, impacts on migratory species, and effects on sharks were considered to be outside the scope of the project, and not pursued further. In consultation with PIRSA Aquaculture, it was also decided not to pursue the effects on the cuttlefish closure zone, but rather to focus on the remaining issues. Thus, this report focuses on three main issues. The first issue is the impacts on parts of the food chain, through an examination of wild fish assemblages around sea-cages. The second issue is the effects of industry inputs on the benthos, primarily infauna and seagrass. The third issue is the impacts on the water column, including both water column chemistry and phytoplankton. Each of these issues is reviewed in de Jong and Tanner (2004), which was produced as a part of the current project, and the reader is referred to this document for background information.

2.2 Methods

2.2.1 Water quality

Water quality was examined at lease and control sites in Fitzgerald Bay on November 11, 2004. Sampling was conducted in two leases (north and south), as well as two associated control areas 1 km away from the associated lease (and at least 1 km from any other lease). Within each lease, samples were taken from just downstream (< 10m) of three cages, with three sites in each control area having a similar separation (~100 m) also being sampled (Figure 2.1, Table 2.1). Samples were spread over the tidal cycle for logistic reasons. At each site, a Niskin bottle was used to collect two water samples for nutrient analysis from 2 m below the water surface. Each sample was filtered (0.45 μm) immediately after collection, with the sample container being rinsed in filtered water before 40 ml was retained and placed on ice. On return to shore, samples were frozen, and then sent to the Water Studies Centre at Monash University for analysis. Each sample was analysed for Total Phosphorus, Total Nitrogen, Nitrate + Nitrite and Ammonia using flow injection analysis on a QuickChem 8000 Automated Ion Analyser. At the time of sampling, a Horiba water quality meter (model W22XD) was used to measure water temperature, salinity and turbidity, again at 2 m below the water surface.

To determine if water quality varied significantly between lease and control sites, a permutational multivariate analysis of variance (PERMANOVA, Anderson 2001) was conducted. PERMANOVA is a multivariate ANOVA technique that does not require the assumption of multivariate normality, and that allows different measures of distance to be used, thus making it much more suitable for most ecological data than standard parametric MANOVA. Location (north/south) and site nested within location were random factors, while Aquaculture (cage/reference) was fixed. The analysis was based on Euclidean distances and untransformed data. 4999 permutations of the residuals under a reduced model were used to calculate probability values. As temperature and salinity varied little, and there is no a priori reason to expect them to vary between cage and control sites, these variables were not included in the analyses. To provide a visual representation of any difference, principal components analysis (PCA, McCune and Grace 2002) was used with PC-ORD (ver 4.0, MjM Software, Oregon). To examine the response of individual water quality measures, univariate ANOVA's were then conducted for each individual parameter using the statistical package SPSS (ver 13.0, SPSS Inc, Chicago). As only a single measurement for each of turbidity, salinity and temperature was made at each site, for these parameters sites are treated as replicates.

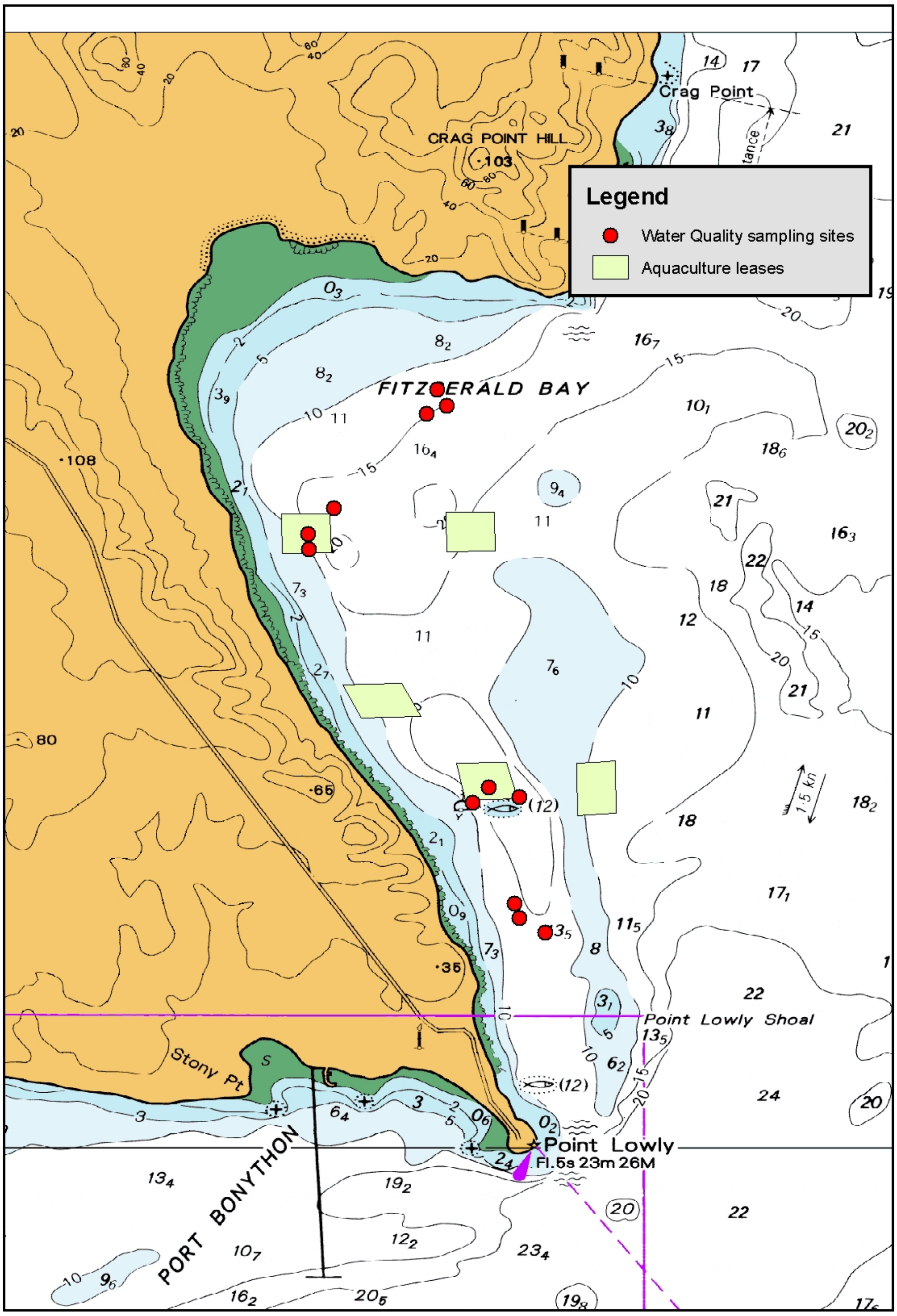

Figure 2.1. Location of water quality sampling sites in Fitzgerald Bay.
Note that while some water samples were taken from just outside lease areas, these were actually immediately adjacent to cages.

Table 2.1. Co-ordinates of water quality sampling sites (WGS 84)

Lease	Treatment	Site	Latitude (^{o}S)	Longitude (^{o}E)
North	Cage	1	32.9361	137.7659
North	Cage	2	32.9387	137.7633
North	Cage	3	32.9402	137.7634
North	Control	4	32.92426	137.7762
North	Control	5	32.92583	137.7772
North	Control	6	32.92664	137.7752
South	Cage	7	32.9639	137.7813
South	Cage	8	32.9655	137.7797
South	Cage	9	32.9649	137.7844
South	Control	10	32.97552	137.7839
South	Control	11	32.97699	137.7844
South	Control	12	32.97841	137.787

2.2.2 Chlorophyll and phytoplankton

Chlorophyll a concentration and phytoplankton composition were examined at lease and control sites in Fitzgerald Bay on November 11, 2004, and August 16-18 2005. In 2004, sampling was conducted at the same time as for water quality, using the same collection methods. In 2005, the same sites were sampled using the same methodology, with the exception that samples were taken from just above the bottom (1m) as well as just below the surface, and some sites were not sampled due to deteriorating weather. Sampling was conducted in two leases (north and south), as well as two associated control areas 1 km away from the associated lease (and at least 1 km from any other lease). Within each lease, samples were taken from just outside three cages (two in 2005), with three (two in 2005) sites in each control area having a similar separation also being sampled (Figure 2.1, Table 2.1). At each site, a Niskin bottle was used to collect three water samples of 1250 ml each for chlorophyll a and phytoplankton. Chlorophyll samples were immediately placed on ice, and on return to shore 1000ml was filtered under vacuum through a 0.7 micron GF/F filter paper. Filter papers were then stored in liquid nitrogen until analysis. Chlorophyll a analysis was based on the methods developed in Golterman et al. (1978). Filter papers were transferred to test tubes to which 5ml of methanol was added and then placed in the fridge for 24 hours to facilitate chlorophyll a extraction. Chlorophyll a analysis was carried out on a Thermo Helios Gamma Spectrophotometer. Phytoplankton samples were kept dark and preserved with 5 ml of Lugol's solution, prior to being sent to Microalgal Services (Ormond, Victoria) for determination of species composition.

The influence of proximity to aquaculture cages on chlorophyll a concentrations was determined using univariate ANOVA. As only surface samples were collected in 2004, and poor weather meant that only surface samples were collected from the northern farm and reference sites in 2005, a full analysis incorporating all factors could not be conducted. Instead the data were analysed in two subsets. The first subset only considered the surface samples, and the second only the 2005 samples. Depth and Aquaculture (lease/reference) were treated as fixed factors, while Month, Site and Location (north/south) were treated as random. Site was nested within the Location by Aquaculture interaction, with all other terms being orthogonal.

To determine if phytoplankton varied significantly between lease and control sites, a PERMANOVA was conducted. Due to the addition of bottom samples in August 2005, and the incomplete sampling at this time, several different analyses had to be undertaken, as PERMANOVA requires a fully balanced data set in its current implementation. First, the November 2004 data were analysed to determine the effects of location and aquaculture. Second, the August 2005 surface data were analysed for the same purpose. Third, the August 2005 data from the southern section of the bay were analysed to examine the interaction between aquaculture and depth. Finally, a comparison of the 2004 and 2005 data was made using surface samples only, and two randomly chosen sites for 2004. Location (north/south) and site nested within location were random factors, while aquaculture (cage/reference) was fixed. The analysis was based on Bray-Curtis dissimilarities and fourth-root transformed data, as is standard for community analyses, to eliminate the effects of joint absences and downweight the influence of highly abundant species respectively (McCune and Grace 2002). 4999 permutations of the residuals under a reduced model were used to calculate probability values. To provide a visual representation of any difference, non Metric Multidimensional Scaling (nMDS, McCune and Grace 2002) was conducted with PC-ORD. The number of axes used in each nMDS was selected using a Monte Carlo test. To determine which species were responsible for differences between significant factors of interest, a SIMPER analysis was conducted (Clarke and Warwick 2001) using PRIMER (ver 5.2.9, Primer-E Ltd, Plymouth).

To assess whether any patterns may be obscured by high levels of variation within species, but not higher taxonomic groups, the 2004 data were also analysed with species grouped as Diatoms, Dinoflagellates or Other. Both PERMANOVA and nMDS were conducted as per above.

In addition, the use of remote sensing was trialled to determine if it could be successfully used to monitor chlorophyll levels in Fitzgerald Bay. Results from the remote-sensed imagery were compared to those from the in-situ water sampling discussed above. Data were obtained from level 2 Aqua-MODIS local area coverage images with 1 km resolution. Images for the week encompassing the sample dates were examined using the level 2 browser. Clear images were only available for 8 and 15 November 2004, and 16 August 2005. These were selected and cropped to focus on Spencer Gulf. Selected images were downloaded by ftp as HDF files and loaded into SeaDAS software, and re-scaled to a range of 0.01-3 μg/l Chlorophyll a for display. For detailed imagery of Fitzgerald Bay, the 1 km resolution image was expanded using the zoom facility in SeaDAS. At this level, the image was quite pixellated, but still usable.

2.2.3 Infauna

To determine if YTK farming has any effects on infaunal assemblages around cages, sediment cores were sampled along a gradient away from cages in August/September 2004. Eight replicate samples were taken using a HAPS corer (Figure 2.2) at each of 0, 20, 50, 100 and 1000 m along transects radiating out from 2 cages on each of 2 separate leases (Figure 2.3, Table 2.2). Each core had a diameter of 67 mm, and was taken to a depth of 10 cm. Cores were extruded from the barrel and preserved in Bennett's solution prior to later processing in the laboratory. Each sample was sieved on a 1 mm mesh in the laboratory, and then sorted to extract all infauna remaining on the sieve. While eight samples were taken at all sites, in some cases a single sample was missing, resulting in only seven replicates at these sites. Taxa were identified and enumerated to the lowest taxonomic level possible, generally family. This sampling protocol follows that used for the Tuna Environmental Monitoring Program (TEMP) off Port Lincoln (see for example Loo and Drabsch 2005).

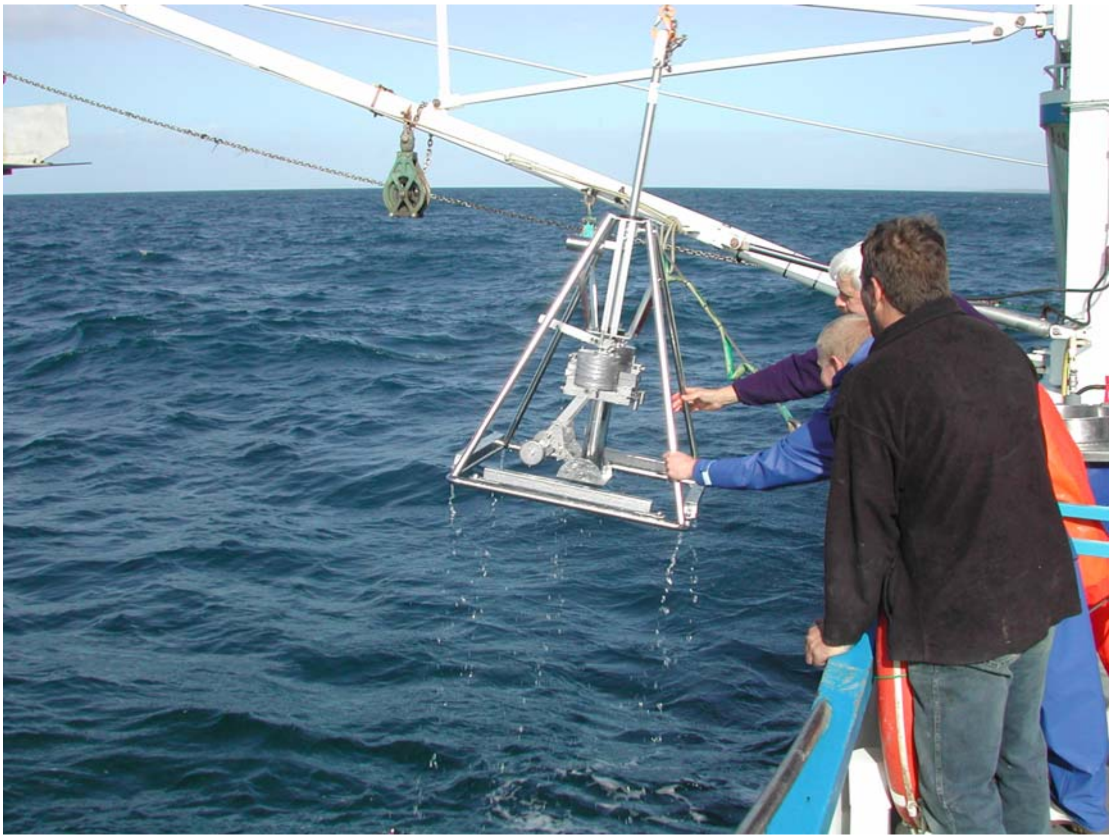

Figure 2.2. HAPS corer being retrieved after collection of sediment sample (photo S. Madigan).

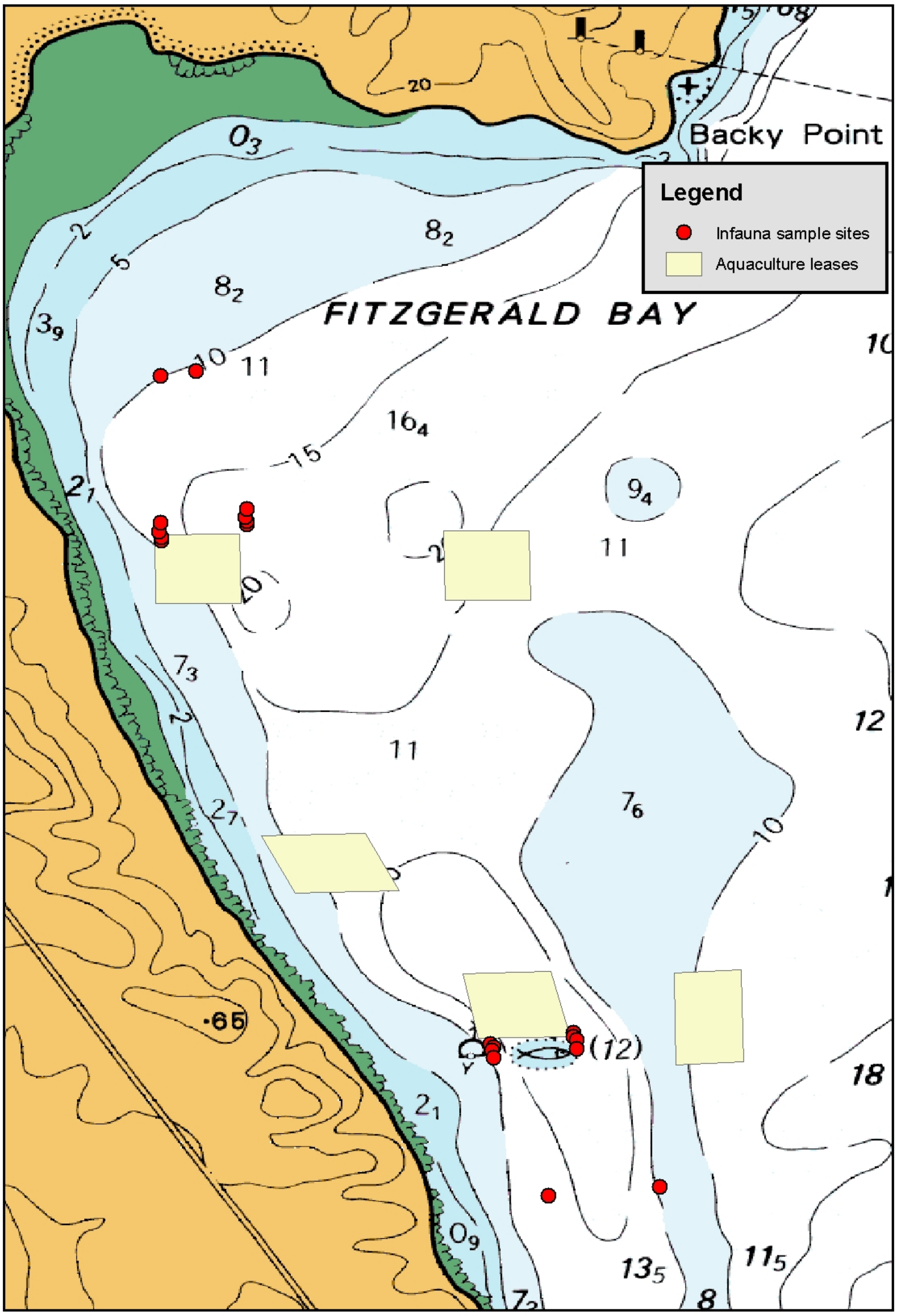

Figure 2.3. Map of Fitzgerald Bay showing lease sites and locations of infauna sampling sites. Note that while some samples were taken from just outside lease areas, these were actually immediately adjacent to cages.

Figure 2.4. Typical sediment core from Fitzgerald Bay (photo: Milena Fernandes).

Table 2.2. Co-ordinates of infaunal sampling sites (WGS 84)

Lease	Cage	Distance from cage (m)	Latitude (^oS)	Longitude (^oE)
North	NW	0	32.937	137.7611
North	NW	20	32.9368	137.7611
North	NW	50	32.9365	137.7609
North	NW	100	32.936	137.761
North	NW	1000	32.9277	137.761
North	NE	0	32.9361	137.7659
North	NE	20	32.9359	137.7659
North	NE	50	32.9357	137.7658
North	NE	100	32.9352	137.7659
North	NE	1000	32.9274	137.763
South	SE	0	32.9649	137.7844
South	SE	20	32.9651	137.7844
South	SE	50	32.9653	137.7846
South	SE	100	32.9658	137.7846
South	SE	1000	32.9736	137.7893
South	SW	0	32.9655	137.7797
South	SW	20	32.9657	137.7799
South	SW	50	32.9659	137.7798
South	SW	100	32.9663	137.7799
South	SW	1000	32.9741	137.783

To determine if distance significantly affected faunal composition, a PERMANOVA was conducted. Distance was treated as a covariate, while lease, and cage nested within lease, were random factors. The influence of highly abundant species was downweighted by using fourth root transformed data, as is standard for community analyses, and Bray-Curtis dissimilarities were used to remove any effects of joint absences. The current implementation of PERMANOVA requires balanced data, so for sites with eight replicates, one replicate was randomly deleted prior to analysis. 4999 permutations of the residuals under a reduced model were used to calculate probability values. Analyses were conducted with all 59 taxa, as well as the 40 taxa that were represented by more than a single individual. As the results for the two data sets were qualitatively identical, and quantitatively very similar, only the analysis based on all 59 taxa is presented. To provide a visual representation of any difference, non-metric multidimensional scaling (nMDS McCune and Grace 2002) was used, again with fourth root transformed data and Bray-Curtis dissimilarities.

The response of total infaunal abundance, and taxonomic richness, were analysed using standard univariate ANOVA. Both variables were natural log transformed prior to analysis, to meet the assumption of normality, and both met the assumption of homogeneity of variances after transformation (based on Levene's test). Similarly, the four most abundant taxa were analysed individually, with the Apseudidae, Capitellidae and Spionidae requiring natural log transformations and Cirratulidae a square root transformation to meet assumptions. To test the assumption of equality of slopes for ANCOVA, initial analyses included terms for the interaction between distance and lease and distance and cage. If these terms were non-significant ($p>0.05$), then the analysis proceeded. Otherwise, analyses were conducted for individual leases or cages as appropriate.

2.2.4 Epifauna

Epifaunal assemblages were assessed at both lease and control sites in two separate remote video surveys, to determine if increased deposition of organic matter was having an affect on them. In the first survey, a series of 100 m transects were conducted radiating out in the four cardinal directions from the edge of each of two lease sites and two control sites (Figure 2.5). In addition, a similar series of transects were videoed radiating out from a single fallowed lease site. For each transect, a digital video camera was lowered to approximately 0.2-0.5 m off the bottom and the substrate filmed while the boat motored slowly along the length of the transect. A GPS was used to record the location where the image first became clear, and the distance from this was monitored to ensure that each transect was 100 m long. The location of the camera relative to the bottom, and quality of the footage, were monitored on board the boat via a live feed to a surface monitor. These transects were conducted from 21-25 June 2004.

Video footage was analysed back in the laboratory to determine the identities of the species present along each transect. The abundance of each taxon was recorded for a central strip of the transect approximately 0.5 m wide. Sessile organisms were recorded as number of individuals, whereas clonal species such as seagrass and algae, and substrates such as sand and rubble, were recorded as percent cover. To examine how epifaunal assemblages change with distance from the edge of the lease, transects were divided into 25 m segments, with abundance and cover recorded separately for each segment. Abundance was based on total numbers of individuals in each segment, while cover was mean percent cover recorded in each of ten non-overlapping frames. Due to the poor taxonomic knowledge of the sessile fauna of southern Australia, and the nature of video footage, taxa were often only identifiable to morphological grouping or genus, rather than species.

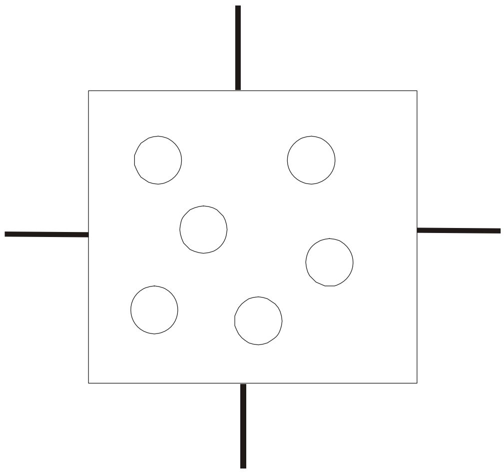

Figure 2.5. Layout of first (left) and second (right) video surveys.

To assess the effects of location (lease versus control), site within location, and distance along the transect on epifaunal composition, a PERMANOVA was conducted, followed by an nMDS to visualise differences, as described above. Twenty-nine different taxa and substrate categories were used in the analysis. For this analysis, location and segment number were treated as fixed and orthogonal, while site was random and nested within location. The four directions were treated as replicates. Due to the requirements of PERMANOVA for a balanced data set, the fallowed lease was excluded from this analysis, although it was included in the nMDS.

The second survey followed the first, with the exception that lease site transects actually radiated out from cages (Figure 2.5), and there were no control site transects. Two cages in each of three lease sites were surveyed, with transects running roughly north and south from each cage. No transect ended < 100 m from an adjacent cage. These transects were conducted on 28 November 2005. Data analysis was as described above, with the exception that both abundance and percent cover were recorded for each of ten non-overlapping frames in each quarter of each transect. This allowed direction to be a factor in the analysis, and it was treated as fixed.

2.2.5 Seagrass

To determine if the farming of yellowtail kingfish in Fitzgerald Bay is having any affect on nearby seagrasses, samples of seagrass were collected on 28-29 November 2005 from adjacent to leases, at control sites within the bay but greater than 1 km from any lease, and from control sites outside the bay in a region not known to be directly influenced by anthropogenic nutrient inputs (ie lacking aquaculture activity, coastal development or riverine runoff). In each of these three areas, ten replicate quadrats (25 x 25 cm) at each of two sites were harvested of aboveground biomass, which was then frozen prior to later analysis. The impact sites were located as close as possible to active yellowtail kingfish leases (within 250 m of a lease boundary – Figure 2.6, Table 2.3), as the leases are placed in deeper water and thus do not lie directly over seagrass. To assess if there are any localised impacts of aquaculture on seagrasses, within-bay control sites were located in the north of Fitzgerald Bay, and to assess any bay-wide effects, regional control sites were located approximately 3.5 km further north, outside the bay. In all cases, sampling was conducted in 4-5 m water depth, and the seagrass species collected was *Posidonia australis.*

In the laboratory, the total number of leaves and shoots within each quadrat were counted. The maximum leaf length and width were also measured. Ten intact shoots were then haphazardly selected, and the longest leaf from each removed to determine epiphyte loading. These leaves were dried at 60°C for 48 hours and weighed. All ten leaves from each quadrat were then placed in 5% hydrochloric acid, rinsed in freshwater, and scraped to remove all epiphytes. Leaves were then redried at 60°C for another 48 hours and reweighed to determine epiphyte free dry weight. The dry weight of epiphytes was then calculated by subtraction. A further ten leaves with low epiphyte loading were randomly selected to determine carbon and nitrogen content. Each leaf was scraped clean on aluminium foil cleaned with methanol, and placed in a glass jar that had been muffled at 450°C overnight and then weighed. Each jar was then reweighed to obtain wet leaf biomass, and frozen prior to CN analysis. Samples were freeze dried overnight and then ground to a fine powder. An aliquot (150mg) was then analysed for % C and % N on a LECO Truspec CNS Elemental Analyser. The remainder of the sample was dried at 60°C for 48 hours and weighed to obtain total dry weight (seagrass plus epiphytes) for each quadrat.

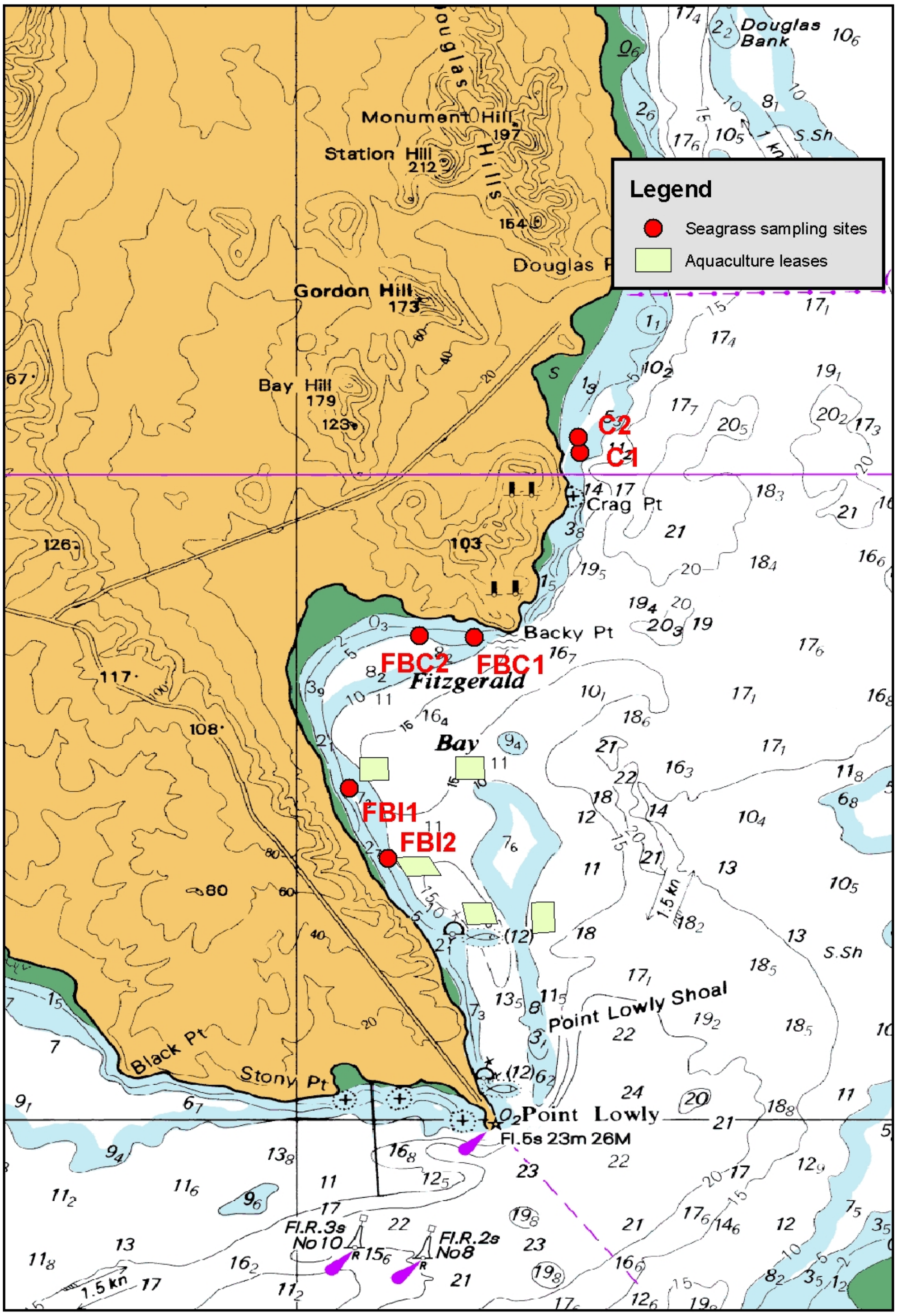

Figure 2.6. Map of Fitzgerald Bay showing lease sites and locations of seagrass sampling sites.

Table 2.3. Co-ordinates of seagrass sampling sites (WGS 84)

Location	Site	Latitude (°S)	Longitude (°E)
Lease	FBI1	32.9419	137.7589
Lease	FBI2	32.9539	137.7656
Lease control	FBC1	32.9161	137.7803
Lease control	FBC2	32.9158	137.7708
Bay control	C1	32.8844	137.7983
Bay control	C2	32.8819	137.7981

To determine if seagrass morphology and biomass varied between locations or sites within locations, a PERMANOVA was conducted. The variables used for this analysis were biomass, maximum leaf length and width, number of leaves and number of shoots (all at the quadrat level). As each variable is measured on a different scale, all variables were standardised by their range prior to analysis, to ensure that those with large ranges did not dominate. Location was treated as a fixed factor, while site was random nested within location. The analysis was conducted on untransformed data, using Euclidean distances. 4999 permutations of the residuals under a reduced model were used to calculate probability values. To provide a visual representation of any difference, principal components analysis (PCA) was conducted using PRIMER.

The response of individual seagrass parameters was analysed using standard univariate ANOVA. Location was treated as a fixed factor, with site nested within location. Biomass was square root transformed prior to analysis, to meet the assumption of normality, while all other variables were approximately normal without transformation. It was not possible to meet the assumption of homogeneity of variance for any variable other than biomass and %N (based on Levene's test), so a conservative alpha level of 0.01 was used for these variables (Underwood 1997).

Epiphyte biomass was expressed on a mg cm^{-2} basis by multiplying the length of each leaf by its width, adding these together for all ten leaves from each quadrat, and then multiplying by 2 to account for both sides of the leaf. Univariate ANOVA was used to determine if epiphyte biomass varied among locations or sites. Untransformed data were used, as these met the assumptions of normality (QQ plots) and homogeneity of variances (Levene's test).

2.2.6 Wildfish

Inputs of feed from aquaculture into the surrounding environment may attract wildfish assemblages to lease areas, and influence their demography through feed supplementation. The wildfish assemblages around cages in Fitzgerald Bay were divided into two components, each of which was studied separately. Attraction of demersal fish to cages was studied as part of an honours project (Williams 2004), and the resultant thesis is reproduced here as Appendix 3. This component is not considered further until the discussion of this chapter. A number of methods were used to try and survey pelagic fish around cages, also in Fitzgerald Bay. Williams (2004, see Appendix 3) trialled a remotely operated vehicle, along with both baited and unbaited remote underwater video cameras. On January 5 and 6, 2005, a series of both floating and sinking multipanel gill nets were set at reference and cage sites. Each net was set just prior to dusk, and pulled up one hour later. This short soak time was used due to animal ethics requirements. The gill nets used were 40 m long and 2 m high with panels of alternating mesh sizes. Nets were set at a range of depths to cover the entire water column. Four nets were set each night, one floating and one sinking at a reference site 1 km away from any lease, and one each within 100 m of a cage. It was not possible to set nets immediately adjacent to a cage due to risks of entanglement with anchor lines etc. Diver surveys were also trialled in May 2005, with two divers doing both point counts of pelagic fish from a stationary position on the bottom adjacent to a cage, and a swim transect at a constant depth of ~ 10 m around a cage.

2.2.7 Sediments

Sediment samples were collected in May 2005 at the edge of the commercial pens and at two control sites located at least 1 km from any aquaculture lease. Sediments were collected by divers using 67 mm (i.d.) PVC tubes capped with rubber bungs. Upon retrieval, the overlying water in the tube was carefully discarded to minimise surface disturbance and the sediment extruded onto a clean stainless steel table. Four cores were collected for the analysis of total carbon (TC). The top layer (0-1 cm) of each core was sliced, transferred into a pre-combusted glass jar and stored frozen (-30 ^{o}C). Sediment samples were freeze-dried, sieved to 500 µm to remove large shell fragments, and homogenized with a mortar and pestle. Aliquots were weighed into foil capsules and analysed for TC by Continuous-Flow stable Isotope Ratio Mass Spectrometry (CF IRMS) using a Europa Scientific ANCA-SL elemental analyser coupled to a Geo 20-20 Mass Spectrometer after decarbonation with 1N HCl. TC concentrations are reported as a percentage of total dry sediment. Two cores were collected for the determination of ammonium and phosphate in porewaters. The top layer (0-2 cm) of each core was sliced, transferred into a pre-weighed centrifuge tube of known volume and stored refrigerated (4 ^{o}C) for up to 3 h before transfer to the laboratory. Sediments collected for porewater analyses were centrifuged at 3,000 rpm for 10 minutes, the supernatant filtered (0.45 µm) and stored frozen (-30 ^{o}C). Ammonium and phosphate were determined spectrophotometrically by flow injection analysis (FIA) in a QuickChem 8000 Automated Ion Analyser (APHA-AWWA-WPCF, 1998).

For all three variables, the effects of aquaculture and location within Fitzgerald Bay were tested using standard univariate ANOVA. In the case of both porewater ammonia and phosphorus concentrations, the data were natural log transformed prior to analysis to improve normality and heterogeneity of variances.

2.3 Results

2.3.1 Water quality

The PERMANOVA indicated that water quality varied significantly between sites, but not between the northern and southern parts of Fitzgerald Bay, nor between cages and control areas > 1 km away from cages (Table 2.4). The PCA, does, however, indicate some separation between cage and control sites (Figure 2.7, Table 2.5), although the southern lease sites do overlap with the control sites. The reason for this discrepancy is the very high variation between sites within the lease by treatment interaction, which makes it impossible to pick up any effects of either location or aquaculture. Sites on the far right of the PCA plot (Figure 2.7) have high levels of ammonia and total nitrogen, but low levels of nitrate + nitrite (and vice-versa for sites on the left of the plot), while sites at the bottom of the plot have high turbidity (and sites at the top low) (Table 2.6).

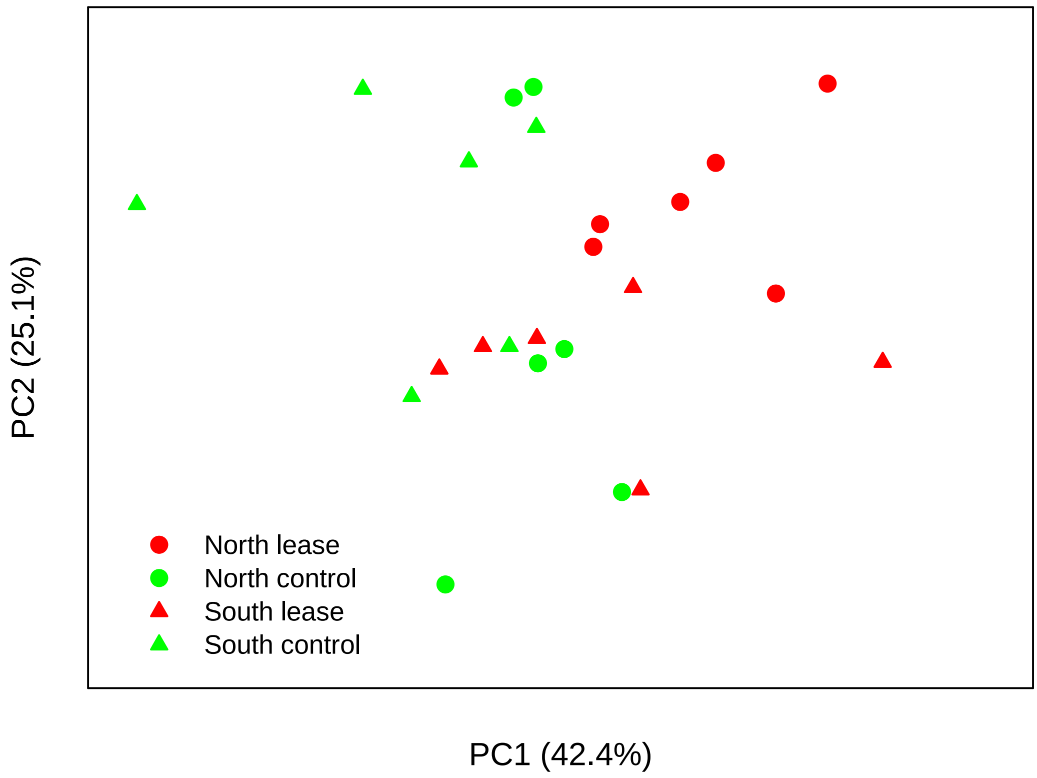

Figure 2.7. Principal components analysis showing effects of location and aquaculture on water quality.

Table 2.4. PERMANOVA results for water quality in Fitzgerald Bay.

Source	df	SS	F	P
Location	1	0.20	0.023	0.87
Aquaculture	1	4.68	0.21	0.62
Site(L x A)	8	71.99	20827	**0.0002**
Locn x Aqua	1	22.82	2.54	0.16
Residual	12	0.005		
Total	23	99.70		

Table 2.5. Variance extracted on each axis by a PCA on Fitzgerald Bay water quality data.

Axis	Eigenvalue	% variance	Cumulative % variance
1	1.697	42.43	42.43
2	1.003	25.08	67.51
3	0.764	19.10	86.61
4	0.536	13.39	100

Table 2.6. Eigenvector loadings obtained from the PCA on Fitzgerald Bay water quality data.

	Eigenvector			
	1	2	3	4
TN	0.61	0.27	-0.13	0.74
NH_4	0.58	0.27	-0.41	-0.65
NO_x	-0.50	0.24	-0.82	0.17
Turbidity	0.23	-0.89	-0.39	0.07

Univariate analyses showed significant variation in ammonia levels associated with both the proximity to finfish cages and location (north/south) in Fitzgerald Bay (Table 2.7, Figure 2.8). Ammonia levels in the north of the bay were 75% higher than in the south, and adjacent to cages they were 81% higher than at control sites. The measured concentrations of nitrate + nitrite also varied between sites, but not with proximity to aquaculture cages or location. Total nitrogen, turbidity, salinity and temperature did not vary with any factor, and total phosphorus was below detectable levels (0.01 mgL^{-1}) in all samples.

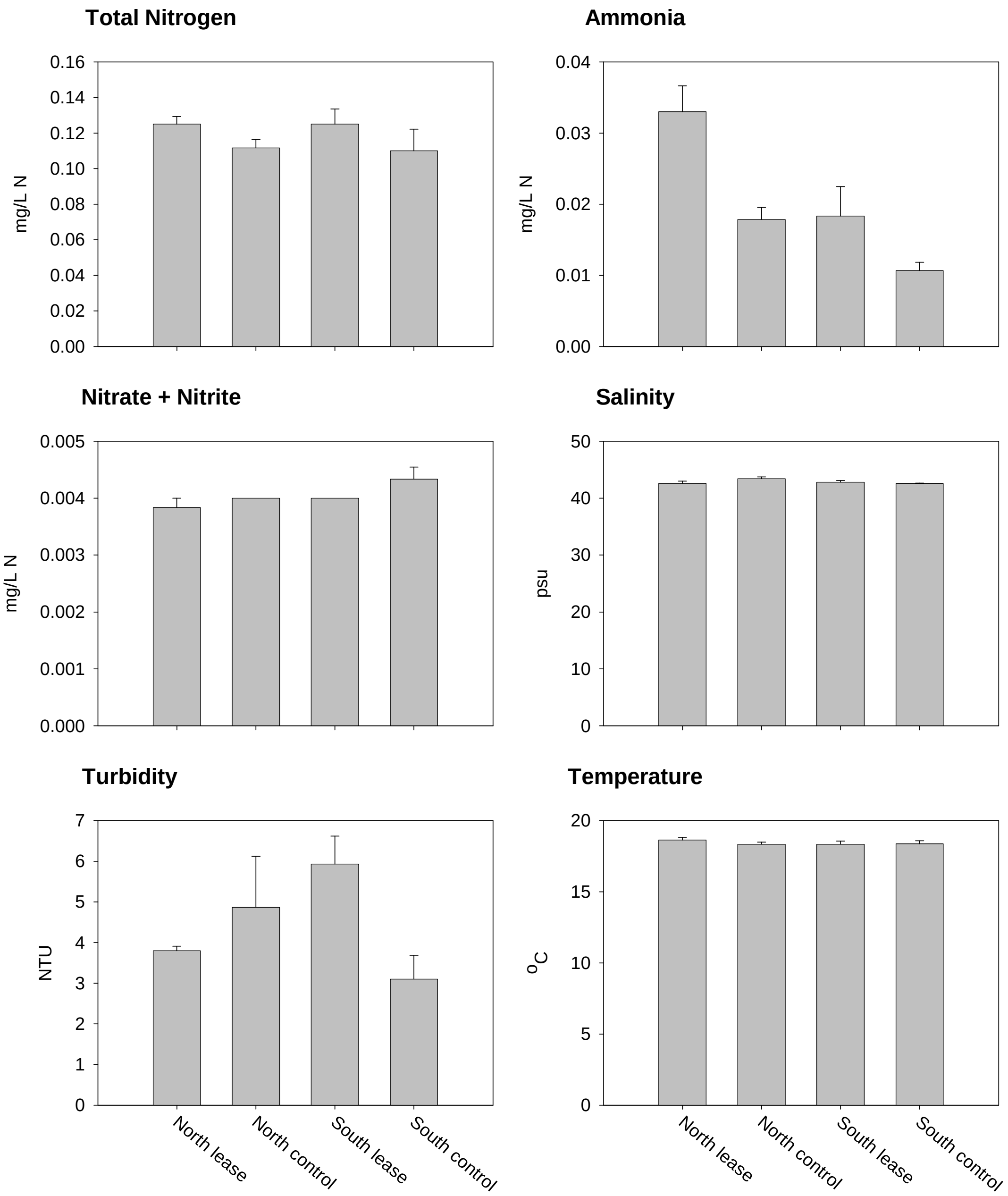

Figure 2.8. Variation in individual water quality parameters at Fitzgerald Bay as a function of location and proximity to finfish aquaculture cages. Error bars are se.

Table 2.7. Univariate ANOVA results for effects of aquaculture on water quality in Fitzgerald Bay.

Source	df	SS	F	P	SS	F	P
		Total Nitrogen			**Turbidity**		
Location	1	0.000004	0.01	0.92	0.10	0.022	0.89
Aquaculture	1	0.001	2.98	0.12	2.34	0.52	0.49
Locn x Aqua	1	0.000004	0.01	0.92	11.4	2.54	0.15
Site(L x A)	8	0.003	1.07	0.44	36.0		
Residual	12	0.005					
		Ammonia			**Salinity**		
Location	1	0.001	14.19	**0.005**	0.30	0.46	0.52
Aquaculture	1	0.001	15.52	**0.004**	0.24	0.37	0.56
Locn x Aqua	1	0.00008	1.68	0.23	0.80	1.23	0.30
Site(L x A)	8	0.0004	0.95	0.51	5.21		
Residual	12	0.001					
		Nitrate + Nitrite			**Temperature**		
Location	1	0.0000004	1.8	0.22	0.053	0.19	0.68
Aquaculture	1	0.0000004	1.8	0.22	0.053	0.19	0.68
Locn x Aqua	1	0.00000004	0.2	0.67	0.083	0.29	0.60
Site(L x A)	8	0.0000017	5	**0.007**	2.27		
Residual	12	0.0000005					

2.3.2 Chlorophyll and phytoplankton

With the exception of sampling time, chlorophyll a concentrations did not vary as a function of any of the factors included in the experimental design, either for the surface samples over both years or for the 2005 samples (Table 2.8, Figure 2.9). That is, none of proximity to an aquaculture lease, position within Fitzgerald Bay, position within a lease, or depth, affected the amount of chlorophyll in the water. Chlorophyll levels were 30% higher in November 2004 than in August 2005, however.

Phytoplankton composition in 2004 was not affected by either proximity to yellowtail kingfish cages or location within Fitzgerald Bay (Table 2.9, Figure 2.10). There was significant small-scale variability between sites, however. A similar result was found for the 2005 surface samples, and in the south of the bay there was an interaction between depth and site (Table 2.9, Figures 2.11, 2.12). When the surface samples for both years were analysed together, there was a significant interaction between site and sampling time, as well as significant main level effects of these two factors (Table 2.9). The nMDS shows a clear difference in species composition between years (Figure 2.13). This difference is due to a large suite of species being much more abundant in November 2004 than in August 2005, and only a few species being less abundant (Table 2.10), as would be expected for a spring versus winter comparison.

When the data for 2004 were analysed at a higher taxonomic level (diatoms/dinoflagellates/other), the results were similar to what was found at the species level, with the exception that there was no longer significant variation between sites (Table 2.9, Figure 2.14). This result indicates that the high level of variability on small spatial scales is due to changes in composition between closely related species.

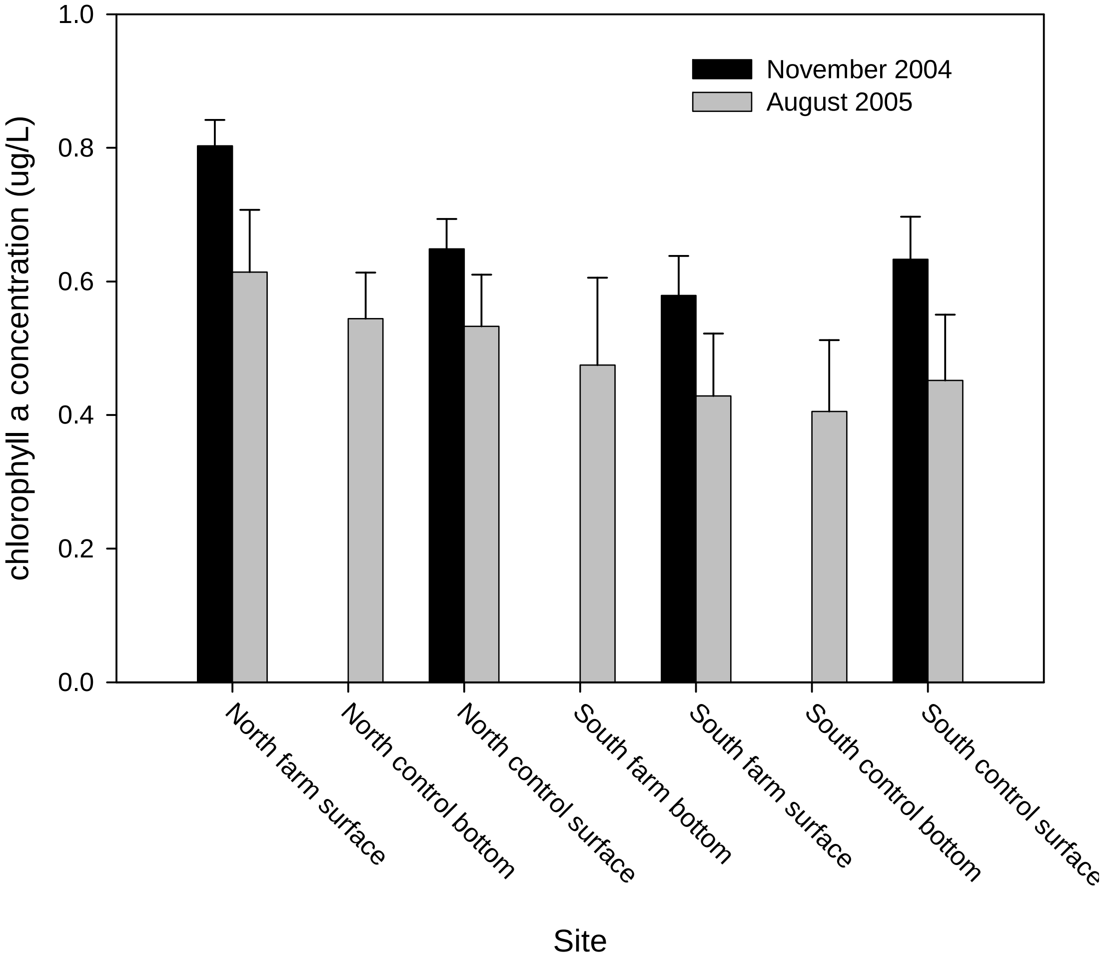

Figure 2.9: Chlorophyll a concentrations in the waters of Fitzgerald Bay. Error bars are se.

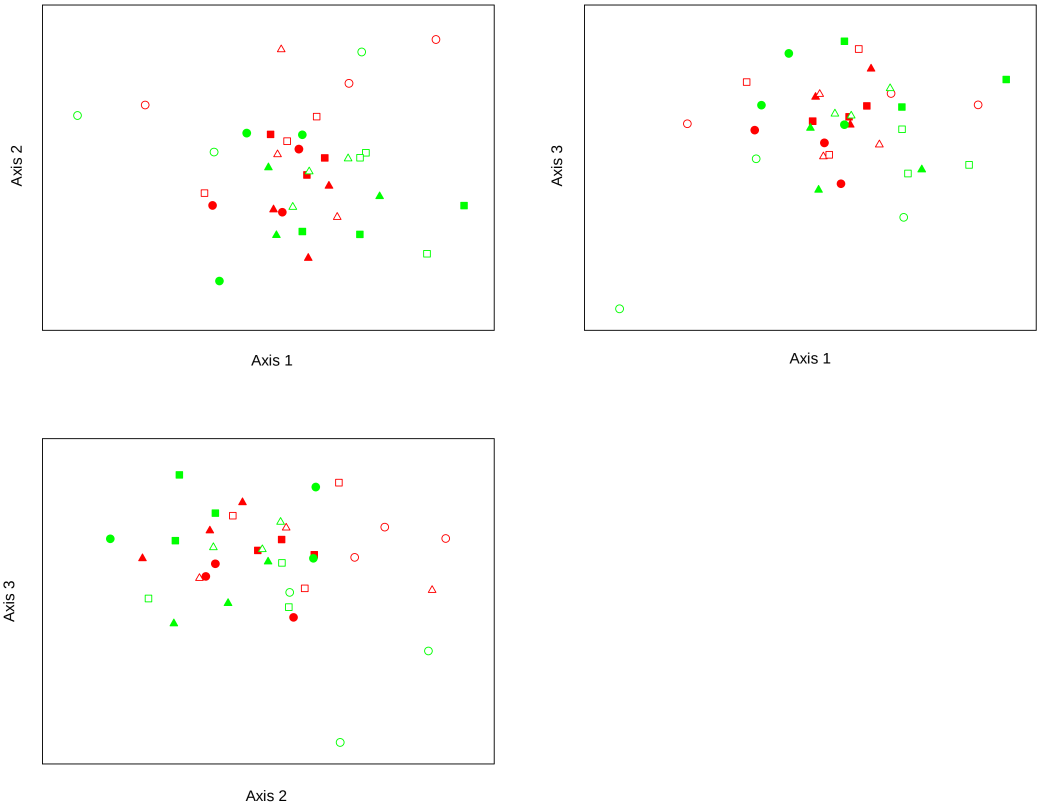

Figure 2.10. nMDS of phytoplankton composition in Fitzgerald Bay for November 2004. Green symbols represent reference sites; red, aquaculture sites. Closed symbols are for the northern part of the bay and open, southern. Shapes represent sites. Stress = 16.1.

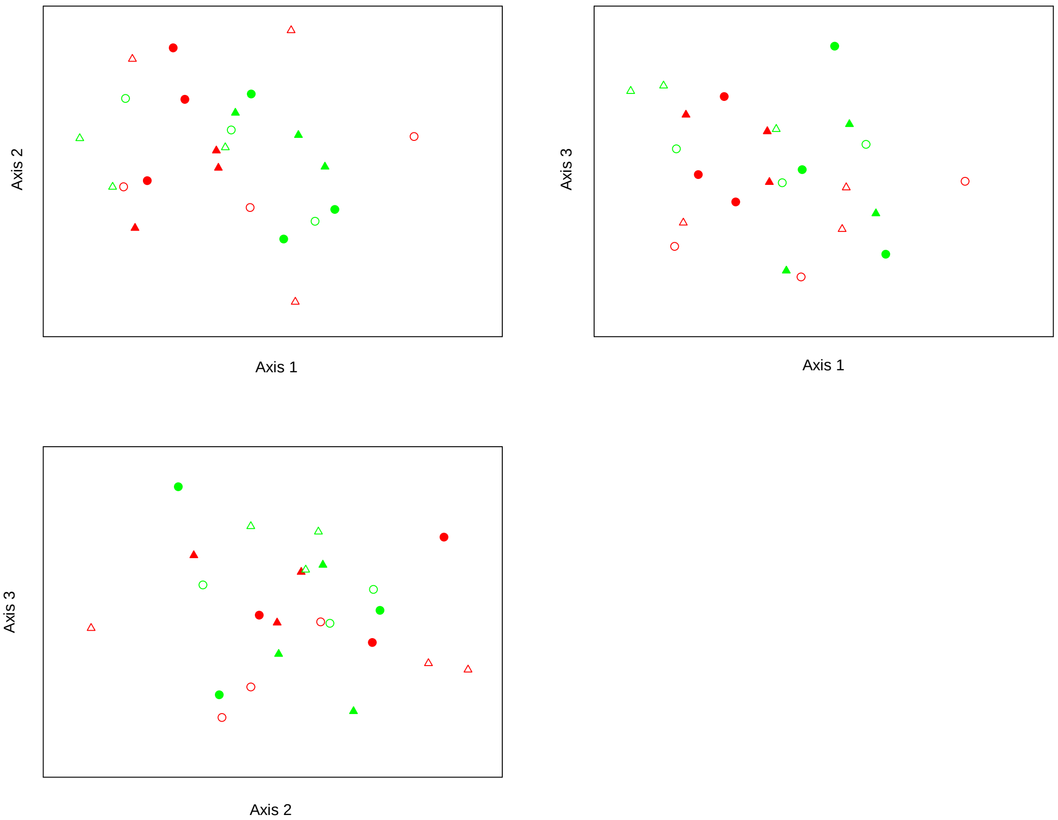

Figure 2.11. nMDS of surface phytoplankton composition in Fitzgerald Bay for August 2005. Green symbols represent reference sites; red, aquaculture sites. Closed symbols are for the northern part of the bay and open, southern. Shapes represent sites. Stress = 19.2.

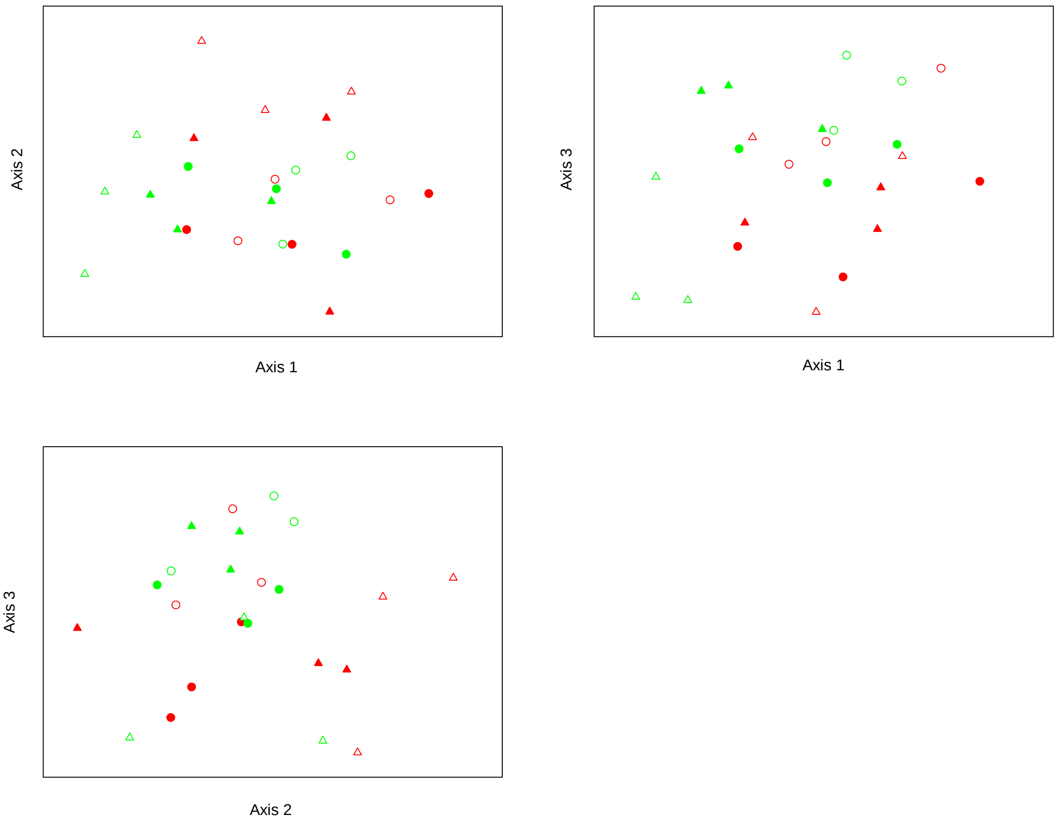

Figure 2.12. nMDS of phytoplankton composition in southern Fitzgerald Bay for August 2005.

Green symbols represent reference sites; red, aquaculture sites. Closed symbols are for surface samples and open, bottom. Shapes represent sites. Stress = 19.2.

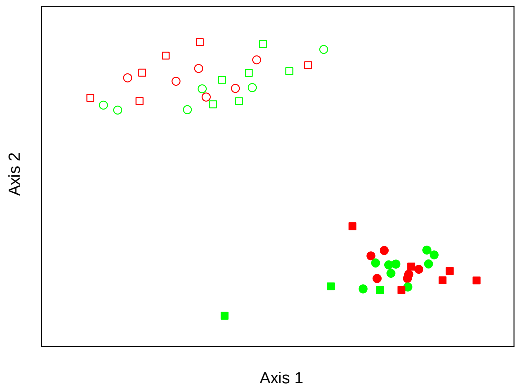

Figure 2.13. nMDS of phytoplankton composition at the surface of Fitzgerald Bay for November 2004 and August 2005.
Green symbols represent reference sites; red, aquaculture sites. Closed symbols are for 2004 and open 2005. Circles represent north sites; squares, south. Stress = 10.1.

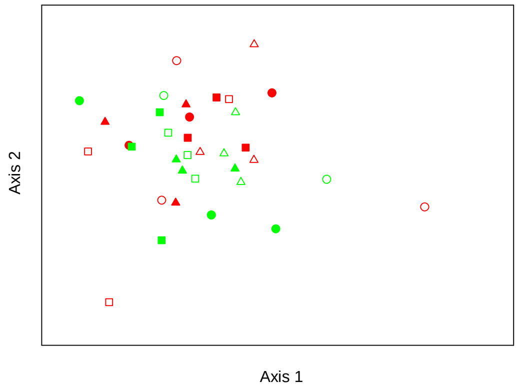

Figure 2.14. nMDS of phytoplankton composition in Fitzgerald Bay for November 2004, with taxa grouped as Diatoms, Dinoflagellates or Other.
Green symbols represent reference sites; red, aquaculture sites. Closed symbols are for the northern part of the bay and open, southern. Shapes represent sites. Stress = 11.0.

Table 2.8. ANOVA results for chlorophyll a concentrations in Fitzgerald Bay.

Source	df	SS	F	P
Surface samples				
Month	1	0.204	2025	**0.014**
Aquaculture	1	0.015	0.20	0.73
Lease	1	0.215	4.56	0.51
Site(Lease*Aqua)	8	0.200	1.69	0.32
Month*Treat	1	0.008	0.36	0.66
Month*Lease	1	0.000	0.004	0.96
Lease*Treat	1	0.075	2.12	0.33
Month*Lease*Treat	1	0.023	1.53	0.28
Month*Site(Lease*Aqua)	4	0.059	1.00	0.42
Residual	40	0.589		
2005 samples				
Aquaculture	1	0.03	1.53	0.45
Lease	1	0.175	8.62	0.24
Depth	1	0.003	0.49	0.66
Site(Lease*Aqua)	4	0.056	3.16	0.19
Lease*Aqua	1	0.016	1.65	0.25
Aqua*Depth	1	0.013	2.91	0.19
Lease*Depth	1	0.005	1.14	0.37
Depth*Site(Lease*Aqua)	3	0.13	2.03	0.13
Residual	28	0.061		

Table 2.9. PERMANOVA results for phytoplankton composition in Fitzgerald Bay.

Source	df	SS	F	P
NOVEMBER 2004				
Location	1	274	1.01	0.48
Aquaculture	1	235	0.87	0.62
Site (L x A)	8	2167	1.70	**0.0002**
Locn x Aqua	1	220	0.81	0.65
Residual	24	3824		
NOVEMBER 2004 (Grouped taxa)				
Location	1	6.9	0.65	0.60
Aquaculture	1	7.6	0.72	0.57
Site (L x A)	8	84.9	0.82	0.72
Locn x Aqua	1	7.3	0.69	0.69
Residual	24	309.9		
AUGUST 2005 - SURFACE				
Location	1	169	0.47	0.90
Aquaculture	1	338	0.94	0.52
Site (L x A)	4	1444	1.34	**0.049**
Locn x Aqua	1	671	1.86	0.13
Residual	16	4309		
AUGUST 2005 - SOUTH				
Aquaculture	1	503	0.74	0.67
Site (A)	2	1357	2.30	**0.0002**
Depth	1	253	0.49	0.68
Depth x Aqua	1	546	1.06	0.42
Site (A) x Depth	2	1030	1.75	**0.0068**
Residual	16	4719		
BOTH YEARS - SURFACE				
Year	1	13479	42.84	**0.0038**
Location	1	192	0.57	0.85
Aquaculture	1	197	0.59	0.85
Site (LxA)	4	1347	1.54	**0.013**
Year x Locn	1	222	0.71	0.59
Year x Aqua	1	313	0.99	0.44
Year x Site (LxA)	4	1258	1.44	**0.03**
Locn x Aqua	1	337	1.00	0.45
Year x Locn x Aqua	1	468	1.49	0.25
Residual	32	6996		

Table 2.10. SIMPER analysis showing main phytoplankton species driving differences between November 2004 and August 2005 samples.

Species	**Nov 2004 Mean Abundance**	**Aug 2005 Mean Abundance**	**% Contribution to Dissimilarity**	**Cumulative % Dissimilarity**
Minidiscus sp.	22979.17	927.92	3.99	3.99
Pleurosigma spp.	3000	22.08	3.95	7.93
Chaetoceros spp.	54395.83	9425	3.2	11.13
Bacteriastrum elegans	16979.17	2139.17	3.1	14.24
Teleaulax acuta	166.67	1129.17	2.86	17.1
Rhizosolenia setigera	250	1897.92	2.84	19.94
Thalassiosira cf. mala	4104.17	306.25	2.74	22.68
Calycomonas sp.	20.83	672.08	2.68	25.36
Leptocylindrus minimus	0	626.67	2.47	27.83
Gymnodinioid spp.	17000	3368.75	2.4	30.24
Hemiselmis sp.	19062.5	3450	2.38	32.61
Plagioselmis prolonga	24812.5	5133.33	2.35	34.96
Cocconeis spp.	2458.33	390	2.22	37.18
Naviculoid spp.	5395.83	516.67	2.21	39.4
Apedinella spinifera	125	539.17	2.18	41.58
Protoperidinium spp.	0	394.58	2.18	43.76
Dactyliosolen fragilissimus	3437.5	417.08	2.14	45.91
Scrippsiella spp.	83.33	456.25	2.14	48.05
Guinardia striata	1333.33	550.42	2.07	50.12
Chrysochromulina spp.	13104.17	2170.83	2.05	52.17
Licmophora sp.	3354.17	512.08	1.99	54.16
Mesodinium rubrum	41.67	399.17	1.96	56.12
Thalassiosira sp.	1854.17	415.42	1.96	58.08
Pyramimonas spp.	9125	1596.25	1.95	60.04
Thalassionema sp.	1250	285.42	1.93	61.97
Leptocylindrus danicus	125	390	1.76	63.73
Ochromonas spp.	2750	677.08	1.63	65.36
Heterosigma sp.	20.83	279.58	1.61	66.97
Eutreptiella spp.	458.33	221.67	1.59	68.56
Rhodomonas salina	0	239.58	1.56	70.12
Minutocellus spp.	520.83	57.08	1.48	71.6
Leucocryptos marina	2458.33	892.92	1.45	73.05
Cerataulina pelagica	250	223.75	1.41	74.47
Cylindrotheca closterium	6833.33	2154.17	1.37	75.83
Nitzschia spp.	3291.67	1155.83	1.36	77.19
Gyrodinium spp.	5562.5	2045.42	1.32	78.51
Heterocapsa rotundata	10416.67	6085.42	1.21	79.72
Unidentified bodonids	0	167.5	1.14	80.86
Skeletonema costatum	208.33	303.33	1.1	81.96
Entomoneis spp.	395.83	35	1.09	83.05
Pseudonitzschia delicatissima complex	2729.17	1401.25	1.06	84.11
Tetraselmis spp.	1729.17	968.33	1.03	85.14
Prorocentrum cordatum	41.67	142.92	1	86.14

Chlorophyll a levels throughout most of Spencer Gulf were considerably less than 1 µg L^{-1} on November 8, 2004 (Figure 2.15), as detected via MODIS satellite imagery. However, values of up to 3 µg L^{-1} were recorded along the shores of the northern section of the gulf, including Fitzgerald Bay (Figure 2.16). This compares to values measured from water samples collected in situ on November 11 that ranged between 0.6-0.8 µg L^{-1} on average (Figure 2.9). While the results for November 15 were broadly similar, overall chlorophyll a levels appear to have increased over the intervening week by up to 0.5 µg L^{-1} in places. On 16 August 2005, the extent of high chlorophyll concentrations had decreased, although inshore waters in Fitzgerald Bay still appeared to have ~ 3 µg L^{-1}. This compares to average values of 0.4-0.6 µg L^{-1} from in situ water samples collected between August 16 and 18 (Figure 2.9). The reasons for these discrepancies are examined in the discussion below.

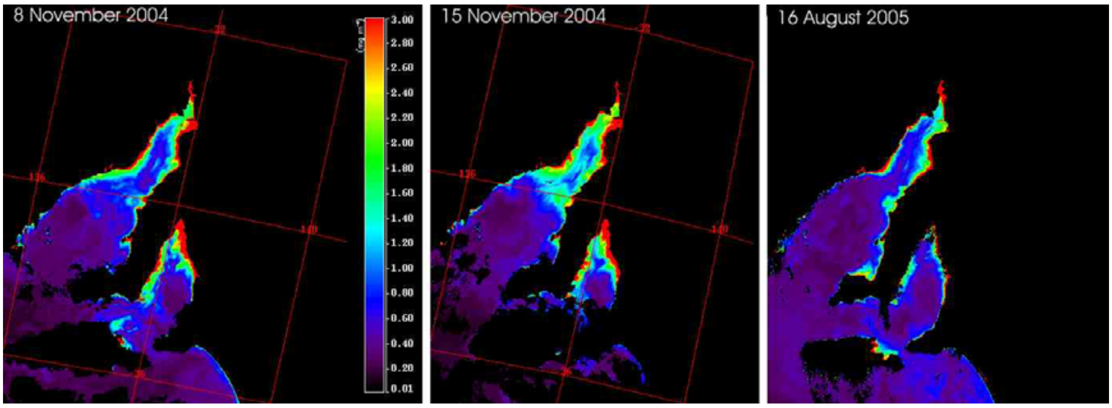

Figure 2.15. Chlorophyll levels in the South Australian gulfs derived from remote sensing. Values expressed in mg m^{-3}, or µg L^{-1}.

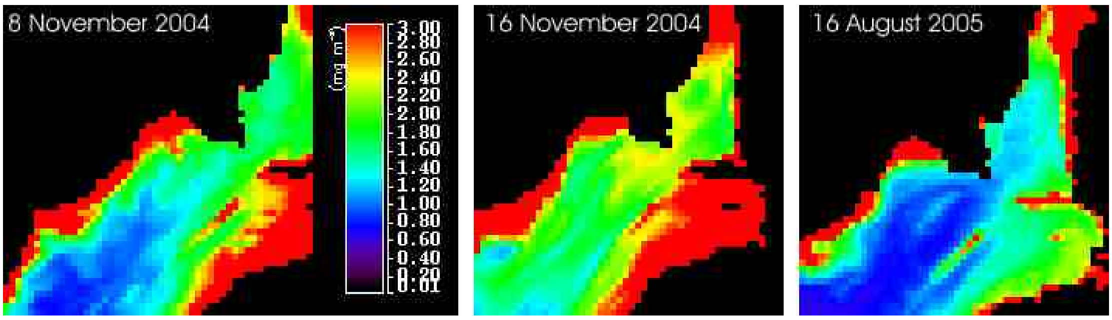

Figure 2.16. Detail of chlorophyll levels in northern Spencer Gulf, including Fitzgerald Bay, derived from remote sensing. Values expressed in mg m^{-3}, or µg L^{-1}.

2.3.3 Infauna

While the PERMANOVA indicated a highly significant affect of distance, as well as lease, on the infaunal composition (Table 2.11), this was not obvious in the nMDS (Figure 2.17). While for some cages there are differences between different distances, these are not generally consistent with a gradient effect, and often the 1000m samples substantially overlap the 0 m samples.

Table 2.11. PERMANOVA results showing gradient effects on infaunal composition away from YTK cages.

Source	df	SS	F	P
Distance	1	21481	6.38	**0.0002**
Lease	1	26210	5.65	**0.0014**
Cage(Lease)	2	9271	1.38	0.13
Residual	135	454663		
Total	139	511625		

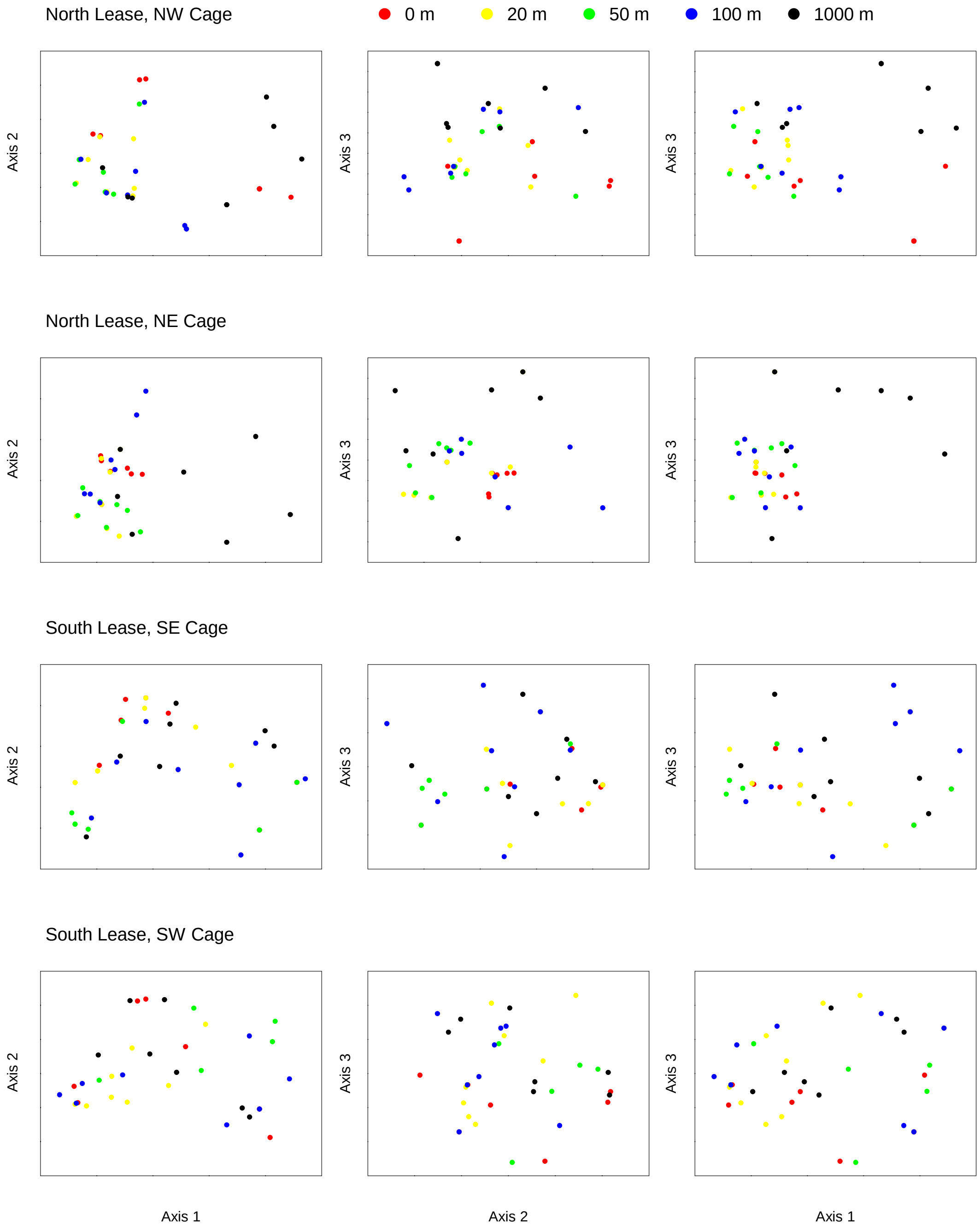

Figure 2.17. nMDS showing relationship between distance from YTK cage and infaunal composition. Note, this is a 3D nMDS, and while each cage has been plotted separately, the plots represent a single analysis. Stress = 0.20.

Only four taxa of the 59 found in the samples occurred with a total abundance of 20 or more (in 146 samples). Three of these were families of polychaetes: Capitellidae (63), Cirratulidae (206) and Spionidae (65), while the fourth was a family of tanaid crustaceans: Apseudidae (62). The next most abundant taxon was the polychaete family Maldanidae, which was represented by only 17 individuals.

The Apseudidae only occurred at the north lease site between 0 and 100 m from the cages, with abundance being fairly constant over these distances (Figure 2.18). Due to the absence of Apseudidae from the south lease, ANCOVA was performed for data from the north lease only. The assumption of homogeneity of slopes was met ($F_{1,69}$=0.188, p=0.666). There was a significant effect of distance, but no effect of cage (Table 2.12).

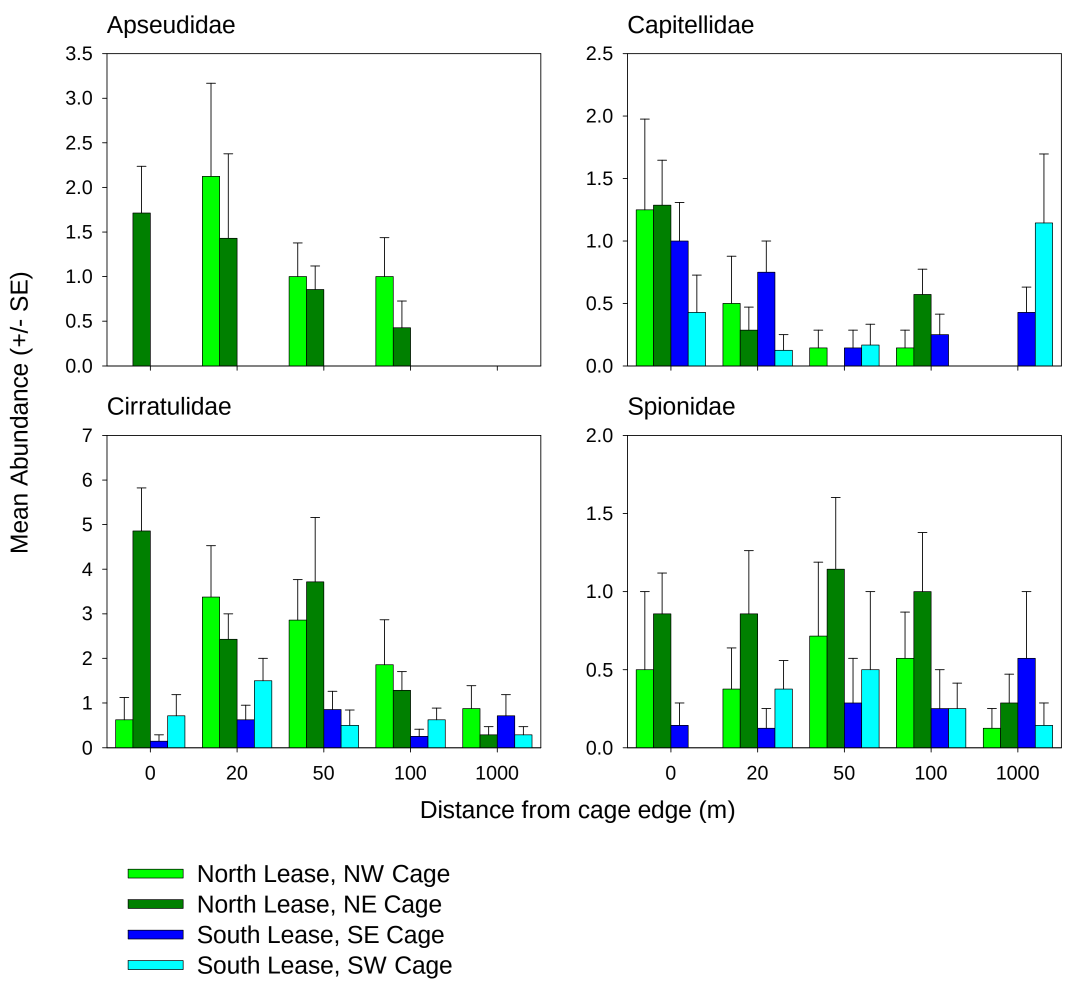

Figure 2.18. Abundance of abundant infaunal taxa as a function of distance from YTK cages.

Table 2.12. ANCOVA results for effects of distance from cage on abundance of the four most common infauna taxa at Fitzgerald Bay.

Source	df	SS	F	P	df	SS	F	P
	Apseudidae				**Capitellidae (north lease)**			
Distance	1	3.35	11.75	**0.001**	1	1.22	7.19	**0.009**
Cage	1	0.062	0.22	0.641	1	0.062	0.37	0.546
Error	70	19.937			70	11.86		
	Capitellidae (south, cage 1)				**Capitellidae (south, cage 2)**			
Distance	1	0.036	0.22	0.64	1	1.18	8.4	**0.007**
Error	35	5.737			34	4.76		
	Ciratullidae (north lease)				**Ciratullidae (south lease)**			
Distance	1	10.22	13.87	**<0.001**	1	0.23	0.55	0.46
Cage	1	1.62	2.20	0.14	1	0.48	1.17	0.28
Error	70	51.56			70	28.68		
	Spionidae							
Distance	1	0.26	1.52	0.22				
Lease	1	1.64	2.86	0.23				
Cage(Lease)	2	1.15	3.34	**0.038**				
Error	141	24.19						

The Capitellidae were abundant at the 0 m sites in both leases, declining out to the 100 m sites. However, at the south lease, abundance then increased again at 1000 m. In this case, while there was no interaction between Cage & Distance ($F_{2,138}$=2.55, p=0.082), there was between Lease & Distance ($F_{1,140}$=9.03, p=0.003), necessitating separate analyses for the 2 leases. For the northern lease, there was no interaction between Cage & Distance ($F_{1,69}$=0.068, p=0.795), and no cage effect, but there was a significant effect of distance (Table 2.12). For the southern lease, the interaction between Cage & Distance was significant ($F_{1,69}$=5.35, p=0.024), so each cage had to be analysed separately. For cage 1, there was no effect of distance, while for cage 2 there was (Table 2.12), although the r^2 was only 0.2, indicating that distance only explains 20% of the variance in the abundance of Capitellids around this cage.

The Cirratulidae declined in abundance at the north lease with increasing distance from the cages, especially past the 50 m site, but at the south lease abundance was relatively constant with distance, although consistently low. There was no interaction between Cage & Distance ($F_{2,138}$=2.10, p=0.127), but there was between Lease & Distance ($F_{1,140}$=6.22, p=0.014), thus the 2 leases were analysed separately. For the north lease, there was no interaction between Cage & Distance ($F_{1,69}$=2.61, p=0.111), and no cage effect, but there was a significant distance effect (Table 2.12). For the south lease, there was again no interaction between Cage & Distance ($F_{1,69}$=1.19, p=0.279) and no Cage effect, but this time there was no Distance effect either (Table 2.12).

Finally, the Spionidae did not show any clear trends with distance. Distance did not interact with either Cage ($F_{2,138}$=0.77, p=0.47) or Lease ($F_{1,140}$=3.62, p=0.059), meaning that both leases could be analysed together. For this taxon, there were significant differences between Cages within Leases, but no effects of Lease or Distance (Table 2.12).

Neither total abundance nor taxonomic richness showed any clear patterns in relation to distance from cages (Figure 2.19). For Abundance, there was no interaction between Cage & Distance ($F_{2,138}$=1.75, p=0.177), but there was between Lease & Distance ($F_{1,140}$=11.18, p=0.001), necessitating analysis at the Lease level. For the northern lease, there was no interaction between Cage & Distance ($F_{1,69}$=2.71, p=0.10), and no Cage effect, but there was a significant effect of Distance (Table 2.13). Total abundance declined approximately 3-fold between the 0 m and 1000 m sites (Figure 2.19). For the southern lease, there was no interaction between Cage & Distance ($F_{1,69}$=0.83, p=0.36), and no Cage or Distance effects (Table 2.13).

For Richness, there was no interaction between Cage & Distance ($F_{2,138}$=1.26, p=0.29), but there was between Lease & Distance ($F_{1,140}$=5.74, p=0.018), necessitating analysis at the Lease level again. For the northern lease, there was no interaction between Cage & Distance ($F_{1,69}$=2.51, p=0.12), and no Distance effect, but there was a significant effect of Cage (Table 2.13). For the southern lease, there was also no interaction between Cage & Distance ($F_{1,69}$=0.57, p=0.45), and no Cage or Distance effects (Table 2.13).

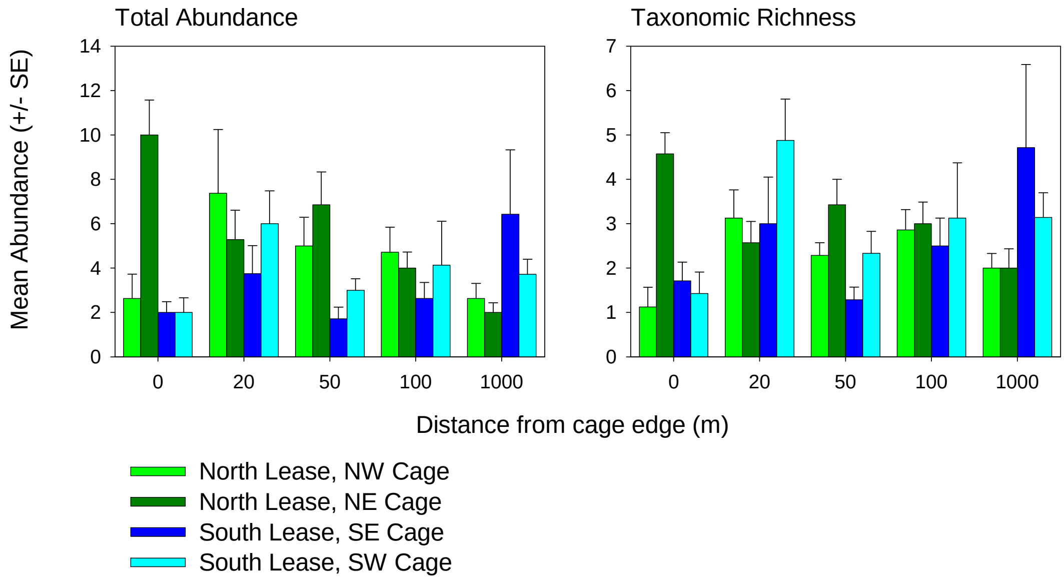

Figure 2.19. Total abundance and taxonomic richness of infaunal taxa as a function of distance from YTK cages.

Table 2.13. ANCOVA results for effects of distance from cage on infaunal abundance and richness (both ln transformed)

Source	df	SS	F	P	SS	F	P
		Abundance (north lease)			**Abundance (south lease)**		
Distance	1	3.80	9.27	**0.003**	1.21	2.91	0.09
Cage	1	1.32	3.23	0.077	0.45	1.10	0.30
Error	70	28.68			29.01		
		Richness (north lease)			**Richness (south lease)**		
Distance	1	0.40	2.15	0.15	1.09	3.29	0.074
Cage	1	1.16	6.20	**0.015**	0.27	0.83	0.37
Error	70	13.04			23.14		

2.3.4 Epifauna

For the video transects radiating out from lease edges, there was a significant lease by location along transect effect on assemblage structure (Table 2.14). However, there were no differences detected between lease and control sites. At both control sites, pairwise tests revealed differences between most quarters, with the exception of quarters 1 and 2 at the first control site, and quarter 4 was the same as both 2 and 3 at the second. At one lease site, all quarters were the same, while at the second only 1 and 2 were the same ($p<0.05$). The nMDS shows that lease sites have much more variable assemblage structure than control sites (Figure 2.20), and suggests that the assemblages at each site are relatively distinct. A PERMANOVA with all five sites, and no aquaculture term, showed similar results, with a significant Site by Quarter interaction ($F_{3,24}$=5.1, P=0.0002).

Table 2.14. PERMANOVA results showing response of epifaunal composition at lease edges to the presence of YTK cages.

Source	df	SS	F	P
Aquaculture	1	9650	0.91	0.67
Site(Aqua)	2	21100	34.2	**0.0002**
Quarter	3	7876	1.08	0.42
Aqua x Q	3	8479	1.16	0.37
Site(Aqua) x Q	6	14569	7.87	**0.0002**
Residual	48	14809		
Total	63	76483		

For the second series of video transects, those radiating out from individual cages, there was a complex pattern of small-scale variation in epifaunal composition, indicated by the numerous higher-order interactions in the PERMANOVA (Table 2.15). While there do appear to be effects of distance along the transect (ie quarter) on epifaunal composition, these effects change depending on what direction the transect went from the cage, and what cage it radiated out from. There is thus no clear indication that YTK farming influences epifaunal assemblages.

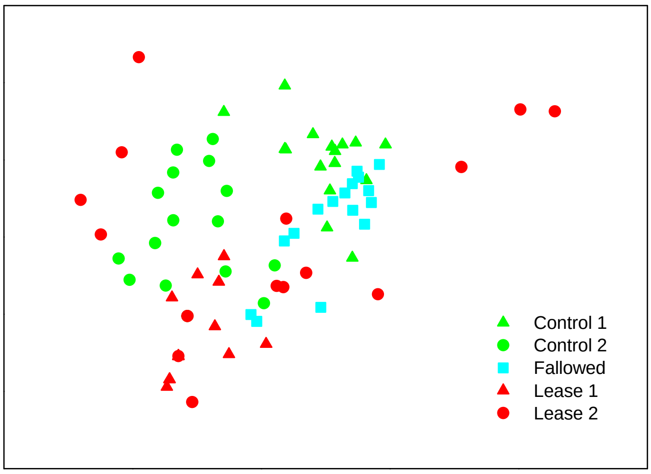

Figure 2.20. nMDS showing effect of active and fallowed YTK leases on epifaunal composition.

Table 2.15. PERMANOVA results showing response of epifaunal composition to the presence of individual YTK cages.

Source	df	SS	F	P
Lease	2	188799	16.8	0.065
Cage(Lease)	3	16870	3.25	**0.012**
Direction	1	11699	1.03	0.35
Quarter	3	3801	0.58	0.71
Lease x Direction	2	22621	1.73	0.28
Lease x Quarter	6	13140	1.00	0.48
Cage(L) x Direction	3	19545	3.76	**0.0058**
Cage(L) x Quarter	9	19796	1.27	0.23
Direction x Quarter	3	5006	0.16	0.94
Lease x Direction x Quarter	6	61271	3.11	**0.044**
Cage(L) x Direction x Quarter	9	29599	1.90	**0.033**
Residual	432	748127		
Total	479	1140273		

2.3.5 Seagrass

While the PERMANOVA indicated a highly significant affect of site, there was no overall affect of location on seagrass morphology and biomass (Table 2.16). The first two axes of the PCA explained 78.4% of the variation in the data, and were the only axes with eigenvalues greater than one (Table 2.17), and hence interpretation is restricted to these axes. There appears to be a separation between one of the lease sites and the two control sites within Fitzgerald Bay, and the other lease site and the two control sites outside of Fitzgerald Bay (Figure 2.21). The later group tend to have longer, thinner leaves, with lower overall biomass compared to the former, but with no differences in leaf or shoot densities (Table 2.18).

Table 2.16. PERMANOVA results showing effects of proximity to YTK cages on seagrass morphology and biomass.

Source	df	SS	F	P
Location	1	2.36	1.72	0.35
Site(Location)	2	2.74	6.88	**0.0002**
Residual	36	7.17		
Total	39	12.27		

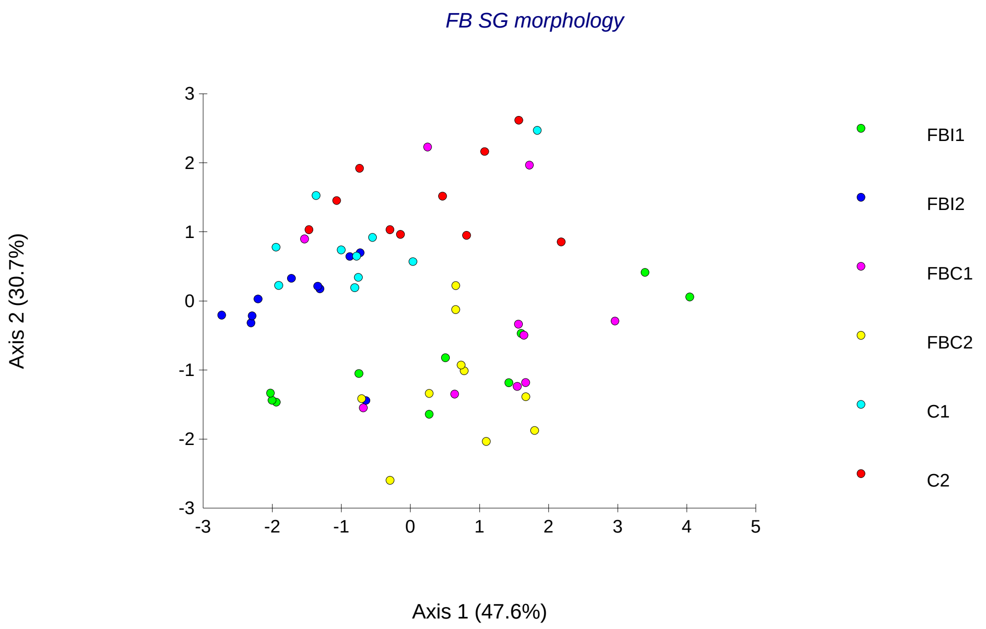

Figure 2.21. PCA showing site difference in seagrass morphology and biomass (see Table 2.17 and 2.18 for PCA results). See Figure 2.6 for site locations.

Table 2.17. Results of PCA on seagrass morphology and biomass.

PC	Eigenvalue	% Variation	Cumulative % Variation
1	2.38	47.6	47.6
2	1.54	30.7	78.4
3	0.69	13.9	92.2
4	0.33	6.6	98.8
5	0.06	1.2	100

Table 2.18. Axis loadings for first two axes of PCA on seagrass morphology and biomass.

Variable	PC1	PC2
# leaves	0.620	0.117
# shoots	0.615	0.001
Max leaf length	0.146	-0.602
Max leaf width	-0.255	0.643
Biomass	0.389	0.460

Individual univariate ANOVA's indicated that all four morphology parameters followed the multivariate pattern, with differences between sites within locations, but not between locations (Table 2.19). The number of leaves per quadrat varies from 41.5 at one of the lease sites, to 113 at one of the Fitzgerald Bay control sites, with a large amount of variation between lease sites (Figure 2.22). The number of shoots shows a very similar pattern, ranging from 13 to 43. Maximum leaf length is 600-650 mm for most sites, with the exception of one of the Fitzgerald Bay control sites, where it is 848 mm, and one of the external control sites, where it is only 511 mm. Leaf width is more variable, ranging between 6.9 mm and 12.6 mm, with the external control sites having the widest leaves.

In contrast to seagrass morphology, total aboveground biomass did vary between locations, but not between sites within locations (Table 2.19). The lease sites had the lowest total biomass, followed by the within bay control sites, with the external control sites having the highest biomass (Figure 2.22). Tukey's test indicates that the lease sites have significantly lower biomass than either the within bay control sites ($p<0.001$) or the external control sites ($p=0.01$). The two control locations did not differ ($p=0.17$). The biomass at lease sites was 60% of that at within-bay control sites, and 44% of that at external control sites.

Epiphyte biomass did not vary between locations, although it did vary between sites within a location (Table 2.19). The highest biomass (3.9 mg cm^{-2}) occurred at one of the external control sites, while the lowest was at one of the lease sites (1.8 mg cm^{-2}). The within-bay control sites tended to have low epiphyte biomass, while the external control sites tended to have high epiphyte biomass (Figure 2.22).

Neither %N nor %C varied with location (Table 2.20), although %N did vary significantly with site nested within location. The average N content at both the within and outside bay controls was 0.98%, whereas the two lease sites were 0.9% and 1.27% respectively (Figure 2.23). Carbon content averaged 33.1%, with the lease sites being intermediate between the within-bay controls and outside bay controls (Figure 2.23).

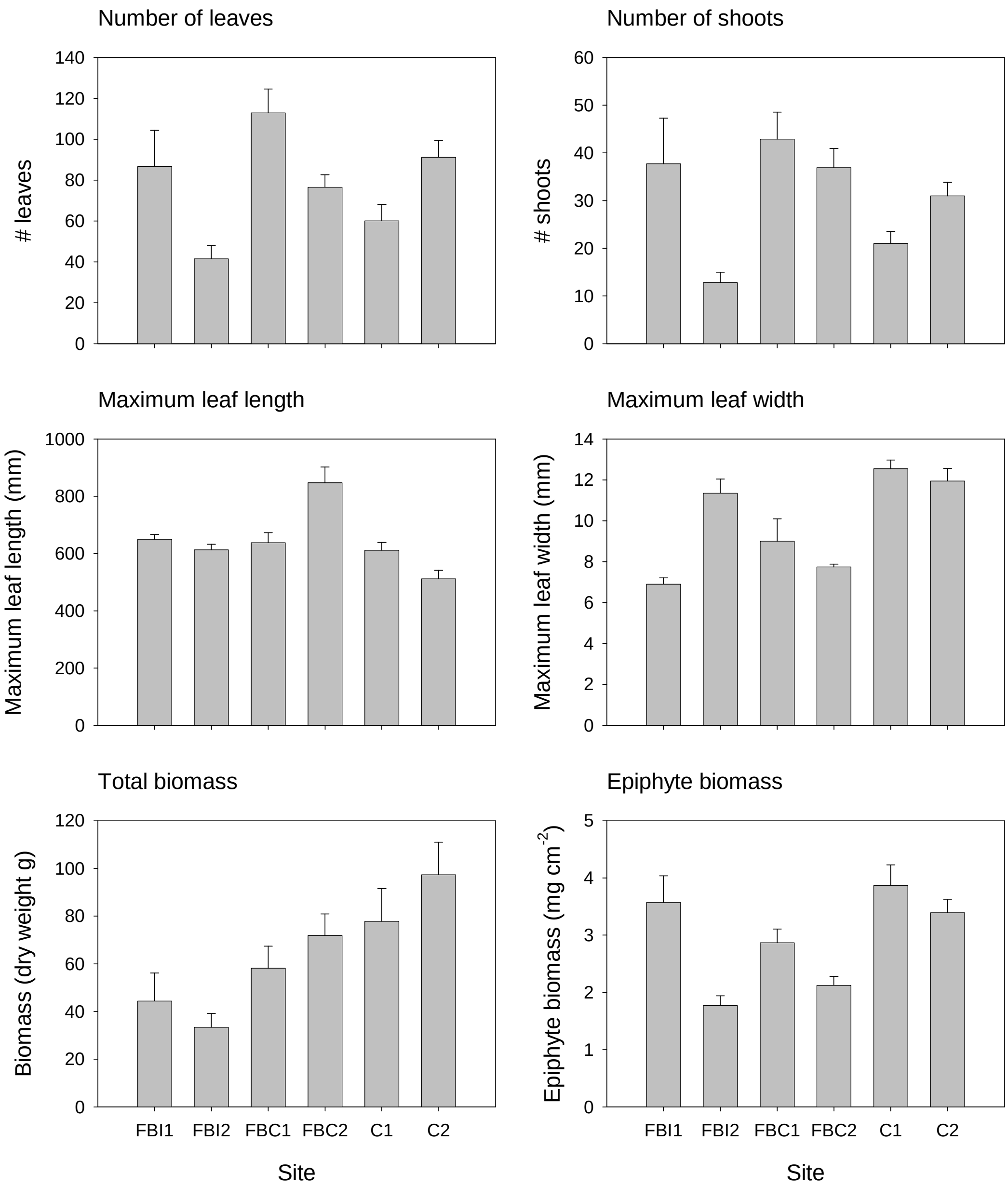

Figure 2.22. Variation in individual seagrass morphology parameters, total above-ground biomass and biomass of epiphytes between sampling sites in and around Fitzgerald Bay. FBI indicates sites adjacent to leases, FBC indicates control sites within Fitzgerald Bay, and C are control sites outside the bay. Error bars are se.

Table 2.19. Univariate ANOVA results for effects of location and site on the morphology and biomass of seagrasses and biomass of epiphytes at Fitzgerald Bay.

Source	SS	df	F	P	SS	df	F	P
	# leaves				**Maximum leaf width**			
Location	9584	2	0.667	0.576	168.96	2	2.33	0.245
Site(Locn)	21600	3	6.56	**0.001**	108.63	3	9.22	**<0.001**
Error	59248	54			212	54		
	# shoots				**Seagrass biomass**			
Location	2723	2	1.08	0.443	109.11	2	14.2	**0.03**
Site(Locn)	3780	3	4.75	**0.005**	11.52	3	0.85	0.47
Error	14324	54			244.35	54		
	Maximum leaf length				**Epiphyte biomass**			
Location	334944	2	1.82	0.304	14.93	2	1.11	0.435
Site(Locn)	276505	3	8.54	**<0.001**	20.12	3	7.96	**<0.001**
Error	582873	54			45.48	54		

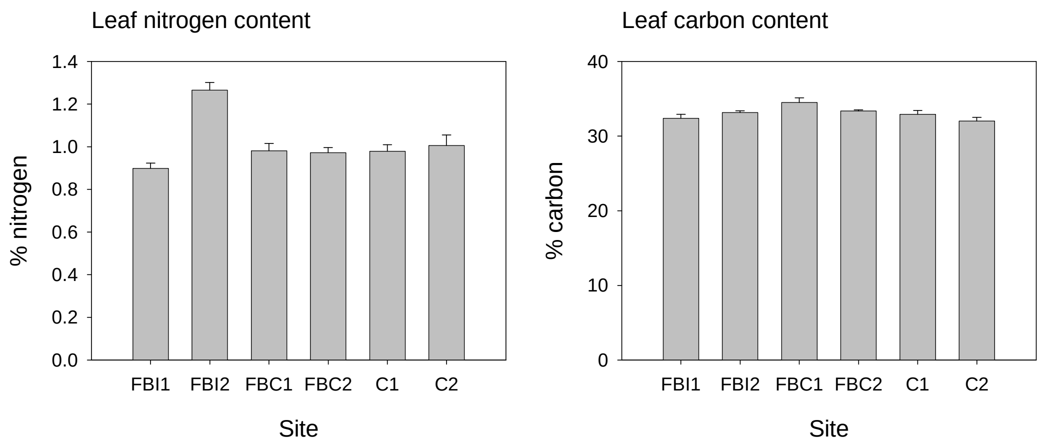

Figure 2.23. Variation in seagrass leaf elemental composition between sampling sites in and around Fitzgerald Bay. FBI indicates sites adjacent to leases, FBC indicates control sites within Fitzgerald Bay, and C are control sites outside the bay. Error bars are se.

Table 2.20: Univariate ANOVA results for effects of location and site on the elemental composition of seagrasses leaves at Fitzgerald Bay.

Source	SS	df	F	P	SS	df	F	P
	% N				**% C**			
Location	0.13	2	0.287	0.769	23.93	2	2.68	0.215
Site(Locn)	0.678	3	19.214	**<0.001**	13.38	3	2.24	0.094
Error	0.635	54			107.42	54		

2.3.6 Wildfish

There did not appear to be any effect of YTK farms on demersal fish, as reported in detail in Appendix 3. Attempts at surveying pelagic fish assemblages in Fitzgerald Bay proved to be unsuccessful. The gill nets only produced four captures; 1 Port Jackson shark, 1 Tommy Rough, and 2 western king prawns. Several other attempts to set gill nets in the area failed due to worse weather conditions than forecast. Similarly, the diver surveys in May 2005 were not successful, with the divers reporting few identifiable fish. This lack of success was in part due to few fish apparently being present, and high turbidity levels making identification difficult. Subsequent attempts to follow up this survey also failed due to poor weather and/or visibility.

2.3.7 Sediments

Organic carbon concentrations in the sediments varied as a function of the interaction between the presence of aquaculture and location within Fitzgerald Bay (Table 2.21, Figure 2.24). While at both locations the carbon concentrations were clearly higher at the farm site than the control site, this difference was accentuated at the northern location, where carbon levels were lower. At the northern location, organic carbon at the farm site was 1.46 times that at the control, versus 1.07 times for the southern location. Nitrogen content also varied with location, and was 14% higher at control sites than at pontoon sites. Although differences in carbon isotope ratios were all highly significant, they were only very small, while there were no difference in the nitrogen isotope ratios.

Table 2.21. Univariate ANOVA results for effects of location and site on the organic carbon levels in the sediments at Fitzgerald Bay.

Source	df	SS	F	P	SS	F	P
		% Carbon			**% Nitrogen**		
Aqua	1	1.56	3.19	0.33	0.001	9.8	**0.009**
Location	1	6.76	13.8	0.17	0.024	192.2	**<0.001**
Locn x Aqua	1	0.49	7.49	**0.018**	0.000025	0.2	0.66
Error	12	0.79			0.002		
		δ 13 Carbon			**δ15 Nitrogen**		
Aqua	1	0.39	27.17	**<0.001**	0.076	0.395	0.54
Location	1	4.10	285.26	**<0.001**	0.016	0.082	0.78
Locn x Aqua	1	0.60	41.78	**<0.001**	0.456	2.38	0.15
Error	12	0.17			2.298		

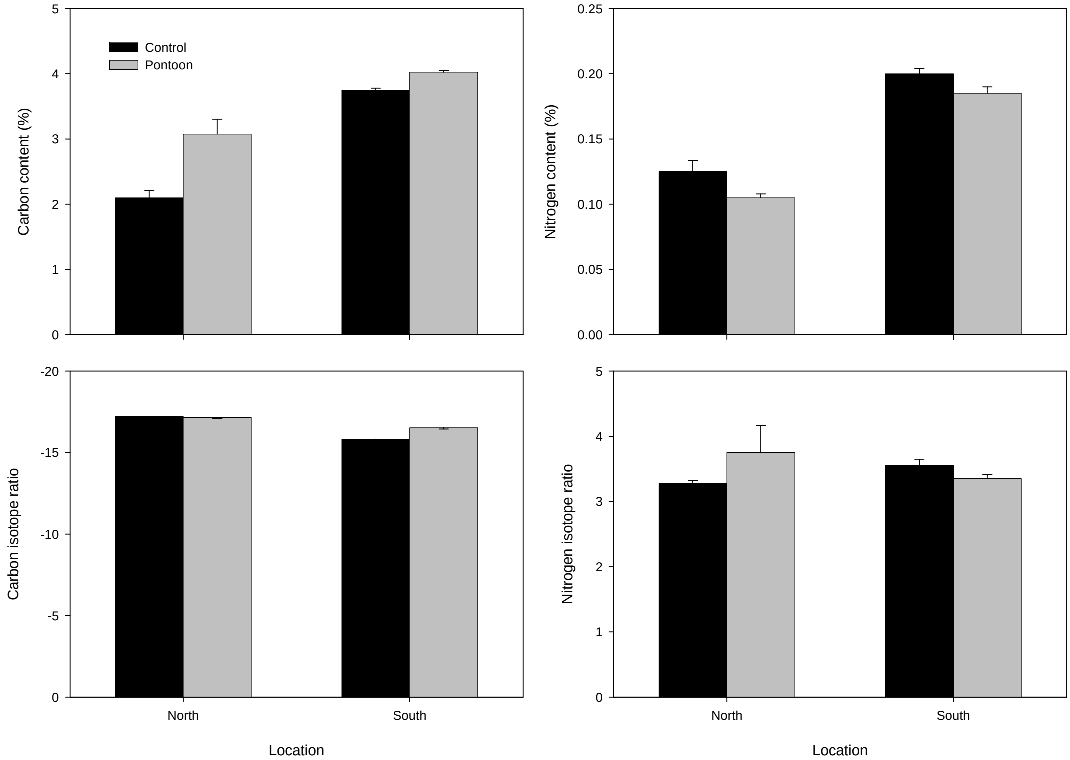

Figure 2.24. Variation in sediment organic carbon and nitrogen composition, and stable isotope ratios, between sampling sites in and around Fitzgerald Bay. Error bars are se.

Similarly, porewater phosphorus concentrations varied as a function of the interaction between the presence of aquaculture and location within the bay (Table 2.22, Figure 2.25). Concentrations were particularly high at the southern farm site, and in both cases the farm sites had higher concentrations than the control sites. These increases were 3077% and 80% respectively. However, for ammonia, there were no statistically significant differences due to any of the factors tested (Table 2.22).

Table 2.22. Univariate ANOVA results for effects of location and site on sediment porewater concentrations Fitzgerald Bay.

Source	SS	df	F	P	SS	df	F	P
	Ammonia				**Phosphorus**			
Aquaculture	0.575	1	5.13	0.27	7.75	1	1.88	0.40
Location	0.079	1	0.71	0.56	0.74	1	0.18	0.75
Aqua x Locn	0.112	1	0.48	0.53	4.13	1	36.3	**0.004**
Error	0.942	4			0.45	4		

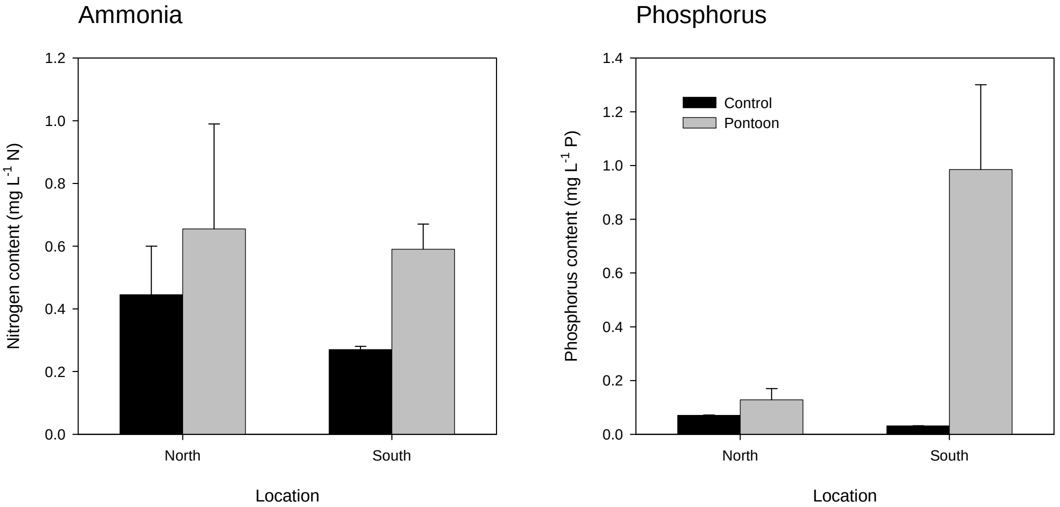

Figure 2.25. Variation in sediment porewater content between sampling sites in and around Fitzgerald Bay. Error bars are se.

2.4 Discussion

The only water quality parameter that varied with proximity to Yellowtail Kingfish cages was ammonia, with concentrations being 81% higher next to cages than at control sites 1 km away from any cages. Ammonia is the main excretory product produced by teleost fish (Forster and Goldstein 1969, Gowen and Bradbury 1987), and as such it is not surprising that concentrations of this nutrient are elevated next to cages. It is likely that this elevation is a persistent feature, although as sampling has only occurred on one occasion, this has not been tested. The samples analysed here were collected in late spring (November 2004), when it is likely that phytoplankton would be at or near their peak in productivity, and thus absorbing a maximal amount of ammonia, thus this pattern is unlikely to be due to seasonally low phytoplankton abundance. This conclusion is supported by the chlorophyll levels measured, which were elevated in November 2004 compared to August 2005. The spatial scale of this increase in ammonia concentration is also not known. However, as ammonia is a highly biologically available form of nitrogen, it is likely to be taken up quickly by phytoplankton, macroalgae or seagrasses. Thus it is unlikely that increased levels of ammonia will be found very far from cages. However, there was no evidence of an increase in phytoplankton abundance, chlorophyll levels, seagrass epiphyte biomass, or seagrass nutrient composition in close proximity to cages, suggesting that while this ammonia may be rapidly utilised, it is also rapidly dispersed and is not retained in plant biomass close to the cages.

The lack of response of most water quality parameters to the presence of yellowtail kingfish pens parallels the results of a number of other studies. For example, a number of studies have looked for effects of tuna farming off Port Lincoln on water quality, and failed to detect them (Clarke et al. 1999, 2000). It is generally considered that this lack of an effect is due to high uptake rates by phytoplankton, and high water movement, which has also been implicated in a lack of detectable effects elsewhere (Doglioli et al. 2004). Due to its geographic location, in northern Spencer Gulf and close to shore, water movement is likely to be less in Fitzgerald Bay than in the Port Lincoln tuna farming zone, and thus nutrient dispersal is also likely to be lower, accounting for the fact that elevated levels of ammonia were detected. Elevated nutrient levels have been detected in several other studies of aquaculture sites located in semi-enclosed areas (e.g. Frid and Mercer 1980, Gowen et al. 1988, Pita et al. 1999). In particular, ammonia tends to be elevated (Merceron et al. 2002), as was found here. Even in these situations, however, aquaculture does not appear to lead to increases in phytoplankton abundance (Gowen and Bradbury 1987, Merceron et al. 2002). In contrast, Modica et al. (2006) found elevated levels of chlorophyll at cage sites compared to upstream control sites, with levels increasing for at least 1 km downstream of cages. Yap et al. (2004) also found increased nutrient levels and phytoplankton abundance inside fish pens in the Phillipines, and attributed this to the very low flushing rates of the pens studied. While the above studies have all been undertaken at small spatial scales, comparing sites adjacent to an individual pen or farm to a reference site only a few kilometres away, Pitta et al (2005) studied the effects of aquaculture zones on nutrient enrichment and phytoplankton. This study in the eastern Mediterranean showed that dissolved inorganic nitrogen increased close to the zones in September, when feeding rates were highest, but not at other times of the year. This increase was then translated into an increase in chlorophyll values, although effects on phytoplankton composition and abundance were sporadic and difficult to interpret. In this study, sites 2-3 nautical miles from 3 farming zones (producing 800, 1500 and 4000 tonnes of fish per annum) were contrasted with sites ~ 20 nm distant, suggesting effects on water quality may be occurring at regional scales rather than local scales. While we did not examine in situ water quality at regional scales, we did use remote sensing to look for these larger scale effects, and failed to find any indication of increased chlorophyll levels at these scales, although the shallow depths in Fitzgerald Bay do make this approach problematic.

While there is a statistically significant effect of distance from cages on the infaunal composition, the responses of individual taxa are not always consistent with the hypothesis that this is due to organic enrichment around cages. The Capitellidae are well known to be indicators of organic enrichment (Grassle and Grassle 1974, Pearson and Rosenberg 1978, Karakassis et al. 2002, Yokoyama 2002, Brooks et al. 2003), and while they did display increased abundance in the immediate vicinity of cages (0-20 m), at the southern site they also showed high abundance at the control site. Even at the 0 m sites, capitellid abundance was below 1.5 per core (maximum mean abundance = 365 m^{-2}), which is well below typical abundances of 10's – 100's of thousands m^{-2} seen in areas with high levels of organic enrichment (e.g. Brooks et al. 2003, Pereira et al. 2004). Cirratulidae are also known indicators of organic enrichment (Glasby 2000), and their pattern of abundance at the north site suggests organic enrichment, although that at the south site does not. While mean abundances reach higher peaks than do the Capitellidae, again the peak abundance of 1378 m^{-2} is relatively low by worldwide standards. Spionidae have been recorded in densities up to 5670 m^{-2} in organically enriched areas (Yokoyama 2002), but in this study did not show any clear trends with distance, and only reached maximum abundances of 325 m^{-2}.

The low densities of taxa generally considered to be good indicators of organic enrichment suggest that while there is some level of organic enrichment in these sediments, it is very low. This conclusion is supported by the lack of effects on infaunal taxon richness and total abundance, with the exception of a 3-fold decline in total abundance at the north lease. In fact, total abundances are very low in comparison to what has been found in the tuna farming zone off Port Lincoln using exactly the same methods. For example, in the 2004 Tuna Environmental Monitoring Program, mean abundances per core at the Rabbit Island control sites ranged between 9.9 and 35.6, with an overall mean of 19.9 (Loo and Drabsch 2005). This compares to means between 2 and 6.4 at the 1000m sites in Fitzgerald Bay, with an overall mean of 3.7. In organically enriched areas, there is typical a low richness, but high abundance, with a few opportunistic taxa dominating the assemblage (Pearson and Rosenberg 1978, Pereira et al. 2004). Previous studies in other areas have shown that impacted infaunal assemblages tend to quickly recover from organic enrichment once an area is fallowed (e.g. McGhie et al. 2000, Brooks et al. 2003), although very high levels of organic enrichment can take some years to recover (Pereira et al. 2004). With the low levels of impact shown here, it is expected that the assemblages will fully recover in a period of months after the commencement of fallowing.

Because of the sensitivity of a range of taxa to organic enrichment, infauna are used to monitor environmental impacts in soft sediments in many areas of the world (eg Tasmania: Macleod et al. 2004, Edgar et al. 2005; Hawaii: Lee et al. 2006; Canada: Brooks et al. 2003; Scotland: Nickell et al. 2003; Mediterranean: Karakassis and Hatziyanni 2000; Norway: Carroll et al. 2003). In South Australia, monitoring of infauna on a lease-by-lease basis is a mandatory requirement for all finfish aquaculture. This monitoring has been occurring annually for the tuna industry since 2001, and has since commenced for other finfish aquaculture sectors. While infauna provide a sensitive method to monitor the environmental impacts of marine aquaculture activities, they have the disadvantage of being costly and time consuming to enumerate (Wildish et al. 2001, Carroll et al. 2003). The traditional approach to monitoring involves collection of sediment cores and manual sorting and identification, as done here. Each core can take up to half a day to process, and the identification stage requires significant expertise in infaunal taxonomy. As an alternative, the Aquafin CRC and FRDC have been funding a project to examine the use of DNA probes to assess the faunal composition of sediment cores, removing the need for manual sorting and identification (2001/102 Development of novel methodologies for cost effective assessment of the environmental impact of aquaculture, or PCR project for short). This project has proven successful, and the technique developed will be applied for the 2005 monitoring of the environmental impacts of tuna aquaculture.

The PCR project relies on using real-time PCR to identify the presence and abundance (in terms of DNA presence) of a pre-defined array of taxa. It is necessary to know beforehand the appropriate taxa to use, as specific DNA probes for them have to be developed and tested to ensure that they effectively quantify the presence of the target taxa with no cross-contamination from other organisms. For the tuna industry off Port Lincoln, probes have been developed and tested for Capitellidae, Lumbrineridae, Cirratulidae, Nephtyidae and Spionidae. This list includes 3 of the 4 most abundant infaunal taxa found at Fitzgerald Bay. However, only 4 Nephtyidae and 13 Lumbrineridae were found in the 146 samples from Fitzgerald Bay, suggesting that they are not ideal for discriminating patterns in infaunal assemblages at this site. It is thus likely that probes will need to be developed for additional taxa to effectively transfer this methodology to Fitzgerald Bay, although more data would be needed to have confidence in which taxa should be targeted, as the patterns seen in this sampling event may not be consistent over time. The extension of this technique to other areas in Spencer Gulf, including Fitzgerald Bay, is being addressed in a new project (FRDC 2006/078 "Aquafin CRC: Development of rapid environmental assessment and monitoring techniques for application to finfish aquaculture in South Australia").

A major problem with both manual identification and PCR techniques is the large number of taxa potentially present in any set of samples. Identifying infaunal taxa to species level is difficult, time-consuming and costly, as would be developing DNA probes at the species level. Instead, the approach taken here has been to identify infauna to the family level only. This is a common approach in environmental impact studies, and has shown to provide the best balance between cost and precision in several studies (Warwick 1993, James et al. 1995). In particular, Karakassis and Hatziyanni (2000) have shown identification at the family level to be appropriate for detecting the impacts of finfish farming in the Mediterranean. This approach is feasible because species in the same family tend to respond similarly to organic enrichment, and indeed it can even eliminate small-scale variability due to other influences. Conducting analyses at the family level should be considered cautiously if the data are to be used for more than just detecting an impact of aquaculture however, as this taxonomic lumping can obscure other patterns (e.g. Maurer 2000, Anderson et al. 2005).

As an alternative to monitoring the biological impacts of aquaculture on the benthos, epifaunal assemblages can also be studied. Typically, this is done using video footage, or alternatively direct diver surveys. This technique relies on the presence of a relatively abundant and diverse epifaunal assemblage that can readily be surveyed using video, or else is more suited to detecting major impacts. In Tasmania, for example, remote video surveys have been successfully used to monitor impacts of salmon farming (Crawford et al. 2001), although they are less sensitive than infaunal sampling (Edgar et al. 2005). Off Port Lincoln, the tuna environmental monitoring program originally included both infaunal sampling and remote video surveys, however, due to the sparse nature of the epifaunal assemblage, the video surveys were considered to be uninformative, and have now been discontinued (M. Loo, SARDI Aquatic Sciences, pers. com.).

In the current study, the results for both infauna and epifauna were equivocal. While there was an apparent effect of distance from cages on both groups, the patterns found did not clearly indicate an effect of organic enrichment. In the case of infauna, it is possible that there are patterns in the level of organic enrichment of the sediments that are unrelated to aquaculture, and that these could have confounded or ability to detect clear gradient effects away from cages. Similar patterns have been found in the tuna farming zone off Port Lincoln, where there are distinct regions of sediments with naturally high and low organic loadings (Fernandes et al. 2006). To determine if this is the case in Fitzgerald Bay, a grid-based sampling effort would be required to map sediment characteristics. In the case of epifauna, there appears to be significant small-scale spatial variability that is not associated with aquaculture. However, it also appears that proximity to aquaculture sites increases this variability, a pattern that is fairly common in assemblages exposed to low-moderate levels of disturbance (see Dernie et al. 2003, Wear and Tanner 2007). This higher level of variability makes it more difficult to detect temporal trends (Thrush et al. 2001), and differences in mean abundance/composition between sites, however.

A third approach to assessing benthic impacts is to examine physico-chemical characteristics of the sediment, rather than biological effects. Possible options include organic matter content, grain size, porewater nutrients, and stable isotope ratios. These parameters have the advantage that they are often easier, quicker and cheaper to measure than biological variables. However, there are also several disadvantages. Firstly, without a detailed knowledge of the benthic systems around the aquaculture lease, it can be difficult to determine what level of change in the physico-chemical variables is biologically meaningful. Secondly, some of these variables may show high levels of temporal variability, whereas the biological variables often measured can integrate effects over a time period of several months. In the current study, we found clear differences in organic carbon and nitrogen content, and porewater ammonia and phosphorus, with samples collected adjacent to cages having elevated levels of all three compared to those collected at control sites at least 1 km away from any cage. Stable isotope ratios, however, did not show any clear patterns. In contrast, previous studies have found sediment isotope ratios to be useful indicators of environmental impacts of aquaculture (Yamada et al. 2003), although in some cases only nitrogen was affected (Sarà et al. 2004). Sediment organic carbon loadings have also been used successfully at other locations to detect benthic impacts of aquaculture (Carroll et al. 2003), although they were not useful in a study of fallowing of salmon farming sites in Tasmania (Macleod et al. 2004). In the study by Carroll et al. (2003), however, the infaunal assemblages at some sites indicated a severe disturbance at a site 50 m from the edge of a cage, while sediment chemistry indicated that the site was normal. Thus, at least in this study, sediment chemistry is a much less sensitive indicator of disturbance than is infaunal composition. Other physico-chemical parameters that have been shown to at least sometimes respond to the presence of aquaculture are sediment redox and sulphide levels (Wildish et al. 2001, Brooks et al. 2003, Macleod et al. 2004, Edgar et al. 2005). Again, however, the study by Brooks et al. (2003) showed recovery of a suite of sediment physico-chemical parameters almost as soon as harvesting was completed, whereas infauna took a further 6 months to recover.

With the exception of seagrass biomass, we failed to detect any effects of aquaculture on either seagrasses or fish. Interestingly, while seagrass biomass was lower at sites close to cages, all other seagrass parameters, such as leaf density, leaf length and leaf width, did not change with location, although they did with site. Epiphyte biomass, and leaf composition showed similar patterns. Previous studies across a number of locations have shown that finfish aquaculture generally causes a decrease in seagrass shoot density, or even complete dieback (Verneau et al. 1995, Delgado et al. 1997, Delgado et al. 1999, Pergent et al. 1999, Cancemi et al. 2000, Dimech et al. 2000a, Ruiz et al. 2001). Both primary productivity (Delgado et al. 1999, Cancemi et al. 2000) and standing biomass (Delgado et al. 1999, Dimech et al. 2000a) have also been reported to decrease with proximity to finfish aquaculture farms. Most studies that have investigated epiphyte cover or biomass, have found increased epiphytic loading on seagrass leaves closer to the finfish farms (Delgado et al. 1997, Delgado et al. 1999, Pergent et al. 1999, Cancemi et al. 2000, Dimech et al. 2000a). However, Ruiz et al. (2001) found that epiphytic cover did not decrease in close proximity to farms, and Bryars (2003) reported that seagrasses in the vicinity of two salmon cages in South Australia appeared ‘healthy’ in respect to epiphyte load, although no quantitative data were collected.

It is unclear whether the decrease in seagrass biomass close to cages is related to their presence, or if it is due to other factors. Unfortunately, no data is available from prior to the commencement of aquaculture in Fitzgerald Bay, and we have not been able to conduct sufficient sampling in this study to look at trends over time to determine if biomass is remaining constant or decreasing. Any effects on seagrasses are likely to be via an increase in nutrient availability. In a parallel study, we (Tanner et al. 2006) have shown that patterns of carbon deposition around pens in Fitzgerald Bay show highest rates north and south of pens, with deposition dropping off very rapidly in an east-west direction. Given that seagrasses were collected several hundred metres west of the lease sites, it is very unlikely then that they would experience decreased rates of sedimentation. This location also means that they are not affected by shading from the pens, or other physical impacts. However, if nutrients are causing a problem, it would almost certainly be manifested as an increase in epiphyte biomass. As discussed above, most studies that have examined epiphytes around finfish cages have found an increase, and this would be due to increased nutrient availability. An increase in epiphyte loads as a result of eutrophication is also the main mechanism that has been implicated in most instances of seagrass decline around the world (e.g. Short et al. 1991, Short and Wylie-Echeverria 1996, Dennison et al. 1993, McGlathery 2001). This link between epiphytes and seagrass loss in eutrophic areas, and the lack of an observed effect of proximity to aquaculture on epiphyte loads in Fitzgerald Bay, suggests that maybe the decreased seagrass biomass close to cages is due to some other factor, and may have been present prior to the commencement of farming.

Amphibolis seagrass has also been reported in Fitzgerald Bay previously (Hone et al. 1996). Whether or not this species was present in 2005 was not determined, as conditions when the seagrass surveys were being conducted precluded broad-scale surveys to determine distribution. However, *Amphibolis* is considered to be more sensitive to nutrient enrichment than *Posidonia* (Shepherd *et al.* 1989, Ralph *et al.* 2006), so would potentially be a better indicator of environmental effects related to nutrient enrichment.

Wildfish assemblages are frequently attracted to finfish cages, either for the food supply that they represent, or for the habitat structure that they form (e.g. Dempster et al. 2002, 2005, Machias et al. 2004, Vita et al. 2004). Elevated abundances of wildfish (and crustaceans) have also been found around tuna cages off

Port Lincoln (Svane, unpublished data). It is therefore surprising that no effects were found on wildfish in the present study. However, we only managed to survey the demersal fish assemblages successfully, with all attempts at surveying pelagic assemblages being unsuccessful, despite the multiple methods tried. This lack of success was due in large part to the poor visibility and exposed nature of Fitzgerald Bay, which ruled out visual survey techniques and made it logistically difficult to sample. There have been a number of previous reports of aggregations of wildfish around the Fitzgerald Bay lease sites, however (S. Stone, pers. com.), although no quantitative surveys have been undertaken. It is thus likely that while aggregation does occur, it is a temporally variable phenomenon, and would require an extensive sampling protocol to properly document.

2.5 Conclusions

Overall, there were limited environmental impacts of yellowtail kingfish aquaculture detected in this study. Those impacts that were detected only occurred on relatively small spatial scales, generally within the lease, and do not appear to be substantially altering ecosystem functioning. Those impacts that were detected are:

- Increased ammonia concentrations adjacent to cages (by 81% compared to control sites), but no changes in other water column nutrients.
- A complex effect on infaunal assemblages, which varied with distance from cage, but not in a manner consistent with organic enrichment
- Variable effects on epifauna, which again are difficult to interpret with respect to impacts of organic enrichment, due to variability between sites.
- A decrease in seagrass biomass, which was only 60% of that at within bay controls adjacent to cages, and 44% of outside bay controls. However, this was not associated with significant changes in seagrass morphology or epiphyte biomass, making it difficult to determine if it was due to aquaculture or was a pre-existing pattern.
- There were slight increases in sediment organic carbon (7-46%) and total nitrogen (14%) content adjacent to cages, which varied as a function of location in the bay.
- There were substantial, although highly variable, increases in porewater phosphorus concentrations adjacent to cages (80-3077%).

The only effects that can be unequivocally related to the presence of aquaculture are the increases in water-column and sediment nutrient (ammonia, phosphorus, organic carbon and total nitrogen) concentrations. There are some suggestions of possible associated changes in faunal and seagrass assemblages (but not phytoplankton or wildfish), although the patterns detected do not clearly correspond to patterns expected in response to increased nutrients.

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Chapter 3: Potential impacts of finfish aquaculture in the south-east of South Australia through modifications to the light environment experienced by seagrasses.

Greg Collings, Keith Rowling and Jason Tanner

3.1 Introduction

Over the past decades, there has been small-scale farming of salmonid fishes in marine net-pens off the south-east coast of South Australia, particularly at Cape Jaffa and Beachport. This farming has typically occurred in more sheltered locations, where the benthos is dominated by seagrass beds. As a number of studies elsewhere in the world have established that this form of aquaculture can have a substantial negative impact on seagrasses (Cancemi et al. 2000, Dimech et al. 2000, Ruiz et al. 2001), the aquaculture system used consists of pens attached to a single-point mooring. This system means that cages are not permanently located over a fixed area, but rather that they move with the currents and wind. As a result, the seagrasses underneath them are only exposed to a direct impact for short periods of time, when cages happened to be directly above them. As some interest has been expressed by industry in expanding aquaculture operations within the area, there is a need to understand how this might impact on seagrasses. These impacts could occur through the deposition of sediments on the seagrasses, increases in water column nutrient concentrations, and thus epiphytic growth, or through direct shading by the cage itself. This later is perhaps the greatest concern while the industry is still small, and thus this chapter looks at the potential for seagrasses off Cape Jaffa and Kingston to withstand a decrease in light, as would be caused by placing finfish cages over them.

Light is scattered and absorbed by both particulate and dissolved material in the water column. As a result, the amount of photosynthetically active radiation (PAR) reaching the seafloor (where it can drive the photosynthesis of seagrasses) decreases with increasing depth (Figure 3.1). How rapidly light (i.e. how much per linear metre of water column) is absorbed is dependent upon the type and amount of material in the water column. This will determine the “steepness” of the curve shown in Figure 3.1. A high attenuation coefficient implies low optical water quality, and a lower fraction of light passing through each metre of water. Thus the amount of light reaching the seafloor is a product of both depth and the attenuation coefficient. High attenuation coefficients dictate that less light will reach the seafloor and the critical depth (i.e. where light becomes limiting) will be reduced. Thus, a decrease in clarity may result in the loss of seagrass because of lost productivity.

Nett productivity represents the balance between the light driven process of photosynthesis, which provides for carbon assimilation, and the respiratory processes which use this stored energy. An inability to supply what is required by the respiratory processes over an extended period would make a seagrass bed unsustainable. Furthermore, this takes into account only the respiratory losses and ignores those of breakage, herbivory, sloughing of material etc.

Seagrasses, with their large underground biomass, have an ability to withstand periods of low light by using stored carbohydrates to make up the deficit between metabolic demands and the insufficient supply of photosynthetically derived carbon. It is generally accepted that the deep margin of seagrass beds is determined by light levels, whereby any further depth leads to a decrease in light, resulting in inadequate light to support long term survival of seagrass (Duarte 1991; Fourqurean and Zieman 1991).

Thus an important comparison can be made with the seagrasses of the Adelaide region, applying the light levels measured in Adelaide over the same period of time (exactly one year earlier). A nett productivity that is lower than that of the deep (light limited) edge of the Adelaide seagrass beds would be cause for concern.

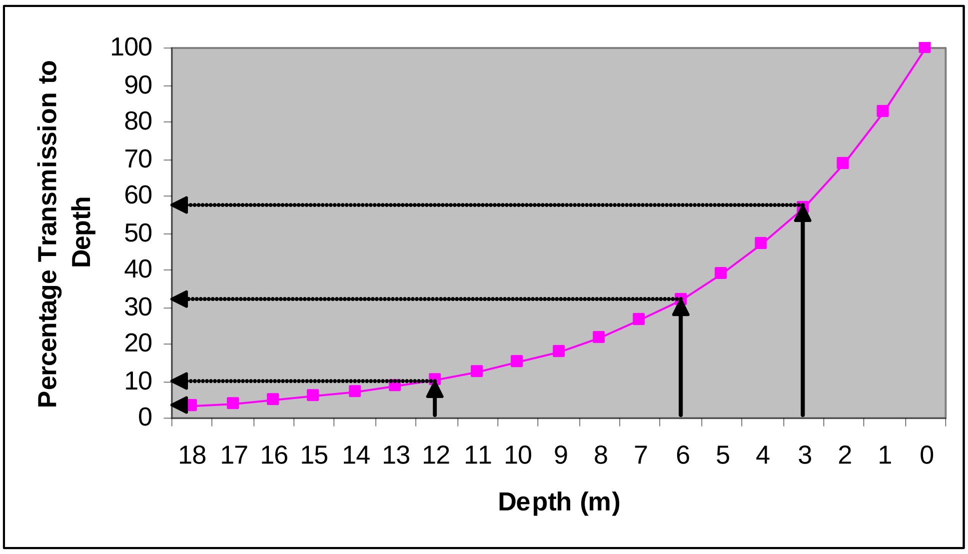

Figure 3.1. The relationship between depth and percentage transmission of irradiance to the seafloor if constant attenuation coefficient is assumed. Under these conditions it is only the depth of the water that affects transmission.

3.2 Methods

3.2.1 Field methods

To investigate the light regimes at potential finfish farming regions in the Lacepede Bay region, a series of Odyssey photosynthetically active radiation (PAR) (400-700 nm) cosine type light meters (Dataflow Systems, New Zealand) were deployed off Cape Jaffa and Kingston SE at a range of depths on March 29th 2006 (Figure 3.2), recording average light intensity for each half hour period. At each site a pair of these meters were attached to the top of a star picket with cable ties such that they sat 60 cm above the sand substratum to approximate the canopy height of an *Amphibolis* / *Posidonia* meadow. Meters were located at distances of approximately

1.5, 3 and 4.5 km from the shore in depths of 5.5, 9.5 and 11.5 m. These are referred to as the inshore, mid and deep sites. The inshore sites were both in *Amphibolis* beds. The mid sites both were within *Posidonia* beds and the deep site was mixed *Amphibolis* / *Posidonia* at Kingston SE, and *Posidonia* at Cape Jaffa. All meters were retrieved on 10th May 2006 and none were affected by algal growth at that time.

In order to investigate light attenuation on a single day (10th May 2006) at each of the sites a series of profiles were taken using a pair of Licor light meters fixed to a frame so that one of the sensor heads was positioned 1 m above the other. This frame was then lowered to the substrate and simultaneous light readings were taken every 2 m until the top sensor was positioned 5 cm below the water surface.

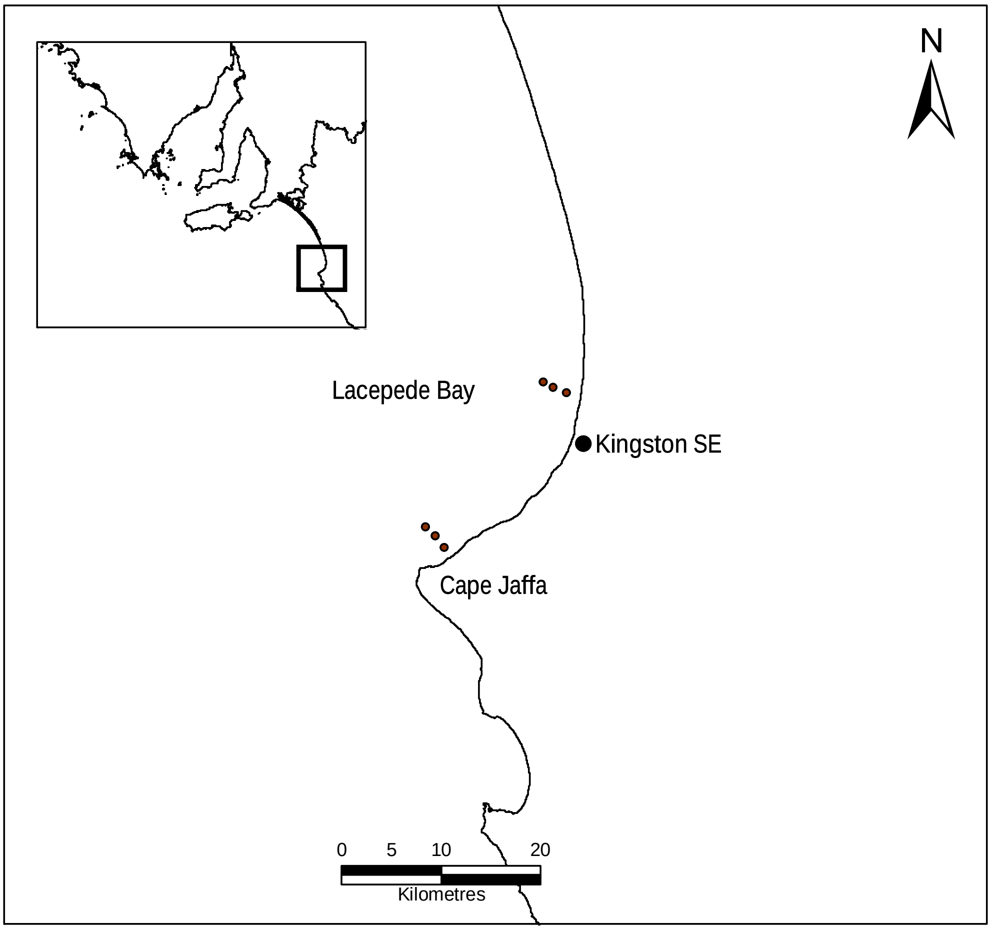

Figure 3.2. Lacepede Bay showing light transects at Cape Jaffa and Kingston SE (see text for details of transects)

3.2.2 Calculation of attenuation coefficient

Light at any given depth can be modelled using the equation:

$$I_{D2} = I_{D1} \times e^{-K.D} \qquad \text{(equation 3.1)}$$

Where I_{D1}=Light at a given depth, I_{D2} = Calculated light intensity at a greater depth, and D is the difference (in metres) between the two depths.

K is the linear attenuation coefficient (LAC).

Rearranging this equation allows us to calculate the attenuation coefficient K.

$$K = \frac{\ln(\frac{I_{D2}}{I_{D1}})}{-D} \qquad \text{(equation 3.2)}$$

As the difference in depth is always 1 metre in this study,

$$K = -\ln(\frac{I_{D2}}{I_{D1}}) \qquad \text{(equation 3.3)}$$

3.2.3 Productivity modelling

The productivity of all photosynthetic organisms are dependent upon the level of light to which they are subjected. However, this relationship is complex, demonstrating a linear response at low irradiance levels, but tapering off to an asymptote at higher levels (Figure 3.3). Furthermore, this response is not consistent between species (e.g. Masini et al. 1995a; Masini and Manning 1997 Cummings and Zimmerman 2003), nor within species where differences may occur according to environment; in particular light (e.g. Mazella and Alberte 1986; Masini et al. 1995a; Masini and Manning 1997) and temperature (e.g. Perez and Romero 1992; Touchette and Burkholder 2000).

Several mathematical models have been fitted to describe the relationship between productivity and ambient light conditions (Fourqurean and Zieman 1991, Falkowski and Raven 1997). Each represents a line of best fit to describe productivity based on the ambient light and on certain parameters of the plant which describe the maximal productivity (P_{max}), efficiency at sub-saturating light intensities (I_K), and dark respiration (Rd).

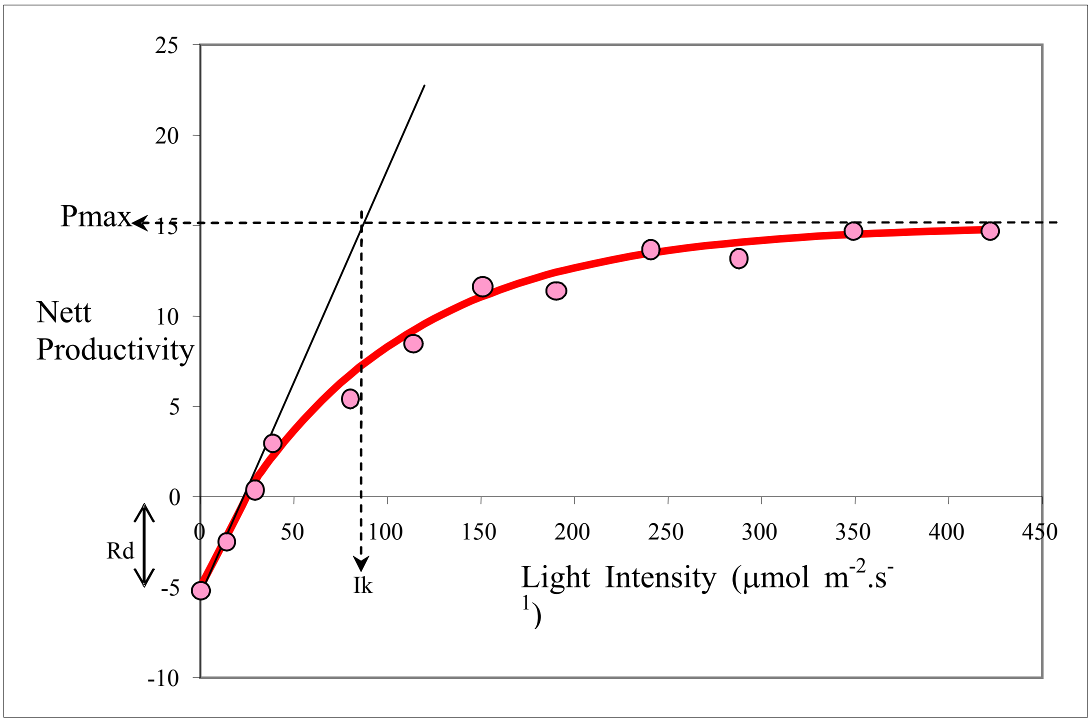

Figure 3.3. A typical Photosynthesis – Irradiance (PI) curve showing nett (red line) productivity as a function of light intensity. Dark respiration (Rd) represents the loss of carbon to cellular metabolism. $P_{max(nett)}$ indicates the maximal productivity achievable. I_K is a measure of the photosynthetic efficiency at sub-saturating light and is determined by extending a line as a tangent to the initial slope of the curve until it intercepts the $P_{max(nett)}$ and dropping a line to the x axis. A small I_K value is indicative of a plant which is making efficient use of available light. Note that $P_{max(nett)}$ is equal to the gross maximal productivity (see model below) minus the dark respiration.

For the purposes of our study, Chalker's (1981) model of primary productivity was utilised:

$$P_I = P_{\max} \times (1 - e^{(\frac{-I}{I_k})}) - Rd \qquad \text{(equation 3.4)}$$

where P_I represents the nett productivity at a given level of illuminance, I;
P_{max} represents the gross maximum productivity;
I_K represents the subsaturating light level and,
Rd is the dark respiration rate

Chalker's (1981) model has been shown in many studies to provide a good fit to measured P-I data (e.g. Cheshire et al. 1995, 1997).

For the purposes of this model, nett productivity is calculated for each half hour period using a constant irradiance level equal to that recorded in the field based on Chalker's (1981) equation above. The photokinetic parameters, P_{max}, Rd and I_k which define the productivity – irradiance relationship shown in Figure 3.1 are, in the absence of data from South Australia, defined on the basis of the parameters provided by Masini and Manning (1997) for *Posidonia sinuosa* off the coast of Perth:

P_{max} = 1.05 g C gdwt^{-1}
I_k = 44 µmol photons m^{-2} s^{-1}
Rd = 0.47 g C gdwt^{-1}

These data were measured at a water temperature of 18ºC, which is similar to the area that is the focus of this study. These parameters were maintained across all sites. Productivity across the period of the study (30/3/2006 – 3/5/2006) was calculated by summing productivity across all calculated half hour periods. The entire model was created and run under a Microsoft Excel 2000 spreadsheet environment.

3.3 Results

Daily irradiance, in terms of photosynthetically active radiation (PAR) was highly variable from day to day (Figure 3.4), with irradiance levels reaching approximately 7.5 mol photons m^{-2} day^{-1} at the Kingston SE site and 6.1 mol photons m^{-2} day^{-1} at the Cape Jaffa site. The period from day 20 to 23 was a time of very low light intensity at all sites.

It is evident that quite high attenuation values occur at both Cape Jaffa and Kingston SE (Figure 3.5). Furthermore, attenuation varies considerably through the extent of the water column. There was no consistent relationship between attenuation coefficient and depth. At some sites the attenuation coefficient was greatest at shallow depths, at others it was at deeper depths, and others varied in an apparently random fashion.

Daily PAR is slightly higher at Kingston SE than Cape Jaffa. Inshore sites at both locations receive more PAR than those further offshore (Figure 3.6). Importantly, only the inshore sites at each of the southeast locations receive more PAR than the 18 m site at Adelaide. Variability (as represented by the magnitude of the error bars) is larger at the southeast sites than at an Adelaide site of equivalent mean irradiance.

Productivity at all sites examined resulted in nett negative productivity across the course of the period of study, i.e. respiration was not balanced by photosynthesis across this period (Figure 3.7). With the exception of the two inshore sites, nett productivity was more negative for all sites in the southeast than it was for the 18 m site at Adelaide. All southeast sites demonstrated lower productivity than the 12 m site at Adelaide. It is worth noting that using the parameters provided by Masini and Manning (1997), nett productivity across the course of the year at all Adelaide sites was negative. This result suggests that the parameters derived for seagrasses in Western Australia are not directly transferable to South Australia, and that local seagrasses may be adapted to survive at lower light intensities.

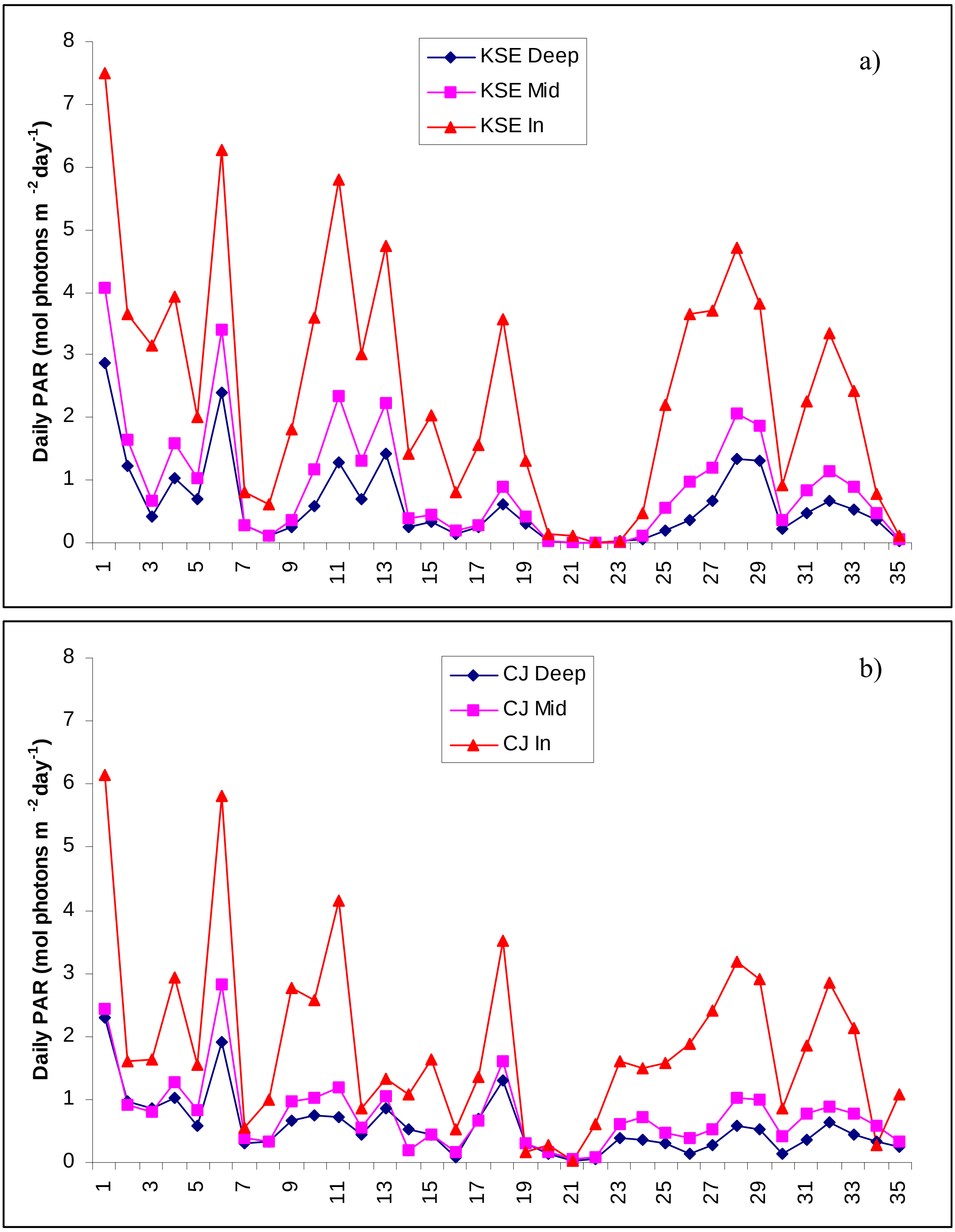

Figure 3.4. Daily irradiance in April/May 2006 at three depths at a) Kingston SE (KSE) and b) Cape Jaffa (CJ).

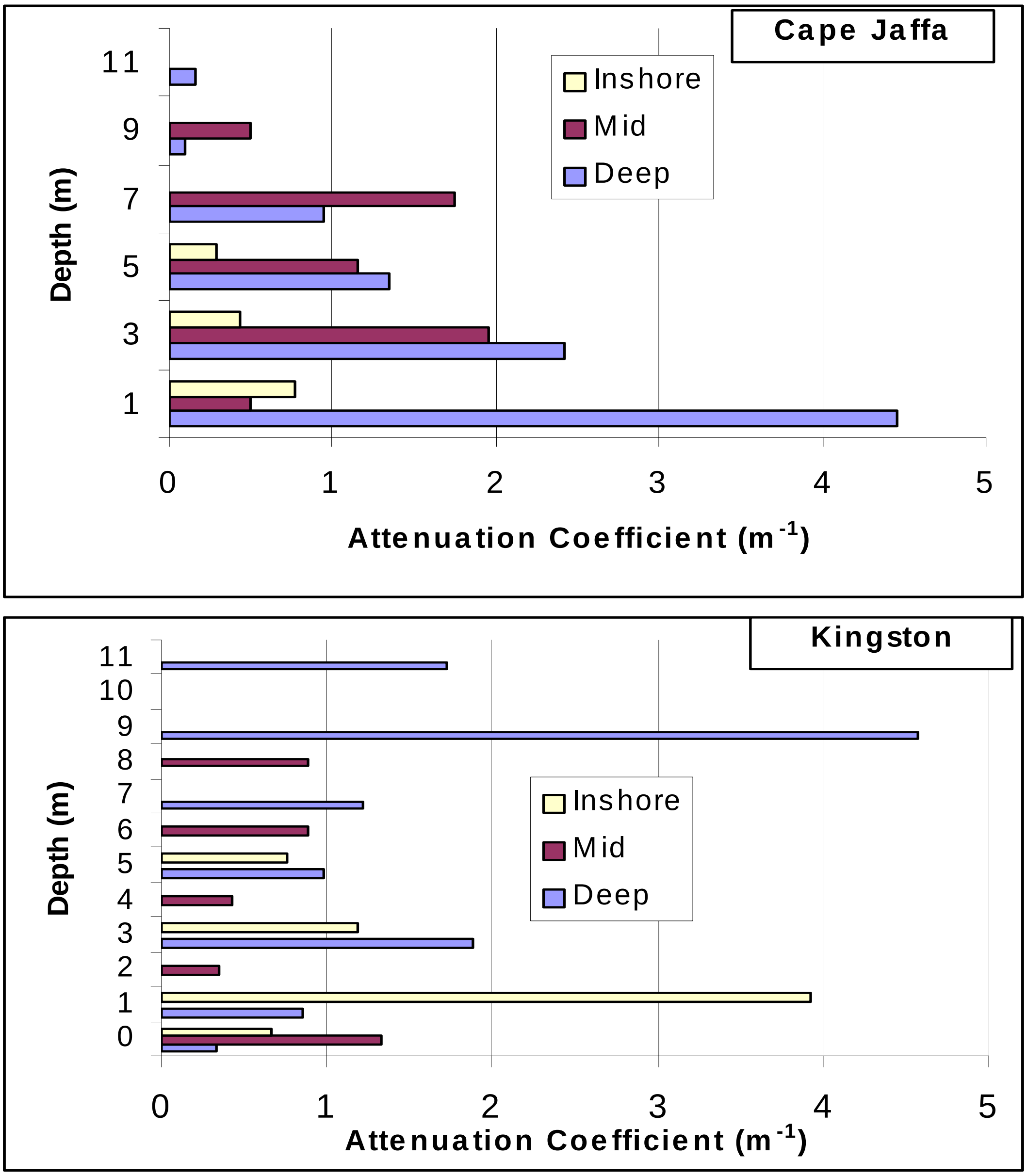

Figure 3.5. Attenuation coefficients recorded at different points in the water column at Cape Jaffa and Kingston SE on May 10, 2006. Where no bar is evident at any given depth, no recording of attenuation was made at that depth.

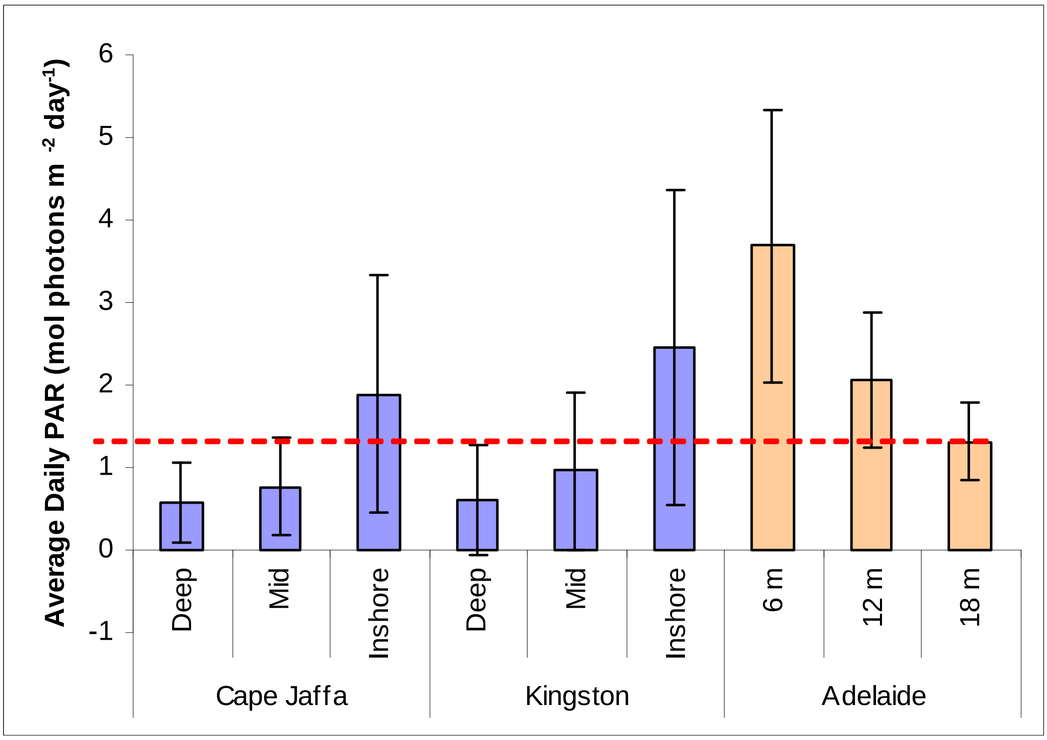

Figure 3.6. Average daily PAR at three depths at Cape Jaffa and Kingston SE from 30/3/2006 to 3/5/2006. Also included are the averages and standard deviations recorded off the Adelaide coast at 6, 12 and 18 m depth. Data for the Adelaide readings was recorded across the period 30/3/2005 to 3/5/2005. Error bars represent standard deviation. The red line represents the level of light that is considered to be the minimum capable of sustaining seagrass beds off the coast of Adelaide.

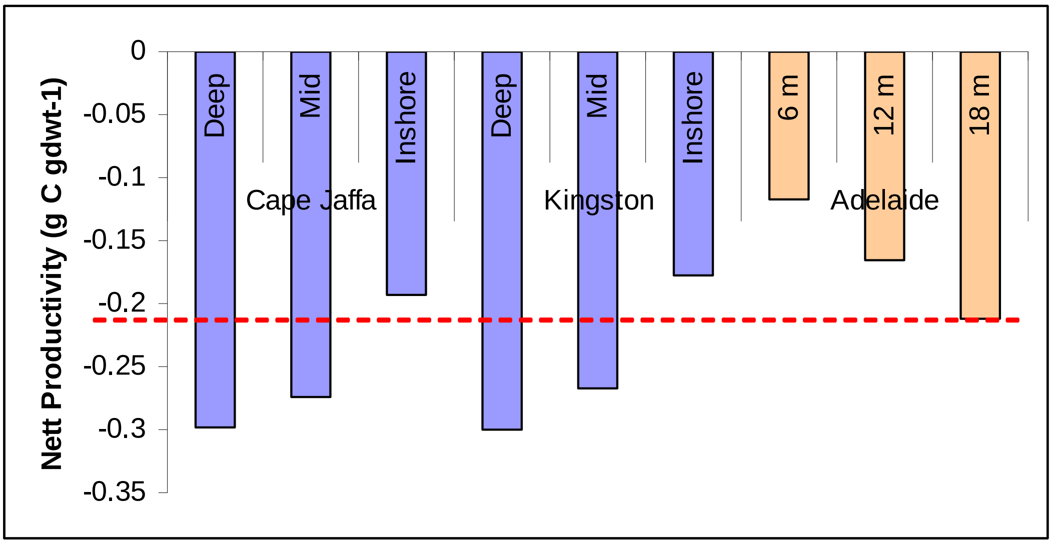

Figure 3.7. Productivity estimates at each of the three depths at two southeast sites and three sites off the Adelaide coast. These indicate the nett productivity that might be achieved by *Posidonia australis* with photokinetic parameters identical to those obtained for *P. australis* in Perth, Western Australia. The red line represents the modelled nett productivity demonstrated by seagrass beds off the coast of Adelaide at a point where the beds are considered limited by critically low light intensity.

3.4 Discussion

Measured attenuation coefficients in this region were relatively high, indicating a low degree of optical water quality. This is the product of turbidity and dissolved materials in the water column, and results in substantial amounts of light being filtered out before it reaches the seagrass canopy. Furthermore, it is worth pointing out that these measurements were carried out on a single day, and this was a day of particularly good weather. As a result, these attenuation figures are likely to represent an underestimate of the typical turbidity of this body of water. High attenuation coefficients may be due to both the amount of suspended and dissolved material, and also the relatively dynamic mature of this shallow exposed region.

Unsurprisingly, the apparently high attenuation recorded in this region results in relatively low levels of light reaching the seafloor. In fact, only the shallow inshore sites (at approximately 6 m depth) have light levels that are greater than that achieved at the deep edge of the seagrass beds off Adelaide. This is of concern given that the 18 m site at Adelaide represents the deep edge of the seagrass beds there, a point generally accepted to be defined by critically low light (Duarte 1991; Fourqurean and Zieman 1991).

Productivity depends on the light climate experienced by the seagrass, and the relationship between light and productivity defined by the photokinetic parameters. In this instance, the important finding is not that the nett productivity over this period is negative. Only a single month has been examined in this study, and numerous authors have demonstrated the ability of seagrasses to withstand long periods of nett negative productivity, so long as photosynthesis is able to replenish below-ground carbohydrate stores in times of high light. Thus one month of low productivity is not, in itself, cause for concern. Furthermore, using the parameters provided by Masini and Manning (1997) results in annual nett negative productivity at sites in Adelaide which *do* support seagrass beds. Thus, the estimates for the Adelaide beds (and therefore the sites in the southeast) represent an underestimate of productivity.

However, the critical issue here is one of relativity, rather than one of absolute values. Both the quantity of PAR and productivity at most of the sites in the current study are lower than that at the 18 m site at Adelaide, which is believed to represent the point at which light becomes critically low and productivity just breaks even. While it is probable that the productivity is underestimated by this model, it is also underestimated for the light limited Adelaide site. Thus, with demonstrably lower light and productivity, all southeast sites except the inshore sites have light environments that are apparently marginal, and therefore would be of great concern if further compromised.

The conclusions drawn here are based on a short term light study and productivity estimates drawn from Western Australian studies at a single point in time (and therefore not incorporating seasonal changes in photosynthetic efficiencies. Better understanding would be provided by an annual dataset describing the light data and estimates of the P-I relationship for seagrasses in this region across the course of a year. It is possible that different periods of the year will result in a more favourable light climate than that of Adelaide, resulting in greater annual productivity, but on the basis of the data collected in this study, there is no evidence suggesting this. Similarly, photosynthetic efficiency may be greater in the southeast seagrasses than those off the coast of Adelaide. However, it is worth noting that Adelaide's seagrasses are already surviving at light levels as low as recorded for any seagrasses in the world (4% of sub-surface irradiance; Collings et al. 2006), so any improvement on this efficiency would be surprising. It is also assumed that the light data collected both in the south-east and off Adelaide are representative for those locations in April, and that the particular years surveyed did not have unusually poor water quality. However, there are no long-term data on subsurface light availability at either site, so this assumption has had to be made.

3.5 Conclusion

Although light data has been collected over a limited period, and is therefore unable to provide an indication of annual carbon budget, a comparison is possible with the situation off the Adelaide coast over the same time of year. Both the average PAR levels and a simple productivity model indicate that the situation at all sites except the inshore sites, in terms of productivity is at least as marginal as that at the deep edge of the Adelaide seagrasses. As this is a point defined by low light, it is clear that any reduction of light in these regions may be detrimental to the long-term survival of any seagrass beds in the region.

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Chapter 4: Impacts of land-based abalone aquaculture discharges on the adjacent marine environment

Simon Bryars, Mandee Theil, and Keith Rowling

4.1 Introduction

Coastal habitats are under threat worldwide from anthropogenic disturbances. Eutrophication caused by various human-related activities, including the discharge of wastewater and stormwater, dredging, and aquaculture, is thought to be one of the biggest threats to temperate coastal reef and seagrass ecosystems (Walker and McComb 1992, Ziemann et al. 1992, Ralph et al. 2006, Turner et al. 2006). Elevated nutrients are thought to cause a shift on temperate reefs from canopy-forming macroalgae to turf-forming algae (Gorgula and Connell 2004, Russell and Connell 2005) and have also been linked with increased epiphyte loads in temperate reef systems (Russell et al. 2005). Likewise in seagrass systems, there has been a clear link between increased nutrients, increased epiphyte loads, and the degradation of seagrasses (Ralph et al. 2006). In addition, the effect of increased nutrients may be most severe in oligotrophic temperate systems that are adapted to low nutrient levels (Russell et al. 2005; Ralph et al. 2006). Thus it is imperative that the effects of nutrients from coastal activities are clearly understood and that appropriate mitigation strategies are employed.

In South Australia, there are several land-based abalone (*Haliotis* spp.) aquaculture farms situated along the coast. Greenlip abalone (*H. laevigata*) is the main species farmed, with much lesser quantities of blacklip abalone (*H. rubra*) produced. The land-based farms pump seawater directly from within a few hundred metres offshore into onshore gravity-fed flow-through farming systems and then discharge the seawater above the high tide mark of the adjacent intertidal environment (Figure 4.1). Annual discharge volumes at some farms are substantial at around 30-35 GL (Table 4.1). The industry commenced in the 1990s and currently produces around 100T abalone year^{-1} with a farm gate value of around \$3.1M (Knight et al. 2005). Farmed abalone are grown in shallow tanks where they are fed a plant-based manufactured feed.

During 2003/2004, a formal environmental risk assessment was conducted on the land-based abalone aquaculture industry in South Australia (Theil et al. 2004; see Chapter 1). From the workshop part of the risk assessment, 62 of the 78 issues that were discussed relating to the effects of farming on the environment, were given a negligible or low ranking. The remaining 16 'priority' issues were given a risk ranking of moderate or moderate to high. No issues were rated as an extreme risk. After reviewing the literature, Theil et al. (2004) changed most of the 16 priority issues to a low ranking, with the exception of a few issues receiving a low to moderate or moderate final ranking:

Figure 4.1. Outfall pipes discharging land-based abalone farm seawater to the intertidal shoreline at Smith Bay, Kangaroo Island. Note how the outflow of water has essentially transformed the area from an intertidal to a subtidal habitat.

- The effect of water quality (nutrients) on the marine environment (moderate risk)
- The impact of land-based abalone aquaculture on erosion (moderate risk)
- The impact of land-based abalone aquaculture on sensitive habitats (low to moderate risk)

Importantly, one of the remaining priority issues from the assessment by Theil et al. (2004) was the potential impact on adjacent marine communities from elevated nutrients caused by farm discharges containing uneaten feed, faeces and metabolic wastes. Consequently, we conducted biological surveys at the three key South Australian farming regions in order to determine if any nutrient-related impacts have occurred since the commencement of the industry. As there are no quantitative data prior to the farms starting, we had to rely on comparisons between appropriate control (non-farm) and putatively impacted (farm) sites. Also, because our surveys could only be over a limited time-period, we deliberately compared key indicators of long-term environmental change that might be associated with elevated nutrients. Appropriate indicators were deemed to be algal, seagrass, and sessile invertebrate communities in both intertidal and subtidal areas. Key indicators of subtidal change were the loss and/or degradation of the long-lived, slow-growing seagrass species *Amphibolis* and *Posidonia*, the loss of canopy-forming macroalgae in favour of non-canopy-forming macroalgae, and changes in invertebrate communities, particularly grazers and filter feeders. In the intertidal, we surveyed for macroalgae and invertebrates that may be indicative of nutrient-driven changes related to farm outfalls. We also sampled water column nutrients in discharge, intertidal and subtidal areas for comparison with any observed biological patterns. Other aspects of abalone farming that we specifically investigated were the escape and survival of farmed greenlip abalone, and possible erosional scouring of seagrasses around intake pipes (see above and Theil et al. 2004).

4.2 Methods

4.2.1 Field sites

Surveys were conducted at the three farming regions of Smith Bay located on the north coast of Kangaroo Island, Point Boston located in the southern part of Louth Bay on lower Eyre Peninsula, and Streaky Bay located on the west coast of South Australia (Figure 4.2). At Smith Bay, there is one main farm (Table 4.1), with a much smaller farm adjacent to its western boundary that was not targeted in the surveys. In addition, at the time of the surveys, a new farm was also under construction to the east of the main farm that was discharging water but which had very few abalone stocked. The new abalone farm began operation in late 2004. At Point Boston, there are two major farms, while at Streaky Bay one relatively small farm operates (Table 4.1). The Smith Bay and Point Boston regions are characterised by intertidal reef/boulder habitat and mixed subtidal reef and seagrass habitat. The Streaky Bay region has intertidal beach and reef habitats and mainly subtidal seagrass habitat.

Each farming region was visited on two occasions, once during February/March 2005 and once during May/June 2005. Water quality surveys were conducted on both occasions, with a subtidal biological survey conducted on the first occasion, and an intertidal biological survey on the second occasion. Artificial seagrass units were also deployed subtidally on the first trip and then collected on the second trip. Due to inherent differences in the number of farms, number of outfall pipes, coastal geomorphology and hydrodynamics, the experimental arrangement of sites for each of the three survey types differed both between and within regions. As the surveys were conducted after farming had commenced, it was imperative to select multiple ‘farm’ and ‘non-farm’ sites that were comparable in terms of exposure, depth, gradient, and substrate. Furthermore, large variation in algal community structure is known to occur at scales of km (Fowler-Walker and Connell 2002). Thus, at each of the three regions, all sites were located along the same section of coast just a few kilometres in total length (Figure 4.2). It was assumed that the farm(s) were not affecting the entire section of coast at hand.

At Smith Bay, the water quality and subtidal surveys were conducted using two farm sites (F1, F2) adjacent to the main farm and three non-farm sites (NF1-3) that were >1km away from F1 and F2 (Figure 4.2). Strictly speaking, F1 and F2 are psuedoreplicates of the farm ‘treatment’, but as the main farm is large with multiple outfall pipes and F1 and F2 were separated by >100m, it was felt that F1 and F2 could be treated as being independent of one another. For the Smith Bay intertidal survey, one farm site (F1), one site at the new farm (New F), and two non-farm sites (NF4, NF2) were used (Figure 2; see later for further explanation).

At Point Boston, the water quality and subtidal surveys were conducted using two farm sites (F1, F2) adjacent to each of the two farms, and two non-farm sites (NF1, NF2) that were several hundred metres away from any farming activities (Figure 4.2). The Point Boston intertidal survey utilised only F1, F2 and NF1, as there was no suitable intertidal habitat at NF2 for comparison (see later for further explanation).

At Streaky Bay, where only one farm is present, the water quality and subtidal surveys were conducted using a farm site (F) adjacent to the farm and four non-farm sites (NF1-4) in an array each side of F (Figure 2). NF1 and NF4 were >1km away from F, while NF2 and NF3 were ~100m away from F (Figure2). The Streaky Bay intertidal survey was conducted at sites F, NF2 and NF3 (Figure 2), where suitably comparable intertidal substrate could be found.

At each of the farm and non-farm subtidal sites, three linear 20m (at Smith Bay and Point Boston) or 50m (at Streaky Bay) transect lines were haphazardly laid out about 20m apart, perpendicular to the shore starting from the low-water mark. These transect lines formed the basis of the subtidal biological surveys and the subtidal water samples collected at each region. Transects at the farm sites were directly adjacent to, or close by, outfall flows across the intertidal.

Table 4.1. Summary of land-based abalone farms surveyed for possible environmental impacts. Farm codes relate to Figure 4.2. At Smith Bay, F1/F2 denotes one farm where two survey sites were located. Note that neither total biomass of abalone or annual farm production is presented due to confidentiality reasons.

Farming region	Farm	Year farming commenced	Annual discharge volume (GL)	Annual feed used (T)	Settlement ponds (Y/N)	# Outfall pipes
Smith Bay	F1/F2	1996	31	96	N	5
Point Boston	F1	1999	16	57	N	5
	F2	1993	36	44	N	6
Streaky Bay	F	2000	1.6	4.5	Y	1

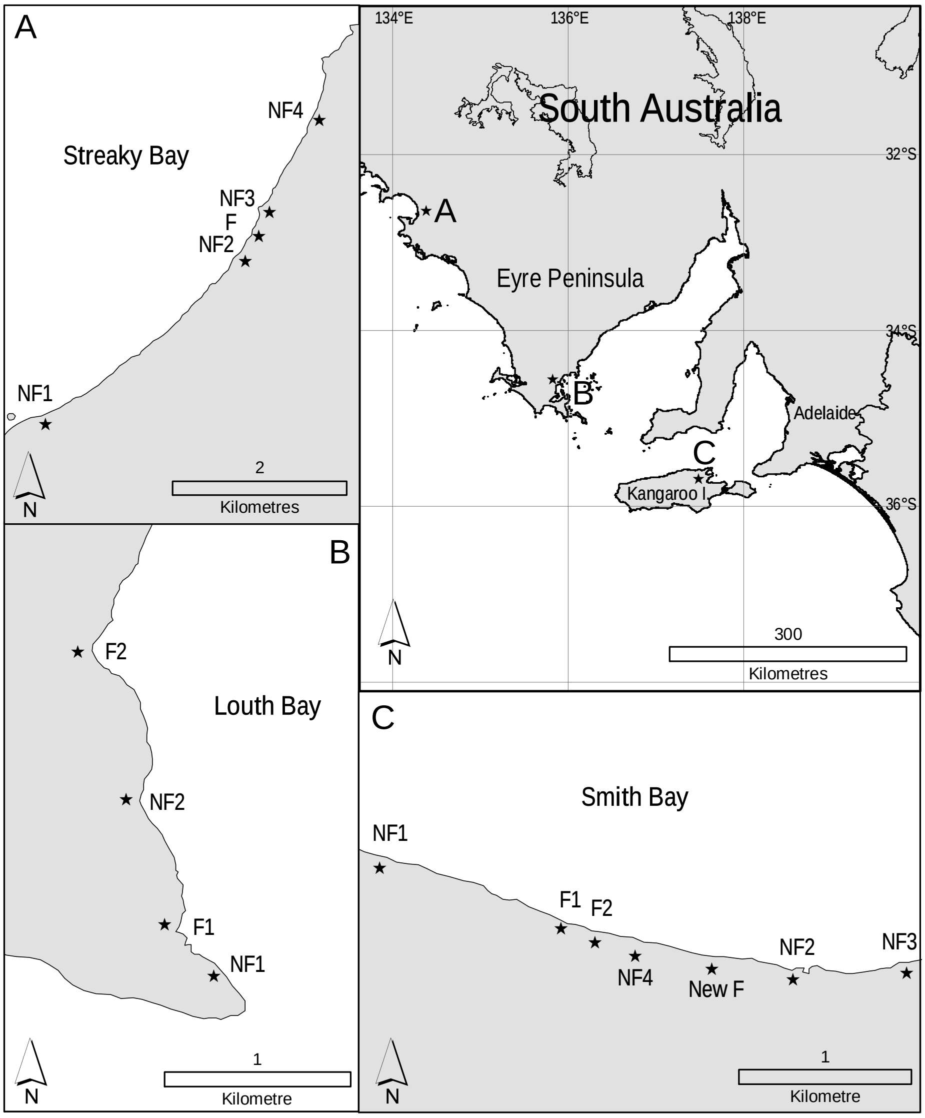

Figure 4.2. Location of sampling sites for field surveys at the three land-based abalone farming regions of (A) Streaky Bay, (B) Point Boston at the southern end of Louth Bay, and (C) Smith Bay. F, farm site; NF, non-farm site.

4.2.2 Water quality surveys

To assess the level of nutrients associated with farming activities, three methods were used; water sampling to measure dissolved nutrient concentrations, artificial seagrass units to quantify epiphyte growth, and seagrass nutrient content to quantify nutrient accumulation.

4.2.2.1. Dissolved nutrients

Water samples were collected at a range of locations within each region to track the nutrient status of water from the farm tanks to the adjacent subtidal waters. Thus, samples were taken from within farm drains (at those farms where open drains occurred), at the end of outfall pipes above the intertidal region, at the edge of the intertidal region where the outflow water meets the ocean, and at the farm and non-farm subtidal sites (Table 4.2). Three replicate samples were taken within a few minutes of one another at one spatial point from the drain, outfall pipe, and intertidal area nearest to each of the farm subtidal sites (Table 4.2). At Point Boston, samples were also collected during winter from an additional drain and outfall pipe at F2, and during summer at Streaky Bay, samples were collected from two drains within the farm (Table 4.2). The protocol for collection of subtidal samples was different to the other samples that were collected in fast flowing discharge waters; a single subtidal sample was collected from the surface water at a distance of <20m offshore adjacent to each transect, such that three replicate samples were collected ~20m apart along the coast for each subtidal site.

Water samples were collected in a 500mL container, from which ~45mL was then filtered (43µm) into a 50mL plastic container and placed on ice in darkness in the field. Samples were then transferred to a -30°C freezer once ashore. Within two months of being frozen, samples were analysed for total phosphorus (TP), total nitrogen (TN), ammonia as nitrogen (NH_3), and oxidized nitrogen (nitrate + nitrite, NO_x) at the Water Studies Centre, Monash University, Victoria, using flow injection analysis on a QuickChem 8000 Automated Ion Analyser. Limits of detection for each nutrient were: TP, <0.01 mg P L^{-1}; TN, <0.01 mg N L^{-1}; NH_3, <0.001 mg N L^{-1}; and NO_x, <0.001 mg N L^{-1}.

Table 4.2. Water sampling regimes used at various sites around land-based abalone farms in three growing regions during summer and winter of 2005. Numbers denote replicate samples collected on each occasion. See text for further details.

Region	Site	Summer	Winter
Smith Bay	F1 - outfall	3	3
	F1 - intertidal	3	3
	F2 - outfall	3	3
	F2 - intertidal	3	3
	F1 - subtidal	3	3
	F2 - subtidal	3	3
	NF1 - subtidal	3	3
	NF2- subtidal	3	3
	NF3 - subtidal	3	3
Point Boston	F1 - drain	3	3
	F1 - outfall	3	3
	F1 - intertidal	3	3
	F2 - drain	3	3
	F2 – drain2	-	3
	F2 - outfall	3	3
	F2 - outfall 2	-	3
	F2 - intertidal	3	3
	F1 - subtidal	3	3
	F2 - subtidal	3	3
	NF1 - subtidal	3	3
	NF2 - subtidal	3	3
Streaky Bay	F - drain	3	3
	F - drain 2	3	-
	F - outfall	3	3
	F - intertidal	3	3
	F - subtidal	3	3
	NF1 - subtidal	3	3
	NF2 - subtidal	3	3
	NF3 - subtidal	3	3
	NF4 - subtidal	3	3

4.2.2.2 Artificial seagrass units

Accumulation of epiphytes on artificial seagrass units (ASUs) was assessed at each of the subtidal sites in each region (Figure 4.2) following the technique of Bryars et al. (2003). Artificial seagrass leaves were constructed from 600mm long sections of 12mm wide, blue packing tape (Manufacturer – Gerrard Signode Pty Ltd). Each artificial seagrass leaf had a 5mm diameter hole punched 5-10mm from one end, and both ends were heat sealed with a flame. Seven artificial seagrass leaves were attached to a star dropper (1500mm length) at 150-200mm intervals using electrical ties (200mm length) (Figure 4.3). To enable the artificial seagrass leaves to move freely, electrical ties were not tightened fully. At each of the subtidal transect locations, a star dropper with artificial seagrass leaves attached was secured horizontally to the seabed using tent pegs and/or boulders about 10-30 m offshore from the low water mark in ca. 1-3 m depth (Figure 4.3).

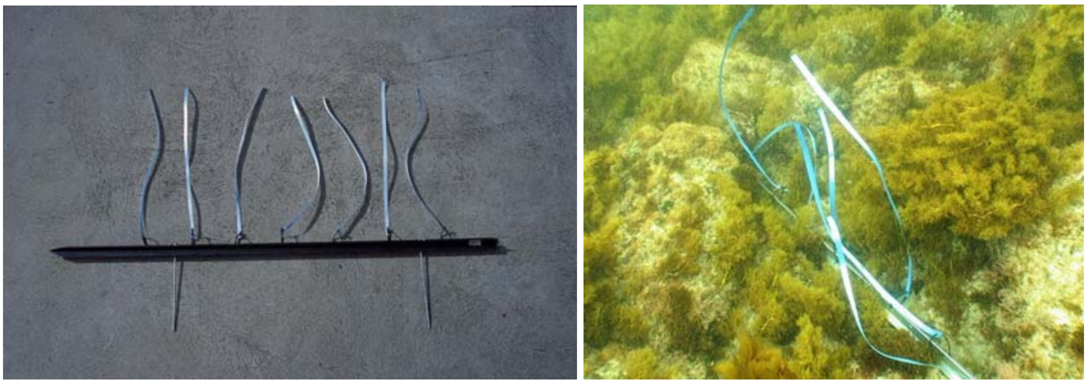

Figure 4.3. Artificial seagrass unit on land (left) and deployed in reef habitat (right).

After 15 weeks of deployment, all ASUs were retrieved. All remaining leaves (some were missing upon retrieval) from each ASU were removed from the star droppers, placed in separate labelled plastic bags, drained of water, sealed, and then frozen. Each leaf was later thawed and a standardized 500mm long section (120cm^2 total surface area) was cut from the upper end. This section excluded the tip (10mm length) of the leaf. Each of the standard length sections was pooled for each ASU, dried at 65°C for 24h, and then weighed to the nearest 0.01g. The technique used assumes that a clean 120cm^2 standard section of packing tape has a uniform weight; a test found that the mean dried weight of a section was 1.65g (n = 7) with a range of 1.63-1.77 and a standard error of 0.02. A value of 1.65g was subsequently subtracted from the final dry weight values (but taking into account the total number of leaves remaining on each ASU) to give a value for dry weight epiphytes 120cm^{-2} for each ASU.

4.2.2.3 Seagrass nutrient content

To investigate if seagrasses adjacent to farms were accumulating nitrogen to a greater extent than those away from farms (see Udy and Dennison, 1997), a subsample of leaves was collected from each of the three transects at each subtidal site of the Smith Bay region during February 2005. The other two farming regions were not sampled. Samples were kept on ice until returned from the field, whence they were kept in a –30ºC freezer. Epiphytes were then removed using a blunt razor blade. A ~5 cm long section of young leaf was then taken, placed in a vial and returned to the freezer. Samples were freeze-dried over night and then ground to a fine powder using a Fritsch stainless steel ball mill. An aliquot (150mg) was then analysed for total nitrogen on a LECO Truspec CNS Elemental Analyser at SARDI Aquatic Sciences, South Australia.

4.2.3 Subtidal surveys

4.2.3.1 Line-Intercept and Point-Intercept Transects

At Smith Bay and Point Boston, a line-intercept-transect (LIT) was used to score percent cover of various macroalgal classes (Table 4.3), as well as bare rock, bare sand, zooanthids, sponges, colonial ascidians, and the seagrass classes of *Amphibolis*, *Posidonia*, and Zosteraceae (Figure 4.4). Rather than an LIT, a point-intercept-transect (PIT) with 1m intervals was used at Streaky Bay to score the seagrass classes of *Posidonia australis*, *P. sinuosa*, and Zosteraceae, as well as bare sand and any other relevant macroalgal classes (Table 4.3, Figure 4.4). The PIT method was more suited to the Streaky Bay region where the benthos was relatively homogeneous seagrass, and as a PIT is more rapid than an LIT, it enabled longer transects and additional sites to be surveyed. The LIT and PIT methods each enabled calculation of percent cover of the various classes present and thus provided a description of the benthos. The LIT method will detect 3-dimensional changes in structure such that losses of canopy-forming algae will lead to increased scoring of low-lying and turf forming algae.

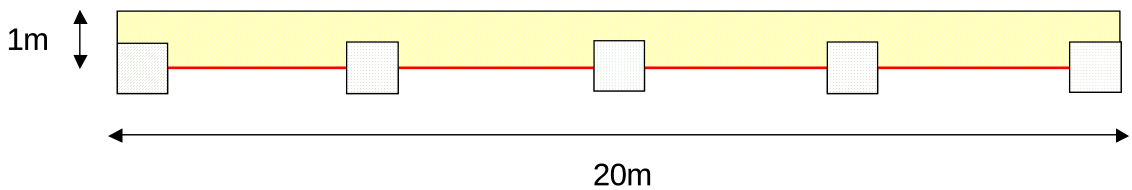

Figure 4.4. Schematic of techniques used in the subtidal surveys. The red line represents the transect line used for LIT (20m) or PIT (50m), the yellow area represents the belt transect, and the squares represent the 5 invertebrate quadrats.

Table 4.3. Physical characteristics and taxonomic examples of the 12 macroalgal classes used in the subtidal and intertidal biological surveys. Classes adapted from Turner et al. (2006).

Macroalgal class	Size (cm)	Shape	Texture	Taxonomic examples
Brown branching	10 – 100+	Robust, branched, often bushy in appearance	Leathery	*Acrocarpia*, *Caulocystis*, *Cystophora*, *Ecklonia*, *Sargassum*, *Scaberia*
Brown foliaceous	2 -20	Bushy, many branches, can be delicate	Soft	*Dicyota, Dilophus, Zonaria*
Brown lobed	2 -20	Flattened and rounded or fan shaped lobes	Firm	*Lobophora, Padina*
Brown lumpy	2 –20	Fleshy fronds or ball-like	Firm	*Colpomenia, Lethesia*
Green foliaceous	2 - 20	Bushy, many branches, can be delicate	Soft	*Bryopsis, Caulerpa, Codium, Struvea*
Green lumpy	2 -20	Fleshy fronds or ball-like	Firm	*Caulerpa, Codium*
Green membranous	2 -20	Membranous, thin sheets	Soft/slimy	*Enteromorpha, Ulva*
Red coralline	2 -7	Branched and spiky or fernlike	Hard	*Amphiroa, Metagoniolithon, Sporolithon*
Red encrusting	N/a	Surface crust	Hard	Unknown encrusting species, *Phymatolithon, Synarthrophyton*
Red foliaceous	2 - 20	Bushy, many branches, can be delicate	Soft	*Dictymenia, Gracilaria, Laurencia, Plocamium*
Red robust	2 – 100+	Robust, branched with robust or leaf-like blades	Leathery	*Osmundaria*
Turfing	< 2	Fine, feathery	Soft/slimy	Any green, red or brown algae <2 cm

4.2.3.2 Belt transects

A 1m-wide belt transect running parallel to the main transect (Figure 4.4) was used to quantify larger invertebrates at all three regions. Sessile and slow-moving, relatively large (> ~3 cm in size) invertebrates were counted from the following four groups: sponges/ascidians, gastropods, bivalves, and echinoderms. Some of the more common and easily identifiable organisms within the four groups were also scored at the lowest taxonomic level possible.

4.2.3.3 Invertebrate quadrats

Along each transect at Smith Bay and Point Boston, a set of 5 evenly-spaced quadrats (Figure 4.4) was used to quantify visible, sessile and slow-moving invertebrates from the following six groups: gastropods, ascidians/sponges, bivalves, echinoderms, anemones, and red coralline algae. Some of the more common and easily identifiable organisms within the six groups were also scored at the lowest taxonomic level possible. Due to the sandy substrate at Streaky Bay, invertebrate quadrats were not sampled.

4.2.3.4 Seagrass quadrats

On each transect, five quadrats of $0.0625m^2$ (250x250mm) were haphazardly placed within a randomly located patch of seagrass (*Posidonia* at Smith Bay and Streaky Bay, *Amphibolis* at Point Boston) in 1-3 m depth. All aboveground material within each quadrat was then harvested at the level of the sediment. All harvested materials were placed in separate labelled plastic bags, drained of excess water, sealed, and then frozen at –30° C. Seagrass samples were later thawed and processed. For *Posidonia*, leaf counts were firstly conducted on the entire quadrat sample based upon the number of intact leaf sections with meristematic tissue at the base. A subsample of 30 leaves was then randomly removed and scraped clean with a sharp blade to remove epiphytes. The length and width of each of the 30 leaves was also measured. For *Amphibolis*, stem counts were conducted on the entire sample based upon the number of primary stems or 'plants'. A subsample of 10 plants was then randomly selected to measure primary stem length. Five of the 10 plants were randomly selected and all leaves were removed from them. A subsample of 30 leaves was then randomly chosen from these leaves and the length and width of each of the 30 leaves was measured. The stems of the five plants and the 30 leaves were scraped clean of epiphytes with a sharp blade.

Leaves, stems, and epiphytes from all sub-samples and remaining samples were placed into separate labelled plastic containers. All subsamples and remaining quadrat samples were dried at 65°C for 72h and then weighed to the nearest 0.01g. Seagrass leaf/stem density was calculated from leaf/stem counts and converted to number of leaves/stems.m^{-2}. Seagrass epiphyte load was calculated from dry weight values of seagrass and epiphyte material as the ratio of epiphyte biomass to seagrass biomass. Seagrass aboveground biomass was calculated from dry weight values of leaves (for *Posidonia*) or leaves + stems (for *Amphibolis*) using the epiphyte: seagrass biomass ratio and then converted to g DW.m^{-2}. For leaf length and width data, means from the sub-samples of leaves within each quadrat were calculated and used in subsequent analyses.

4.2.4 Intertidal surveys

A series of intertidal surveys were conducted on reef/boulder habitat at each of the three farming regions during May-June 2005. As the intertidal topography differed between each region, the sampling protocol had to be modified accordingly. At Smith Bay, the intertidal was relatively uniform with a ~30m wide strip of boulders around the entire bay. At Point Boston, the intertidal was far more complex with a mixture of flat rock, large boulders and high relief rock that created crevices and rock pools, and which varied in width along the coast from a few metres to many 10's of metres. In the vicinity of the farm at Streaky Bay, the intertidal was relatively uniform with a ~20m wide strip of flat rock with some vertical relief. Thus, within each region, it was critical to select farm and non-farm sites that had comparable topography.

At Smith Bay, 30m transects were laid from the high water mark (denoted by lichen-covered rocks) directly towards the sea, with five transects within three separate outfall flows near F1, two within two outfall flows at the new abalone farm site (New F), five at NF4, and five near NF3. All transects were ca. 20-30 m apart. At every 2m along each transect, a 250 x 250 mm quadrat was laid within 0.5m distance from the transect line, at the lowest point possible between the boulders. Thus, there were 16 quadrats per transect. At Point Boston, plots of about 5 x 5m in area were selected on flat rock platforms in the lower intertidal, with two plots (one within the outfall flow, one immediately adjacent to the outfall flow <30m away) at each of four outfalls at both F1 and F2, and four plots at NF1, several hundred metres away from farming areas. In each plot, 10 quadrats were haphazardly placed. At Streaky Bay, plots of about 5 x 5m were located in the upper part of the intertidal band of rock, with two plots within the one outfall at site F, two plots near NF2, and two plots near NF3. Within each site, plots were separated by ~10m. Ten quadrats were haphazardly placed within each plot.

At all three regions, the number of visible solitary invertebrates within each quadrat was counted and classified to the highest level possible in the field. For colonial mussels and macroalgae (see Table 4.3), the % cover within each quadrat was estimated in size intervals of 10%. All scoring was done in the horizontal plane only (i.e. we did not look under boulders or overhangs). At each faming region, the total area of intertidal coverage by farm outfall water was also estimated.

4.2.5 Scouring around intake pipes

A qualitative assessment of possible seagrass scouring around intake pipes was made at Smith Bay and Streaky Bay. The intake pipes at the Point Boston farms reportedly lie over hard substrates and, thus, were not surveyed. At Smith Bay, underwater video deployed from a small boat was used to survey four sets of intake pipes: two off the main farm, one off the new farm to the east, and one off the small farm to the west. At Streaky Bay, SCUBA was used to visually assess the three intake pipes.

4.2.6 Data analyses

The main aim of the study was to compare physical and biological data from farm and non-farm sites at the three different farming regions. Due to inherent differences in geomorphology and hydrodynamics between the three farming regions, analyses on physical and biological data were restricted to separate within-region comparisons.

4.2.6.1 Water quality data

Nutrients

Any values less than the detectable level were treated as zero. While this biases towards lower mean values, it is a precautionary approach when attempting to detect differences between unimpacted (non-farm) sites, with expected low nutrient levels, and putatively impacted (farm) sites with expected higher nutrient levels. Thus, it lowers the chance of a Type II error (i.e., accepting the null hypothesis of no significant difference when, in fact, there is one), which is the error of most concern in impact assessments. Graphical comparisons of nutrient concentrations were made between regions (Smith Bay, Point Boston, Streaky Bay) and locations (farm drain, farm outfall, farm intertidal, farm subtidal, non-farm subtidal) within regions. Mean and maximum nutrient values were also compared to the EPA (2003) water quality criteria for marine ecosystems, that are applicable for aquaculture operations, and against the more stringent ANZECC (2000) water quality guideline values for protection of marine waters in south-central Australia (low rainfall area).

For each region, statistical comparisons were made only between farm and non-farm subtidal sites, as this is the comparison that will indicate if farms may be affecting the adjacent marine environment. Differences in nutrient concentrations were examined using Bray-Curtis dissimilarity measures (Bray and Curtis 1957). This measure was chosen, as it is not affected by joint absences (noting that less than detectable values were treated as zero). Differences in Bray-Curtis dissimilarity measures were tested using the analysis of similarities (ANOSIM) routine of Clarke and Gorley (2001), while spatial patterns were plotted using a non-metric multi-dimensional scaling (nMDS) algorithm. In interpreting nMDS ordinations, if stress values are <0.1, then this corresponds to a good ordination with no real prospect of a misleading interpretation (Clarke and Warwick 1994). Nutrient data were square root transformed prior to analyses. In the first instance, a 2-way crossed ANOSIM of treatment (farm versus non-farm for Smith Bay and Point Boston) or site (for Streaky Bay) by season (summer, winter) was conducted using all samples as replicates. If this analysis showed a significant effect of treatment and/or season, then a more robust 2-way nested ANOSIM of site within treatment was performed on the entire data set or the winter and summer data sets separately (when season had a significant effect). However, it must be noted that due to the low number of sites per treatment, the power of the nested ANOSIMs was greatly reduced and in some cases a test statistic could not be calculated.

ASU weight and seagrass nitrogen content

Comparisons of ASU weight and seagrass nitrogen content were made using ANOVA. At Smith Bay and Point Boston, a nested model was used with Site as a fixed factor nested in Treatment (farm, non-farm) as a fixed factor. At Streaky Bay where there was only one farm site (and thus no replication for Treatment), comparisons were made only between sites, with Site as a fixed factor. Site was deemed to be a fixed factor in all ANOVA models, as the sites were not selected randomly from a larger set of sites; all possible farm sites were sampled and the non-farm sites were 'fixed' in terms of limited available sites that were deemed to be comparable (depth, gradient, exposure, occurrence along same stretch of coast). Normally, nested factors are treated as random, but in cases where there are great restrictions with options for random samples in space and time, this may be difficult to argue (Kingsford 1998).

4.2.6.2 Subtidal data

Benthic cover

Raw benthic cover values were adjusted according to the relative percent cover of hard and soft substrate. The categories of bare rock and bare sand were retained as they are habitats in their own right and may be important indicators of algal and seagrass loss, respectively. Differences related to farms were investigated using the multivariate techniques described for the nutrient data, except data were untransformed. For the Smith Bay and Point Boston data, a 2-way nested ANOSIM was performed with Site nested within Treatment. In addition to the ANOSIM and nMDS ordination, a similarity percentage (SIMPER) analysis (Clarke and Warwick 1994) was also performed to identify key taxa contributing to any differences between farm and non-farm groups. If multivariate analyses on benthic cover indicated differences, then ANOVA was used on the most abundant and/or influential macroalgal groups to test for differences between farm and non-farm treatments/sites with the same models outlined for the ASU data. ANOVA was also used on a modified data set of seagrass cover to test for differences between farm and non-farm treatments/sites with the same models outlined for the ASU weight data.

Seagrass morphology

At Smith Bay, both *P. angustifolia* and *P. sinuosa* were inadvertently harvested. As the two species have slightly different morphologies (Robertson 1984), this complicates comparisons across the five sites. Consequently, quadrats containing one or other of the two species were separated for further analyses, with quadrats containing mixed assemblages being discarded. As a result of this, only sites F1, NF1, NF2 and NF2 could be compared for *P. angustifolia*, and sites F1, F2 and NF3 for *P. sinuosa*. At Streaky Bay, the dominant seagrass was *P. australis*. However, some quadrats contained a mix of *P. australis* and *P.sinuosa*, or *P. sinuosa* only. As these two species have very different morphologies (Robertson 1984), any quadrats containing *P. sinuosa* were omitted from further analyses.

In comparing seagrass morphology it is pertinent to firstly analyse seagrass biomass and if this shows significant differences, make further comparisons of other seagrass variables, such as leaf length, that may contribute to the overall biomass. Thus, comparisons of biomass were firstly made with ANOVA. As replication existed at the quadrat level within transects for the seagrass morphology data, any statistical models needed to account for this. While testing of the other variables would best be done with a multivariate technique, this could not be achieved because (1) the datasets were unbalanced and the non-parametric testing procedure of PERMANOVA (Anderson 2001, 2005) in its present form cannot deal with this scenario, and (2) parametric MANOVA in SPSS (see below) requires all factors to be fixed, which affected outcomes greatly. Thus, separate ANOVAs were conducted on each of the other variables.

At Point Boston the ANOVA model was: Treatment + Site(Treatment) + Transect(Site(Treatment)), with Treatment and Site as fixed factors, and Transect as a random factor. Due to the separation of species at Smith Bay, Treatment (farm, non-farm) could not be used as a factor, with ANOVAs based on a model of: Site + Transect(Site) with Site as a fixed factor and Transect as random. At Streaky Bay, where only one farm exists, the same model as Smith Bay was used.

Invertebrates

Invertebrate belt transect and quadrat data were analysed using ANOSIM, nMDS, and SIMPER, as described earlier. All data were untransformed prior to analyses. For the quadrat data, the total number of invertebrates was summed from all 5 quadrats for each transect and then used in analyses. For the Smith Bay and Point Boston data, a 2-way nested ANOSIM of site within treatment was performed, while for Streaky Bay, a 1-way ANOSIM of site was used. Invertebrate groups were excluded from analyses where there was only one organism across all transects at a site, as these chance encounters will add little to multivariate analyses (Clarke and Warwick 1994). Consequently, the following groups were used for belt transect analyses; Smith Bay and Point Boston: gastropods, ascidians/sponges, echinoderms; Streaky Bay: all four groups sampled; and for quadrat analyses; Smith Bay: gastropods, ascidians/sponges, echinoderms; Point Boston: gastropods, ascidians/sponges, bivalves, anemones, red coralline algae.

4.2.6.3 Intertidal data

Macroalgal cover

Mean percent cover of macroalgae was calculated for each transect (Smith Bay) or plot (Point Boston, Streaky Bay) from all quadrats. For graphical analyses, taxa were omitted where the highest value for mean cover per plot was <5%, as these chance encounters added little to the interpretation of farm effects. Statistical analyses were not required for macroalgal cover (see Results).

Invertebrates

The total number of invertebrates was summed from all quadrats for each transect (Smith Bay) or plot (Point Boston, Streaky Bay) (= replicates) and converted to density (no.m^{-2}). For graphical and statistical analyses, taxa were omitted where there was only one individual across all replicates or there were no scores >1 in any of the replicates, as these chance encounters added little to the interpretation of farm effects. Data were analysed using ANOSIM, nMDS, and SIMPER, as described earlier. However, due to the varying experimental designs of the intertidal surveys, the ANOSIM models were different to those used on subtidal data. For Smith Bay and Streaky Bay, comparisons were made between individual sites, while at Point Boston a comparison was made of farm (F1, F2), adjacent-farm (F1A, F2A), and non-farm (NF1) treatment groups by pooling plots into each group. Due to large variation in abundances for Point Boston and Streaky Bay, data were fourth-root transformed to reduce the influence of abundant groups on the similarity matrix (Clarke and Warwick 1994).

4.2.6.4 Statistical analyses

ANOSIM, nMDS, and SIMPER analyses were performed with the software package PRIMER (ver 5.2.9, Primer-E Ltd, Plymouth). All ANOVA tests were conducted using the software package SPSS (ver 14.0, SPSS Inc., Chicago). Before conducting an ANOVA, the assumption of homogeneity of variances was tested using Levene's test and the normality of data was visually assessed using QQ plots. When assumptions were not met, data were transformed in an attempt to meet the assumptions. Where this failed, analyses were performed on untransformed data using a conservative alpha level of 0.01, instead of 0.05 (Underwood 1981). As is appropriate for percentages (proportions), benthic cover data and nitrogen content data were arcsine transformed according to Zar (1984) prior to analyses.

4.3 Results

4.3.1 Nutrients

Gradients of nutrient concentration were generally evident at each farming region, with highest levels in the farm drain, outfall and intertidal sites, lowered levels in the farm subtidal sites, and lowest levels in the non-farm subtidal sites (Figure 4.5, Table 4.4). However, patterns were often inconsistent between seasons and regions. For example, while total phosphorus was not detected at any of the sites in Smith Bay or Streaky Bay during summer, extremely high levels of TP (up to 0.25 mg P L^{-1}) were detected in and around the F2 discharge at Point Boston. It was later discovered that this was due to the periodic practice of adding fertiliser to promote algal growth on larval settlement plates occurring at the time of sampling; similarly high levels were not detected during winter. Patterns were also inconsistent from drain through outfall to intertidal sites, possibly due to biological processes in the open drains (Point Boston, Streaky Bay), settlement pond (Streaky Bay), and intertidal areas (all regions) which all contain macroalgae. While mean nutrient levels at Smith Bay and Streaky Bay were comparable, mean nutrient levels (apart from ammonia) at Point Boston were considerably higher (Figure 4.5). This was mainly due to the extremely high values of TP, TN and NO_x detected during summer at F2.

In intertidal and subtidal waters adjacent to farms, TP ranged from 0 to 0.25 and 0 to 0.23 mg P L^{-1}, respectively, while in subtidal waters away from farms it never exceeded 0.01 mg P L^{-1} (Table 4.4). However, TP was often not detected in the drain, outfall or intertidal waters, and rarely detected in subtidal waters; it was detected at farm subtidal sites just six times and once at a non-farm subtidal site. Excluding the F2 discharge at Point Boston during summer, TP only ranged from 0 to 0.04 mg P L^{-1} in the intertidal and subtidal sites; well below the EPA (2003) and ANZECC (2000) guideline values for marine waters of 0.5 and 0.1 mg P L^{-1}, respectively.

In intertidal and subtidal waters adjacent to farms, TN ranged from 0.08 to 2.9 and 0.06 to 2.8 mg N L^{-1}, respectively, while in subtidal waters away from farms it never exceeded 0.32 mg N L^{-1} (Table 4.4). At Point Boston, maximum farm values of TN in both intertidal and subtidal waters exceeded the ANZECC (2000) water quality guideline value, but not the EPA (2003) value (Table 4.4). These high values were apparently related to the fertilizer treatment (see above).

In intertidal and subtidal waters adjacent to farms, NH_3 ranged from 0.007 to 0.068 and 0 to 0.056 mg N L^{-1}, respectively, while in subtidal waters away from farms it never exceeded 0.01 mg N L^{-1}. Furthermore, NH_3 levels adjacent to farms are not considered to be extreme, barely exceeding the EPA (2003) and ANZECC (2000) water quality guideline values (Table 4.4).

In intertidal and subtidal waters adjacent to farms, NO_x ranged from 0.006 to 2.7 and 0.002 to 2.5 mg N L^{-1}, respectively, while in subtidal waters away from farms it never exceeded 0.025 mg N L^{-1}. Mean levels of NO_x were substantially higher at the Point Boston farm versus non-farm subtidal sites (Figure 4.5), and during summer, high mean values of 0.051 and 1.33 mg N L^{-1} were detected adjacent F1 and F2, respectively. The F2 value was also apparently related to the fertilizer treatment (see above). Nonetheless, maximum levels of NO_x in the intertidal exceeded the ANZECC (2000) value at all three regions (Table 4.4), and in both intertidal and subtidal waters adjacent to the Point Boston farms, maximum values of NO_x exceeded the ANZECC (2000) and EPA (2003) values (Table 4.4).

While absolute differences between farm and non-farm subtidal nutrient concentrations were not great (Figure 4.5), multivariate statistical comparisons of these two groups showed some significant differences when treating all samples as replicates. A 2-way crossed ANOSIM of the Smith Bay data (without TP for which all values were less than detectable limits), showed a significant difference between treatment (Global R = 0.78, P = 0.001) and season (Global R = 0.811, P = 0.002). This conclusion is visually supported by an nMDS plot, which shows a clear separation of farm versus non-farm samples in both summer and winter. Nonetheless, a 2-way nested ANOSIM of the Smith Bay data showed no significant effect of treatment in summer (Global R = 0.583, P = 0.10) or winter (Global R = 0.75, P = 0.10). At Point Boston, a 2-way crossed ANOSIM showed a significant effect of treatment (Global R = 0.423, P = 0.001) and season (Global R = 0.194, P = 0.02), which is visually supported by nMDS plots (Figure 4.6). It is apparent that three of the farm samples are clear outliers; these are the F2 summer samples described earlier. A 2-way nested ANOSIM of the complete Point Boston data set showed no significant treatment effect for summer (Global R = 0.5, P = 0.333), while for winter a Global R statistic could not be calculated. At Streaky Bay, a 2-way crossed ANOSIM (without TP, which was detected in just one sample) showed no significant effect of treatment (Global R = 0.204, P = 0.088) but a significant effect of season (Global R = 0.75, P = 0.001), which is visually supported by an nMDS plot (Figure 4.6).

Table 4.4. Minimum and maximum nutrient concentrations (mg L^{-1}) for farm intertidal and subtidal, and non-farm subtidal sites at the land-based abalone farming regions of Smith Bay, Point Boston, and Streaky Bay. Values in bold are above the EPA (2003) guideline values for marine ecosystems of 0.5, 5, 0.05 and 0.2 mg L^{-1} for TP, TN, NH_3 and NO_x, respectively. Values with * are above the ANZECC (2000) guideline values for marine waters in south central Australia (low rainfall area) of 0.1, 1, 0.05 and 0.05 mg L^{-1} for TP, TN, NH_3 and NO_x, respectively. TP = total phosphorus, TN = total nitrogen, NH_3 = ammonia as N, NO_x = oxidized nitrogen.

Farming region	Sites	TP min	TP max	TN min	TN max	NH_3 min	NH_3 max	NO_x min	NO_x max
Smith Bay	Farm - inter	0	0.04	0.15	0.28	0.031	**0.063***	0.01	0.069*
	Farm -sub	0	0	0.06	0.18	0.003	**0.056***	0.008	0.026
	Nonfarm -sub	0	0	0.04	0.11	0.002	0.004	0.002	0.015
Point Boston	Farm - inter	0	0.25*	0.08	2.9*	0.007	**0.068***	0.006	**2.7***
	Farm -sub	0	0.23*	0.1	2.8*	0.003	0.042	0.01	**2.5***
	Nonfarm -sub	0	0.01	0.07	0.15	0.001	0.01	0.004	0.025
Streaky Bay	Farm - inter	0	0.01	0.14	0.3	0.032	0.041	0.008	0.094*
	Farm -sub	0	0.01	0.09	0.16	0	0.027	0.002	0.005
	Nonfarm -sub	0	0	0.05	0.32	0	0.008	0.002	0.008

As the land-based abalone farms are discrete units, annual nutrient loads to the adjacent marine environment can be estimated based on annual discharge volumes and the difference in concentration of incoming and outgoing water, such that any difference in nutrient mass is due to production within a farm. To this end, annual nutrient loads for each farm were calculated by multiplying the discharge volume (Table 4.1) by the difference in mean total dissolved nutrient concentration between the farm outfall sites (i.e., outgoing water) and the non-farm subtidal sites (i.e., representative of incoming water). Mean values were calculated using all available data from summer and winter, except in the case of Point Boston F2, where the peak nutrient levels that were detected during the fertiliser treatment were excluded from calculations.

Estimates of annual nutrient loads displayed some clear patterns (Figure 4.7). At all farms, nitrogen was released in far greater quantities than phosphorus, and apart from Streaky Bay, ammonia was the most prevalent inorganic form of nitrogen being released to the marine environment (Figure 4.7). It is possible that the settlement pond at Streaky Bay was contributing to the apparent shift from ammonia to oxidized nitrogen (see earlier). Smith Bay had much greater nitrogen loads than the other farms. This can be explained by the greater amount of feed input at Smith Bay (Table 4.1) and, presumably, a commensurately greater biomass of abalone. Streaky Bay, with its small annual discharge volume (Table 4.1) and small-scale of operation, had the lowest annual nutrient loads.

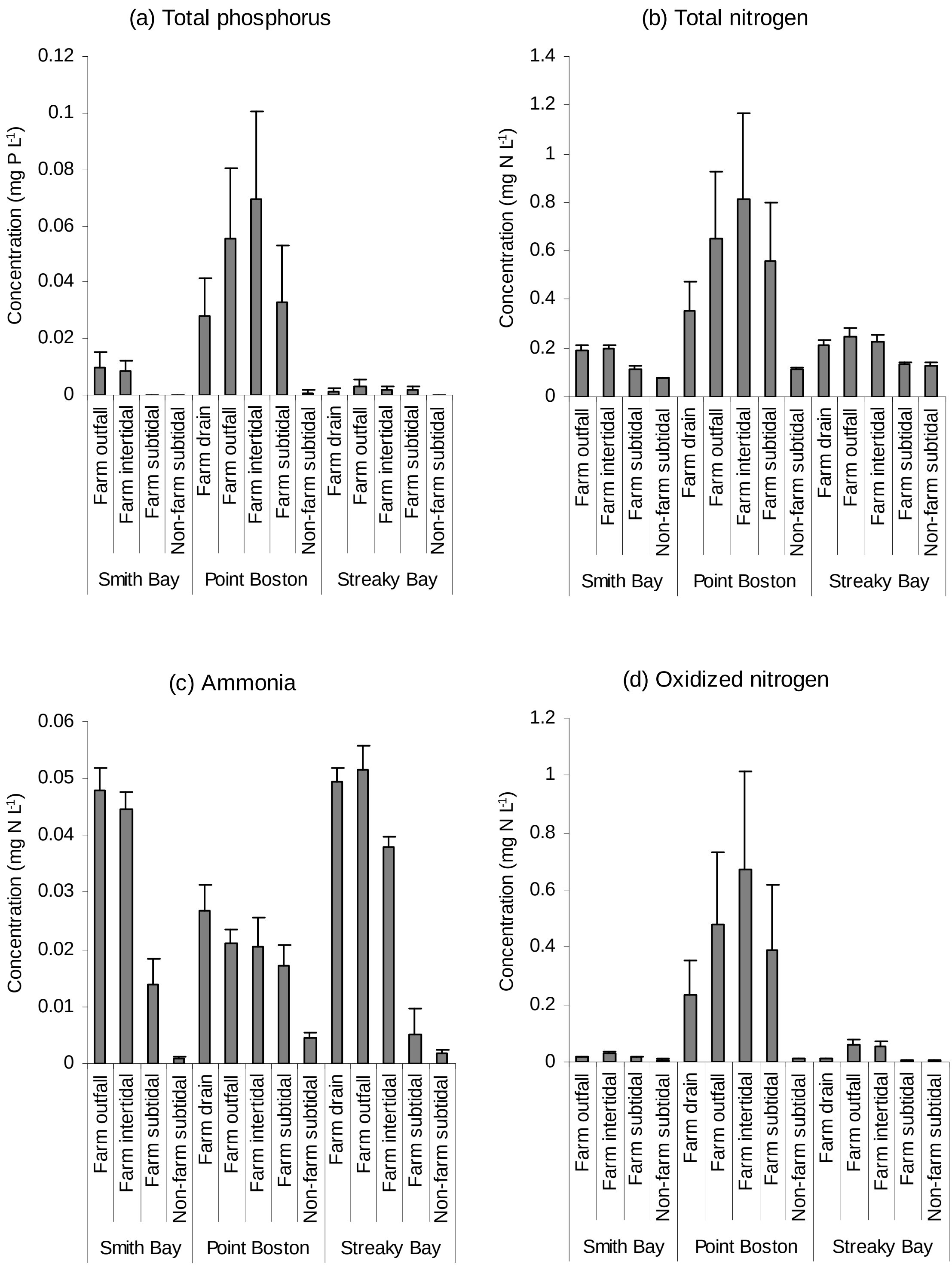

Figure 4.5. Dissolved concentrations (mean + SE) of (a) total phosphorus, (b) total nitrogen, (c) ammonia (total as nitrogen), and (d) oxidized nitrogen at farm and non-farm sites for the land-based abalone farming regions of Smith Bay, Point Boston, and Streaky Bay. Means are from all data pooled for each site (n = 12, 12, 12, 17, 15, 15, 12, 11, 12, 9, 6, 6, 6, 24, for each site plotted from left to right). Dashed horizontal lines indicate the EPA (2003) guideline values for marine ecosystems (note that those for total phosphorus and total nitrogen are not shown as they are off the scale).

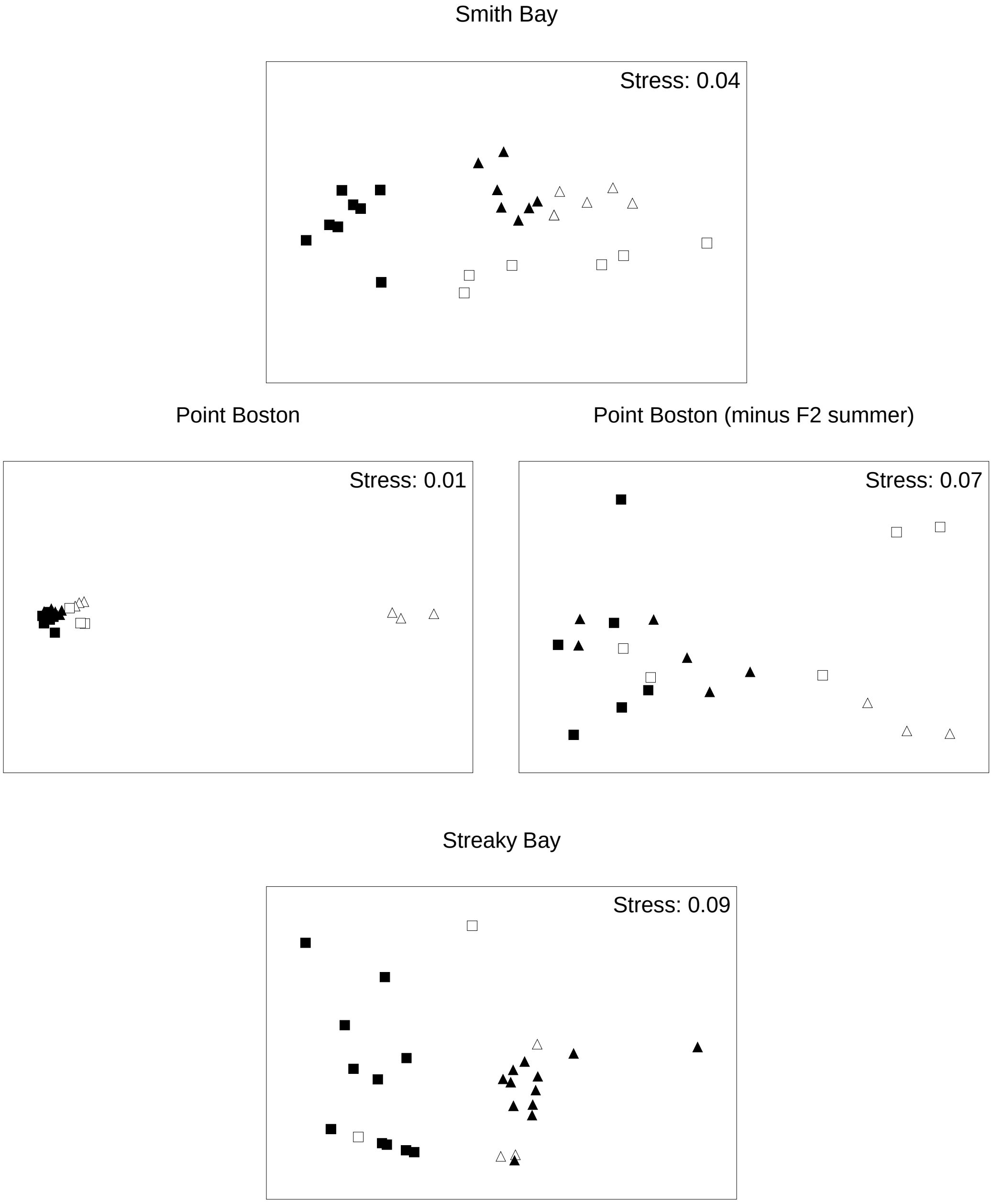

Figure 4.6. nMDS plots of nutrient concentrations from farm and non-farm subtidal water samples collected during the summer and winter of 2005 at the three land-based abalone growing regions of Smith Bay, Point Boston, and Streaky Bay.
Open triangle, Farm Summer; Closed triangle, Non-farm Summer; Open square, Farm Winter; Closed square, Non-farm Winter. Smith Bay and Streaky Bay plots include TN, NH_3, and NO_x; Point Boston plots include TP, TN, NH_3, and NO_x, with the right-hand plot excluding the 3 summer samples from F2.

The estimates of annual nutrient loads have some obvious shortcomings; at Point Boston F1, the ammonia load was greater than the total nitrogen load, and at Point Boston F2, the estimate for oxidized nitrogen was negative (-78). Nonetheless, the estimates do provide some indication of the annual nutrient loads from land-based abalone farms in SA. More frequent sampling of inlet and outlet water should provide more reliable estimates of annual nutrient loads, as it is apparent from the sampling at Point Boston F2 (see earlier), that nutrient concentrations in the outfalls are highly variable. An alternative, but more complicated, approach to estimating loads would be to collect detailed data on feed inputs, feed conversion ratios, and waste outputs.

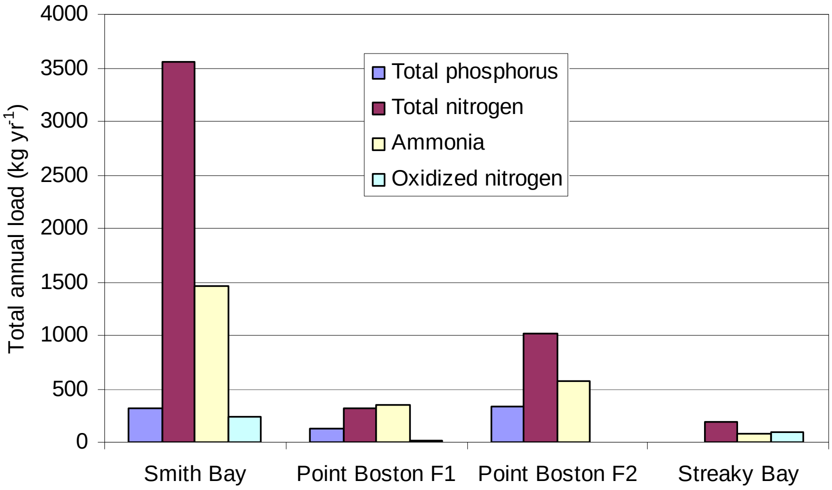

Figure 4.7. Estimates of total annual nutrient loads for 2005 (kg yr^{-1}) for the four land-based abalone farms surveyed for water quality. The estimate for oxidized nitrogen at Point Boston F2 was negative and is not shown.

No spatial patterns of ASU epiphyte load related to farm versus non-farm groups were evident at any of the farming regions (Figure 4.8; ANOVAs: Smith Bay $F_{1,10} = 0.105$, $P = 0.753$; Point Boston $F_{1,7} = 0.004$, $P = 0.952$; Streaky Bay $F_{4,10} = 0.312$, $P = 0.864$). A clear pattern of % nitrogen in *Posidonia* leaves was evident, with slightly elevated levels at the farm sites of Smith Bay (Figure 4.9). Differences between farm and non-farm treatments were statistically significant ($F_{1,6} = 37.119$, $P = 0.001$).

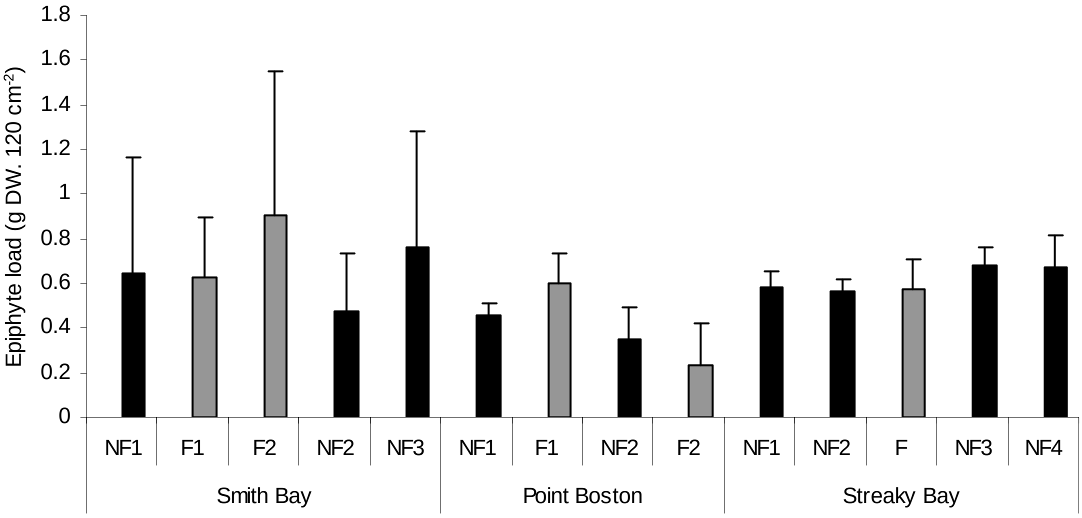

Figure 4.8. Epiphyte load (mean + SE) on artificial seagrass units after 15 weeks of deployment at farm (F, grey bars) versus non-farm (NF, black bars) subtidal sites for the land-based abalone farming regions of Smith Bay, Point Boston, and Streaky Bay. n = 3 for all sites, except n = 2 for Point Boston F2.

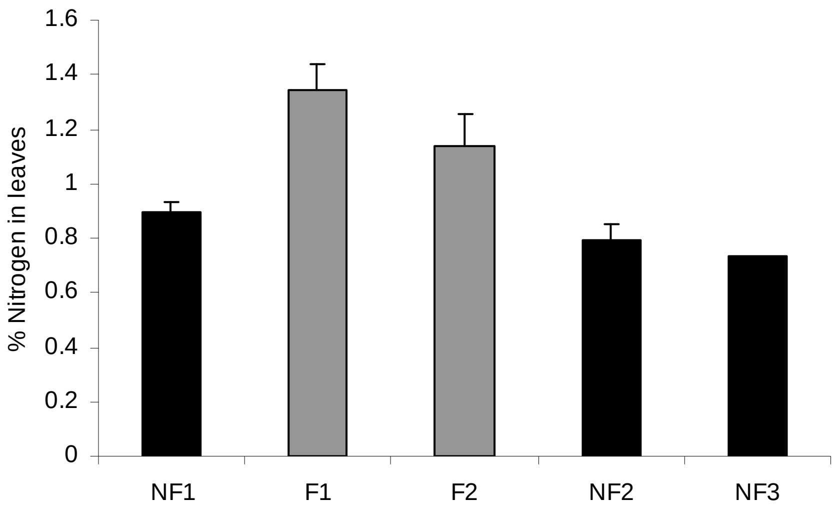

Figure 4.9. % nitrogen (mean + SE) in *Posidonia* leaf tissue at farm (F, grey bars) and non-farm (NF, black bars) subtidal sites for the land-based abalone farming region of Smith Bay. n = 3 for NF1, NF2; n = 2 for F1, F2; n = 1 for NF3 (as some samples were misplaced).

4.3.2 Subtidal habitats

4.3.2.1 Benthic cover

Benthic cover differed considerably between the three regions with a complex mixture of hard- and soft-substrate classes occurring at Smith Bay and Point Boston, and mainly soft-substrate classes at Streaky Bay (Figure 4.10). At Smith Bay the benthos was dominated by brown branching algae (mainly *Cystophora*), the seagrasses, *Amphibolis* and *Posidonia*, and red coralline algae. There was very little cover of turfing algae at any of the Smith Bay sites. Brown branching algae, *Amphibolis*, and turfing algae dominated the benthos at Point Boston, with a notable presence of red coralline algae at Site F2. The remaining benthic cover at both Smith Bay and Point Boston was composed of small contributions of other forms of brown, red and green algae, Zosteraceae seagrass, various invertebrates, and bare sand and rock. *Posidonia* dominated the benthos at Streaky Bay, with brown branching algae, Zosteraceae, and bare sand also present in noticeable, but varying, amounts across the five sites.

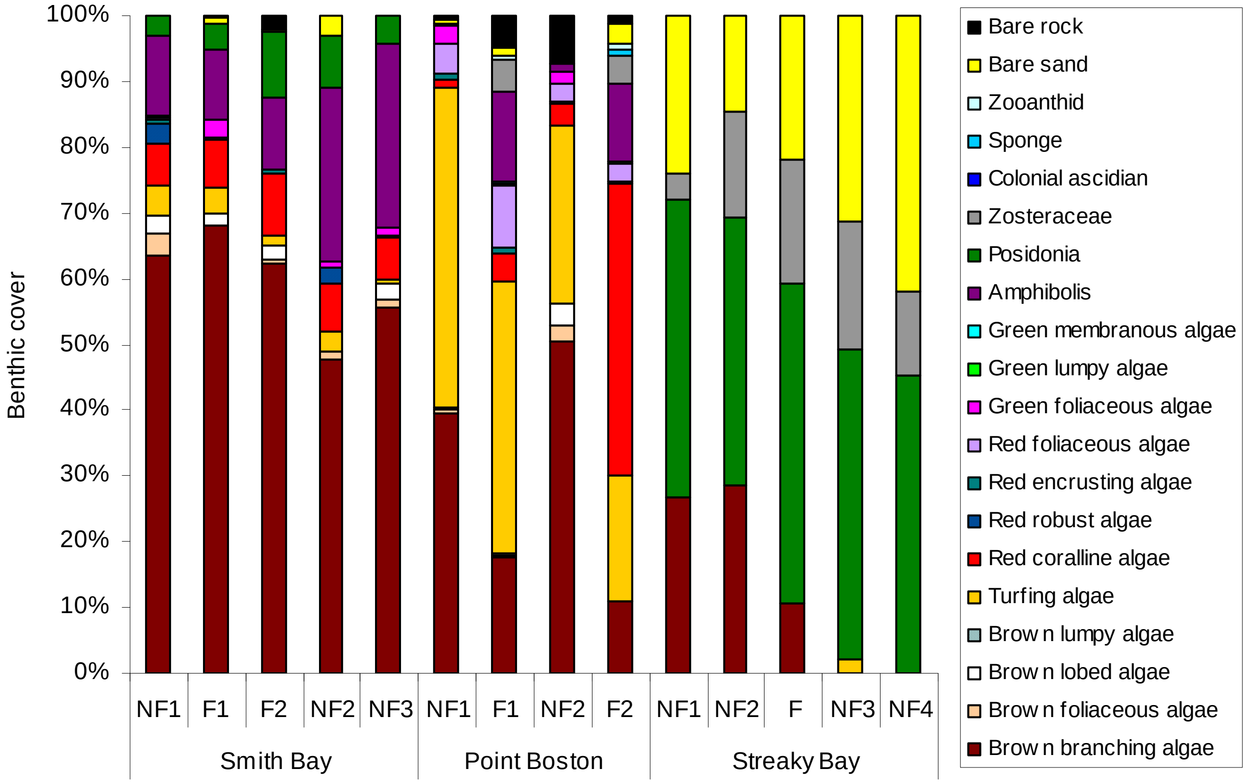

Figure 4.10. Benthic cover of 20 habitat classes at farm (F) and non-farm (NF) subtidal sites for the land-based abalone farming regions of Smith Bay, Point Boston and Streaky Bay. Values are means of three transects at each site.

While Smith Bay and Point Boston were dominated by hard substrate, Streaky Bay was predominantly soft substrate (Figure 4.11). However, within each region, there were differences in composition between sites, with some evidence of alongshore trends at Smith Bay and Streaky Bay. Therefore, to enable meaningful site comparisons, measures of macroalgal and seagrass cover required standardisation for substrate type (see later). Hard substrates were characterised by boulders at Smith Bay, boulders and rock slabs at Point Boston, and low relief rock at Streaky Bay.

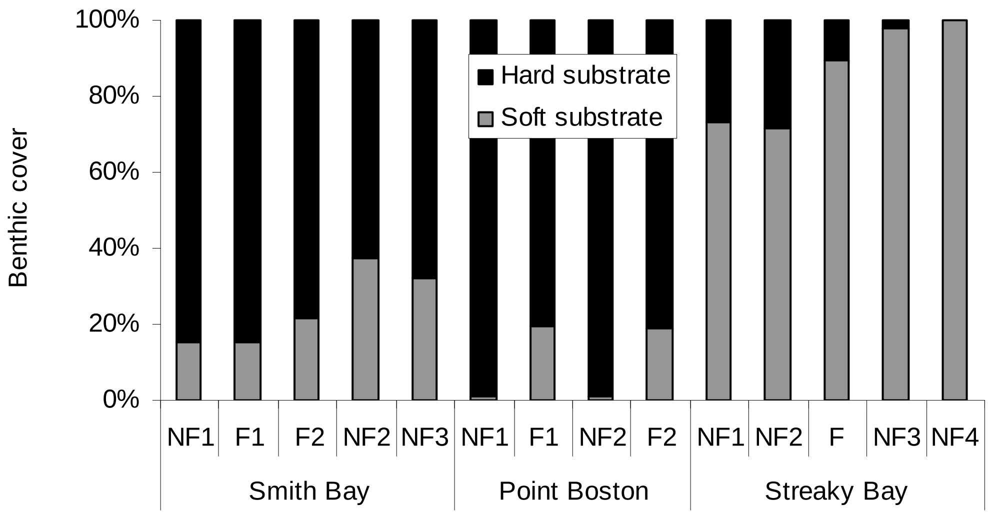

Figure 4.11. Benthic cover of hard and soft substrate at farm (F) and non-farm (NF) subtidal sites for the land-based abalone farming regions of Smith Bay, Point Boston and Streaky Bay. Values are means of three transects at each site.

2-way nested ANOSIMs of benthic cover (adjusted for substrate type) showed no significant difference between farm and non-farm treatments at Point Boston (Global R = 0.25, P = 0.333) or Smith Bay (Global R = -0.583, P = 1.00), and an ANOSIM showed no difference between sites at Streaky Bay (Global R = 0.067, P = 0.285). However, while the nMDS plots of benthic cover visually support the statistical results for Smith Bay and Streaky Bay, with no separation of farm versus non-farm transects, at Point Boston there is a clear separation of farm and non-farm transects (Figure 4.12). A SIMPER analysis of the Point Boston data showed that 80% of the difference between the two groups was due to brown branching algae (29%), red coralline algae (28%), and turfing algae (23%).

Figure 4.12. nMDS plots of benthic cover on hard and soft substrates at Smith Bay, hard substrates at Point Boston, and soft substrates at Streaky Bay. Open symbols represent farm transects, closed non-farm transects.

4.3.2.2 Macroalgae

Brown branching algae, red coralline algae, turfing algae, and red foliaceous algae accounted for most of the macroalgal cover on hard substrates across the three farming regions. Across all five sites at Smith Bay, about 70-80% of hard substrate was covered with brown branching algae, while red coralline algae, turfing algae, and red foliaceous algae accounted for around 10, ≤5, and <1% cover, respectively (Figure 4.13). As the multivariate comparisons and data of benthic cover indicate no difference between farm and non-farm sites at Smith Bay (Figures. 4.12, 4.13), no univariate comparisons were made of the four dominant macroalgal groups. However, at Point Boston, there was multivariate evidence of a difference in benthic cover due mainly to the macroalgal groups of brown branching algae, red coralline algae, and turfing algae. For brown branching algae, which had about half the cover at the farm sites (around 15-20%) compared to the non-farm sites (around 40-50%; Figure 13), ANOVA showed a significant effect of farm treatment ($F_{1,8} = 56.004$, $P < 0.001$). Red coralline algae, which had about 50% cover at one of the farm sites (F2) but <10% at the other three sites, also showed a significant effect of farm treatment ($F_{1,8} = 17.988$, $P = 0.003$). Turfing algal cover showed no consistent pattern across farm and non-farm sites, with around 50% cover at NF1 and F1, and around 25% cover at F2 and NF2. Red foliaceous algae contributed about 10% of cover at F1. ANOVAs showed no significant effect of farm treatment for turfing algae ($F_{1,8} = 0.413$, $P = 0.539$) or red foliaceous algae ($F_{1,8} = 3.629$, $P = 0.093$). It was not possible to reliably compare macroalgal cover at Streaky Bay, as only NF1 and NF2 had hard substrate on all three transects; F1 and NF4 had hard substrate on only one transect each and NF3 had none. On the one relevant transect at F1, there was 100% cover of brown branching algae on hard substrate, as was the case at all transects on NF1 and NF2; thus indicating no loss of this group at the farm site.

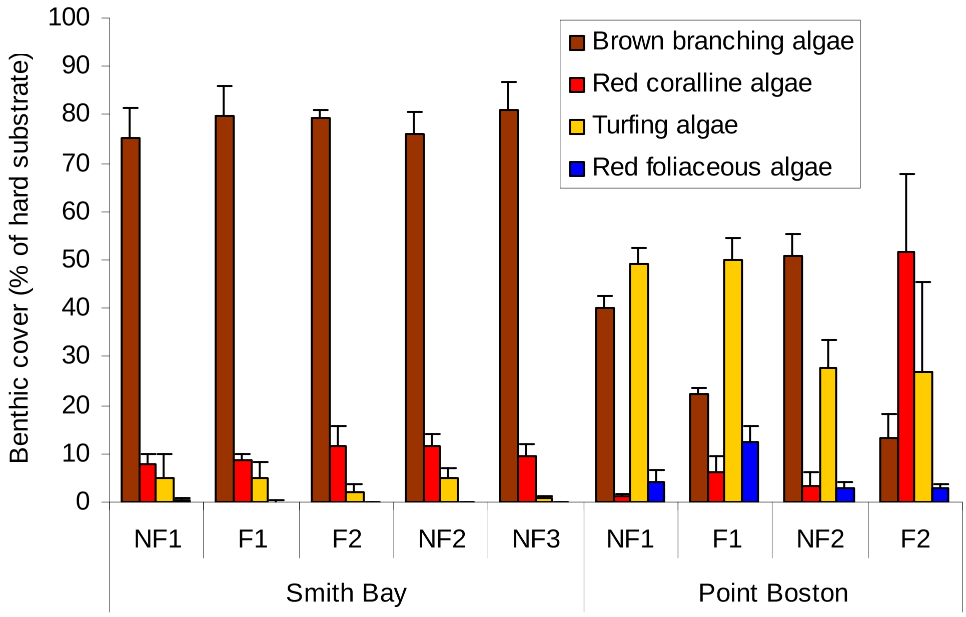

Figure 4.13. Benthic cover (mean + SE, n = 3) of the four dominant algal habitat classes at farm (F) and non-farm (NF) subtidal sites for the land-based abalone farming regions of Smith Bay and Point Boston.

4.3.2.3 Seagrasses

Whilst multivariate analyses of benthic cover indicated no difference between farm and non-farm sites due to seagrasses, further analyses of a modified seagrass data set were considered appropriate. Consequently, cover (%) of seagrasses was analysed using three classes: *Amphibolis*/*Posidonia*, Zosteraceae, and sand (Figure 4.14). *Amphibolis* and *Posidonia* were grouped because relatively low cover of the slow spreading, long-lived *Amphibolis* and *Posidonia*, in comparison to bare sand or the fast spreading, colonisers in the family Zosteraceae, may indicate that an impact has occurred. In the case of Smith Bay, where a mix of *Amphibolis* and *Posidonia* occurs, any relative differences between the two genera would provide no indication of long-term losses; but a decline in cover of the two combined could provide evidence.

At Smith Bay, the soft substrate areas were almost entirely *Amphibolis* and *Posidonia* with no indication of a negative effect at the farm sites (Figures. 4.10, 4.14). Indeed, an ANOVA showed no significant effect of farm versus non-farm treatment on the cover of *Amphibolis*/*Posidonia* ($F_{1,10} = 0.469$, $P = 0.509$). At Point Boston, a spatial pattern could not be reliably determined as the amount of soft substrate at the two non-farm sites was very low (1% of benthos). Nonetheless, at the two farm sites, *Amphibolis* covered between about 50 and 80% of soft substrate, with about 15% cover of Zosteraceae. Furthermore, bare sand contributed <30% at the two farm sites. Overall, the benthic cover of soft sediments provides no suggestion that major losses of seagrasses have occurred adjacent to the Point Boston farms.

At Streaky Bay, *Posidonia* covered about 45-60% and Zosteraceae about 5-20% of the soft substrate across all sites, with no indication of an effect on these seagrasses at the farm site (Figures. 4.10, 4.14). A MANOVA using the three benthic classes of *Posidonia*, Zosteraceae, and sand showed no significant difference between sites (Pillai's Trace, $P = 0.204$). A reasonable amount of bare sand (about 20-40%) occurred across the five sites. At Streaky Bay, the distance from the low water mark to the inshore margin of seagrass meadows was also measured, as a regression of this margin could indicate an impact. Distances were 129, 75, 75, 79, and 90m for sites NF1, NF2, F, NF3, NF4, respectively. Based upon these data there is no indication that the seagrass margin has regressed at F compared to NF2 and NF3. Comparisons of F with NF1 and NF4 are problematic due to inherent differences in beach profiles of these sites.

Seagrass morphology

At all three farming regions, there were no patterns across sites that would suggest an environmental gradient along the coast (Figures. 4.15-4.17). At Smith Bay, the only consistent pattern of seagrass morphology at farm versus non-farm sites appeared to be leaf length and maximum leaf length, which were both lowest at the two farm sites (Figure 4.15). However, Figure 4.15 shows both *P. angustifolia* and *P. sinuosa* combined; separate analyses are provided below.

An ANOVA of *P. angustifolia* biomass at Smith Bay showed a significant difference between sites ($F_{3,8.545}$ = 27.643, P < 0.001), with the farm site having a consistently lower biomass than the three non-farm sites (LSD test: F1 < NF1, P < 0.001; F1 < NF2, P < 0.001; F1 < NF3, P < 0.001). However, separate ANOVAs of the morphological variables for *P. angustifolia* showed no significant difference between sites for leaf length ($F_{3,6.410}$ = 1.054, P = 0.431), maximum leaf length ($F_{3,7.134}$ = 0.144, P = 0.930), leaf width ($F_{3,6.293}$ = 2.519, P = 0.151), leaf density ($F_{3,7.137}$ = 2.244, P = 0.169), or epiphyte load ($F_{3,6.469}$ = 0.854, P = 0.510). ANOVA of *P. sinuosa* biomass at Smith Bay showed no significant difference between sites ($F_{2,5.084}$ = 0.642, P = 0.564). Thus, further testing of other morphological variables was not conducted.

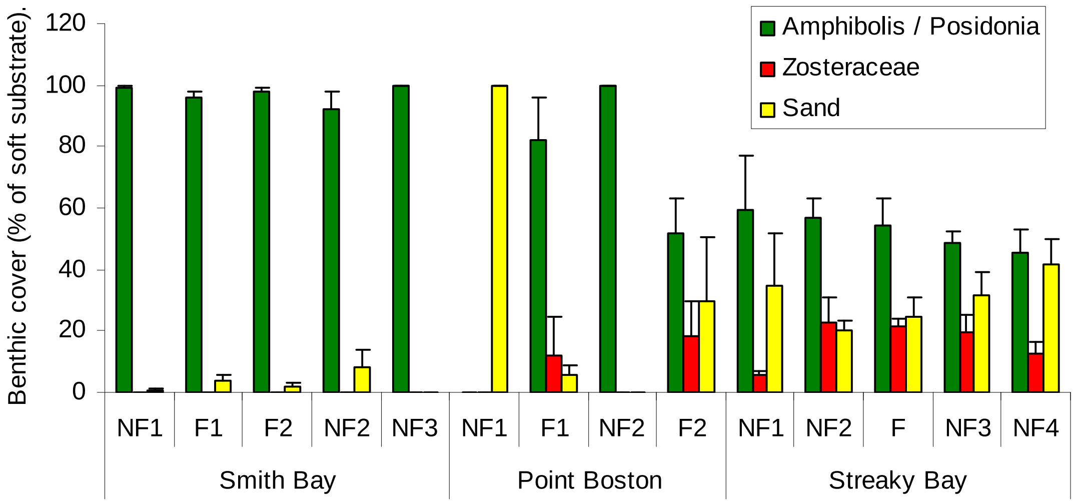

Figure 4.14. Benthic cover (mean + SE) of seagrasses and sand at farm (F) and non-farm (NF) subtidal sites for the land-based abalone farming regions of Smith Bay, Point Boston, and Streaky Bay. n = 3 for all sites, except Point Boston sites NF1 (n = 2) and NF2 (n = 1) where the total amount of soft substrate was also very low (see Figure 4.9).

At Point Boston, the only consistent visual pattern of farm versus non-farm sites was for stem length, which was lowest at the two farm sites (Figure 4.16). However, there was no significant difference between farm and non-farm treatment for biomass ($F_{1,8}$ = 4.347, P = 0.071) and further testing of the other morphological variables was deemed unnecessary.

At Streaky Bay there were no visual patterns of seagrass morphology that might indicate a negative effect of the farm (Figure 4.17), and ANOVA showed no significant differences in biomass between sites ($F_{4,9.045}$ = 2.804, P = 0.091). Further testing of the other morphological variables was deemed unnecessary.

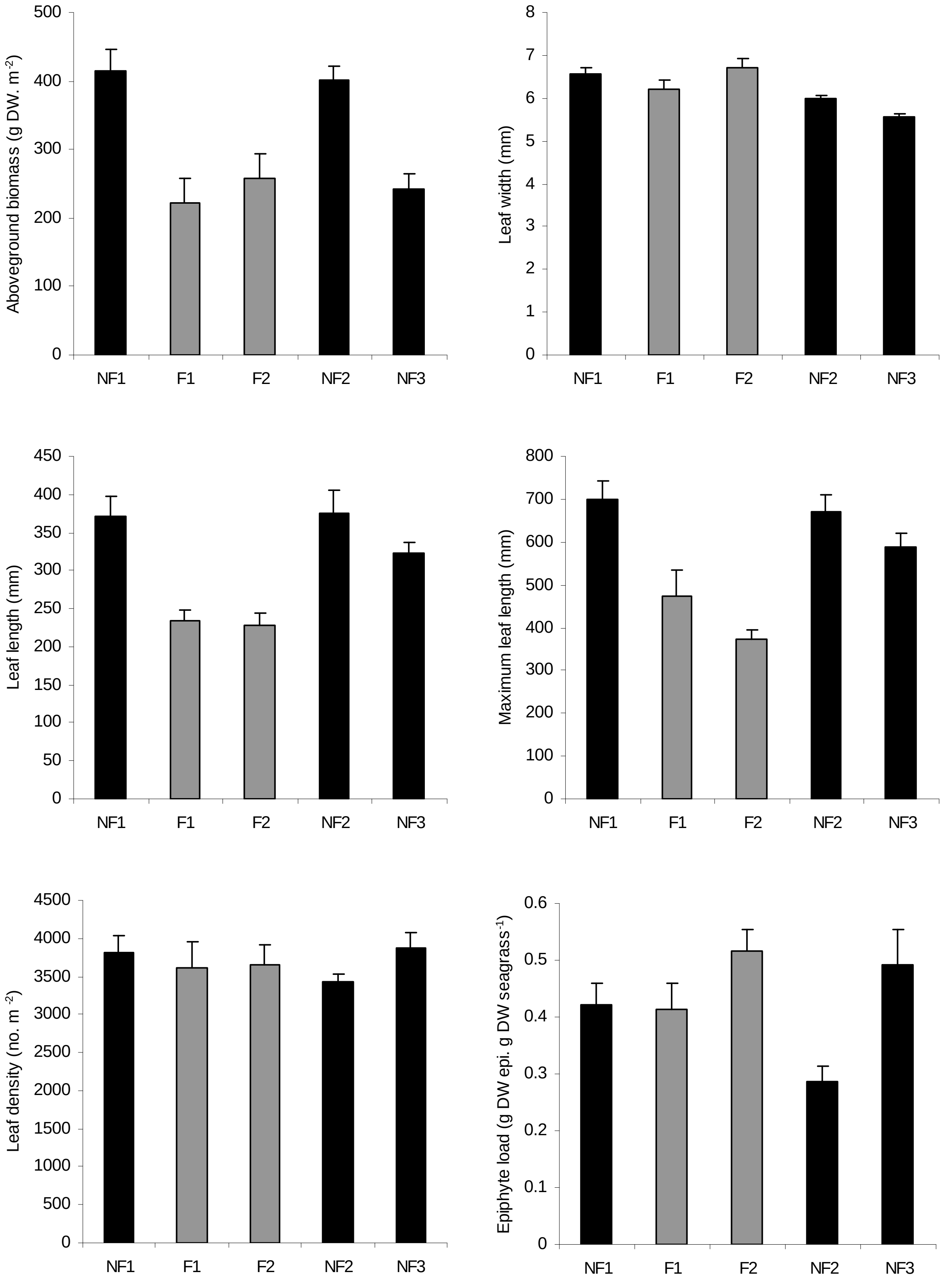

Figure 4.15. Parameters of seagrass morphology (mean + SE, n = 15) for *Posidonia angustifolia*/*sinuosa* at farm (F, grey bar) and non-farm (NF, black bar) subtidal sites for the land-based abalone farming region of Smith Bay.

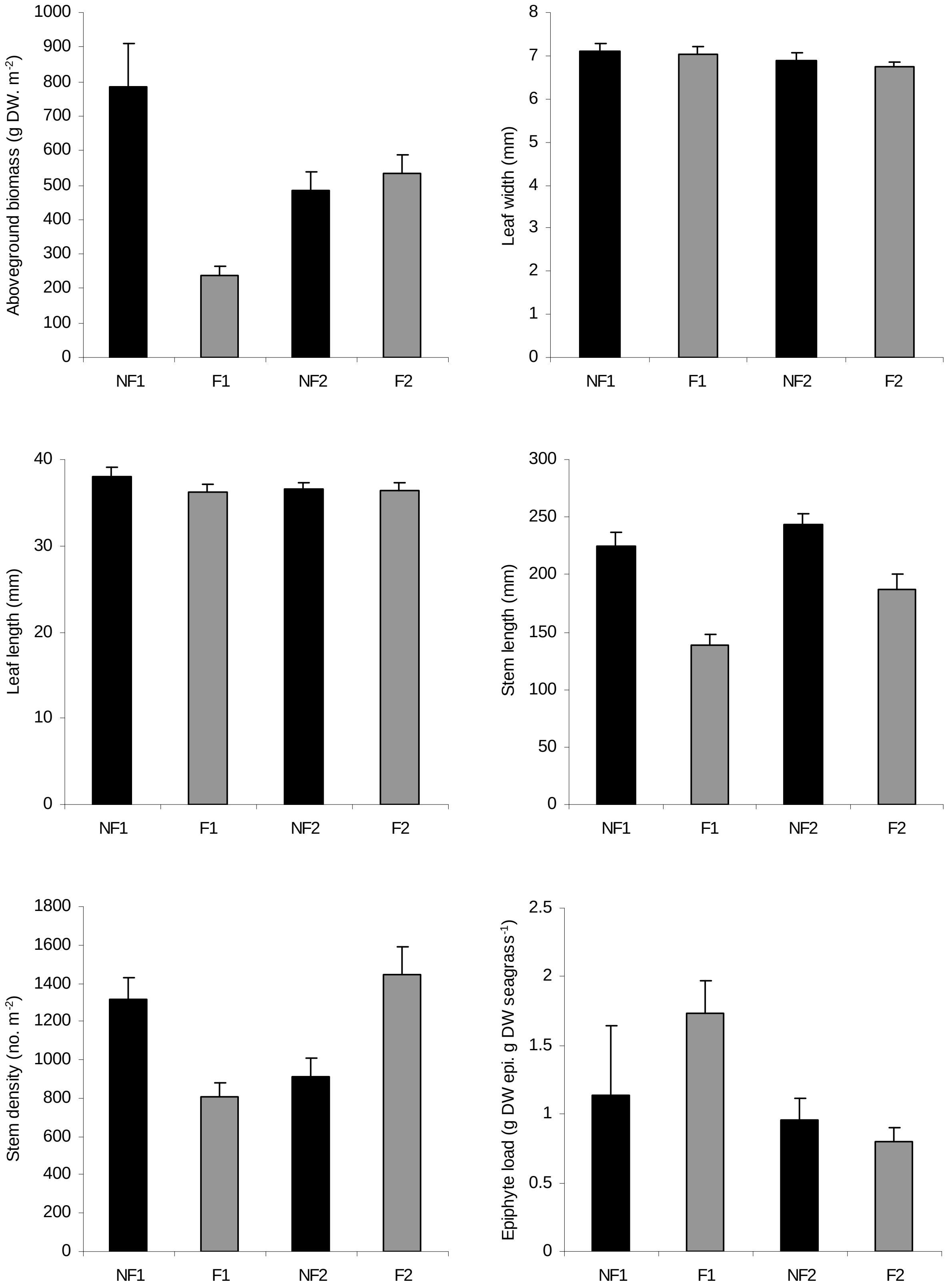

Figure 4.16. Parameters of seagrass morphology (mean + SE, n = 15) for *Amphibolis antarctica* at farm (F, grey bar) and non-farm (NF, black bar) subtidal sites for the land-based abalone farming region of Point Boston.

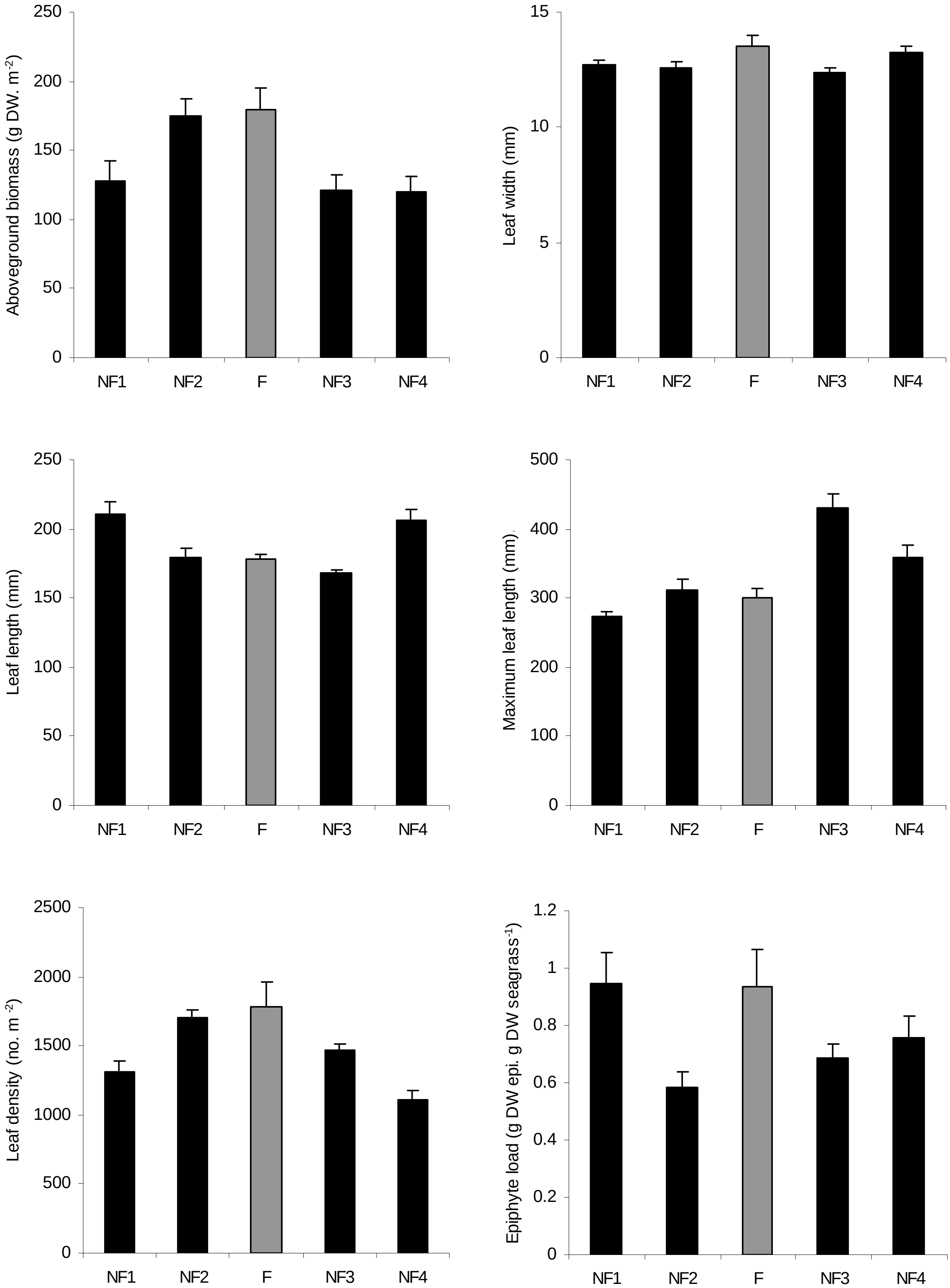

Figure 4.17. Parameters of seagrass morphology (mean + SE, n = 15) for *Posidonia australis* at farm (F, grey bar) and non-farm (NF, black bar) subtidal sites for the land-based abalone farming region of Streaky Bay. n = 10 at F and n = 13 at NF4 for all parameters (except aboveground biomass), n = 14 for NF2 leaf density, n = 12 and 10 for aboveground biomass at NF1 and NF4, respectively.

4.3.2.4 Invertebrates

Belt transects

While a 2-way nested ANOSIM showed no significant difference between farm and non-farm treatments at Smith Bay (Global R = 0.667, P = 0.20), the nMDS plot showed a clear separation of farm and non-farm transects (Figure 4.18). SIMPER analysis showed that 69% of the difference was due to gastropods, which clearly had greatest densities at the two farm sites (Figure 4.19). ANOVA of gastropod density showed a significant effect of farm treatment ($F_{1,10}$ = 57.622, P < 0.001). Closer inspection of the gastropod data showed that the increased numbers adjacent to the Smith Bay farm were not due to escaped farmed abalone (of which none were found) but to increased numbers of *Turbo torquatus*/*undulatus*, which accounted for 92% of all gastropods in the belt transects. Indeed, a total of 257 *Turbo* were found across the 6 farm transects, compared to just 15 across the 9 non-farm transects. No clear patterns of farm versus non-farm sites were evident for the other invertebrate groups at Smith Bay (Figure 4.19), which collectively contributed little (31%) to the SIMPER analysis.

ANOSIM tests showed no significant differences between farm and non-farm groups at Point Boston (Global R = -0.25, P = 1.00) or Streaky Bay (Global R = 0.167, P = 0.082), and the nMDS ordination (Figure 4.18) and data plots (Figure 4.19) generally reflected these statistical outcomes. However, there appeared to be greater gastropod densities at the Point Boston farm sites (Figure 4.19) and ANOVA showed a significant effect of farm treatment ($F_{1,8}$ = 7.797, P = 0.023). As with Smith Bay, the numbers of *Turbo torquatus*/*undulatus* at the Point Boston farm sites contributed to these differences, with a total of 45 found across the 6 farm transects compared to just 8 across the 6 non-farm transects; *Turbo* contributed almost half (46%) of all the gastropods found at Point Boston. However, due to large variation between transects, differences in mean *Turbo* densities between treatments were not statistically significant ($F_{1,8}$ = 4.050, P = 0.079).

The obvious difference in invertebrate communities between Streaky Bay and the other two regions (Figure 4.19) is due to the predominantly soft substrate at Streaky Bay (Figure 4.11), which supports large numbers of the bivalve, *Pinna bicolor*, and relatively low numbers of gastropods and ascidians/sponges. Indeed, no bivalves were scored at Smith Bay or Point Boston. There was no evidence of an increase in greenlip abalone densities adjacent to any of the farms; no greenlip abalone were found at Smith Bay or Streaky Bay, while just five were found at NF1 and one at F1 at Point Boston.

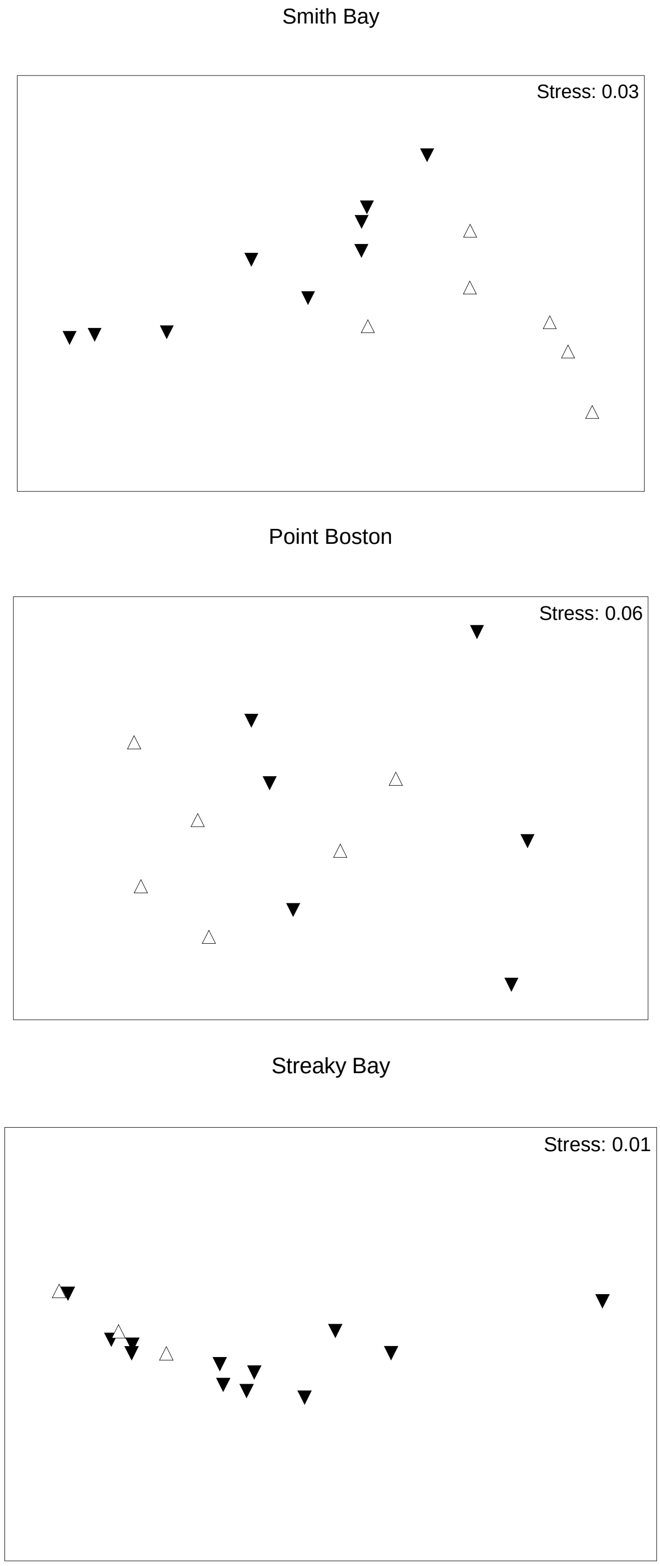

Figure 4.18. nMDS plots of subtidal invertebrate communities documented using belt transects at the three land-based abalone growing regions of Smith Bay, Point Boston, and Streaky Bay. Open symbols represent farm transects, closed non-farm transects.

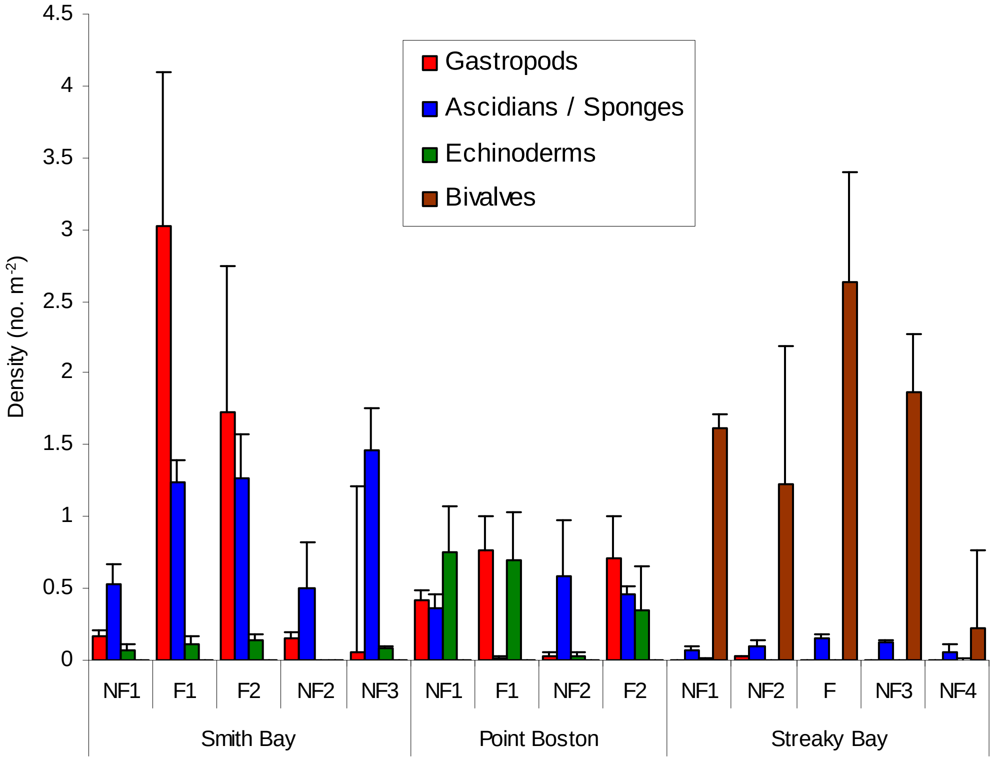

Figure 4.19. Density (mean + SE, n = 3) of invertebrates scored on belt transects at farm (F) and non-farm (NF) subtidal sites for the land-based abalone farming regions of Smith Bay, Point Boston and Streaky Bay.

Quadrats

Invertebrate abundances in the quadrats were generally very low (≤ 20 individuals in total from the 5 quadrats/transect). While an ANOSIM of the Smith Bay data showed no significant effect of treatment (Global R = 0.5, P = 0.20), an nMDS plot indicates some separation of farm and non-farm transects (Figure 4.20) and a SIMPER analysis showed that 75% of the differences were due to gastropods. As with the belt transect data, closer inspection of the quadrat data revealed that *Turbo* abundance was largely responsible for the gastropod differences, with 17 *Turbo* found across the 6 farm transects, and only 1 *Turbo* found across the 9 non-farm transects. An ANOSIM of the Point Boston data showed no significant difference between farm and non-farm treatments (Global R = -0.5, P = 1.00). Due to the sandy substrate type at Streaky Bay, invertebrate quadrats were not sampled there.

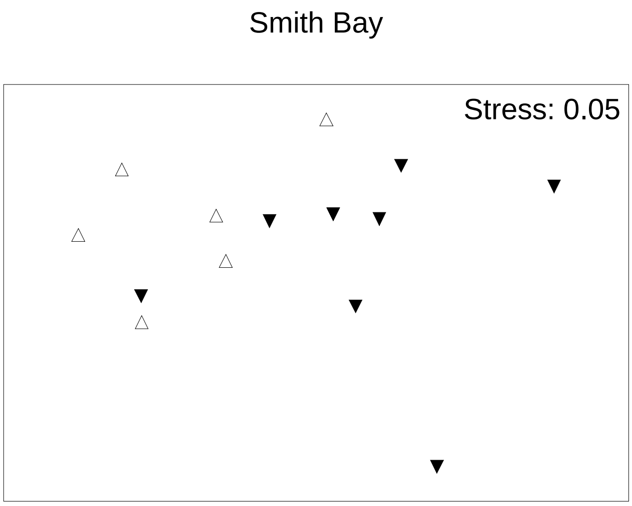

Figure 4.20. nMDS plot of subtidal invertebrate communities documented using quadrats at the land-based abalone growing region of Smith Bay. Open symbols represent farm transects, closed non-farm transects. Note that two of the non-farm transects are lying exactly on top of one another.

4.3.3 Intertidal habitats

4.3.3.1 Area of outfall influence

At Smith Bay, the total area of influence from the main farm outfalls was estimated at 4500 m^2. Based upon an estimated 3600m of coastline in Smith Bay and a 30m-wide intertidal strip, the relative area affected by outfalls is 4.2% of Smith Bay. Due to the complex topography, it was not possible to estimate the spatial extent of outfalls in the Point Boston region. Nonetheless, it is worth noting that at the time of the Point Boston surveys, there were five outfalls operating at F1 and six at F2, which is comparable to the five outfalls at F1/F2 in Smith Bay (Table 4.1). At Streaky Bay, the total area of influence from the only outfall there was estimated at 800m^2.

4.3.3.2 Macroalgae

At all three regions, substantial macroalgal cover was found at sites located directly in farm outfalls, with virtually no cover at sites away from farms or directly adjacent to outfalls (Figure 4.21). At Smith Bay, brown lobed algae (*Colpomenia*), and green membranous algae (including *Ulva*) were found at F1 and the new farm site, with red coralline algae also prevalent at the established farm site (F1). Similarly, at the Point Boston farm sites, brown lobed algae (*Colpomenia*), green membranous algae (including *Ulva*), and red coralline algae were present, with turfing algae also prevalent. Significantly, red coralline algae were highly abundant at F2 (around 55% cover, Figure 4.22), where that group also had a high cover in the subtidal (Figure 4.13). Turfing algae dominated the outfall at Streaky Bay, with a notable lack of red coralline algae. Due to the clear differences between farm and non-farm sites at all three regions, statistical comparisons were deemed unnecessary.

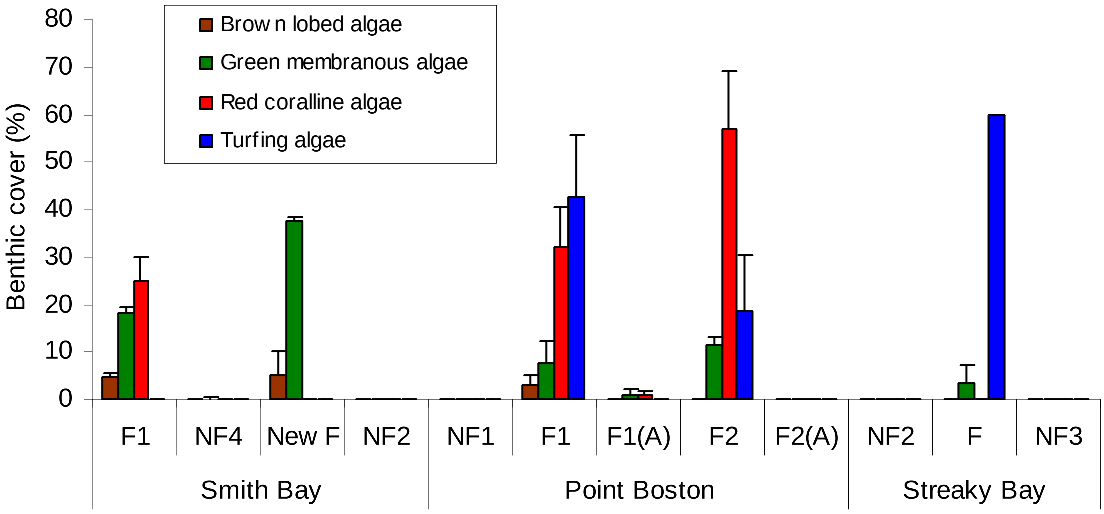

Figure 4.21. Benthic cover (mean + SE) of the four dominant macroalgal groups at farm (F) and non-farm (NF) intertidal sites for the land-based abalone farming regions of Smith Bay, Point Boston, and Streaky Bay. Sites denoted with (A) are adjacent to outfall farm discharges. N = ???

Figure 4.22. Red coralline algae and green membranous algae (*Ulva*) growing in intertidal habitat due to the outfall water from a land-based abalone farm (F2) at Point Boston.

4.3.3.3 Invertebrates

An ANOSIM of the Smith Bay data showed a significant difference between sites (Global R = 0.293, P = 0.016), with pair-wise differences between F1 vs. NewF (P = 0.048), F1 vs. NF4 (P = 0.024), and F1 vs. NF2 (P = 0.016). SIMPER analysis of the Smith Bay data showed that 82% of the difference between farm and non-farm groups was due to *Nodilittorina* (42%) and *Nerita atramentosa* (40%). An nMDS plot of the Smith Bay data supports the statistics, with a clear separation of farm and non-farm plots, but with the two new-farm plots being more closely aligned with the non-farm plots than the farm plots (Figure 4.23). The gastropods *Nerita atramentosa* and *Nodilittorina*, were the most abundant solitary invertebrates at Smith Bay, with much smaller numbers of *Austrocochlea constricta* / *porcata* (other gastropods), anemones, barnacles and chitons present (Figure 4.24). There were substantially less *Nodilittorina* at the F1 site than the other three sites. Anemones were present only at the two farm sites, while barnacles were absent from the F1 site. Chitons were present only at F1.

An ANOSIM of the Point Boston data showed a significant difference between groups (Global R = 0.582, P = 0.001), with pair-wise differences between farm and non-farm groups (P = 0.029) and farm and adjacent-farm groups (P = 0.002). SIMPER analysis of the Point Boston data showed that 82% of the difference between farm and adjacent-farm groups was due to *Nerita atramentosa*. An nMDS plot of the Point Boston data showed a clear separation of the adjacent-farm plots from the farm plots and, except for one plot, the adjacent-farm plots were closely aligned with, but separated from, the non-farm plots (Figure 4.23). Four of the farm plots (2 from F1, 2 from F2) had to be removed in order to perform the analysis, as they contained no invertebrates. *Nerita atramentosa* was by far the most abundant invertebrate (of which all were gastropods) in the Point Boston region (Figure 4.25). However, it was completely absent from the two farm sites (F1, F2) that were directly in the outfall flow. While all other gastropods were found in low densities (<20 m^{-2}), *Lepsiella* and *Nodilittorina* were also absent from F1 and F2.

An ANOSIM of the Streaky Bay data showed no significant difference between sites (Global R = 0.167, P = 0.20), and an nMDS plot supported this conclusion (Figure 4.23), although there was a very large separation between the two farm plots. Two species of limpet (unidentified limpet, *Siphonaria*) were highly abundant across all three sites at Streaky Bay (Figure 4.26). Several other gastropods and an anemone were also present in low densities with no apparent spatial patterns related to farm versus non-farm sites.

Aggregations of small (<20mm in length) mussels were also noted to have some cover at Point Boston and Streaky Bay, but were absent from Smith Bay. At Point Boston, discharges appeared to promote small amounts of mussel cover with 5 and 2% cover at F1 and F2, respectively, and 0% cover at the sites adjacent to outfalls and at NF1. Conversely, at Streaky Bay, mussels had 13 and 21% cover at the two non-farm sites and just 5% cover at the farm site, but these differences were not statistically significant ($F_{2,3}$ = 7.614, P = 0.067).

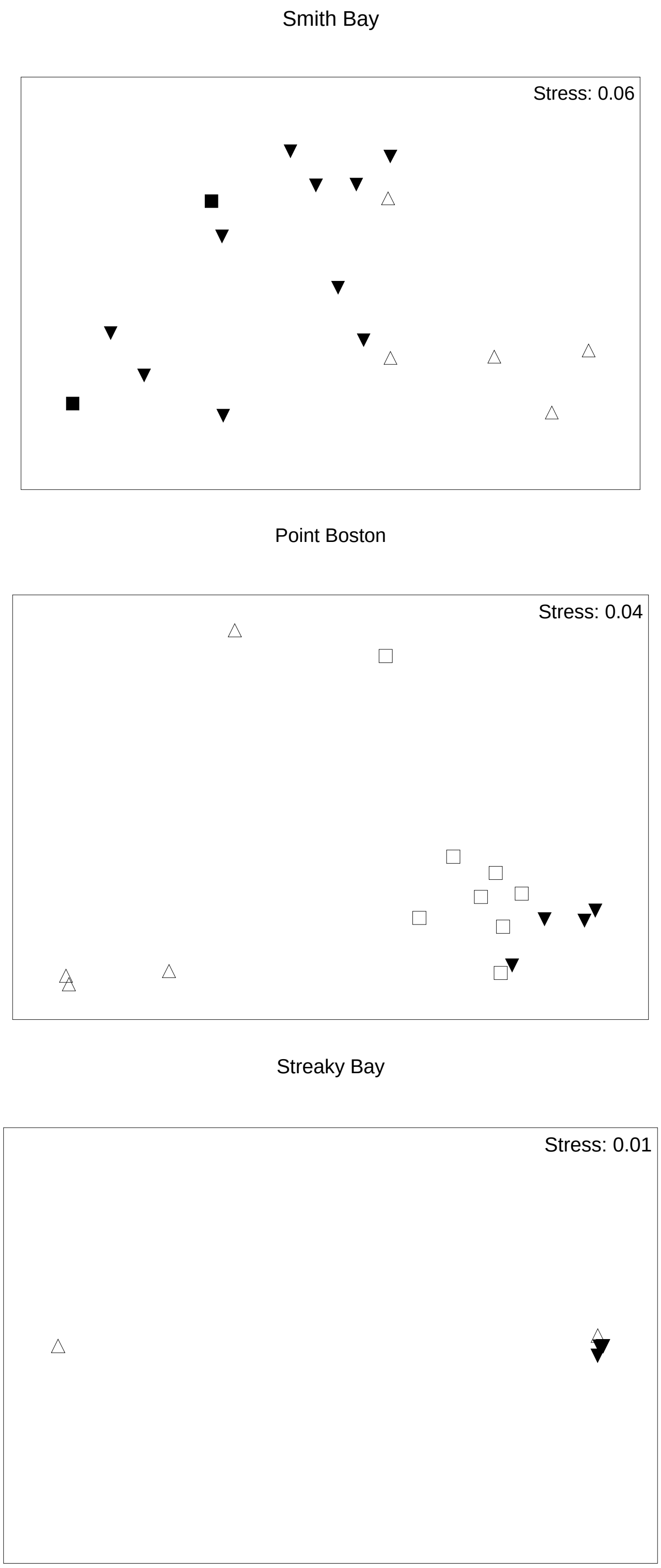

Figure 4.23. nMDS plots of intertidal invertebrate communities at the three land-based abalone growing regions of Smith Bay, Point Boston, and Streaky Bay. Open triangles represent farm plots; closed triangles non-farm plots; closed squares new farm plots at Smith Bay; open squares adjacent-farm plots at Point Boston.

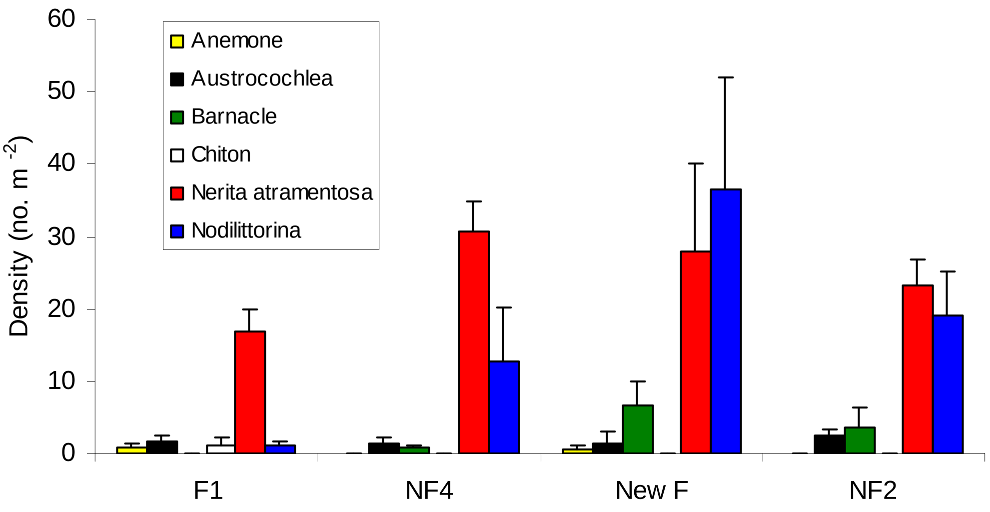

Figure 4.24. Density (mean + SE) of invertebrates at farm (F) and non-farm (NF) intertidal sites for the land-based abalone farming region of Smith Bay. n = 5 for all sites except New F, where n = 2.

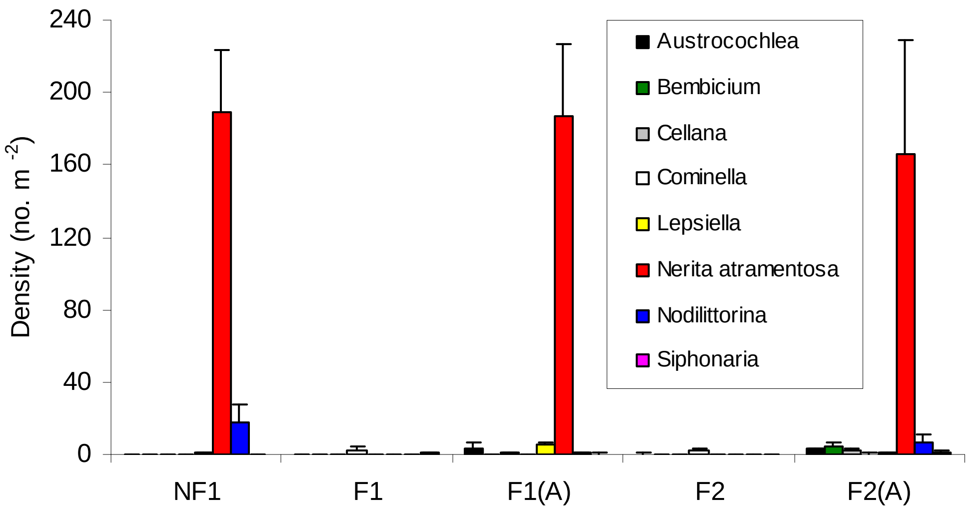

Figure 4.25. Density (mean + SE, n = 4) of invertebrates at farm (F) and non-farm (NF) intertidal sites for the land-based abalone farming region of Point Boston. Sites denoted with (A) are directly adjacent to outfall farm discharges.

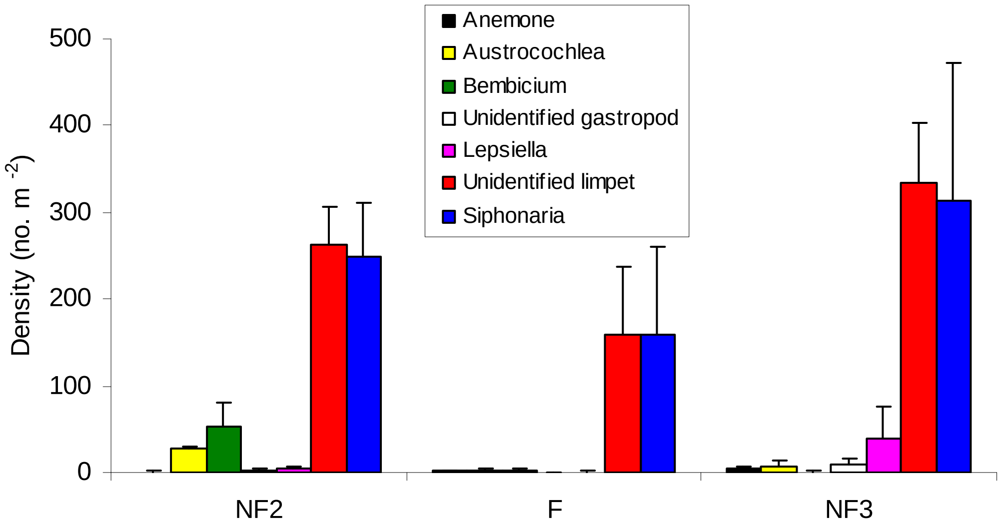

Figure 4.26. Density (mean + SE, n = 2) of invertebrates at farm (F) and non-farm (NF) intertidal sites for the land-based abalone farming region of Streaky Bay.

4.4 Discussion

4.4.1 Nutrients

Nutrient levels in farm drains, outfalls and adjacent intertidal waters were clearly elevated in comparison to non-farm subtidal sites (which are representative of nutrient levels in offshore waters that would be entering the farms). An examination of nutrient concentration data collected by industry as part of their licence-based environmental monitoring programs, also indicates an increase in nutrients when comparing levels leaving the farm (outfall) as opposed to levels in offshore waters entering farms (control; Figure 4.27). Thus, dissolved nitrogen (and to a much lesser extent, dissolved phosphorus) is being produced within the abalone farms and discharged to the adjacent marine environment. However, overall we found only a small detectable increase in dissolved inorganic and organic nitrogen levels in the subtidal waters adjacent to farms at the Smith Bay and Point Boston regions, and virtually no detectable increase adjacent to the farm in Streaky Bay. It is apparent that the nitrogen, which is mostly in the bio-available forms of ammonia and oxidized nitrogen, is rapidly assimilated and/or dispersed. In support of this conclusion are the ASU results, which showed no significant increase in epiphyte load adjacent to any of the farms. However, the nitrogen content of seagrasses was slightly elevated adjacent to the Smith Bay farm, indicating that seagrasses were deriving additional nitrogen from the farm discharges. Similarly, Udy and Dennison (1997) showed that seagrasses close to nutrient sources, such as prawn farms, had a higher nutrient content than those distant from nutrient sources.

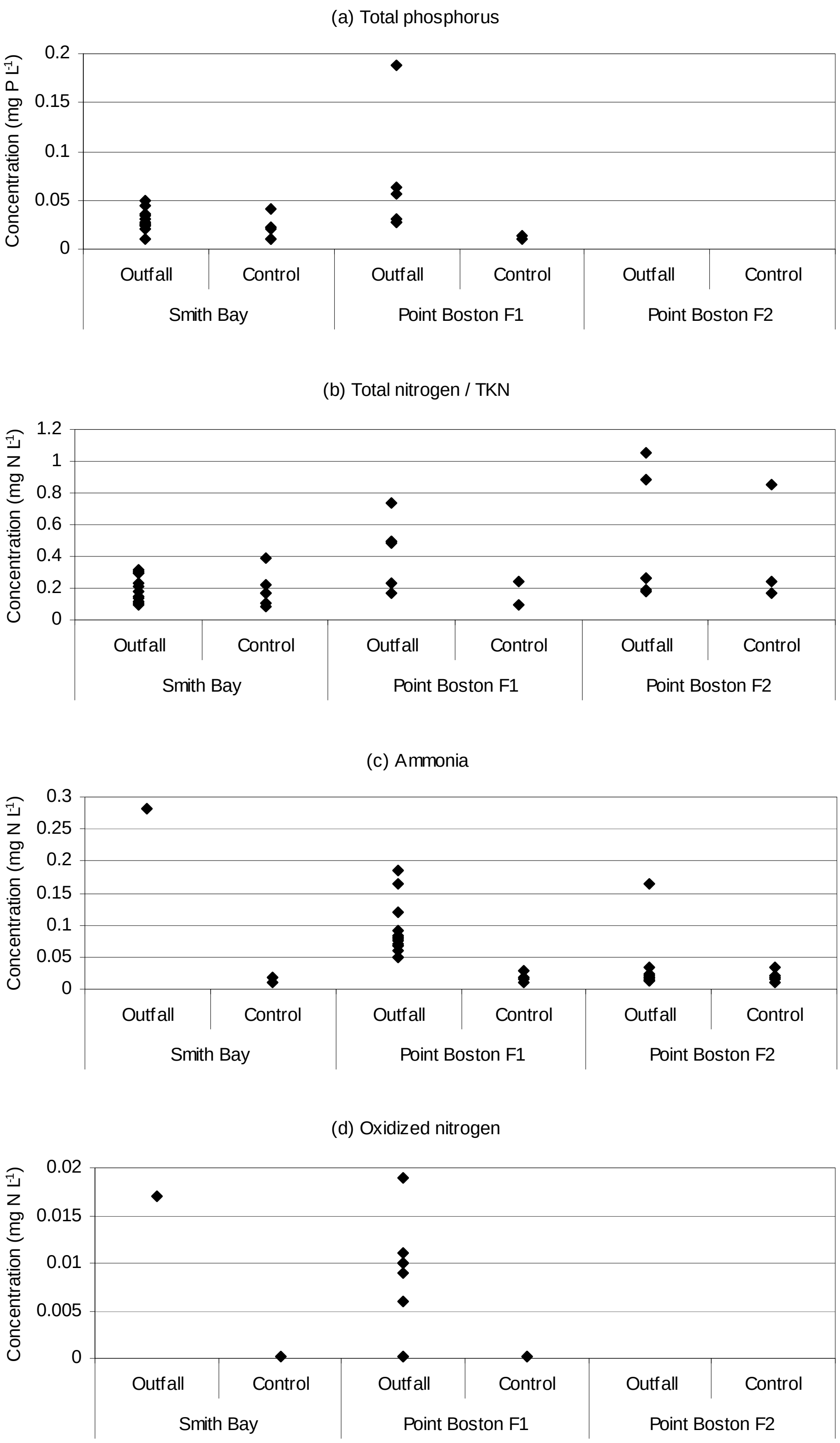

Figure 4.27. Data collected during 2003-2005 as part of the land-based abalone farming licence-based environmental monitoring program.

While nutrients are produced in the farms and discharged to the marine environment, the total annual loads of dissolved nitrogen are relatively low in comparison to some wastewater treatment plant outfalls with similar annual discharge volumes (Figure 4.28). This possibly helps to explain the apparently low level of subtidal impact adjacent to the farms (see later). The annual loads of nutrients associated with particulates from the S.A. abalone farms are unknown. Maguire (1998, cited in Theil et al. 2004) estimated that the total annual nitrogen load (as faeces, solid waste and ammonia) from a farm using 150 T feed was 5.37 T. When Maguire's (1998) value is adjusted to be comparable with the feed inputs of the S.A. farms (see Table 4.1), the resultant estimates of 3.4, 2.0, 1.6, and 0.2 T N yr^{-1} are mostly quite comparable with the estimates of 3.5, 0.3, 1.0, and 0.2 T N yr^{-1}, for Smith Bay, Point Boston F1, Point Boston F2, and Streaky Bay, respectively (Figure 4.28).

Levels of total phosphorus and oxidized nitrogen were significantly elevated on one occasion at Point Boston site F2, which was apparently related to fertilizer inputs from one of the farms. It is possible that pulse events of extremely high nutrients may be more detrimental to the adjacent marine environment than chronically low levels, and the practice of fertilising tanks could perhaps be reviewed. It is worth noting that the macroalgal community of site F2 differed the most from control sites (see below).

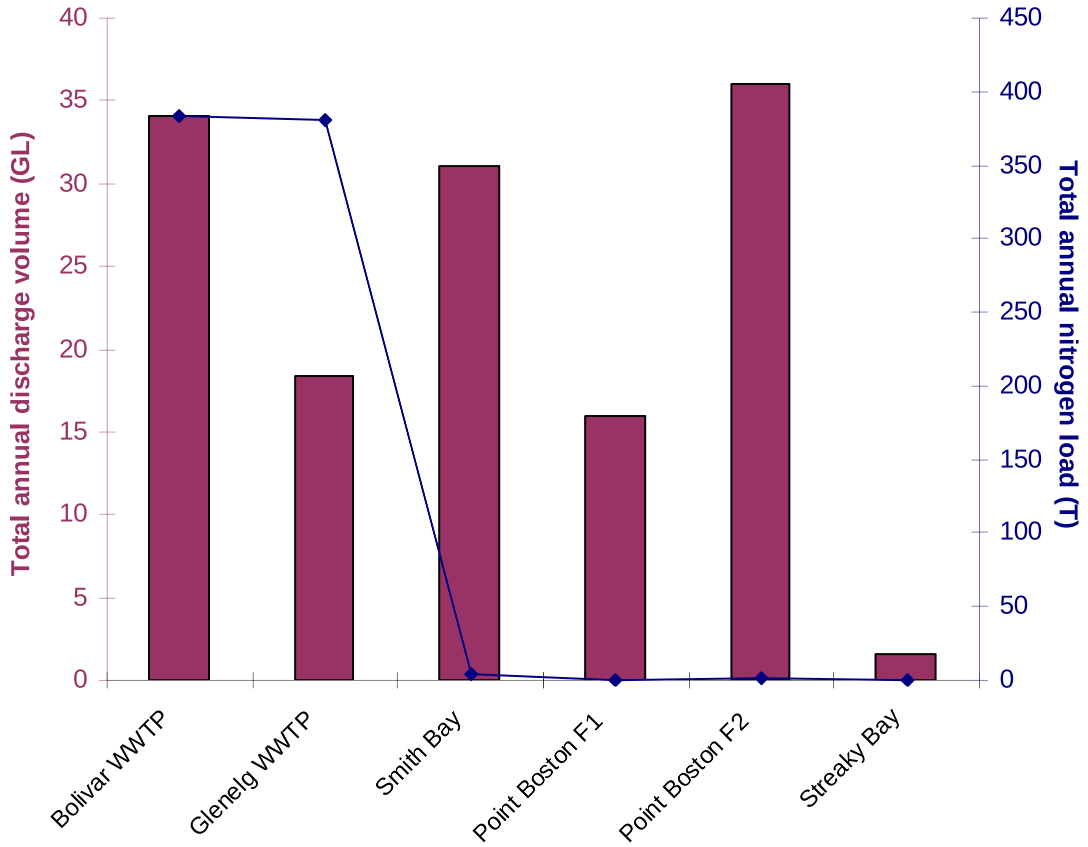

Figure 4.28. Comparison of total annual discharge volumes and dissolved nitrogen loads from two wastewater treatment plants (WWTPs) off Adelaide with the four land-based abalone farms surveyed at Smith Bay, Point Boston and Streaky Bay. Data for the WWTPs were taken from Wilkinson et al. (2003). Nutrient load data for the abalone farms were taken from Figure 4.6. Discharge data for the farms were taken from Table 4.1.

4.4.2 Subtidal reefs

There is growing evidence that elevated water column nutrients can cause a decline in canopy-forming algae with a replacement by turf-forming algae (Gorgula and Connell 2004, Russell and Connell 2005, Eriksson et al. 2006). At Smith Bay, there was no evidence of this having occurred, despite the main farm being in operation for ca.10 years. However, it appears that some macroalgal changes have occurred at Point Boston. In particular, the cover of canopy-forming brown-branching algae was lower and the cover of low-lying red coralline algae higher at farm sites, than at non-farm sites. Significantly, the reef system adjacent to farm F2, where high pulses of nutrients were detected, and which has been operating longer than F1 (Table 4.1), was apparently the most impacted in terms of loss of canopy-forming macroalgae and an increase in red coralline algae. Whilst the red coralline algae that we documented are not strictly turf-forming algae, they have a similar low-lying, smothering habit (Figure 4.22) to turf-forming algae. Interestingly, the cover of turfing algae was not higher at the farm sites, as Fowler-Walker and Connell (2002) found that in South Australia, turfing algae (but not articulated coralline algae) are far more prevalent in open areas than in kelp forests. It is apparent that in the case of F2 at Point Boston, articulated coralline algae (i.e. red coralline algae) have become dominant in both the subtidal (from where canopy-forming algae have disappeared) and in the intertidal (see below).

At both Smith Bay and Point Boston, there was a significantly greater abundance of gastropods (viz. *Turbo*) at the farm sites compared to the non-farm sites. It could be argued that gastropod numbers may vary seasonally, such that the pattern seen at the time of our survey may not always be apparent. However, the work of Andrew and Underwood (1989) suggests that turbinid densities vary little through time (3 years). Thus it is likely that the patterns we observed are a reflection of long-term change. It is unclear why the farm discharges had apparently caused an increase in *Turbo* density. Clarkson and Shepherd (1985) studied the diet of *T. torquatus* and *T. undulatus*, reporting that they feed mainly on geniculate coralline algae and macroalgae. At Point Poston the cover of red coralline algae (a form of geniculate coralline algae) was increased at the farm sites relative to the non-farm sites, but at Smith Bay, this was not the case. Increases in *Turbo* numbers might be related to sediments in the form of uneaten feed and faeces from the farm, as *Turbo* has a wide range of diet items (Clarkson and Shepherd 1985) and they were often found near the base of boulders in areas where organic material appeared to be accumulating. A study by Terlizzi et al. (2005) on temperate reefs in the Mediterranean found that molluscan assemblages were different between a site adjacent a sewage outfall and two control sites, and they speculated that the changes could have been driven by increased sedimentation or changes in habitat complexity. Thus mechanisms that reduce sediment loads from land-based abalone farms to the adjacent marine environment may be beneficial in this regard, but this requires further research. It is also possible that nutrient inputs caused an increase in benthic microalgae, and thus an increase in food availability. In this regard it is worth noting that the LIT method used was not designed to detect benthic microalgae on subtidal rock, and that intertidal rocks associated with farm outfalls did have a slippery covering of microalgae (Figure 4.29). However, *Turbo* do not appear to be well adapted to radula rasping of hard surfaces (Clarkson and Shepherd 1985), and it is more likely that they were feeding on other items, including drift material.

Figure 4.29. Gastropods (mainly *Austrocochlea constricta*) feeding on algal-covered boulders in farm outfall water at Smith Bay.

Several workers have shown the importance of interactions between macroinvertebrate grazers and canopy-forming macroalgae (e.g. Fletcher 1987, Edgar et al. 2004, Russell and Connell 1985). At Point Boston, changes in both macroalgal and macroinvertebrate (gastropod) communities were apparently associated with farm discharges. However, the work of Fletcher (1987) indicates that turbinids do not play a major role in controlling macroalgal cover, and this appears to be particularly so in southern Australia where grazing by macroinvertebrates is less effective than in eastern Australia (Fowler-Walker and Connell 2002). Furthermore, at Smith Bay where there was a marked increase in gastropod abundance associated with the farm discharges, no change in the macroalgal community was detected. Thus the collective observations from Point Boston and Smith Bay indicate that the changes in macroalgal cover observed at Point Boston were not due to increased macroinvertebrate herbivory on canopy-forming algae. More likely, the macroalgal changes were due to increased nutrients.

There was no evidence of increased greenlip abalone densities adjacent to farms. As greenlip abalone have relatively small (10’s to 100’s of metres) home ranges (Shepherd 1986, Shepherd and Godoy 1989), if substantial numbers were successfully escaping and surviving, then densities might be expected to be higher adjacent to farms. This was not the case. At Smith Bay and Point Boston, empty shells of abalone were often found in the intertidal region adjacent to the farms and, due to their small size and shell colour, could be identified as originating from farms. However, it is apparent that these escapees do not survive the journey across the intertidal and/or their survival rate is very low if they do make it to subtidal waters. It is highly likely that predation by birds prevents most of the escapees from reaching the adjacent marine environment. At Streaky Bay, the presence of a settlement pond that contains predatory fish and is also frequented by seabirds, would also act as a barrier to the escape of farmed abalone. Indeed, no empty shells were observed in the adjacent intertidal area at Streaky Bay. Furthermore, even if escapees did reach the adjacent subtidal environment at Streaky Bay, it is not particularly suited to greenlip abalone, being predominantly sand and seagrass.

4.4.3 Subtidal seagrasses

The seagrasses, *Amphibolis* and *Posidonia*, are slow growing, long-lived perennial species. Thus any previous environmental perturbations are likely to be manifested over time in the cover and aboveground biomass of these two genera. If seagrass loss had occurred adjacent to farm sites, then the cover of bare sand might be higher there. However, none of the cover data suggest that losses of *Amphibolis* or *Posidonia* have occurred.

At Smith Bay, there was some evidence that the biomass of *P. angustifolia* was being affected by the farm discharges. However, because both species were not present in sufficient numbers for comparison of multiple farm and non-farm sites together, any conclusions about the effect of the Smith Bay farm on seagrass morphology should be treated cautiously. It is also possible that the apparently sublethal impact on seagrass morphology at Smith Bay could be seasonal and may not lead to long-term loss; Wear et al. (2006) found a seasonal pattern of impact and recovery of *Posidonia* adjacent to drain discharges in the SE of South Australia.

4.4.4 Intertidal reefs

The farm outfalls have essentially transformed an intertidal habitat into a subtidal one. Within the intertidal area covered by outfall water, substrates have become covered with various types of algae. We also noted that sediments had been trapped in red coralline and turfing algae within the outfalls at all three regions. It is unknown whether the cover of algae in the intertidal is related to nutrients from the farms, or simply to permanent immersion. Experimental work by McGuinness and Underwood (1986) indicates that algal composition on intertidal boulders can be influenced by the period of emersion. Significantly though in our study, the red coralline algae found in the outfalls at Point Boston were also found in the adjacent subtidal waters, and their occurrence appears to be related to nutrients. Furthermore, the green membranous macroalgae (such as *Enteromorpha* and *Ulva*) that were found in the intertidal, are fast-growing ephemeral species known to respond to increased nutrient levels (Karez et al. 2004), including rocky intertidal shores affected by sewage outfalls (e.g. Fairweather 1990). While the intertidal macroalgae found adjacent to farm outfalls would almost certainly be deriving nutrients from the farm discharges, it is unclear to what extent they modify nutrient concentrations before they enter the subtidal environment. The results of the water quality surveys showed little variation between samples taken at outfalls (before water enters the intertidal) and samples taken at the intertidal (just before outfall water mixes with subtidal water, Figure 4.5). However, it must be noted that metabolic wastes from invertebrates and other biota associated with the outfalls (see below) would also affect nutrient concentrations in the intertidal region.

The gastropod communities in the intertidal have also changed due to the farm outfalls, probably to those species that can tolerate subtidal conditions, although this appears to depend on the topography of each site. For example, at Smith Bay, *Nodilittorina* species that prefer the high intertidal (Edgar 1997) were found in substantially lower densities at the farm site when compared to the three non-farm sites. However, *Nerita atramentosa* and *Austrocochlea constricta* / *porcata*, which can all survive in the mid intertidal (Edgar 1997), were found in comparable densities at all four sites at Smith Bay. This is probably related to the boulder habitat of the Smith Bay region, which allows these gastropods to escape permanent immersion (Figure 4.29). The situation at Point Boston was different to Smith Bay with surveys occurring on flat rock faces. In this instance, there was a notable absence of *Nerita atramentosa* and *Austrocochlea* at the outfall sites, which were completely submerged. It was apparent that the zone of influence and change due to the outfall water was highly localised at Point Boston as the communities directly adjacent to outfalls (F1(A), F2(A)) were significantly different to those in outfalls (F1, F2) and were more closely aligned with those at a control site (NF1) that was well away from the farming area. At Streaky Bay, we detected no pattern of invertebrate communities related to outfall flow.

While the area of true intertidal habitat was reduced at the farm outfalls, our surveys did not indicate that the density of intertidal gastropods was also reduced. In fact more extensive sampling of the intertidal area may well show an increase in gastropod abundance around farm outfalls as the method that we used placed quadrats at the lowest vertical point possible to sample emersed substrate, yet it was apparent that most of the gastropods in the outfall areas were emmersed on the tops of boulders (e.g. Figure 4.29). This is probably due to the increased abundance of macro- and micro-algae that would provide a food source (note that microalgae growing on rocks were not quantified but, as with macroalgae, were also likely to more abundant in the outfalls), and to the ecophysiological tolerances of different gastropod species (see above). Indeed, at Smith Bay, large aggregations of gastropods (mainly *Austrocochlea constricta* and *Nerita atramentosa*) were noted to be clearing algae in and around exposed sections of boulders that were not permanently covered by outfall water (Figure 4.24, foreground of Figure 4.1). At Smith Bay we sampled within outfalls from a new farm (New F) and found that these plots had macroalgal communities similar to the established farm site (F1) but invertebrate communities similar to non-farm sites (NF4, NF2). Thus it appears that the intertidal communities at the new farm site were undergoing a transition due to the outfall.

4.4.5 Seagrass scouring around intake pipes

Qualitative observations revealed no evidence of seagrass scouring around any of the intake pipes surveyed. At Smith Bay, there was a mixture of macroalgae (including *Cystophora*) and seagrass (*Amphibolis*, *Posidonia*) growing directly adjacent to, and (in the case of macroalgae) over, the pipes. Some small sand patches were noted near the pipes, but these could not be termed 'blow-outs'. At Streaky Bay, where the pipes are clearly raised from the bottom on concrete blocks, *Posidonia* was growing directly adjacent to, and under, the pipes in the nearshore region where seagrasses occurred. Further offshore, the benthos changed to macroalgae.

4.5 Conclusions

- Discharge waters contain elevated levels of dissolved nutrients that can be detected in adjacent intertidal and subtidal waters, and in the nitrogen content of subtidal seagrass (*Posidonia*).
- Discharge water transforms intertidal habitats into subtidal habitats that have vastly altered communities to adjacent intertidal areas. As well as water flow and cover, nutrients and sediments are probably also contributing to the communities found in the discharge areas with green membranous algae (such as *Ulva*), red coralline algae and turfing algae present. The spatial extent of this impact is restricted to the areal cover of discharge water, which is relatively small compared to the areal cover of intertidal habitat in each farming region. However, intertidal invertebrates do still exist in the outfalls when boulders and rocks are available to provide vertical relief from continual immersion, e.g. some intertidal gastropods (such as *Austrocochlea* and *Nerita*) which probably thrive due to the algal growth stimulated by the outfall water.
- Due to the lack of data before farms commenced discharging, it is not possible to conclude whether any differences observed between farm and non-farm sites are due to farm discharges. Furthermore, the likelihood of detecting differences was reduced by the limited number of farm and non-farm sites surveyed in each region and the cost-limited scope of the biological surveys. Nonetheless, some site-specific and subtle differences were detected adjacent to the abalone farms that are consistent with the known effects of increased nutrients on subtidal marine communities, viz. some evidence of a negative impact on seagrass (*Posidonia*) at Smith Bay, strong evidence of a negative impact on canopy-forming macroalgae at Point Boston, and a greater gastropod (viz. *Turbo*) density at Smith Bay and Point Boston. Importantly though, no major changes to subtidal communities have occurred due to farm discharges, with diverse communities of macroalgae, seagrasses and invertebrates still present directly adjacent to farm outfalls. The spatial extent of the apparent subtidal changes at Smith Bay and Point Boston is unknown.
- The apparent low level of impact in subtidal waters adjacent to farms is probably related to the relatively low nutrient concentrations and low total annual nutrient loads. For example, even though the annual discharge volume of two of the sites (Smith Bay F1, Point Boston F2) is equivalent to a major WWTP outfall off Adelaide (Bolivar) that has been linked to major

seagrass decline (Shepherd et al. 1989), the total annual loads of nitrogen at the farm sites are 100x lower (Figure 4.23).

- Highly elevated nutrient levels do occur when fertilizer is added to the farms and these pulse events could be more detrimental to the marine environment than chronic low doses. The practice of fertilizing nursery tanks could be better controlled, as it is apparent that a large amount of the fertilizer is being wasted. It may also be possible to divert outfall water through a treatment system (e.g. settlement pond, nutrient stripper) at times of fertiliser addition.
- Escaped abalone are having no impact on the adjacent marine environment, as they apparently do not survive once they leave the farm outfall pipes.
- There was no evidence of seagrass scouring around intake pipes at the two farms investigated.

4.6 Acknowledgments

Thanks to: Dr Jason Tanner for advice and support; Dr Stephen Madigan and Dr Peter Lauer (PIRSA Aquaculture) for constructive feedback and input; Dr David Currie and Dr Greg Collings (SARDI) for statistical assistance; Dr David Turner (SARDI) for assistance with the sampling design, some of which was based on the Reef Health Program; Gen Mount, Kylie Johnson, Emma O'Loughlin and Jodi Lill (SARDI) for laboratory analyses; and Dr Milena Fernandes (SARDI) for intellectual input. PIRSA Aquaculture and FRDC are thanked for their financial support.

Special thanks to the land-based abalone farming industry members, particularly Bruce Green, Tom Hyde, John Hall, and Roger Goodenough.

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Chapter 5: Impacts of BST long-line oyster aquaculture on epibenthic and infaunal communities at South Spit, Stansbury

Simon Bryars, Mandee Theil, and Keith Rowling

5.1 Introduction

Coastal habitats are under threat worldwide from anthropogenic disturbances, including aquaculture operations. Thus it is imperative that the effects of aquaculture are clearly understood and that, if impacts are evident, then appropriate mitigation strategies are employed. In South Australia (SA), there are numerous Pacific oyster (*Crassostrea gigas*) farms situated along the coast in sheltered embayments that have good water circulation. Successful culture of oysters in SA began in 1970 (Olsen 1994) and has steadily increased in size and production since that time. In 2004/05, over 55 million adult oysters were produced, with a combined value (including spat) of $21.19M (Econsearch 2006). Oysters are grown mainly over intertidal sand and seagrass habitats using baskets suspended on a ‘rack and rail’ system or on Baker-Schultz-Turner (BST) long-lines (Figure 5.1). Oyster growers visit farms periodically via small vessels whereupon the growers may tend the baskets by walking directly on the tidal flats. The oysters are filter feeders and, as such, no feed inputs are associated with their farming.

During 2003/2004, a formal environmental risk assessment was conducted on the intertidal oyster aquaculture industry in South Australia (Wear et al. 2004; see Chapter 1). From the workshop part of the risk assessment, 41 of the 51 issues that were discussed relating to the effects of farming on the environment, were given a negligible or low ranking. The remaining 10 ‘priority’ issues were given a risk ranking of moderate. No issues were rated as high or extreme risk. After reviewing the literature, Wear et al. (2004) changed most of the 10 priority issues to a low or low-moderate ranking. Nonetheless, Wear et al. (2004) identified that further research is required on the effects of physical disturbance and shading on the nearby environment. Consequently, we proposed to conduct biological surveys at several key South Australian farming regions in order to determine if any impacts have occurred since the commencement of the industry. However, a change in priorities of the Audit Project (FRDC No 2003/223) that was driven by the project steering committee meant that only one region (Stansbury) was investigated. In particular, possible effects on *Posidonia* seagrass due to shading by BST long-lines and trampling by growers were investigated at Stansbury (note that no rack and rail systems are currently in operation at that location). As there are no quantitative data prior to the oyster farms starting, we examined spatial patterns of *Posidonia* around oyster farming structures that may indicate an effect of shading and/or trampling. As *Posidonia* is a long-lived, slow-growing seagrass, any unusual patterns of loss and/or degradation may be indicative of long-term change. We also examined the distribution of the razorfish (*Pinna bicolor*), which is a bivalve that lives in the intertidal and has a large section of its shell exposed above the seabed, thus making it vulnerable to breakage from trampling. Under the direction of PIRSA Aquaculture, one other aspect of oyster farming that we specifically investigated at Stansbury was the possible effect on the marine environment (viz. benthic infauna) of Copper Chromium and Arsenate (CCA) treated timber which is used for oyster racks and posts (see Wear et al. 2004).

Figure 5.1. Baker-Schultz-Turner (BST) designed long-line culture of oysters over dense *Posidonia* seagrass at South Spit, Stansbury, South Australia.

5.2 Methods

5.2.1 Field site

Epibenthic and infaunal surveys were conducted during April 2005 on South Spit at Stansbury, South Australia (Figure 5.2).

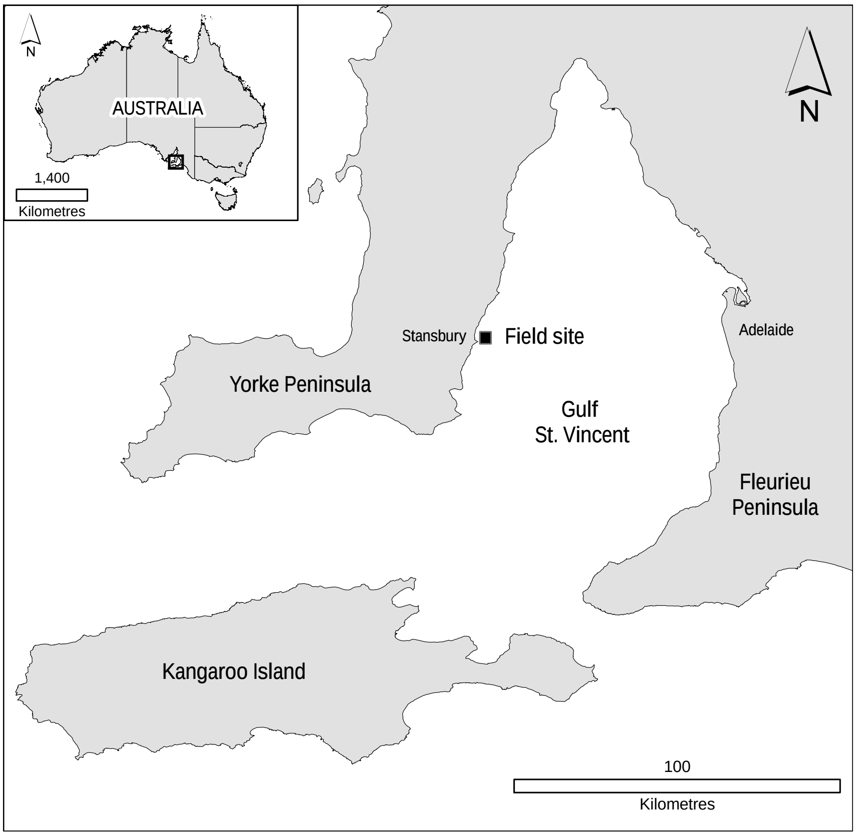

Figure 5.2. Location of the field site on South Spit at Stansbury, South Australia.

5.2.2 Epibenthic survey

At each of three sites (A, B, C) separated by <500m, three replicate linear 50m transect lines were laid out for each of four treatment types: 'Outside' of long-lines, 'Between' long-lines, 'Adjacent' to long-lines, and 'Under' long-lines (Figure 5.3). It was anticipated that the Between and Outside treatments would act as suitable controls where shading and trampling did not occur. At each site, the Between, Adjacent and Under transects were associated with three adjacent pairs of long-lines in an area of the lease that had apparently uniform seagrass cover. A more randomised method of selecting transects could not be used due to the inherently patchy nature of seagrass on South Spit. The Outside transects were laid about 20m apart in an area <100m away from the long-lines that was either within or just outside the oyster lease but was free of farming structures. Each of the treatment transect lines formed the basis of the survey. At Site A, long-lines were oriented E-W and Adjacent and Under transects were aligned with the northern side of the paired lines (Figure 5.3), with the hope that potential effects of trampling and shading could be separated, as the level of shading adjacent to the northern paired line would be much lower than that for the southern paired line. At Sites B and C, long-lines were oriented N-S and Adjacent and Under transects were aligned with the western (Site B) and eastern (Site C) sides of the paired lines. Long-lines within each pair were ~2m apart (Figure 5.1), while the distance between pairs of lines was around 10-15m.

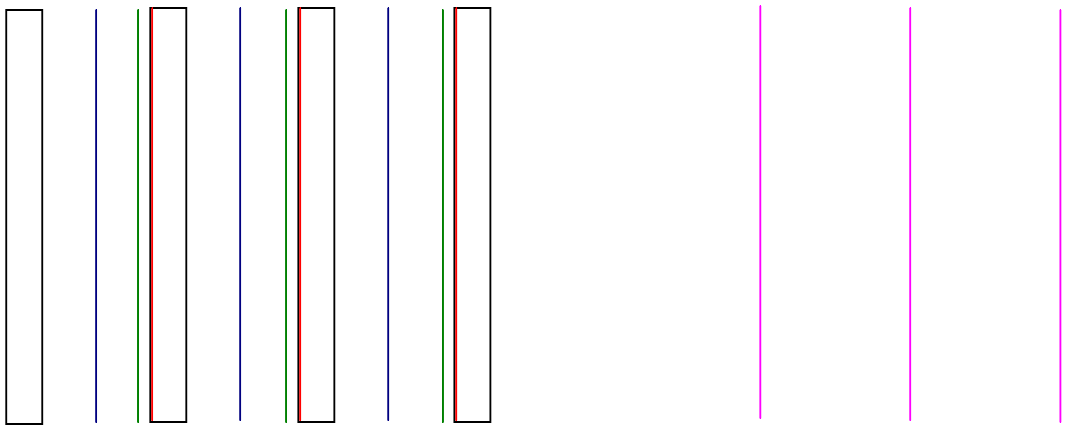

Figure 5.3. Schematic showing the spatial arrangement of the three replicate transects for each of the four treatments used for the intertidal survey of BST long-line oyster culture at Stansbury, South Australia. Black rectangles represent paired rows of long-lines; blue lines are Between treatments; green lines are Adjacent treatments; red lines are Under treatments; and pink lines are Outside treatments. See text for further details.

A point-intercept-transect (PIT) method using 1m intervals was used to score seagrass species (*P. australis*, *P. sinuosa*) and bare sand, enabling calculation of percent cover of *Posidonia*. A 1m wide, belt transect running parallel to the main transect (Figure 5.4) was used to quantify live razorfish (*Pinna bicolor*). A set of five 250x250mm quadrats (0.0625m^2) randomly positioned along each transect was used to quantify *Posidonia* leaf density *in situ*. One to two shoots (with several leaves attached) were harvested from a designated corner of each quadrat for later confirmation of species (*P. australis*, *P. sinuosa*) and measurement of leaf length and width. If both seagrass species were observed in a quadrat then shoots of both species were harvested. For the duration of the survey, two pairs of Odyssey photosynthetically active radiation (PAR) (400-700 nm) cosine type light meters (Dataflow Systems, New Zealand) were deployed on the seabed at Site A; one pair directly underneath baskets on a long-line and one pair between long-lines. Meters measured and logged light intensity at 30-minute intervals.

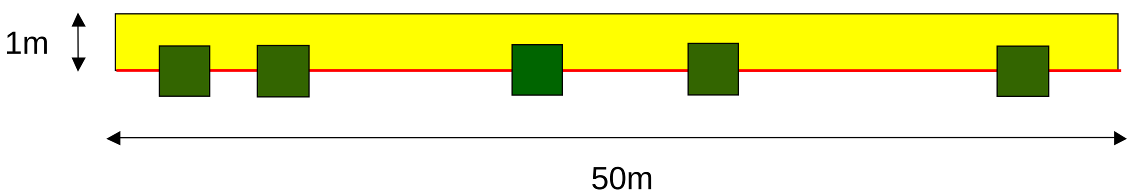

Figure 5.4. Schematic of techniques used in the epibenthic survey. The red line represents the point-intercept-transect line, the yellow area represents the belt transect, and the squares represent the five randomly placed quadrats.

5.2.3 Infaunal survey

At Site B that was used for the epibenthic survey, two sets of sediment samples were collected to examine the possible effects of timber posts on infaunal communities. One set of samples ('Post' treatment) was collected within 50mm of posts, and the other ('Control' treatment) was collected halfway between the rows of long-lines, as in the 'Between" treatment of the epibenthic survey (Figure 5.3). For the Post treatment, 8 replicate samples were collected from the first 8 posts in one row of a long-line from the northern end, such that samples were ~3m apart. For the Control treatment, 7 replicate samples were taken randomly (minimum distance apart was set at 1m) along a 25m transect laid adjacent to the Post treatment. Sediment core samples were collected with a 400mm long section of 40mm inner diameter PVC pipe inserted 200m into the substrate. Cores were stored in plastic jars filled with seawater and 5% formalin. Each core was later washed through a 1mm sieve and the retained infauna were then sorted, identified and enumerated.

5.2.4 Data analyses

As some quadrats contained mixed assemblages of *P. australis* and *P. sinuosa*, which are known to have different leaf morphologies (Robertson 1984), separate analyses for leaf length and width were performed for each species. This separation of species meant that datasets of leaf width and leaf length were unbalanced as not all transects/sites contained both species. Furthermore, information on individual quadrats was deemed unimportant for the leaf length/width data, such that mean values were calculated from all leaves collected and measured in the relevant quadrats along each transect, and this value was used in subsequent analyses. Analyses of seagrass cover and leaf density did not discriminate between the two species.

A two-way ANOVA was used to test for differences in seagrass cover, leaf length, leaf width, and *Pinna* density between Treatments (Outside, Between, Adjacent, Under) and Site (A, B, C) with the following model: Treatment + Site + Treatment * Site, where Treatment was fixed and Site was random. The following model was used for leaf density to account for variation between quadrats within transects: Treatment + Site + Treatment * Site + (Transect(Treatment* Site)), where Treatment was fixed and Site and Transect were random. If a significant interaction of Treatment x Site was found, then separate 1-way ANOVAs on Treatment were used for each site.

All ANOVA tests were conducted using the software package SPSS (ver 14.0, SPSS Inc., Chicago). Before conducting an ANOVA, the assumption of homogeneity of variances was tested using Levene's test and the normality of data was visually assessed using QQ plots. When assumptions were not met, data were transformed in an attempt to meet the assumptions. Where this failed, analyses were performed on untransformed data using a conservative alpha level of 0.01, instead of 0.05 (Underwood 1981). As is appropriate for percentages (or proportions), seagrass cover data were arcsine transformed according to Zar (1984) prior to analyses. While testing of the seagrass morphological variables (leaf density, leaf width, leaf length) would ideally be done with a multivariate technique, this could not be achieved because (1) the datasets were unbalanced and the non-parametric testing procedure of PERMANOVA (Anderson 2001, 2005) in its present form cannot deal with this scenario, and (2) parametric MANOVA in SPSS requires all factors to be fixed, which affected outcomes greatly. Thus, separate ANOVAs were conducted on each of the variables.

Differences in infaunal communities were examined using Bray-Curtis dissimilarity measures (Bray and Curtis 1957). This measure was chosen, as it is not affected by joint absences (noting that many zero values were present in our dataset). Differences in Bray-Curtis dissimilarity measures were tested using the analysis of similarities (ANOSIM) routine of Clarke and Gorley (2001), while spatial patterns were plotted using a non-metric multi-dimensional scaling (nMDS) algorithm. In interpreting nMDS ordinations, if stress values are <0.1, then this corresponds to a good ordination with no real prospect of a misleading interpretation (Clarke and Warwick 1994). Infaunal data were square root transformed prior to analyses. An ANOSIM of treatment (Post, Control) was conducted using all samples as replicates. ANOSIM and nMDS analyses were performed with the software package PRIMER (ver 5.2.9, Primer-E Ltd, Plymouth).

5.3 Results

5.3.1 Shading by baskets

Patterns of underwater light intensity over a 2-day period at Site A indicated a depression in light intensity reaching the seabed during the afternoon (Figure 5.5). This pattern is consistent with the East-West alignment of the long-lines at Site A and the sun moving to the west such that the baskets cast a shadow over the seabed directly underneath the baskets, where the loggers were sited, only during the afternoon (Figure 5.6).

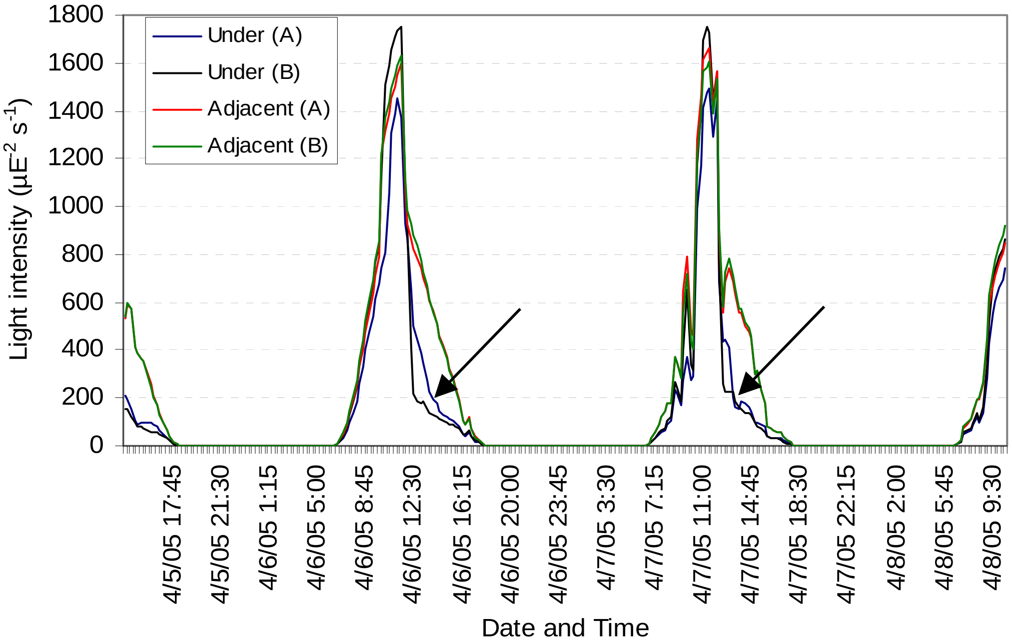

Figure 5.5. Light intensity from pairs (A, B) of light loggers deployed for two full days under and adjacent to a Baker-Schultz-Turner (BST) designed oyster long-line stocked with baskets at South Spit, Stansbury, South Australia, during April 2005. The arrows indicate a depression in light intensity reaching the seabed during the afternoon.

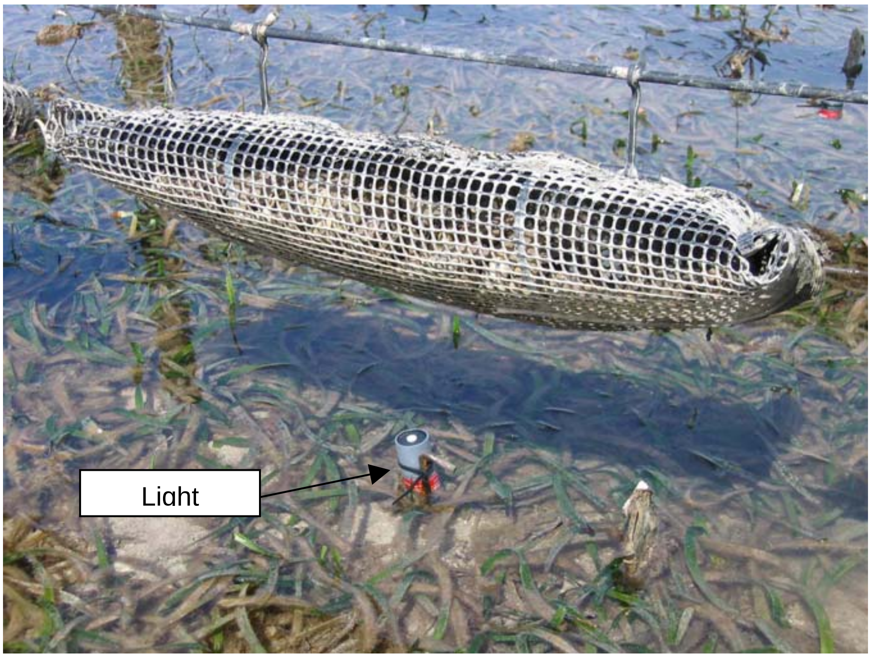

Figure 5.6. Baker-Schultz-Turner (BST) designed oyster long-line stocked with baskets at South Spit, Stansbury, South Australia, showing a light logger used to measure seabed light intensity at Site A. Note that the positioning of the shadow (photographed here at 12.20pm) is not directly underneath the basket where the logger is located, but is to the south of the logger due to the angle of the sun and the East-West alignment of the long-lines.

5.3.2 Seagrass cover

There is some indication of an effect of treatment on seagrass cover at two of the three sites (Figure 5.7). Seagrass cover was lowest at the Under treatments within both Sites B and C, with about 15% less cover under long-lines than between long-lines for the two sites. As the 2-way ANOVA of seagrass cover showed a significant interaction of Treatment x Site ($F_{6,24} = 2.624$, $P = 0.042$), separate 1-way ANOVAs of Treatment were run for each site. There was no significant effect of Treatment for Site A ($F_{3,8} = 0.185$, $P = 0.903$), but a significant effect at Site B ($F_{3,8} = 5.956$, $P = 0.020$) and Site C ($F_{3,8} = 6.151$, $P = 0.018$). Post-hoc comparisons showed that the differences at Site B lay between the Under treatment and the other three treatments (LSD test: Under < Outside, $P < 0.016$; Under < Between, $P < 0.004$; Under < Adjacent, $P < 0.020$). At Site C, the differences lay between the Under treatment and the Outside (LSD test: Under < Outside, $P = 0.007$) and Between treatments (Under < Between, $P = 0.016$), and between the Adjacent and the Outside (Adjacent < Outside, $P = 0.019$) and Between (Adjacent < Between, $P = 0.043$) treatments.

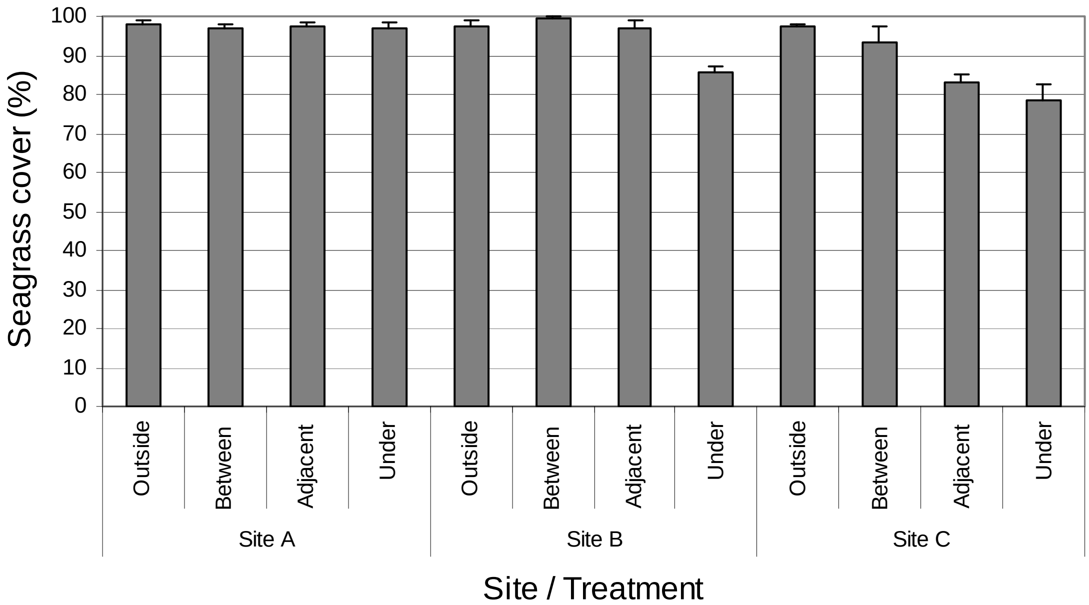

Figure 5.7. Seagrass (*Posidonia*) cover (mean + SE, n = 3) for the four experimental treatments of Outside, Between, Adjacent, and Under BST oyster long-lines at Sites A-C on South Spit, Stansbury, South Australia.

5.3.3 Seagrass morphology

At all three sites there is a clear trend for lowered leaf densities at the Adjacent and Under treatments in comparison to the Between and Outside treatments (Figure 5.8). At Site B, leaf density was almost 30% lower under the long-lines than between the long-lines. However, an ANOVA failed to show any effect of Treatment ($F_{3,6}$ =

3.070, P = 0.113). Nonetheless, the consistent trends in leaf density observed at all three sites may be an indication of a sublethal effect of shading and/or trampling.

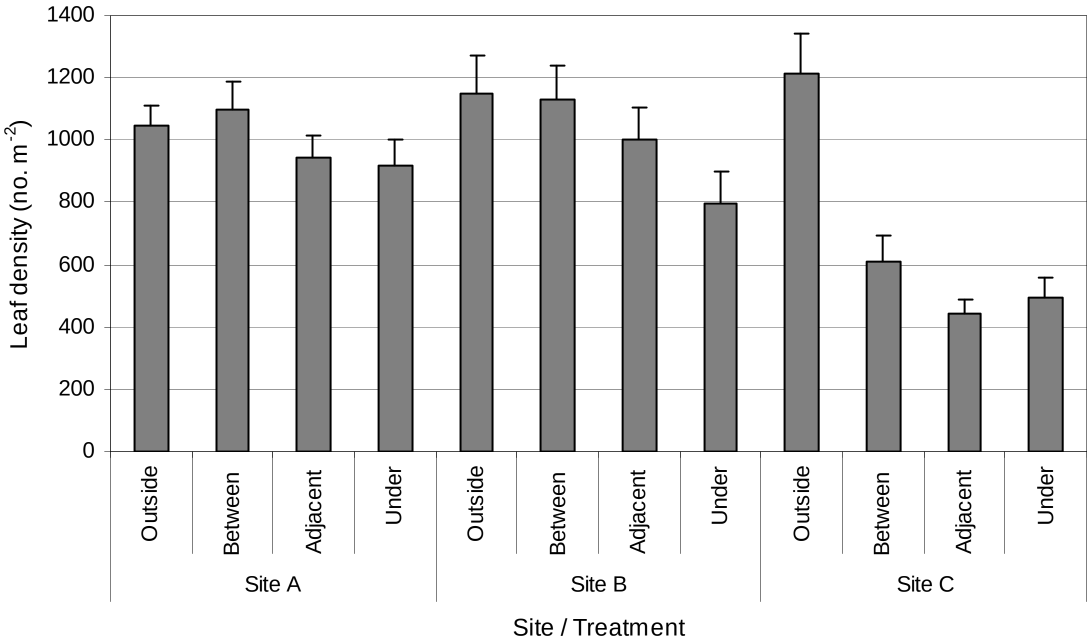

Figure 5.8. Seagrass (*Posidonia*) leaf density (mean + SE, n = 3) for the four experimental treatments of Outside, Between, Adjacent, and Under BST oyster long-lines at Sites A-C on South Spit, Stansbury, South Australia.

There were no clear trends in *P. australis* and *P. sinuosa* leaf length or leaf width that might suggest an effect of long-lines (Figures 5.9, 5.10). In support of this, 2-way ANOVAs showed no effect of Treatment on leaf length or leaf width for both species (*P. australis*: leaf length $F_{3,6}$ = 1.535, P = 0.299; leaf width $F_{3,6}$ = 0.901, P = 0.494; *P. sinuosa*: leaf length $F_{3,3}$ = 2.424, P = 0.243; leaf width $F_{3,3}$ = 1.537, P = 0.366). Due to the lack of *P. sinuosa* at Site A (Figure 5.10), statistical analyses were performed using Sites B and C only.

While conducting the intertidal surveys, we observed unnatural spatial patterns of bare sand within some parts of the lease at Site B (Figure 5.11). These areas of bare sand were clearly correlated with previous positioning of spat trays across the long-lines (Figure 5.11a,b) and represent areas of seagrass loss.

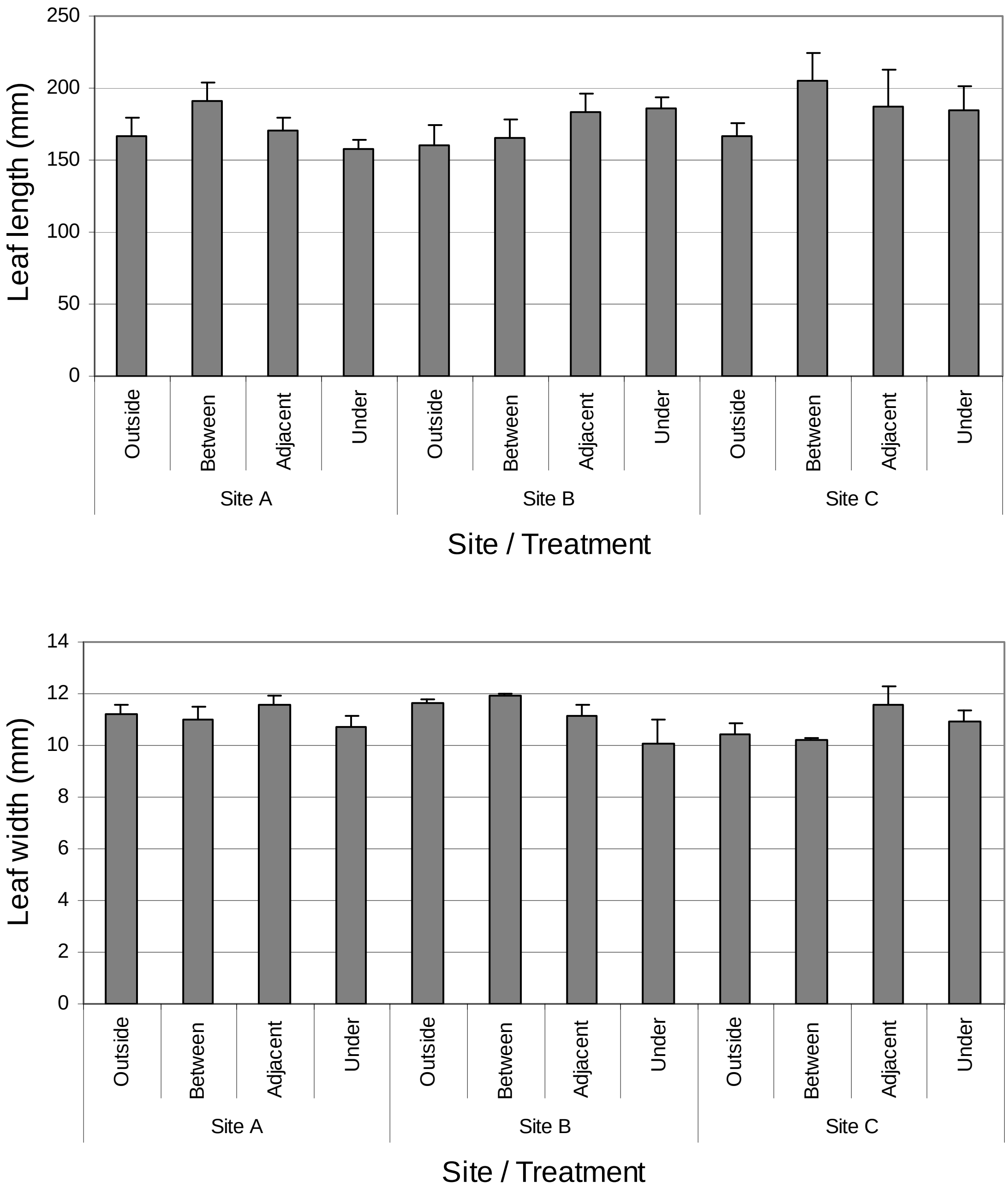

Figure 5.9. Leaf length and width (means + SE, n = 3) of *Posidonia australis* for the four experimental treatments of Outside, Between, Adjacent, and Under BST oyster long-lines at Sites A-C on South Spit, Stansbury, South Australia.

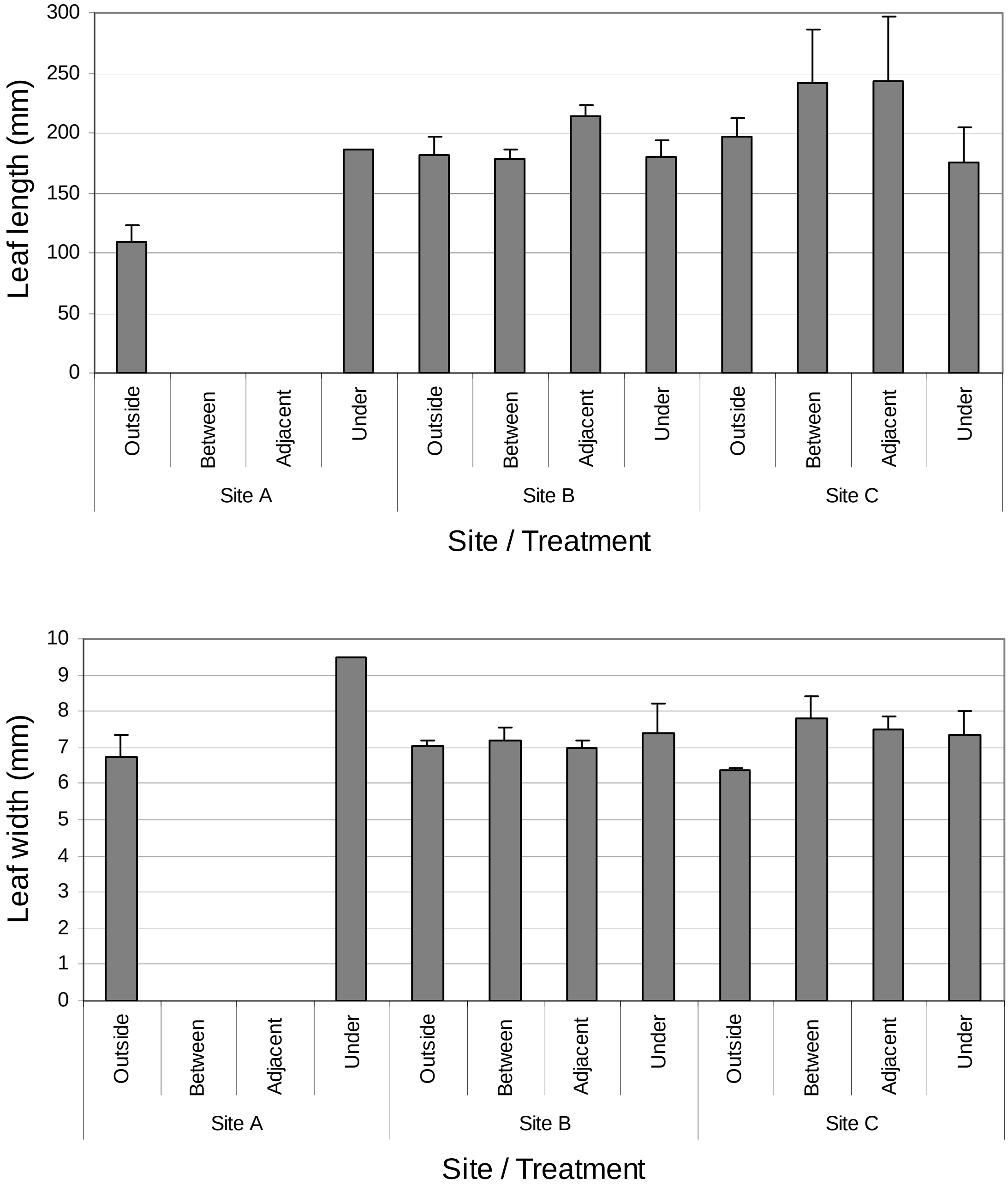

Figure 5.10. Leaf length and width (means + SE) of *Posidonia sinuosa* for the four experimental treatments of Outside, Between, Adjacent, and Under BST oyster long-lines at Sites A-C on South Spit, Stansbury, South Australia. n = 3, except for Site A, Outside, n = 2, and Site A, Under, n = 1. Missing bars indicate a lack of *P. sinuosa* for that site and treatment.

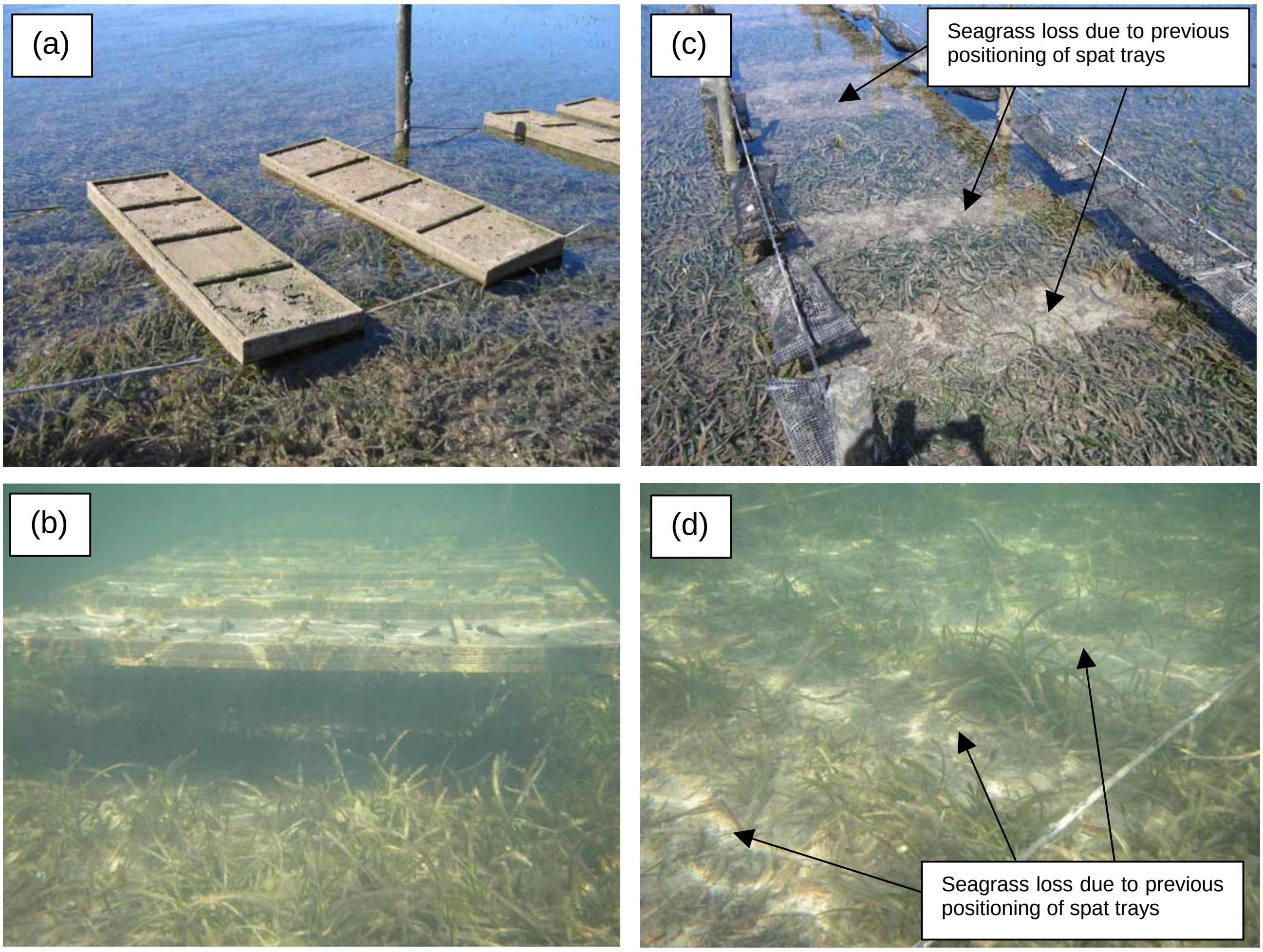

Figure 5.11. Localised strips of seagrass (*Posidonia*) loss due to shading from spat trays at Site B on South Spit at Stansbury, South Australia, with the heavy shading caused by the trays shown (a) at low tide from above water and (b) at high tide underwater, and previous losses due to trays that have since been removed shown (c) at low tide from above water and (d) at high tide underwater. Note that oyster baskets shown in (c) are not the cause of the seagrass loss shown.

5.3.4 Razorfish

If a trampling impact on *Pinna* was occurring, one might expect lowered numbers adjacent to long-lines where farmers walk. However, the data do not show any such pattern (Figure 5.12). As the 2-way ANOVA showed a significant interaction of Treatment x Site ($F_{6,23} = 3.840$, $P = 0.008$), separate ANOVAs were run for each site separately; these tests showed no significant Treatment effect at Site A ($F_{3,7} = 2.629$, $P = 0.132$) or Site B ($F_{3,8} = 1.295$, $P = 0.341$), but a significant effect at Site C ($F_{3,8} = 13.432$, $P = 0.002$). Nonetheless, a post-hoc test showed that the only differences lay between the Outside treatment and the other three treatments (LSD tests: Outside ≠ Adjacent, $P = 0.001$, Outside ≠ Between, $P = 0.002$, Outside ≠ Under, $P < 0.001$).

5.3.5 Infauna

Infaunal samples were characterised by very low abundances across a wide range of taxa (Appendix 5.A, Table 5.). There was no evidence of any difference between Post and Control treatments, with an ANOSIM showing no significant difference (Global R = 0.073, P = 0.172), and an nMDS plot supporting this conclusion (Figure 5.13).

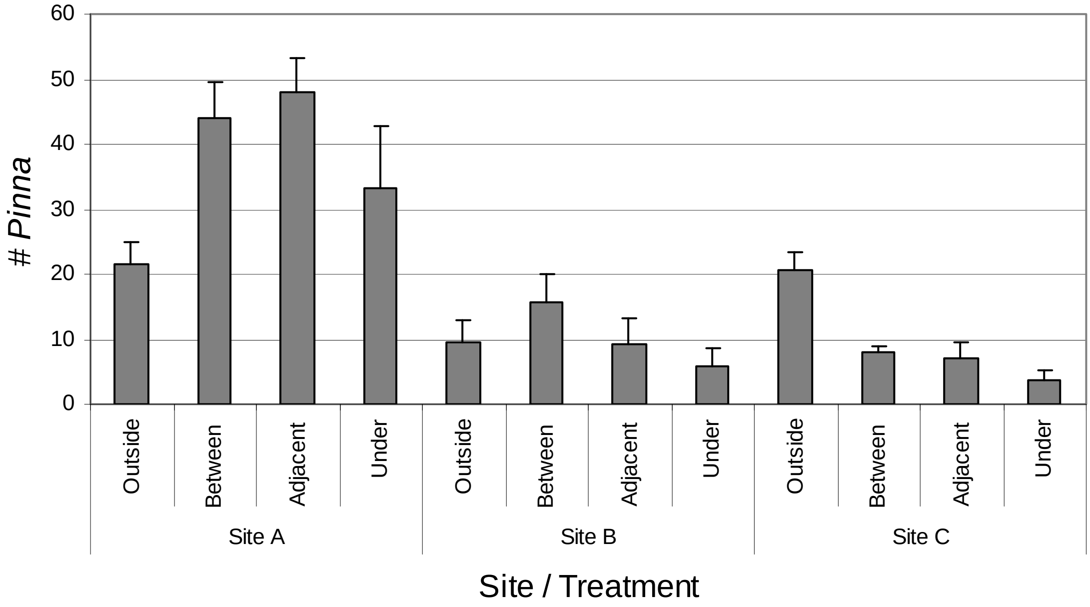

Figure 5.12. Number (mean + SE) of *Pinna bicolor* in belt transects for the four experimental treatments of Outside, Between, Adjacent, and Under BST oyster long-lines at Sites A-C on South Spit, Stansbury, South Australia. n = 3, except for Site A, Outside, where n = 2.

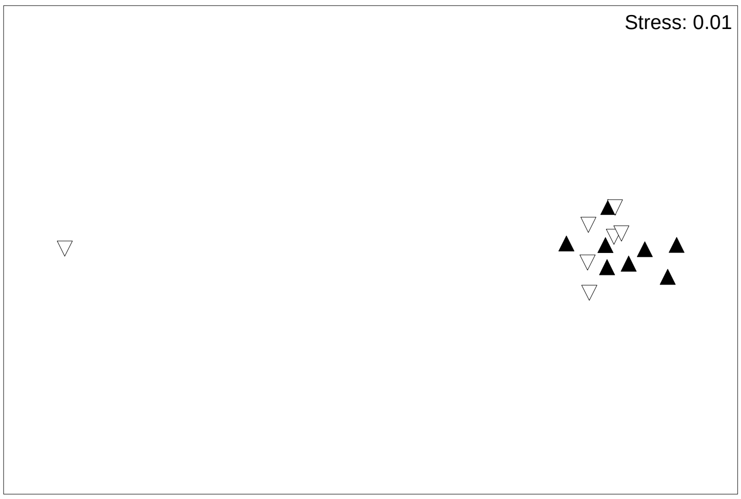

Figure 5.13. nMDS plot of benthic infauna in Post and Control treatments. Closed symbols are Post samples, Open symbols are Control samples.

5.4 Discussion

South Spit was found to have a dense coverage of two meadow-forming seagrasses, *Posidonia australis* and *P. sinuosa*, which are both completely exposed at spring low tides. Whilst *Posidonia australis* is commonly found in intertidal areas around South Australia, South Spit is somewhat unusual in that *P. sinuosa*, which is usually restricted to subtidal waters, is also common in the intertidal there. Razorfish (*Pinna bicolor*) were also found to be prevalent on South Spit, but with a patchy distribution, which is usual for this species (Butler and Keough 1981).

Based upon previous work by Madigan et al. (2000), it was anticipated that no effect of BST long-line culture on the *Posidonia* meadows of South Spit would be found. However, the collective results of seagrass cover and leaf density from our survey indicate a localised, but variable, effect of BST long-line basket culture on seagrass. Nonetheless, the impact associated with basket culture appears to be sublethal with the reductions in cover and density being relatively minor compared to control areas between long-lines. If long-line basket culture was having a lethal impact on seagrass, then one might observe a band of bare sand underneath and/or adjacent to the long-lines (similar to the spat tray impact - see below); clearly this is not the case for basket culture.

5.4.1 Shading

Prolonged shading of *Posidonia* can cause a reduction in leaf density and leaf length (Neverauskas 1988, Gordon et al. 1994, Fitzpatrick and Kirkman 1995). The apparent subtle impact associated with BST long-line basket culture on *Posidonia* is probably due to shading from the baskets. It was evident from the light logger data that baskets do cause some level of shading. The timing and degree of shading on a daily basis will depend on the orientation of the long-lines. In the case of Site A, in which long-lines are oriented East-West, the shading directly underneath baskets occurred during the afternoon. The situation at Sites B and C, which are oriented North-South, will be quite different. As the timing and duration of shading could affect seagrass productivity, it is possible that some of the differences in cover and morphology between sites could be due to the differing orientation of the long-lines. It is also evident that heavy localised shading does occur underneath the BST long-line baskets when they are suspended close to the seabed (Figure 5.14). Due to their low positioning, the shadow cast by these baskets will not move far during the day, as opposed to baskets that are suspended higher. It is predicted that low-lying baskets will have a greater impact on seagrass productivity through the effects of shading.

In contrast to baskets suspended along BST long-lines, it is apparent that spat trays suspended across BST long-lines can have a lethal impact on *Posidonia*. The areas of seagrass loss observed were clearly correlated with the shape of the spat trays, and are almost certainly due to excessive shading. It is unknown how long it took for the seagrass to die in these cases, nor when, or if, seagrass recovery will occur. There did not appear to be any regrowth from within the bare sand areas where spat trays had been removed, which is consistent with shading experiments on *P. australis* in which the shoots have died due to excessive shading (Fitzpatrick and

Kirkman 1995). In these situations, recovery will need to occur by rhizome spreading from surrounding seagrass and/or by seedlings. However, it is apparent that recovery of denuded areas by *Posidonia* is a very slow process (Bryars and Neverauskas 2004, Shepherd et al. 1989).

Figure 5.14. Baker-Schultz-Turner (BST) designed oyster long-line stocked with baskets at Site B on South Spit, Stansbury, South Australia. Note the low-lying nature of the baskets and the heavy shading over seagrass underneath the baskets.

Whilst a sublethal impact from basket culture and a lethal impact from tray culture was apparent, the total area of impact is very minor relative to the total area of seagrass cover on South Spit, which we estimate as being several km^2. Furthermore, we found no evidence of flow-on erosional effects that could be triggered by the seagrass impacts. It is also possible that the sublethal effect is a seasonal one; Wear et al. (2006) found seasonal sublethal impacts of drain discharge water on *Posidonia* in the SE of South Australia.

5.4.2 Physical disturbance

It is difficult to separate the possible effects of shading versus trampling adjacent to long-lines where both activities may occur. The spatial patterns of seagrass cover and morphology that we surveyed adjacent to long-lines were inconsistent across sites. Certainly, we did not observe any bare ‘walking tracks’ alongside the long-lines. Furthermore, there was no indication of an impact on razorfish, which would be highly susceptible to breakage from excessive trampling. The differences observed between treatments at Site C were probably due to small-scale patchiness and natural differences between the lease area and the outside control area, as *Pinna* is renowned for having a patchy distribution (Butler and Keough 1981) and this was clearly evident when we conducted fieldwork on South Spit (Figure 5.15). Furthermore, the trend seen at Site C was reversed at Site A (Figure 5.12), with higher numbers inside the lease than outside. Overall, there was no evidence for lowered numbers of *Pinna* that might be associated with trampling. It is apparent from talking to oyster growers that each oyster basket is visited infrequently (every few months) and therefore each adjacent patch of tidal flat is trampled at that frequency. Madigan et al. (2000) also reported visitation rates of <10 times per year for BST long-lines and racks in Murat Bay and suggested that the lowered *Posidonia* biomass they recorded adjacent to the racks was due to shading rather than trampling (note that they found no effect adjacent to the BST long-lines). Experimental work with trampling on the seagrass *Thalassia testudinum* (a robust perennial species that is morphologically comparable to *Posidonia*, see Green and Short 2003) showed that seagrass biomass was inversely related to trampling intensity and duration (Eckrich and Holmquist 2000). However, the ‘lightly-trampled’ treatment of Eckrich and Holmquist (2000) consisted of trampling 20 times per month for 4 months; a rate far greater than would normally be expected adjacent to the oyster lines at Stansbury. We therefore conclude that trampling is not currently a major issue for *Posidonia* or *Pinna* within the oyster leases of South Spit at Stansbury. Indeed, Krastev (2001), in a study of *Pinna bicolor* around oyster leases in Smoky Bay, found that razorfish were actually more prevalent inside than outside leases and suggested that the leases acted as refuges for the razorfish from recreational fishers. Nonetheless, effects of trampling on razorfish and seagrasses could potentially occur in leases where visitation rates of lines are very high.

Whilst conducting the intertidal surveys we noted scars within the *Posidonia* beds that are consistent with damage from boat propellers (Figure 5.15). However, it is not possible to say if oyster farm boats caused the damage. Recreational boaters do visit the area to collect razorfish and during the time of our survey we observed a non-oyster farm boat forcing its way across South Spit during mid-tide causing damage to the seagrass beds.

5.4.3 Infauna

Although very limited in scope, the infaunal survey showed no difference between samples collected directly against posts and those collected about 5m away from posts. Assuming that the entire lease area is not being affected, then there was no indication of an effect of chemical (viz. CCA) leaching from posts on benthic infauna. As benthic infauna is known to be highly sensitive to the chemical composition of sediments, this suggests that effects on other biota from leaching are unlikely.

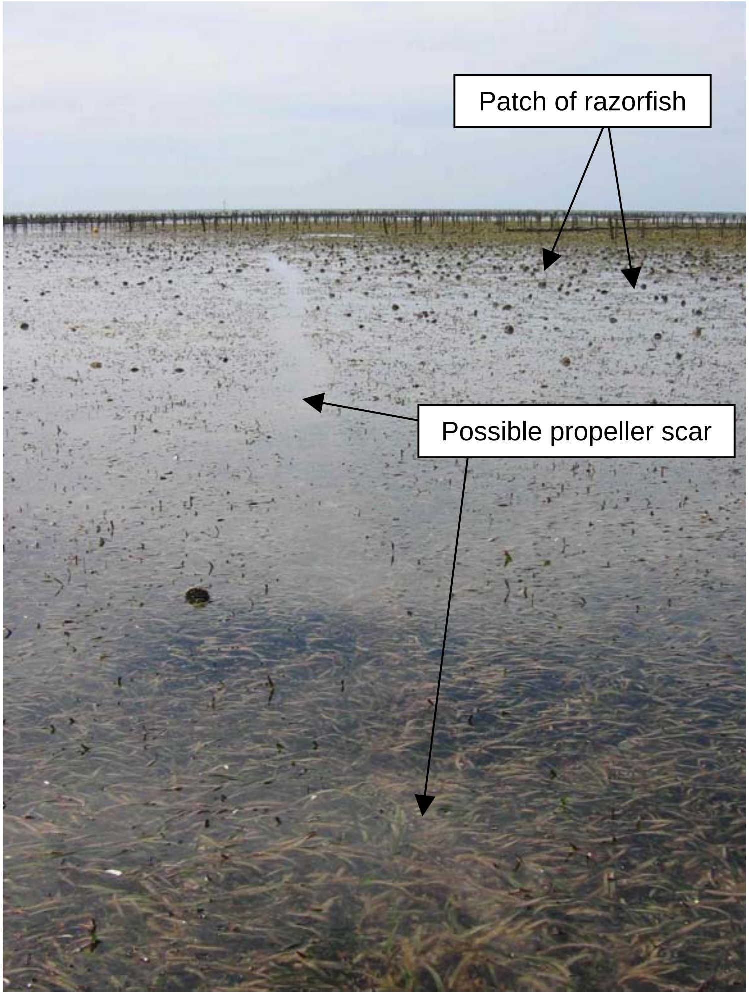

Figure 5.15. Evidence of damage to a *Posidonia* seagrass meadow caused by a boat propeller on South Spit, Stansbury, South Australia. Note the patch of razorfish (*Pinna bicolor*) indicated by the dark tips appearing above the seagrass canopy at low tide.

5.5 Conclusions

Based on the outcomes of the present investigation and previous work, the following conclusions and recommendations are made:

- Prolonged heavy shading is known to kill *Posidonia* seagrass.
- A sublethal shading effect of BST long-line baskets on *Posidonia* seagrass was apparent at the three sites surveyed at South Spit, Stansbury, South Australia.

- A lethal shading effect of BST long-line spat trays on *Posidonia* seagrass was observed at one of the sites at South Spit, Stansbury, South Australia.
- Whilst the relative area and degree of shading impact on *Posidonia* meadows from BST long-line culture at South Spit is low, lethal and sublethal impacts could be reduced with the following farming practices:
    - BST long-line baskets to be kept suspended above the seabed as high as practicable.
    - Use of a rotational schedule for stocking BST long-lines with baskets.
    - BST long-line spat trays to be moved every few months.
- While trampling can affect seagrass aboveground biomass and could potentially affect razorfish, there was no evidence for a trampling affect on *Posidonia* or *Pinna* at South Spit, Stansbury, South Australia.
- There was no evidence for an effect of chemical leaching (viz. CCA) from treated posts on benthic infauna at South Spit, Stansbury, South Australia.

5.6 Acknowledgments

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5.8 Appendix 5.A.

Table 5.1. Taxonomic groups and counts for the 8 Post treatment samples (P1-8) and the 7 Control treatment samples (C1-7) collected for the infaunal survey at South Spit, Stansbury, South Australia.

Taxa	P-1	P-2	P-3	P-4	P-5	P-6	P-7	P-8	C-1	C-2	C-3	C-4	C-5	C-6	C-7
AMPELISCIDAE	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
AMPHARETIDAE sp 1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
AORIDAE	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Armandia	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0
Artacamella	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
ASELLOTA	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Cheirocratus group	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0
CIRRATULIDAE	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
cockle	0	0	1	0	0	1	0	0	1	0	0	0	0	0	0
COPEPODA	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
DEXAMINIDAE sp2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
DORVILLEIDAE group 1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Eunice	0	0	1	0	0	0	0	0	0	1	1	0	0	0	0
Glycera	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
HESIONIDAE	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Hesionura australiensis	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0
Leptosynapta	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
MALDANIDAE	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Mediomastus	0	1	1	0	0	1	2	0	0	1	2	0	0	0	0
NEMATODA	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
NEMERTEA	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Nephtys gravieri	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
NEREIDIDAE	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Phylo	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Platynympha longicaudata	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Schistomeringos loveni	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0
Scyphoproctus	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Solemya	0	0	0	0	0	1	0	0	1	0	0	0	0	1	0
Spio	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Syllinae sp 2	2	0	0	0	0	0	0	0	0	1	1	1	1	0	0
TEREBELLIDAE	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0

Chapter 6: Environmental monitoring programs for marine finfish in South Australia.

Jason Tanner

6.1 Introduction

PIRSA Aquaculture specify mandatory environmental monitoring and reporting requirements for all aquaculture sectors in South Australia. These sectors are currently intertidal shellfish, subtidal shellfish, marine finfish, marine tuna, subtidal abalone and land-based aquaculture.

The marine tuna EMP is the most developed, and is based on extensive investigations of the environmental impacts of tuna farming off Port Lincoln. These investigations commenced with the work of Cheshire et al. (1996a, b), who found that effects could be detected up to 150 m from cages in the early days of farming when leases were primarily in shallow water in Boston Bay, and before many of the current management practises that reduce feed loss and other environmental impacts were introduced. The tuna environmental monitoring program (TEMP) has been conducted annually since, and has evolved through a number of iterations based on the findings of these surveys and a range of research projects (see Clarke 1998, Clarke et al. 1999, Madigan et al. 2001). The current TEMP is based on Madigan et al. (2001), and requires sampling of benthic infauna from a compliance point for each lease, located 150 m outside the lease boundary, as well as associated control points. In addition, up to 2003, video assessments were required of the seafloor, with 150 m transects conducted north and south from a pontoon, and south from the midpoint of the southern lease boundary. In 2004, this requirement for video surveys was dropped as footage from previous years indicated that there was little fauna or flora present that could be reliably assessed using video. In 2005, the methodology to assess the infaunal component of the TEMP changed from the traditional manual counting and identification to the use of genetic assays (Loo et al. 2006a), allowing a more rapid turn-around and decreasing the cost of the analysis.

Other aquaculture sectors also have EMPs that have evolved over time, and with an underpinning from research projects and earlier monitoring programs that have been conducted by various organisations (reviewed in Bryars (2004)). The current EMP requirements for all sectors are specified in the *Aquaculture Regulations 2005*. In this chapter, the current marine finfish sector EMP requirements are assessed against the findings presented in Chapter 2, and recommendations are made as to how the EMP could change in the future.

6.2 Marine finfish EMP options

The current EMP requirements for marine finfish are based on those developed for tuna, reflecting the recommendations of Madigan et al. (2001). The *Aquaculture Regulations 2005* require an annual video survey as described above (see Figure 6.1), and at the Minister's discretion, an assessment of benthic infauna. In addition to documenting the outcomes of the video surveys, the annual EMP report is also required to include details of the location of the lease and all structures on it, the location of fallowed areas, details of the amount of stock held, feed inputs and any chemicals used, and details of all known interactions with large marine vertebrates.

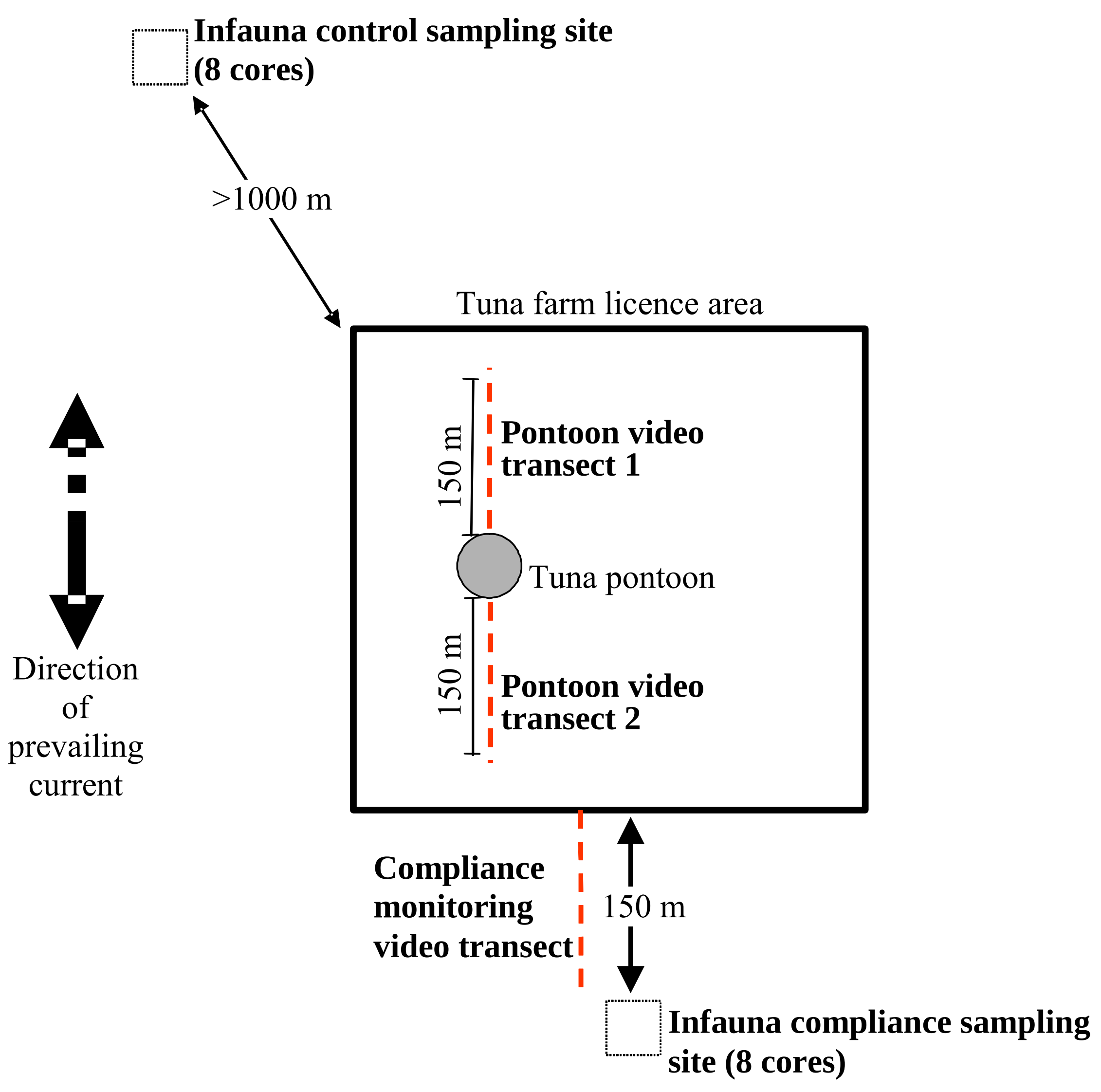

Figure 6.1. Schematic showing design of video and infaunal surveys for the 2003 TEMP. The video surveys currently required for marine finfish EMPs under the *Aquaculture Regulations 2005* follow this protocol.

Chapter 2 documents the findings from a series of video transects conducted both from the edge of pontoons and the edge of leases. The purpose of these transects was to assess the presence of large epifauna/epiflora around pontoons and leases, and to determine if the assemblage changed with distance along the transect. For the transects radiating out from lease edges, there was variability between sites, but no distance effects were detected (Table 2.14). Nor were there any differences between aquaculture leases and control sites at least 1 km away from aquaculture. For the transects radiating out from cage edges the results were more complex, with a number of higher order interactions that indicate a substantial degree of small-scale variability in epifaunal composition. It was thus not possible to determine if there were effects due to aquaculture or not, although if there were they would have been fairly subtle. While factors such as sediment colour, and the presence of wastes, were not formally assessed from these videos, there was no apparent accumulation of wastes close to cages, and no other noticeable differences between cage transects and off-lease transects.

While video footage provides a clear indication of the presence or absence of wastes and major impacts such as microbial mats, the results presented in Chapter 2, along with the experience from the tuna EMPs, indicate that video transects are only likely to detect substantial impacts such as build-up of excess feed. This differs to what has been found in some other areas, such as Tasmania (Macleod et al. 2004), where video surveys have been found to be highly useful. This difference is likely due to the nature of the farming environment in South Australia compared to Tasmania. Marine finfish (and tuna) farming tends to occur in areas with relatively high water movement and coarse sediment, whereas salmon farming in Tasmania tends to occur in more sheltered areas with low water movement and fine sediments. As a consequence, epifauna are sparser and more variable in South Australian aquaculture areas than they are in Tasmanian areas. Video transects may prove useful in other areas of South Australia if aquaculture is conducted overs substrates with a high cover of benthic organisms, although the current management arrangements discourage this.

The other current component of the marine finfish EMP that is only required at the Minister's discretion is infaunal sampling. Again, Chapter 2 shows that the response of infauna to the presence of yellowtail kingfish cages is variable, and not necessarily consistent with organic enrichment, as would be expected if aquaculture was causing an impact. The data presented in Chapter 2 allow us to conduct a power analysis to determine the sample size required to detect a given level of impact on infauna for any of the univariate indices that can be measured. Here, we present a power analysis for total abundance and taxon richness, assuming that a four-fold increase in abundance or a 50% decrease in richness are the effects that are required to be detected. These levels are those that were used for tuna farming when infaunal assemblages were enumerated manually, and were designated by PIRSA Aquaculture, the relevant regulatory authority in South Australia.

Power analysis for univariate ANOVA was conducted using the package Power and Precision v 2.00. The algorithm used requires an estimate of the standard deviation, as well as of the expected means for each sample site. Estimates of standard deviation were obtained from the data described in Chapter 2, along with estimates of the means for control sites. An estimate of the mean for the compliance site was then calculated based on the above effect sizes. For both abundance and taxonomic richness, natural logarithm transformed data were used. In both cases, five separate scenarios were examined. The first used the overall mean and standard deviation from all four 1000 m sites to provide an estimate of control values, while the other four used estimates for each 1000 m site individually. In all cases, it was assumed that the control sites had the same mean.

For abundance, it is obvious that high power is achieved under all five scenarios with a relatively low number of replicates (Table 6.1). In the worst case scenario, using data from the cage 2 1000 m site only, a power of 0.8 is achieved with 6-7 replicates, and of 0.95 with 9 replicates. These are the two generally accepted standards for power (e.g. Underwood 1993, Murphy and Myors 1998), and the design for the tuna environmental monitoring program is based on a power of 0.8. As expected, power increases fairly rapidly as sample size (number of replicates) increases, but increases much more slowly as the number of control sites increases. Indeed, power can actually peak for an intermediate number of control sites.

Table 6.1. Power analysis for total infaunal abundance in Fitzgerald Bay. Analyses are carried out on natural log transformed data. k is the number of control sites, n the number of replicate samples per site, and f the effect size.

k	1	2	3	4	5
Mean abundance and variance of all four 1000 m sites					
Control mean = 1.31					
Compliance mean = 2.69					
SD = 0.57					
f	1.21	1.14	1.05	0.97	0.90
n = 5	0.92	0.94	0.95	0.95	0.95
6	0.96	0.98	0.98	0.98	0.98
7	0.99	0.99	0.99	0.99	0.99
Abundance and variance for cage 1 1000 m site					
Control mean = 1.19					
Compliance mean = 2.35					
SD = 0.44					
f	1.32	1.24	1.14	1.05	0.98
n = 5	0.95	0.97	0.98	0.98	0.98
6	0.98	0.99	0.99	0.99	0.99
7	0.99	0.99	0.99	0.99	0.99
Abundance and variance for cage 2 1000 m site					
Control mean = 1.62					
Compliance mean = 3.25					
SD = 0.92					
f	0.89	0.84	0.77	0.71	0.66
n = 5	0.69	0.72	0.72	0.71	0.70
6	0.79	0.82	0.83	0.82	0.81
7	0.86	0.89	0.90	0.89	0.88
8	0.91	0.94	0.94	0.94	0.93
9	0.95	0.96	0.97	0.96	0.96
10	0.96	0.98	0.98	0.98	0.98
Abundance and variance for cage 3 1000 m site					
Control mean = 1.04					
Compliance mean = 2.08					
SD = 0.37					
f	1.41	1.33	1.22	0.12	1.05
n = 5	0.97	0.99	0.99	0.99	0.99
6	0.99	0.99	0.99	0.99	0.99
Abundance and variance for cage 4 1000 m site					
Control mean = 1.31					
Compliance mean = 2.69					
SD = 0.57					
f	1.68	1.58	0.46	1.34	1.25
n = 5	0.99	0.99	0.99	0.99	0.99

Table 6.2. Power analysis for infaunal taxonomic richness in Fitzgerald Bay. Analyses are carried out on natural log transformed data. k is the number of control sites, n the number of replicate samples per site, and f the effect size.

k	1	2	3	4	5
Mean richness and variance of all four 1000 m sites					
Control mean = 1.09					
Compliance mean = 0.39					
SD = 0.49					
f	0.71	0.67	0.62	0.57	0.53
n = 5	0.51	0.53	0.52	0.51	0.49
6	0.61	0.63	0.63	0.61	0.60
7	0.69	0.72	0.72	0.71	0.69
8	0.76	0.84	0.79	0.78	0.77
9	0.81	0.89	0.85	0.84	0.83
10	0.89	0.92	0.89	0.88	0.88
Richness and variance for cage 1 1000 m site					
Control mean = 1.06					
Compliance mean = 0					
SD = 0.29					
f	1.83	1.72	1.58	1.46	1.36
n = 5	0.99	0.99	0.99	0.99	0.99
Richness and variance for cage 2 1000 m site					
Control mean = 1.46					
Compliance mean = 0.86					
SD = 0.79					
f	0.38	0.36	0.33	0.30	0.28
n = 10	0.36	0.36	0.35	0.33	0.32
15	0.52	0.53	0.52	0.50	0.48
20	0.65	0.68	0.67	0.65	0.63
25	0.75	0.78	0.78	0.77	0.75
30	0.82	0.86	0.86	0.85	0.84
35	0.88	0.91	0.91	0.91	0.90
40	0.92	0.94	0.95	0.94	0.94
Richness and variance for cage 3 1000 m site					
Control mean = 1.04					
Compliance mean = 0					
SD = 0.37					
f	1.41	1.33	1.22	1.12	1.05
n = 5	0.97	0.99	0.99	0.99	0.99
6	0.99	0.99	0.99	0.99	0.99
Richness and variance for cage 4 1000 m site					
Control mean = 1.37					
Compliance mean = 0.45					
SD = 0.32					
f	1.44	1.36	1.24	1.15	1.07
n = 5	0.98	0.99	0.99	0.99	0.99
6	0.99	0.99	0.99	0.99	0.99

For taxonomic richness, higher levels of replication are needed to detect a 50% decrease with a power of 0.8 than are required to detect a four-fold increase in abundance when using the average mean and standard deviation for the four sites (Table 6.2). However, it is obvious that this result is due to high variability in taxonomic richness at the control site for cage 2, where ~ 30 replicates are needed for a power of 0.8, compared to 5 replicates for a power of 0.97 or greater based on the other 3 control sites.

Ignoring the anomalous results for taxonomic richness at the cage 2 1000 m site, the above results suggest that 8 replicates per site are sufficient to give a power of 0.8 for both total abundance and taxonomic richness. The highest power occurs with either two or three control sites, although this assumes that the only difference between control sites and the compliance site is due to aquaculture. Given that there are other differences, it would be prudent to sample five or six control sites.

While infaunal sampling is a well recognised technique for assessing the environmental impacts of aquaculture (Macleod et al. 2004, Edgar et al. 2005, Lee et al. 2006, Brooks et al. 2003, Nickell et al. 2003, Karakassis and Hatziyanni 2000, Carroll et al. 2003), and the power analysis presented above shows that impacts that are deemed to be substantial can be detected with a reasonable number of replicates, the cost of this approach is a drawback. This problem is being addressed through a new project to extend PCR based assessment of infaunal assemblages to the major yellowtail kingfish farming areas (Boston Bay, Arno Bay and Fitzgerald Bay), which if successful, has the potential to reduce the cost of the laboratory analysis component of infaunal sampling. This technique has been recently developed for tuna aquaculture (Loo et al. 2006a), and has been used for the first time to analyse the 2005 tuna environmental monitoring program samples (Loo et al. 2006b). As a result, there was a cost saving of 28.5% on the laboratory component compared to 2004. These savings are scale dependent as the set-up costs for this technique are relatively high, but the additional per sample costs low. Thus, it is unlikely to be economical to use PCR for monitoring of a single lease, whereas the savings may be higher than those indicated above if monitoring is co-ordinated between tuna, finfish and possibly other sectors or industries, such that the set-up costs can be shared. As indicated in Chapter 2, there are currently PCR probes for 3 of the 4 most abundant taxa found at Fitzgerald Bay, however, further work needs to be done to ensure that the technique produces valid results for this region. This work is being conducted in a new project (FRDC 2006/078 "Aquafin CRC: Development of rapid environmental assessment and monitoring techniques for application to finfish aquaculture in South Australia"). Despite this, it is worthwhile considering alternative approaches to investigating the impacts of yellowtail kingfish aquaculture, other than infaunal sampling.

The results presented in Chapter 2 suggest that the chemical composition of sediments may provide an alternative means of detecting the environmental impacts of individual leases. We found clear differences in organic carbon and nitrogen content, and porewater ammonia and phosphorus, with samples collected adjacent to cages having elevated levels of all four compared to those collected at control sites at least 1 km away from any cage. The analysis of porewaters is a relatively complicated activity, with sampling requiring careful handling, as well as centrifuging and filtering prior to freezing within a few hours of collection, so is not a logistically feasible approach to take for routine monitoring. In comparison, sediment carbon and nitrogen content only require samples to be frozen prior to later laboratory analysis. In Chapter 2, significant effects were found for these variables based on 4 samples per site, indicating that relatively low numbers of replicates are needed. The sampling procedure used was to collect cores using a HAPS corer (see Figure 2.2), with the top 1 cm of each core being sliced off and frozen at –30ºC prior to laboratory analysis. While this method of sampling is the same as for infauna (with the exception that more of the core is kept for infauna, and it is not frozen), fewer replicates are needed reducing the sampling cost. The main saving, however, is in the laboratory analysis, which only costs ~$105 per sample for carbon and nitrogen, compared to $300-350 for infauna using manual sorting, and $130-200 for infauna using the PCR technique. An alternative sampling procedure would be to use a pipe dredge (essentially a piece of steel pipe with one end blocked on a chain), which is much easier and the equipment costs only a few dollars compared to ~$10,000 for a HAPS corer. The disadvantage of a pipe dredge is that there is no way to control how deep it penetrates into the sediment (generally only a few cm), and its use may lead to additional variability in the levels of carbon and nitrogen detected, requiring extra replication to maintain power. Before this approach is considered, a study would need to be done to determine how total organic carbon and total nitrogen vary as a function of depth in the sediment profile, and what level of replication was required when a pipe dredge is used.

There are several other disadvantages with relying on measurements of C and N in the sediments for monitoring. Firstly, without a detailed knowledge of the benthic systems around the aquaculture lease, it can be difficult to determine what level of change in the physico-chemical variables is biologically meaningful. Secondly, some of these variables may show high levels of temporal variability, whereas the biological variables often measured can integrate effects over a time period of several months. Sediment organic carbon loadings have been used successfully at other locations to detect benthic impacts of aquaculture (Carroll et al. 2003), although they were not useful in a study of fallowing of salmon farming sites in Tasmania (Macleod et al. 2004). In the study by Carroll et al. (2003), however, the infaunal assemblages at some sites indicated a severe disturbance 50 m from the edge of an operational salmon cage, while sediment chemistry (TOC, pH, H_2S and redox) indicated that the site was normal. Thus, Caroll et al. (2004) concluded that sediment chemistry is a much less sensitive indicator of disturbance than is infaunal composition. This decreased sensitivity could be related to several factors. If sediment chemistry varies on shorter temporal scales than infauna, then a negative impact may be seen on infauna even though the chemistry has returned to normal. Infauna may also be responding to environmental variables that have not been measured. Thus infauna have the advantage that they integrate different impacts from a disturbance, as well as integrating over time. Other physico-chemical parameters that have been shown to at least sometimes respond to the presence of aquaculture are sediment redox and sulphide levels (Wildish et al. 2001, Brooks et al. 2003, Macleod et al. 2004, Edgar et al. 2005). Again, however, the study by Brooks et al. (2003) showed recovery of a suite of sediment physico-chemical parameters almost as soon as harvesting was completed, whereas infauna took a further 6 months to recover.

The measurement of these variables also tends to be more complicated, requiring either divers or careful handling and processing of sediment cores immediately after collection, whereas infaunal samples can be collected remotely and only need to be preserved for processing at a later time. Conversely, processing infauna samples tends to be more time-consumiung and costly, especially using traditional manual sorting techniques.

Given these disadvantages, it is not recommended that monitoring switch fully to the analysis of sediment composition instead of infauna without further work. One way of accomplishing this would be to do a full infaunal and sediment analysis one year, alternating with just sediment analysis in intervening years, providing that the infaunal analysis showed that there were no impacts. For this approach, it is suggested that two separate trigger levels be set (Figure 6.2). The level 1 trigger would be the trigger against which compliance is judged, while the level 2 trigger would be used to determine the extent of monitoring required in the following year. If the level 2 trigger is more stringent than the level 1 trigger, then leases that are compliant but do not have a long way to go before becoming non-compliant, will be required to undertake the full infaunal monitoring program. This will ensure that any deterioration in their environmental performance is picked up early. On the other hand, leases that are a long way from being non-compliant, and thus considered not to have any potential to be non-compliant the following year, can be rewarded for their good performance through reduced monitoring requirements for the following year. Thus, if a four times increase in abundance or a 50% decrease in taxonomic richness is set as the level 1 trigger, as previously used for SBT before the PCR technique was adopted, then the level 2 trigger might be a two times increase in abundance or a 25% decrease in richness. On this basis, for the 2003 TEMP, only 1 lease would have failed to meet the level 2 criteria (having a 2.1 times increase in abundance). In making the decision to reduce monitoring requirements however, farm management practises should also be considered to ensure that the lease is not at increased risk of causing an unacceptable negative impact. So, for example, if production doubles, or feed changes from pellets to baitfish, the environmental risk profile of the lease will change substantially, and it may be desirable to require the more stringent monitoring program regardless of performance in the previous year. Extending this approach, there could either be a requirement to undertake infaunal monitoring every second year, regardless of the results of sediment monitoring, or there could be two sets of criteria for sediments, with sediment only monitoring being allowed in the third and subsequent years if the more stringent criteria are met. Based on Figure 2.24, it appears that a 50% increase in organic carbon and 14% increase in total nitrogen next to cages has minimal influence on infauna, and thus these values are suggested as a suitable level for compliance (trigger level 1) in the first instance, with a 25% and 7% increase respectively suggested as the more stringent level 2 triggers. While this approach would produce cost savings for leases with good environmental performance, if different leases are out of phase within the cycle these savings would be reduced if PCR analysis of infuana is used, as the per sample cost increases as the number of samples decreases.

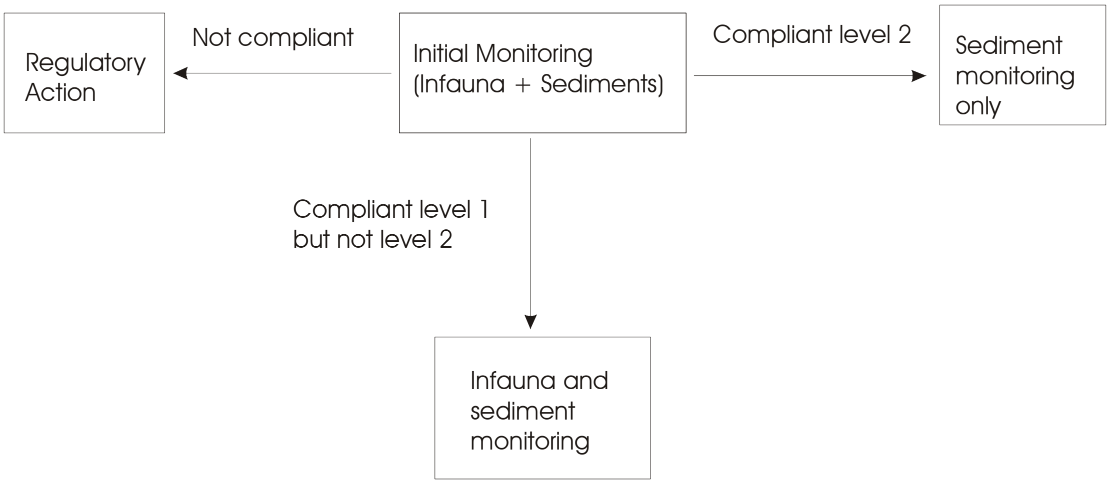

Figure 6.2. Overview of 2-tiered monitoring strategy.

In addition to compliance-based monitoring of individual leases, it is also useful to consider regional monitoring, to ensure that the combined effects of multiple leases are not having larger-scale effects that would not be detected by compliance-based monitoring on individual leases. Based on the results presented in Chapter 2, the most likely approach for this is to monitor the *Posidonia* seagrass (or *Amphibolis* if it still occurs) at sites within Fitzgerald Bay, as well as sites outside the bay (and away from other sources of pollution). While seagrass biomass was the only variable to show a response to proximity to aquaculture, measuring biomass requires destructive sampling of a number of quadrats, and so is not recommended for a routine monitoring program. Instead, it is suggested that leaf density and maximum leaf length be monitored non-destructively, with a subset of leaves collected for determination of epiphyte biomass. The first 2 variables show direct impacts on the seagrasses themselves, while epiphyte biomass is often a good indicator of nutrient enrichment, and can provide an early warning indicator of pending declines in biomass (e.g. Borum 1985, Cambridge et al. 1986, Silberstein et al. 1986, Tomasko and Lapointe 1991, Bryars et al. 2003). To assess the required sampling design to reliably detect changes in these variables, power analysis was undertaken based on the data presented in Chapter 2, using the software developed by Lenth (2006). The high site-to-site variability within a location for all three variables means that the best sampling design has a large number of sites, with minimal replication at each site (Table 6.3). For leaf density, 10 sites would be required for a power of 0.8, while for maximum leaf length and epiphyte biomass, only 5 and 7 sites would be required respectively, but all with only a single replicate. This is problematic in Fitzgerald Bay, however, as there are few sites with seagrasses close to finfish cages, meaning that the required site level replication could not be achieved. An alternative approach would be to simply compare within bay sites to outside bay sites, regardless of the proximity of within bay sites to finfish leases, which would allow adequate replication to be achieved within the bay.

The other issue with monitoring seagrasses is that they will respond to total nutrient loads in the bay, and not just those inputs from aquaculture. Other nutrient sources in Fitzgerald Bay include the shacks along the coastline (Bryars 2003), as well as anything that is exported from Whyalla and Port Bonython, immediately south of the bay. Currently the level of inputs from these sources is unknown, although the Whyalla wastewater treatment plant discharges 48 tonnes of N per year into the marine environment (SA Water 2003), and the Whyalla steelworks 210 tonnes of N per year (DEH 2003). Determining what fraction of this enters Fitzgerald Bay would require detailed hydrodynamic, and possibly biogeochemical, modelling of the area. However, even if all of this entered Fitzgerald Bay, the current nitrogen inputs from finfish farming, which are on the order of 350 tonnes per year (Tanner et al. 2006), would still account for over 50% of anthropogenic inputs. An alternative approach to investigating the sources of nutrient inputs into Fitzgerald Bay is to use stable isotope analysis of seagrasses (Lepoint et al. 2004, Fourqurean et al. 2005). This technique integrates nutrient inputs over time, and allows terrestrial sources to be distinguished from marine. With further detailed studies of the isotope signatures of inputs from the Whyalla wastewater treatment plant, the steelworks, local shacks, and aquaculture feed, it may be possible to distinguish what percentage of N used by the seagrasses comes from each source.

Table 6.3. Power analysis for seagrass variables in Fitzgerald Bay. Analyses are carried out on untransformed data. Rows indicate site level replication, and columns within site replication.

Leaf Density						
SD (Location) = 69.32						
SD (Site) = 84.85						
SD (error) = 33.12						
	1	2	3	4	5	10
6	0.5575					
7	0.6436					
8	0.717					
9	0.778					
10	0.8278					0.8747
Maximum leaf length						
SD (Location) = 409						
SD (Site) = 306						
SD (error) = 104						
	1	2	3	4	5	10
2						0.2915
3						0.6123
4	0.7783					0.8191
5	0.896					0.9238
Epiphyte biomass						
SD (Location) = 2.73						
SD (Site) = 2.59						
SD (error) = 0.92						
	1	2	3	4	5	10
5	0.6953					0.7435
6	0.7997					
7	0.8724					
8	0.9208					
9	0.952					
10	0.9715	0.9785	0.9805	0.9815	0.9821	0.9833

One of the problems with the use of seagrasses, and many other variables, in assessing environmental impacts of aquaculture is that of natural spatial variability. With any once-off sampling event, it is difficult to determine if variability is due to the putative impact, or some other source of variability in the system. Ideally, this would be overcome by employing a BACI (before after control impact) design, preferably with sampling at multiple temporal scales both before and after the impact, and at multiple control sites (e.g. Underwood 1993). In the context of aquaculture, however, this is generally not feasible. In the current study, all aquaculture operations were already well-established when the study commenced, so it was not possible to obtain before data. At the commencement of this project, it was hoped to monitor a new operation off Port Giles, including sampling prior to commencement of farming, but this operation has not been established. Regularly monitoring a defined set of variables will help to overcome this problem, although it is still not possible to state unequivocally that different trends at the putative impact site compared to the control sites are not due to natural differences. Seagrasses, however, offer the advantage that some analyses can be undertaken retrospectively. Growth rates can be determined by examining leaf scars on shoots and rhizomes (Short and Duarte, 2001). In *Posidonia* species, there is a seasonal pattern of growth that can be discerned from these scars, allowing the growth rates over a period of several years to decades to be estimated by destructive sampling. There is thus the potential to examine growth rates before aquaculture commenced in Fitzgerald Bay, as well as after, to determine if patterns have changed since aquaculture commenced.

While the carbon and nitrogen content of seagrasses located in close proximity to aquaculture leases did not differ to that of those at control sites (Chapter 2), it might be useful to consider the elemental composition of seagrasses in more detail. Previous studies of seagrasses have successfully utilised stable isotope ratios of nitrogen to determine the sources of nitrogen utilised by seagrasses (anthropogenic vs natural, e.g. Udy and Dennison 1997, Lapointe et al. 2004a, b, Papadimitriou et al. 2005, Bryars et al. 2006). While the focus of these studies has generally been on detecting signals from wastewater treatment plants, Udy and Dennison (1997) also studied a site near a prawn farm, and could distinguish altered isotope ratios in seagrasses. In addition, isotope ratios in sediments have also been used to characterise the extent and level of nutrient enrichment from a variety of finfish farms around the world (e.g. Sara et al. 2004, Grey et al. 2004, Vizzini and Mazzola 2004, Lojen et al. 2005, Yokoyama et al. 2006). This method relies on different sources of nitrogen having different isotope ratios. For example, in Moreton Bay, treated sewage has a ratio of 9.2‰, while natural sources have a ratio close to zero. The signature for YTK pellets is similar (Milena Fernandes, SARDI Aquatic Sciences, unpublished data). The ability to distinguish aquaculture-derived nitrogen from other sources will depend on the ratios of the different sources, which would all need to be measured, and how distinctive that for aquaculture is. For Fitzgerald Bay, consideration needs to be given to any seepage of effluent from the shacks along the shore, and potential inputs transported from Whyalla. Additional resolution could come from using carbon isotope ratios, especially if feeds contain a substantial component of terrestrial plant material, which has a different signature to marine sources. However, the currently used feed only has a slightly different signature to what would be expected from the natural background (Milena Fernandes, SARDI Aquatic Sciences, unpublished data), which means that it is likely to be difficult to distinguish a signature from it..

6.3 Future research

Several areas for potential future research have been identified above, and these need to be filled before some of the monitoring options discussed here can be implemented. These are:

1. PCR probes need to be tested in Fitzgerald Bay, and additional probes developed if necessary, in order for this technology to be applied on a routine basis. This testing and development is part of a recently funded FRDC project (FRDC 2006/078 "Aquafin CRC: Development of rapid

environmental assessment and monitoring techniques for application to finfish aquaculture in South Australia"), and so is not discussed further here.

2. Before a pipe dredge could be used to simplify sampling for total organic carbon and total nitrogen, the consequences of using this sampling method need to be investigated. To understand the results in the context of current knowledge about variation in these elements around aquaculture leases, sediment cores would need to be examined to assess the vertical profile in both elements. If there is a lot of variation in the top few centimetres, then this method would not be applicable, as there is no way to control the depth that it penetrates into the sediment. In addition, pilot sampling with a pipe dredge would be necessary to allow a power analysis to be conducted, thus allowing the required sample size to be estimated.

3. While not discussed above, some experimental sediment enrichment studies could provide valuable insights into what levels of enrichment of both TOC and TN can be sustained before substantial changes in infaunal assemblages occur. This would then lead to a more informed decision on what the trigger level(s) for monitoring of sediment composition should be. Alternatively, more intensive sampling of infauna from sites with a range of levels of TOC and TN could be undertaken, and the association between infauna and sediment composition studied, although this is likely to miss the high levels of enrichment that are of particular interest.

4. For isotope analysis of seagrasses to be useful for regional monitoring, an assessment needs to be made of the signatures of the different nutrient sources in Fitzgerald Bay. These sources include aquaculture feed, for which some preliminary data are available, any seepage from the shacks along the shoreline, and anything that might be exported from the Whyalla wastewater treatment plant or BHP. Isotope ratios in waters entering the bay should also be measured to help in determining the sources.

5. It may be useful to undertake hydrodynamic modelling of Fitzgerald Bay, along with the associated data collection (deployment of current meters and other oceanographic instrumentation for a period of 1 year). It is possible that a detailed model of Fitzgerald Bay could be embedded in the Spencer Gulf model being developed as part of Risk & Response (FRDC 2005/059), thus reducing the cost of such an undertaking. This model would provide an understanding of whether waste discharges from Whyalla are exported to Fitzgerald Bay, thus aiding in the interpretation of stable isotope analyses of seagrasses. In addition, the model could be used to examine patterns of waste deposition around cages (via the carbon deposition model produced as part of FRDC 2001/104), and if required could be integrated with a biogeochemical model to examine overall system-wide impacts of aquaculture, although this would probably only be justified if production increases substantially. Similar models could be developed for other production areas (such as Arno Bay).

6. Retrospective analysis of seagrass growth could be used to help clarify whether aquaculture is having an impact on seagrasses or not, as this question is not resolved by the analyses reported in Chapter 2. Such an

analysis would allow a comparison of growth rates before and after aquaculture commenced at both control and putatively impacted sites, and thus allow the criteria for a fully rigorous environmental impact study to be met (e.g. Underwood 1993). These criteria cannot be met for any of the other variables utilised in this study, as comprehensive before data are not available.

6.4 Conclusion

Given the high level of small-scale spatial variability found in the epifaunal assemblages of Fitzgerald Bay, it is suggested that remote video alone is not an adequate means for assessing environmental impacts of finfish aquaculture, although it would be suitable for detecting build-up of excess feed and other gross changes. The power analysis for infauna indicates that changes can be detected with a reasonable level of replication, and it is suggested that 8 replicates from each compliance site and each of 4-5 control sites would provide a good sampling regime.

There are several possible alternatives to the current strategy of focusing on video analysis of epifauna, and infaunal analysis, and that could be implemented with little or no additional research. The first of these is to incorporate an analysis of the chemical characteristics of the sediments into any monitoring program. Initially, this should be based on total organic carbon and total nitrogen. In the first instance, a trigger level of a 50% increase in organic carbon content of the sediments relative to control sites, and a 14% increase in total nitrogen, is suggested, and from Chapter 2 it appears that 4 replicates at each site are sufficient. However, given problems with using sediment characteristics in other jurisdictions, it is recommended that initially this be done in conjunction with infaunal sampling. Two alternatives for this would be to conduct both infaunal and sediment sampling every year, or to alternate between the two providing that each individual lease is compliant for infauna at a more stringent level than is currently used. It is suggested that sediment monitoring alone only be allowed if infaunal abundance at compliance sites is less than double that at controls, and if taxonomic richness is no less than 75% of controls. If a pipe dredge is to be used for sediment sampling (as opposed to a HAPS corer or divers), to reduce costs, then it is recommended that further work be done to establish how much extra variability is introduced by not being able to accurately set the depth to which sediments are sampled, as it is likely that the level of replication to achieve adequate power will need to be increased.

The other approach to monitoring is to monitor seagrasses to determine if they are being impacted by finfish aquaculture. As leases do not occur directly over seagrass, it would not be possible to use this approach for compliance-based monitoring of individual leases, but rather it is a regional monitoring approach. As such, it will also monitor for the impacts of total nutrient inputs from all sources, not just aquaculture, although it is possible that stable isotope analysis will allow the different nutrient sources to be partitioned out. If desired, seagrasses could also be used to try and get a handle on longer-term changes through a retrospective analysis of seagrass growth rates.

In the medium term, the PCR technique is being further developed to allow implementation at Fitzgerald, Arno and Boston Bays. Once this project is complete, it will allow a switch away from manual identification and enumeration of infauna, leading to a reduction in costs for infaunal monitoring, especially if laboratory processing is done in a single batch in association with other aquaculture sectors. This could then be applied as a standalone monitoring technique, or in conjunction with sediment sampling as discussed previously for manual enumeration.

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Chapter 7: An optimum environmental monitoring program for land-based abalone aquaculture

Simon Bryars

7.1 Introduction

As part of their annual licence requirements and stipulated in the *Aquaculture Regulations 2005*, land-based abalone growers are required to undertake an environmental monitoring program (EMP) and submit an annual EMP report that details information on farm management practices and water quality sampling. EMP design for land-based aquaculture is based on a consideration of the fate and treatment of each farm's effluent water and feed usage (Appendix 7.A). Licensees are ranked into three categories (A, B or C) according to their potential risk to the receiving environment, with the associated EMP modified accordingly. Most land-based abalone farms are ranked as Category C (High risk), which requires a set number of samples and sampling events to be undertaken at discharge sites and intake sites for the water quality monitoring component of the EMP (Appendix 7.B). For Category C farms, water samples are analysed for oxidised nitrogen, ammonia (total as nitrogen), soluble phosphorus, and total suspended solids. PIRSA Aquaculture then compares the discharge levels against the EPA water quality guidelines for aquaculture stipulated in the EPA *Water Quality Policy 2003*. If trigger levels are breached, then further management action is considered. Thus, the land-based abalone industry operates in an adaptive management framework within the context of ecologically sustainable development (ESD).

Also as part of the land-based abalone EMP report, farm management information is required, including monthly weight and type of feed used, monthly volume of influent and effluent water, type and amount of any chemicals used, and any treatment of discharge water (e.g. settlement ponds) (Appendix 7.B). Monitoring of biological variables in the adjacent marine environment is not currently required as part of land-based abalone aquaculture EMP reporting.

This chapter presents and assesses various options for a land-based abalone EMP in the context of:

- The results of Chapter 4 on biological impacts.
- An adaptive management framework within the context of ESD.
- An appropriate industry EMP needs to be scientifically robust, simple, and cost-effective.
- The EPA's *Water Quality Policy 2003*, which has largely driven the current framework for the land-based aquaculture EMP.

7.2 Possible options for a land-based abalone aquaculture EMP

7.2.1 Option 1: Continue with the current EMP

Methodology: see Introduction

Positives:

- Provides some information on levels of nutrients being discharged to the marine environment from water quality results
- Nutrient levels in discharge waters can be unambiguously compared against EPA water quality guidelines.
- Water samples are easily collected by industry and relatively inexpensive to analyse. Costs per water sample (total phosphorus, total nitrogen, oxidized nitrogen, ammonia) is approx. $80 (estimate based on analysis by the Australian Water Quality Centre).
- Provides information on the amount and type of feed added.
- Provides information on influent and effluent volumes.
- Provides information on chemical usage.

Negatives:

- The nature of water sampling means that the nutrient levels detected may vary greatly depending on the timing of sampling (see Chapter 4). For example, if sampling is conducted during feeding or fertilising times, then elevated levels may be detected. Conversely, if sampling occurs at other times, then peak levels may be missed. Furthermore, sampling can deliberately be conducted when nutrient levels are expected to be low.
- The limited number of water sampling events required for a Class C farm (3 per year) cannot provide a holistic indication of the total annual nutrient load from a farm. Furthermore, nutrient loads cannot be calculated from the farm management data collected on feed tonnage and influent/effluent volumes, without other data on abalone stock tonnage, feed and abalone nutrient composition, feed conversion ratios, and amount and composition of fertiliser added, none of which is collected as part of the EMP (note that stock tonnage is not reported due to confidentiality agreements).
- Water sampling provides no information on possible biological effects in the adjacent marine waters.
- If water quality guidelines at the point of discharge (above high water mark) are exceeded, it does not mean that guidelines would also have been exceeded, or that environmental harm is occurring, in adjacent subtidal marine waters.

Conclusion: Due to the very nature of land-based abalone farming with its high tonnages of stock and feed input, one would expect that nutrient levels in effluent water would be elevated in comparison to influent water. Indeed, this was shown to be the case in Chapter 4. However, nutrient concentrations in effluent water are highly variable, and they provide little insight to nutrient concentrations in adjacent marine waters and any possible biological effects. Thus, the value of monitoring effluent water quality probably needs to be reviewed. The farm management data on feed tonnage and effluent volume is also of little value without accompanying data that enables estimation of nutrient loads. For example, a coarse estimate of nutrient load based on feed inputs could be made using data on the nutrient composition of feed and abalone, feed conversion ratio of abalone, and tonnage of feed and stock.

7.2.2 Option 2: Extend current EMP to include marine sites

Methodology: Continue with water sampling at intake and discharge points, but also include marine sites in waters adjacent to farms.

Positives:

- See Option 1 above.
- Provides information on nutrient concentrations in waters that are relevant to biota and potential for environmental harm.

Negatives:

- See Option 1 above.
- Water quality monitoring may be inadequate for detecting an increase in nutrients in oligotrophic SA waters where nutrients are rapidly assimilated by biota (see Chapter 4).
- EPA water quality guidelines may be inappropriately high for oligotrophic SA waters. For example, in the present study (see Chapter 4), there was mostly no indication of elevated nutrients adjacent to farms in relation to EPA guideline values, yet seagrass nitrogen content was significantly raised adjacent to the farm at Smith Bay (indicating that nitrogen from the farm was in fact elevated in adjacent waters and being utilised by the seagrasses) and nutrient-related changes to benthic communities were apparent adjacent to the farms at both Smith Bay and Point Boston.

Conclusion: Option 2 provides more information than Option 1 (the current EMP), as it may give a better indication of potential environmental harm from elevated nutrient concentrations in waters adjacent to farms. However, the program still has limitations (see Option 1 above) and the value of monitoring water quality may be questionable (see Option 3 below).

7.2.3 Option 3: Use alternative biological measures of water quality in adjacent marine waters

Methodology: Integrated biological measures of water quality may be more appropriate than water sampling. For example, the use of nitrogen content in seagrasses shows some promise, with data from Smith Bay showing a significant elevation adjacent to the farm (Chapter 4). While the biological significance of the elevation is unknown (see Chapter 4), the important point from a monitoring perspective is that nitrogen content was statistically detectable using a limited number of samples. Thus an appropriate monitoring program could use seagrass nitrogen content with a trigger level based upon a set significance level and number of samples that achieves a certain level of statistical power. The use of nitrogen stable isotope ratios in seagrasses may also be worth investigating, as the plant-based aquaculture feed used for abalone may have a specific nitrogen isotopic signature that can be traced in seagrasses deriving nitrogen from abalone farm discharge water.

Positives:

- Provides a more integrated indication of eutrophication than water column nutrients.
- Levels can be unambiguously compared against set guidelines.
- Nitrogen content of seagrasses is relatively simple and inexpensive to measure. Seagrass material could be collected at low tide on snorkel gear to reduce costs and would take only a few hours to complete. Cost per sample for preparation and analysis is approx. $70 (based on SARDI preparation and outsourced analysis). Analysis results are generally available in 1-2 months.

Negatives:

- Depending on the measure used, it may provide no information on whether the biological effect is a negative one (e.g. elevated nitrogen content may not necessarily mean that seagrass is stressed).
- Requires development and testing of potential integrated measures before implementation.

Conclusion: Traditional water quality monitoring of nutrient concentrations has several limitations, including issues with large temporal variation and the rapid assimilation of nutrients in SA waters (see above). Alternative biological measures of water quality, such as seagrass nitrogen content and nitrogen stable isotope ratios, that capture and integrate nutrient concentrations over time, present a potentially viable option to traditional water quality sampling that warrants further research and development.

7.2.4 Option 4: Allow a mixing zone

Methodology: Under the *Water Quality Policy 2003*, the SA EPA can allow a mixing zone for some operations, which is an allocated area where water quality objectives for the receiving waters at the point of discharge may not be achievable. However, a number of requirements must be met for a mixing zone to be approved, including “In marine waters the zone must not have a radius exceeding 100 m and must be at least 200 m from the mean low water mark of the coast at spring tides”. Clearly, the discharges from land-based abalone farms cannot meet this requirement as they occur just above the high tide mark. Nonetheless, a modification of the requirements by the EPA for land-based abalone farms may be justified. Indeed, the current practice of discharging water into the high intertidal, where nutrients undergo some assimilation and dilution through an ‘artificial’ intertidal community before entering the subtidal environment, is probably more acceptable than discharging water 200m offshore directly into the subtidal where naturally occurring seagrass and reef communities would be exposed to peak nutrient levels from discharge waters. Thus, a 100m radius mixing zone based around the present point of discharge (i.e., high water mark) may be appropriate.

Positives:
- Water quality monitoring is no longer required within the mixing zone.
- Water quality guidelines were rarely exceeded adjacent to farms (see Chapter 4), such that allocation of a mixing zone would not be a drastic measure.

Negatives:
- Before a mixing zone could be endorsed, the zone of influence on water quality adjacent to farms would need to be determined. Presently, this is unknown for those few occasions when water quality objectives in subtidal waters adjacent to farms may have been exceeded. Modelling work by Sanderson (2004) for an abalone farm in Port Stephens predicted that ammonia concentrations would decline rapidly within the first 50-100 m away from a subtidal discharge outlet.
- It is possible that a farm discharge may increase over time whereupon the zone of influence becomes larger than the mixing zone. Without monitoring outside the mixing zone, this could not be detected. Thus, water quality monitoring would presumably still be required outside the mixing zone.
- Provides no information on biological effects.

Conclusion: Due to the very nature of land-based abalone farming with its high tonnages of stock and feed input, one would expect that nutrient levels in adjacent marine waters would be elevated. Indeed, this was shown to be the case in Chapter 4. Nonetheless, the degree of elevation of nutrients is typically low, possibly due to the limitations of water quality sampling in oligotrophic waters (Chapter 4). Thus, while Option 4 is preferable to Option 1 because it provides information on the receiving marine waters, it is still reliant on water quality monitoring with its inherent limitations (see above).

7.2.5 Option 5: Undertake biological monitoring

Methodology: Using appropriate techniques, monitor biological parameters that may indicate nutrient-mediated changes due to farms (see Chapter 4) at farm sites and suitable control sites. Appropriate techniques could involve the use of SCUBA and/or remote underwater video to survey benthic composition. The use of fixed markers at the edge of seagrass beds could be used to indicate seagrass loss.

Positives:
- Can monitor if benthic communities continue to change adjacent to farms (given that farms have been operating for some years now and that changes are apparent at some farms – see Chapter 4).

Negatives:
- An EMP based on the benthic surveys outlined in Chapter 4 would be complex and expensive for industry to conduct or outsource. Costs of a 3

person SARDI dive team per day (including boat) are approx. $3500 (estimate based on SARDI commercial rates and does not include travel and accommodation costs associated with surveying regional locations). Detailed surveys at multiple sites will also take several days; it took approximately 1 week per region for a trained 3-person team to survey the sites detailed in Chapter 4. It then takes many days for trained personnel to process and interpret the data, which are not available for 1-2 months later.
- Cannot be sure that biological differences detected between farm and non-farm sites are due to farm discharges, as no data exist prior to commencement of farming (Chapter 4).
- Biological responses may vary between farming regions, and some changes (e.g. increased gastropod numbers) do not represent a biologically significant 'state' change (see Chapter 4). Thus, monitoring of all variables may not be warranted at all regions.
- We already suspect that any change in reef communities has occurred over a long time period; further monitoring would therefore appear of little value (unless monitoring can detect changes in the spatial extent of impacts, see Option 6).

Conclusion: Due to the very nature of land-based abalone farming with its high tonnages of stock and feed input, it is likely that some level of impact will occur on the adjacent marine environment. From the investigations detailed in Chapter 4, it is apparent that some changes have occurred to the biological communities adjacent to farms, but that these changes have occurred over a period of many years. Nonetheless, without prior data, the precise nature and extent of changes can never be demonstrated unequivocally. Furthermore, quite rigorous investigations were required to detail apparently subtle changes adjacent to farms. Thus, based on the high costs and complexity of benthic surveys, and the uncertainty of survey results in demonstrating an impact, it is suggested that detailed benthic surveys should not be used in an EMP. However, the careful selection of some more basic techniques (e.g. permanent markers at the edge of seagrass beds, permanent transects to monitor the cover of canopy versus turfing/red coralline algae, see Chapter 4) might be very useful for monitoring long-term trends and would be relatively inexpensive to monitor compared to detailed benthic surveys.

7.2.6 Option 6: Allow a zone of influence and undertake monitoring adjacent to this zone

Methodology: As with Option 5, monitoring sites can be located directly adjacent to, and away from, farms. However, if the measured variables exceed a trigger level, then what is the response? We know that water quality levels will be exceeded on some occasions and that impacts to benthic communities have probably already occurred, i.e., a zone of influence already exists adjacent to the abalone farms. In the case of water quality, trigger levels already exist, but what of biological parameters? For example, what percentage of macroalgal loss is deemed to be acceptable? If we are to accept some impact on marine communities adjacent to land-based abalone farms, then it becomes a question of spatial extent and temporal stability of this zone of influence. Thus it may be more appropriate to monitor benthic communities, water quality, and other relevant variables, at site(s) just outside the present zone of influence and at appropriate control sites well away from the zone of influence. This approach would be similar in principal to the tuna EMP where sites adjacent to the farm lease boundaries are compared with multiple control sites, and any impacts within the lease are unknown (but acceptable).

Positives:

- The benefits of this approach lie in adaptive management of farm discharges and the level of monitoring required. For example, if changes are detected outside the zone of influence (the boundaries of which are set based on present data), then this indicates that the zone of influence is increasing because either the present nutrient loads are unsustainable or nutrient loads have been increased. Under this scenario, a farm may increase nutrient loads, and if trigger levels outside the zone are not exceeded, then the new load is deemed to be sustainable. If the trigger levels are exceeded, then the load must be reduced.
- The use of low-cost, early warning integrated indicators, such as seagrass nitrogen content (see Option 3), may be most appropriate in the first instance, rather than long-term indicators, such as seagrass and macroalgal loss that are likely to be more costly to monitor. If the early warning indicators are triggered, then measurements of long-term indicators may be required.

Negatives:

- The present spatial extent of the zone of influence needs to be determined.
- Baseline data on benthic communities outside the zone of influence are required.
- Costs will be high for the biological monitoring aspects (see Option 5).

Conclusion: This option accepts that some impact (a zone of influence) will occur adjacent to an abalone farm. However, the proposed EMP is designed to detect changes in the size of the zone of influence and thus provides protection for seagrass and reef communities. It also allows adaptive farm management in a framework of ESD.

7.2.7 Option 7: Apply limits to annual nutrient loads and cease monitoring

Methodology: Rather than monitoring water quality and benthic communities, it may be appropriate to simply set limits for annual nutrient loads from abalone farms. Based upon present and historical nutrient loads, some subtle long-term changes in subtidal marine communities have apparently occurred at Smith Bay and Point Boston, with no change detected at Streaky Bay (see Chapter 4). Some elevation of nutrients in receiving waters can also occur. However, if we are willing to except these changes, then the present annual loads appear to be sustainable in the context of the parameters surveyed in Chapter 4. Thus, monitoring of potential impacts is not required.

Positives:

- No off-lease monitoring required.

Negatives:

- Annual loads need to be calculated accurately (see above and Chapter 4).
- Further biological change may be occurring without knowing it.
- Nutrient loads cannot be increased from present levels, which may limit farm production.
- Difficult to allocate a nutrient load limit for new farms in new regions, as information is available from only a limited number of currently-operating farms on how certain nutrient loads may alter an ecosystem and how this may vary with different types of receiving ecosystems (e.g. seagrass, reef). Nonetheless, a review could be conducted of other SA cases of anthropogenic nutrient-mediated changes to coastal ecosystems (e.g. WWTP outfalls) to place land-based abalone farm nutrient loads into context and to guide nutrient load limits for different ecosystems.

Conclusion: This option accepts that biological impacts may have already occurred adjacent to existing farms and will also occur adjacent to new farms. While it may appear dangerous to have no benthic monitoring adjacent to farms, this is actually the current situation with existing farms (see Option 1). However, the main difficulty with Option 7 is in setting load limits for new farms and in not being able to increase loads at existing farms. Nonetheless, existing farms still have scope to increase their production through improved farm management (e.g. improving feed conversion ratios).

7.3 Discussion

Current and historical land-based abalone EMPs have focused on the water quality associated with discharge waters and farm management practices. However, prior to the investigations of the present report (see Chapter 4), the effects of land-based abalone farms on marine communities in SA were unquantified. It is now apparent that some changes to marine communities may occur and are probably related to increased nutrients derived from farm discharges. While some level of environmental impact is usually associated with an aquaculture operation, the EPA and PIRSA Aquaculture deem an impact within a lease site to be acceptable. It is the off-lease impacts that are of particular concern. Nonetheless, under the *Aquaculture Act 2005*, all attempts should be (and are) made by licensees to mitigate environmental harm within leases. However, for land-based abalone farms that derive water from, and discharge water to, the marine environment that is outside of their land-based lease boundaries, the issue of environmental impact becomes more problematical.

This chapter has provided a number of options for an alternative to the existing EMP. An appropriate EMP needs to be scientifically robust, simple, and cost-effective. In this context, the development of innovative environmental indicators that provide rapid feedback and are inexpensive needs to be pursued. At present, such indicators have not been developed to an appropriate stage for use in a land-based abalone EMP and, in the interim, more traditional methods (e.g. water quality monitoring) should probably be continued. However, in terms of environmental monitoring associated with land-based abalone farming, several key questions have been raised by the present study:

- Is it important to monitor total nutrient load?
- Are current water quality monitoring and guideline values appropriate?
- Are water quality and/or biological monitoring of adjacent marine waters required?
- What level of off-lease environmental impact is deemed acceptable?
- Can a mixing zone or zone of influence be applied for off-lease impacts?

Such questions need to be resolved by the EPA and PIRSA Aquaculture before an optimum EMP can be developed and implemented for the land-based abalone aquaculture sector, and their resolution is beyond the scope of the present study.

7.4 Future research

Several areas for potential future research have been identified above, and these need to be investigated if some of the monitoring options discussed here can be implemented. These are:

1. Development and testing of potential integrated biological measures of water quality such as nitrogen content of seagrasses, and stable nitrogen isotope signatures of seagrasses.
2. Determine links between integrated biological water quality measures and environmental harm, e.g. at what level of nitrogen concentration in seagrass tissue does harm occur?
3. Determine the zone of influence from abalone farms on adjacent coastal water quality.
4. Determine the zone of influence from abalone farms on adjacent biological communities.
5. Accurately calculate total annual nutrient loads, which requires data collection on feed tonnage and influent/effluent volumes (which is currently done), abalone stock tonnage, feed and abalone nutrient composition, feed conversion ratios, and amount and composition of fertiliser added (which is not currently done).
6. Conduct a Before-After-Control-Impact (BACI) designed study on any new land-based abalone farms to determine unambiguously any effects of farming on the adjacent marine environment.
7. Conduct further fundamental research on the effects of nutrient loads on reef and seagrass communities to identify suitable biological indicators for

environmental monitoring programs and to determine sustainable annual nutrient loads.

7.5 Acknowledgments

Dr Stephen Madigan and Dr Peter Lauer (PIRSA Aquaculture) provided valuable input to this chapter.

7.6 References

Environment Protection Authority (2003). Environment Protection (Water Quality) Policy and Explanatory Report 2003.

Sanderson, B.G. (2004) Calculation of dilution and transport of nitrogen discharged from a proposed abalone farm. Unpublished draft report.

7.7 Appendix 7.A

Table7.1. Guidelines for determining land-based licence categories and corresponding environmental monitoring program (EMP) risk profiles (Table supplied by PIRSA Aquaculture).

LAND-BASED AQUA-CULTURE CATEGORY	CATEGORISATION	EXAMPLES	EMP RISK PROFILE
A	Ponds or dams on premises, <1500kg of natural feed used per annum, no physical discharge of water from the ponds or dams	Example: 4 yabbie ponds on a farm, occasional feed of lupins, ponds may overflow occasionally in winter.	Low
B	Either: A) No water released from premises and no water released into natural water bodies (including aquifers) and >1500kg manufactured feed per annum; or B) Water released from premises and <1500kg manufactured feed per annum; or C) Water released from premises and >1500kg natural feed used per annum <u>OR</u> Filter feeding molluscs where water is released from the premises or to a natural water body	This category includes a variety of systems of moderate risk – including farms that grow non-established exotic species or that use artificial feeds or that discharge to open systems but do not use artificial feeds. Examples: Land-based recirculating or pond-based aquaculture system – no discharge, finfish fed pellets frequently or in large volume. Oyster hatchery – pump ashore, flow thru tanks, algae used as feed, discharge to sea.	Medium
C	Water released from premises or water released into natural water bodies (including aquifers) expect for filter feeding molluscs (which fall in Category B); frequent/routine feeding (>1500kg of manufactured feed used per annum)	Farms in this category are generally categorised by use of artificial feeds and also discharge into sea or other open system. Examples: Finfish hatchery – pump ashore, flow thru tanks, feeding manufactured feed, discharge into salt creek or land-based. Abalone farm – pump ashore, raceways, feeding manufactured feed, discharge to sea.	High

7.8 Appendix 7.B

Land-based Aquaculture
Environmental Monitoring Program (EMP) Report
Category C
2005 – 2006

For reporting period **1 July 2005 to 30 June 2006**

Send the completed EMP via mail or fax to:
PIRSA Aquaculture
GPO Box 1625
ADELAIDE SA 5001
Facsimile: (08) 8226 0330
Email: lauer.peter@saugov.sa.gov.au

Due by 31 August 2006

All holders of an aquaculture licence, regardless of the level of development on their licensed area, are required to complete and return this Proforma.

CONTACT DETAILS

Please ensure the following details are complete and correct.

Principal Contact:			
Licensee(s):			
Postal Address:			
Phone:		Mobile:	
Fax:		Email:	

NB: A change of principal contact may require the completion of a transfer form.

SITE DETAILS

Details for the EMP are specific to the individual licence – **DO NOT** include additional licenses.

Licence Number:	
Licence Category:	

Land-based Aquaculture EMP Report – Proforma

For reporting period 1 July 2005 to 30 June 2006

The Land-based Aquaculture EMP Report Proforma can be used to assist with EMP reporting requirements as directed by the *Aquaculture Regulations 2005* (Division 1, Regulation 14 and Division 2, Regulation 27).

All EMP reporting requirements under the *Aquaculture Regulations 2005* can be viewed at: http://www.parliament.sa.gov.au/Catalog/legislation/Regulations/a/2005.205.un.htm or by calling PIRSA Aquaculture on (08) 8226 0314.

Note that under *Aquaculture Regulations 2005*, failure to complete **ALL** the reporting requirements may incur an expiation fee of $315 and a penalty of up to $5000.

Within this Proforma, sections should be completed in legible writing using a black pen with responses written, circled or drawn where appropriate.

Part 1 – Farm management information

1. Is the licensee the author of the following information? (if 'Yes', go to question 3)

Yes	No

2. If 'No', then please provide the name and address of the author.

Name:	
Address:	
Signature:	

3. Has there been any stock on site during the reporting period? (If 'No', go to question 5)

Yes	No

4. Overall, how many months has stock been on site during the reporting period?

1 or less	2	3	4	5	6
7	8	9	10	11	12

5. Does the facility operate mainly as a hatchery or grow-out?

Hatchery	Grow-out	Combination

6. Does the facility mainly operate as a pond, flow-through or recirculation system?

Pond	Flow-through	Recirculation	Combination

If 'Combination' please list details.

7. How many species do you currently farm? (If no species are currently farmed, go to question 9)

0	1	2	3	4	5
6	7	8	9	10	11

If more than one species is farmed please list them.

8. What was/were the principal species farmed during the reporting period?

Yabbies	Marron	Trout	Murray Cod
Silver Perch	Golden Perch	Greenlip Abalone	Blacklip Abalone
Barramundi	Kingfish	Goldfish	Other

If 'Other', please specify.

9. What is the source of production water? (If 'Do not source any production water', go to question 13)

Do not source any production water		
Mains	Surface Water	Combination
Ocean	Ground	Other

If ‘Combination’ or ‘Other’, provide details of the source.

10. What is the salinity of the production water?

Fresh	Brackish	Marine	Hyper-saline

11. Is the volume of water used in production per month constant?

Yes	No

12. State the volume (in thousands of litres – ‘kl’) of water used in production per month.

Month	Jul 05	Aug 05	Sep 05	Oct 05	Nov 05	Dec 05
Volume (kl)						
Month	Jan 06	Feb 06	Mar 06	Apr 06	May 06	Jun 06
Volume (kl)						

13. Is water used in aquaculture production discharged from the property?

Yes	No

14. Regardless of whether water leaves the property or not, state the volume (in thousands of litres – ‘kl’) of water discharged from the aquaculture facility per month.

Month	Jul 05	Aug 05	Sep 05	Oct 05	Nov 05	Dec 05
Volume (kl)						
Month	Jan 06	Feb 06	Mar 06	Apr 06	May 06	Jun 06
Volume (kl)						

15. Regardless of whether water leaves the property or not, is the discharged water treated in any manner?

Yes	No

If 'Yes' describe how the discharged water is treated and what is removed by the treatment process (e.g. suspended solids, nutrients etc).

16. Regardless of whether it leaves the property or not, does the discharged water interact with any other water body? (If 'No', go to Question 18)

Yes	No

17. If 'Yes', which type of water body does the discharged water interact with?

Ocean	Ground Water	Mains Water	Surface Water	Other

Provide details of how the discharged water interacts with the other water body (eg flows into a dam).

18. Is feed added for aquaculture production? (If 'No', go to Question 22)

Yes	No

19. State the type of aquaculture feed provided.

Natural	Manufactured	Combination

20. Is the quantity of feed used for aquaculture production per month constant?

Yes	No

21. State the quantity (kg) of feed (natural or manufactured) used for aquaculture production per month.

Month	Jul 05	Aug 05	Sep 05	Oct 05	Nov 05	Dec 05
Natural (kg)						
Manufactured (kg)						
Month	Jan 06	Feb 06	Mar 06	Apr 06	May 06	Jun 06
Natural (kg)						
Manufactured (kg)						

22. Have any chemicals or medicines (e.g. disinfectants, prophylactics, therapeutics, antifoulants) been used during the reporting period?

Yes	No

If 'Yes', state the (1) date, (2) type, (3) amount and (4) purpose the chemicals or medicines were used for (attach a separate sheet if required).

Date	Chemical	Amount (state if 'kilogram' or 'litre')	Purpose

23. Provide (and attach to this report) a map showing the layout of the farm including;

1. Location of any intake and discharge pipes/channels.
2. Ponds, dams or tanks.
3. Sheds or other structures associated with aquaculture.
4. The location of any other water bodies that may interact with the farm's operations.
5. The location of where water samples were taken.

Also provide GPS co-ordinates of the water sampling sites in GDA94 Easting and Northing coordinates <u>or</u> WGS84 decimal degrees and a date when water samples were taken.

24. Have there been any modifications to your farm layout during the reporting period?

Yes	No

If 'Yes', identify the modifications on the map drawn in question 23.

Part 2 – Water quality sampling

As your licence is a C (high) category environmental risk profile the licensee is required to take and submit 3 series of water quality information.

All licensees must ensure…	✓
Collect and analyse 3 series of water samples per year.	
Each of the series of water samples was taken at intervals no closer than 3 months	
Each of the series of water samples was taken at the same times and dates each year.	
Production water samples (1 or 2)	✓
(1) If there is only 1 intake point, 2 samples must be collected from a representative location of intake water.	
(2) If there is more than 1 intake point, 2 samples must be collected from each of 2 different sites from representative locations of intake water. Outline with regs	
Provide a date of when samples were taken	
Discharged water samples (3 or 4)	✓
(3) If there is only 1 discharge point, 2 samples must be collected from a representative location of discharged water.	
(4) If there is more than 1 discharge point, 2 samples must be collected from each of 2 different sites from representative locations of discharged water.	
Provide a date of when samples were taken	
Water sample analysis	✓
Water samples tested for oxidised nitrogen (as nitrogen NO_3+NO_2)	
Water samples tested for ammonia (total as nitrogen NH_4+NH_3)	
Water samples tested for soluble phosphorous	
Water samples tested for total suspended solids	
Water samples collected according to requirements of the testing laboratory.	
Water samples analysed by an accredited laboratory	

25. As your licence is a C (high) category environmental risk profile, has the licensee;

1. Attached a copy of the results obtained from the accredited laboratory?

Yes	No

2. Identified on the site map (done in question 23) where samples were taken from (including GPS co-ordinates of the sampling sites and a date sampled)?

Yes	No

3. Taken and analysed the samples as directed by *Aquaculture Regulations 2005*?

Yes	No

Checklist of ATTACHED EMP Report requirements (tick where the stated action has been attached with this Proforma).

All licensees must…	✓
Provide a map showing the layout of the farm, as prescribed by question 23.	
Licensees categorised as having a medium or high-risk profile must…	✓
Provide a copy of the original water quality results with this report.	
Provide the location of all water quality sample sites on the site map and provide GPS co-ordinates of the water quality sample sites.	

DECLARATION BY LICENSEE

A person must not make a statement that is false or misleading in a material particular (whether by reason of the inclusion or omission of any particular) in any information provided under the Aquaculture Act 2001. Maximum penalty: $5000[3]

I/We declare that the information I/we have provided in this form is true and accurate.

Signature of all licensees:

Signature	**Name**	**Date**
……………………………..	..…………………………….	…………………………

[3] Section 85 Aquaculture Act 2001

Chapter 8: Conclusions

8.1 Benefits and adoption

This project was instigated and partially funded by PIRSA Aquaculture, with the main goal being to provide scientifically defensible options for environmental monitoring of the marine finfish and land-based abalone sectors. There are a number of clearly identified options for monitoring of each of these sectors provided, and these can be used as the basis for revisions of the environmental monitoring requirements for these sectors in the near future. Which options are followed is a policy decision, and will depend on the exact balance of outcomes that PIRSA Aquaculture wish to achieve. By providing up-to-date and scientifically defensible monitoring programs, both the industry and the regulators will have increased confidence in security of resource allocation, and data with which to refute any unfounded allegations about environmental harm. The hierarchy of monitoring stratgeies presented will allow this information to be obtained in the most cost-effective way possible, without compromising the validity of the data and resultant conclusions.

In addition, the project provides both PIRSA Aquaculture and the relevant industries with a rigorous and independent assessment of the marine environmental impacts associated with each of these sectors. Alongside this, the range of stakeholder concerns with the marine finfish, land-based abalone, and intertidal oyster aquaculture sectors have been identified, and the current state of knowledge with regard to each issue of concern is summarised. This provides both PIRSA Aquacultue and industry with a good understanding of what the perceived issues are, and how these differ to what the real issues are likely to be.

8.2 Further development

There are a number of ways in which the work described in this report can be extended. Most obviously, there are a number of smaller sectors that have not been covered (e.g. subtidal shellfish such as mussels and abalone), and as these sectors grow they will need to be examined in a similar way.

For finfish, several areas for potential future research have been identified above, and these need to be filled before some of the monitoring options discussed here can be implemented. These are:

1. PCR probes need to be tested in Fitzgerald Bay, and additional probes developed if necessary, in order for this technology to be applied on a routine basis. This testing and development is part of a recently funded FRDC project (FRDC 2006/078 “Aquafin CRC: Development of rapid environmental assessment and monitoring techniques for application to finfish aquaculture in South Australia”), and so is not discussed further here.
2. Before a pipe dredge could be used to simplify sampling for total organic carbon and total nitrogen, the consequences of using this sampling method need to be investigated. To understand the results in the context of current knowledge about variation in these elements around aquaculture leases, sediment cores would need to

be examined to assess the vertical profile in both elements. If there is a lot of variation in the top few centimetres, then this method would not be applicable, as there is no way to control the depth that it penetrates into the sediment. In addition, pilot sampling with a pipe dredge would be necessary to allow a power analysis to be conducted, thus allowing the required sample size to be estimated.

3. While not discussed above, some experimental sediment enrichment studies could provide valuable insights into what levels of enrichment of both TOC and TN can be sustained before substantial changes in infaunal assemblages occur. This would then lead to a more informed decision on what the trigger level(s) for monitoring of sediment composition should be. Alternatively, more intensive sampling of infauna from sites with a range of levels of TOC and TN could be undertaken, and the association between infauna and sediment composition studied, although this is likely to miss the high levels of enrichment that are of particular interest.
4. For isotope analysis of seagrasses to be useful for regional monitoring, an assessment needs to be made of the signatures of the different nutrient sources in Fitzgerald Bay. These sources include aquaculture feed, for which some preliminary data are available, any seepage from the shacks along the shoreline, and anything that might be exported from the Whyalla wastewater treatment plant or BHP. Isotope ratios in waters entering the bay should also be measured to help in determining the sources.
5. It may be useful to undertake hydrodynamic modelling of Fitzgerald Bay, along with the associated data collection (deployment of current meters and other oceanographic instrumentation for a period of 1 year). It is possible that a detailed model of Fitzgerald Bay could be embedded in the Spencer Gulf model being developed as part of Risk & Response (FRDC 2005/059), thus reducing the cost of such an undertaking. This model would provide an understanding of whether waste discharges from Whyalla are exported to Fitzgerald Bay, thus aiding in the interpretation of stable isotope analyses of seagrasses. In addition, the model could be used to examine patterns of waste deposition around cages (via the carbon deposition model produced as part of FRDC 2001/104), and if required could be integrated with a biogeochemical model to examine overall system-wide impacts of aquaculture, although this would probably only be justified if production increases substantially. Similar models could be developed for other production areas (such as Arno Bay).
6. Retrospective analysis of seagrass growth could be used to help clarify whether aquaculture is having an impact on seagrasses or not, as this question is not resolved by the analyses reported in Chapter 2. Such an analysis would allow a comparison of growth rates before and after aquaculture commenced at both control and putatively impacted sites, and thus allow the criteria for a fully rigorous environmental impact study to be met (e.g. Underwood 1993). These criteria cannot be met for any of the other variables utilised in this study, as comprehensive before data are not available.

For aquaculture over seagrass, further work needs to be done on developing the light model. Two notable gaps are the lack of a full-year of light data for the regions in the south-east that were studied. Ideally, the existing data should be supplemented by deploying light loggers for 2-3 weeks every 2-3 months, thus allowing a full-year assessment of the consequences of light reduction for the affected seagrasses. Also, many of the model parameters comes from studies on seagrasses in Western Australia, and it is not known if these are directly transferable to South Australia or not.

For land-based abalone aquaculture, several areas for potential future research have been identified above, and these need to be filled before some of the monitoring options discussed here can be implemented. These are:

1. Development and testing of potential integrated biological measures of water quality such as nitrogen content of seagrasses, and stable nitrogen isotope signatures of seagrasses.
2. Determine links between integrated biological water quality measures and environmental harm, e.g. at what level of nitrogen concentration in seagrass tissue does harm occur?
3. Determine the zone of influence from abalone farms on adjacent coastal water quality.
4. Determine the zone of influence from abalone farms on adjacent biological communities.
5. Accurately calculate total annual nutrient loads, which requires data collection on feed tonnage and influent/effluent volumes (which is currently done), abalone stock tonnage, feed and abalone nutrient composition, feed conversion ratios, and amount and composition of fertiliser added (which is not currently done).
6. Conduct a Before-After-Control-Impact (BACI) designed study on any new land-based abalone farms to determine unambiguously any effects of farming on the adjacent marine environment.
7. Conduct further fundamental research on the effects of nutrient loads on reef and seagrass communities to identify suitable biological indicators for environmental monitoring programs and to determine sustainable annual nutrient loads.

8.3 Planned outcomes

1. A comprehensive understanding of the current knowledge of the environmental effects of South Australian aquaculture, and an assessment of current information collection protocols. This will allow a more comprehensive approach to further data collection and monitoring to be developed.

   The outcomes of the review of current knowledge (as at the commencement of this project) of environmental effects are documented in Chapter 1. This review involved a thorough examination of environmental monitoring reports submitted to PIRSA Aquaculture, as well as both published and grey literature from relevant research agencies.

2. An assessment of industry perceptions as to likely and actual environmental impacts of aquaculture, which will form a key information resource for targeting field investigations.

   The perceptions of both industry and a range of other stakeholders were assessed in a series of environmental risk assessment workshops. These workshops were conducted for the marine finfish (excluding tuna), land-based abalone, and intertidal shellfish (oysters) sectors. The outcomes of these workshops, and subsequent literature reviews of high priority issues, are documented in Chapter 1.

3. An understanding of the actual environmental impacts associated with Yellowtail Kingfish, Salmon, Oyster and Abalone culture, that will allow future environmental monitoring programs to be designed on a firm scientific basis, and that will help in the identification of strategies to minimise future environmental impacts.

   The actual impacts of both yellowtail kingfish and land-based abalone aquaculture are comprehensively assessed in Chapters 2 and 4. There were only a few relatively minor impacts detected for either sector, despite sampling being conducted either immediately adjacent to cages for finfish, or at the outfall for abalone. Salmon could not be investigated, as this species is not currently being actively farmed in South Australia. However, a preliminary assessment of the potential consequences of light reduction to seagrasses in the currently designated finfish aquaculture zones in the south-east of South Australia was undertaken, and is presented in Chapter 3. A small study of intertidal oysters was conducted at one site only (Chapter 5), as PIRSA Aquaculture indicated that they did not wish to continue work on this sector prior to additional sites being sampled.

4. A series of optimal environmental monitoring programs for the industry sectors examined.

   A series of options for environmental monitoring for both marine finfish and land-based abalone are documented in Chapters 6 and 7. These options are based on the outcomes of the intensive field investigations, and provide PIRSA Aquaculture with a range of monitoring strategies that they could employ if desired.

Appendix 1: Intellectual property

This report will be made freely available to the public via FRDC, PIRSA Aquaculture, and SARDI?

Appendix 2: Staff

Dr Jason Tanner (Principal investigator)
Dr Simon Bryars (Co-investigator)
Ms Serena DeJong
Ms Sharon Drabsch
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Ms Rachel Wear
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Appendix 3: An investigation into the effect of finfish aquaculture on the demersal macrofauna in Fitzgerald Bay (Spencer Gulf, South Australia) using remote underwater video

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A research paper submitted in partial fulfilment of the requirements for the degree of Bachelor of Science with Honours

Supervisors: Dr Jason Tanner and Dr Simon Bryars

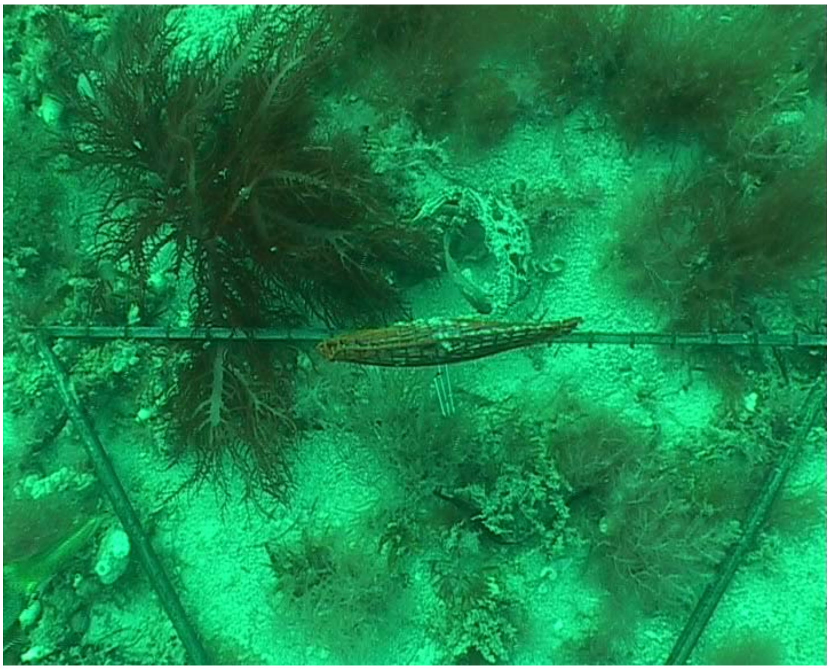

Figure A3.1. Photograph of several macrofaunal species (juvenile snapper – *Pagrus auratus*, blue swimmer crab – *Portunus pelagicus*, southern calamary – *Sepioteuthis australis*) extracted from footage recorded by the baited remote underwater video technique used in the present study.

A3.1 Abstract

Wild-fish aggregation is an environmental effect of finfish aquaculture that has been reported from the Mediterranean, Scotland, Norway and Australia. Wild macrofauna often contribute to waste mitigation around fish farms. Baited remote underwater video (RUV) surveys in Fitzgerald Bay (northern Spencer Gulf, South Australia) detected no differences

among the demersal macrofaunal assemblages on sites containing sea-cages and on sites located at least one kilometre from finfish aquaculture. The demersal assemblages present within Fitzgerald Bay were found to be similar to those in two other areas of upper Spencer Gulf that were not used for any form of aquaculture. The assemblages in the bay sampled by the RUV were found to vary over time. Significant spatial heterogeneity in the habitat was observed within both Fitzgerald Bay and another nearby location, and these habitat differences (mixed sponge/macroalgae vs. bare sand) might explain the assemblage differences detected at the two locations. RUV was found to be a suitable method for macrofaunal surveys around fish farms and bait was found to enhance its effectiveness. The study suggests that demersal macrofauna do not play a major role in waste mitigation around the fish farms in Fitzgerald Bay.

A3.2 Introduction

Sea-cage aquaculture has expanded rapidly during the past few decades, and now occurs in more than 60 countries worldwide (FAO 2002). In 2002, global sea-cage production accounted for 2.4 million tonnes of finfish (FAO 2002), which equates to approximately 69 000 sea-cages when assuming individual cage biomass is 35 tonnes (pers. comm. - Pheroze Jungalwalla, Executive Officer, Tasmanian Salmonid Growers Association). Due to the size and expansive nature of the sea-cage aquaculture industry, there is now greater emphasis on the identification and assessment of its environmental effects. Efforts to date have concentrated on many biological and chemical aspects, including impacts associated with eutrophication, escapes, underlying sediment condition and diseases/parasites (Karakassis 1999; Naylor *et al.* 2000; Pearson and Black 2001; Loo *et al.* 2004). Despite the attention directed towards sea-cage farming, many ecological effects of sea-cages have received relatively scant attention. In particular, little is known about the ecological implications for mobile macrofauna directly associated with sea-cage aquaculture developments. The aggregation of wild fish species around sea-cages is a case in point. This phenomenon has been reported throughout the Mediterranean (Spain(Dempster *et al.* 2002; Dempster *et al.* 2004), France (Pergent *et al.* 1999), Greece (Papoutsoglou *et al.* 1996; MacDougall and Black 1999; Thetmeyer *et al.* 2003), Israel (Golani 2003), and also in Norway (Bjordal and Johnstone 1993), Scotland (Carss 1990) and Australia (Felsing *et al.* 2002; Dempster *et al.* 2004), but has received little detailed investigation.

The attractive effect of sea-cages is assumed to be due to a combination of factors; habitat provision (Papoutsoglou *et al.* 1996), resource supply either directly (excess feed, faeces) or indirectly (prey species associated with sea-cages) (Pearson and Black 2001), and possibly chemical attraction elicited by farmed stock (Dempster *et al.* 2002). Whilst aggregation is the primary ecological effect on wild fauna, there are further environmental and ecological consequences associated with these aggregations that are poorly understood and vary between locations. Flow-on effects can include waste mitigation (Gowen and Bradbury 1987; Papoutsoglou *et al.* 1996; Felsing *et al.* 2002), disease/parasite transfer (Saunders 1991; Bjorn *et al.* 2001), changes in local assemblage composition (Thetmeyer *et al.* 2003), and altered reproductive output and fishing success (Bjordal and Johnstone 1993; MacDougall and Black 1999; Dempster *et al.* 2002; Dempster *et al.* 2004). If fishing is prohibited, aquaculture sites could function as marine protected areas (Dempster *et al.* 2002), and enhance local stocks by both increasing reproductive output (Chiappone and Sullivan 2000) and providing emigrants to the surrounding environment (McClanahan and Mangi 2000). Aggregating fauna may also contribute to waste mitigation by consuming cage waste (excess feed, faeces, bio-fouling) (Katz *et al.* 2002), and at the same time improve their growth and reproductive condition (Dempster *et al.* 2004). Conversely, where legislative protection from fishing is not afforded, aggregations around sea-cages are easy targets for fishermen (Morgan 2000; Dalgetty 2002), which may exacerbate the over-exploitation of stocks (Dempster *et al.* 2004). In most cases, however, the interplay between the positive and negative ecological consequences associated with sea-cage aggregations is much more complicated.

The first step towards understanding the implications of, and to, wild stock assemblages associated with sea-cages is to determine their composition. Nonetheless, the composition of macrofaunal assemblages associated with sea-cages has rarely been examined, with only four previous studies having been undertaken (Carss 1990; Dempster *et al.* 2002; Golani 2003; Dempster *et al.* 2004) and one currently underway (Thetmeyer *et al.* 2003). Most of this research investigates wild fish aggregations on aquaculture sites in the Mediterranean, where fish are attracted to sea-cages in large numbers (Thetmeyer *et al.* 2003). Two previous studies have investigated the wild fauna that associate with finfish aquaculture developments in Australian waters. Only one of these studies, which reported extensive aggregation of pelagic fish species around sea-cages off the northern coast of New South Wales (Dempster *et al.* 2004), investigated the composition of the assemblage. No research of this kind has been performed in South Australia, despite marine finfish production using sea-cages being the largest sector of the state's aquaculture industry (ABARE 2004) with major operations in Boston Bay, Arno Bay and Fitzgerald Bay. Consequently, the present study sought to survey the mobile macrofaunal communities associated with finfish aquaculture in Fitzgerald Bay, South Australia.

To date, the most commonly used survey method for fish has been diver conducted transects. *Dempster et al.* (2002, 2004), Golani (2003) and Thetmeyer (2003) used diver transects to sample aquaculture sites in areas of the Mediterranean where conditions were suitable. However, dive operations are not always possible around sea-cages in South Australian waters, or indeed on many aquaculture sites worldwide. Environmental constraints that limit the applicability of diver-conducted surveys include restricted visibility, strong currents, depth and temperature of the water, and the potential presence of sharks. The additional expense and weather dependent nature of dive operations also often discourages their use, especially when suitable alternatives are available.

The potential of remote techniques for macrofaunal surveys has been highlighted by many authors in recent times (Ellis and DeMartini 1995; Willis *et al.* 2000; Cappo *et al.* 2002). During the past decade there has been a gradual shift towards the use of remote techniques to sample environments that are not accessible with traditional diver-conducted surveys, and now these methods are even being used in areas that were formerly sampled exclusively by divers (Okamoto 1989; Ellis and DeMartini 1995; Francour *et al.* 1999; Willis *et al.* 2000). The advantages of remote techniques stem from the fact that they are not subject to the limitations imposed upon divers by factors such as depth, temperature, time and safety requirements (Cappo *et al.* 2002). Remote techniques comprise both destructive and non-destructive forms, each with inherent strengths and weaknesses (Charbonnel *et al.* 1997; Francour 1999). Destructive sampling results in the death of the sampled organisms, and often the destruction of associated habitats. Examples include trawls (Wassenberg *et al.* 1997), angling (Ellis and DeMartini 1995), poisons (Willis and Anderson 2003) and nets (Carss 1990). The presence of on-site infrastructure (e.g. cages, nets, moorings, anchors), stock, and vessels involved in the operation of a commercial sea-cage aquaculture site prevents the application of most destructive methods. Conversely, many non-destructive remote techniques are ideally suited to sea-cage aquaculture and provide several inherent advantages over traditional diver surveys, as well as the universal benefits of remote techniques mentioned above. Non-destructive remote methods avoid the behavioural modifications induced in fish by the presence of divers (Francour *et al.* 1999), do not harm the species or the habitat sampled and can provide information on the habitat and species behaviour (Cappo *et al.* 2002).

Commonly used non-destructive remote methods include a variety of forms of remote underwater video (RUV)(Moser *et al.* 1998; Francour *et al.* 1999; Harvey *et al.* 2002; Cappo *et al.* 2003; Willis *et al.* 2003; Meekan and Cappo 2004) and remotely operated vehicles (ROVs) (Alin *et al.* 1999; Norcross and Mueter 1999; Caselle *et al.* 2002), which are all candidates for use around sea-cages. Although non-destructive remote techniques have been used previously to survey many types of fish species (Sainte-Marie and Hargrave 1987; Priede *et al.* 1994; Ellis and DeMartini 1995; Willis *et al.* 2000; Trenkel *et al.* 2002; Svane and Saunders 2003), in numerous marine environments (Priede and Merrett 1996; Cappo *et al.* 2003; Willis *et al.* 2003), they have never been applied to an aquaculture setting. The present study developed a form of RUV suitable for use around sea-cages. This method used bait as an attractant, as have many previous remote survey techniques (Sainte-Marie and Hargrave 1987; Priede and Merrett 1996; Meekan and Cappo 2004). Different baits are known to selectively attract different species (pers. comm. – Dr. Trevor Willis, Universita di Bologna), however the extent of this effect has never been investigated. In this study a comparison of bait types will be undertaken at sea-cages and control sites to evaluate the variability in sample composition elicited by bait.

Sea-cage aquaculture provides a varied environment, resulting in the high abundance and diversity of species encountered on many aquaculture sites (Carss 1990; Dempster *et al.* 2002; Thetmeyer *et al.* 2003). The spatial (ie. pelagic, demersal, benthic), temporal (diurnal, seasonal) and trophic (ie. piscivorous, planktivorous, herbivorous, detritivorous) distribution of the resident assemblages prevents one technique from adequately surveying the entire suite of species. Therefore the identification and sampling of faunal components within the assemblage is required (Lincoln Smith 1989), taking into consideration the behavioural and ecological traits of the resident species (Samoilys and Carlos 2000). For the present investigation, there are at least two distinct components of the sea-cage associated macrofauna that need to be considered: pelagic and benthic. The pelagic component comprises fish species that are often associated with the mid-water structure provided by sea-cage nets (Dempster *et al.* 2002; Thetmeyer *et al.* 2003; Golani 2003). The benthic component is associated with the underlying substrate and may comprise a range of taxa, including fish, crustacea and demersal elasmobranchs. Each of these faunal divisions may incorporate species with a range of behaviours, such as being cryptic, schooling and seasonally present, which can complicate survey efforts further.

Wild fish species have been shown to play an important role in the mitigation of waste from finfish aquaculture cages both in Mediterranean (Thetmeyer *et al.* 2003) and Australian (Felsing *et al.* 2002) waters. Despite this, the species involved in waste mitigation have received little direct attention in most relevant studies, mainly due to the use of indiscriminate sampling methods (see Carss 1990; Dempster *et al.* 2004). The RUV survey technique in the present study targets the demersal macrofauna that are likely to be involved in waste mitigation around sea-cages by using bait to attract these species. The major objective of the present study was to determine whether finfish aquaculture has affected the demersal macrofaunal assemblages in Fitzgerald Bay. The demersal assemblages were sampled by baited RUV and compared on a local scale (between sites - aquaculture vs no aquaculture) within Fitzgerald Bay, regional scale (with other nearby locations that do not contain finfish aquaculture) and over time to detect any differences attributable to aquaculture.

A3.3 Methods

A3.3.1 Study area

Fitzgerald Bay is located in northern Spencer Gulf, South Australia (Figure A3.2a). Sea-cage aquaculture has been undertaken within the bay continuously since 1999, initially producing snapper (*Pagrus auratus*) but now exclusively farming yellowtail kingfish (*Seriola lalandi*). Currently there are five 20 hectare lease sites in Fitzgerald Bay (Figure A3.2b), four of which contain stock, with a combined annual production of approximately 620 tonnes. The sites containing fish are distributed along a channel that runs through Fitzgerald Bay, to the west of an offshore sandbank. The channel ranges in depth from 10-23 m and experiences substantial tidal flows (up to 39.1 cm/sec, Parsons Brinckerhoff and SARDI 2003). Current direction is approximately north-south along the channel, alternating every six hours in a semi-diurnal pattern. The two lease sites chosen for the study were located at either end of the channel, to allow for the selection of suitable control sites (Figure A3.2b). The benthic habitat is variable throughout the bay apart from a continuous narrow coastal fringe of seagrass in shallower depths (less than 6 to 8 m, (Shepherd 1974; Hone *et al.* 1996). Therefore control sites were selected to be as similar as possible to each lease in terms of geographic location and water depth. Twenty hectare control sites were used to allow for site-level spatial variation. A minimum distance of one kilometre was specified between any lease and control site to avoid possible impacts associated with aquaculture development.

Two sizes of sea-cage are present in Fitzgerald Bay; 80 and 120 m circumference net collars, with the latter used far less commonly. The 80 m cages were the only type sampled in this study. Cages are of standard sea-cage design, with a weighted net suspended below a buoyant ring, which is moored to the seafloor with four anchors spaced equidistantly around the net collar (Figure A3.3). Net depths on 80 m cages range between 6-8 m, resulting in a distance of 5-15 m between the net and the seafloor. All stocked sea-cages are fed once per day with extruded pellets.

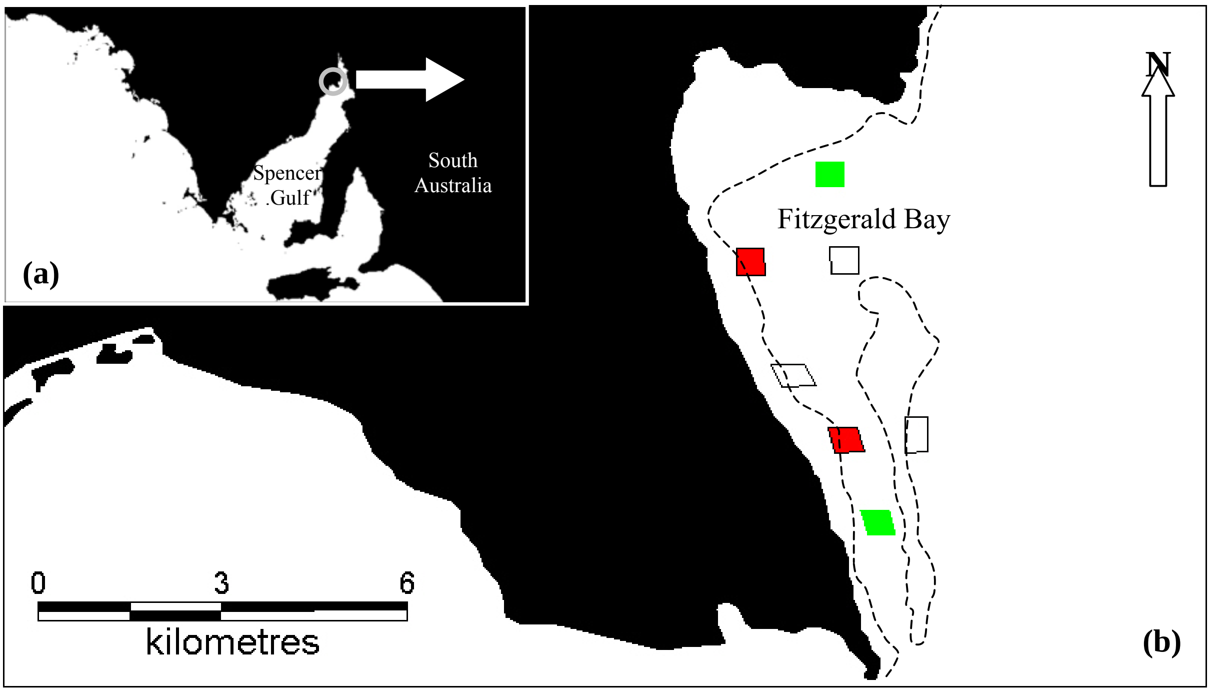

Figure A3.2. Map showing the location of (a) Fitzgerald Bay within South Australia and (b) the lease and control sites used in the present study (black outline boxes = licensed lease sites, red boxes = lease sites sampled in this study, green boxes = control sites, dashed line = 10 m depth contour).

A3.3.2 RUV deployment

Benthic RUV was chosen as the sole survey technique for the main part of the study following the identification and assessment of potential survey techniques suitable for use in Fitzgerald Bay, and investigation of the composition of wild faunal assemblages found adjacent to the sea-cages there (Appendix A3.A). All sampling was undertaken during daylight hours (0800 – 1700) using two RUVs and a small (6 m) research vessel as a platform for operations. Lease site deployments were made with reference to randomly selected sea-cage sectors. Each sea-cage sector was never sampled more than once during a field trip. Lease site deployments were made within 5 m of a sea-cage, and at least an hour after feeding had ceased at a cage. Control sites were divided into 5 by 5 grids (i.e. 25 cells), cells were randomly chosen and RUVs were deployed at their midpoint. Successive RUV deployments were usually made 2-10 minutes apart, separated by a minimum distance of 200 m, but as much as several kilometres depending upon the weather conditions. Once set, the boat was moved a suitable distance away from the RUVs (>200 m) and the motors turned off until retrieval.

Two Amphibico Dive Buddy housings were used with the RUVs; one containing a Sony Digital Handycam DCR-TRV20E, the other a Sony 3CCD Network Handycam DCR-TRV950E. Cameras were mounted with a distance of 1 m between the lens and the seafloor, which resulted in an estimated "out of water" FOV of 1220 mm by 860 mm. Deployment lengths of 30 minutes were chosen based on the early arrival times and low species numbers detected in a pilot study (maximum numbers of species (1-4) usually occurred before 20 minutes recording time had elapsed). A single small (~400 g) pack of frozen brined pilchards was used as bait for each deployment. Prior to placement in a bait basket, pilchards were thawed and crushed to maximise the bait plume.

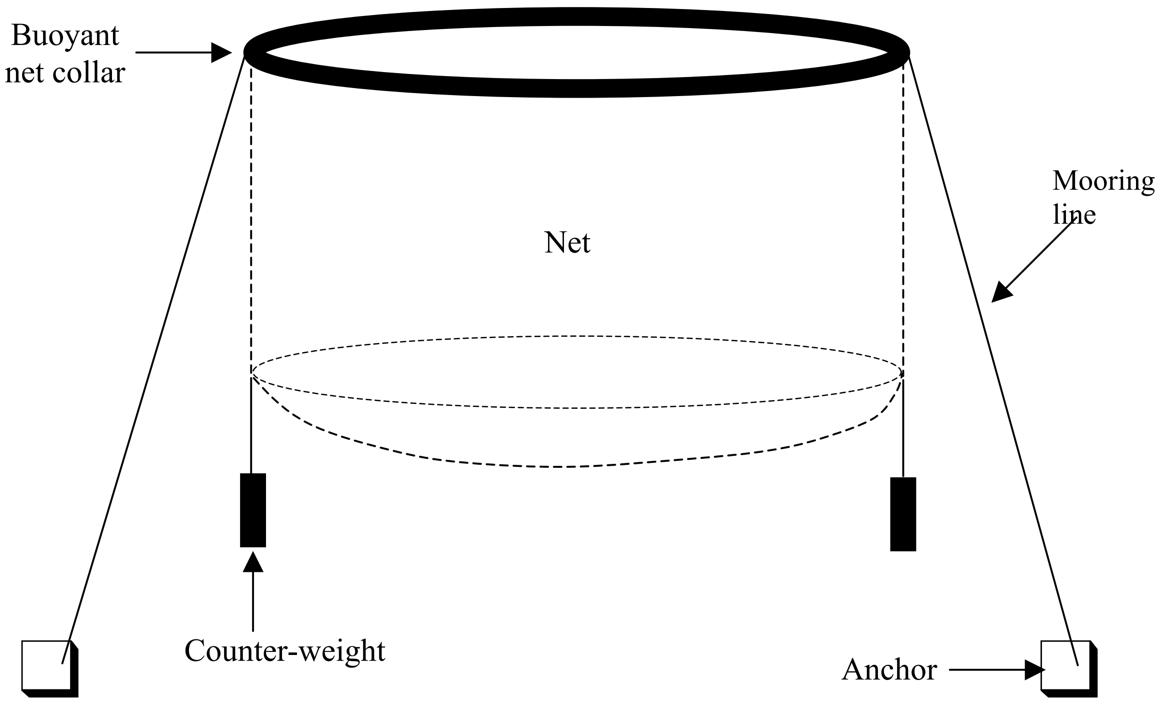

Figure A3.3. Schematic diagram of a typical sea-cage used in Fitzgerald Bay (NB. Only two anchors shown for illustrative purposes).

A3.3.3 Footage analysis

Footage was viewed using a Sony MiniDV recorder with a real-time counter, and analysis commenced from the moment that the RUV settled on the seafloor. Only mobile macrofauna were counted, therefore sessile organisms such as razorfish (*Pinna bicolor*) and ascidians were ignored. Relative abundance estimates were made by recording the maximum number of individuals of a single class visible within one frame of footage (defined as "MaxN" by (Cappo *et al.* 2002). MaxN is a conservative measure of relative abundance because it usually underestimates the true numbers of species visiting the bait (Cappo *et al.* 2002). Using MaxN avoids the problem of recounting the same individual on separate visits to the bait, and has been found to be an accurate estimate of "true" density (Willis *et al.* 2000). Being a conservative estimate of relative abundance does, however, mean that any differences detected between areas of high and low faunal densities are likely to be understated (Cappo *et al.* 2002). Due to difficulties with identifying small cryptobenthic fish species from the dorsal view recorded by the RUVs, these species were grouped in a "benthic" category. The presence of two distinct cohorts of snapper (*Pagrus auratus*) in the surveys necessitated separation of the classes for statistical analysis based upon the minimum legal length (juvenile <38 cm, adult >38 cm). Some *Portunus pelagicus* individuals were easily distinguished from others (e.g. male/female, missing claw, markings) and thus each new arrival in the FOV was included in the MaxN count regardless of whether they were all present in one frame of footage. Qualitative observations of the visible benthic environment were made for each deployment to provide habitat information for each site. Species identification was verified with the use of several reference books (Gommon *et al.* 1994; Edgar 1997; Poore 2004) and via consultation with experts at SARDI Aquatic Sciences (Graham Hooper, Paul Jennings).

A3.3.4 Statistical analyses

Non-parametric multivariate analysis of variance (NPMANOVA Anderson 2001) was used to test for differences in assemblage composition between treatments. All NPMANOVA analyses in the present study were restricted to two factor designs as this is the maximum of factors that the current version of the program allows. The analysis is based on permutation tests and calculates a test-statistic analogous to Fisher's F-ratio. The Bray-Curtis coefficient was chosen for all analyses because it disregards joint absences between samples. No transformation of the data was required as no taxa contributed disproportionately to the data. A consistent number of permutations (4999) was used throughout. Pair-wise *a posteriori* comparisons were made for factors that were found to have a significant effect. To visualise the similarities between samples, non-metric multi-dimensional scaling (nMDS) ordination plots were generated using the PC-ORD program (MjM software, Oregon(McCune and Grace 2002). The Bray-Curtis distance measure was again used. To test for differences in sample richness (i.e. species per RUV deployment) between treatments univariate analyses were performed using the SPSS program (Version 10, 1999), which comprised one-way ANOVAs and post hoc analysis with Tukey's HSD test.

A3.3.5 First survey: Local effects

To detect the local-scale effects of finfish aquaculture, RUVs were used to survey the benthic mobile macrofauna present on lease and control sites in Fitzgerald Bay. A two-way orthogonal sampling design was used, with site (lease/control) and location (north/south) as two fixed factors. Sampling was undertaken on the 22nd, 24th and 25th June 2004 during which time 24 deployments (2 lease sites + 2 control sites x 6 replicates) were conducted. Throughout this sampling period, nine 80 m sea-cages were present on the northern lease site and three on the southern lease. A 120 m cage containing fish was also present on the southern lease, however it was not included in the survey. To determine whether the tide in Fitzgerald Bay had an effect on the RUV sample composition, the deployments were structured using the tidal phase. Three replicates were conducted on each site around the high tide and three around the low tide. Tide and site were treated as fixed factors in a 2-way experimental design.

A3.3.6 Second survey: Regional effects

Commercial finfish aquaculture sites have been operational throughout Fitzgerald Bay for approximately five years, thus potentially having the scope to affect the ecosystem of the entire bay. A true before-after-control-impact (BACI)(Green 1979) design could not be implemented due to the absence of suitable data prior to the establishment of sea-cage aquaculture in Fitzgerald Bay. Therefore two comparable areas that have never contained finfish aquaculture were selected to assess whether the macrofaunal assemblages within

Fitzgerald Bay are typical of the upper Spencer Gulf region. The two Fitzgerald Bay control sites were sampled once again, as were two 20 hectare sites both 28 kilometres to the north (Douglas Point) and 22 kilometres to the south (Cowleds Landing) of Fitzgerald Bay (Figure A3.4). Sites within each area were positioned to match those in Fitzgerald Bay in terms of water depth, separation and site dimensions (Figure A3.4). The distance offshore was similar for the Douglas Point (600–1400 m) and Fitzgerald Bay (400–1100 m) sites, however, sufficient depth was only accessible further offshore (3.5-7.5 kilometres) in the vicinity of Cowleds Landing. RUVs baited with crushed pilchards were used to sample the macrofaunal assemblages in all areas. A total of 36 deployments (6 sites x 6 replicates) were conducted over three days (27th, 28th and 29th July, 2004). Sampling was undertaken at two times each day: morning and afternoon, with 6 deployments at each time. Area and site were treated as fixed factors, with site nested in area for a 2-way experimental design.

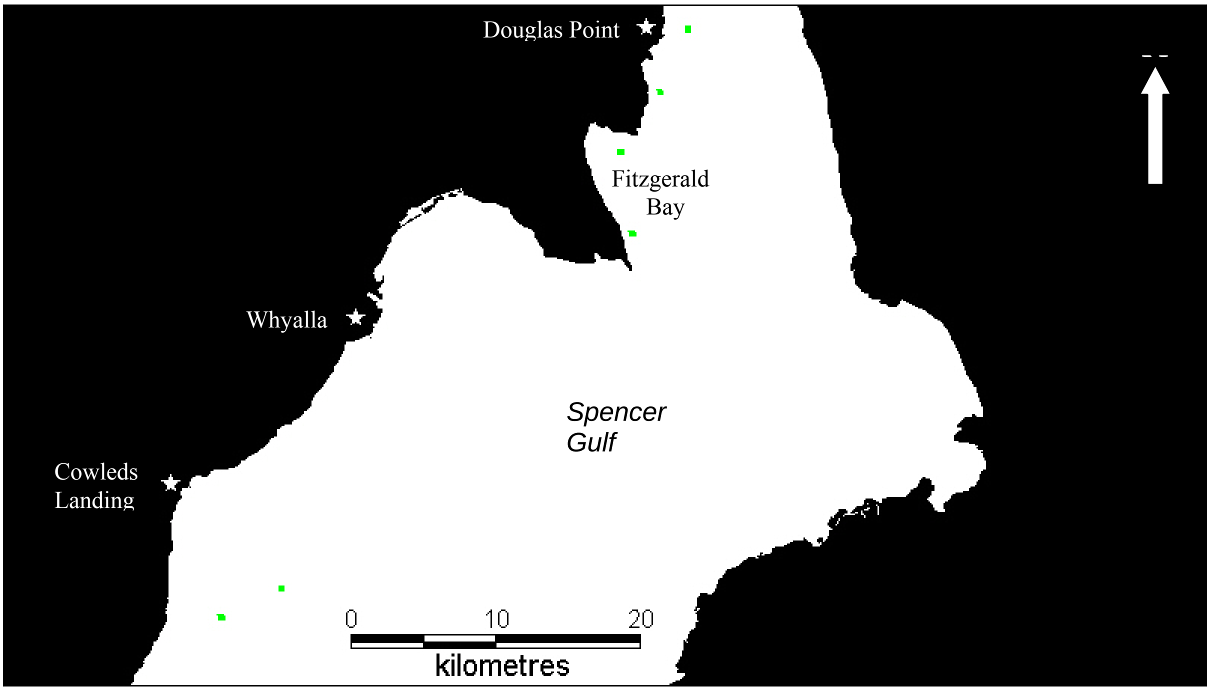

Figure A3.4. Site locations for the regional surveys within the upper Spencer Gulf at Douglas Point, Cowleds Landing and Fitzgerald Bay (green boxes = sites sampled in the second survey).

A3.3.7 Third survey: Bait and temporal effects

To evaluate bait efficacy and the effect that different bait types had on the sample composition of RUV surveys in Fitzgerald Bay, three bait treatments were assessed: crushed pilchards, extruded aquaculture pellets and no bait. The pilchard treatment matched the bait used in previous surveys. Pellets used for daily feeding were sourced directly from the aquaculture operators (9 mm diameter). The “no bait” treatment consisted of an empty bait basket. Sampling was undertaken throughout the day on three consecutive days (31st August, 1st and 2nd September), during which time the northern lease site contained eight 80 m sea-cages and the southern lease contained six. Each bait treatment was applied to each of the two lease and two control sites surveyed previously (Figure A3.2b) in Fitzgerald Bay (3 baits x 4 sites x 5 replicates = 60 deployments). Strong tides during sampling resulted in the loss of several deployments from the southern sites and thus a reduction in the number of usable replicates from five to three. To include these sites in the NPMANOVA analysis, which required a balanced design, all other treatment levels would have had to be reduced to three replicates also. To avoid the loss of statistical power associated with such a reduction, the complete data (n=5) set from the northern sites was used to compare bait types. Bait type and site (aquaculture/control) were treated as fixed factors in a 2-way experimental design.

To determine whether the effects of finfish aquaculture varied over time and examine the temporal stability of the assemblages within Fitzgerald Bay, a temporal comparison of RUV samples was undertaken. The pilchard data from the third survey (31st August – 2nd September) allowed such a comparison with the results from the first survey (22nd – 25th June). The first survey found that the assemblages differ between areas of the Bay. Therefore to determine whether temporal effects are consistent among assemblages, all four sites were included in the analyses. Due to the data loss mentioned above, all treatments from the first survey were randomly reduced to three replicates. Sites (analysed as four individual sites) and time were treated as fixed factors in a 2-way experimental design. The interaction term was used to determine whether the differences between sites were consistent over time.

A3.4 Results

A3.4.1 Local effects

There was no apparent effect of aquaculture on the demersal macrofaunal assemblages in Fitzgerald Bay (site: lease vs control, NPMANOVA: $F_{1,20} = 0.21$, $P = 0.96$) and this lack of an effect was geographically consistent (site x location, NPMANOVA: $F_{1,20} = 1.84$, $P = 0.10$). The assemblages did, however, differ between locations within the Bay (location: north vs south, NPMANOVA: $F_{1,20} = 6.41$, $P = 0.0002$). The geographic separation can clearly be seen in the nMDS plot (Figure A3.5). Tidal phase at the time of sampling did not affect species composition (NPMANOVA: $F_{1,20} = 0.92$, $P = 0.47$, (Figure A3.5), permitting subsequent deployments to be undertaken throughout the tidal cycle.

Fitzgerald Bay had low taxonomic richness, with only six separate taxa observed in the 24 RUV deployments performed during this part of the study (Appendix A3.B, Table A3. 1). Five of these taxa comprised only one species, which were easily recognisable in the footage, while the "benthic" category may have incorporated 2-3 separate species, as accurate identification and discrimination was impossible from the dorsal view recorded in the footage. Western king prawns (*Melicertus latisulcatus*), skipjack trevally (*Pseudocaranx wrighti*), juvenile snapper (*Pagrus auratus*) and small cryptobenthic finfish ("benthic" category) were recorded on both the northern and southern sites. Blue swimmer crabs (*Portunus pelagicus*) and Port Jackson sharks (*Heterodontus portusjacksoni*) appeared solely on the southern sites. The highest species richness was recorded on the southern lease (n=5), with the remaining three sites having identical species richness values (n=4, Figure A3.6). The number of species sampled per RUV deployment varied significantly between sites (One-way ANOVA: $F_{3,20}= 3.61$, $P = 0.03$, Figure A3.6). Post hoc Tukey HSD analysis revealed that the southern lease had a significantly higher number of species per deployment than both the southern control ($P = 0.047$) and northern lease ($P = 0.047$) sites.

Qualitative analysis of the video footage revealed that the southern lease incorporated two habitat types; coarse substrate with substantial coverage of macroalgae and sponges ("mixed" habitat) on the near-shore edge, and very sparsely vegetated soft substrate ("bare" habitat) further offshore. The other three sites contained only one habitat type each. The southern control site is characterised by the mixed habitat, while both northern sites contained the bare habitat type.

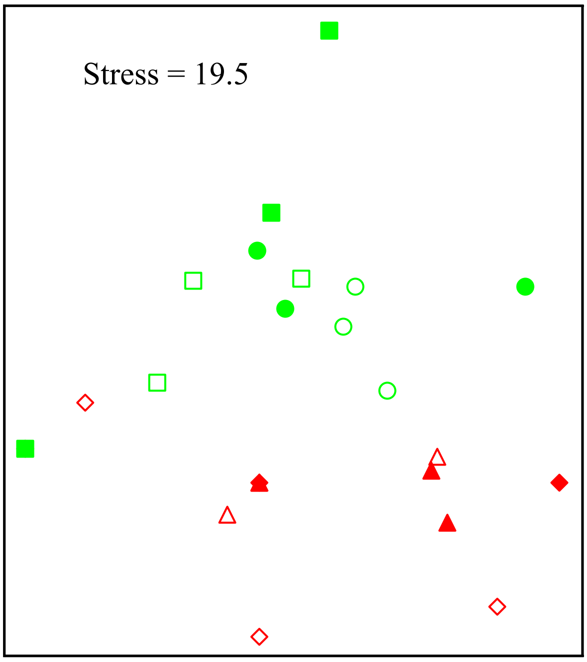

Figure A3.5. Non-metric MDS plot of the macrofaunal assemblages sampled at high and low tide from four sites within Fitzgerald Bay ▲ = northern control, high tide, ∆ = northern control, low tide, ♦ = northern lease, high tide, ◇ = northern lease, low tide, ● = southern control, high tide, ○ = southern control, low tide, ■ = southern lease, high tide, □ = southern lease, low tide. While the stress value is high, it still indicates a useable plot.

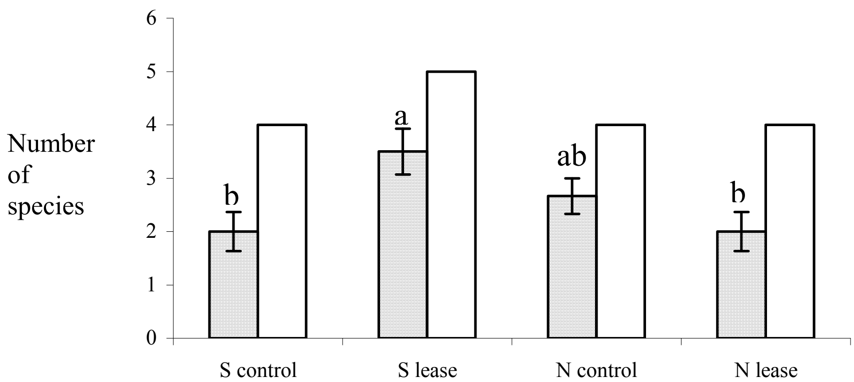

Figure A3.6. Average number of species ± standard error recorded per RUV deployment (shaded) and total species richness (clear) at the northern (N) and southern (S) lease and control sites in Fitzgerald Bay. Letters indicate the results of post-hoc tests (α = 0.05).

A3.4.2 Regional effects

There were no regional differences in the demersal assemblages surveyed at the three areas within upper Spencer Gulf (NPMANOVA: $F_{2,30}$ = 0.58, P = 0.94), as evidenced by the substantial overlap of locations in the nMDS plot (Figure A3.7). There was, however, a significant difference among sites within each location (NPMANOVA: $F_{3,30}$ = 4.23, P = 0.002). Post hoc pairwise comparisons of the sites at each location indicated that there was spatial heterogeneity at both Douglas Point (t = 2.06, P = 0.004) and Fitzgerald Bay (t = 3.65, P = 0.002), although not at Cowleds Landing (t = 1.30, P = 0.08). Qualitative observation of the RUV footage revealed that both Cowleds Landing sites were similar, characterised by varying levels of seagrass cover (*Heterozostera tasmanica*) interspersed with patches of bare sediment. The sites within the remaining two locations were noticeably different: one containing the bare habitat, the other the mixed habitat. These habitat differences were reflected in the diversity and types of species recorded on each site. A total of eight species were recorded on the vegetated sites at both Douglas Point and Fitzgerald Bay, whereas on the unvegetated sites only four were observed.

Of the 11 species recorded (Appendix A3.B, Table A3.2), all 11 were found at Cowleds Landing, 5 at Fitzgerald Bay and 8 at Douglas Point. Cowleds Landing 2 had the highest total species richness for an individual site (n=7), with the lowest species richness values being found on the two sites where only unvegetated sediment was visible in the footage; Fitzgerald Bay 1 (n=2) and Douglas Point 2 (n=3) (Figure A3.8). There were significant differences in the number of species sampled per RUV deployment among sites (One-way ANOVA: $F_{5,30}$ = 3.67, P = 0.01, Figure A3.8). However, only Douglas Point 1 and 2 were shown to be significantly different during post hoc analysis (Tukey's HSD test: P = 0.032).

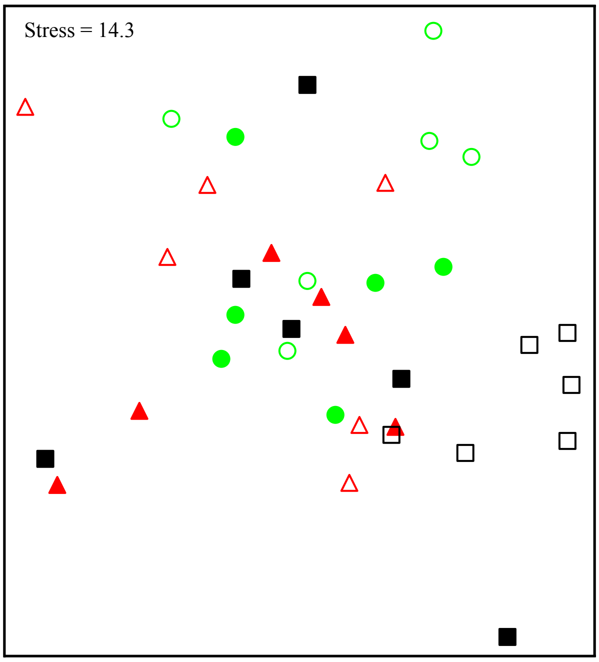

Figure A3.7. Non-metric MDS plot of the macrofaunal assemblages sampled from three locations within upper Spencer Gulf ▲= Douglas Point 1, △ = Douglas Point 2, ● = Fitzgerald Bay 1, ○ = Fitzgerald Bay 2, ■ = Cowleds Landing 1, □ = Cowleds Landing 2.

A3.4.3 Bait effects

Bait type was found to significantly affect the results of the RUV surveys undertaken within Fitzgerald Bay (NPMANOVA: $F_{2,24} = 4.73$, $P = 0.001$, Figure A3.9). Pair-wise post hoc comparisons revealed that both aquaculture pellets ($t = 2.03$, $P = 0.02$) and pilchards ($t = 3.09$, $P = 0.0006$) produced samples that were significantly different to those of an unbaited RUV, but that there were no differences between the bait types ($t = 1.13$, $P = 0.30$). Consistent with the initial local effects survey, the presence of finfish aquaculture did not affect the composition of the demersal assemblages (NPMANOVA: $F_{1,24} = 1.08$, $P = 0.38$, Figure A3.11).

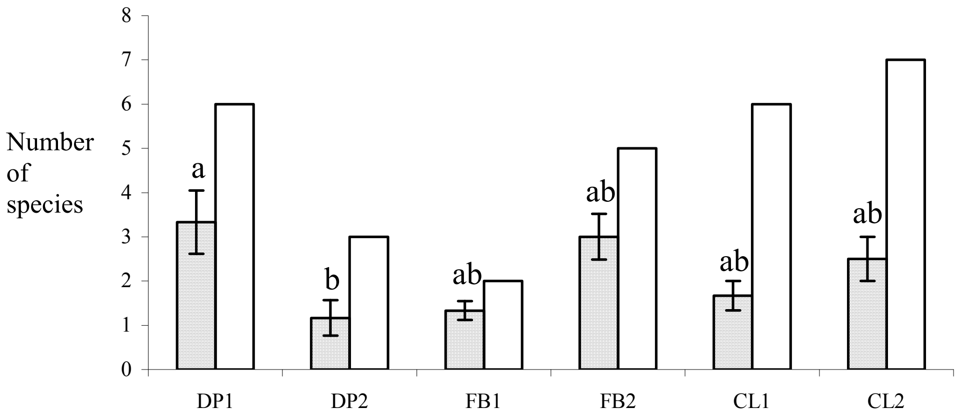

Figure A3.8. Average number of species ± standard error per RUV deployment (shaded) and total species richness (clear) from two sites at each of three locations in upper Spencer Gulf. DP = Douglas Point, FB = Fitzgerald Bay, CL = Cowleds Landing. Letters indicate the results of post-hoc tests ($\alpha = 0.05$).

Only three species categories were recorded on the northern Fitzgerald Bay sites during this study: *Pseudocaranx wrighti*, adult *Pagrus auratus* and small cryptobenthic finfish species (the "benthic" category). Benthic species and *Pseudocaranx wrighti* were recorded on both the control and lease sites, however adult *Pagrus auratus* were only detected on the lease site. All three species categories were sampled by the RUV deployments that used pilchards and pellets, but only the benthic species category was recorded in the unbaited treatment. Furthermore, seven of the ten unbaited deployments recorded no mobile macrofauna. In contrast, only two replicates of the pellet treatment and one of the pilchard treatment failed to record any species. Qualitative observations of the RUV footage were similar between sites, depicting the bare habitat type described previously.

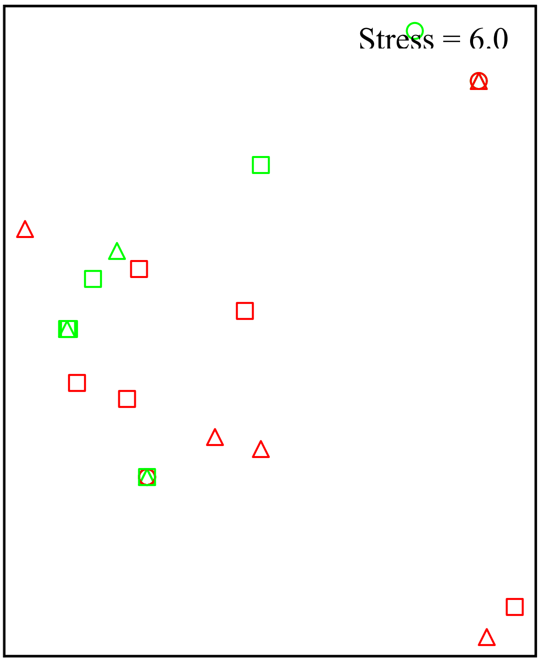

Figure A3.9. Non-metric MDS plot of the macrofaunal assemblages sampled using three different baits from a lease and control site within Fitzgerald Bay △ = pellets, control site, △ = pellets, lease site, ○ = no bait, control site, ○ = no bait, lease site, □ = pilchards, control site, □ = pilchards, lease site.

A3.4.4 Temporal effects

In the nine weeks between the first (T1 = 22nd – 25th June) and third (T2 = 31st Aug – 2nd Sept) surveys, the demersal assemblages within Fitzgerald Bay had changed (NPMANOVA: $F_{1,16}$ = 7.41, P = 0.0002, Figure A3.10). Differences were again detected between sites (NPMANOVA: $F_{3,16}$ = 3.92, P = 0.0004) and these differences were consistent over time (NPMANOVA: $F_{3,16}$ = 1.15, P = 0.32). Post hoc pair-wise comparisons revealed a significant difference between the northern and southern control sites (t = 2.84, P = 0.001) and the northern lease and southern control sites (t = 2.20, P = 0.002) for both sampling periods. These effects are related to geographic position, and not to the presence of aquaculture.

The number of species sampled per RUV deployment was similar for all sites during the second survey (One-way ANOVA: $F_{3,8}$ = 1.29, P = 0.34). Although sample richness had decreased between surveys for three of the sites (Figure A3.11), the only significant difference was on the northern control site (One-way ANOVA: $F_{1,7}$ = 6.22, P = 0.04). Total species richness also decreased for three of the sites, with the southern lease remaining unchanged (Figure A3.12)

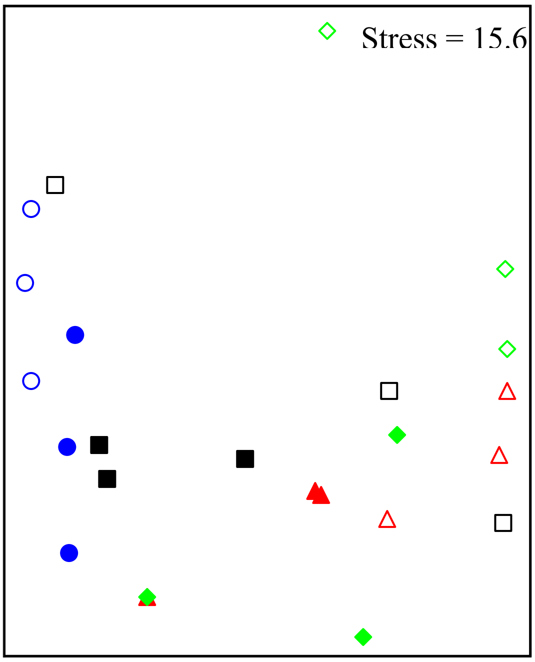

Figure A3.10. Non-metric MDS plot showing the effect of time (T1 – first survey, T2 – third survey) and site location on the macrofaunal assemblages in Fitzgerald Bay ▲= northern control, T1, △ = northern control, T2, ♦ = northern lease, T1, ◇ = northern lease, T2, ● = southern control, T1, ○ = southern control, T2, ■ = southern lease, T1, □ = southern lease, T2.

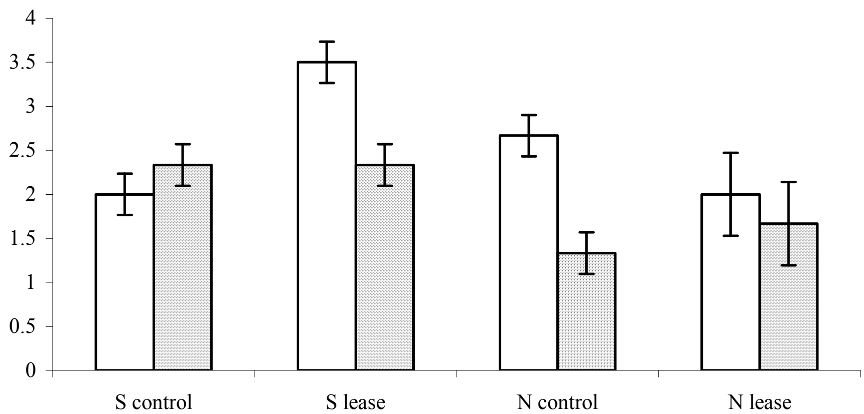

Figure A3.11. Average number of species ± standard error recorded per RUV deployment for the first (clear) and third (shaded) surveys on the northern (N) and southern (S) lease and control sites in Fitzgerald Bay.

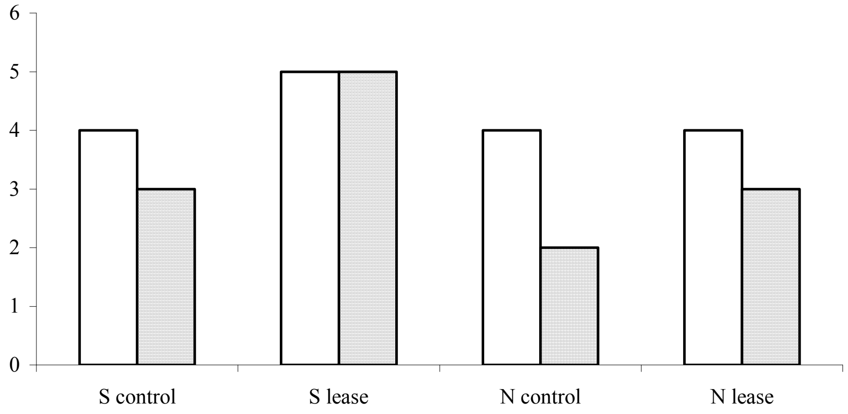

Figure A3.12. Total species richness values recorded during the first (clear) and second (shaded) surveys on the northern (N) and southern (S) lease and control sites in Fitzgerald Bay.

A combined total of eight taxa were recorded over the two surveys, with six being observed during each trip (Appendix A3.B, Table A3.3). Species absent from the first survey (but present in the second) were adult *Pagrus auratus* and bridled leatherjackets (*Acanthaluteres spilomelanurus*), while those present in the first survey but not appearing in the second were *M. latisulcatus* and *H. portusjacksoni*. Mature *Pagrus auratus* were found exclusively on the northern lease during the latter (August/September) sampling period. Juvenile *Pagrus auratus* were present for both surveys, however they were only detected on the southern sites in the latter survey, after having initially been recorded on both northern sites also. *Melicertus latisulcatus* were ubiquitous on the sites containing soft sediment (northern control and lease, southern lease) during the first sampling period, but were not recorded at all during the second survey.

From a qualitative perspective the RUV footage indicated no obvious temporal variation in the benthic environment. The northern lease and control sites contained the bare habitat type. The southern control site contained the mixed habitat, and the southern lease incorporated both habitat types.

A3.5 Discussion

A3.5.1 Effects of aquaculture

The presence of finfish aquaculture was found to have no effect on the composition of the demersal macrofaunal assemblages in Fitzgerald Bay on a local or regional scale, or over time. Despite several previous studies having detected aggregative effects of finfish aquaculture on wild fish assemblages (Dempster *et al.* 2002; Thetmeyer *et al.* 2003; Golani 2003; Dempster *et al.* 2004), this phenomenon did not extend to the demersal macrofauna of Fitzgerald Bay. The role of wild macrofaunal assemblages in waste mitigation on finfish aquaculture sites has been recognised in both Australia (Felsing *et al.* 2002) and the Mediterranean (Thetmeyer *et al.* 2003; Golani 2003). It is reasonable to assume, therefore, that this relationship could also have existed in Fitzgerald Bay, given the similar climate and aquaculture practises. There are two potential reasons why demersal macrofauna were not more prevalent on aquaculture sites in Fitzgerald Bay: (1) insufficient waste (ie.food source) is deposited on the seafloor under the cages to attract and retain resident demersal scavengers in the area, and (2) behavioural modification of scavengers in response to farm practises prevented the survey technique from sampling them adequately.

Settling velocities of waste material vary according to Stokes' Law (Yrong-Song *et al.* 1999) and these variable settling rates have been shown to produce concentric impact zones around sea-cages (Brown *et al.* 1987). Given the substantial tidal flows through Fitzgerald Bay (up to 39.1 cm/sec, Parsons Brinckerhoff and SARDI 2003) and the seafloor clearance (5 to 15 m) of the sea-cage nets, there is ample opportunity for waste dispersal to occur over a substantial area, especially for light-weight wastes (e.g. epiphyte biofouling, faeces). Conversely, pelleted feed sinks rapidly and is not carried far from the farm, with substantial local accretion under sea-cages detected previously in the Mediterranean (Thetmeyer *et al.* 2003). Pelagic fish have been documented to consume much of this excess feed before it reaches the seafloor (Thetmeyer *et al.* 2003), and fish in the sea-cages in Fitzgerald Bay are only fed once per day. The combination of these factors may prevent sufficient waste deposition beneath the sea-cages in Fitzgerald Bay to attract resident demersal scavengers. In support of this argument is the fact that during the course of this study excess pellets were never detected on the seafloor. Furthermore, during the bait effects study, pellets held in bait baskets were observed to disintegrate within the 30 minute duration of an RUV deployment. Any pellets, therefore, that did reach the seafloor would most likely disintegrate rapidly and either be consumed by the resident demersal fauna or dispersed by the tide within a very short time. Such limited food availability would provide little direct incentive for scavengers to remain in the area, over and above the existing natural assemblage.

If the scavengers most involved in waste mitigation in Fitzgerald Bay did not remain associated with the sea-cages for long periods, they may not have been sampled by the techniques used in this survey. By avoiding sea-cage feeding times during sampling, I may have been missing the opportunity to sample the scavenging fauna. Wild species have been observed to modify their behaviour in response to aquaculture practises. Sea birds follow feed boats from cage to cage (Harrison 2003) and wild fish follow inter-tidal oyster farmers during infrastructure defouling (pers. obs.). It is possible, therefore, that the scavengers in Fitzgerald Bay may have also modified their behaviour. Regardless of the cue (eg. boat engines, the noise of pellets hitting the surface of the water, the feeding activity of farmed fish), the scavengers may have moved from cage to cage during feeding and thus were not observed in the RUV deployments, which is a distinct possibility for highly mobile species (e.g. dolphins, *Pseudocaranx wrighti*). Dolphins and *Pseudocaranx wrighti* were regularly observed on the Fitzgerald Bay sites and could easily move between cages on an aquaculture site (100s of metres) or even between sites (several kilometres) in the course of a day. To examine this scenario, RUVs or divers could be used to observe the activity of scavengers during feeding events, identifying the species involved and their persistence times once feeding had ceased. Additional investigation of the scope of waste dispersal from the Fitzgerald Bay sea-cages is required in order to determine the rate at which it occurs and the area affected.

Thetmeyer et al (2003) detected negative impacts of finfish aquaculture on benthic and epibenthic wild fish assemblages, which included small, cryptic, territorial species. These taxa were found to be subject to predation and competitive exclusion as a result of certain wild fish species being attracted to sea-cages (Thetmeyer *et al.* 2003). No such effect, however, was detected in Fitzgerald Bay. In fact, the "benthic" species category, which included small cryptobenthic species, was the most common in this study, being observed in all three surveys that were undertaken and on every site that was sampled. "Benthic" species were also the only type recorded by the unbaited RUV deployments, indicating their abundance and possibly an attraction to the RUV structure.

The frequent observation of adult *Pagrus auratus* on the northern lease site, but at no other location sampled during the study, raises the possibility of their aggregation at sea-cages. Sparids have been found to aggregate on finfish aquaculture sites in the Mediterranean (Dempster *et al.* 2002; Thetmeyer *et al.* 2003; Golani 2003; Dempster *et al.* 2004) and contribute to the consumption of excess pellet feed (Thetmeyer *et al.* 2003; Golani 2003). Anecdotal reports of large recreational catches of *Pagrus auratus* around Fitzgerald Bay sea-cages lends further support to this argument. During this study recreational fishermen were observed fishing adjacent to sea-cages on the northern site. Spawning aggregations of adult *Pagrus auratus* assemble annually in northern Spencer Gulf between October and March (Fowler and Jennings 2003). This fishery has been closed during November of each year since 2000 to afford some protection to this spawning stock from fishermen, for whom large *Pagrus auratus* are a prized catch. In South Australia, fishermen are allowed access to aquaculture lease sites, which in the case of Fitzgerald Bay could put added pressure on local populations of aggregating species. The closure of aquaculture sites to fishing, conversely, may provide some refuge for recreationally and commercially targeted species that aggregate there. Further investigation of the distribution and residence times of mature *Pagrus auratus* on Fitzgerald Bay aquaculture sites is required to elucidate this potential aggregative effect and the potential risks associated with it.

A3.5.2 Habitat effects

Differences in benthic habitat type are thought to be the major driver behind differences observed in the demersal assemblages in upper Spencer Gulf during the present study. Despite the use of qualitative habitat classifications, spatial heterogeneity was evident within both Fitzgerald Bay and Douglas Point, with areas of unvegetated bare sand interspersed by areas of extensive benthic biota (sponges, macroalgae) at both locations. Thus habitat differences could explain the disparity detected in macrofaunal assemblages at these locations. There were obvious differences in the species richness among these habitats. Australian sand habitats have previously been found to have low macrofaunal species richness (Jenkins and Wheatley 1998; Travers and Potter 2002; Butler 2003), which was again detected during the present study. Most species recorded from these unvegetated sites were small, cryptobenthic and difficult to identify from dorsal footage. The "benthic" category was used to group these species and probably included several taxa. Trawl surveys from the Fitzgerald Bay region sampled several common cryptobenthic species (Fowler and Jennings 2004), which were likely to comprise most of those grouped in the "benthic" category used in the present study. These species included wavy grubfish (*Parapercis haackei*), spotted stinkfish (*Repomucenus calcaratus*) and sand flathead (*Platycephalus* *bassensis*). In the present study, all taxa found in bare areas were also found in the mixed habitat, except for adult *Pagrus auratus* and *Melicertus latisulcatus*. *Melicertus latisulcatus* bury themselves in the soft sediment of the bare areas to avoid predation (Tanner and Deakin 2001), a strategy that may have been less effective in the coarser substrate observed in the mixed habitat. In contrast, the sponge habitat deployments recorded a more diverse range of species, with *Heterodontus portusjacksoni*, *Portunus pelagicus* and two species of leatherjacket only being sampled from these areas. Likely explanations for this high fidelity to the mixed habitat centre on refuge and diet requirements. Predator avoidance in bare sand environments is commonly achieved via camouflage (ie. cryptobenthic species, (Ryer *et al.* 2004) and schooling (Laurel *et al.* 2004), both of which are likely be equally effective in the sponge habitat. Species that rely on habitat complexity to avoid predators, however, are more vulnerable in bare areas (Beukers and Jones 1998), and therefore avoid them or modify their behaviour to access these less favourable habitats (Travers and Potter 2002; Laurel *et al.* 2004). Species for which predation is a less immediate threat are probably distributed according to diet preferences (adult *Pagrus auratus, Heterodontus portusjacksoni*), and inhabit areas that contain appropriate food sources (Levinton 2001). The impending annual reproductive season for *Pagrus auratus* may also have contributed to their habitat selection, whereby they appeared to be aggregating around sea-cages located over soft sediment.

Seagrass (*Heterozostera tasmanica*) was recorded at Cowleds Landing in depths similar to those sampled at Fitzgerald Bay and Douglas Point, however no seagrass was ever observed in footage from the latter sites. Cowleds Landing was also the only area that did not display inter-site differences in either the demersal assemblages or qualitative comparisons of the benthic environment. Cowleds Landing had the highest overall species richness of the three sites, but lower sample richness than the sponge habitat at Fitzgerald Bay and Douglas Point. Decreased sample richness could indicate that bait is less effective at attracting species in seagrass habitat or that it has a lower species density than the sponge habitat. Decreased flow rates within seagrass canopies (Peterson *et al.* 2004) could result in a smaller bait plume and thus a smaller sample volume compared to bare sand habitat. Another potential explanation, partially related to bait plume dispersal, could be that the deployment time used in this study was insufficient for a seagrass environment. The RUV deployment time (30 minutes) for this study was based on the results of trials undertaken in Fitzgerald Bay, where species accumulation in the footage was generally observed to plateau at around three species after approximately 20 minutes. These trials occurred mostly on bare sand substrate, which is known to be species poor (Jenkins and Wheatley 1998; Travers and Potter 2002; Butler 2003). In seagrass habitat, which is known to have relatively high biodiversity (Jenkins and Wheatley 1998; Hindell *et al.* 2000; Travers and Potter 2002), species accumulation may not have levelled off after 20 minutes, resulting in the need for longer deployment times. One hour RUV deployment times have been used in southern Australian seagrass habitats (Butler 2003), as a result of trials detecting maximum numbers of species after 30-40 minutes. However, the need for consistency between replicates in this study resulted in the use of 30 minute deployments throughout all habitats.

The role of seagrass areas as nurseries (Bell and Pollard 1989), could also have contributed to the low sample richness. The presence of potential predators at the RUV bait basket may have deterred small or juvenile macrofauna from approaching the bait and thus entering the FOV. The small size of juvenile organisms and the obstructive nature of seagrass habitat could have prevented observation of some individuals in the FOV. Nonetheless, this effect would be expected to also extend to the sponge habitat.

A3.5.3 Temporal stability

Dempster (2002, 2004) found that wild fish aggregations associated with sea-cages in the Mediterranean were temporally stable over periods ranging from several weeks to months. The macrofaunal assemblages in Fitzgerald Bay, however, were found to vary over the course of the present study (nine weeks). This difference could be due to the fact that this study was essentially sampling natural communities, whereas the aggregations examined by Dempster (2002, 2004) were not present in that location prior to the establishment of aquaculture. The differences detected in the present study, therefore, were possibly due to natural seasonality: species responding to the transition from early (June) to late (August/September) winter. The wild fish sampled by Dempster (2002, 2004), however, were attracted to the sea-cages, which are a temporally stable factor. The seasonal (September to October) variation encountered during Dempster's (2002, 2004) study was either insufficient to elicit a response, produced an undetected response or was ignored by the assemblage.

While some species were detected throughout the present study (*Portunus pelagicus, Pseudocaranx wrighti,* juvenile *Pagrus auratus,* "Benthic" category), there were several interesting temporal trends for other species. Mature *Pagrus auratus*, *M. latisulcatus*, *H. portusjacksoni* and *A. spilomelanurus* were recorded exclusively during one sampling period. Very low individual counts and sporadic sightings of the latter two species prevent temporal inferences from being made from the existing data. *Melicertus latisulcatus*, however, was common during the first survey (June) and absent from the third survey (August/September). Activity in this species is directly related to water temperature, with minimum activity occurring during the cooler winter months (King 1977). During August/September, water temperatures in Fitzgerald Bay are usually around 12-13^{o}C (Schilg and Hyde 2003). The lower limit of activity for penaeid prawns is 10-12^{o}C; therefore, most were likely to have been buried in the sediment during the third survey (King 1977). The species is also migratory with individuals moving in a southerly and easterly direction as they mature (Carrick 1982) and thus likely to leave Fitzgerald Bay during the year.

Adult *Pagrus auratus* were recorded only on the northern site during the second survey, which corresponds with the lead-up to their annual reproductive season in upper Spencer Gulf from October to March (Fowler and Jennings 2003). An interesting coincidence is that juvenile *Pagrus auratus* were recorded on the soft substrate of northern Fitzgerald Bay during the first survey, in the absence of adult conspecifics, but were only found in the southern sponge habitat in the latter survey. Differential juvenile mortality between habitats could explain this observation. An alternative explanation is behavioural change that occurs during the first year of life (pers. comm. – Dr. Anthony Fowler, SARDI Aquatic Sciences) and results in a change in habitat selection by juvenile *Pagrus auratus* from the bare, flat, muddy substratum on which they settle (Fowler and Jennings 2003) to more complex habitats (Thrush *et al.* 2002).

A3.5.4 Project limitations

The decision to use a baited, vertically mounted RUV in this study was based on an initial review of the potential sampling methods available and the results of a pilot study comparing several viable techniques that were identified (Appendix A3.A). Many positive factors contributed to the final selection of the equipment (footage quality, fixed FOV, ability to survey the species present), however several weaknesses emerged during the fieldwork and subsequent footage analysis. Schooling species presented a problem, as often only part of a large school was visible in the footage and could therefore be included in the results (Willis *et al.* 2000), even though it was likely that many more individuals were present. The decision to include relatively immobile demersal macrofauna (eg. holothurian, echinoid) in the count data was based on the fact that they are technically "mobile macrofauna". When these species were observeded during the study they were already present in the FOV when the RUV settled on the seafloor. While they were obviously not active scavengers responding to the bait, RUV captures every species that passes within the FOV regardless of whether the bait was the reason for their presence. The two taxa mentioned above were a minor component of the assemblages sampled from one area (Cowleds Landing) and did not influence the results greatly.

A further limitation of the present study was that accurate identification of small benthic and demersal finfish species from the dorsal view afforded by the RUV was impossible in many cases and this led to the grouping of some species into a single "benthic" category during the footage analysis. Non-identification of these species decreased the resolution of the surveys and thus made it more difficult to detect differences between areas. The baited RUVs used in this study, nevertheless, proved to be effective at sampling cryptobenthic species, which comprised a substantial proportion of the macrofauna sampled in Fitzgerald Bay. Small cryptic species are notoriously difficult to survey (Brock 1982; Lincoln Smith 1989; Thetmeyer *et al.* 2003), and destructive methods such as trawls (Harmelin-Vivien and Francour 1992; Letourneur *et al.* 2001) and piscicides (St.John *et al.* 1990; Willis and Anderson 2003) have been used extensively in the past to sample this faunal component. Baited techniques have been endorsed for surveying cryptic species (Stewart and Beukers 2000) and the present study lends support to this finding, although occasional individuals were recorded on unbaited RUV. Bait has been shown to make remote macrofaunal surveys faster and more comprehensive (Cappo *et al.* 2002), however its use has associated consequences. Selective attraction of species by different bait types can affect the composition of samples obtained using baited techniques (pers. comm. – Dr Trevor Willis, Universita di Bologna). Bait plume effects can also cause inconsistencies between replicates of baited techniques. The variation in current speed between deployments may produce differences in the bait plume size and thus sample area (Sainte-Marie and Hargrave 1987; Willis *et al.* 2000). Only three species groups were sampled during the bait comparison undertaken in the present study, and thus the potential to detect selective attraction was reduced. A bait comparison study would be more likely to demonstrate selective species attraction if it was undertaken in an area with greater species richness.

Portunus pelagicus was a common visitor to the RUVs during this study. The highly aggressive nature of this species was evident during footage analysis, where it was observed to attack conspecifics that approached the bait. This behaviour, in conjunction with the relatively small FOV of the RUV equipment, probably acted to decrease the MaxN values for both *P. pelagicus* and other species, by excluding them from entering the FOV. The count method of including *P. pelagicus* individuals that were easily distinguished from others in the MaxN value reduced the problem for that particular species, however their effect on other taxa could not be avoided.

The loss of replicates during the August/September surveys due to the strong tides in Fitzgerald Bay reduced the statistical power of the temporal analysis and the scope of the bait comparison. RUV toppling could be avoided by undertaking deployments during weaker tides (e.g. dodge, neap) or incorporating more weight into the base of the frame. However, deployments were retrieved by hand in this study, and thus there was a trade-off between RUV weight and ease of retrieval. Low species richness and MaxN count values were consistent observations throughout the study, which in conjunction with a conservative measure of relative abundance (ie. MaxN) also restricted the likelihood of detecting statistically significant assemblage composition differences. Nonetheless, differences in demersal assemblages were still discovered during the project, indicating sufficient statistical power was present.

A3.6 Conclusions and future research

Finfish aquaculture in Fitzgerald Bay does not appear to have affected the resident demersal assemblages, indicating that the benthic environment within the Bay is not being significantly affected by waste from the sea-cages. There is undoubtedly additional organic input to the Bay from finfish aquaculture and further research is required to quantify this contribution and determine its fate. As demersal macrofauna do not appear to be playing a significant role in the consumption of any additional organic input, the pelagic faunal component becomes a likely suspect. The role that pelagic scavengers play in waste mitigation in Fitzgerald Bay was not included in the present study and needs to be investigated (although the pilot study indicated low numbers of pelagic macrofauna - see Appendix A3.A). The possible aggregation of mature *Pagrus auratus* at sea-cages in the Bay, and the possible implications of fishing access to these areas, needs to be examined. Nevertheless, based on our current understanding of local benthic effects, Fitzgerald Bay appears to be a suitable site for finfish aquaculture.

A3.7 Acknowledgments

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A3.9 Appendix A3.A: Equipment selection and trial

Two forms of non-destructive remote survey technique were identified as potentially suitable for use around sea-cages: remotely operated vehicle (ROV) and remote underwater video (RUV). A pilot study was undertaken to determine which of these methods was most effective in Fitzgerald Bay. The use of a VideoRay ProII ROV was intended to survey both the benthic and pelagic macrofauna associated with sea-cages. Due to a combination of strong currents, low underwater visibility and the risk of entanglement with sea-cage infrastructure, this method proved unfeasible. Subsequently, several types of baited RUV were designed, built and tested, using brined pilchards as bait. Although the use of bait raises some concerns regarding plume size effects and selective species attraction (Butler 2003), it also allows a faster, more comprehensive survey of macrofaunal assemblages (Cappo *et al.* 2002). The use of bait is particularly important in habitats dominated by sand, such as Fitzgerald Bay, where the abundance and diversity of resident assemblages is low (Butler 2003). A laterally mounted RUV system (Figure A3.13a) was deployed on the seafloor, however sediment resuspension was high close to the substrate, thus reducing visibility and making reliable identification and enumeration of fauna impossible. A lateral RUV with an adjustable stand (Figure A3.13b) was then tested, however the benthic nature of many of the resident macrofaunal species prevented them from being recorded during sampling.

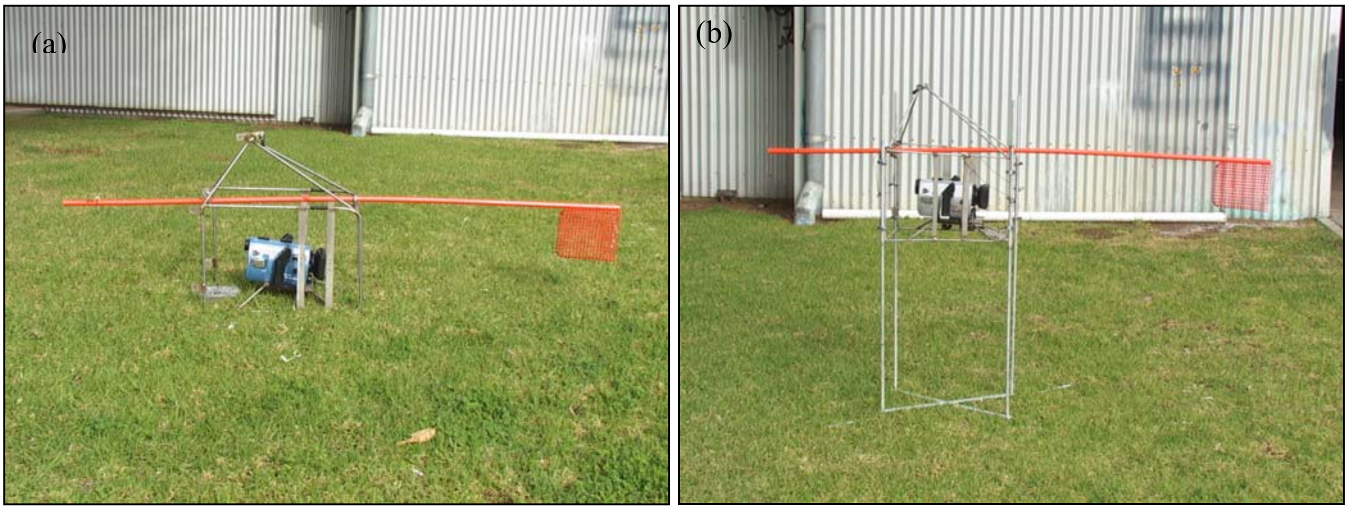

Figure A3.13. Views of laterally mounted RUV designs with the camera (a) close to the seafloor and (b) raised off-bottom, that were tested during the pilot study.

In response to this, a vertically mounted RUV (Figures A3.14a and 14b) was developed to ensure that benthic species were detectable and to avoid the worst of the visibility. This design incorporated a bait arm, which raised the bait basket off-bottom to prevent sediment being resuspended by species attempting to feed. The base of each frame was also marked in 50mm increments to allow rudimentary length estimates of individuals in the FOV. Plastic coated 12 mm wire mesh bait baskets manufactured for the crayfish industry, were used exclusively throughout this study. Wire mesh baskets prevent larger species from consuming the bait, and ensure a relatively consistent bait mass throughout the deployment. These baskets also reduced the time spent at the bait (known as "persistence time") for most species, by ensuring that attempts to feed go largely unrewarded and resulting in a continual turnover of individuals at the bait. Another advantage of the vertical RUV is a constant field of view (FOV), resulting from a fixed depth of field. In contrast, for laterally mounted RUVs the depth of field varies with the water clarity, which in Fitzgerald Bay changes with the tidal phase, thus altering the sampling volume between deployments.

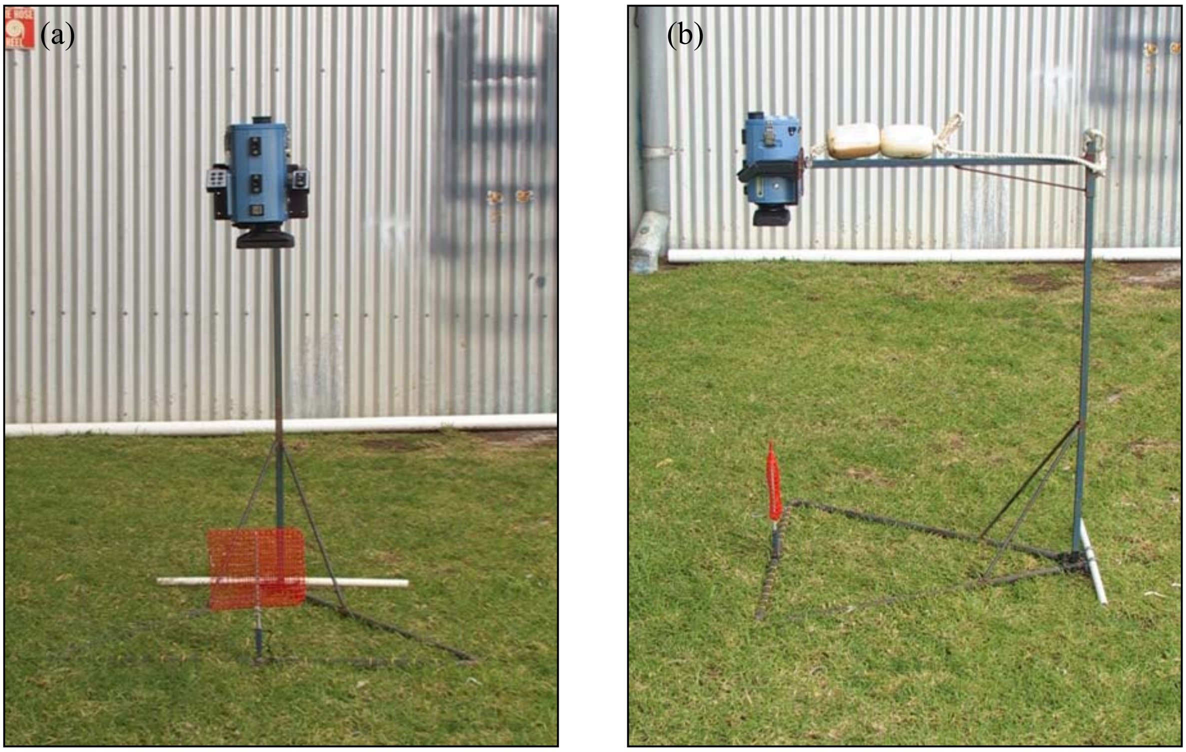

Figure A3.14. Front (a) and side (b) views of the vertically mounted RUV used in this study.

Following the failure of the ROV, both RUV designs (lateral and vertical) were deployed in mid-water to assess the pelagic component of the macrofauna, and to compare their efficacies. A mid-water mooring system (Figure A3.15a) was developed, which was suitable for both RUVs, consisting of an anchor, mid-water float and retrieval line (Figure A3.15b). Noise associated with the equipment was minimised by insulating all metal components that were in direct contact using vulcanising rubber adhesive tape. The mid-water mooring system could be adjusted to ensure that the RUVs were suspended approximately adjacent to the bottom of the sea-cage net, an area where pelagic species are known to aggregate (Dempster *et al.* 2002). However, the mid-water RUV deployments in Fitzgerald Bay failed to consistently record any pelagic species and so were not utilised in this investigation.

The results of the pilot study indicated that the benthic component of the macrofaunal assemblage in Fitzgerald Bay was the most consistently detectable using the techniques that had been assessed. The vertically mounted RUV was the most effective method of sampling this habitat; therefore an additional frame was built for use in the main study.

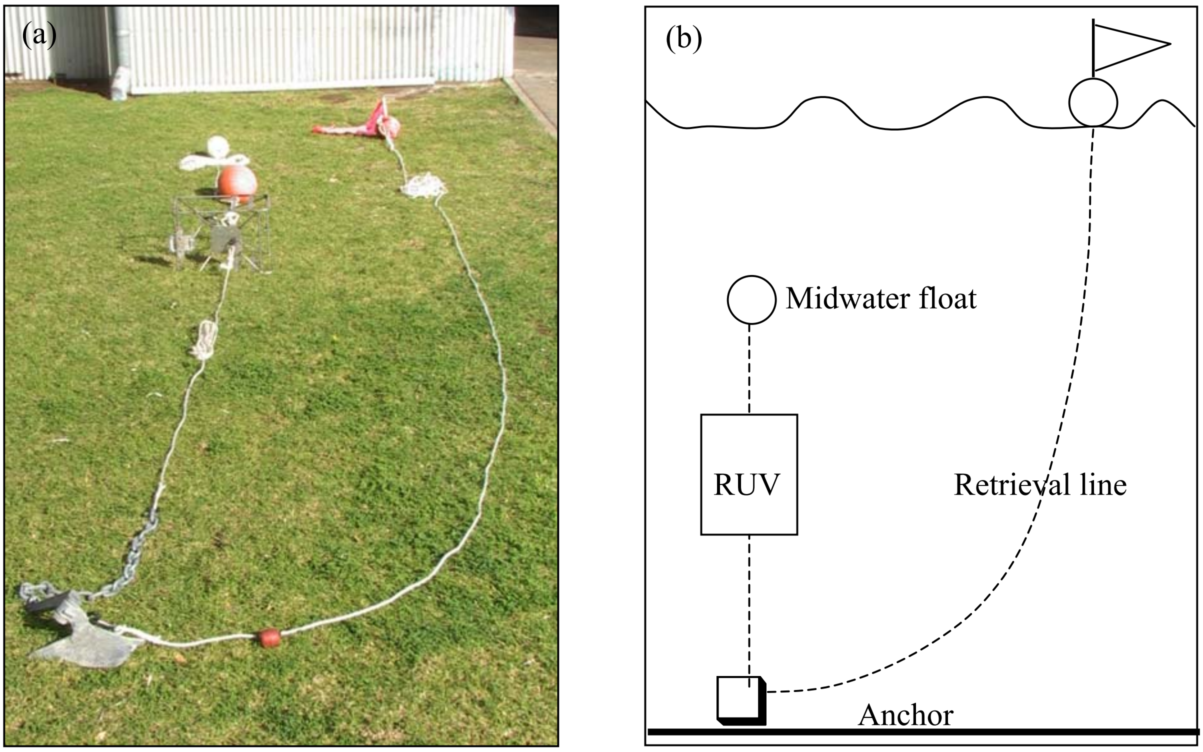

Figure A3.15. (a) Photograph and (b) schematic diagram of the mid-water deployment system tested during the pilot study.

A3.10 Appendix A3.B: Taxa recorded during surveys

Table A3.1: All species observed during the "local effects" survey of Fitzgerald Bay and the sites on which they occurred.

Common name	Scientific name	Presence on site			
		Northern control	**Northern lease**	**Southern lease**	**Southern control**
Blue swimmer crab	*Portunus pelagicus*			X	X
Western king prawn	*Melicertus latisulcatus*	X	X	X	
Skipjack trevally	*Pseudocaranx wrighti*	X	X	X	
Snapper (juv.)	*Pagrus auratus*	X	X		X
Port Jackson shark	*Heterodontus portusjacksoni*			X	X
Benthic		X	X	X	X

Table A3.2. All species observed during the "regional" survey and the sites on which they were recorded (Douglas Point = DP, Fitzgerald Bay = FB, Cowleds Landing = CL).

Common name	**Scientific name**	**Presence on site**					
		DP1	**DP2**	**FB1**	**FB2**	**CL1**	**CL2**
Blue swimmer crab	*Portunus pelagicus*	X			X		X
Red swimmer crab	*Nectocarcinus integrifrons*					X	
Skipjack trevally	*Pseudocaranx wrighti*			X	X		X
Snapper (juv.)	*Pagrus auratus*	X			X		
Benthic		X	X	X	X	X	X
Bridled leatherjacket	*Acanthaluteres spilomelanurus*	X					
Pygmy leatherjacket	*Brachaluteres jacksonianus*	X					
Sand flathead	*Platycephalus bassensis*		X				X
Smalltooth flounder	*Pseudorhombus jenynsii*						X
Unidentified finfish 1			X				
Unidentified finfish 2							X
Unidentified finfish 3						X	
Port Jackson shark	*Heterodontus portusjacksoni*				X	X	X
Southern calamary	*Sepioteuthis australis*	X					
Holothurian	*Stichopus mollis*					X	
Echinoid	*Allostichaster polyplax*					X	

Table A3.3. All species observed during the "temporal" surveys (T1, T2) of Fitzgerald Bay and the sites on which they were recorded.

Common name	Scientific name	Presence on site							
		Northern control		Northern lease		Southern lease		Southern control	
		T1	T2	T1	T2	T1	T2	T1	T2
Blue swimmer crab	*Portunus pelagicus*					X	X	X	X
Western king prawn	*Melicertus latisulcatus*	X		X		X			
Skipjack trevally	*Pseudocaranx wrighti*	X	X	X	X	X	X		
Snapper (adult)	*Pagrus auratus*				X				
Snapper (juvenile.)	*Pagrus auratus*	X		X			X	X	X
Bridled leatherjacket	*Acanthaluteres spilomelanurus*						X		
Benthic		X	X	X	X	X	X	X	X
Port Jackson shark	*Heterodontus portusjacksoni*					X		X	