Critical water needs to sustain freshwater ecosystems and aquatic biodiversity in Queensland's Mitchell River

Final report

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Acronyms and abbreviations

ALA ....alpha-linoleic acid
AM ....aquatic macroinvertebrates
AP.................aquatic plants
ARA..............arachidonic acid
BAFA............bacterial fatty acid
CDOM...........coloured dissolved organic matter
CSIRO ....Commonwealth Scientific and Industrial Research Organisation
DAF ....Department of Agriculture and Fisheries
DHA..............docosahexaenoic acid
DKN..............dissolved Kjeldhal nitrogen
DKP ....dissolved Kjeldhal phosphorus
EOS ....end-of-system
EPA ....eicosapentaenoic acid
F....................Fish
GL.................gigalitre (1,000,000,000 L)
JBG ....juvenile barramundi growth
LA ....linoleic acid
MJO..............Madden–Julian Oscillation
ML.................megalitre (1,000,000 L)
NAWRA........Northern Australia Water Resource Assessment
ppt ....parts per thousand
PUFA............polyunsaturated fatty acid
S ....sulfur
SOI................Southern Oscillation Index
Sr ....strontium
t C.................tonnes of carbon
TKN ....total Kjeldhal nitrogen
TKP...............total Kjeldhal phosphorus
TM.................terrestrial macroinvertebrates
TN.................total nitrogen
TP ....terrestrial plants
TPh...............total phosphorus
TSS...............total suspended solids
WRD ....water-resource development

Acknowledgements

Executive summary

Critical water needs to sustain freshwater ecosystems and aquatic biodiversity in Queensland's Mitchell River

The Australian Government has identified water-resource development in the north of Australia as a priority for the coming years and Queensland's Mitchell River is a focus of this interest. The Queensland Government currently regulates small amounts of extraction from the Mitchell River meaning that the flow regime is largely unregulated. In its current condition, the catchment supports existing water users, including Traditional Owners, and significant ecological assets, including wetlands of national significance, threatened species, traditional harvests, and commercial and recreational fisheries. Information is needed on the water needs of freshwater ecosystems and their associated biodiversity to guide decision-making around water planning and development.

This project examined the links between freshwater flow and floodplain inundation, aquatic plant biomass accumulation, floodplain subsidies to freshwater food webs, fish movement and fisheries production. This project had five major objectives to understand the flow requirements of freshwater assets:

1. Identify the dependence of floodplain inundation and aquatic plant biomass accumulation on river flow
2. Map key 'hotspots' of freshwater primary productivity within the floodplain
3. Quantify riverine and floodplain connectivity to assess the likely impact of varying wet seasons and water-resource development on aquatic food webs, and the movement of fish between river channels and floodplain wetlands
4. Quantify the importance of river flows to estuaries by examining the flow dependence of barramundi growth, population dynamics and habitat use
5. Assess the implications of flow alteration on freshwater ecological communities, including movement of fish throughout the catchment.

These objectives were achieved through a range of desktop and field-based studies. Objectives 1 and 2 relied on remotely sensed data and modelling, with field-based surveys used for ground truthing. The methodology for Objective 3 involved field-based data collection combined with tissue analyses and modelling. Objective 4 relied on analyses of otoliths from fish caught in the barramundi fishery. Objective 5 relied on field-based surveys and tracer analyses of fish and invertebrates caught in the surveys.

Key findings

Objectives 1 and 2

- Analyses of remotely sensed satellite data and gauged river-flow data showed that the inundation of the floodplain and accumulation of aquatic plant biomass in floodplain wetlands were dependent on flows from upstream as well as local rainfall. Inter-annual variation in river flows explained approximately 60% of the variation in floodplain inundation.
- Field surveys of algal productivity combined with remotely sensed satellite data showed that over 90% of algal primary productivity on the floodplain occurred within floodplain wetlands, as opposed to river channels flowing through the floodplain, due to the vast

surface area of the wetlands themselves. Aquatic macrophytes growing in floodplain wetlands provide an important substrate for algal growth that fuels the base of aquatic food webs.

Objective 3

- Floodplain connectivity, facilitated by inundation from river flows, ensured up to 58% of algal productivity in floodplain wetlands was connected to the main Mitchell River channels over the course of inundation events, providing a potential energy subsidy to the riverine food webs. The duration and extent of inundation were primary drivers of the connectivity of wetlands and river channels, and this connectivity would be reduced by dams and drier wet seasons.
- Evaluation of water-resource development scenarios showed that potential floodplain primary productivity input to riverine aquatic food webs would be reduced by up to 26% due to reduced access of aquatic biota to floodplain wetlands.
- Key consumers of the algae produced in the Mitchell River system, including the floodplain wetlands, were invertebrates such as dragonfly and mayfly larvae, which contain high levels of essential fatty acids and are a high-quality food for fish. Fish used wet-season connections to the floodplain to access this high-quality food.

Objective 4

- Analysis of catch and age data identified that wetter years with larger flows support increased growth, abundance and biomass of coastal barramundi. Additional metrics highlighted the importance of the sequential pattern of river flow over multiple years as a key driver of barramundi population dynamics. Reduced flows as a result of water-resources development may lead to smaller fish and increased natural mortality.
- Barramundi that access freshwater habitats in the wet season had faster growth than those that stayed in the coastal zone. Changes in the inter-annual river-flow patterns and connectivity caused by water-resources development may impact the opportunities for barramundi to migrate and access freshwater habitat.

Objective 5

- Fish movement varied between species and across the catchment; however, most fish made at least one large-scale movement in their lifetime, facilitated by wet-season flows that connect dry-season habitats. The widespread movement across all species indicated that such behaviour is most likely associated with foraging and seeking refuge in habitats that persist over the dry season. The mainstem of the Mitchell River itself acted as an important source habitat of individuals that move around the rest of the catchment.
- Analysis of fish movement and flow-mediated connectivity showed that the extent of fish movement to a given location is dependent on the connectivity of that location to the rest of the river network. Locations that have higher levels of connectivity had a higher proportion of fish moving to them, supporting biodiversity and ecosystem function. Predictive modelling showed that reductions in connectivity through the development of dams, without mediating environmental flows, will reduce fish movement across the catchment.
- The ability of fish to access the Mitchell River itself from the other sub-catchments would nearly halve (45%) if all five dams used in our modelling are constructed and operated without mitigation measures.

The Mitchell River's freshwater biodiversity – and the food and habitats that sustain it – are dependent on dry-season refugia, and wet-season flows and connectivity. Wet-season flows are important for all habitats, including rivers, floodplains and coasts. They inundate floodplains, and the nutrients they deliver fuel the growth of aquatic plants and algae, the quantity and quality of which subsidise and support the entire Mitchell River aquatic food web. Wet-season flows also connect highly productive floodplains with their rivers. Flows and freshwater habitat availability support the abundance, age-structure, biomass and growth rate of the Mitchell River barramundi population in the coastal fishery in the Gulf of Carpentaria. Fish move more easily when rivers and floodplains are connected by flows. Given the importance of flows for the Mitchell's freshwater biodiversity, these findings have direct relevance to environmental risk assessments for future water planning in the Mitchell catchment.

There are several specific implications of these findings for water planning and river management:

- The biodiversity and ecosystem function of the Mitchell River catchment is dependent on flow-mediated connectivity. Floodplain primary productivity, which supports aquatic food webs as well as birds and other animals, relies on overbank flows that result from upstream catchment river flow and local rainfall.
- The extent of movement of fish around the catchment, particularly from the Mitchell River mainstem to the other major sub-catchments, indicates that planning of water-resource development needs to consider the entire catchment. The Palmer, Walsh and Mitchell rivers are not separate ecological systems but rather interconnected by flow and fish movement.
- Floodplain inundation is a crucial process with considerable evidence showing that aquatic food webs rely on access to inundated areas including wetlands or floodplain creeks and rivers. While local rainfall is important, catchment flows from upstream are vital for inundation. Any reductions in wet-season flows are likely to impact floodplain inundation, particularly in years with limited cyclone activity in the Gulf of Carpentaria.

These findings and implications are summarised in the three conceptual diagrams below (Figures ES-1, ES-2 and ES-3), depicting the ecological processes in the dry season and wet season, and the predicted impacts of changes to the catchment.

Wet season
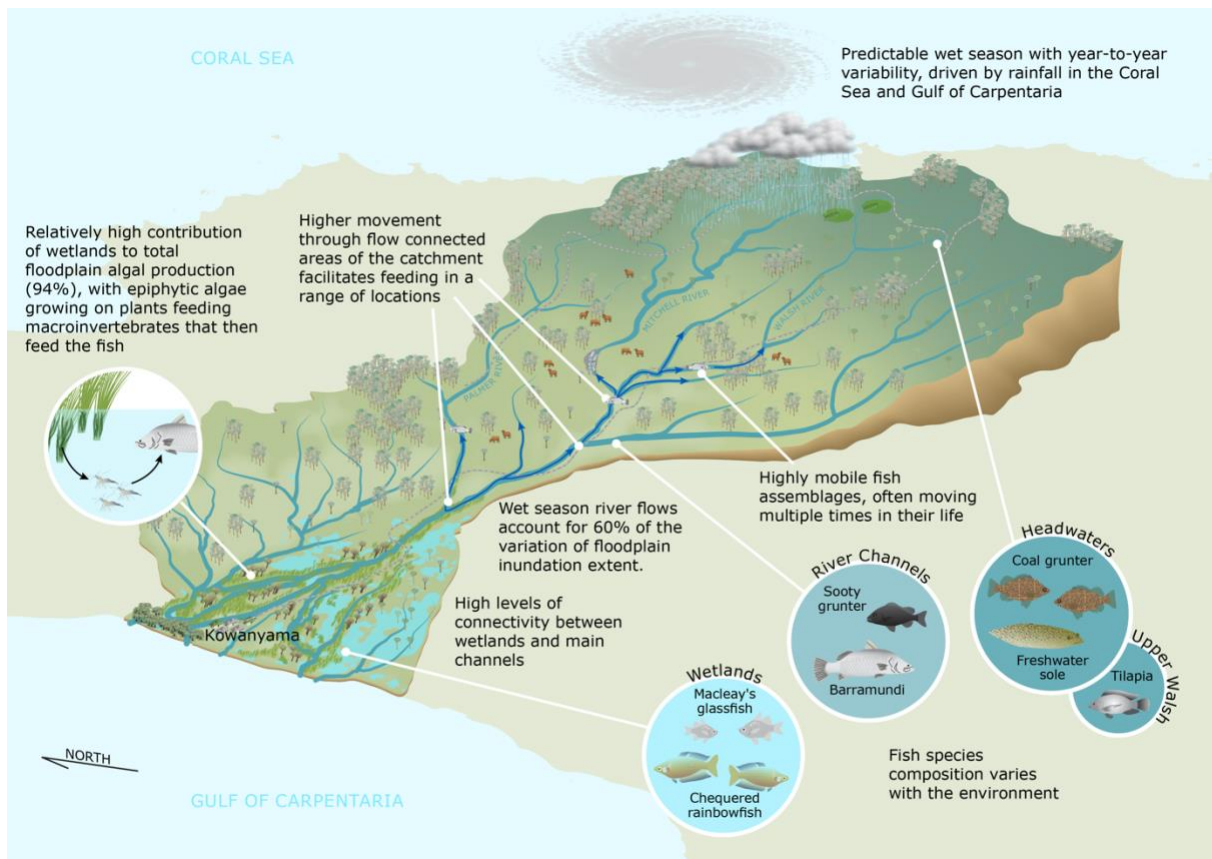
Figure ES-1. Conceptual model of Mitchell River ecosystem function in the wet season.

- All rivers flow in the wet season and floodplain wetlands are periodically inundated, connecting these wetlands with river channels. Connectivity and its seasonal changes determine the habitat and food available for fish, and these vary across the catchment. High turnover of species occurs in the headwaters and between adjacent sites of low connectivity, indicating that species composition is influenced by opportunities for movement between sites.
- Floodplain inundation and aquatic plant biomass accumulation are dependent on upstream flows and rainfall. At the end of the 2017–2018 wet season, most of the algal primary productivity on the floodplains occurred in wetlands. Floodplain wetlands are a major source of high-quality food for fish. Macroinvertebrates feed on epiphytic algae, concentrate lipids such as polyunsaturated fatty acids, and are then eaten by fish. Fish use wet-season connections to the floodplain to access this food source. Inundation periods can be short, and many of the fish that move into floodplain wetlands do not have the opportunity to move back to the river until the following wet season. Sufficient wet-season flows are required to ensure floodplain wetland habitats persist through the dry season to support this biodiversity.
- Fish movement varies between species and across the catchment, and most fish make at least one large-scale movement in their lifetime, showing that even non-reproductive individuals access different parts of the catchment. Most fish caught in the Walsh and upper Palmer rivers had moved there from other areas of the catchment. Short-term fish

movement is dependent on river-reach connectivity, with higher movement occurring through more connected areas.

- Barramundi that access freshwater habitats in the wet season have faster growth than those that stay in the coastal zone. Wetter years with larger flows support increased growth, abundance and biomass of coastal barramundi.

Dry season
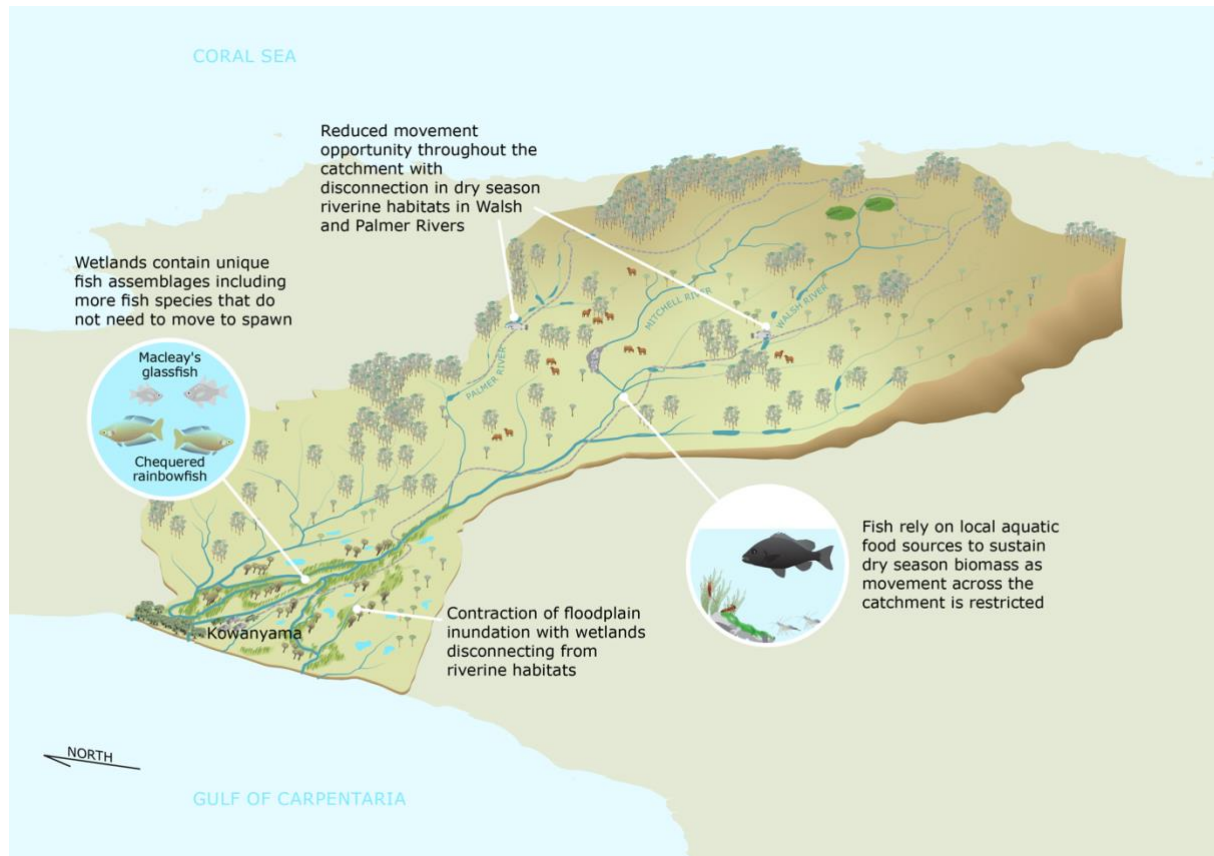
Figure ES-2. Conceptual model of Mitchell River ecosystem function in the dry season.

- The mainstem of the Mitchell River flows perennially, while the Walsh (below the irrigation scheme), Lynd and upper Palmer rivers cease to flow in the dry season and form a series of disconnected pools. Floodplain inundation recedes after wet-season floods and water only remains in depressions in the landscape (some wetlands and creeks) until the next wet season. These variations in flows throughout the catchment create habitats with different degrees of connectivity to the rest of the river network. Connectivity changes across the catchment and with longitudinal landscape changes (e.g. elevation, geology).
- The Mitchell Falls on the Mitchell River are a natural barrier to upstream fish migration. Below this barrier, fish in the main channel of the Mitchell can move freely within this reach during the dry season, while all other fish in the catchment find refuge in wetlands or pools. Off-channel wetlands contain functionally unique fish assemblages, with fish that do not need to move to spawn in higher proportions than at sites in the main channel, making these wetlands important habitats for the biodiversity of the catchment.
- As the dry season progresses and habitats contract, fish become more reliant on local food sources to sustain dry-season biomass.

Possible threats to Mitchell River ecosystem functioning from water-resource development
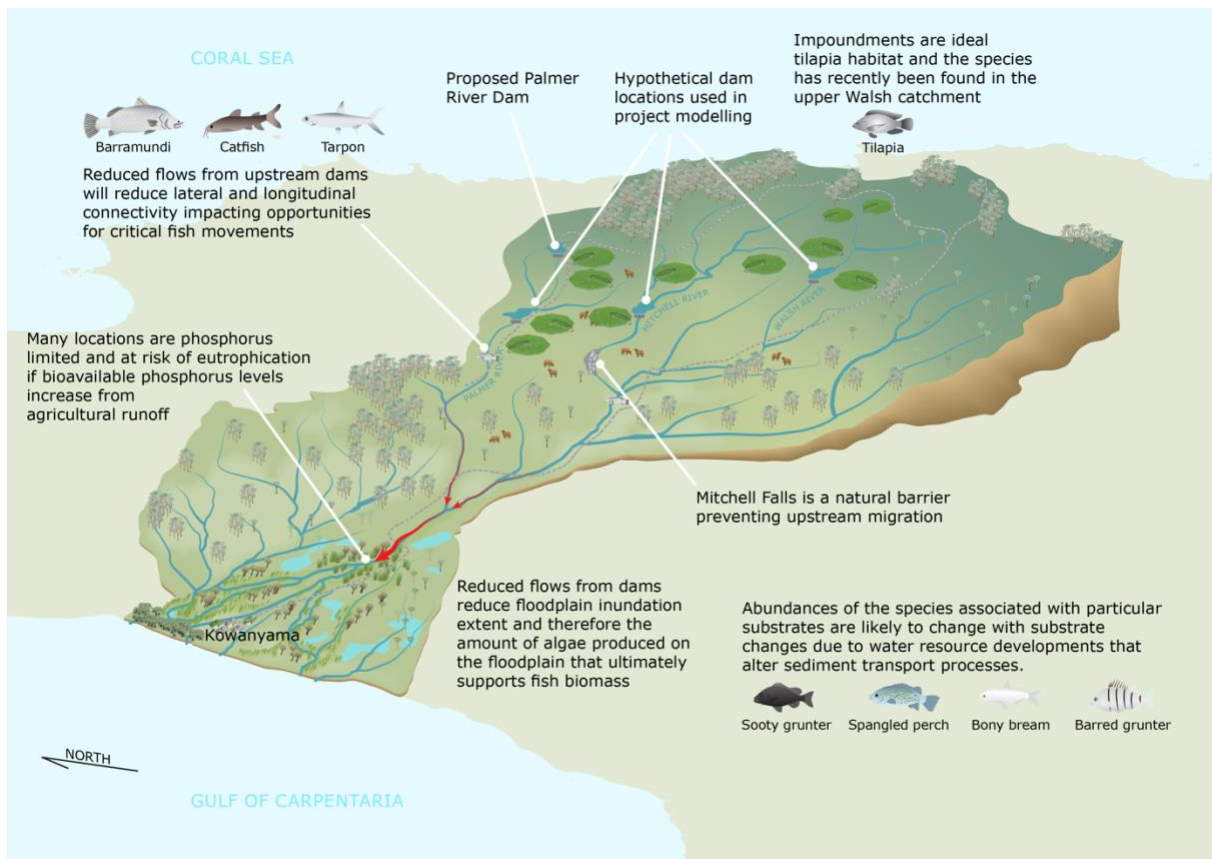
Figure ES-3. Conceptual model of possible threats to Mitchell River ecosystem function from water-resource development.

- Dams impact lateral and longitudinal connectivity by reducing river flow and acting as a barrier. This reduces the potential for fish movement and access to primary productivity in floodplain wetlands as fish can no longer access the main river channel.
- Reduced flows under water-resource development scenarios are predicted to affect the growth of barramundi in the coastal fishery, with younger fish likely to be most impacted.
- Connectivity changes characteristics of the environment which, in turn, influences fish species composition and functional diversity. Reduced connectivity will alter fish assemblages, thereby impacting the biodiversity of the catchment.
- Impoundments created from dams and weirs are ideal habitat for invasive tilapia.
- Many locations in the catchment, including floodplains, are phosphorus-limited. Eutrophication, harmful algal blooms, and ultimately oxygen depletion and fish kills would be a risk if agricultural run-off resulted in increased phosphorus levels.

Publications

The following publications arose from the data collected during the project.

Leahy, S.M., and Robins, J.B. (2021). River flows affect the growth of a tropical finfish in the wet-dry rivers of northern Australia, with implications for water resource development. Hydrobiologia, 848:4311–33. https://doi.org/10.1007/s10750-021-04641-7

Molinari, B., Stewart-Koster, B., Adame, M.F., Campbell, M.D., McGregor, G.B., Schulz, C.P., Malthus, T.J., and Bunn, S.E. (2021). Relationships between algal primary productivity and environmental variables in tropical floodplain wetlands. Inland Waters, 11(2):180–90. https://doi.org/10.1080/20442041.2020.1843932.

Molinari, B., Stewart-Koster, B., Malthus, T.J., and Bunn, S.E. (2021). Assessing spatial variation in algal productivity in a tropical river floodplain using satellite remote sensing. Remote Sensing, 13(9):1710. https://doi.org/10.3390/rs13091710.

Molinari, B., Stewart-Koster, B., Malthus, T.J., and Bunn, S.E. (submitted to Journal of Applied Ecology). Impact of water resources development on connectivity and primary productivity across a tropical river floodplain.

Ndehedehe, C.E., Onojeghuo, A.O., Stewart-Koster, B., Bunn, S.E., and Ferreira, V.G. Upstream flows drive the productivity of floodplain ecosystems in tropical Queensland. Ecological Indicators, 125:107546. https://doi.org/10.1016/j.ecolind.2021.107546.

O'Mara, K., Venarsky, M., Stewart-Koster, B., McGregor, G.B., Schulz, C.P., Kainz, M., Marshall, J., and Bunn, S.E. (2021). Connectivity of fish communities in a tropical floodplain river system and predicted impacts of potential new dams. Science of the Total Environment, 788:147785. https://doi.org/10.1016/j.scitotenv.2021.147785.

O'Mara, K., Stewart-Koster, B., Venarsky, M., McGregor, G.B., Schulz, C.P., Kainz, M., Marshall, J., and Bunn, S.E. (submitted to Freshwater Biology). Dietary energy flow through food webs and across habitats in a tropical river system.

O'Mara, K., Stewart-Koster, B., Venarsky, M., McGregor, G.B., Schulz, C.P., Marshall, J., and Bunn, S.E. (in progress). Indirect effects of river connectivity on fish species composition, turnover and functional diversity across a tropical floodplain river catchment.

1. Introduction

1.1 River-flow ecology

Natural flow regimes connect river habitats to create diverse ecosystems across riverine landscapes, through which water, sediment and organisms move (Poff *et al.*, 1997). River flows provide longitudinal connectivity between river reaches within river channels (in-stream), and floods create lateral connectivity to floodplain habitats such as wetlands (Ward & Stanford, 1995). Both longitudinal and lateral connectivity are important for the functioning of the ecosystem as they provide opportunity for mobile organisms to move, as well as transporting water and nutrients between habitats (Reid, Delong & Thoms, 2012; Vivancos *et al.*, 2021). For many tropical regions across the globe with wet and dry seasons, flows change predictably throughout the year according to rainfall (Poff *et al.*, 1997). This seasonality provided by the natural flow regime is a key factor influencing aquatic food webs and biological processes such as reproduction and recruitment of fishes (Agostinho *et al.*, 2004; Douglas, Bunn & Davies, 2005).

1.1.1 River flows and in-stream ecosystems

High river flows are important for maintaining connectivity and water quality throughout in-stream river reaches, allowing the exchange of materials and energy between different areas of the catchment (Jardine *et al.*, 2015). Naturally occurring low flows are equally important in maintaining healthy ecosystems by providing optimal foraging opportunities for different sets of species. Cessation of flow in intermittent rivers results in river fragmentation as previously flowing reaches become a series of disconnected pools or dry completely (Ward & Stanford, 1995; Rolls, Leigh & Sheldon, 2012). Inter-annual variation in flows can affect habitat quality, alter food webs and impact the reproductive success of some species (Agostinho *et al.*, 2004). Habitat changes (including aquatic vegetation changes) can also occur as water levels recede, reducing the extent of physical habitat and access to riverbank habitats such as marginal and overhanging vegetation and undercut banks (Rolls *et al.*, 2012). The seasonal expansion and contraction of aquatic habitat changes the composition and abundance of biological communities present, resulting in different food webs in low-flow conditions compared to high-flow conditions (Leigh & Sheldon, 2009; McIntosh *et al.*, 2017). As drying continues, fish become increasingly dependent on food produced within remnant pools (Venarsky *et al.*, 2020), and the number of top predators in disconnected pools influences the nature of the food web and its biomass (Pereira *et al.*, 2017). Finally, fish genetic diversity and reproductive success is also strongly influenced by the seasonal occurrence of low flows and river fragmentation that limits life history migrations (Gido *et al.*, 2016).

1.1.2 Floodplain wetlands ecosystems

Floodplain wetlands are a key component of many riverine ecosystems. They provide habitat for many species and support important ecosystem services (Ward & Stanford, 1995; Cowardin *et al.*, 2005; Keddy *et al.*, 2009). During the wet season, high river flows can create flooded areas of land adjacent to a river, which stretch from the banks of its channel onto the floodplain, inundating floodplain wetlands. As the flood recedes, the floodplain wetlands that remain inundated provide refuge for aquatic organisms and sources of energy for aquatic food webs (Venarsky *et al.*, 2020). Primary producers, such as aquatic macrophytes, take

advantage of the physical conditions of floodplain wetlands, where they grow rapidly and provide structure for algae to also flourish (Jardine *et al.*, 2013; Adame *et al.*, 2017; Ndehedehe *et al.*, 2020). Fish seek refuge from predators in the aquatic plants in the wetlands, where they feed on these algae and also invertebrates (Winemiller & Jepsen, 1998). Wetlands also provide habitat for migrant birds and nesting opportunities for waterbirds such as geese and ducks (Franklin, 2008). Additionally, the fertile soils, flat landscapes and accessibility to water resources have made floodplains attractive for agriculture and urban development (Tockner *et al.*, 2008).

1.2 Water-resource development

Water scarcity and poor water-quality issues are faced across the globe, and as the world changes, so do the pressures on freshwater ecosystems (Loucks & Van Beek, 2017). While climate change may exacerbate water scarcity in some regions, population growth, land use change and urbanisation also threaten to increase the number of degraded waterbodies and further degrade those with existing issues (Karr, 1991). Additionally, climate change, population growth and land-use change can increase demand for water, leading to the emergence of new water-resource developments (Milly *et al.*, 2008). There are various technologies for extracting water from natural rivers and lakes. In terms of physical characteristics, these mostly fit into the following categories: surface storage (reservoirs or lakes with artificial outflow control); channelisation (for irrigation, drainage, flood protection or erosion control); direct extraction from the river channel (potentially to off-stream storages); diversion of water via inter-basin transfers, waste treatment and assimilation; and groundwater extraction (Jain & Singh, 2003). While water extraction may provide economic and societal benefits, there are usually (if not always) trade-offs, with loss or reduction of existing ecosystem services (benefits people obtain from ecosystems), including biodiversity loss (Grizzetti *et al.*, 2016). There are several ways in which water-resource developments can impact the ecology of natural rivers and lakes, including creating barriers to movement, reducing flow, and reducing floodplain inundation extent and frequency (Branco *et al.*, 2014).

1.2.1 *Impact pathways of water-resource development*

Barriers to movement

Water-resource infrastructure such as dams and weirs create physical barriers to movement for mobile aquatic species (Larinier, 2001). The physical structure of the barrier influences the degree of fish movement (Weigel, Connolly & Powell, 2013), and some barriers such as dams can be completely impassable, isolating fish from productive habitats and aquatic refugia (Pelicice, Pompeu & Agostinho, 2015). Diadromous species that rely on migrations between fresh and marine waters to reproduce are heavily impacted by these barriers (Liermann *et al.*, 2012). To address this issue, various forms of fish passages have been incorporated into many of these structures (Agostinho *et al.*, 2002). However, these passages are selective, and only strong swimming species such as salmonids can generally use these migration aids effectively (Larinier, 2001; Volpato *et al.*, 2009). By creating physical barriers to movement, water-resource developments cause fragmentation of populations and loss of genetic structuring (Fu *et al.*, 2003).

Flow reductions and habitat changes

The natural flow regime can be severely disrupted by water-resource development (Bunn & Arthington, 2002) and large artificial dams and reservoirs are likely to have the single most important direct impact on the flow regimes of rivers, affecting the timing and magnitude of seasonal flow pulses (Smakhtin, 2001). Smaller floods and longer periods of low flows reduce the connectivity of the river network, leading to extended periods of river fragmentation and ultimately habitat degradation and loss (Ward & Stanford, 1995; Karim *et al.*, 2015). Trapping of sediment by dams and reservoirs can further alter upstream and downstream habitat, for example, by causing downstream channel degradation, which can lower the water table and promote erosion and habitat desiccation (Ward & Stanford, 1995; Schmutz & Sendzimir, 2018) As habitat structure and complexity influences the composition and abundance of biological communities, changes in fish community composition often occur following habitat changes caused by water-resource development (Marchetti & Moyle, 2001).

Reduced floodplain inundation

Floodplain habitats are a mosaic of waterbodies that have varying degrees of flow disturbances that temporarily renew lateral connectivity to the river network (Junk, Bayley & Sparks, 1989; Ward *et al.*, 2002). River regulation affects the rate of river-channel erosion and sedimentation, which influences lateral connectivity through physical changes in river and floodplain morphology (Shields Jr, Simon & Steffen, 2000). Flood peaks are also typically reduced by river regulation, which reduces the frequency and extent of floodplain inundation (Ward & Stanford, 1995). The reduced frequency, extent and duration of floodplain inundation reduces the transfer of water and nutrients between in-stream and floodplain waterbodies (Thoms, 2003). Floodplain wetlands can be highly productive environments, and because of this, they are recognised as important feeding habitats for fish (Opperman *et al.*, 2010; Pettit *et al.*, 2011). Reduced floodplain inundation, and therefore reduced connection times between in-stream and floodplain waterbodies, reduces movement opportunities for fish to feed in wetlands or migrate out of wetlands following stranding from previous seasons (Karim *et al.*, 2016). Floodplain wetlands are not only important for aquatic species but are an important source of food and water for many terrestrial animals and humans (Jackson, Finn & Scheepers, 2014). Drying wetlands may force terrestrial animals and humans to move to new locations in search of food and water (Kingsford, 2000).

1.3 Northern Australian context

1.3.1 Pressures from water-resource development

Pressure from an expanding population and degradation of existing cultivated land in Australia has increased demand for new land suitable for agriculture (Robertson, 2010). The Mitchell River catchment in northern Australia has received considerable attention for agricultural expansion (Commonwealth of Australia, 2015). However, because of the wet–dry tropical climate, year-round irrigation of crops is not possible without the construction of new water-resource developments. In their water-resource assessment for the Mitchell catchment, the CSIRO discussed off-channel water storage and dams as solutions, and identified locations for five potential dams spread across the Mitchell, Palmer and Walsh sub-catchments (Petheram *et al.*, 2018). A key finding of the CSIRO report was that the Mitchell catchment has the potential to support 140,000 ha of year-round irrigated agricultural

development, such as sugarcane. Irrigation on this scale would be based on four large in-stream dams that could release 2,800 GL for agriculture in 85% of years. Off-stream water harvesting could extract 2,000 GL with 85% reliability, which would be sufficient to irrigate 200,000 ha of a single dry-season crop such as cotton. A small dam on the upper Mitchell River (Southedge Dam) is currently the only development in the Mitchell catchment with hydrological impact, though an inter-basin transfer exists from the Barron River to the upper Walsh River, which supports the Mareeba–Dimbulah Water Supply Scheme (Marshall, 2016). The Mitchell catchment is one of the few remaining large catchments in Australia that is mostly unregulated in terms of flows.

1.3.2 Existing users of water

Traditional Owners

There are seven Indigenous groups spread across the Mitchell catchment: the Western Yalanji, Djungan and Bar-Barum peoples on the upper Palmer, Mitchell and Walsh rivers, respectively; the Wakaman people inhabiting the middle-to-lower Walsh River and middle Mitchell; and the Kokominjena, Kokoberra and Kunjen people of Kowanyama, the largest Indigenous community in the catchment, on the lower Mitchell floodplain. Indigenous Australians across these communities make up ~26% of the catchment population and harvest fish, turtles and other resources from wetlands, creeks and river channels as an important part of their diet (Jackson *et al.*, 2014; Lyons *et al.*, 2018). Reliance on the natural resources is particularly high within the floodplain community of Kowanyama, where wet-season floods restrict incoming food supply (Petheram *et al.*, 2018).

Coastal fisheries

The Mitchell River flows westward into the Gulf of Carpentaria through braided estuary channels lined with mangroves, including the Mitchell River Fan Aggregation which is listed in the Directory of Important Wetlands of Australia. The Gulf of Carpentaria supports inshore gillnet and pot fisheries targeting barramundi, king and blue threadfin, barred javelin, sharks, and mud crabs (Griffiths *et al.*, 2010). Offshore commercial fisheries employ otter and fish trawls (targeting prawns), gillnets (targeting tropical scalefish and sharks) and troll-lines (targeting mackerel) (Griffiths *et al.*, 2010). The Gulf also supports several species that are rare, endangered or extinct in other parts of their distribution, including sawfish, whiprays, river sharks and dolphins (Lyons *et al.*, 2018). The estuarine and coastal areas of the Gulf of Carpentaria are low in nutrients, and nutrients transported from the catchment to the coastal zone during wet-season floods are quickly taken up into the food web (Burford *et al.*, 2012; O'Mara, 2019). While floods in the Gulf of Carpentaria may not stimulate increased productivity in the coastal zone (Burford *et al.*, 2010), they cue migrations of important fisheries species including banana prawns *Penaeus merguiensis* (Burford *et al.*, 2010) and barramundi *Lates calcarifer* (Halliday *et al.*, 2012) resulting in increased catches in the Gulf of Carpentaria.

Irrigators in the Walsh catchment

The Mareeba–Dimbulah Water Supply Scheme (MDWSS) irrigates an area of agriculture in the upper Walsh River, which is supplemented with water from an inter-basin transfer from Tinaroo Dam on the Barron River (Lyons *et al.*, 2018). This irrigated agricultural area comprises only 0.3% of the catchment and is dominated by sugarcane and horticulture

(Petheram *et al.*, 2018). Part of the MDWSS extends into the upper reaches of the Mitchell catchment (Lyons *et al.*, 2018).

Graziers throughout the catchment

Pastoralism on large grazing leases is currently the most extensive land use throughout the Mitchell catchment (95%), followed by conservation reserves (3%) (Petheram *et al.*, 2018). About half of the $225 million/year gross value of agricultural production comes from beef production, with cropping (mangoes, sugarcane and avocados) making up most of the remainder (Lyons *et al.*, 2018). Beef production in the Mitchell catchment relies on grazing of natural shrubs and grasses, which are influenced by soil fertility and the natural seasonal variability in rainfall (Lyons *et al.*, 2018). Even though these constraints keep the cattle-carrying capacity low, there are approximately 185,000 cattle in the catchment (Petheram *et al.*, 2018). Livestock grazing on natural resources has reduced ground-cover vegetation, degraded riparian zones and increased alluvial-gully and channel-bank erosion in the Mitchell catchment (Brooks *et al.*, 2009).

1.4 Objectives

There is significant interest in agricultural expansion in the Mitchell River catchment. The region also supports many ecological assets of high value associated with the river systems, including wetlands of national significance, important recreational fisheries and threatened species. There is concern that these ecological assets may be impacted by intensive land and water-resource development. To address this concern, this project aimed to identify flow-dependent ecological assets in the Mitchell River catchment, their critical links to flows, and the implications of future land and water-resource development. This aim is achieved through five objectives:

1. Identifying floodplain inundation and aquatic plant biomass accumulation dynamics and their dependence on hydrologic processes (Section 4.1)
2. Mapping key 'hotspots' for freshwater primary productivity within the Mitchell River floodplain (Section 4.2); this provides information on where important ecological food resources are likely to be located
3. Assessing riverine and floodplain connectivity to assess the likely impact of varying wet seasons and water-resource development on aquatic food webs (Section 5.1), and assessing movement from the river channel to floodplain wetlands (Section 5.2)
4. Assessing the importance of river flows to estuaries by examining the flow dependence of barramundi population dynamics (Section 6.1) and barramundi growth (Section 6.2), and the relationship between the use of marine and freshwater habitats and barramundi length (Section 6.3)
5. Assessing the implications of development on flow-dependent ecological communities (Section 7), including fish assemblage composition (Section 7.1), fish movement through the catchment (Section 7.2) and food sources for the fish (Section 7.3).

These objectives were addressed through a range of ecological studies conducted throughout the catchment, from headwaters to the coast. Collectively, this study tells the eco-hydrological story of the Mitchell River, describing the role of flows and how they facilitate connectivity among aquatic habitats and how ecological communities respond to that.

Section 2 provides an overview of the Mitchell River and its hydrology. The methods used in each study are described in Section 3; Sections 4, 5, 6 and 7 comprise the key findings of each study. Finally, Section 8 provides the conclusions and synthesis of the research.

2. Mitchell River overview

The Mitchell River catchment is situated in the wet–dry tropics of north Queensland, Australia, and covers an area of approximately 72,000 km^2 (Petheram *et al.*, 2018) (Figure 2-1). The westward-flowing main channel of the Mitchell River stretches from the headwaters in the Daintree rainforest in the east to the river mouth in the Gulf of Carpentaria in the west. Major tributaries that flow into the Mitchell River include the Walsh, Lynd, Alice and Palmer rivers. Geology and river form vary throughout the catchment, with the eastern third comprising bedrock varying between sedimentary, granitic and volcanic lithology (Batlle-Aguilar *et al.*, 2014). A fan-shaped sediment deposit at lower elevation spreads west from its apex at the confluence of the Mitchell and Palmer rivers, producing a network of braided channels and creeks on the floodplain (Rustomji *et al.*, 2010). Rainfall in the Mitchell catchment is highly seasonal, with only 4% of annual rainfall (on average) falling across the catchment during the dry season from May to October (Petheram *et al.*, 2018).

Figure 2-1. Location of the Mitchell River catchment in north Queensland.

2.1 Hydrology

Historical data show that the main channel of the Mitchell River experiences perennial flow in most years, while the Palmer and Walsh rivers experience cease-to-flow conditions in most years (an average of 45 and 33 zero-flow days per year, respectively). The high variation of discharge between the wet and dry seasons can be observed in four illustrative gauging stations (Table 2-1) in the Mitchell River main channel (Figure 2-2). The mean monthly discharge data were obtained from the Queensland Water Monitoring Information Portal (water-monitoring.information.qld.gov.au, accessed on 4 June 2021).

Table 2-1. Gauging stations on the mainstem of the Mitchell River. Note Mitchell River and Koolatah and Mitchell River at Dunbar are adjacent and effectively represent a single time series of data.

Station code	Name	Latitude	Longitude	Elevation (m)
919003A	Mitchell River at O.K. Bridge	−16.4708	144.2894	174
919009A	Mitchell River at Koolatah	−15.9509	142.3772	0
919009B	Mitchell River at Dunbar	−15.9424	142.3743	0
919011A	Mitchell River at Gamboola	−16.5345	143.6772	105
919014A	Mitchell River at Cooktown Crossing	−16.5631	144.8897	290
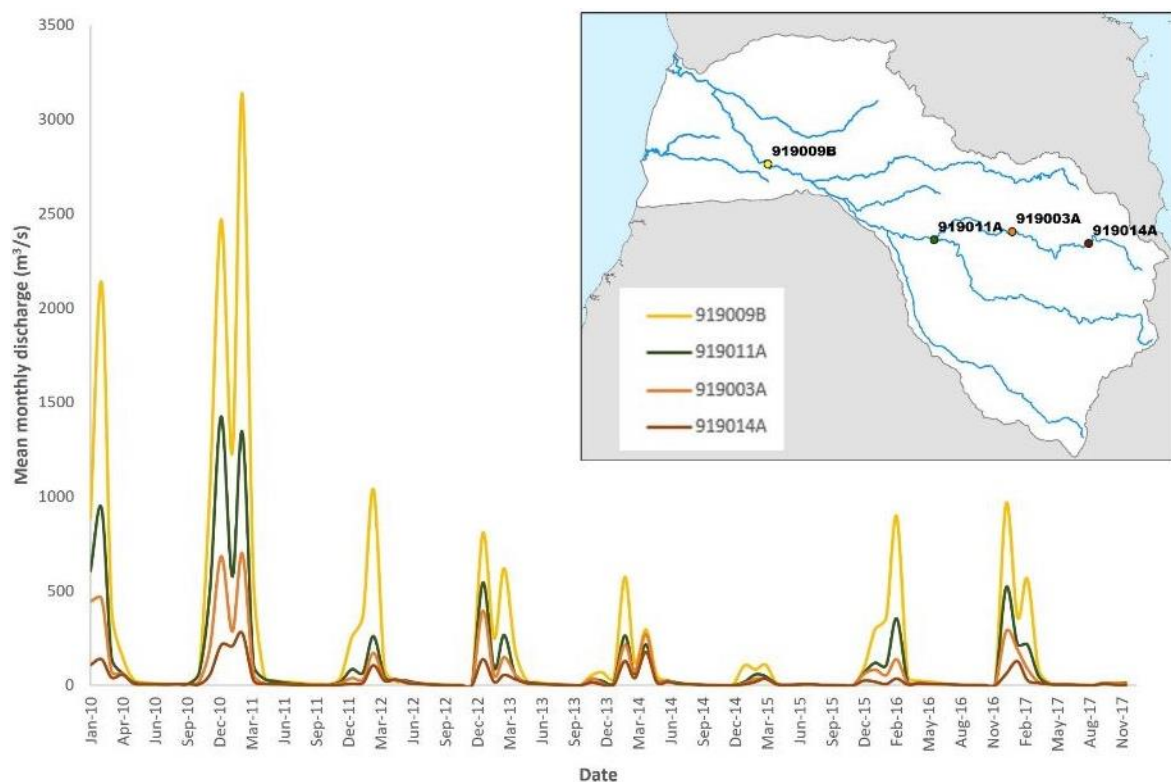
Figure 2-2. Hydrographs at four gauging stations (919009B, 919011A, 919003A and 919014A) from 1 January 2010 to 31 December 2017. The mean monthly discharge (cubic metres per second) data from those dates were obtained from the Queensland Water Monitoring Information Portal (water-monitoring.information.qld.gov.au, accessed 4 June 2021).

In this research, we used the CSIRO's hydrodynamic model (Karim et al. 2018) and river-flow model (Hughes et al. 2017) from the Northern Australia Water Resource Assessment

(NAWRA) to derive water-resource development scenarios for evaluation (Table 2-2). Each of these scenarios were used in different ecological studies that are described below.

Table 2-2. Water-resource development scenarios for scenario evaluation.

Scenario	Description	Model settings	Characteristics
WRD (water-resource development) scenario	WRD scenario for floodplain connectivity analysis	Inclusion of three potential dams: the Pinnacles dam on the main Mitchell River; the Rookwood dam, on the Walsh River; and the Palmer dam, on the Palmer River	Reduced floodplain inundation extent, reduced connectivity among river and wetland sites
Scenario JBG (juvenile barramundi growth)	WRD scenario for changes in JBG rates analysis	Inclusion of a 6,000 GL total extraction from three different nodes, with a low pump-start threshold (200 ML/day) and zero allocation to end-of-system flow	Reduced river discharge in the estuarine areas of the catchment
All-dams scenario		Inclusion of five potential dams: three Walsh River dams, one Mitchell River dam (Pinnacles dam), and one Palmer River dam	
Three-dams scenario	WRD scenarios for river-channel connectivity index analysis	Inclusion of Palmer dam, Pinnacles dam, and one dam on the Walsh River (Chillagoe dam)	Reduced connectivity among river sites, dams act as barriers for fish movement
Chillagoe dam scenario		Inclusion of Chillagoe dam only	
Palmer dam scenario		Inclusion of Palmer dam only	
Pinnacles dam scenario		Inclusion of Pinnacles dam only	

2.2 Study sites

Sampling was conducted across the entire catchment to collect data on fish assemblage composition and the food web, primary productivity and water quality (Figure 2-3). Given the broad array of studies, the sampling locations can be divided into sites on the floodplain (Table 2-3), and sites in creeks and river channels (Table 2-4).
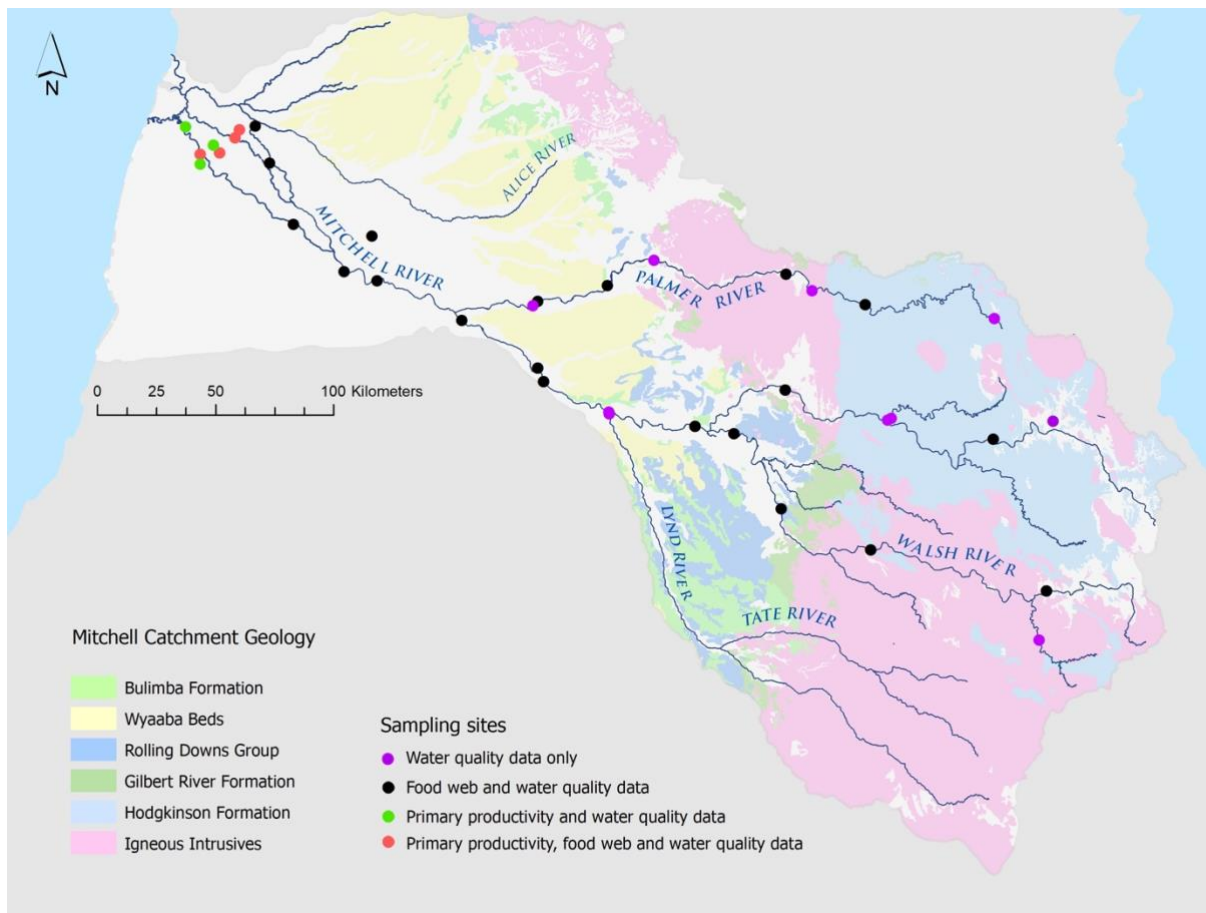
Figure 2-3. Map of the Mitchell River catchment showing the location of sampling sites and the underlying geology.

2.2.1 Floodplain wetlands

Across the Mitchell River floodplain, 11 floodplain wetlands (Table 2-3) were sampled for water quality assessment, primary productivity measurements, and fish collection and food web analysis.

Table 2-3. Wetland sites information table. FW codes are the codes used in the food web analysis (Section 7.2 and Section 7.3). Types of samples: FC = fish collection and food web analysis, PP = algal primary productivity, WQ = water quality.

Site code	FW code	Site name	Latitude	Longitude	Elev- ation (m)	Date sampled	Type of sample
9191006	A1	Koolatah Lake	−15.7917	142.4059	42	04/07/2017	WQ, FC
9190083	M30	Yelko Waterhole	−15.4748	141.825	14	27/04/2018	WQ, PP
						06/06/2018	WQ, FC
						11/05/2019	WQ, FC
9190086	M36	Sandy Creek	−15.4177	141.8839	10	02/05/2018	WQ, PP
						07/06/2018	WQ, FC
						08/05/2019	WQ, FC
9190087	M29	Leichardt Waterhole	−15.7469	142.1055	30	11/06/2018	WQ, FC
9190073	M24	Twelve Mile Lagoon	−16.2949	143.0368	90	15/10/2017	WQ
						14/06/2018	WQ, FC
9190034	M35	Magnificent Creek	−15.4805	141.7507	14	25/04/2018	WQ, PP
						09/05/2019	WQ, FC
9190085	M34	Brumby Waterhole	−15.3862	141.8998	13	01/05/2018	WQ, PP
						10/05/2019	WQ, FC
9190088	M38	White Water	−15.5138	142.0152	19	13/05/2019	WQ, FC
9190081	M37	Ten Mile	−15.3756	141.6947	7	24/04/2018	WQ, PP
9190082	M32	Wilson Hole	−15.5178	141.7511	13	26/04/2018	WQ, PP
9190084	M33	Racecourse	−15.4453	141.8017	13	30/04/2018	WQ, PP
						14/05/2019	WQ

2.2.2 Riverine/in-stream reaches

Across the Mitchell River catchment, 27 riverine/in-stream reaches were sampled (Table 2-4) for water quality assessment (Section 3.1), and fish collection and food web analysis (Section 3.6).

Table 2-4. Riverine and in-stream sites information table. FW codes are the codes used in the food web analysis (Section 7.2 and Section 7.3). Types of samples: FC (fish collection and food web analysis), WQ (water quality).

Site code	FW code	Site name	Lat	Long	Elev (m)	River distance to Mitchell River mouth (km)	Date sampled	Type of sample
9190015	M7	Mitchell River upstream of Hurricane Station	−16.5661	144.7733	287	598	19/06/2017	WQ, FC
							21/10/2017	WQ
9190011	M15	Mitchell River at Gordon Arthur Crossing	−16.3791	143.9811	146	444	21/06/2017	WQ, FC
							20/10/2017	WQ
							15/06/2018	WQ, FC
9190010	M19	Mitchell River at Gamboola	−16.5171	143.6364	119	381	29/06/2017	WQ, FC
							12/10/2017	WQ
							19/10/2017	WQ
9190035	M23	Mitchell River at Hughes Crossing	−16.3474	143.0595	89	277	02/07/2017	WQ, FC
							14/10/2017	WQ
							12/06/2018	WQ, FC
9190043	M27	Mitchell River at Koolatah Crossing	−15.9626	142.4246	49	163	03/07/2017	WQ, FC
							16/10/2017	WQ
919009a	M28	Mitchell River at Koolatah	−15.9281	142.2986	44	143	10/06/2018	WQ, FC
9190036	M31	Mitchell River at Shelfo	−15.3727	141.96	15	54	08/06/2018	WQ, FC
9190037	M21	Mitchell River at Lynd Junction	−16.4631	143.3089	97	n/a	14/10/2017	WQ
9190079	M25	Mitchell River upstream of junction with Palmer River	−16.114	142.7479	67	n/a	15/10/2017	WQ
9190009	M4	McLeod River at Mulligan Highway	−16.4983	145.0022	334	n/a	21/10/2017	WQ
9190023	M12	St George River upstream of junction with Mitchell River	−16.4866	144.3853	200	n/a	21/10/2017	WQ
9190069	M11	Mitchell River at St George River junction	−16.494	144.371	195	n/a	22/10/2017	WQ

Site code	FW code	Site name	Lat	Long	Elev (m)	River distance to Mitchell River mouth (km)	Date sampled	Type of sample
919205a	P1	Palmer River at Maytown	−16.0539	144.2856	252	494	22/06/2017	WQ, FC
							18/10/2017	WQ
9192008	P3	Palmer River downstream of Fern Hill Bend	−15.9375	143.984	190	436	23/06/2017	WQ, FC
919203a	n/a	Palmer River downstream of gauging station 919203A	−15.9809	143.3025	114	315	01/07/2017	WQ, FC
9192007	P5	Palmer River at Drumduff	−16.0414	143.0373	95	269	13/06/2018	WQ, FC
9192006	P6	Palmer River at Drumduff Crossing	−16.057	143.0187	82	n/a	14/10/2017	WQ
9192002	P2	Palmer River at Palmerville Station	−16.0002	144.0834	199	n/a	18/10/2017	WQ
9192009	P4	Palmer River near King Junction	−15.8845	143.4807	126	n/a	19/10/2017	WQ
919201a	n/a	Palmer River at Palmer River Goldfields	−16.1071	144.7786	410	n/a	23/10/2017	WQ
9193028	W3	Walsh River downstream of junction with Bulluburrah Creek	−17.1448	144.9771	427	653	28/06/2017	WQ, FC
							12/10/2017	WQ
							20/10/2017	WQ
9193022	W4	Walsh River at Picnic Hole	−16.9889	144.3062	225	527	25/06/2017	WQ, FC
							20/10/2017	WQ
9193007	W1	Emu Creek at Petford	−17.333	144.9487	469	n/a	20/10/2017	WQ
9193014	W5	Walsh River downstream of gauging station 9193014	−16.8315	143.9652	155	459	24/06/2017	WQ, FC
919309a	W6	Walsh River at Trimbles Crossing	−16.546	143.7856	121	407	05/07/2017	WQ, FC
							12/10/2017	WQ
9193010	W8	Eureka Creek at junction with Walsh River	−17.1438	144.9808	428	n/a	12/10/2017	WQ
							20/10/2017	WQ
9190007	L1	Lynd River at Junction with Mitchell River	−16.4703	143.3096	97	n/a	12/10/2017	WQ

3. Methodology

This chapter provides a broad and general overview of the methodology and brief explanations of the specific methods. More detailed explanations of the methods can be found in each of the open-access publications listed on page 7.

3.1 Water quality

Water quality measurements were used to characterise water quality in the Mitchell River catchment across different years and seasons. The water quality parameters measured in the floodplain were also used in the process of building a statistical model to predict algal primary productivity (Section 3.3, Section 4.2). Additionally, water samples provided an estimate of the strontium ($^{87}Sr/^{86}Sr$) and sulfur ($^{34}S/^{32}S$, expressed as $\delta^{34}S$) stable isotopes at the time of sampling, to define isoscape regions in food web analyses (Section 3.8, Section 5.2, Section 7.2).

Water samples for water quality analysis were collected throughout the Mitchell River catchment. In total, water samples from 38 sites were collected over five trips in June and July 2017, October 2017, April/May 2018, June 2018 and May 2019. Several water quality parameters were measured including total suspended solids (TSS), coloured dissolved organic matter (CDOM), ammonium (NH_4-N), total Kjeldhal nitrogen (TKN), oxidised nitrogen (NOx-N), total nitrogen (TN), total phosphorus (TPh), phosphate (PO_4-P), total Kjeldhal phosphorus (TKP), dissolved Kjeldhal nitrogen (DKN) and dissolved Kjeldhal phosphorus (DKP). Analyses were conducted by the Science Division Chemistry Centre, Department of Environment and Science, Queensland Government in Brisbane, according to standard methods (APHA-AWWA-WEF, 2005).

3.2 Floodplain inundation and plant accumulation

Primary productivity in the floodplains supports numerous important freshwater ecological assets, traditional harvests and coastal fisheries. This section of the study addressed **Objective 1,** to understand the key drivers of floodplain inundation and primary productivity of freshwater ecosystems in the Mitchell River. To achieve that, a framework was applied that included remote sensing techniques and statistical modelling. Remotely sensed images were used to predict floodplain inundation and aquatic plant biomass accumulation across the Mitchell River floodplain. Additionally, the dependence of floodplain inundation and aquatic plant accumulation on the hydro-climatology of the upstream catchment was measured. For a detailed explanation of this method, refer to Ndehedehe et al. (2021).

The remotely sensed data used in this section were from the Global Surface Water (GSW) dataset, Landsat 5 and 8, SPOT 7, the Dynamic Land Cover Dataset (DLCD) and Sentinel-2 (Table 3-1). The GSW dataset (available from the European Joint Research Centre [Pekel et al. 2016]) provides the worldwide extent of surface water between 1984 and 2015 (Table 3-1). This dataset was used to identify water occurrence (inundation frequency) and long-term changes in surface-water permanence in the catchment. Level 1 Landsat 5 and Landsat 8 observations for the 2007–2019 period were retrieved from Google Earth Engine and used to compute total surface inundation in the wet and dry seasons (Table 3-1). The median values of overlapping cells for each available Landsat scene across each dry/wet season were used to obtain the optical indices to identify open-water features. These data

were also used in change detection of open-water features in analyses for two periods: 2007–2014 and 2014–2019. The GSW dataset and the DLCD for Australia (Lucas *et al.*, 2019) were used to perform accuracy assessment of the open-water features mapping. To estimate aquatic plant biomass accumulation, Sentinel-2 top-of-atmosphere reflectance data were used for the summer wet seasons (December–March) and dry periods (April–August) between 2015 and 2020. The median value for each period was used to retrieve vegetation metrics and to identify hotspots of aquatic primary productivity in the Mitchell River floodplain.

Table 3-1. Satellite images and products, resolutions, and acquisition dates.

Name	Resolution	Acquisition dates
Landsat 5	30 m	Dec 2006 to Mar 2007 Jun 2007 to Aug 2007
Landsat 8	30 m	Dec 2013 to Mar 2014 Jun 2014 to Aug 2014 25 Apr 2018 Dec 2018 to Mar 2019 Jun 2019 to Aug 2019
SPOT 7	6 m	25 Apr 2018
Sentinel-2	10 m	2015–2020
Global Surface Water dataset	30 m	2007–2019
Australian Dynamic Land Cover Dataset	250 m	2001–2015

3.2.1 Drivers of floodplain inundation – rainfall and flows

The main drivers of floodplain inundation are upstream run-off, localised rainfall, or a combination of the two (e.g. Ndehedehe *et al.*, 2020). The coupled relationship of discharge and local rainfall was modelled to assess the drivers of floodplain inundation (Section 4).

The observed rainfall data used were the monthly Australian interpolated gridded rainfall data of the Mitchell River. Accumulated total rainfall, from 2000 to 2019, was accessed from the Australian Bureau of Meteorology climate database (bom.gov.au/climate/data). For the historical flow data for 2000 to 2019, five gauging stations were used: 919003A (Mitchell River at O.K. Bridge), 919009B (Mitchell River at Dunbar), 919011A (Mitchell River at Gamboola), 919014A (Mitchell River at Cooktown Crossing) and 919203A (Palmer River at Strathleven) (Table 2-1). For a complete methodology, refer to Ndehedehe et al. (2021).

3.3 Floodplain algal productivity

Algae are the major basal component of freshwater food webs (Lewis Jr *et al.*, 2001; Jardine *et al.*, 2013; Saigo *et al.*, 2015). In tropical Australia, a large percentage of fish biomass in in-stream systems is derived from floodplain algae (range 42%–97%; Jardine et al. 2013). This section of the study addressed **Objective 2**, which is to map 'hotspots' of algal productivity on the Mitchell River floodplains.

3.3.1 Field data collection

The Mitchell River floodplain includes three wetland types: riverine (contained within a channel, e.g. river, stream), lacustrine (large waterbodies with permanent water, e.g. lakes and lagoons) and palustrine (wetlands that have fluctuating water residency, e.g. swamps) (Shine & Klemm, 1999). Within these wetlands, four categories of habitats were described according to their dominant macrophyte structure: emergent habitats, floating habitats, submerged habitats and open water. Emergent habitats have grasses and sedges rooted in shallow water with vegetative parts emerging vertically above the water surface. Floating habitats have plants rooted in shallow water with floating leaves. Submerged habitats have plants rooted in the benthos with the vegetative parts predominantly submerged, and open-water habitats are largely free of aquatic plants (Molinari et al. 2021a; Figure 3-1). These aquatic habitats were used to assess algal productivity (Section 4).
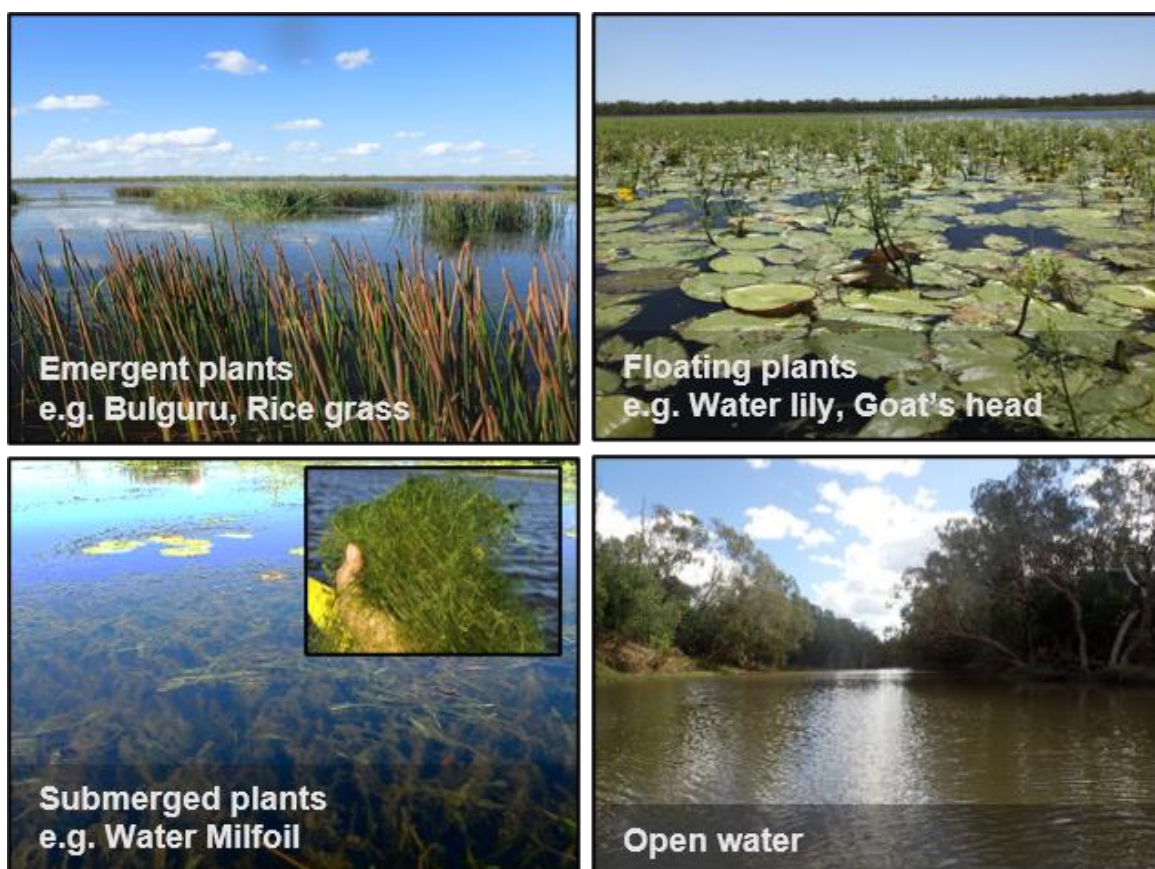
Figure 3-1. Habitats observed on the Mitchell River floodplains. Photos: Cameron Schulz, Ben Stewart-Koster and Bianca Molinari.

There was a single field trip to measure algal primary productivity in late April 2018, at the end of the wet season when the wetlands were still inundated. The sample design included seven wetlands: two lacustrine wetlands (Wilson Hole and Brumby Waterhole), two palustrine wetlands (Ten Mile and Racecourse), and three riverine wetlands (Magnificent Creek, Yelko Waterhole and Sandy Creek) (Table 2-3). Within each wetland, we selected between one and three locations of each habitat type (Figure 3-2) resulting in a total of 63 observations: 23 in emergent habitats, 19 in floating habitats, 14 in submerged habitats, and seven in open-water habitats. For a complete description of the sampling, refer to Molinari et al. (2021a).
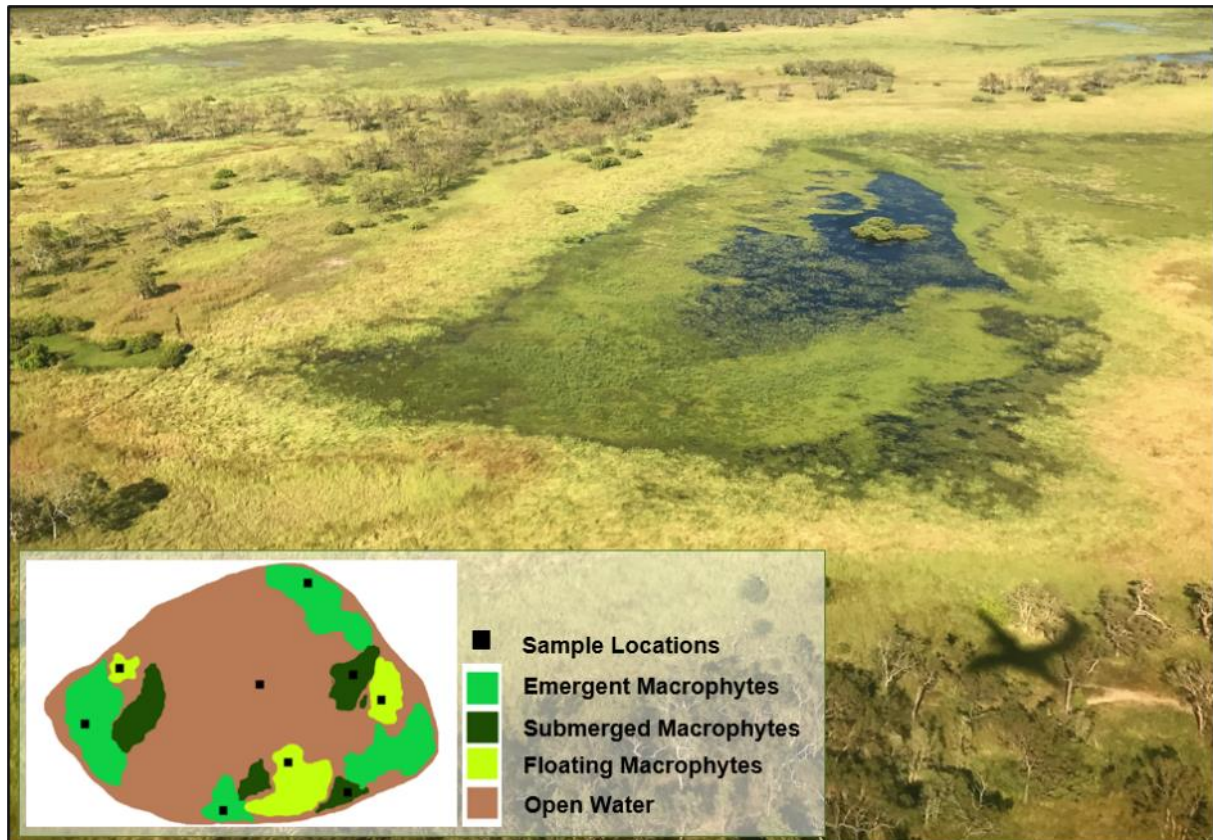
Figure 3-2. Aerial view of a Mitchell River wetland on 23 April 2018. Inset shows a conceptual illustration of the sampling design at a hypothetical wetland. Photo: Bianca Molinari.

During the floodplain productivity trip, photosynthetically active radiation profiles were measured at each floodplain site using a Licor logger (Figure 3-3). Temperature, salinity, pH, dissolved oxygen and turbidity were measured using a calibrated sonde (EXO). At each wetland, we collected water samples to measure the nutrients and physical variables such as CDOM and TSS, as described in Section 3.1.

Figure 3-3. Collection of photosynthetically active radiation profile and plant biomass. Photos: Bianca Molinari.

In each replicate habitat containing macrophytes, we harvested plants using a 0.5×0.5 m quadrat and measured the plant biomass (Molinari et al. 2021a). Algal biomass was measured using chlorophyll *a* (Chl-*a*) as a proxy. In the plant habitats (emergent, floating and submerged habitats), algae were scrubbed from the leaves and stems of a known weight of macrophyte with a soft brush and tap water, and later filtered onto a glass fibre filter. In the open-water habitats, a known volume of water with floating algae was taken from the water sample and filtered onto a glass fibre filter. The samples were immediately frozen and kept in the dark until returned to the laboratory, where Chl-*a* was extracted with 90% acetone using a sonicator, and read with a spectrophotometer (Shimatzu) (Jeffrey & Humphrey, 1975).

Algal productivity was measured as carbon (C) uptake from incubations with ^{13}C-bicarbonate (Adame et al. 2017; Burford et al. 2012; Molinari et al. 2021a). The plants (emergent, floating and submerged plants) with the attached algae were placed inside clear polyethylene bottles

(500 mL) filled with local water (Figure 3-4). For the open-water habitats, the same bottles were filled with the water from each sample location. The bottles were placed inside large containers with three different light treatments: full light, 50% light and dark treatment (5% light). To begin the experiment, we added ^{13}C-bicarbonate (99 atom % ^{13}C; Cambridge Isotope Laboratories, Tewksbury, MA, USA; 40 µM solution) to each bottle. Samples were incubated for 2–3 hours during the day at full sunlight. At the end of the experiment, we stopped the incubation by refrigerating the bottles (4°C) in the dark. Samples were analysed for ^{13}C/^{12}C with an elemental-analyser isotope ratio mass spectrometer (EA-IRMS, Sercon System, Crewe, UK), from which we calculated algal productivity. For a detailed description of the methods, refer to Molinari et al. (2021a).

Figure 3-4. Incubation bottles and three light treatments used for measuring algal primary productivity on the Mitchell River floodplain. Photos: Bianca Molinari.

3.3.2 Statistical modelling

The locally sampled variables were used to build a statistical model to predict algal primary productivity at new locations (Section 4.2). In the construction of the model to estimate algal productivity, all available variables were tested: habitat type (macrophyte structures and open water); nutrients such as ammonium, DKN, TKN, phosphate, DKP, TPh, and the molar ratio of TN to TPh (TN/TPh); light variables; and physical variables including CDOM, TSS and turbidity.

3.3.3 Spatial variation in algal productivity

Remotely sensed images were combined with the previously developed statistical model to predict algal productivity across the Mitchell River floodplain (Section 4.2.1). Landsat 8 Operational Land Imager and SPOT 7 imagery, obtained on 25 April 2018, were used (Table 3-1). The Landsat scene (185×185 km, path 98, row 71) was converted to surface reflectance, from which we extracted bands and indices that were used for algal primary productivity predictions. A single, high-resolution SPOT 7 scene was used to validate the classification of habitats. See Molinari et al. (2021b) for a complete description of the methodology.

3.4 Floodplain connectivity using spatial methods

To assess the likely impact of varying wet seasons and water-resource development on aquatic food webs, floodplain connectivity was assessed to address **Objective 3** of this study.

3.4.1 Flood event data for floodplain connectivity analysis

Modelled floodplain inundation data were used to evaluate the connectivity of floodplain wetlands across the whole of the Mitchell River floodplain (Section 5.1). The inundation model was created by NAWRA using a two-dimensional hydrodynamic model (MIKE21) (Karim *et al.*, 2018). The hydrodynamic models were calibrated using observed daily discharge and stage height data, and Moderate Resolution Imaging Spectroradiometer (MODIS) and Landsat imagery. Daily inundation depth (30×30 m pixel resolution) was modelled for a 40-day period for a dry year (2006), an average year (2001) and a wet year (2009) (Table 3-2).

Table 3-2. Northern Australia Water Resource Assessment historical floodplain inundation scenarios.

Year	Start date	End date	Peak inundation day	Duration (days)
2001	15/02/2001	26/03/2001	05/03/2001	40
2006	10/04/2006	19/05/2006	28/04/2006	40
2009	01/02/2009	12/03/2009	12/02/2009	40

3.4.2 Water-resource development scenario for floodplain connectivity analysis

Floodplain inundation data in the 'WRD (water-resource development) scenario' considered the inclusion of three of the five potential dams in the Mitchell River: the Pinnacles dam on the main Mitchell River, approximately 80 km upstream of the confluence of the Mitchell and Walsh rivers; the Rookwood dam on the Walsh River; and the Palmer dam on the Palmer River (Karim et al. 2018; Petheram et al. 2018; Figure 3-5, Table 2-2). Daily inundation depth (30×30 m pixel resolution) for the WRD scenario was performed for the same 40-day period as the historical scenarios for 2006, 2001 and 2009 (Table 3-2). These scenarios were used to evaluate the impact of water-resource development on floodplain algal productivity and connectivity (Section 5.1).
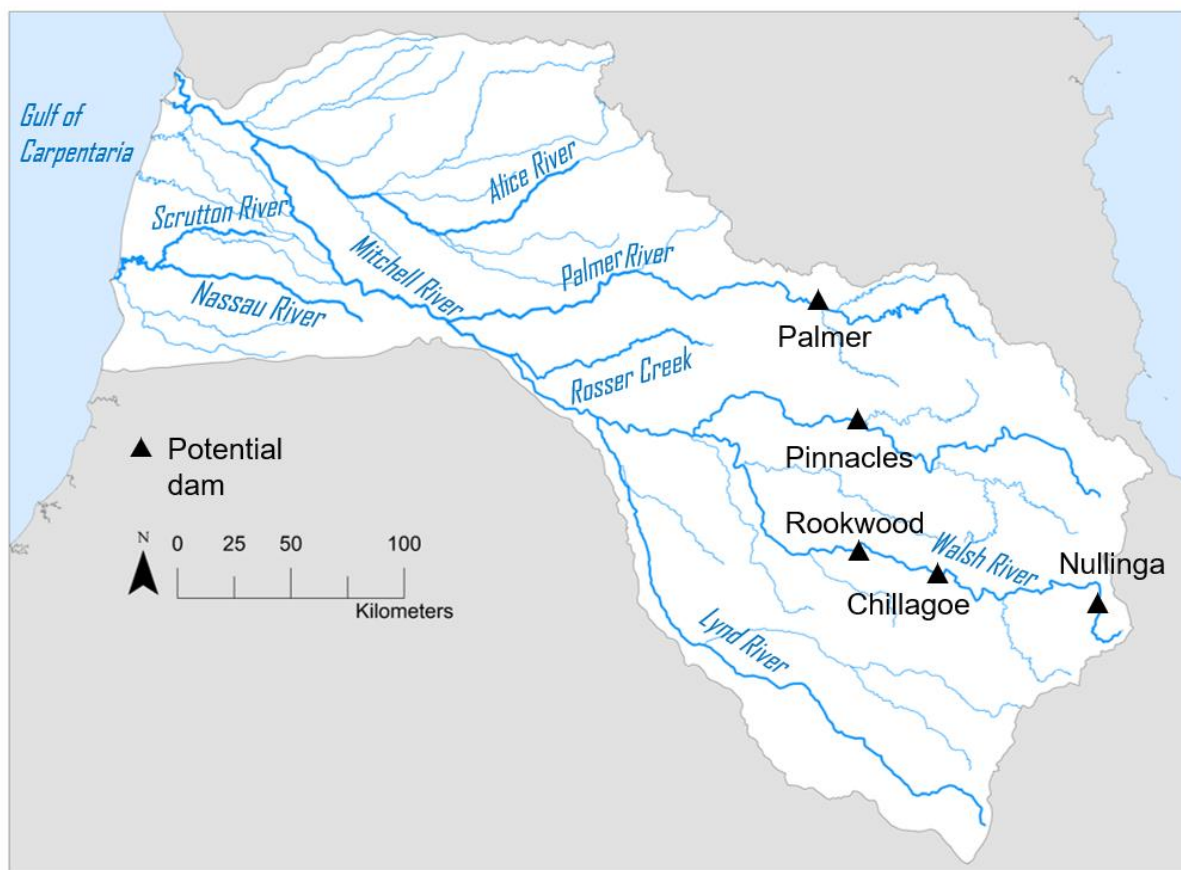
Figure 3-5. Location of five potential dams (Palmer, Pinnacles, Rookwood, Chillagoe and Nullinga) in the Mitchell River catchment.

3.4.3 Graph theoretical approach

Floodplain connectivity was quantified via a graph theoretical approach (Section 5.1). From the daily inundation depth maps (Section 3.4.1), pixels with an estimated depth of water over 10 cm were considered inundated by surface water. Connections between habitats (pixels in the inundation model) were made when a pixel and any of its eight immediate neighbours were inundated with surface water (Figure 3-6). This process was repeated for each pixel to identify connectivity across the floodplain and create a floodplain spatial graph.
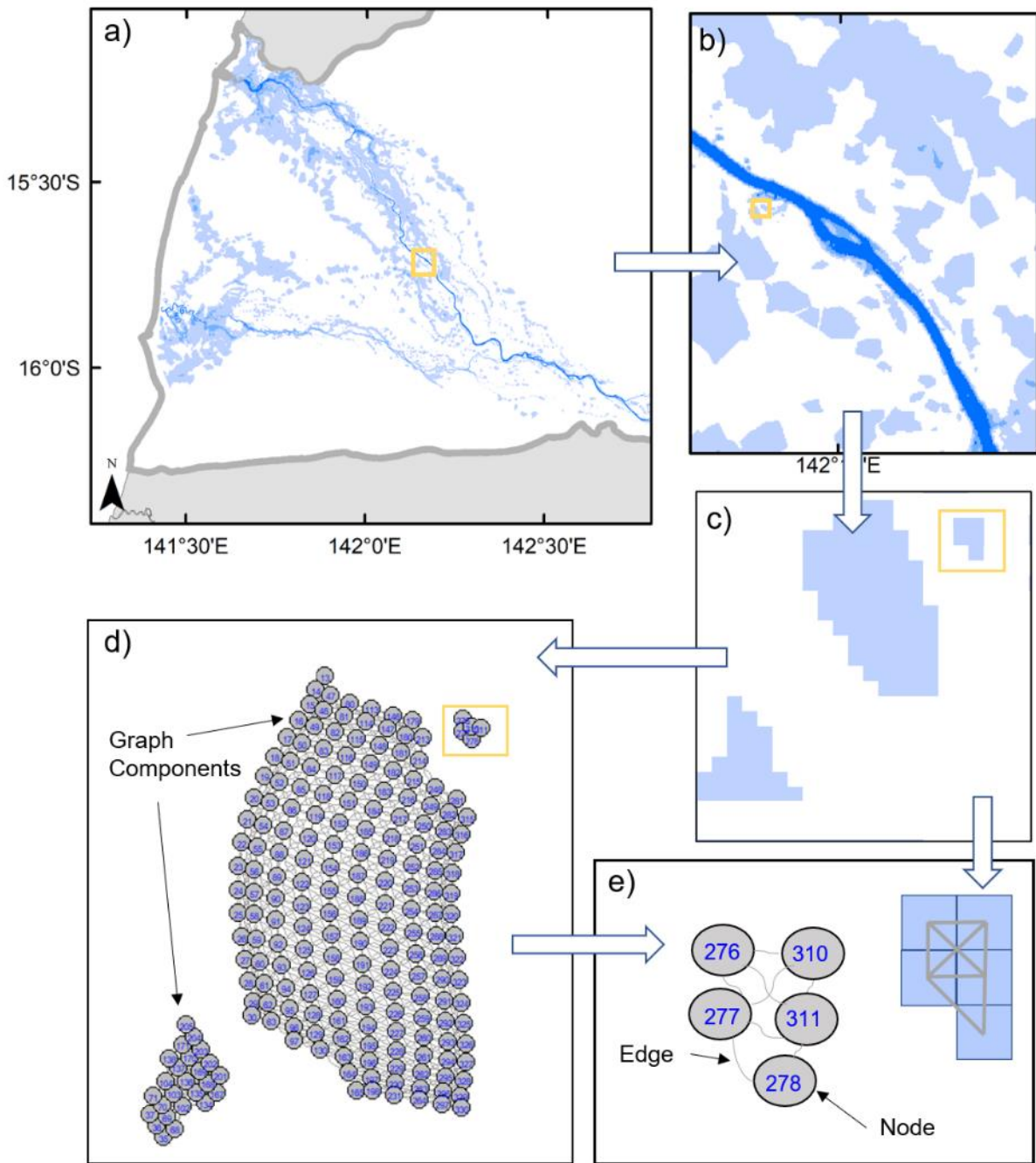
Figure 3-6. The process of transforming floodplain inundation into a spatial graph: a) the extent of Mitchell River floodplain inundation (simulated for 20 February 2009); b) the portion of the floodplain inside the yellow box in a); c) the portion at the floodplain inside the yellow box in b); d) conceptual illustration of a spatial graph of the pixels in c); and e) illustration of key terms in spatial graphs.

One floodplain spatial graph was created for each day of the historical 40-day period of 2001, 2006 and 2009 (Table 3-2), with and without a WRD scenario. A set of connectivity metrics were derived for each day in the 40-day periods, to quantify the connectivity of the whole floodplain. By comparing connectivity metrics between years and scenarios, we can identify how varying inundation, either due to variable wet seasons or water-resource development, can impact connectivity of productive habitats across the floodplain.

The connectivity metrics used were total number of edges, number of components and maximum component size (Table 3-3). Additional metrics created to measure the impact of changing connectivity on productive habitats were inundated habitats, inundated productivity, connected habitats and connected production. For further detail on the graph creation and metrics calculation, refer to Molinari et al. (2021c).

Table 3-3. Graph theoretic connectivity metrics and landscape productivity metrics: description and ecological interpretation.

	Metric name	Description	Ecological interpretation
Graph metrics	Graph edges	Number of edges of the graph	Higher number of edges indicates higher connectivity, providing greater potential patches for animal movement among inundated areas
	Number of components	Number of components of the graph	Higher number of components indicates reduced connectivity due to greater level of landscape fragmentation
	Maximum component size	Size of the largest component of the graph	Larger components indicate higher connectivity providing increased opportunities for animal movement among inundated areas
Landscape productivity metrics	Inundated habitats	Number of 'productive habitats' inundated	Larger numbers indicate a larger number of productive habitats that are inundated
	Connected habitats	Number of 'productive habitats' at the largest component of the graph	Larger numbers indicate a larger number of productive habitats that are accessible to higher consumers from the main river channels
	Inundated productivity	Sum of algal primary productivity in the inundated 'productive habitats'	Higher inundated productivity indicates high primary productivity being generated as a consequence of larger inundated areas
	Connected productivity	Sum of algal primary productivity in the 'productive habitats' on the largest component of the graph	Higher connected productivity indicates more primary productivity that is accessible to higher consumers from the main river channels

3.5 Coastal finfish catch and growth

To assess the importance of river flows to the Mitchell River estuary (**Objective 4**), the iconic barramundi (*Lates calcarifer*) was used as an indicator species. Barramundi catch, age, growth and length were assessed in relation to flows and freshwater habitat use. For a complete methodology, refer to Robins et al. (Robins *et al.*, 2021).

3.5.1 Commercial finfish catch

Barramundi was used as an indicator because of their long life cycle which includes a variety of aquatic habitats (coastal, estuary, and freshwater habitats such as wetlands) (Milton & Chenery, 2005; Milton *et al.*, 2008). While the positive influence of rainfall and river flows on barramundi catches is well known (e.g. Robins et al. 2005), the analysis of age-structure can provide insights into likely causes of increases in catch. A series of analyses of catch and growth were conducted to quantify their relationships to river flows and potential impacts of flow alteration from water-resource development (Section 6).

Catch and age data

The historical barramundi catch, length, gender and age data were obtained by the Queensland Department of Agriculture and Fisheries (DAF), which has monitored barramundi at the genetic stock level for the southern Gulf of Carpentaria between 2000 and 2018. The southern Gulf barramundi stock extends across numerous major river systems. For the current analysis at the Mitchell River, age-frequencies were analysed at a sub-stock level, in the region between 13ºS and 16ºS ('mid' region of the southern Gulf) (Robins *et al.*, 2021). Harvest data for barramundi and associated effort data (days fished) were derived from the compulsory commercial fishery logbook system (CFISH) database (DR2830).

In the Gulf of Carpentaria, barramundi spawn over spring and summer (i.e. October to March), with 1 January adopted as the nominal Queensland birth date. The year-class (i.e. birth year) was assigned on the basis of spawning year. For example, fish spawned between October 2008 and March 2009 are allocated to the 2009 year-class. The age-frequencies considered in the current work are relative frequencies, as sample sizes vary between collection years, and are equivalent to an age-specific catch per unit effort (Morrongiello *et al.*, 2014).

Year-class strength

Indices of recruitment were based on a catch-curve approach (Maceina, 1997), using the sampled age-frequencies. In the current work, analyses considered 19 sequential years of barramundi age-frequencies sampled between 2000 and 2018. Age-classes between 3 and 9 years of age were considered to be fully available to the fishery, reflecting the gillnet mesh sizes used in the Gulf. The mesh size of a gillnet determines the size of the fish that are caught. Larger mesh sizes allow smaller fish to swim through the net, while smaller mesh sizes exclude larger fish as they are blocked by the net and do not become ensnared in it.

Catch-at-age

As an extension to standard year-class strength indices, the age-frequency data with reported harvest and effort data were also used to derive a catch-at-age frequency per unit effort. Catch-at-age is appropriate for barramundi from the southern Gulf of Carpentaria stock, because harvest is size-selective not age-selective.

Analysis with flow

Statistical models were built to identify the relationships between age-frequencies over time and potential explanatory flow variables. The end-of-system (EOS) flows for the Mitchell River used in the models were provided by the CSIRO via Fisheries Research and Development Corporation (FRDC) project 2018/079 ('Ecological modeling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the Northern Prawn Fishery').

As the purpose of the analysis was to inform planning for water-resource development scenarios, analyses were conducted for: i) monthly flows, and ii) seasonal flows that occurred within the year matching the 'birth year'. It became apparent during the current project that the age-structure of barramundi harvested from the Gulf of Carpentaria was the consequence of not only flows in the birth year, but also the sequential pattern of flows over multiple years, based on the patterns in the catch-at-age matrix. To account for the influence of flows over sequential years on year-class strength and catch-at-age, flow variables were also constructed for: iii) seasonal flows that occurred between the first and second birthday; iv) seasonal flows that occurred between the second and third birthday, and v) flows occurring between the third birthday and capture, which includes the flows in the sample year.

3.5.2 Finfish growth

Barramundi otoliths (ear bones) were analysed to quantify growth in the barramundi caught in the fishery. The otoliths were sourced from the DAF archive for the collection years between 2000 and 2018. Samples from the Mitchell region were individuals captured from the following locations: the Mitchell River, commercial fishery logbook grids AB13 and AC14 grids, and the Nassau River. Sample sizes were supplemented using barramundi collected by the Tropical Rivers and Coastal Knowledge research program from March 2008 to January 2011 (Halliday *et al.*, 2012).

Age and increment width measurements were taken from the otoliths (Robins *et al.*, 2021). Otolith age is a combination of increment count, edge classification and an age allocation matrix (Fisheries Queensland, 2012). Otolith increment widths were measured for the first 3 'years' of growth, which captures the period of most rapid somatic growth in barramundi (Stuart & McKillup, 2002). The growth rates in this period are most strongly influenced by availability of freshwater nursery habitat (Roberts *et al.*, 2019) and/or increased food availability from productivity responses (Davies, Bunn & Hamilton, 2008; Pettit *et al.*, 2017). Therefore, inter-annual differences in freshwater availability and subsequent growth rates were expected to be most apparent and influential in these first 3 years of life.

Environmental data

Seasonal flooding is an important driver of barramundi population dynamics, as evidenced in growth rates (Robins *et al.*, 2006), year-class strength (Staunton-Smith *et al.*, 2004; Halliday *et al.*, 2012), exploitable biomass (Tanimoto *et al.*, 2012) and subsequently commercial catch (Robins *et al.*, 2005). The exact mechanisms underpinning this are speculated to be the result of flow-stimulated increases in the biological productivity of estuarine habitats and the connection of ephemeral nursery habitats used by juvenile barramundi (Halliday *et al.*, 2012).

In situ discharge data for the Mitchell River were available at Koolatah (919009A) and Dunbar (919009B) gauging stations (Table 2-1). Modelled EOS flow data were provided by

the CSIRO via FRDC project 2018/079. *In situ* and modelled datasets showed strong agreement (r > 0.82). Therefore, modelled flow data from the EOS node was used for all further analyses on barramundi growth rates in the Mitchell region.

Total river discharge per month was calculated for each year for which concurrent barramundi otolith increment data were also available. River flow was minimal or absent in July, August and September for all study years; those months were therefore excluded from further analyses. Total flow was summed by quarter, with flows in October–November–December representing an early start to the wet season, January–February–March capturing typical wet-season flow, and April–May–June capturing a late end to the wet season.

Regional or continental-scale atmospheric indices appear to influence fish growth rates (Martino *et al.*, 2019) and catch rates (Halliday & Robins, 2001). Therefore, the relationship between barramundi growth rates and two major atmospheric indices, the Southern Oscillation Index (SOI) and the Madden–Julian Oscillation (MJO), was explored as an alternative to river discharge.

The SOI is an indicator of El Niño and La Niña events. El Niño (sustained negative SOI values below −8) is associated with hotter and drier than normal conditions in northern Australia, and La Niña (sustained positive SOI values above 8) is associated with cooler and wetter than usual conditions in the region (Bureau of Meteorology, 2012). Monthly SOI values were accessed via the Australian Bureau of Meteorology (bom.gov.au).

The MJO describes an eastward-moving body of moist air that results in locally increased sub-equatorial cloud cover and rainfall in a 30–60-day recurring cycle (Madden & Julian, 1972). MJO intensity values were available every 5 days from the US National Weather Service (cpc.ncep.noaa.gov/products/precip/CWlink/daily_mjo_index/pentad.html). These MJO values were averaged within a shorter wet-season period (January to March) to provide an index of MJO intensity during the peak of the wet season.

Growth rate modelling

Statistical models were developed for the period 1997 to 2009, the period for which sufficient otolith data (≥10 increment widths per year) were available to test the hypotheses that i) otolith increment widths (as a proxy of growth rates) in juvenile barramundi were influenced by river discharge at key times of the year; and ii) atmospheric indices could be used instead of river discharge variables to model growth rates in young barramundi.

Water development scenario

The water development scenario referred to as 'scenario JBG' (juvenile barramundi growth) involves 6,000 GL total extraction from three different nodes, with a low pump-start threshold (200 ML/day) and zero allocation to EOS flow. Data were obtained from the NAWRA portal (nawra-river.shinyapps.io/river) on 24 October 2019. This scenario was used to assess whether changes in juvenile barramundi growth rates could be detected under a severe water abstraction scenario. To produce juvenile barramundi growth estimates under scenario JBG that are directly comparable to the existing flow scenario, a simulation of barramundi growth rates was constructed for the same time period that was used to produce the growth models (1997–2009).

3.5.3 Finfish otolith microchemistry

It is possible that variability in recruitment (as indexed by year-class strength) is related to hydrologic opportunity, that is, opportunities for barramundi to move (upstream and downstream). The consequences (i.e. effects on the population's biomass) are a function of the fit between the opportunity presented and the optimal requirement for each life stage. Barramundi otoliths were drawn from the DAF Fisheries Monitoring collection, which began in 2000. The selected barramundi otoliths were investigated for strontium isotope ratios ($^{87}Sr/^{86}Sr$); methodology can be found in Robins et al. (2021).

Water samples obtained from freshwater and estuarine reaches of Mitchell River (Table 3-4) were analysed for soluble cations (calcium) and soluble metals (including strontium and barium) by the Queensland Chemistry Centre as per National Association of Testing Authorities accredited analysis. Water samples were further analysed for $^{87}Sr/^{86}Sr$ profile by the Isotope Geoscience Group of the University of Melbourne. Mixing models of $^{87}Sr/^{86}Sr$ across the salinity gradient for the Mitchell River were developed using the approach of Fry (2002), with an upstream freshwater endmember and a downstream estuarine endmember. The $^{87}Sr/^{86}Sr$ profile for each sample was used to classify the habitat as freshwater, estuarine or intermediate. Additional water samples were collected in October 2018 to provide a seasonal replicate and determine how closely the hypothetical mixing curves represented the actual water chemistry in lower habitats. At the same time, barramundi were sampled by line fishing in collaboration with Kowanyama Rangers in October 2018. This was at the end of the dry season, when there had been limited opportunity for barramundi to move due to the lack of river flows within the previous 3 months.

Table 3-4. Site details and chemistry of water samples used to develop conservative mixing curves of $^{87}Sr/^{86}Sr$ to infer salinity and likely habitat residency of barramundi in the Mitchell River.

Site, latitude/longitude (ºS, ºE)	Collection date	Salinity (ppt)[a]	$^{87}Sr/^{86}Sr$	Sr (µg/L)
Gordon Arthur Crossing, 16.30, 143.973	08/09/2017	0.04	0.73158	16.8
Estuary main channel, 15.218, 141.608	09/11/2017	31.00	0.70924	3330.2

[a] salinity derived from measured conductivity (parts per thousand).

The $^{87}Sr/^{86}Sr$ profile for each otolith sample was classified for habitat use depending on the strontium isotope values during slow-growth opaque zones (Roberts *et al.*, 2019), which coincides approximately with the dry season. To be classed as freshwater, $^{87}Sr/^{86}Sr$ ablation values needed to indicate a salinity of ≤1 parts per thousand (ppt) derived from the mixing curve for at least one growth season. To be classed as estuarine, $^{87}Sr/^{86}Sr$ ablation values needed to indicate a salinity of >5 ppt for all opaque zones. Strontium isotope profiles that had $^{87}Sr/^{86}Sr$ ablation values during opaque zones of >1 ppt but ≤5 ppt were classified as intermediate.

Statistical modelling was used to identify whether 'faster' growth, as indexed by length-at-age, was associated with freshwater habitat, and to predict total length as a function of age-class and habitat class (see Robins et al. 2021 for details).

3.6 Freshwater fish assemblage composition

Variables describing the environment, connectivity and fish assemblages throughout the Mitchell catchment were collected to address the following aims:

1. Assess environmental changes across the Mitchell catchment, as well as across habitats and across different connectivity levels
2. Assess which environmental variables influenced species composition, turnover and functional diversity
3. Determine whether species composition and functional diversity varied across connectivity levels
4. Determine whether the relationship between species composition and the environment changed across different levels of connectivity.

3.6.1 Fish collections

The low conductivity of freshwater in comparison to seawater allows an electrical current to be used as an efficient means of sampling fish in freshwater, such that fish that are near to the submerged electrical anode are temporarily stunned (Bohlin *et al.*, 1989). Stunned fish were scooped with a net and placed in an aerated bucket of water, where they were then identified to species level, counted and measured. Some fish were kept for tracer analyses and the remaining fish were returned alive to the water (Figure 3-7, Figure 3-8). A boat was used for electrofishing where possible due to the dangers of estuarine crocodiles. However, backpack electrofishing was used in some upstream sites that were either too shallow for a boat or did not have boat access. Sampling was performed throughout the catchment in river-channel, floodplain creek and floodplain wetland sites during the early dry season in July 2017 and June/July 2018. We define floodplain creeks as creeks off the main river channels that are connected to the main channel during the wet season and disconnected in the dry season. We define floodplain wetlands as bodies of water that are located on the floodplain but may become connected to floodplain creeks or the main channel by overland flows during the wet season. All the floodplain wetlands sampled in this studied were permanent waterbodies.

Figure 3-7. Electrofishing and recording fish species composition at sites on the Mitchell River. Photos: Lyndal Scobell and Glenn McGregor.

Figure 3-8. Some of the fish species caught in the Mitchell catchment. Left column top to bottom: barramundi (Lates calcarifer), Hyrtl's tandan (Neosilurus hyrtlii), glassfish (Ambassis macleayi), tarpon (Megalops cyprinoides), snubnose garfish (Arrhamphus sclerolepis). Right column top to bottom: chequered rainbowfish (Melanotaenia splendida inornata), coal grunter (Hephaestus carbo), freshwater sole (Brachirus selheimi), barred grunter (Amniataba percoides), spangled perch (Leiopotherapon unicolor). Photos: Kaitlyn O'Mara and Lyndal Scobell.

3.6.2 Environmental characteristics

Water physicochemical parameters (oxygen, temperature, turbidity, conductivity, pH) and physical characteristics (depth, reach length) were measured at each site and a habitat assessment was performed. We performed multivariate analyses using environmental variables that were likely to have a direct effect on fish species abundance and distribution. After removing correlated variables, these included average depth, water temperature, turbidity, conductivity and dissolved oxygen (mg/L), and habitat coverage variables (% coverage) for mud, sand, bedrock, small woody debris, large woody debris, bank steepness, undercut bank, aquatic macrophytes and overhanging vegetation.

3.6.3 Fish species composition, turnover and functional diversity

To determine fish species composition (abundance of each species), the catch per unit effort was calculated at each site for each species, by dividing the total number of individuals of each species caught at a site by the total electrofishing shocking time. The species turnover was calculated from Simpson's pairwise dissimilarity of presence/absence fish species data. We created a fish metrics dataset to study how the composition of fish communities across the Mitchell catchment was related to the functional diversity of the species present. We chose metrics that were most likely to influence where a species was found (trophic guild, reproductive movement classification, parental care classification, and spawning substrate and frequency). The classifications for these traits were taken from Sternberg and Kennard (Sternberg & Kennard, 2014).

3.7 River-channel connectivity using an index created from stream-flow gauge data

To further address **Objective 3** of this study, the likely impact of water-resource development on aquatic food webs was assessed separately from the floodplain connectivity analyses (described at Section 3.4) by quantifying the connectivity of riverine reaches in the catchment based on the flow regime.

A site connectivity index (connectivity of a site to the rest of the river network) was created to quantify relationships between flow-mediated connectivity and environmental characteristics, fish species composition and functional diversity, and fish movement (Section 7.2). The index was based on 15 years of modelled daily flows (discharge) data (2000–2015), obtained by Hughes et al. (2017) for nine stream gauges in the Mitchell River catchment. These data were modelled from decades of recorded discharge at each stream gauge and were used instead of historical discharge data because of gaps in the daily records for some of the gauges. A depth of 10 cm was chosen as the minimum water depth required for fish to pass a gauge, and the flow (m^3/s) at 10 cm depth for each gauge was set as the minimum flow required for connection between gauges (DSITIA, 2013).

First, we calculated pairwise proportions of days that flow was above the passable depth at each of the gauges the fish needed to pass to reach the second gauge in the pair. For example, for a fish to have opportunity to move from the upper Palmer River to the upper Mitchell River, this pair of gauges, as well as all the gauges passed on the way, had to have flow rates above the minimum passable depth of 10 cm on a given day to be counted as connected for that day. A value of 1 was assigned for days where gauges were above this minimum flow, and a value of 0 was assigned for days where one or more gauges in the

connection pathway had flow below this minimum flow. Each main-channel site was assigned to the nearest gauge in the same sub-catchment, and because there were more main-channel sites than gauges, some gauges were assigned to two sites. Because connectivity is also influenced by distance and elevation, the pairwise proportions of days sites were connected were tabulated in a matrix and divided by the river distance between sites (km). Second, these pairwise distance-weighted values were divided by the slope between sites (obtained from GIS digital elevation model layers).

Lastly, the final connectivity index value for each site was calculated by dividing the mean site connectivity (average connectivity of a site with all other sites) by the river distance to mouth, to anchor the site connectivity network to the actual river network and ensure that connectivity was not just between sites, but to the rest of the river network. The main Mitchell River mouth was used for this calculation rather than the headwaters of each sub-catchment because the mouth has a more clearly defined location and uses a single distance for each site, rather than multiple distances for each headwater location. Floodplain creek and wetland sites were not included in this analysis since flow at the stream gauges does not accurately represent connectivity to these off-channel habitats. For a more detailed explanation of the connectivity index, refer to O'Mara et al. (O'Mara *et al.*, 2021).

The final connectivity index values were plotted, and visual assessment found two clearly separated groups of sites. This separation was used to define 'high' and 'low' connectivity factor levels, with off-channel sites as a third level. These levels were used to test for differences in environmental characteristics, and fish species composition and functional diversity among connectivity levels.

To define the connectivity of each channel site under the water-resource development scenario, five scenarios of dam infrastructure development were considered. For these scenarios, modelled flow data for a 15-year period (2000–2015) was downloaded from the NAWRA River Model app (nawra-river.shinyapps.io/river) (Hughes *et al.*, 2017). The river model data is a prediction of what historical flows would have been under scenarios selected by the model user, including a change in flows at all sites downstream of a potential dam. Petheram et al. (Petheram *et al.*, 2018) identified five potential dams in the Mitchell catchment (Figure 3-5) and we modelled five unique scenarios based around these:

1. All dams (all five potential dams – three Walsh River dams, one Mitchell River dam [Pinnacles dam], and one Palmer River dam)
2. Three dams only (the three with lowest predicted cost per ML yield) (Petheram *et al.*, 2018): Palmer dam, Pinnacles dam, and one dam on the Walsh River (Chillagoe dam)
3. Chillagoe dam only
4. Palmer dam only
5. Pinnacles dam only.

Sites upstream of dams were also given a pairwise connectivity of 0 to sites downstream of the dam, assuming the dams would be impassable to fish. Therefore, there is both a flow and barrier effect on connectivity under the selected scenarios.

3.8 Tracer analyses

To address **Objective 5**, a variety of tracer analyses were used to infer movement of fish throughout the catchment and the flow of energy from the base of the food web to the fish assemblages (Sections 7.2 and 7.3).

Some of the fish from the 2017 and 2018 electrofishing sampling were humanely euthanised and kept for tracer analyses. Fish were also collected from floodplain habitats in May 2019 by cast netting, baited fish traps and angling. Species kept from each site were chosen based on the fish present at the time of sampling. Because we studied whole fish communities, sampling targeted all habitat types present and effort was allocated to each habitat based on its relative area within the site.

3.8.1 Tracers of movement for freshwater fish

The tracer analyses for freshwater fish involved two approaches: otolith microchemistry, to infer and predict the occurrence of at least one movement of individuals over their entire lifetime; and stable-isotope analyses to infer recent foraging movements. Both approaches relied on defining an isoscape, which is a quantification of the spatial variation in key stable isotopes around the catchment. The isotopic values of individual fish can then be matched against the spatial variation to predict where a fish may have moved, based on their otolith microchemistry and/or the stable-isotope composition.

Stable isotopes are forms of the same element that differ in the number of neutrons in the nucleus (Fry, 2006). Elements can have multiple isotopes, and ratios of the stable isotopes of individual elements (e.g. $^{13}C/^{12}C$) can be used as tracers in ecological studies because the soft tissue of an organism preserves the stable isotopes of their diet (Fry, 2006). Therefore, the stable-isotope ratios of an animal's muscle tissue are indicative of what it has eaten. Depending on the rate of tissue turnover, which varies between taxa and with growth, animals can preserve the isotopes of their diet on a timeline spanning days to months (Barnes & Jennings, 2007). Because adult fish are in a slow-growth phase, they are likely to incorporate dietary stable isotopes from their most recent feeding on a scale of weeks to months (MacAvoy, Macko & Garman, 2001; Barnes & Jennings, 2007).

Similar to stable isotopes in soft tissues, fish integrate strontium isotopes from the surrounding water into their otoliths. As otoliths grow concentrically, variation in the ratios of strontium isotopes ($^{87}Sr/^{86}Sr$) across the otolith reflect the water of their location throughout their lives. The spatial variation that creates an isoscape is generally linked to the underlying geology of the catchment. Five distinct geological units that have been shown to influence $^{87}Sr/^{86}Sr$ isotope ratios are found across the Mitchell River catchment (Batlle-Aguilar *et al*., 2014) – i) Bulimba Formation, ii) Wyaaba Beds, iii) Rolling Downs Group, iv) Gilbert River Formation and v) Hodgkinson Formation – along with a series of igneous intrusives (Figure 2-3). We used the variation in these geological units to guide sampling and analyses of fish tissues to trace movement of fish.

Movements over the lifetime of fish – otolith microchemistry

To predict whether fish had moved at least once throughout their lifetime, we collected water samples for strontium stable-isotope analyses during the 2017, 2018 and 2019 dry seasons. Water samples were collected from 41 sites across the catchment (see all but green sites on Figure 2-3) and mussel samples were collected from 15 sites. Water samples provide an

estimate of the $^{87}Sr/^{86}Sr$ isotope ratio at the time of sampling, and laser ablation analyses of the mussel shells provide an estimate of the temporal variation in the ratio. Together these data were combined in a clustering routine to define an isoscape of $^{87}Sr/^{86}Sr$ isotope ratios.

Sagittal otoliths were dissected from 987 fish collected in the field, cleaned of adhering tissue and stored to dry. One otolith from each fish was embedded in a two-part epoxy resin and sectioned transversely for laser ablation analysis. The concentric growth of otoliths ensures that results of the cross-sectional laser ablation analysis reflect the $^{87}Sr/^{86}Sr$ isotope ratios in the surrounding water from the fish's birth to its death. Using a statistical classification model, we matched the data from the laser ablation analyses to the isoscape to make preliminary predictions as to whether each fish was captured in a different location from its birth. This preliminary approach may result in an underestimate of movement, as fish that migrated away from their birth location and returned are not identified as having moved in this analysis. Further analyses are continuing, and future publications will provide a deeper assessment of the lifetime movements of individual fish.

Recent movements before sampling – sulfur stable-isotope analyses

Sulfur stable-isotope ratios ($^{34}S/^{32}S$, expressed as $\delta^{34}S$) of freshwater vary spatially with changes in underlying geology (Burke *et al.*, 2018), and show little fractionation between food sources and consumers in freshwater (MacAvoy *et al.*, 2000). This means that an organism's muscle tissue will reflect the location where feeding took place. Fish that have recently moved to a new location will therefore be isotopically different if the background $\delta^{34}S$ signal is different from their previous feeding location. We used the spatial differences in $\delta^{34}S$ across the catchment to identify fish that had made recent movements (Section 7.2; refer to O'Mara et al. [2021] for detailed methodology). This analysis aimed to identify wet-season movements of fish communities across different habitats throughout the Mitchell catchment, and examine relationships between movement and flow-mediated connectivity (river-channel connectivity index outlined in Section 3.7 of this report). The regression equation of the relationship between fish movement and connectivity was used to predict changes to movement under five scenarios of dam infrastructure development, where dams act as absolute barriers to movement and also affect connectivity via reduced flows. The five scenarios are described in Section 3.7. The mean pairwise connectivity of the sites under each scenario was fitted to the equation to predict the percentage of migrant fish at each site under each scenario. The change in migrants (i.e. difference in percentage of migrants between the no-dams scenario and each of the other scenarios) was then calculated to assess how each scenario would impact movement at various locations throughout the catchment. Assumptions of these predictive models are outlined in O'Mara et al. (O'Mara *et al.*, 2021).

3.8.2   *Tracers of dietary energy flow*

Fatty acids are effective tracers of dietary pathways because they are released from ingested lipid molecules during digestion of food without becoming degraded (Iverson, 2009). They are then taken up and preserved in storage fats (such as in muscle tissue) and can be traced back to food web origins (Iverson, 2009). Polyunsaturated fatty acids (PUFAs) are essential components of cell membranes and reproductive and sensory organs in vertebrates and are therefore essential in their diets (Murray *et al.*, 2014; Roy *et al.*, 2020). However, little is known about the fatty-acid composition of aquatic food webs and dietary PUFA sources for fish in northern Australia.

The flow of dietary energy through food webs and across habitats can be assessed by the lipids and fatty acids in food sources and consumers (Kainz, Arts & Mazumder, 2004; Arts, Brett & Kainz, 2009; Guo *et al.*, 2017). Changes in lipids and fatty-acid content from food sources to consumers at different trophic levels can indicate how dietary energy produced by food sources at the base of the food web (plants, bacteria and fungi) is transferred and retained in aquatic consumers (Arts *et al.*, 2009). We collected fish, macroinvertebrate (aquatic and terrestrial) and plant (aquatic and terrestrial) taxa from floodplain and river-channel habitats and analysed taxa at each site for total lipids and fatty acids to address two aims (Section 7.3). Firstly, we aimed to explore the flow of dietary energy through food webs and identify high-quality food sources in terms of PUFA content. To address this aim, the fatty-acid composition of food web groups was examined using multivariate statistics to identify diet pathways, and the ratio of individual fatty acids between trophic levels was examined to identify high-quality food sources. Secondly, we aimed to determine if there were differences in food quality across macrohabitats and assess whether there were particular habitats that were more important for food quality than others. Univariate analyses of variance tests were used to address this aim.

4. Mitchell River floodplain inundation, productivity and connectivity and the implications of development

4.1 Floodplain inundation and plant biomass accumulation

4.1.1 Floodplain inundation and river-channel connectivity

Satellite imagery showed a high level of inter-annual variation in open-water features (i.e. floodplain wetlands) during the wet seasons between 2007 and 2014. Comparing the average inundation extent over the entire wet season, the largest wet season was in 2008 (average inundation extent for the season of 544 km^2) followed by 2009 (259 km^2), in comparison to less than 100 km^2 in 2007, 2013 and 2020 (Figure 4-1). The maximum areas of inundation in each season are much greater than these numbers; however, averages were used to establish the relationship between inundation and river flows. Average inundation was positively related to discharge, with years with higher wet season flows being characterised by larger surface inundation extent. In contrast to the wet season, the estimated floodplain inundation extent for the dry season remained relatively stable between years, with average dry-season inundation extent fluctuating between 55 km^2 and 80 km^2 (Figure 4-1).

	Yr-2007	Yr-2008	Yr-2009	Yr-2010	Yr-2011	Yr-2012	Yr-2013	Yr-2014	Yr-2015	Yr-2016	Yr-2017	Yr-2018	Yr-2019	Yr-2020
Wet months (Dec - Mar)	63	544	259	106		144	64	124		126	159		118	60
Dry months (Jun - Aug)	73	60	58	65	61	70	59	71	56	80	55	62	65	60

Water year

■ Wet months (Dec - Mar)   ■ Dry months (Jun - Aug)
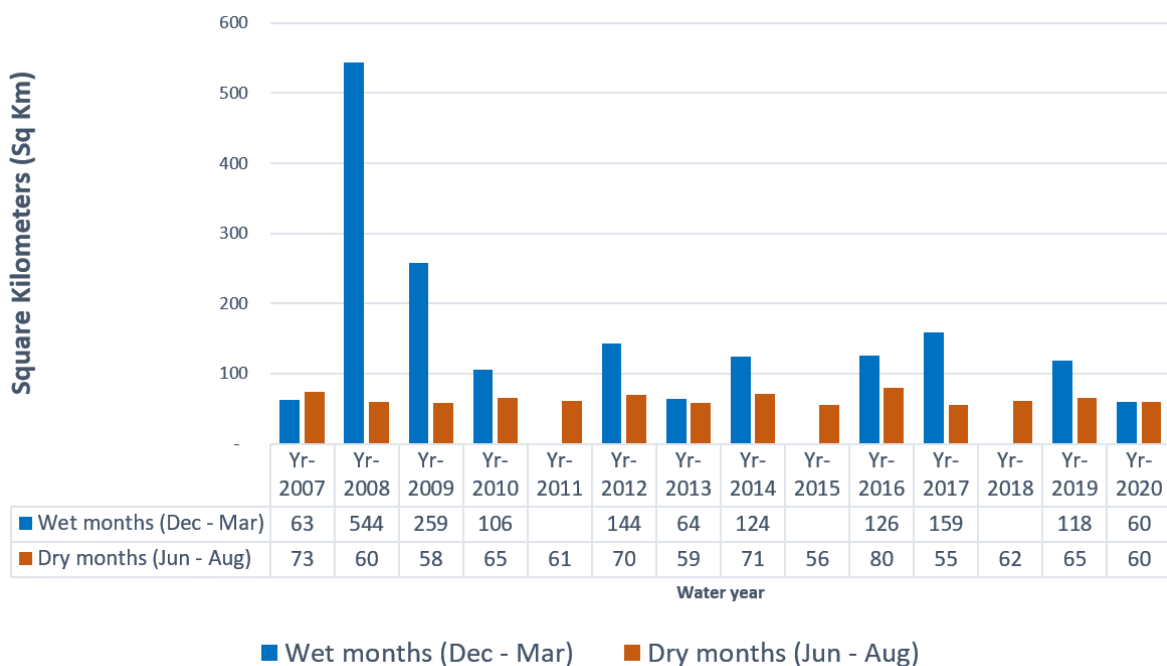
Figure 4-1. Temporal variation in open-water spatial extent, for wet seasons and dry seasons, 2007–2020.

Temporal inundation pattern analysis showed that in the wet seasons, most waterholes were inundated, and rivers remained well connected. As the dry season approached, while most waterholes remained inundated, some sections along major rivers and channels became disconnected. Some mid-sections of river channels remained disconnected in both wet and dry seasons during years with reduced annual rainfall (e.g. 2007). Additionally, extreme floods in the wet season did not guarantee river channels remained connected in the

following dry season of the same year. For example, in the wet year of 2009, many sections of the river channel became disconnected during the dry season despite the large inundation extent during the previous wet season (Figure 4-2).
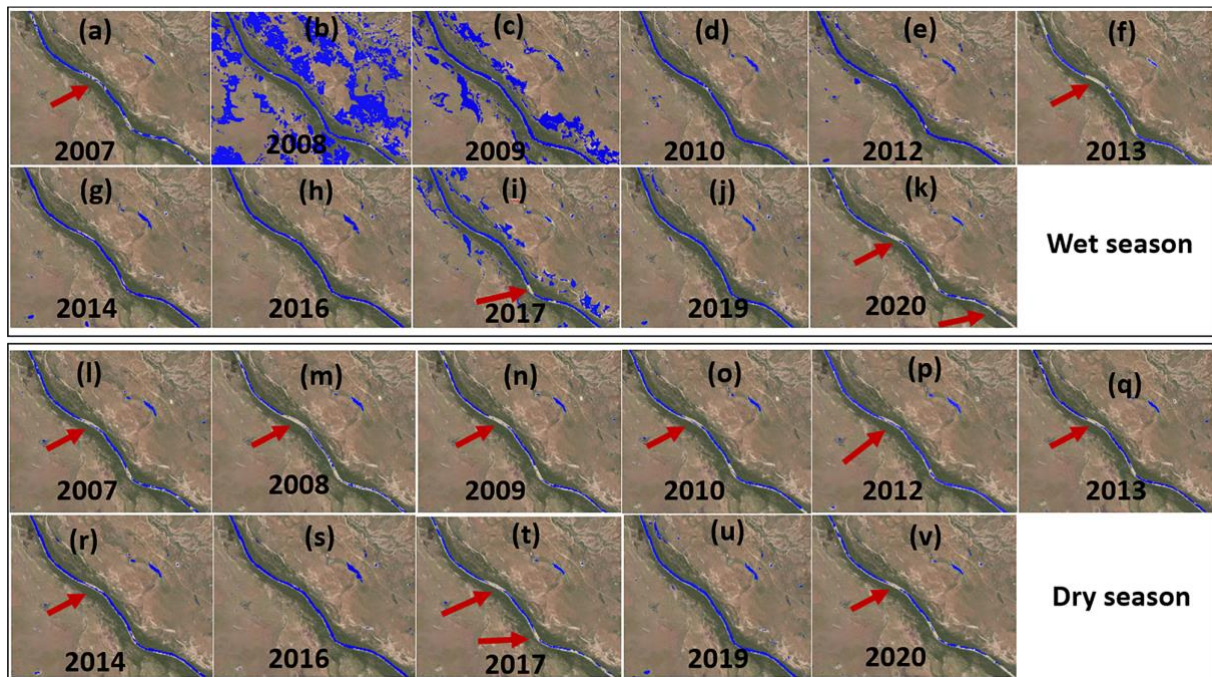
Figure 4-2. Inundation patterns of riverine systems and waterholes during wet (a–k) and dry (l–v) seasons between 2007 and 2020. Wet-season inundation for each year is based on the median of Landsat images for the wet season (December–March); dry-season inundation is based on the median of images between June and August of the same year. The red arrows show disconnection along the mid-stream Mitchell River. Figure reproduced from Ndehedehe et al. (2021).

4.1.2 Aquatic plant biomass accumulation

The distribution of aquatic plant biomass accumulation in the floodplain region followed the seasonal climatic variability between 2016 and 2020 (Ndehedehe *et al.*, 2021). The extent of aquatic plant biomass was shown to have a relatively strong relationship with wet-season river flow. The years with relatively higher rainfall and flow also had higher aquatic plant biomass. For example, the largest extent of aquatic plant biomass on the floodplain was found in 2019 (36.9 km^2), which also had the highest average upstream flows (Figure 4-3).
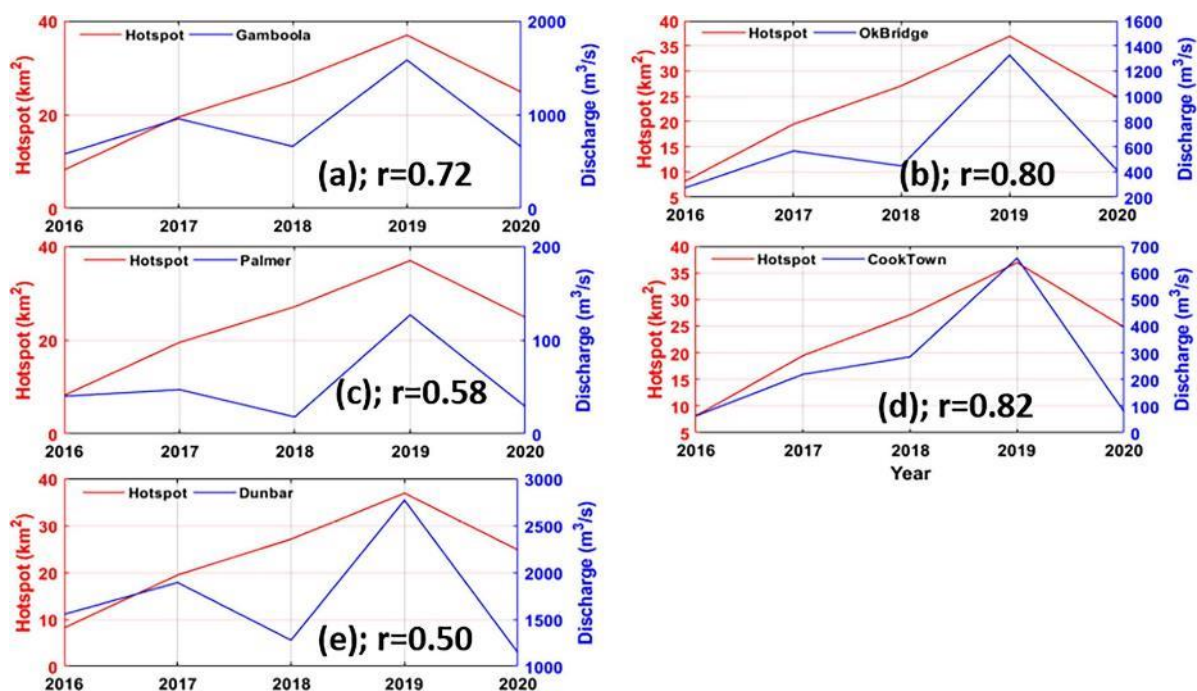
Figure 4-3. Floodplain aquatic plant biomass extent (2016–2020) over the downstream Mitchell catchment, shown as 'hotspot' (km². The relationship between predicted extents of hotspots and observed discharge is indicated for (a) Gamboola, (b) O.K. Bridge, (c) Palmer, (d) Cooktown Crossing and (e) Dunbar. Figure reproduced from Ndehedehe et al. (2021).

4.1.3 Drivers of inundation and aquatic plant biomass accumulation

The results highlighted that inundation and plant biomass accumulation on the Mitchell River floodplain were correlated to upstream flows and local rainfall (Ndehedehe et al. 2021). Observed discharge at the Dunbar gauging station (the most downstream river gauge) explained approximately 60% of the proportion of variability in total floodplain inundation extent, with local rainfall on the floodplain accounting for 25% of this variability. Similarly, plant biomass accumulation was correlated with discharge (though quantified over a shorter period due to the available satellite time series), with the strongest correlation found with the observed discharge at Cooktown Crossing gauging station (Figure 4-3d).

4.2   Floodplain algal productivity

At the end of the 2017–2018 wet season, observed algal productivity ranged between 0.18 and 1013.98 mg C/m^2/day. The rates of algal productivity varied by habitat type and were higher among submerged habitats, with an average of 336 mg C/m^2/day (Table 4-1).

Table 4-1. Average algal productivity per habitat (emergent, floating, submerged and open water) and their standard deviations (SD). n is the number of samples per habitat.

Habitat type	n	Average algal productivity (C/m^2/d)	SD algal productivity (C/m^2/d)
Emergent	24	121.28	157.06
Floating	18	88.54	106.52
Submerged	14	335.66	319.89
Open water	7	112.96	62.83

The statistical analysis showed that habitat type and turbidity can be used as local predictors of algal productivity in the Mitchell River floodplain wetlands and rivers. The submerged habitats were shown to support significantly higher algal primary productivity than the other macrophyte structures (emergent and floating) and open water. Submerged macrophytes usually have complex leaf and branching structures, providing greater surface area for algal colonisation and supporting higher algal productivity (Molinari et al. 2021a; Ward et al. 2016). In contrast, the above-water leaves of dense emergent and floating plants reduce the available light and consequently algal productivity (Adame et al. 2017; Hinojosa-Garro, Mason, and Underwood 2010; Molinari et al. 2021a). Somewhat surprisingly, turbidity was positively related to algal productivity. As the physical effect of turbidity would normally reduce light penetration into the water column, it may be nutrients that are the underlying driver and turbidity is acting as a surrogate variable, given that more turbid water may carry higher concentrations of nutrients.

4.2.1   Algal productivity spatial distribution

Using Landsat 8 imagery, the predictions for algal productivity across the Mitchell River floodplain ranged from 3.2 to 859.2 mg C/m^2/day, with an average of 151 mg C/m^2/day (Figure 4-4). The total daily algal productivity across the whole floodplain area was estimated at 85 t C/day. The higher predictions of algal productivity were located on secondary river channels, small wetlands and littoral areas. In total, 94% of the floodplain algal productivity occurred in those areas rather than in the areas of the main channels of the Mitchell, Alice, Scrutton, Nassau and Palmer rivers contained within the floodplain area. Predicting spatial variation of algal productivity allowed the identification of key productive habitats that are important sources of food for aquatic consumers.
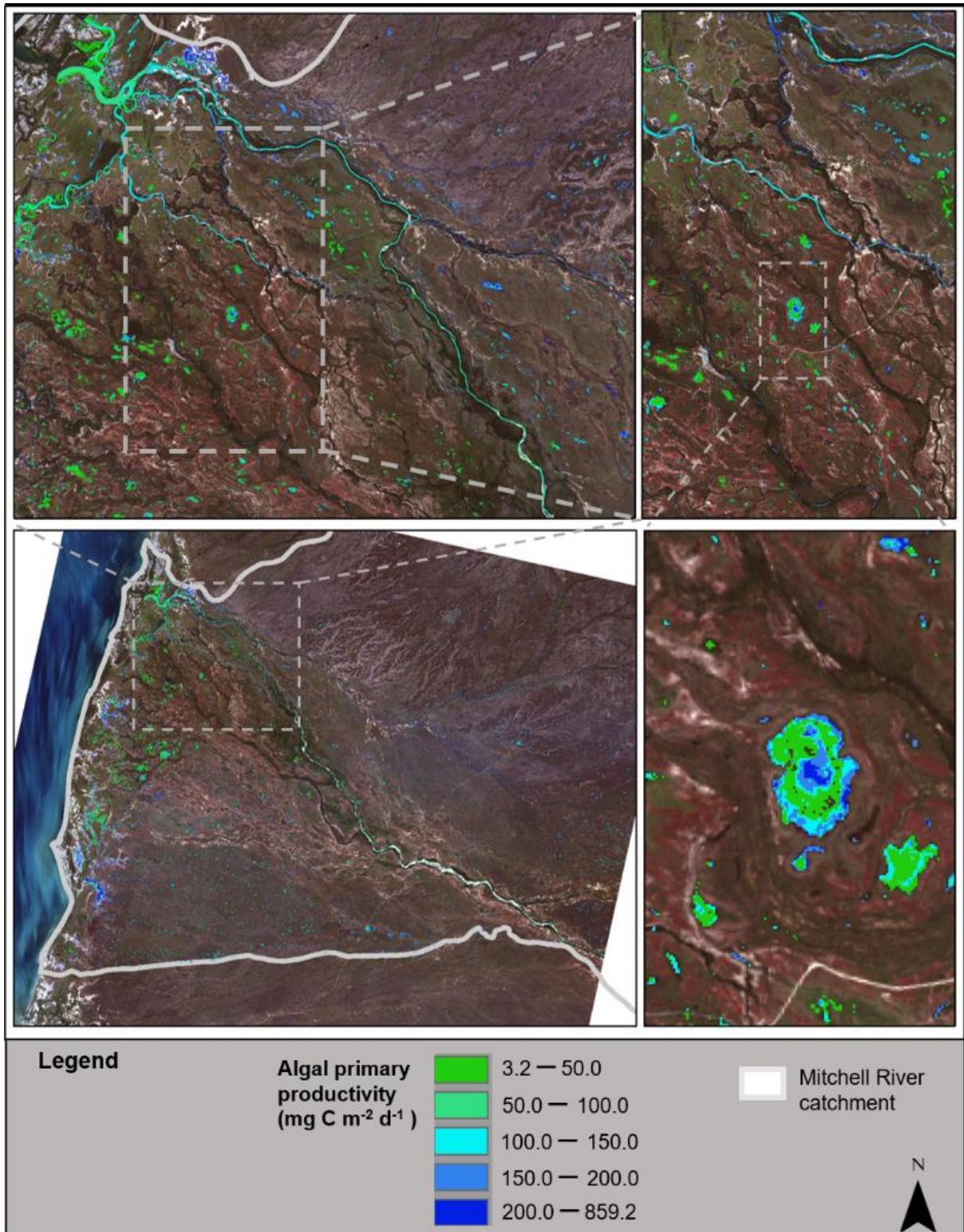
Figure 4-4. Spatial variation of predicted algal primary productivity (mg C/m²/day) across different extents. The background is the Landsat 8 image. The bottom-right figure corresponds to a small fraction of the floodplain consisting of Racecourse Swamp.

4.3 Implications of development

Climate change and anthropogenic hydrological alteration are expected to alter the dynamics of the natural flow regime in large rivers around the world (Poff, 2018). In Section 4.1, upstream flows were shown to be the primary driver of floodplain inundation and to play an important role in supporting aquatic plant mass accumulation on the floodplain. Consequently, reduction of river flows is expected to lead to substantial loss in total surface inundation extent with subsequent loss of plant biomass (Thapa, Thoms & Parsons, 2016; Ndehedehe *et al.*, 2021).

Aquatic plants provide habitat for floodplain fauna including invertebrates and small fish, and while the plants do not contribute substantially to the aquatic food webs, they support algal productivity (Molinari et al. 2021a), which constitutes most of the secondary productivity on floodplains (Pettit *et al.*, 2011). The reduction of plant biomass as a result of an altered flow regime may have a subsequent impact on ecosystem functioning.

In addition to flow alteration, agriculture expansion has the potential to alter water quality (Osteen, Gottlieb & Vasavada, 2012; Kingsford, 2015). Molinari et al. (2021a) showed that Mitchell River wetlands are in a clear-water state under current conditions, where primary productivity and nutrients are relatively balanced. Many locations in the catchment, including on the floodplain, are phosphorus-limited (Appendix A). However, changes in water quality and an increase in bioavailable phosphorus levels caused by agricultural run-off may trigger a shift to an alternative state, characterised by an excess of nutrients, turbidity and/or phytoplankton-dominated waters (Scheffer *et al.*, 2001). Such a shift could lead to eutrophication, harmful algal blooms, and ultimately oxygen depletion and fish kills (Ansari *et al.*, 2011).

5. Connectivity of the Mitchell River floodplain

5.1 Connectivity modelling of wetlands and primary productivity

The connectivity modelling of the Mitchell River floodplain was implemented using the floodplain inundation data from the CSIRO NAWRA hydrodynamic model (Section 3.4). Floodplain inundation extent displayed substantial variability between the wet (2009), average (2001) and dry (2006) years. The floodplain inundation on the wet year (2009) showed a larger surface-water extent, spreading to areas further from river channels, compared with the average and dry years (Figure 5-1). Under the WRD scenario, there was a large reduction in inundation extent for the three years, ranging between 27% and 41% (Figure 5-1, Table 5-1).
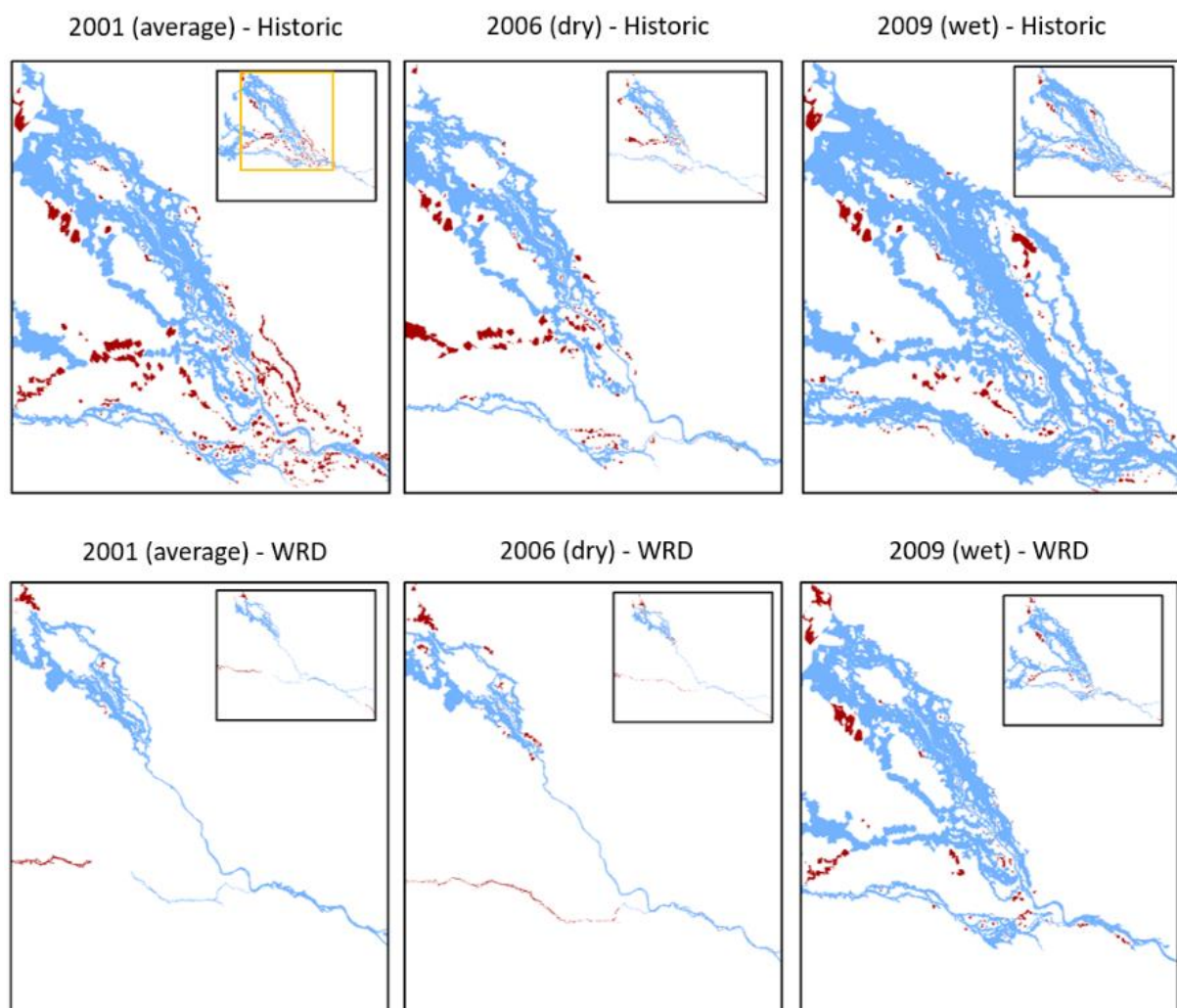
Figure 5-1. Illustration of floodplain inundation on the peak inundation day of each year (05/03/2001, 28/04/2006 and 12/02/2009) for the historical and WRD (water-resource development) scenarios. Blue shading represents pixels that are inundated and connected to the Mitchell River; red shading represents pixels that are inundated and not connected to the Mitchell River. The insets in the top right of each panel show the extent of inundation across the whole floodplain; the yellow box represents the extent to which the images were zoomed. Reproduced from Molinari et al. (2021c).

Table 5-1. Graph theoretic connectivity metrics and landscape productivity metrics on the day of peak inundation for each modelling period – day 19 (05/03/2001), day 19 (28/04/2006) and day 12 (12/02/2009) – for the historical and WRD (water-resource development) scenarios. The reduction due to WRD is shown in the columns of percentage comparisons.

		2001 scenario			2006 scenario			2009 scenario		
		Historical	WRD	100−%WRD/Historical	Historical	WRD	100−%WRD/Historical	Historical	WRD	100−%WRD/Historical
Graph metrics	Graph edges	1.9E+07	9.3E+06	50	1.0E+07	3.9E+06	62	3.0E+07	1.8E+07	41
	Number of components	3.6E+07	3.7E+07	−3	3.7E+07	3.8E+07	−2	3.5E+07	3.6E+07	−5
	Maximum component size	2.1E+06	8.6E+05	59	1.1E+06	4.2E+05	62	3.7E+06	2.2E+06	41
Landscape productivity metrics	Inundated habitats	2.4E+05	1.4E+05	41	1.4E+05	9.8E+04	31	3.1E+05	2.3E+05	27
	Connected habitats	2.2E+05	1.2E+05	48	1.2E+05	6.9E+04	42	3.0E+05	2.2E+05	28
	Inundated productivity (tonnes C)	32.3	19.1	41	18.2	13.2	28	43.9	29.8	32
	Connected productivity (tonnes C)	30.0	16.0	47	15.7	8.8	44	42.3	28.4	33

Consistent with the floodplain inundation extent, connectivity was highest in the wet year (2009), with the spatial graph having a consistently higher number of graph edges than the other two years. On the peak inundation day of 2009, the number of edges (~30 million edges) was three times higher than on the peak inundation day of 2006 (~10 million edges), indicating much greater potential for dispersal of aquatic organisms (Figure 5-2). A consistent reduction in the connectivity metrics were found under the WRD scenario, which was most evident in the drier years (Table 5-1). For example, at the peak inundation day of the dry year (2006), the number of edges were reduced by 62% compared to a reduction of 41% in 2009 under the same scenario (Figure 5-2). The number of components was higher for the dry year, indicating a higher level of landscape fragmentation, which was aggravated by water-resource development (Figure 5-2). The size of the maximum component was larger on the wet and average years, suggesting extended areas accessible to aquatic species from the Mitchell River. Collectively, the graph theoretic metrics indicate that the reduction of floodplain inundation, as a consequence of drier climates or water-resource development, translates into reduced connectivity, which ecologically means reduced opportunities for aquatic organism dispersal and reduced connection among wetlands, small river channels and the mainstem of the Mitchell River.
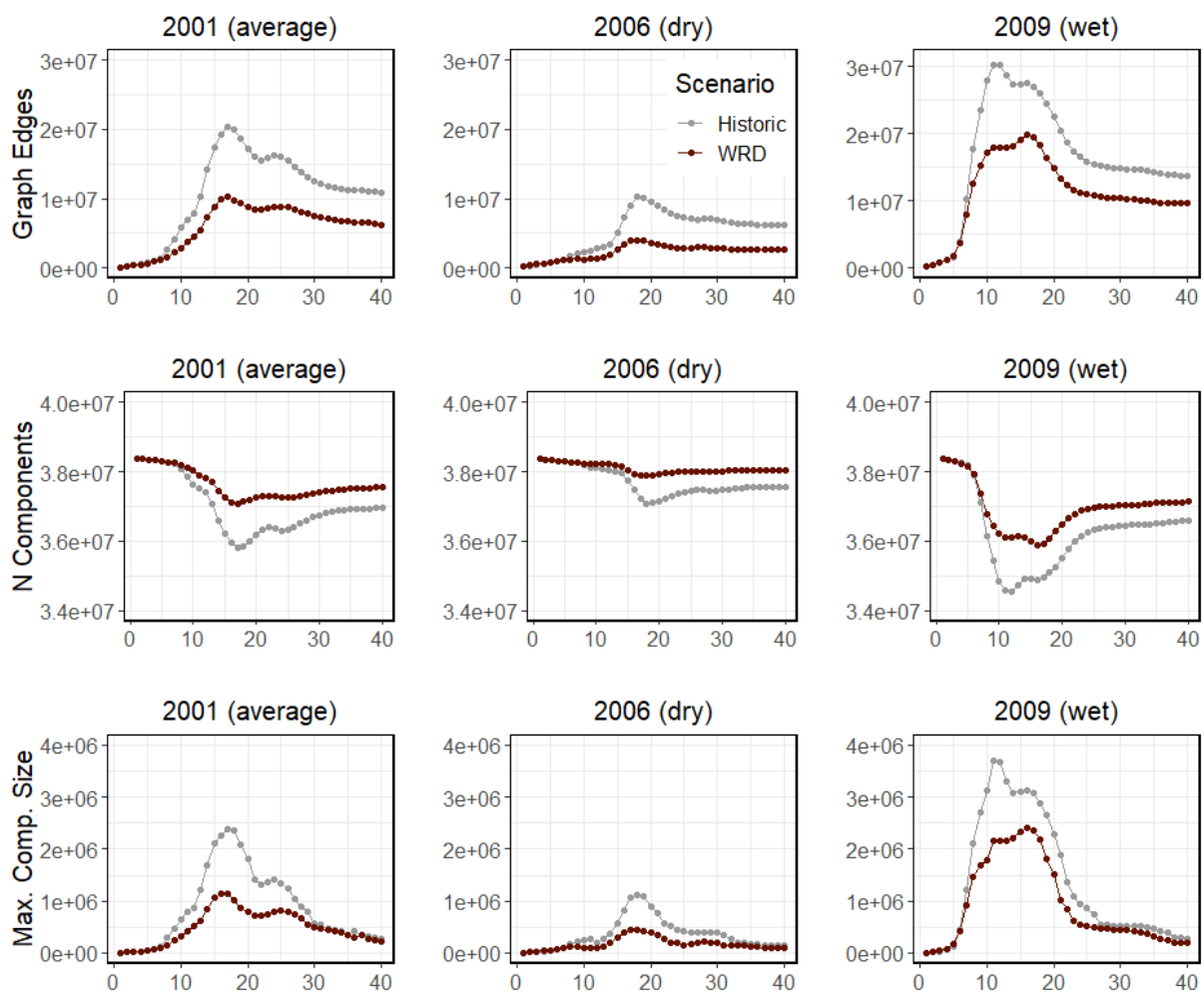
Figure 5-2. Comparison of graph theoretic connectivity metrics for the modelled 40-day period for three years (2001, 2006 and 2009). Grey is the historical scenario; red is the WRD (water-resource development) scenario. The x-axis shows the days over the 40-day inundation period considered for each year.

Changes in floodplain inundation were also shown to impact the inundation and connectivity of productive habitats on the floodplain. At the peak inundation day, the inundated and connected productivity of algae was markedly larger in 2009 – 43.9 and 42.3 tonnes of carbon (t C) respectively – compared with 18.2 and 15.7 t C at the peak inundation day of the dry year of 2006 (Table 5-1, Figure 5-3). The inundated productivity slowly declined with the receding flood, while connected productivity rapidly declined, and in the average and dry years, the peak of productivity was shown to be connected to river channels for shorter durations than in the wet year of 2009 (Figure 5-3). For the wet year of 2009, the drops in inundated productivity and connected productivity under the WRD scenario were similar (32% and 33%, respectively). However, for the dry year of 2006, the loss of productivity as the habitats were no longer inundated was 28% and the loss due to impaired connectivity was 44%.
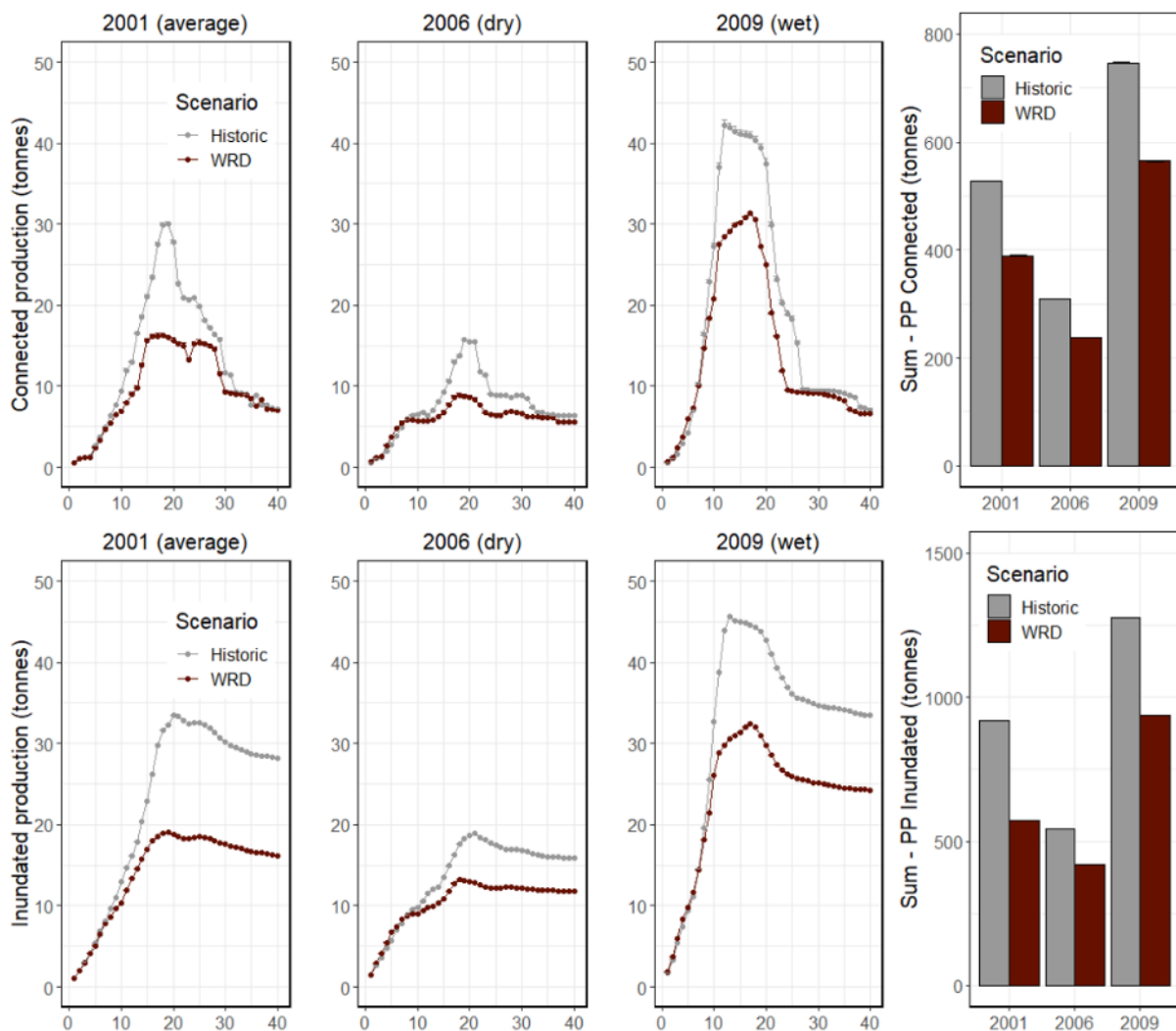
Figure 5-3. Connected and inundated algal productivity for the modelled 40-day period for three years (2001, 2006, and 2009). Grey bars and lines are the historical scenario; red are the WRD (water-resource development) scenario. Prediction intervals are shown as bars for each observation. PP =primary productivity.

The reductions in the extent of inundation were shown to impact the amount of algal productivity connected to the Mitchell River. Both drier wet periods and the WRD scenario were shown to reduce the input of primary productivity, and the combination of both is likely to be substantial. At the peak inundation day of 2009, the connected productivity under the historical scenario was estimated as 42.3 t C. At the peak inundation day of the dry year of 2006, the connected productivity under the WRD scenario was 8.8 t C. This implies that reduced floodplain inundation from a drier wet season and a WRD scenario combined can be translated into a loss of 79% of connected algal productivity that is no longer available to in-stream ecosystems when inundation is at its highest.

Similarly, for the entire 40-day modelled period, flow alteration had a significant impact on total inundated and connected algal productivity. Under the historical scenario, the total inundated and connected productivity for the 40-day period in the wet year was 1,276 and 746 t C, respectively, compared with 543 and 309 t C in the dry year. Under the WRD scenario, the inundated and connected productivity were reduced in both years by almost a quarter. The 23% reduction in the dry year means a loss of 71 t C connected to the main Mitchell River channel. In the wet year, the connected productivity was reduced by 24%, which means a loss of 182 t C input to in-stream food webs. These figures represent substantial losses of energy at the base of the food web that will have consequences for higher order consumers, such as barramundi and other predatory fish.

5.2 Fish movement to floodplains

Based on the sulfur stable-isotope analyses (Section 3.8), migrant fish were found in three out of five wetlands where more than five large fish were collected, and all wetlands containing migrants were situated on the floodplain (Figure 5-4). Yelko Waterhole was the only floodplain wetland visited during two sampling trips, and had migrant fish on both occasions. In 2018, 30% of large fish (three out of 10) analysed for stable isotopes were migrants, including *Leiopotherapon unicolor*, *Scortum ogilbyi* and *Oxyeleotris lineolatus.* In 2019, three large fish (all *Megalops cyprinoides*) were collected from Yelko Waterhole, two of which were recent migrants. Leichardt Waterhole was sampled in 2018 and one out of 11 large fish (*Lates calcarifer*) was a recent migrant. White Water wetland was sampled in 2019 and one out of six fish was found to be a migrant (*Neoarius graeffei*).

These results show that there was evidence for recent movement to most of the wetlands sampled, even though sample numbers for large fish were low in some wetlands. Further, migrants included catadromous species (*Megalops cyprinoides* and *Lates calcarifer*), potadromous species (*Leiopotherapon unicolor* and *Scortum ogilbyi*), and species that do not make reproductive migrations (*Neoarius graeffei*), suggesting that movement to wetlands is not influenced by reproductive movement traits and is likely related to foraging. This study only classified fish that recently moved into wetlands, but it is likely that fish also use lateral connectivity to move out of wetlands.
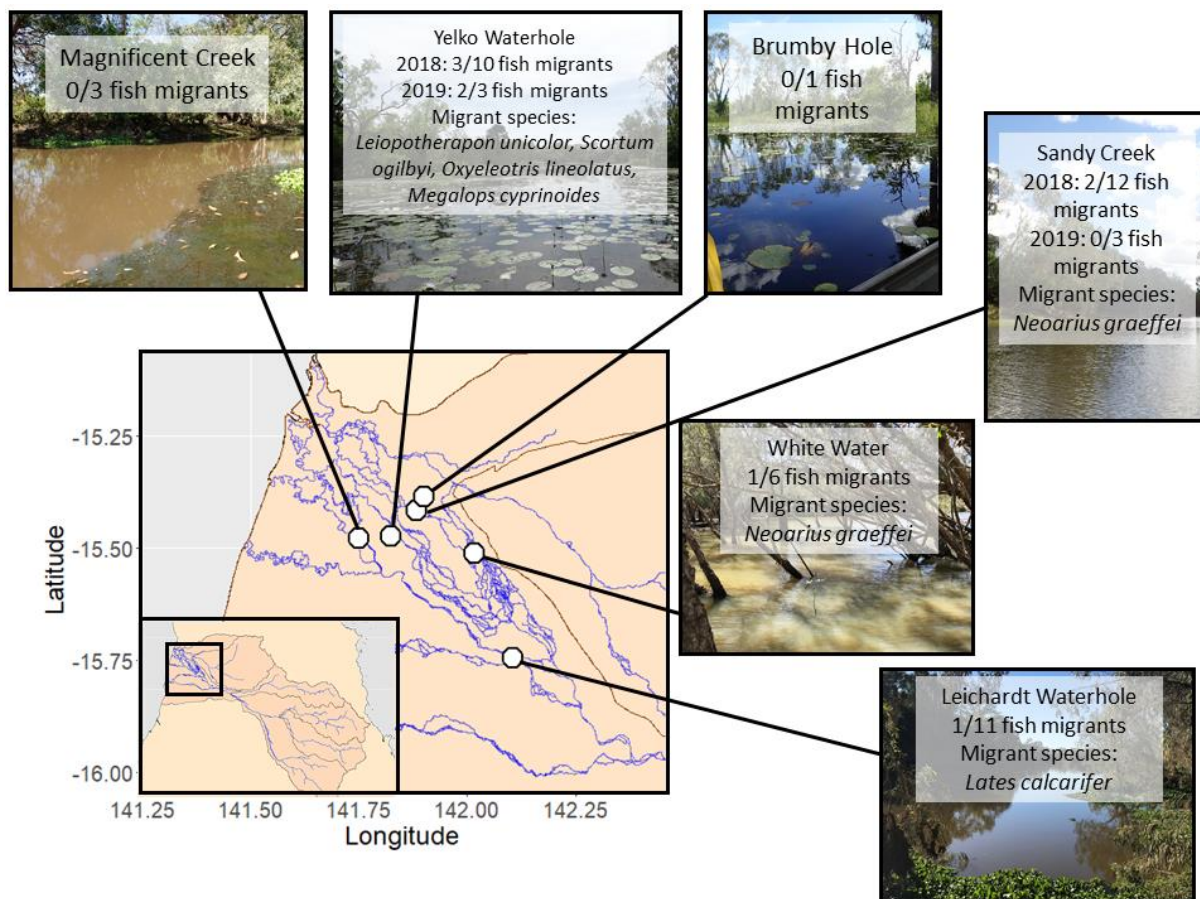
Figure 5-4. Resident and migrant large fish (>10 cm long) in floodplain creeks and wetlands, determined from analysis of sulfur stable isotopes.

6. Fish growth and recruitment

6.1 Catch and age

6.1.1 Catch

Regional commercial catch and nominal catch rate (kg/days fished) for barramundi in the region influenced by the Mitchell River (mid-region of the southern Gulf of Carpentaria) has been variable over time, with the lowest total catch reported in 2015 (~150 t) and the lowest catch rate in 2005 (~60 kg/days fished). The highest total catch reported was in 2008 (>300 t), and the highest nominal catch rate was in 2012 (~120 kg/days fished) (Figure 6-1; Robins et al. 2021).
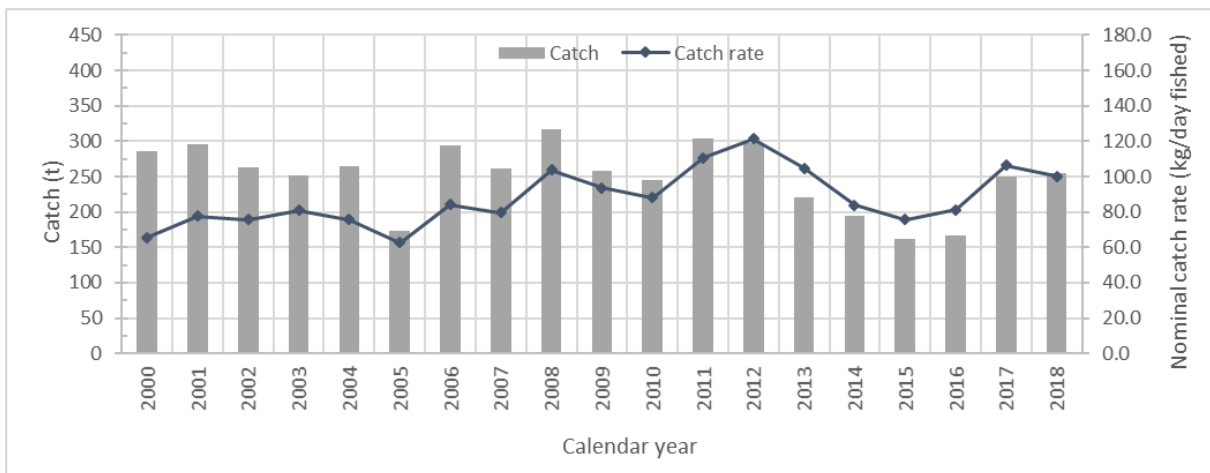
Figure 6-1. Reported commercial harvest and catch rate for barramundi from the mid-region stock. Figure reproduced from Robins et al. (2021).

6.1.2 Age-frequency and catch-at-age

Barramundi from the mid-region of the southern Gulf of Carpentaria stock displayed highly variable length-at-age with a wide spread of total length for any given age-class, indicative of individual growth variability (Figure 6-2). Similarly, there was considerable variation between years in the age-frequencies of barramundi, suggesting variable recruitment between year-classes (Figure B-1, Figure B-2).
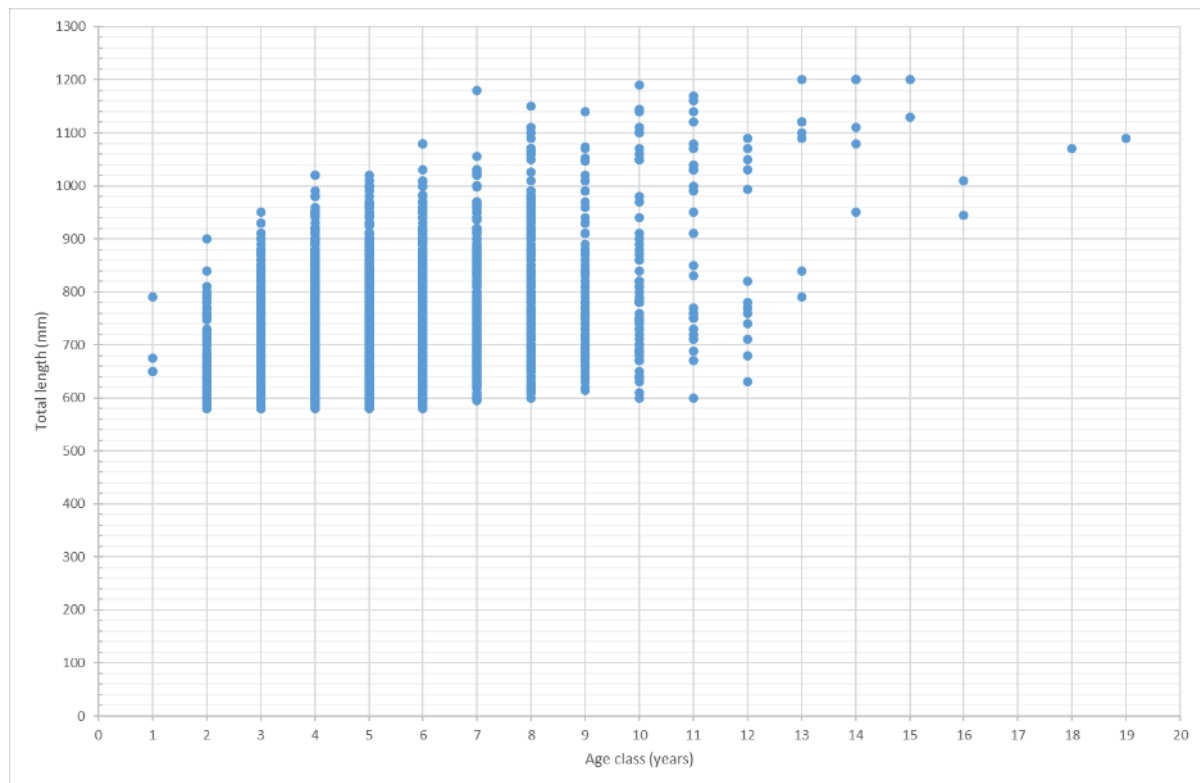
Figure 6-2. Observed length-at-age for legal size barramundi sampled by Department of Agriculture and Fisheries (Queensland) from the mid-region stock between 2000 and 2018 (n=4,140). Figure reproduced from Robins et al. (2021).

Catch-at-age of the commercial harvest was variable. Most of the harvest was dominated by 4- to 6-year-old fish, but in some years (e.g. 2008 and 2014), younger fish were important in the harvest while in other years (e.g. 2018), older fish were important in the harvest (Figure B-2). The statistical model that best explained the variation in catch across the age-classes (3–9 years) included the quarterly flows in the year of birth, as well as quarterly flows in the second year of life and flows from the third birthday to capture (adjusted-R^2 = 0.69). Including quarterly flows from the second and third years of life, as well as those from the capture year, significantly improved the fit compared to the model that used quarterly flows only in the first year of life (i.e. the birth-year flow; see Robins et al. 2021).

6.2 Growth

6.2.1 Drivers of otolith increment width

Higher river discharge volumes increased juvenile barramundi growth rates in the Mitchell River region in each of the three wet-season periods (October–November–December, January–February–March and April–May–June; conditional-R^2 = 0.90; Figure 6-3). The results indicated that older juveniles (12 to 36 months old) benefit more from high flow volumes at the peak of the wet season (January–February–March) (Figure 6-3b), while younger juveniles (0 to 24 months) benefit more from high flow volumes at the end of the wet season (April–May–June) (Figure 6-3c).

In this study, we also quantified the relationship between major atmospheric indices and increment widths of juvenile barramundi. The results indicate that both MJO and SOI have strong positive effects on juvenile barramundi growth rates in the Mitchell region (conditional-R^2 = 0.90). The positive effect of MJO on juvenile barramundi growth rates in the Mitchell region at all increments (Figure 6-3d) could be explained by the increased prey availability triggered by pulse flood events. The largest otolith increment widths were seen during La Niña (strongly positive SOI) resulting in higher growth rates for all measured ages of juvenile barramundi (Figure 6-3e). The speculated mechanistic link between SOI and growth rates of estuarine/marine taxa is that increased freshwater flows contribute to the movement of nutrients and inundation of wetland areas, stimulating high levels of primary productivity (Davies *et al.*, 2008). This is supported by the results in Section 4 of this report, which showed that, for the Mitchell River, floodplain inundation, productivity and connectivity were demonstrated to be greater during years of higher flow than during years of lower flow.
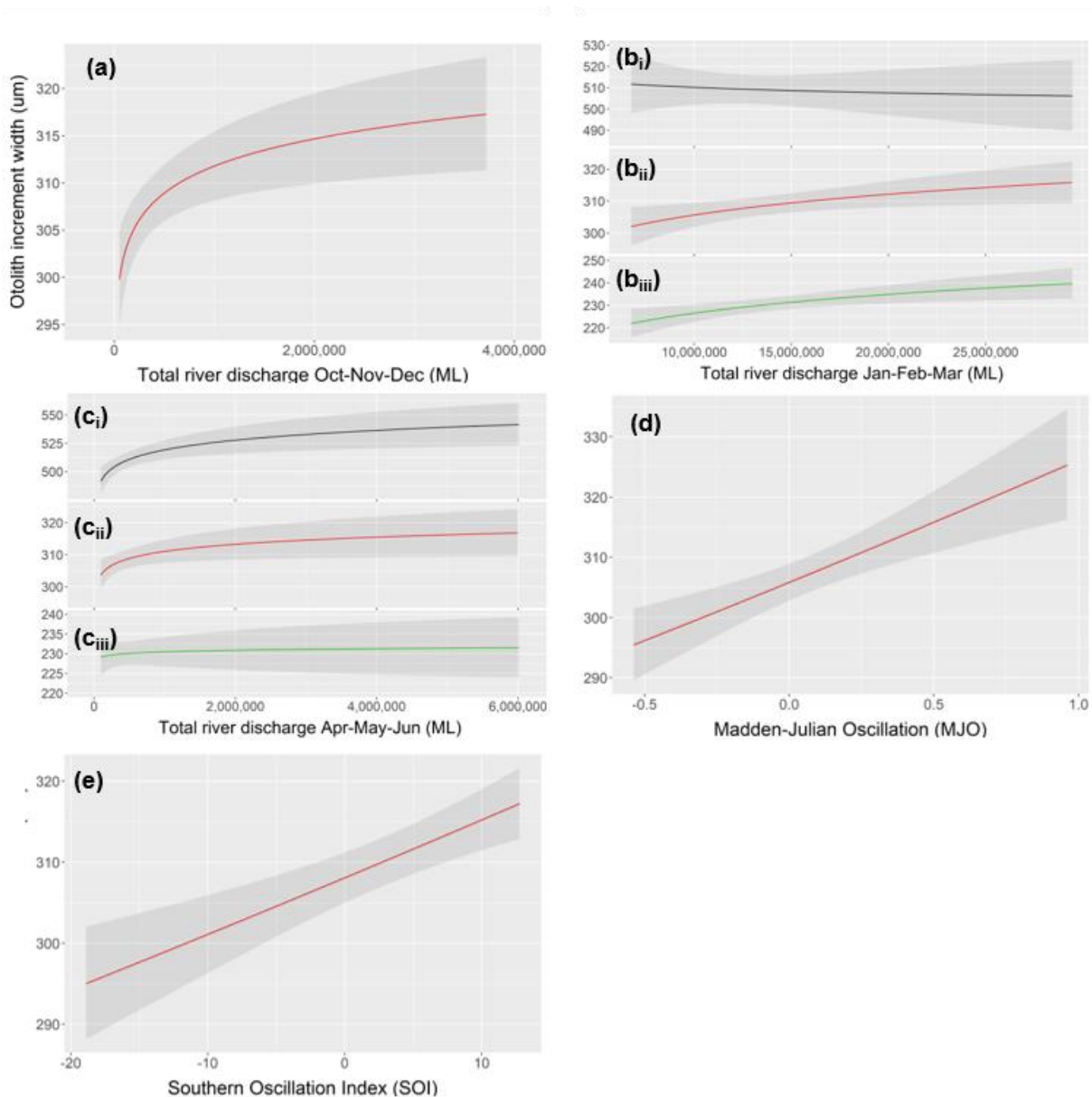
Figure 6-3. Predicted (±95% confidence intervals) otolith increment width as a function of flow and atmospheric variables, derived using the single top model. Note difference in y-axis range between panels. Black line: increment 1; red line: increment 2; green line: increment 3. Where significant interactions with increment were not present, predicted otolith increment widths are illustrated for increment 2. Figure reproduced from Robins et al. (2021).

6.2.1 Water-resource development scenario

Juvenile barramundi growth rates in the Mitchell region were distinctly reduced under flow scenario JBG compared with existing flow conditions. Otolith increment widths were reduced by 14.2–28.2% under this scenario (Table 6-1), resulting in a 19% reduction in cumulative otolith width between the otolith core and third increment. This means that age 2+ barramundi would be expected to be, on average, about 19% smaller (in total length) under flow scenario JBG compared with age 2+ barramundi under existing flow conditions. Given the importance of size-selective survival in young fish (Perez & Munch, 2010), reduced growth rates may have important consequences for cohort survival, with consequent deleterious effects on year-class strength (Halliday *et al.*, 2010), biomass (Tanimoto *et al.*, 2012) and catch rates (Robins *et al.*, 2005; Tanimoto *et al.*, 2012). Water planning decisions

in this region should be mindful of these consequences and seek to weigh potential benefits of water harvesting and storage against the downstream impacts of reduced freshwater flows on this highly productive and valuable system.

Table 6-1. Predicted changes in juvenile barramundi growth rates (otolith increment widths) under scenario JBG. Table reproduced from Robins et al. (2021).

	Percent difference in increment width	Percent difference in cumulative otolith width
Increment 1	−14.2%	−14.2%
Increment 2	−23.1%	−17.6%
Increment 3	−28.2%	−19.3%

6.3 Otolith microchemistry

6.3.1 Habitat use inferred from $^{87}Sr/^{86}Sr$ profile

There were slight differences in habitat residency between barramundi from different year-classes from the Mitchell River (Table 6-2). Overall, 63% of barramundi caught in the estuary of the Mitchell River had resided in freshwater (salinity ≤1 ppt) for at least one dry season, while 10% had resided only in estuarine water (salinity >5 ppt) (Table 6-2). The remainder (27%) were classified as 'intermediate'. The $^{87}Sr/^{86}Sr$ profiles of intermediate-classed fish were commonly cyclical, with salinity values >1 ppt and <5 ppt during the dry season.

Table 6-2. Habitat residency of barramundi harvested from Mitchell River estuaries based on inference from otolith $^{87}Sr/^{86}Sr$ profiles (n=106). Table reproduced from Robins et al. (2021).

	Freshwater	Intermediate	Estuary
Mitchell overall	**63%**	**27%**	**10%**
2002 year-class	66%	28%	6%
2003 year-class	50%	29%	21%
2005 year-class	66%	24%	10%
2006 year-class[A]	43%	43%	14%

[A] samples derived from Halliday et al. (2012) with habitat class interpretation based on five Mitchell River barramundi with both $^{87}Sr/^{86}Sr$ and trace metal/calcium profiles (i.e. strontium, barium, manganese, magnesium).

Of the barramundi from the Mitchell River that moved into freshwater habitats, the majority (~52%) moved upstream during the wet season associated with their first year of life, with 41% moving during the wet season associated with their second year of life. The latter were predominantly from the 2005 year-class, which experienced relatively low flows, being the 26th-lowest wet-season flow between 1900 and 2018.

6.3.2 Total length as related to habitat class

Barramundi inferred as residing in freshwater based on their [87]Sr/[86]Sr profile had significantly larger total length-at-age than non-freshwater fish (Figure 6-4ab). When analysed with non-freshwater fish split into intermediate and estuarine, intermediate-classed fish had smaller total length-at-age than estuary fish, but the difference was not significant (Figure 6-4c).
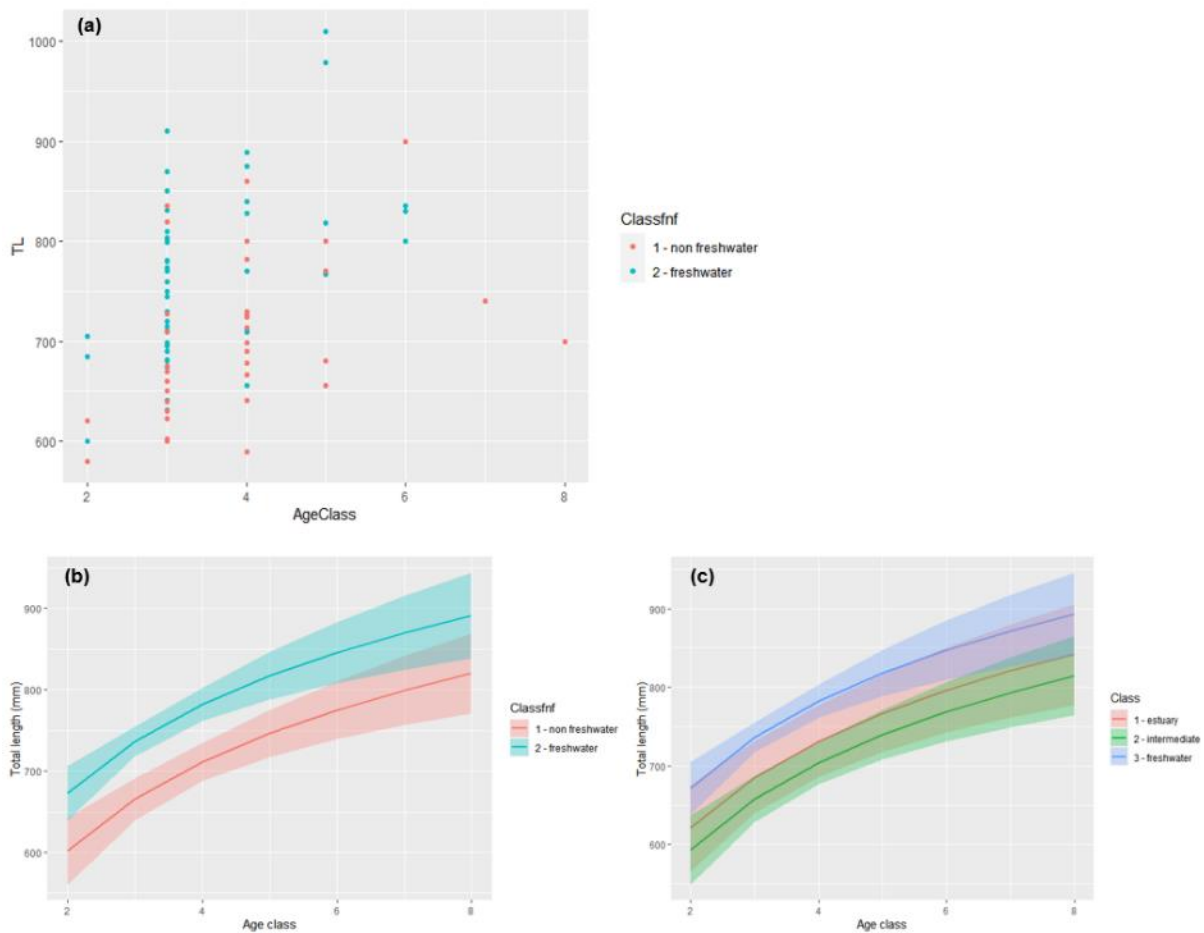
Figure 6-4. Total length by age-class for barramundi harvested from the Mitchell River estuaries: (a) observed for habitat class inferred from [87]Sr/[86]Sr otolith profiles(n=106); (b) and (c) modelled for habitat class inferred from otolith [87]Sr/[86]Sr profiles. Figure reproduced from Robins et al. (2021).

7. Implications of development for flow-dependent assets

7.1 Fish species composition, turnover and functional diversity in the Mitchell River catchment

7.1.1 Current

Connectivity, environment and species composition (abundance)

Analysis of environmental and biological data across varying levels of connectivity showed clear relationships between:

1. connectivity and environmental characteristics
2. environmental characteristics and species composition (abundance).

Environmental characteristics varied greatly between off-channel sites, poorly connected channel sites and highly connected channel sites. Highly connected channel sites were warmer and contained more woody debris (both small and large) than lower connectivity channel sites, while connected channel sites contained more bedrock. Off-channel sites contained more mud substrate and macrophytes than connected channel sites. Off-channel sites also contained more macrophytes and were more turbid than highly connected channel sites. Both poorly and highly connected channel sites were more oxygenated than off-channel sites.

There was no evidence of a strong relationship between fish species composition (abundance) and site-level connectivity; however, there was a relationship between fish species composition and the characteristics of the environment. This relationship was not influenced by connectivity, indicating a predictable association of various species with specific environmental characteristics. Bedrock cover, small woody debris and sand cover explained 31% of the variation in species composition. Substrate type was also related to the distribution and abundance of some species, including sooty grunter, barramundi and others (Figure 7-1). While barramundi did not appear to show substrate preferences, they were more abundant at highly connected sites, suggesting that connectivity may be important in determining barramundi distribution. Their diadromous life history and short-term movement patterns show that barramundi move between connected river, floodplain and estuary areas, and these movements likely support feeding and growth (Roberts *et al.*, 2019; O'Mara *et al.*, 2021).
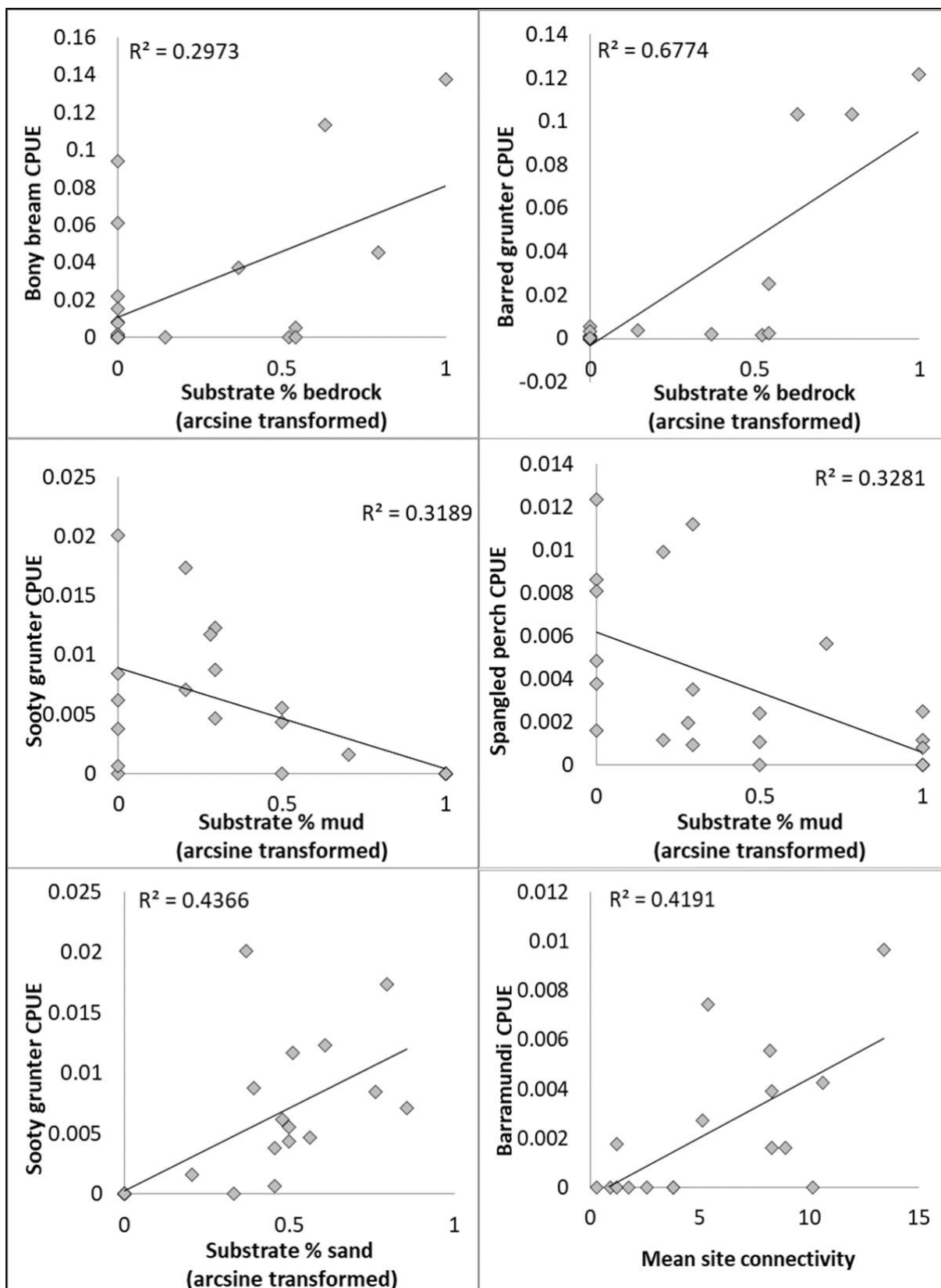
Figure 7-1. Correlations of site abundance (catch per unit effort [CPUE]) of some species with substrate types and site connectivity.

Fish species turnover and functional diversity in the Mitchell

Fish species turnover was calculated from the presence and absence of species across sites and refers to the change in species between sites, where a value of 1 would indicate complete turnover (a unique fish assemblage compared to other sites). Turnover occurred across the Mitchell catchment from the headwaters to the mouth, indicated by a correlation with longitude (r = 0.24, p = 0.001) but not latitude (*p* > 0.05) in a multivariate distance-based linear model (Figure 7-2). When the effect of longitude was accounted for in the model, bedrock cover, small woody debris and average water depth were significant predictors of turnover and together explained 59% of species turnover.
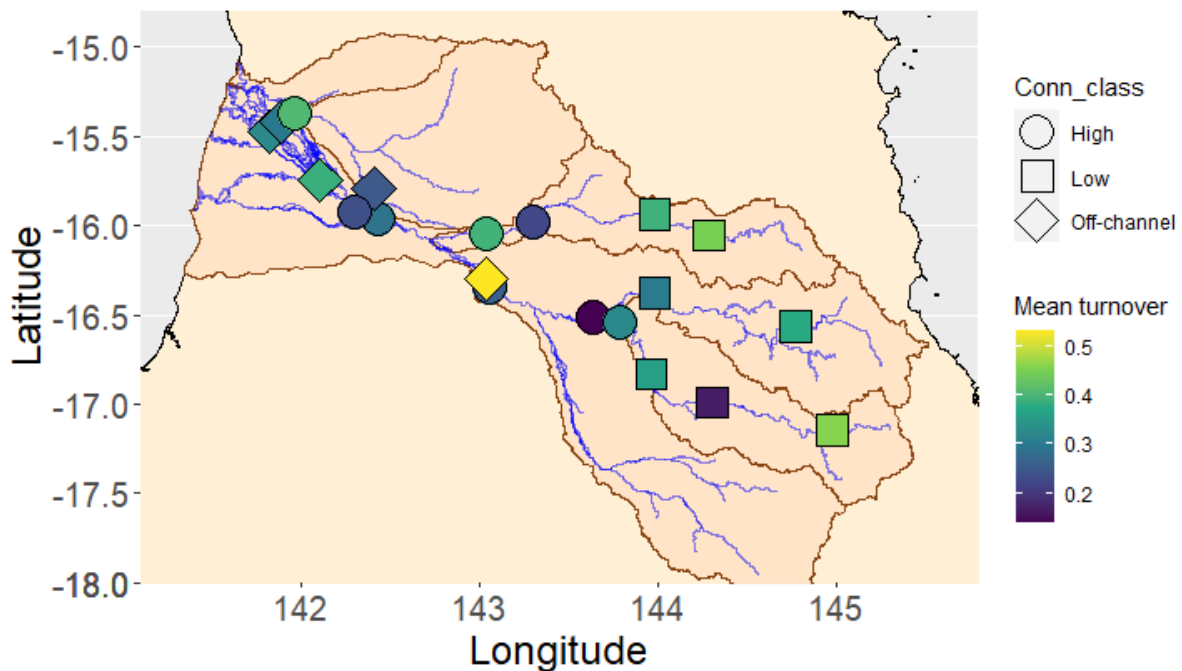
Figure 7-2. Fish species turnover (mean of pairwise site turnover [Simpson dissimilarity]) across the Mitchell catchment. The yellow diamond is the Twelve Mile Lagoon.

The highest turnover was observed in Twelve Mile Lagoon, an off-channel waterhole in the mid-catchment that is unlikely to be connected to the river during floods (Figure 7-2). The most upstream and the most downstream river-channel sites had the next-highest mean pairwise turnover values. Further, the turnover values between the most upstream sites and the next adjacent site downstream in each river were among the highest (turnover of 0.75, 0.45 and 0.33 for the Palmer, Mitchell and Walsh, respectively), indicating that headwaters contain unique fish assemblages. While the furthest upstream site in the Walsh and Mitchell rivers shared similar species (indicated by low turnover between these sites), the fish assemblage in the most upstream Palmer River site was different to the Mitchell and Walsh headwater sites. The most downstream Mitchell River main-channel site contained similar species to the lower catchment sites, but different species from headwater sites. Additionally, highly connected river reaches were more likely to contain similar species than river reaches with lower connectivity, indicated by highest turnover between adjacent sites of low connectivity and lowest turnover between adjacent sites of high connectivity (Figure 7-3). The turnover evident in the fish assemblages in the Mitchell catchment was also related to changes in fish functional diversity (Mantel test: turnover [Simpson dissimilarity] ~ functional diversity [Bray-Curtis dissimilarity], Pearson's r = 0.73).
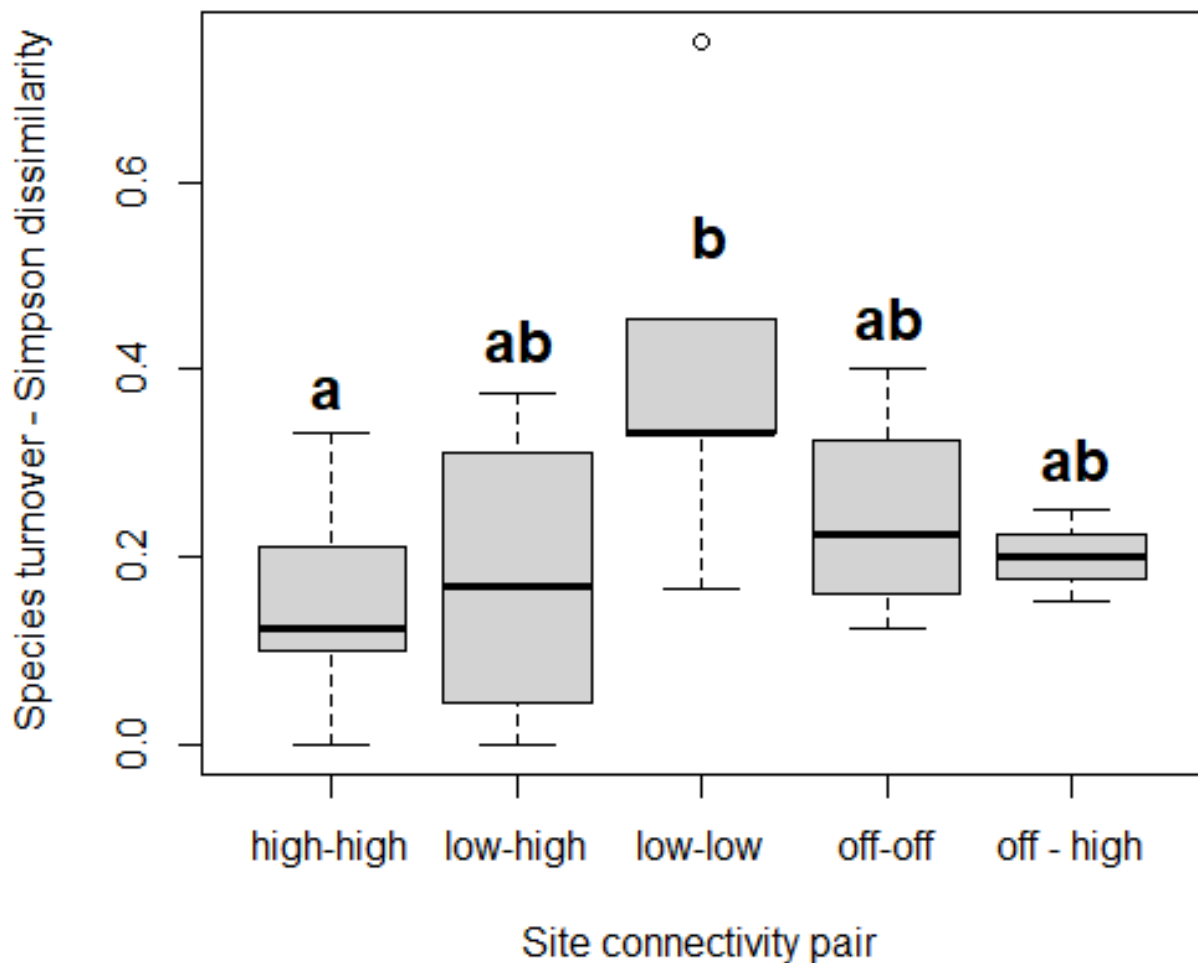
Figure 7-3. Fish species turnover across adjacent sites of varying connectivity, showing highest turnover between pairs of lowly connected sites and lowest turnover between pairs of highly connected sites. Fish movement was less restricted between highly connected sites, facilitating more similar species composition. Different letters indicate statistically significant differences (α=0.05).

Functional diversity is the biological organisation of species traits that influence ecosystem functioning (Tilman, 2001). The functional traits dataset (created from species presence/absence data) included only traits that were most likely to influence where a species was found (trophic guild, reproductive movement classification, parental care classification, and spawning substrate and frequency). Using these traits, functional diversity differed between off-channel and in-channel sites of both high and low connectivity, but not between in-channel sites of high and low connectivity (Figure 7-4). In-channel low- and high-connectivity sites were more abundant among potadromous species and species that use a mineral spawning substrate than off-channel sites. Off-channel sites were more abundant among fish that do not move to spawn than in-channel low- and high-connectivity sites. Mud cover, percentage of steep riverbanks and average water depth were the best environmental predictor variables of functional diversity and cumulatively explained 44% of the variation in functional diversity between sites (using a multivariate distance-based linear model).
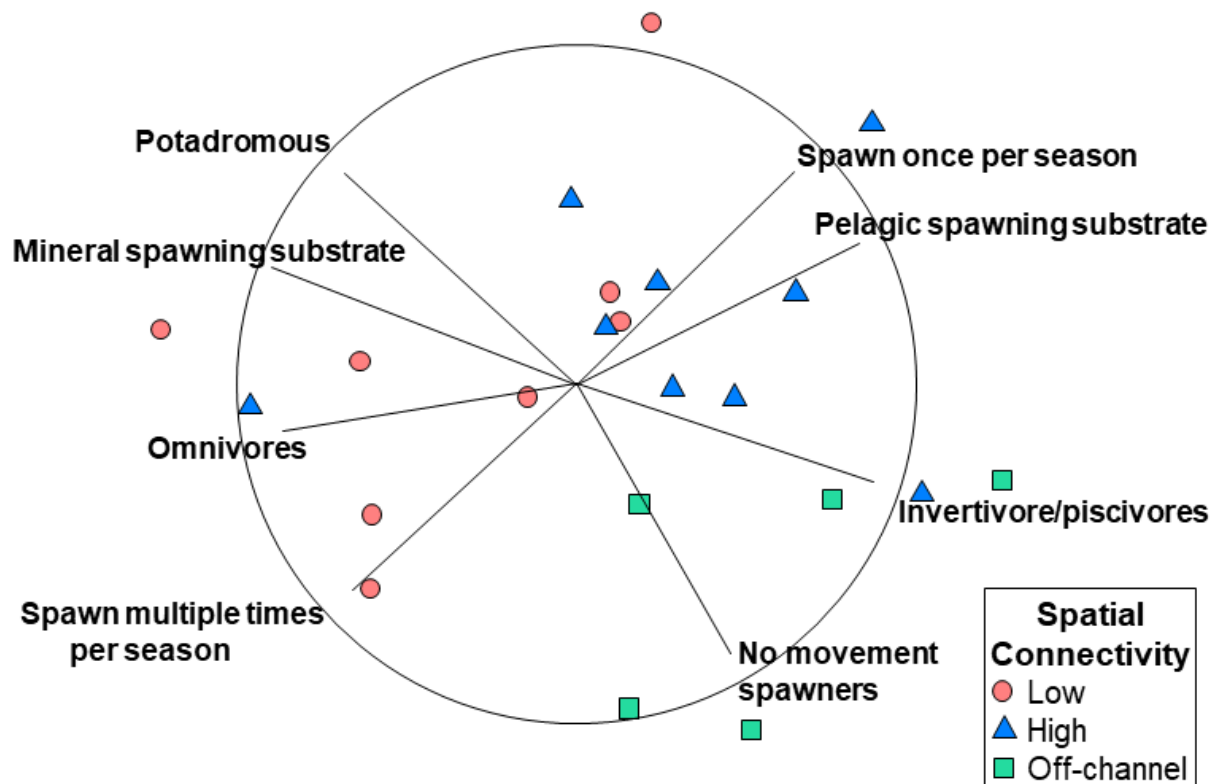
Figure 7-4. 2D visualisation of fish functional diversity traits by multidimensional scaling. Points (sites) are coloured by level of connectivity. Functional traits that best explain variation in the data are displayed as vectors.

7.1.2 Predicted impacts of water-resource development

Reductions to flow from water-resource developments are likely to reduce connectivity (O'Mara *et al.*, 2021). From the two-step connectivity–environment and environment–species composition relationship observed in the Mitchell, we can infer that changes to connectivity will likely cause consequential changes to species abundances. The high turnover between adjacent sites of low connectivity in this study suggests that differences between fish assemblages in adjacent sites are likely to become more pronounced if reaches become less connected due to flow alteration. The environmental characteristics of highly connected areas may transition to conditions typical of low-connectivity areas. Reduced flows from dams may cause areas currently defined as having low connectivity to become disconnected for longer periods. The environment in low-connectivity areas may then transition to an environment with characteristics similar to off-channel habitats, such as higher macrophyte and mud coverage, and turbidity.

While dams are predicted to reduce flows, seasonal floods will still occur (Figure E-1, Figure E-2, Figure E-3, Figure E-4) and are likely to control macrophyte populations in river channels (Riis & Biggs, 2003); however, upstream of the dam, macrophytes may thrive in the reservoir (Jones *et al.*, 2020). New dam reservoirs (pools immediately upstream of the dam wall) cause an environmental transition from lotic to lentic environments, with a decrease in flow, change in substrate and turbidity with sedimentation, and change in plant community composition (often increased floating macrophyte coverage) as the dam wall acts as a trap (Schleiss *et al.*, 2016; Shivers *et al.*, 2018). Increased macrophyte coverage is currently observed in the Mitchell catchment, with high densities of floating macrophytes in the upper

Walsh River at the Leafgold and Bruce weirs (Figure 7-5). These environmental characteristics are similar to off-channel habitats, and fish assemblages in areas that transition from connected river channel to reservoir are likely to change both in terms of species composition and functional diversity.

Species that are currently abundant in off-channel habitats, such as glassfish (*Ambassis macleayi*) and chequered rainbowfish (*Melanotaenia splendida inornata*), are likely to increase in abundance in new reservoirs, while species that are abundant in river channels but uncommon in off-channel habitats, such as barred grunter (*Amniataba percoides*) and sooty grunter (*Hephaestus fuliginosus*), are likely to decrease in abundance in new reservoirs. Additionally, the wetland-like habitat found in weir pools is favoured by invasive tilapia fishes (*Oreochromis mossambicus* and *Tilapia mariae*) that are currently found at the Bruce and Leafgold weirs on the Walsh River. Therefore, additional reservoirs from new water-resource developments are likely to create more habitat that is favoured by invasive tilapia outside of their current range.

Figure 7-5. Floating macrophytes in Bruce Weir, upper Walsh River, October 2020. Photo: Kaitlyn O'Mara.

The barrier effect of a dam is also likely to change the functional fish assemblage because diadromous species will be unable to complete critical life history migrations. Dams are known to reduce the upstream species richness of diadromous species (Gehrke, Gilligan & Barwick, 2002; Rolls *et al.*, 2014), and fishways have not been as effective for Australian fish species as they have been for fish species in other areas of the world (Harris *et al.*, 2017). In this study, barramundi and tarpon were caught, though in low abundance, in the Walsh and Palmer rivers, respectively, at sites above the proposed locations for potential dams outlined in in Petheram et al. (2018). Therefore, dams in the Mitchell catchment will likely create different upstream and downstream assemblages, including a difference in species that do not move to reproduce.

Additional to creating physical barriers and macrophyte-filled reservoirs, dams are also likely to change fish assemblages throughout the Mitchell catchment due to changes in sediment transport and deposition. Trapping of sediment by dams and reservoirs can alter upstream and downstream substrate and habitat, including causing downstream channel degradation, which can lower the water table and promote erosion and habitat desiccation (Ward & Stanford, 1995). In this study, two of the top three environmental predictors of fish abundance were coverage of various substrate types (bedrock and sand). Abundances of the species that were associated with particular substrates (sooty grunter, spangled perch, bony bream and barred grunter) are likely to change with substrate changes caused by water-resource developments. A change in substrate may also cause a functional assemblage change as the reproductive success of species that use a particular substrate to spawn may be impacted (Catalano, Bozek & Pellett, 2007). Other functional assemblage changes may also occur because of changes to connectivity.

The clear difference in proportions of no-movement spawners between river channel and off-channel habitats suggests that these differences may become more pronounced with reduced floodplain inundation. It is likely that as no-movement spawners increase in off-channel habitats, the abundance of fish species that move to spawn, such as catadromous and potadromous species, is likely to decrease. Reduced connections to floodplain wetlands would create increased risk for fish to become trapped in off-channel habitats, and catadromous or potadromous fish that risk migrating to wetlands may miss reproductive opportunities as they wait for reconnection to the river channel. This will then affect the population dynamics of these species in the Mitchell River system, and in particular for barramundi, reduce the biomass available for harvest by downstream fisheries as well as the contribution of mature individuals to the estuarine population.

7.2 Fish movement in the Mitchell River catchment

7.2.1 Fish movement over their lifetime

The spatial analysis of $^{87}Sr/^{86}Sr$ isotope ratios identified an isoscape comprising 15 different regions in distinct parts of the catchment (Figure 7-6). Comparing the $^{87}Sr/^{86}Sr$ isotope ratios in the otoliths of fish to those of the regions allows a classification of movement to be made.
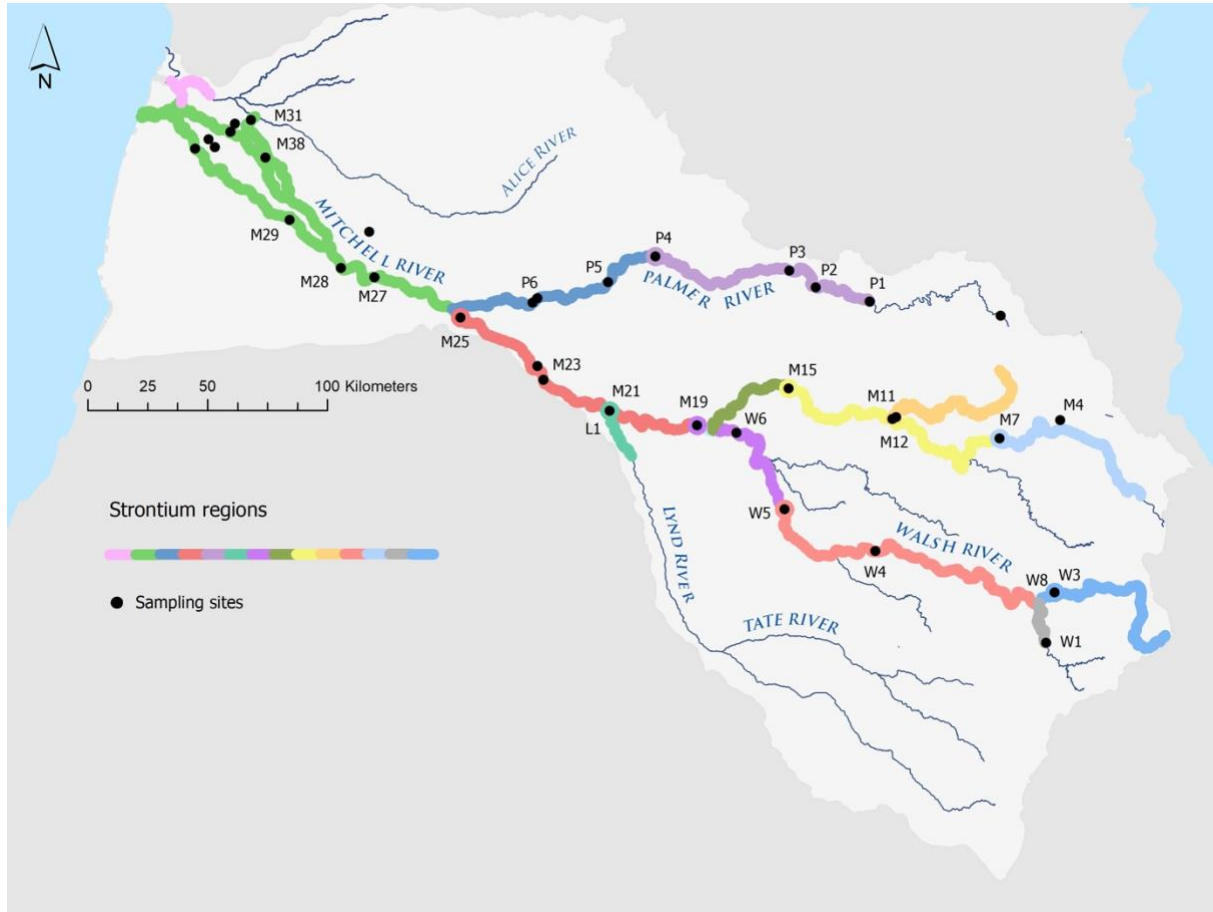
Figure 7-6. The regions in the strontium isoscape defined by $^{87}Sr/^{86}Sr$ isotope ratios. These regions were defined by spatial analysis of water samples and laser ablation of mussel samples.

There was evidence of a relatively high level of movement of fish across the catchment, based on the preliminary otolith $^{87}Sr/^{86}Sr$ isotope ratio analysis of the location of birth and death compared against the isoscape. Approximately 75% of all fish caught in the freshwater reaches showed evidence of at least one movement during their lifetime (736 of 987 fish caught). At the species level, there is a high level of movement within the assemblages, with evidence of movement across almost all species (Figure 7-7). From these analyses, there was no evidence of spawning migrations being the sole reason for movement in the assemblages, with very similar proportions of individuals moving among species that are known to migrate for spawning and those that do not (Figure 7-7).
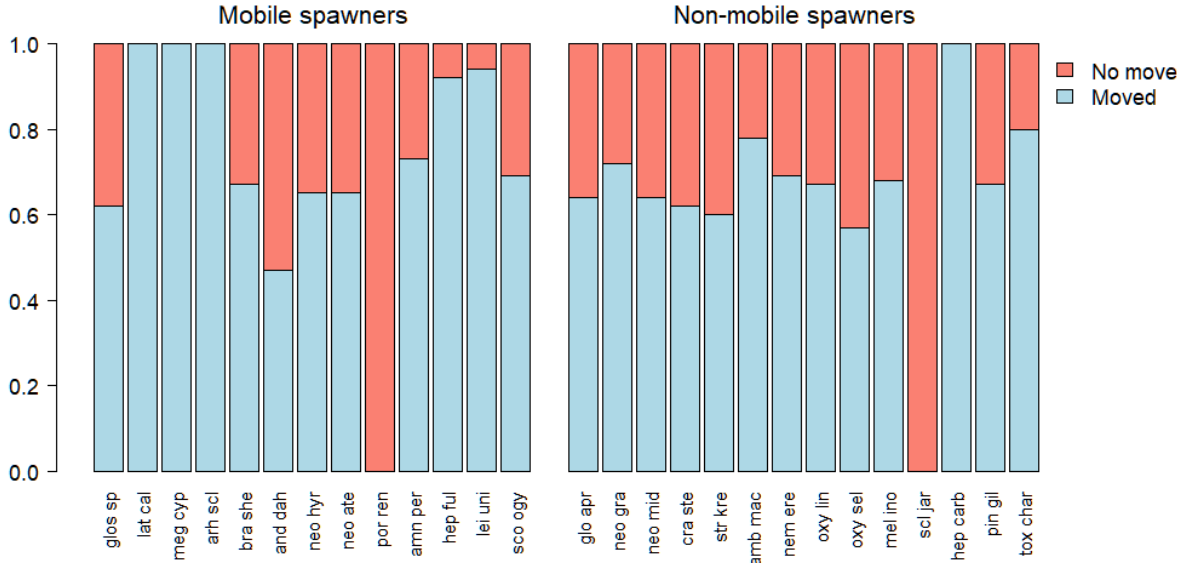
Figure 7-7. Proportion of each species that moved at least once in their lifetime separated by spawning strategy. Classification of spawning strategy comes from Sternberg and Kennard (2013). Species codes are listed in Appendix C.

Complementing the evidence of high levels of movement across all species, there was also evidence of movement throughout the catchment. The proportion of fish that showed evidence of at least one movement over their lifetime was generally high across all sampling sites (Figure 7-8). At some sites in the Walsh and Upper Mitchell, all fish caught showed evidence of movements during their lifetime (blue circles in Figure 7-8). The regions with higher proportions of fish that showed no evidence of moving tended to be in the lower Palmer and down on the floodplain of the Mitchell.
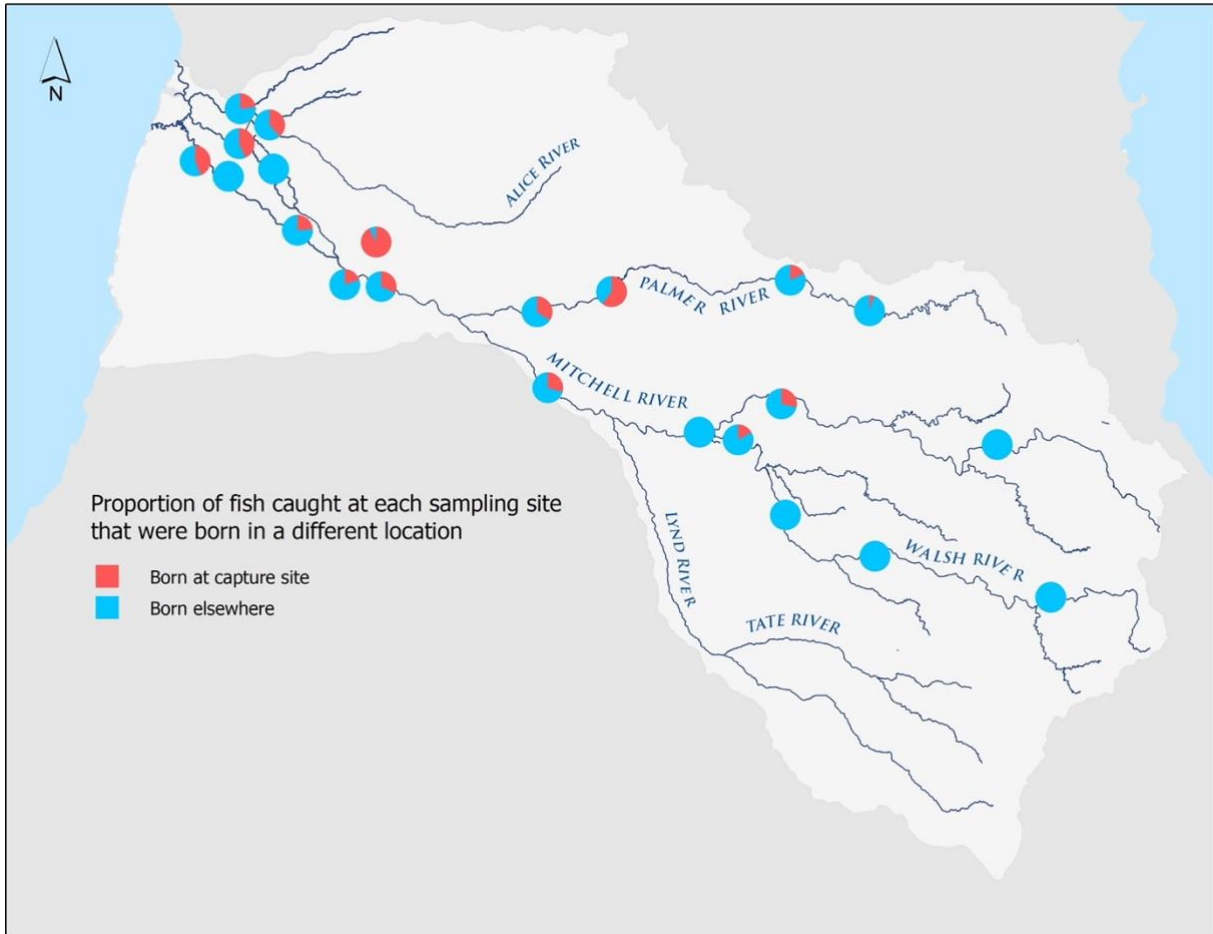
Figure 7-8. The proportion of fish caught at each sampling site that were likely to be born in a different location from where they were caught.

While there was evidence of widespread movement across all species and in all regions of the catchment, there was a distinct pattern to the location of the predicted birthplace of a large proportion of fish in the assemblages. Across the entire catchment, the Mitchell River itself appeared to be an important source habitat for the assemblages (Figure 7-9). The heat map, which shows the proportion of individuals caught at each site that were born at the location of capture and other sub-catchments, shows the Mitchell River itself as the dominant birthplace of individuals (darker colours in Figure 7-9 indicate a higher proportion). As such, access to and from the Mitchell River, from headwaters to coast, appears to be very important to support spawning, foraging and dry-season refuge for assemblages across the whole catchment.
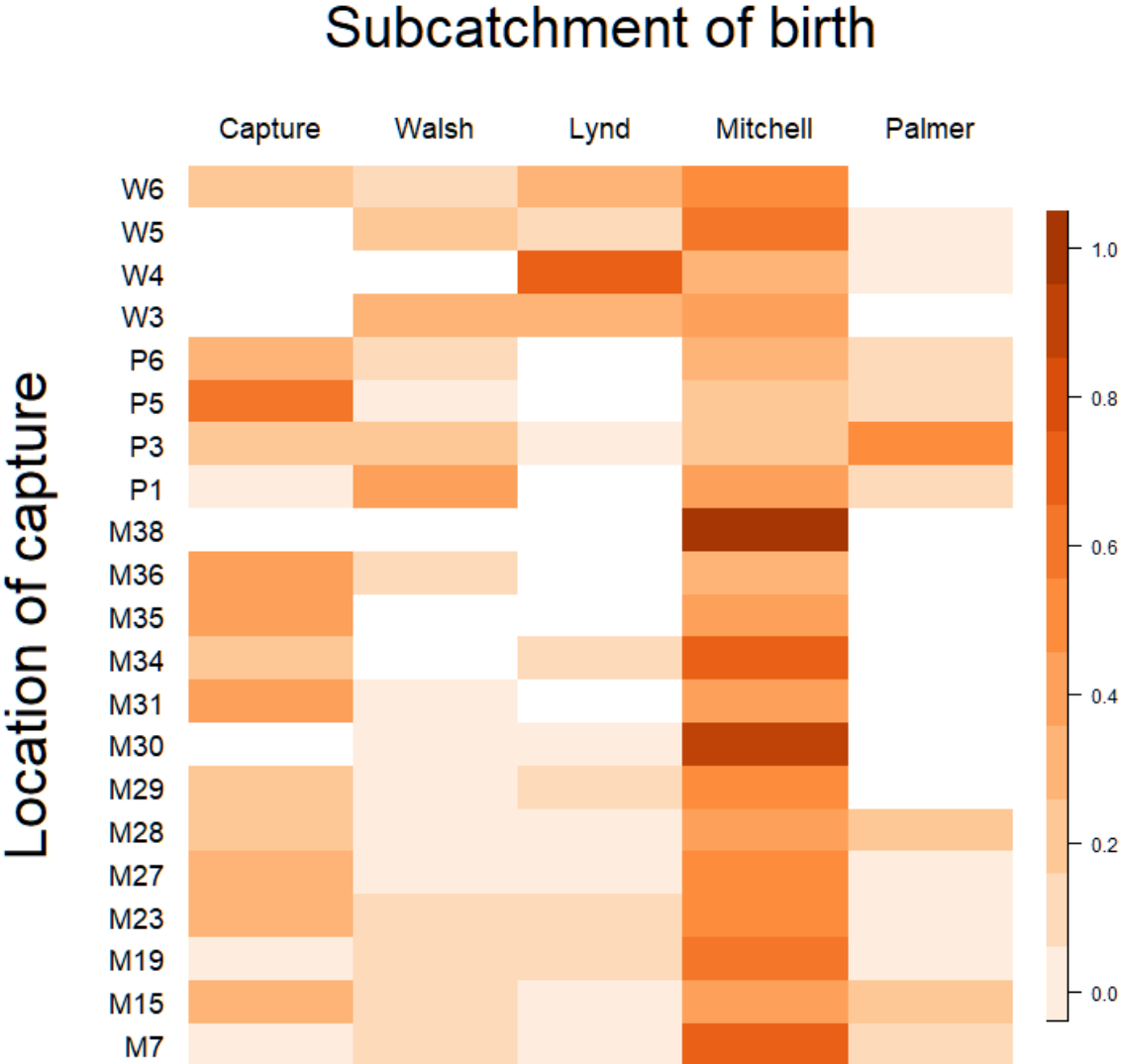
Figure 7-9. Heat map showing the predicted sub-catchment of birth for all fish caught at sampling locations around the catchment. Rows indicate the sites (site codes listed in Table 2-3 and Table 2-4), and columns indicate the location of capture and the four major sub-catchments that were sampled. The darkest colour in each row indicates the location or sub-catchment where the highest proportion of fish are predicted to have been born.

7.2.2 Recent movements

Across the catchment, 16% (48 of 299) of all large fish in this study were identified as recent immigrants to their site of capture via the sulfur stable-isotope analysis; however, this varied on a site-by-site basis, with recent immigration percentages ranging between zero and 67% (zero to seven individual migrant large fish per site; Figure 7-10, Table D-1).

At the species level, evidence for recent movement was apparent for 13 of the 17 large fish species collected in this study, though the percentage of migrants for each species varied between zero and 44% (for species with n ≥ 5). As expected, many individuals of catadromous species were migrants, and unexpectedly, most fish species that were not known to make reproductive migrations also showed evidence of movement (Table 7-1). For example, 43% of freshwater longtom and 27% of salmon catfish individuals were found to be migrants and these species are not considered to be reproductive migrators.
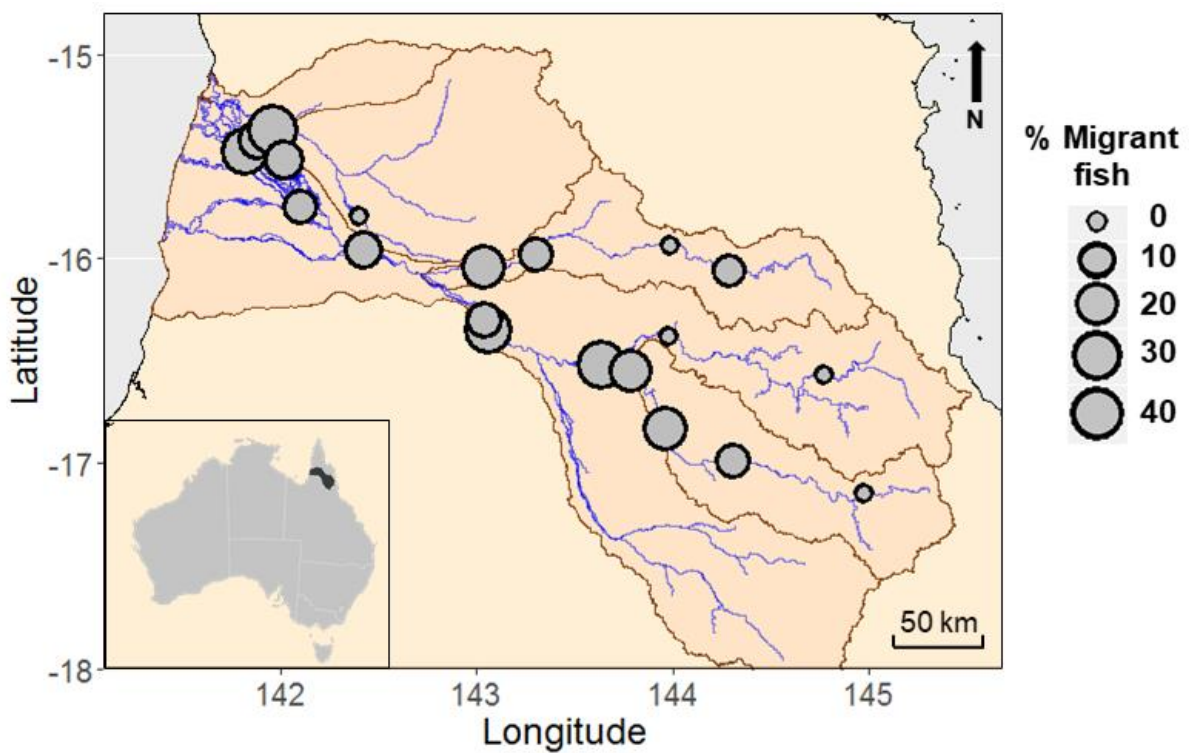
Figure 7-10. Percentage of recent migrants (indicated by size of grey points) at sites with n ≥ 5 large fish.

Table 7-1. δ³⁴S analysis results for large fish species. Fish species with individuals collected from wetlands that had δ³⁴S significantly different from the wetland prey δ³⁴S are indicated in the 'Migrated to wetlands?' column. Species are grouped by reproductive movement classifications from Sternberg and Kennard (2014).
P = piscivore/invertivore, I = invertivore, O = omnivore, HD = herbivore/detritivore.

Reproductive movement classification	Trophic guild	Species	% migrants	n	Migrated to wetlands?
Catadromous	P	Tarpon *Megalops cyprinoides*	33	15	Y
	P	Barramundi *Lates calcarifer*	44	23	Y
Potadromous	I	Toothless catfish *Anodontiglanis dahli*	0	10	
	I	Hyrtl's tandan *Neosilurus hyrtlii*	0	5	
	I	Black catfish *Neosilurus ater*	18	17	
	O	Spangled perch *Leiopotherapon unicolor*	13	16	Y
	O	Sooty grunter *Hephaestus fuliginosus*	7	29	
	HD	Gulf grunter *Scortum ogilbyi*	14	22	
No movement	P	Sleepy cod *Oxyeleotris lineolatus*	7	44	Y
	P	Shovel-nosed catfish *Neoarius midgleyi*	9	11	
	P	Salmon catfish *Neoarius graeffei*	27	34	Y
	P	Northern saratoga *Scleropages jardinii*	0	1	
	P	Mouth almighty *Glossamia aprion*	0	15	
	P	Giant gudgeon *Oxyeleotris selheimi*	17	6	
	P	Freshwater longtom *Strongylura krefftii*	43	14	
	I	Seven-spot archerfish *Toxotes chatareus*	0	11	
	HD	Bony bream *Nematalosa erebi*	9	22	

Barramundi and tarpon were found to be two of the most mobile species, which was expected given their catadromous life history. However, otolith microchemistry studies have found that most barramundi do not migrate to the sea each year to breed, and most sub-adult tarpon (the likely age of all tarpon in this study) would have not yet migrated to the sea to spawn (Milton & Chenery, 2005; Shen *et al.*, 2009; Crook *et al.*, 2017). Therefore, movements of barramundi and tarpon found in this study may represent foraging for food or movement to more permanent habitats, rather than reproductive migrations. Migrant barramundi were found only in deeper sites (>2 m depth) and it is possible that they move to these deeper habitats to ensure they have enough water to survive the dry season (Moore & Reynold, 1982) and/or to increase their chances of success in foraging for prey fish.

Movement of large fish (percentage of migrant fish at a site, determined from $\delta^{34}S$ analysis) was higher at river-channel sites that were more connected to the rest of the river network (Figure D-1). In fact, over half of the variation in large fish movement across sites was explained by flow-mediated site connectivity. The relationship between fish movement and connectivity suggests that realised movement is a function of opportunity. Post wet-season movement may be related to fish moving into deeper pools to find refuge as flow subsides, which has also been observed by Kaus et al. (2018).

7.2.3 Predicted impacts of water-resource development

The connectivity–movement model predicted that dams would have a negative impact on the movement potential of large fish across river-channel sites, with each of the dam scenarios resulting in appreciable reductions in predicted migration. The largest predicted change in the percentage of migrants was under the all-dams scenario, with movement potential into sites in the Mitchell sub-catchment from elsewhere in the catchment predicted to decrease by a mean of 45% (Table 7-2, Figure 7-11). There was little difference in predicted decrease in movement potential between the all-dams and the three-dams scenario, with both scenarios resulting in reductions ranging between 20% and 40%. The individual dam scenarios (Chillagoe, Palmer or Pinnacles) were predicted to most affect the movement potential of fish within the sub-catchment in which they were located. However, the Pinnacles dam scenario had a similar predicted effect on fish movement potential in the Palmer sub-catchment to the Palmer dam scenario, through changes in connectivity of downstream Palmer River sites with the rest of the river network (Table 7-2). Predicted values for individual dam scenarios showed that the Pinnacles dam is likely to have a greater effect on fish movement potential across more sites than the Chillagoe dam or the Palmer dam (Table 7-2 and Table 7-3). Fish at the lower Mitchell River sites (Sites 11 and 16 in Figure 7-11) are likely to be more heavily impacted by reduced connectivity from the Pinnacles dam than the Chillagoe or Palmer dams. These lower Mitchell sites serve as a migration pathway for diadromous species and provide important connections to the floodplain. Reductions in connectivity at these sites may therefore affect reproductive success and restrict access to productive habitats.

Table 7-2. Sub-catchment mean (± standard deviation) percentage change in migrants from current (no-dam) percentages. The three-dams scenario includes the potential Chillagoe, Palmer and Pinnacles dams.

Sub-catchment	All dams	Three dams	Chillagoe dam only	Palmer dam only	Pinnacles dam only
Mitchell	−45 ± 15	−42 ± 14	−8 ± 6	−6 ± 2	−31 ± 12

| Palmer | −33 ± 15 | −31 ± 14 | −5 ± 2 | −14 ± 9 | −16 ± 12 |
| Walsh | −26 ± 15 | −20 ± 12 | −15 ± 10 | −3 ± 2 | −7 ± 6 |
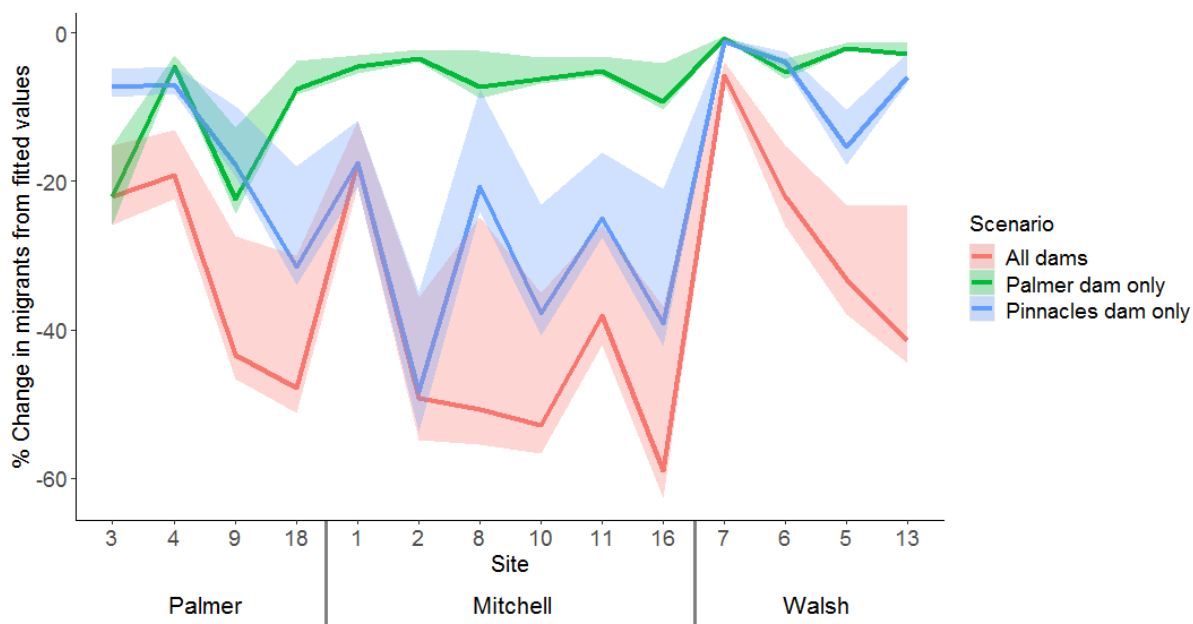
Figure 7-11. Percentage change across sites from current migrant percentages under three different dam scenarios. The all-dams scenario is likely to have the greatest impact on fish movement. The Pinnacles dam is likely to have a greater impact on movement at most sites, particularly lower Mitchell River sites, than a dam on the Palmer River. Only three scenarios were plotted for clarity of the plot.

Table 7-3. Actual, fitted and predicted percentages of migrants (% migrants) with 95% prediction intervals (PI) under five dam scenarios from 15 years (2000–2015) of modelled CSIRO stream gauge data.

Site	Actual % migrants	Fitted % migrants	All dams % migrants	All dams Lower PI	All dams Upper PI	Three dams % migrants	Three dams Lower PI	Three dams Upper PI	Chillagoe dam only % migrants	Chillagoe dam only Lower PI	Chillagoe dam only Upper PI	Palmer dam only % migrants	Palmer dam only Lower PI	Palmer dam only Upper PI	Pinnacles dam only % migrants	Pinnacles dam only Lower PI	Pinnacles dam only Upper PI
M7	0	5	0	1	24	0	1	24	5	1	26	5	1	26	0	1	24
M15	0	9	4	1	25	5	1	25	8	1	37	8	1	38	5	1	25
M19	35	43	21	4	65	24	4	68	34	7	80	40	8	84	34	7	80
W1	30	20	10	2	41	10	2	42	19	3	62	19	3	61	13	2	48
M27	14	12	7	1	35	8	1	36	11	2	45	11	2	45	9	1	39
M31	40	29	12	2	47	13	2	50	26	5	72	27	5	72	18	3	59
P1	8	5	0	1	24	0	1	24	5	1	27	0	1	24	5	1	26
P3	0	6	5	1	26	5	1	27	6	1	29	6	1	29	5	1	29
P5	9	20	11	2	45	12	2	47	18	3	60	15	3	55	16	3	56
M29	25	22	12	2	46	12	2	48	21	4	65	21	4	64	15	3	54
W3	0	4	0	1	24	0	1	24	0	1	24	4	1	24	4	1	24
W4	10	5	0	1	24	4	1	25	5	1	26	5	1	27	5	1	27
W5	24	7	5	1	26	5	1	28	6	1	30	7	1	33	6	1	30
W6	22	27	16	3	56	18	3	59	19	3	62	27	5	72	26	5	71

Reduced river flows can lead to earlier disconnection of habitats, and water-resource development in the form of dams and other infrastructure adds the direct effect of a barrier across the channel (Branco *et al.*, 2014). These dam scenarios are examples of extreme development impacts and present worst-case scenarios. Some of these impacts may be mitigated by the use of fish passages on dams, or developing off-channel water storage instead of major large dams (Harris *et al.*, 2017). However, these also come with their own ecological impacts that would need to be investigated. Our connectivity–movement model provides a tool that can be used in optimisation models to determine whether there are infrastructure solutions that have minimal impact on fish movement but are also viable for agriculture. The modelling tool can also be used to make predictions that can be directly integrated into management decision-making tools, such as ecological risk assessments (e.g. Mcgregor *et al.*, 2018).

The NAWRA River Model app (nawra-river.shinyapps.io/river) (Hughes *et al.*, 2017) provides an avenue to develop additional water-resource development scenarios that can be integrated into the connectivity index and regression model to predict likely changes to connectivity and the environment.

7.3 Energy flow through food webs and across habitats in the Mitchell River catchment

7.3.1 Current

Energy flow through Mitchell River food webs

Two diet pathways were revealed in the study of fatty acids in Mitchell River food webs. Firstly, there was an aquatic pathway through which fish derived most of their fatty acids from consuming aquatic macroinvertebrates like mayfly and dragonfly larvae, which in turn fed on algae and aquatic plants (Figure 7-12). The second diet pathway was terrestrial, where terrestrial macroinvertebrates (grasshoppers) most likely fed on terrestrial plants, which contributed little to the aquatic food webs (Figure 7-12).
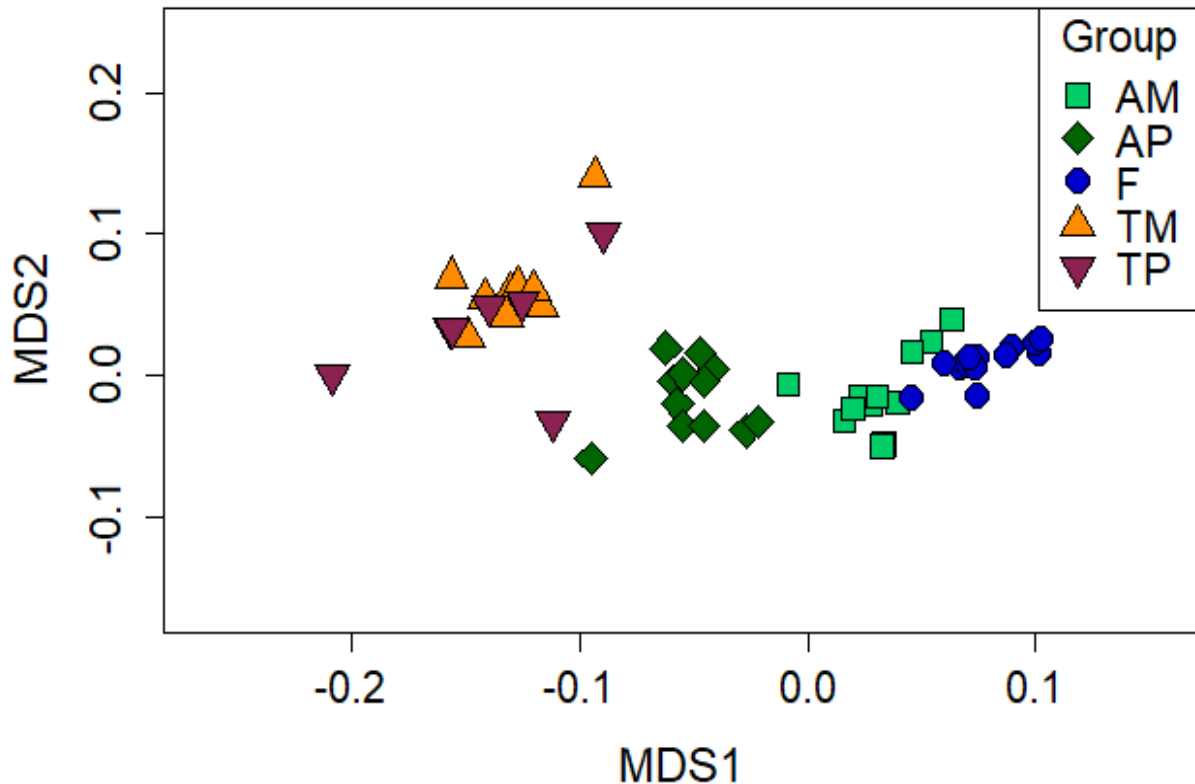
Figure 7-12. Non-metric multidimensional scaling (nMDS) of fatty-acid composition (%) for all food web components. Symbols represent mean site values of food web groups: AM = aquatic macroinvertebrates, AP = aquatic plants, F = fish, TM = terrestrial macroinvertebrates, TP = terrestrial plants. 2D stress = 0.18.

Fish fatty-acid composition varied among trophic guilds, with eicosapentaenoic acid (EPA), docosahexaenoic acid (DHA), oleic acid and docosatetraenoic acid best separating guilds (Figure F-1). Differences in fatty-acid composition among trophic guilds indicated that selective feeding influenced predator–prey similarities in fatty-acid composition. Predator–prey energy transfer and retention was assessed by examining concentration ratios of total lipids and individual fatty acids between trophic levels for aquatic plants, aquatic macroinvertebrates and fish. The macroinvertebrate-to-plant fatty-acid ratio, used as an indicator of fatty-acid bioaccumulation in consumers relative to their plant diet, was high across all fatty acids, except for lignoceric acid and the omega-3 (n-3)/omega-6 (n-6) ratio (Table 7-4). Macroinvertebrates concentrated lipids and the longer-chain polyunsaturated fatty acids (PUFAs) EPA, DHA and arachidonic acid (ARA) from their plant diet, providing a higher-quality and more energy-rich food for fish than plants. In contrast to macroinvertebrates, fish did not concentrate lipids and fatty acids from their diet (macroinvertebrates), except for lignoceric acid and DHA (Table 7-4). This indicates that DHA is mostly produced by these tropical fish, since the fish cannot obtain required amounts of DHA from the diet because of the low DHA in plants and macroinvertebrates. Fish had similar contents of stearidonic acid, terrestrial fatty acids and n-3/n-6 ratios to macroinvertebrates. Macroinvertebrates and fish from the Mitchell River system also contained high amounts of ARA.

Table 7-4. Concentration ratios of fatty acids between major aquatic food web groups (no terrestrial sources were included in these groups). Most fatty acids were more concentrated in aquatic macroinvertebrates than in aquatic plants. Macroinvertebrates were six times more energy-rich (total lipid concentration ratio) than plants and are therefore a high-quality diet for fish. Values are mean ratios across all sites and species. AP = aquatic plants, AM = aquatic macroinvertebrates, F = fish.

Food web groups	AM:AP	F:AM
Total lipids	6	0.5
Myristic acid	20	0.7
Palmitic acid	14	0.6
Stearic acid	22	0.6
Oleic acid	27	0.5
Linoleic acid (LA)	17	0.4
Alpha-linoleic acid (ALA)	8	0.3
Stearidonic acid	15	1
Arachidonic acid (ARA)	96	0.7
Lignoceric acid	0.4	2
Eicosapentaenoic acid (EPA)	77	0.2
Docosahexaenoic acid (DHA)	186	8
Saturated fatty acids	15	0.6
Monounsaturated fatty acids (MUFA)	23	0.6
Polyunsaturated fatty acids (PUFA)	17	0.6
Bacterial fatty acids (BAFA)	31	0.6
Terrestrial fatty acids	5	0.8
n-3	17	0.6
n-6	23	0.6
n-3/n-6 ratio	0.7	1

This study of fatty acids in Mitchell River food webs demonstrated that the energy and dietary quality available to tropical freshwater fish of the Mitchell River is largely driven by aquatic macroinvertebrates (Table 7-4, Table F-1, Figure F-1). Aquatic macroinvertebrates selectively retained fatty acids, in particular the long-chain PUFAs, and contained more total lipids than aquatic plants and even fish, which demonstrates that tropical aquatic macroinvertebrates provide energy-rich and high-quality food for fish. In fact, macroinvertebrates provided six times more energy to fish than plants. Macroinvertebrate abundance is therefore important for fish nutrition in Mitchell River food webs. For some fatty acids, such as alpha-linoleic acid (ALA) and linoleic acid (LA), the lower concentrations in fish were proportional to lower total lipid content; for other fatty acids, lower concentrations may indicate that fish either use more of the fatty acids than macroinvertebrates for higher functions or do not require them to the same degree as macroinvertebrates. Fatty-acid content of fish in the Mitchell River was similar for most fatty acids to fish in other tropical freshwater regions (Zenebe, Ahlgren & Boberg, 1998; Guo *et al.*, 2017), but different from temperate, cold-water fish or marine fish (Parzanini *et al.*, 2020). This indicates that water temperature and salinity, in addition to fish taxa, likely play an important role in determining fishes' lipid and fatty-acid composition.

Energy flow across habitats in the Mitchell River

The fatty-acid composition of samples was similar between samples from the same food web group, regardless of capture habitat; that is, fish from any given habitat were more similar to fish from other sites than to macroinvertebrates or plants from the same habitat (Figure 7-12). Separate analyses of the fish and aquatic macroinvertebrate groups showed that there were some differences among habitats for EPA, but not for total lipids and other PUFAs. Both macroinvertebrates and fish from river-channel sites on the Mitchell River in the middle-to-lower catchment had higher EPA than those in other habitats. Similarly, Larson et al. (Larson *et al.*, 2015) found higher PUFAs (including EPA) in river-channel compared to off-channel fish and macroinvertebrates. Periphyton and epiphyton were high in EPA relative to other plant types and river-channel sites provide abundant surfaces for periphyton growth, such as rocks, sand and wood. It is likely that the abundant periphyton in river-channel habitats are a primary food source for aquatic macroinvertebrates, even though plants have low EPA compared with macroinvertebrates. High EPA in macroinvertebrates compared with plants is likely due to their preferential retention of algal EPA (Kühmayer *et al.*, 2020); however, little is known about their ability to synthesise EPA. Fish may also synthesise some EPA (Murray *et al.*, 2014), making it difficult to distinguish how much of the EPA in fish came from diet.

In contrast to EPA, there was no difference in ALA, LA, ARA, DHA or total lipids across macrohabitats for fish and aquatic macroinvertebrates, indicating that river channels, floodplain creeks and wetlands were of equal dietary quality for fish. These results suggest that the overall dietary quality across taxa in each community is not different between macrohabitats. However, since we did not measure macroinvertebrate abundance or biomass, it was not possible to determine macrohabitat dietary quality in terms of abundant high-quality macroinvertebrate taxa. The abundance and composition of macroinvertebrate taxa present is highly likely to influence dietary quality because total lipids and PUFA were highly variable between macroinvertebrate taxa. Ephemeropterans were rich in total lipids and EPA, while hemipterans and coleopterans were rich in ARA, and decapod shrimp had higher DHA than other macroinvertebrates. The quality of food received by the fish is therefore dependent on the macroinvertebrate taxa present (abundance of taxa high in PUFAs and taxa low in PUFAs), and fish may be advantaged or disadvantaged by the composition of the macroinvertebrate community present at a location. Further, macroinvertebrate community composition (family-level) is different across microhabitats (sandy or rocky bed, edge, riffle, macrophyte: Thomson *et al.*, 2002), and data collected in the project but not shown here found that the overall site abundance of macroinvertebrate taxa (order-level) varies with microhabitat coverage and composition in the Mitchell River. Therefore, both presence/absence and coverage of microhabitat present at a site is likely to influence food quality.

7.3.2 *Predicted impacts of water-resource development*

High densities of macroinvertebrates are associated with macrophytes, and floodplain wetlands are known to have high macrophyte biomass (Gregg & Rose, 1985; Thomaz *et al.*, 2008). Therefore, wetlands may be able to support higher macroinvertebrate densities than habitats with lower macrophyte coverage, such as flowing river channels. Wetland macrophytes provide habitat structure for macroinvertebrates and a surface for epiphyton growth, and are therefore important for supporting fish biomass through the dry season until

the wetlands become reconnected to the river network (Molinari et al. 2021c; Thomaz et al. 2008). The relatively unregulated flows of the Mitchell River system mean fish are able to use seasonal connectivity to move freely across habitats (Jardine et al. 2012; O'Mara et al. 2021). New water-resource developments in the Mitchell catchment would likely reduce connectivity throughout the catchment (O'Mara *et al.*, 2021), including reduced floodplain inundation (Section 5). Reduced floodplain inundation not only reduces opportunities for fish to migrate into and out of wetlands, but also increases reliance on rainfall to sustain wetland area, with decreased area reducing habitat for macroinvertebrates. Therefore, the pressure to expand agriculture and regulate river flows not only threatens access to high-quality food for fish, but also threatens food quality and availability through changes in microhabitat, leading to shifts in community structure and composition of macroinvertebrates (Bowen, Bovee & Waddle, 2003; Leigh *et al.*, 2013).

The community structure and composition of macroinvertebrates and algae have also been shown to be affected by operations of water-resource infrastructure, such as timing, frequency and magnitude of downstream flows (Bednarek & Hart, 2005; Wu *et al.*, 2009). As macroinvertebrates and algae provide the basis of high-quality food for the fish assemblages of the Mitchell River, it is important to factor least-impact flows into the design of new water-resource developments, with the aim of mitigating as many impacts on biodiversity as possible.

8. Conclusions

The broad range of research approaches used in this project demonstrates the importance of flow-mediated connectivity in supporting aquatic ecosystems in the Mitchell River. Several key findings are relevant to water planning, and several predictive tools were developed that can be used to predict the impact of changes to river flows that may occur in the future from water-resource development and/or climate change.

The natural seasonal flow regime drives ecosystem functioning across the catchment, with stark differences observed between the wet and dry seasons (Figure 8-1 and Figure 8-2).

8.1 Wet season

- All rivers flow in the wet season and floodplains are periodically inundated, connecting these wetlands with river channels. Connectivity and its seasonal changes determine the habitat and food available for fish, and these vary across the catchment. High turnover of species occurs in the headwaters and between adjacent sites of low connectivity, indicating that species composition is influenced by opportunities for movement between sites.
- Floodplain inundation and aquatic plant biomass accumulation are dependent on upstream flows and rainfall. At the end of the 2017–2018 wet season, most of the algal primary productivity on the floodplains occurred in wetlands. Floodplain wetlands are a major source of high-quality food for fish. Macroinvertebrates feed on epiphytic algae, concentrate lipids such as polyunsaturated fatty acids, and are then eaten by fish. Fish use wet-season connections to the floodplain to access this food source. Inundation periods can be short, and many of the fish that move into floodplain wetlands do not have the opportunity to move back to the river until the following wet season. Sufficient wet-season flows are required to ensure floodplain wetland habitats persist through the dry season to support this biodiversity.
- Fish movement varies between species and across the catchment, and most fish make at least one large-scale movement in their lifetime, showing that even non-reproductive individuals access different parts of the catchment. Most fish caught in the Walsh and upper Palmer rivers had moved there from other areas of the catchment. Short-term fish movement is dependent on river-reach connectivity, with higher movement occurring through more connected areas.
- Barramundi that access freshwater habitats in the wet season have faster growth than those that stay in the coastal zone. Wetter years with larger flows support increased growth, abundance and biomass of coastal barramundi.

Figure 8-1. Conceptual model of Mitchell River ecosystem function in the wet season.

8.2 Dry season

- The mainstem of the Mitchell River flows perennially, while the Walsh (below the irrigation scheme), Lynd and upper Palmer rivers cease to flow in the dry season and form a series of disconnected pools. Floodplain inundation recedes after wet-season floods and water only remains in depressions in the landscape (some wetlands and creeks) until the next wet season. These variations in flows throughout the catchment create habitats with different degrees of connectivity to the rest of the river network. Connectivity changes across the catchment and with longitudinal landscape changes (e.g. elevation, geology).
- The Mitchell Falls on the Mitchell River are a natural barrier to upstream fish migration. Below this barrier, fish in the main channel of the Mitchell can move freely within this reach during the dry season, while all other fish in the catchment find refuge in wetlands or pools. Off-channel wetlands contain functionally unique fish assemblages, with fish that do not need to move to spawn in higher proportions than at sites in the main channel, making these wetlands important habitats for the biodiversity of the catchment.
- As the dry season progresses and habitats contract, fish become more reliant on local food sources to sustain dry-season biomass.

Figure 8-2. Conceptual model of Mitchell River ecosystem function in the dry season.

8.3 Possible threats to Michell River ecosystem functioning from water-resource development

Possible threats to Mitchell River ecosystem functioning from water-resource development (Figure 8-3):

- Dams impact lateral and longitudinal connectivity by reducing river flow and acting as a barrier. This reduces the potential for fish movement and access to primary productivity in floodplain wetlands as fish can no longer access the main river channel.
- Reduced flows under water-resource development scenarios are predicted to affect the growth of barramundi in the coastal fishery, with younger fish likely to be most impacted.
- Connectivity changes characteristics of the environment which, in turn, influences fish species composition and functional diversity. Reduced connectivity will alter fish assemblages, thereby impacting the biodiversity of the catchment.
- Impoundments created from dams and weirs are ideal habitat for invasive tilapia.
- Many locations in the catchment, including floodplains, are phosphorus-limited. Eutrophication, harmful algal blooms, and ultimately oxygen depletion and fish kills would be a risk if agricultural run-off resulted in increased phosphorus levels.

Figure 8-3. Conceptual model of possible threats to Mitchell River ecosystem function from water-resource development.

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Appendix A. Water quality

Table A-1. Water quality – nutrients observed in the Mitchell River catchment.

Water quality site code	Sample date	Ammonium nitrogen as N (mg/L)	Oxidised nitrogen as N (mg/L)	Dissolved Kjeldahl nitrogen as N (mg/L)	Total nitrogen as N (mg/L)	Phosphate phosphorus as P (mg/L)	Dissolved Kjeldahl phosphorus as P (mg/L)	Total Kjeldahl phosphorus as P (mg/L)
9190007	14/10/2017	0.008	0.001	0.29	0.35	0.001	<0.02	0.03
9190009	21/10/2017	0.004	<0.001	0.18	0.25	0.004	<0.02	0.02
9190010	12/10/2017	0.011	0.001	0.27	0.27	0.001	<0.02	<0.02
9190011	20/10/2017	0.003	<0.001	0.23	0.23	<0.001	<0.02	0.05
9190011	15/06/2018	0.004	0.008	0.15	0.35	<0.001	<0.02	<0.02
9190015	21/10/2017	0.018	0.238	0.51	1.25	0.004	0.02	0.13
9190023	21/10/2017	0.014	0.022	0.49	0.74	0.005	0.03	0.07
9190034	25/04/2018	0.006	0.001	0.35	0.5	0.004	0.04	0.09
9190034	9/05/2019	0.006	0.004	0.29	0.51	0.005	0.03	0.09
9190035	12/06/2018	0.002	<0.001	0.12	0.17	<0.001	<0.02	<0.02
9190035	14/10/2017	0.008	0.002	0.16	0.23	0.002	<0.02	0.02
9190036	8/06/2018	<0.002	<0.001	0.12	0.1	<0.001	<0.02	<0.02
9190037	14/10/2017	0.012	<0.001	0.1	0.11	0.002	<0.02	<0.02
9190043	16/10/2017	0.005	0.009	0.13	0.18	<0.001	<0.02	0.02
9190069	21/10/2017	0.011	0.005	0.38	0.46	<0.001	<0.02	0.03
9190073	15/10/2017	0.004	<0.001	0.19	0.25	<0.001	<0.02	<0.02
9190073	14/06/2018	0.019	0.037	0.32	0.47	<0.001	<0.02	<0.02
9190079	15/10/2017	0.012	0.013	0.21	0.18	0.004	<0.02	<0.02
9190081	24/04/2018	0.008	<0.001	0.57	0.83	0.069	0.11	0.34
9190082	26/04/2018	0.006	0.002	0.37	0.63	0.006	0.04	0.11
9190083	6/06/2018	0.007	0.002	0.19	0.3	<0.001	<0.02	0.04
9190083	11/05/2019	0.006	<0.001	0.24	0.41	0.004	<0.02	0.04
9190083	27/04/2018	0.007	<0.001	0.48	0.77	0.004	0.05	0.12
9190084	14/05/2019	0.011	0.001	0.5	0.68	0.002	<0.02	0.04
9190084	30/04/2018	0.013	<0.001	0.91	1.08	0.011	0.05	0.07
9190085	1/05/2018	0.007	<0.001	0.57	0.72	<0.001	0.02	0.05
9190085	10/05/2019	0.011	<0.001	0.6	0.82	0.008	0.03	0.08
9190086	7/06/2018	0.004	<0.001	0.27	0.32	<0.001	<0.02	0.03
9190086	2/05/2018	0.006	<0.001	0.24	0.34	<0.001	<0.02	0.03
9190086	8/05/2019	0.006	0.001	0.2	0.36	<0.001	<0.02	0.03
9190087	11/06/2018	0.004	0.002	0.28	0.36	<0.001	<0.02	<0.02

| Water quality site code	Sample date	Ammonium nitrogen as N (mg/L)	Oxidised nitrogen as N (mg/L)	Dissolved Kjeldahl nitrogen as N (mg/L)	Total nitrogen as N (mg/L)	Phosphate phosphorus as P (mg/L)	Dissolved Kjeldahl phosphorus as P (mg/L)	Total Kjeldahl phosphorus as P (mg/L)
9190088	13/05/2019	0.006	0.066	0.29	0.53	<0.001	0.02	0.04
9192002	18/10/2017	0.008	0.001	0.36	0.53	<0.001	<0.02	0.04
9192006	14/10/2017	0.006	0.001	0.25	0.24	<0.001	<0.02	<0.02
9192007	13/06/2018	0.006	0.001	0.12	0.17	<0.001	<0.02	<0.02
9192009	19/10/2017	0.013	0.013	0.48	0.75	0.002	<0.02	0.16
9193007	20/10/2017	0.005	0.188	0.21	0.51	<0.001	<0.02	0.03
9193010	20/10/2017	0.005	0.005	0.26	0.28	<0.001	<0.02	0.03
9193010	12/10/2017	0.01	0.006	0.25	0.36	0.001	<0.02	<0.02
9193022	20/10/2017	0.004	<0.001	0.29	0.39	<0.001	<0.02	0.03
9193028	20/10/2017	0.008	0.003	0.28	0.28	<0.001	0.03	0.02
9193028	12/10/2017	0.01	0.002	0.26	0.37	0.002	<0.02	0.03
919009a	10/06/2018	0.002	0.009	0.15	0.16	<0.001	<0.02	<0.02
919201a	21/10/2017	0.016	0.001	0.74	1.28	0.028	0.07	0.2
919205a	18/10/2017	0.022	0.003	0.97	1.52	0.005	0.03	0.17
919309a	12/10/2017	1.39	0.006	2.93	2.21	0.002	0.04	0.12

Appendix B. Finfish catch
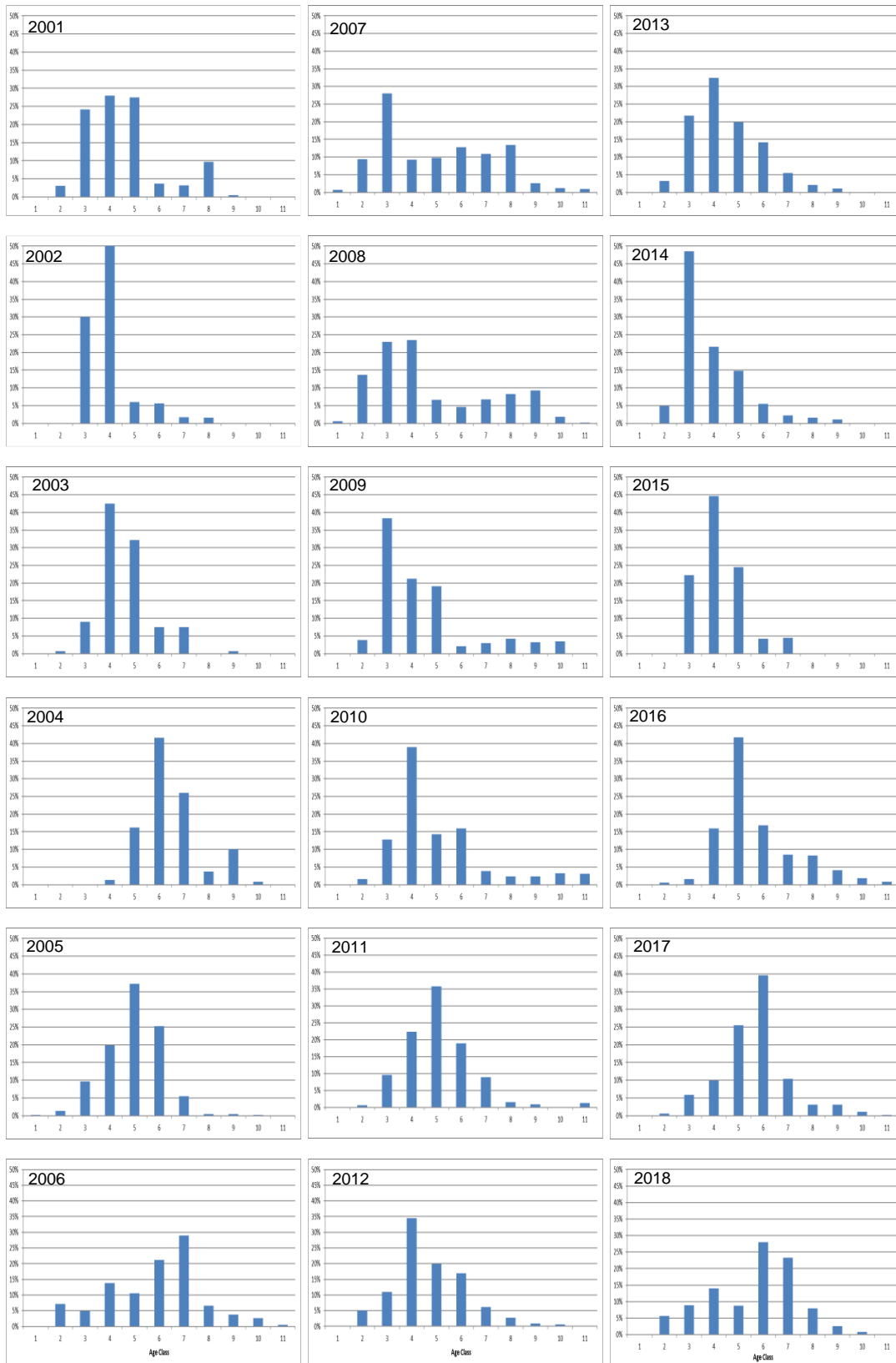
Figure B-1. Estimated age-frequencies of barramundi from the Mitchell River influence region between 2001 and 2018. Figure reproduced from Robins et al. (2021).
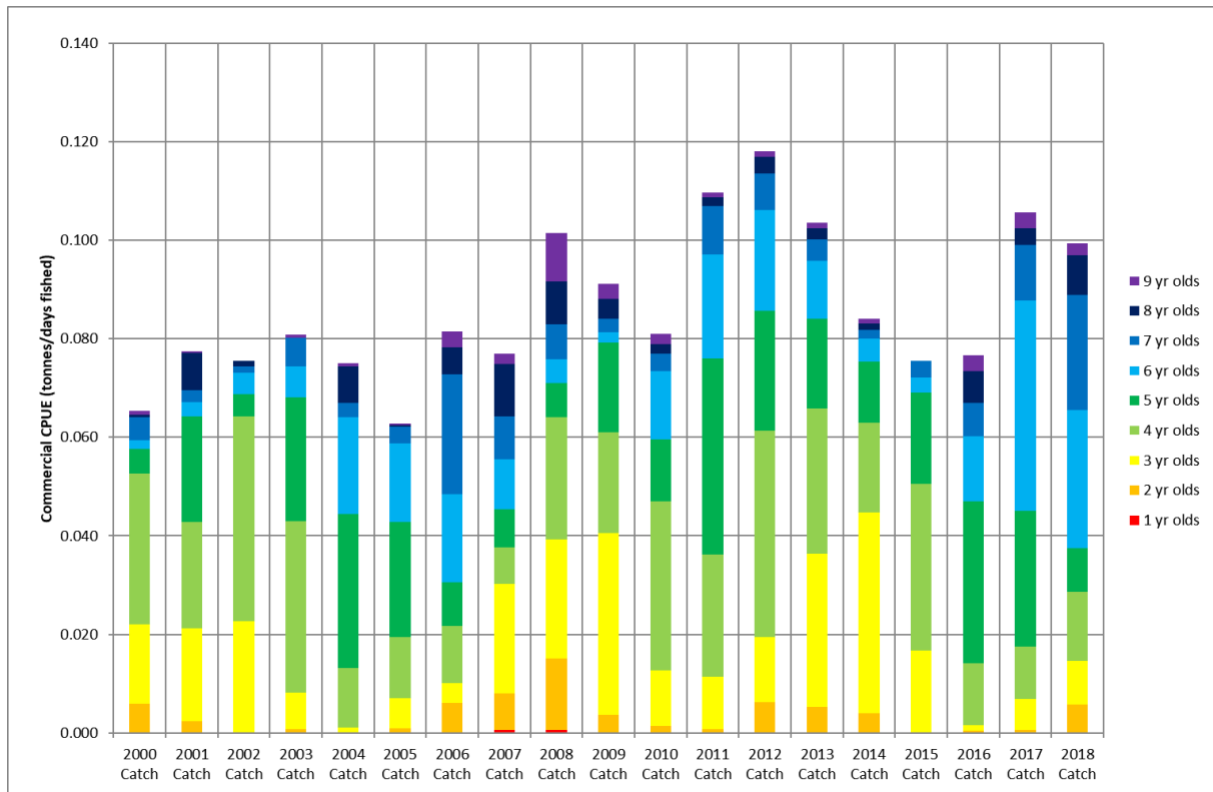
Figure B-2. Estimated age composition of the commercial barramundi catch per year (adjusted for effort) for stock in the Mitchell River influence region. Figure reproduced from Robins et al. (2021).

Appendix C. Fish species

Table C-1.Catch per unit effort (number of individuals per electrofishing power-on second) for each species at each site.

Species name	Species abbreviation	9190015	9190011	919205a	9192008	9193014	9193022	9193028	9190010	9192007	9193007	9190043
Ambassis macleayi	amb mac	0.00	0.03	0.00	0.06	0.05	0.98	0.00	0.14	0.40	0.02	0.00
Amniataba percoides	amn per	1.03	0.25	1.21	0.06	0.04	1.03	0.00	0.01	0.00	0.02	0.00
Anodontiglanis dahli	and dah	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.14	0.01
Brachirus selheimi	bra she	0.03	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.01
Craterocephalus stercusmuscarum	cra ste	0.01	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.00
Glossamia aprion	glo apr	0.01	0.00	0.00	0.11	0.02	0.02	0.13	0.01	0.00	0.00	0.00
Glossogobius giurus	glos spp	0.00	0.06	0.00	0.11	0.07	0.16	0.00	0.02	0.02	0.00	0.04
Hephaestus carbo	hep carb	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00
Hephaestus fuliginosus	hep ful	0.20	0.04	0.00	0.06	0.07	0.09	0.12	0.08	0.05	0.12	0.17
Lates calcarifer	lat cal	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.10	0.06	0.04	0.07
Leiopotherapon unicolor	lei uni	0.09	0.04	0.08	0.00	0.01	0.04	0.11	0.05	0.01	0.02	0.10
Megalops cyprinoides	meg cyp	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
Melanotaenia splendida inornata	mel ino	1.43	0.00	0.00	0.17	0.00	0.00	0.45	0.30	0.20	0.10	0.07
Nematalosa erebi	nem ere	1.13	0.05	1.38	0.00	0.00	0.46	0.00	0.22	0.08	0.37	0.94
Neoarius graeffei	neo gra	0.26	0.03	0.00	0.00	0.00	0.11	0.00	0.00	0.03	0.02	0.01
Neoarius midgleyi	neo mid	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.06	0.01
Neosilurus ater	neo ate	0.03	0.00	0.00	0.00	0.01	0.02	0.03	0.02	0.02	0.29	0.00
Neosilurus hyrtlii	neo hyr	0.00	0.00	4.62	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00
Oxyeleotris lineolatus	oxy lin	0.03	0.10	0.04	0.39	0.07	0.16	0.00	0.16	0.39	0.18	0.06
Oxyeleotris selheimi	oxy sel	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.02	0.00	0.01
Pingalla gilberti	pin gil	0.00	0.00	0.00	0.00	0.01	0.04	0.00	0.01	0.00	0.00	0.05
Porochilus rendahli	por ren	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Scleropages jardinii	scl jar	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Scortum ogilbyi	sco ogy	0.00	0.03	0.04	0.00	0.00	0.02	0.00	0.12	0.08	0.12	0.02
Strongylura krefftii	str kre	0.00	0.03	0.00	0.00	0.00	0.02	0.00	0.06	0.03	0.02	0.02
Toxotes chatareus	tox char	0.00	0.01	0.12	0.00	0.01	0.02	0.00	0.18	0.07	0.25	0.25

Species name	Species abbreviation	9191006	919309a	9190083	9190086	9190036	919009a	9190087	9190035	9192007	9190073	9190011
Ambassis macleayi	**amb mac**	0.01	0.02	0.88	0.01	0.00	0.06	0.05	0.04	0.04	0.03	0.00
Amniataba percoides	**amn per**	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.03	0.00	0.00	0.02
Anodontiglanis dahli	**and dah**	0.00	0.00	0.00	0.05	0.02	0.00	0.00	0.00	0.00	0.00	0.00
Brachirus selheimi	**bra she**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craterocephalus stercusmuscarum	**cra ste**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.00
Glossamia aprion	**glo apr**	0.05	0.03	0.03	0.01	0.01	0.02	0.02	0.00	0.00	0.03	0.00
Glossogobius giurus	**glos spp**	0.02	0.15	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.01
Hephaestus carbo	**hep carb**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Hephaestus fuliginosus	**hep ful**	0.00	0.06	0.00	0.00	0.00	0.04	0.00	0.02	0.00	0.00	0.01
Lates calcarifer	**lat cal**	0.01	0.00	0.00	0.00	0.04	0.03	0.06	0.02	0.02	0.00	0.00
Leiopotherapon unicolor	**lei uni**	0.01	0.12	0.02	0.01	0.00	0.01	0.00	0.06	0.02	0.00	0.02
Megalops cyprinoides	**meg cyp**	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.04	0.00	0.00
Melanotaenia splendida inornata	**mel ino**	0.17	0.00	1.37	0.02	0.01	0.23	0.12	0.00	0.00	0.43	0.02
Nematalosa erebi	**nem ere**	0.61	0.00	0.00	0.02	0.01	0.08	0.02	0.15	0.00	0.00	0.00
Neoarius graeffei	**neo gra**	0.01	0.00	0.00	0.04	0.04	0.02	0.00	0.02	0.00	0.00	0.00
Neoarius midgleyi	**neo mid**	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
Neosilurus ater	**neo ate**	0.00	0.05	0.00	0.00	0.00	0.01	0.00	0.02	0.01	0.00	0.00
Neosilurus hyrtlii	**neo hyr**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Oxyeleotris lineolatus	**oxy lin**	0.18	0.20	0.16	0.11	0.07	0.04	0.08	0.09	0.10	0.00	0.02
Oxyeleotris selheimi	**oxy sel**	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pingalla gilberti	**pin gil**	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
Porochilus rendahli	**por ren**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Scleropages jardinii	**scl jar**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Scortum ogilbyi	**sco ogy**	0.05	0.00	0.01	0.00	0.02	0.03	0.04	0.06	0.06	0.00	0.00
Strongylura krefftii	**str kre**	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.01
Toxotes chatareus	**tox char**	0.02	0.00	0.01	0.06	0.00	0.01	0.01	0.00	0.02	0.02	0.00

Appendix D. Fish movement

Table D-1. Percentage of fish identified as migrants at each site, including the number (n) of large fish (fish >10 cm) collected from each site. Sites M30.a and M30.b represent Yelko Waterhole samples from 2018 and 2019, respectively, and Sites M36.a and M36.b represent Sandy Creek samples from 2018 and 2019, respectively.

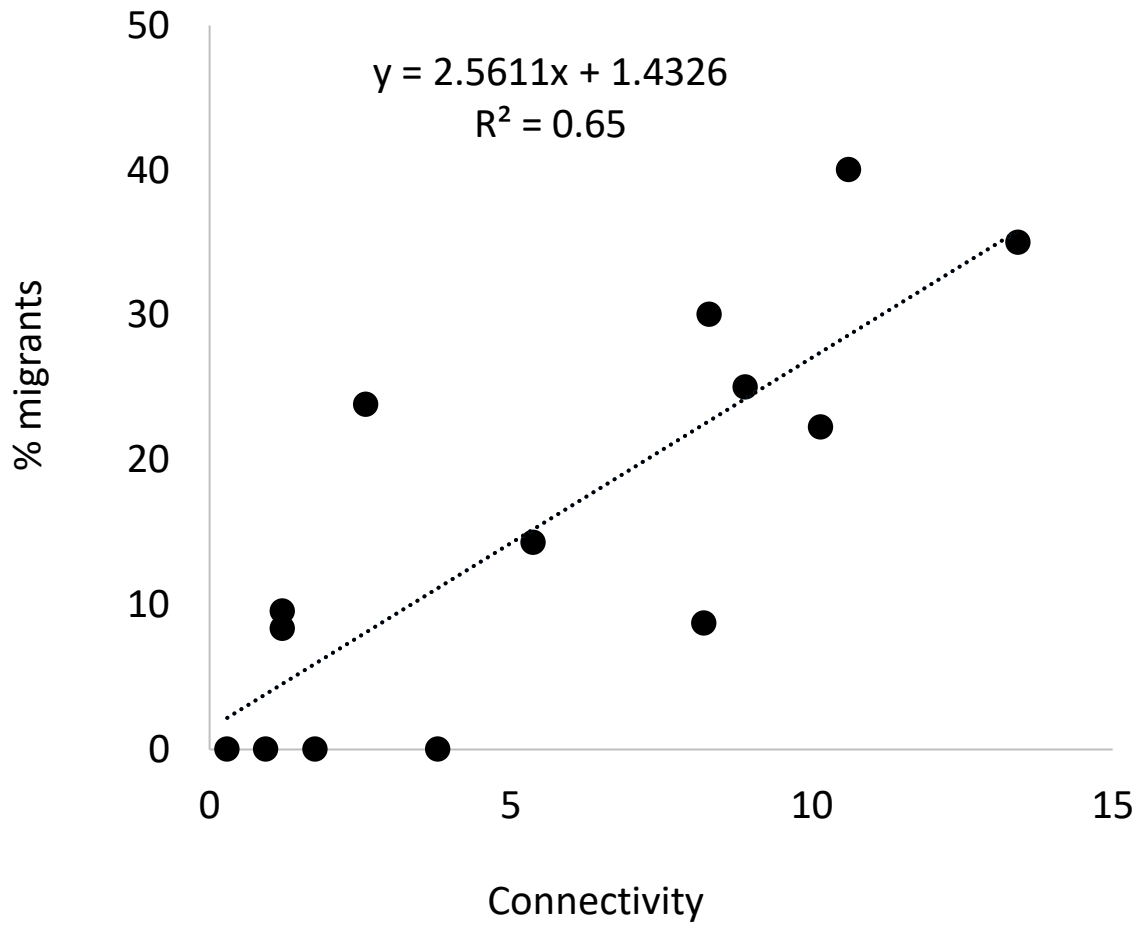| Site	% migrants	n large fish
M7	0	14
M15	0	13
P1	8	12
P3	0	11
W5	24	21
W4	10	21
W3	0	8
M19	35	20
P5	9	23
W1	30	20
M27	14	21
A1	0	17
W6	22	9
M30.a	30	10
M36.a	17	12
M31	40	15
M28	9	11
M29	25	16
P6	11	9
M36.b	0	3
M24	0	3
M35	0	1
M30.b	67	3
M34	17	6

Figure D-1. Relationship between mean site connectivity (weighted by slope, distance between sites, and distance to river mouth) and percentage of migrants at the same site, for river-channel sites.
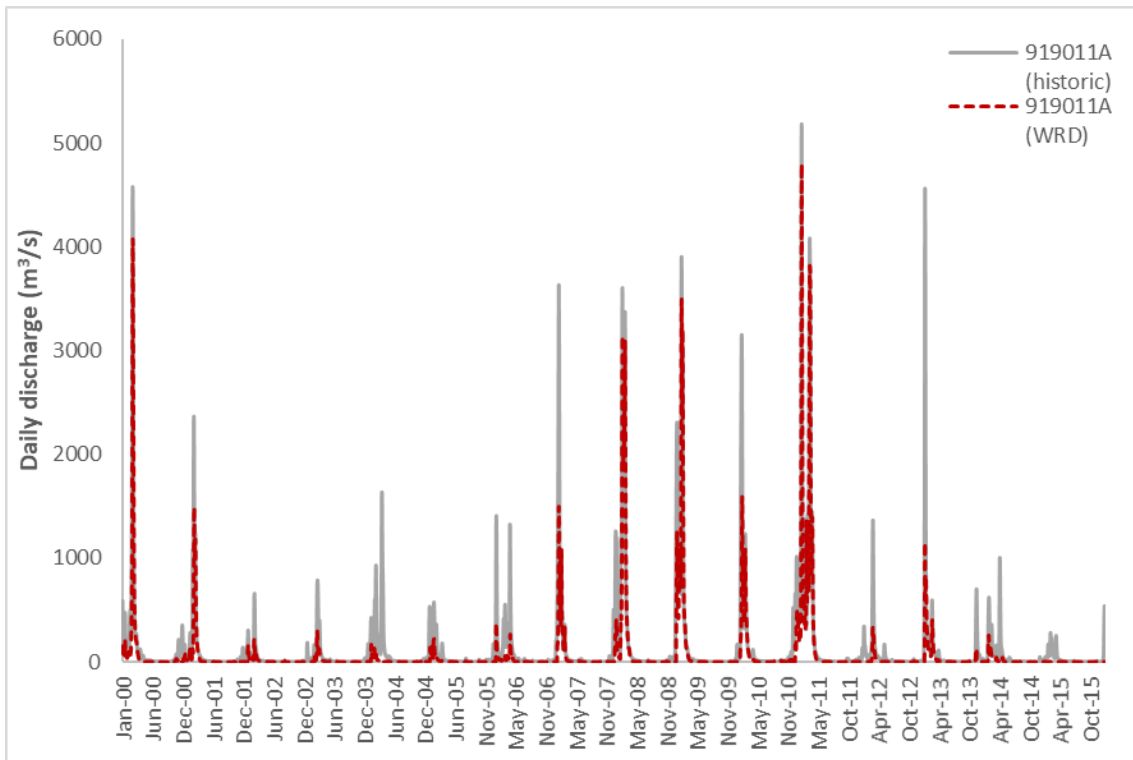
Appendix E. Modelled flow

Figure E-1. Modelled flows rate data at 'Mitchell River at Gamboola (919011A)' gauging station under historical and three-dams scenarios (Table 2-2).
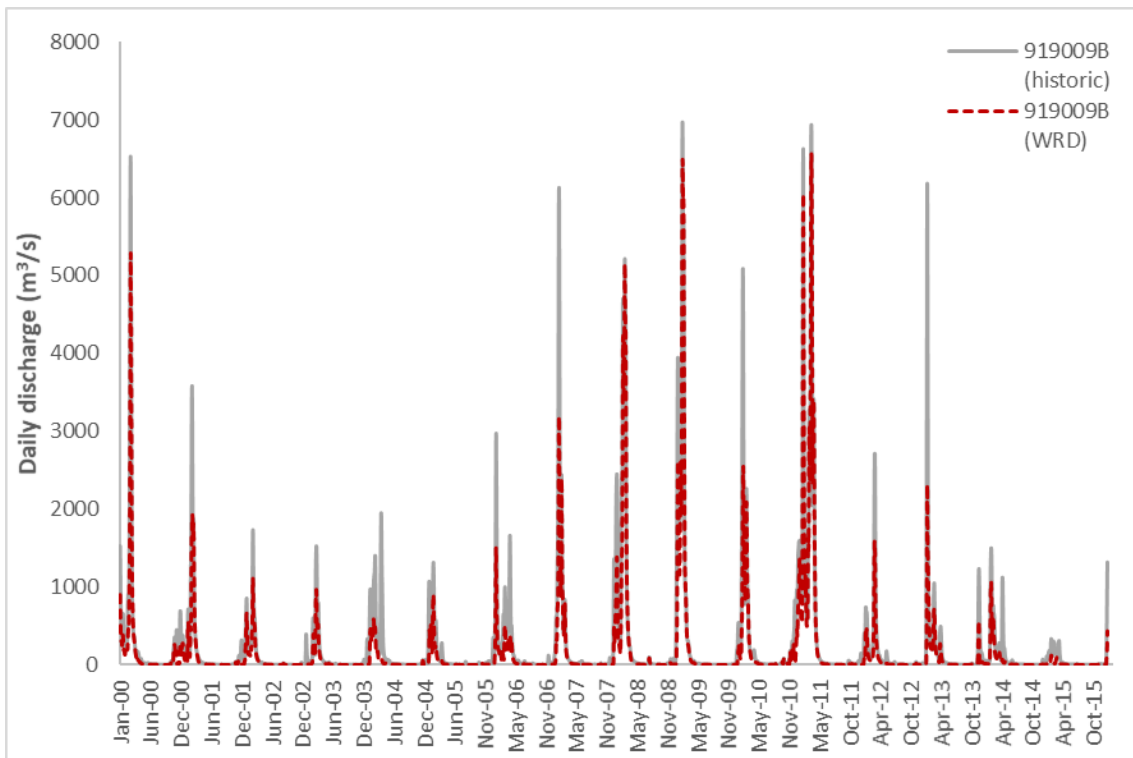

Figure E-2. Modelled flows rate data at 'Mitchell River at Dunbar (919009B)' gauging station under historical and three-dams scenarios (Table 2-2).
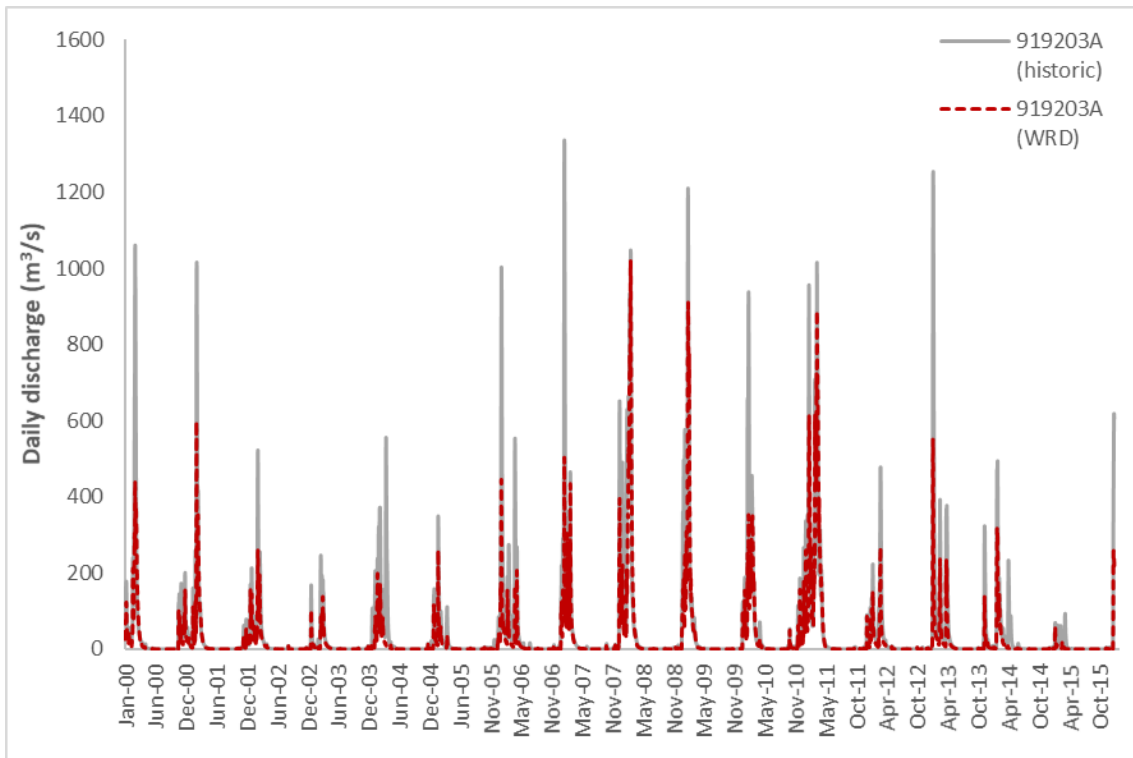
Figure E-3. Modelled flows rate data at 'Palmer River at Strathleven (919203A)' gauging station under historical and three-dams scenarios (Table 2-2).
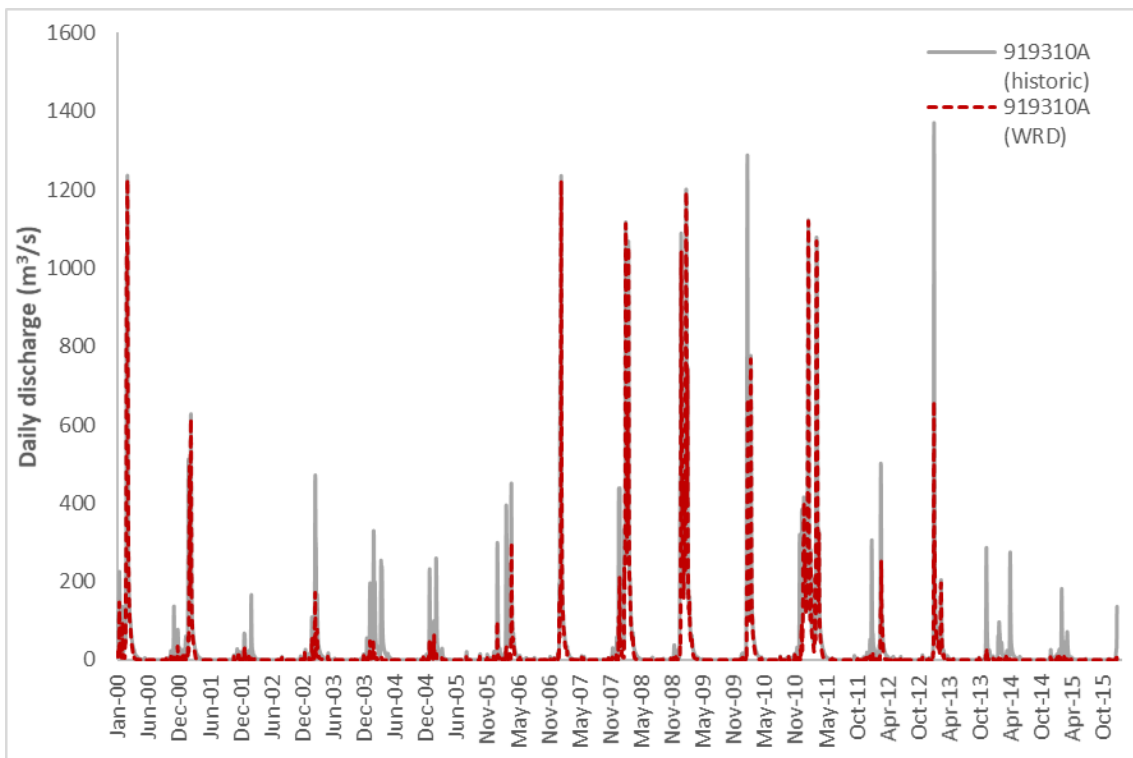

Figure E-4. Modelled flows rate data at 'Walsh River at Rookwood (919310A)' gauging station under historical and three-dams scenarios (Table 2-2).

Appendix F. Fatty acids and energy flow through food webs and across habitats

Table F-1. Euclidean distances between food web groups at each site. Food web groups: AM = aquatic macroinvertebrates, AP = aquatic plants, F = fish, TM = terrestrial macroinvertebrates, TP = terrestrial plants.

| Habitat type	Site	Site	F-AM	F-TM	AM-AP	AM-TP	TM-AP	TM-TP
Upper river channel	Walsh River D/S junction with Bulluburrah Creek	W3	14		19			
	Mitchell River at Gordon Arthur Crossing	M15	10	38	17	45	33	40
River channel	Mitchell River at Hughes Crossing	M23	10	31	22		23	
	Mitchell River at Shelfo	M31	11	40	20		30	
Floodplain creek	Sandy Creek	M36.a	15	30	16		26	
	Sandy Creek	M36.b	15		14	37		
	Magnificent Creek	M35	13	33	23	30	27	22
Floodplain wetland	Yelko Waterhole	M30.a	18	37	19	49	32	55
	Yelko Waterhole	M30.b	19		29			
	Leichardt Waterhole	M29	14	32	22	36	24	20
	Twelve Mile Lagoon	M24	20	41	15	19	22	32
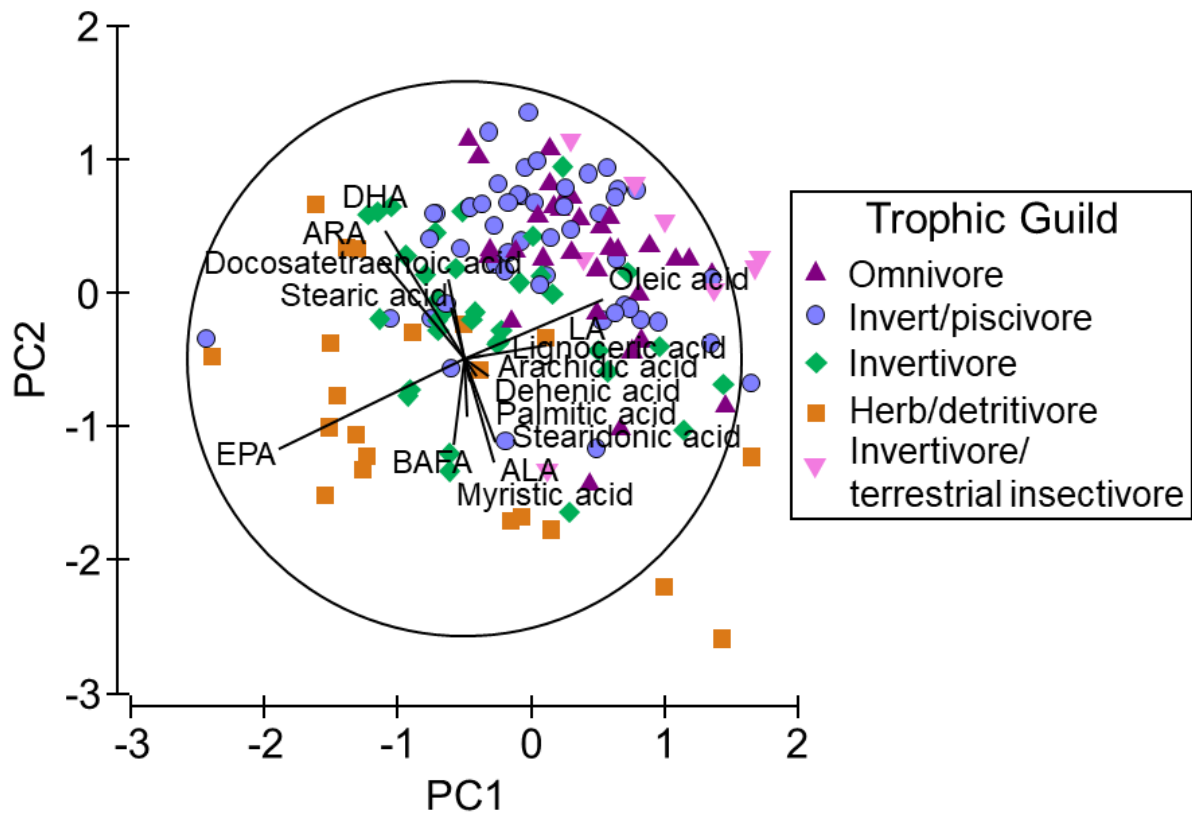
Figure F-1. Fish fatty-acid composition plotted based on principal components multivariate analysis. Grouping of fish by trophic guild indicates that selective feeding influenced predator–prey similarities in fatty-acid composition. ALA = alpha-linoleic acid, LA = linoleic acid, EPA = eicosapentaenoic acid, DHA = docosahexaenoic acid, ARA = arachidonic acid, and BAFA = bacterial fatty acid.