

National Environmental Science Programme

Repairing and conserving Australia's saltmarshes and seascapes

Colin Creighton, Terry Walshe, Ian McLeod, Vishnu Prahalad, Marcus Sheaves and Matt Taylor

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A Report for the National Environmental Science Program, Marine Biodiversity Hub

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Keywords

Restoration Ecology; Ecosystem Repair; Salt Marsh; Seascapes; Prawns; Net Primary Productivity.

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EXECUTIVE SUMMARY

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Australia's coastal marine biodiversity and accompanying benefits such as fisheries have been markedly reduced due to loss of essential inshore habitats. These coastal habitats provide a nursery ground for a multitude of animals, including fish, prawns and birds. Many species depend on inshore habitat during critical early life-stages characterised by rapid growth and development (coastal dependency). These coastal tidally linked wetlands (commonly known as 'seascapes') comprise mangrove-lined channels, salt marshes, mud flats, sedge lands and sub tidal and inter tidal channels and gutters.

Much of the losses in habitat have occurred as a function of Australia's development for urban, infrastructure and agricultural uses - all of benefit to Australia's economy. While better planning and management could have reduced the impact on our coastal marine biodiversity much of the task ahead for Australia is about land use optimisation - seeking the best patterns of use and management that will maximise overall community benefits. Essentially re-creating or repairing key components of coastal habitat for benefits whether it is commercial or recreational fishing, water quality, biodiversity or carbon sequestration. If these restoration efforts are planned and implemented carefully Australia can maximise benefits from its coastal resources while achieving urban, infrastructure and agricultural development.

Optimisations by repair and improved management to maximise coastal benefits is gaining momentum internationally. In Australia this is occurring through the work of multiple community interest groups - recreational and commercial fishers, conservation groups, bird observers and natural resource management organisations to name a few. In many cases their work is encouraged by local, state and federal government agencies,.

In a modern landscape where there is competition between land uses for scarce resources, such as occurs for virtually all Australia's coasts, benefit statements are essential. The types of questions often asked include:

- what benefit will accrue from repair?
- what are the nature of these benefits?
- what is the break even point for investment in repair or when will the economic benefits of repair exceed the costs?
- what are the current benefits that will be foregone?

This research project has attempted to answer the first two of these questions for three broad landscapes - coastal tropical, subtropical and temperate. The project recognised that to accurately answer all four questions it is best undertaken on a location-by-location basis. Therefore the broad goal of this project was to develop scientific information using key indicator-species that can be readily valued financially. Fish and crustaceans have a clear economic marketplace value as a food so the focus was on selected fish and crustacean species. Monetary priced benefits and estimates of increase in monetary benefit in the seafood marketplace can be readily verified for any increase in productivity for taxa such as prawn species because of their clear economic worth.

Tables 1 and 2 in chapter 1 summarise the approach taken in the three case study sites. Each of the broad climatic landscapes have differing features in terms of their natural attributes, beneficial species, current protection arrangements and management activities. NSW the subtropical case study certainly has the most advanced suite of features – saltmarshes are recognised as an endangered habitat, protection is in place for what remains and NSW has a very active recreational and commercial fishing lobby, including levy funds and supportive agencies. At the other end of the spectrum Tasmania has no protection or management systems in place and

indeed up till this project there has been virtually no information of the likely fisheries productivity of these coastal landscapes.

Chapter 2 is a descriptive chapter that investigates the most appropriate ways to make estimates of secondary production in saltmarsh systems. Chapters 3-6 summarise the findings from the three case-studies. Chapter 3 (Tropical case study) details the value of estuarine habitats to banana prawns in North Queensland. Chapters 4 and 5 (Subtropical case-studies) explore the benefits of wetland restoration to School Prawns in New South Wales, from a modelling and field-based perspective respectively. Finally, Chapter 6 (Temperate case-study) describes the value of wetland restoration to Tasmanian fish populations. Each chapter is written in the form of a draft scientific article ready for peer review. This has been intentionally done as there is a paucity of relevant literature and a clear need to communicate rapidly and with a sound scientific basis to a whole range of stakeholders. These four chapters stand alone, detailing findings and benefits for the three climate landscapes. Each can be applied across their specific climate landscapes to aid in the discussions and deliberations towards understanding the benefits of protection and repair of wetlands. Furthermore, they can provide sufficient indicator information of likely financial benefits that can then be factored into investment planning such as break-even analysis.

Chapter 7 brings the three case studies together, articulating likely benefits and opportunities, both financial and non-financial. It concludes with a call for action, recognising we may never have the most precise scientific estimates of likely improvement flowing from coastal marine biodiversity repair but we do know the benefits will be substantial.

The next phases of this work as can be expected vary for each climate landscape. For coastal tropical landscapes of the Great Barrier Reef the Reef Plan 2050 already calls for the repair where possible of these coastal marine seascapes. The next task is to bring together the multiple layers of GIS-based mapping on coastal seascapes and determine which subset of these are the most prospective for repair. For subtropical seascapes the next phase will to be to work on the entire Clarence Estuary. This is the biggest and most productive estuary in NSW and worthy of detailed multi-disciplinary research across ecology, hydrology, flood management, economics and social issues to develop a draft blueprint for repair at the whole of estuary scale. The first task for restoring Tasmanian temperate seascapes is most probably legislative - to gain formal recognition and protection of existing remnants. In parallel, especially for areas such as the case study, repair works could start immediately, benefitting coastal marine biodiversity at no cost to other land uses.

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CHAPTER 1 – BUILDING A FUNCTIONAL UNDERSTANDING OF COASTAL ECOSYSTEMS

1.1 Definitions

Saltmarshes generally refers to the mosaic of coastal wetland vegetation types that occupy areas of low energy with intermittent tidal inundation, typically in bays, inlets and estuaries, on sheltered soft substrate foreshores, often occurring behind mangroves.

Seascapes is a broader more functional definition for the mosaic of intertidal, sub-tidal and supra-tidal areas that include channels, gutters, mud flats, mangrove clumps and fringes lining channels and various communities of salt marsh vegetation. The seascapes concept conceptualizes a nursery as "a spatially explicit seascape consisting of multiple mosaics of habitat patches that are functionally connected" by Nuagelkerken et al (2015) to foster investigation and analysis of the functional attributes of these important ecosystems. These seascapes serve multiple functions including sediment trapping, nutrient cycling, dissipation of wave energy, fish and prawn nursery, carbon sequestration and resting / feeding areas for birds. The nursery concept here refers to the lifecycle of most of our inshore species – having the early part of their life history within the sheltered waters of estuaries, embayments and wetlands.

1.2 The biological and ecological function of seascapes

Seascapes are functionally part of the continuum that drives coastal ecological productivity (e.g. Laegdsgaard 2006; Victorian Saltmarsh Study 2011; Saintilan and Rogers 2013; Creighton et al. 2015), by linking mangroves to freshwater wetlands. The sustainability of seascapes relies on tidal inundation and fresh water inputs from rain, groundwater flows and rivers. Inputs of fresh water can influence soil and water salinities and the nutrient dynamics and movement of sediments (e.g. Victorian Saltmarsh Study 2011). Their provision of ecosystem services relies on connectivity between land and sea being maintained so both fresh water and tidal water have adequate opportunities to meet. Barriers to connectivity occur along almost every river and estuary in the more populated parts of Australia (Creighton et al. 2015). Reinstating connectivity for biological, chemical and hydrological fluxes is key to re-establishing net primary productivity.

Soils and vegetation

Seascape sediments generally consist of poorly sorted anoxic sandy silts and clays. Carbonate concentrations are low, concentrations of organic material are high and the sediments may have salinity levels that are much higher than that of seawater. Acid sulfate soils have the potential to become oxidised, releasing significant amounts of sulphuric acid into coastal waterways leading to chronic poor water quality and fish kills (Creighton 2013).

Vegetated saltmarsh-communities can be dominated by a single plant species or occur as a mosaic, with the diversity of plant species increasing in higher elevated areas that are less frequently inundated. The combination of salinity, elevation and inundation that is responsible for many of the patterns seen in the distribution of saltmarsh plant species has been extensively studied (examples include Adam 1981a; b; 1990; King 1981; Zedler et al. 1995; Streever and Genders 1997). Zonation patterns of vegetation in saltmarshes has been described in detail (e.g. Zedler et al. 1995; Streever and Genders 1997). Vegetation is usually zoned parallel to the shoreline, and there is a general broad scale zonation from the estuary landward. The zones can be described as lower, mid and upper levels, usually each with a distinct mosaic of species that is often complicated by small-scale patchiness. Succulents dominate the lower marsh (e.g. *Sarcocornia* spp.), while the mid-marsh usually contains species such as *Sporobolus* spp. and *Samolus* spp. The upper marsh is a mosaic of species including *Juncus kraussii* and *Baumea juncea*. The area behind the upper marsh under natural conditions can be brackish to fresh back swamps dominated by sedges and

casuarinas (e.g. *Casuarina glauca*) in the saltier, sometimes tidally inundated areas transitioning to melaleucas (e.g. *Melaleuca quinquenerva*) and various reeds and sedges (e.g. *Phragmites australis*) in the fresh swamps and then eucalypts and angophoras (e.g. *Angorphora costata*).

Coastal water quality and net primary productivity

Seascapes, as are all coastal wetlands, are sometimes described as 'nature's kidneys', helping to maintain water quality by collecting, assimilating and recycling nutrients and contaminants from runoff (reviewed in Creighton 2013). As water flow slows, sediments and chemicals drop out of the water column, dissolved oxygen levels increase and nutrient levels reduce. High rates of productivity lead to high rates of mineral uptake, and decomposition processes take place in wetland sediments. Seascapes remove nitrates and phosphates processing and transferring these inputs into biological outputs such as diatoms and phytoplankton – essential components of the coastal and marine food chain.

Seascapes also perform necessary water quantity functions, moderating the rate of catchment runoff. By reducing strength of flow and ensuring a more dampened water flow hydrograph these brackish ecosystems are key to fostering high levels of coastal net primary productivity. Indeed a brackish estuarine system linked to both fresh and tidal water is one of the world's most productive ecosystems. In parallel with their role in moderating and slowing catchment runoff, seascapes with their often-sandy sediments, gutters and channels play a role in coastal groundwater recharge. This again results in longer return periods and expanses of brackish water between fresh and seawater, leading to highly productive ecosystems are assimilating nutrients and translating them into the basic building blocks for net primary productivity, such as diatoms and phytoplankton.

Primary productivity, food chains and biodiversity

Seascapes by virtue of being highly productive linked ecosystems and communities provide substrate, shelter and food for a diverse range of species including fish, invertebrates, mammals, birds and plants. Soils of saltmarshes contain a lot of decomposing plant material that feed a wide range of organisms in food webs, from bacteria and fungi through to mammals (Victorian Saltmarsh Study 2011). The production of biomass in these ecosystems can be over three times higher than in a terrestrial ecosystem. The above ground net primary productivity of Australian coastal saltmarsh plants has been estimated to range between 3-13 tonnes ha⁻¹ year⁻¹ (Congdon and McComb 1980; Clarke and Jacoby 1994; as cited in Victorian Saltmarsh Study 2011).

Specialised plants provide the foundation for food chains in seascapes. As plants die and decompose, bacteria and fungi break down the plant detritus, converting it into carbohydrates and proteins that are more easily digestible by crabs, finfish and filter feeders such as oysters or mussels. Algae also play a significant role in fuelling the biota and high productivity of these ecosystems. In addition, the saltmarsh component sequesters significant quantities of carbon both in plants and in the sediment below them. Colloquially part of 'blue carbon', saltmarshes are among the highest carbon sequesters of all vegetation communities (Lawrence et al 2012). Protection of existing saltmarsh ensures that this accumulated carbon is not released as carbon dioxide into the atmosphere. Repairing saltmarshes, mangroves and seagrasses will add to the volume of carbon sequestered in the Australian landscape. "Blue carbon" unfortunately is not yet part of Australia's National Carbon Accounts so no Australia-wide estimates of sequestration budgets and fluxes is available.

Nursery habitat

Seascapes provide habitat and shelter and therefore often act as nursery grounds for many commercially important fish, as well as for fish and crustacean species that are part of the coastal and marine food chain. Commercial and recreationally important species that exploit seascapes in their nursery phases include yellowfin bream (*Acanthopagrus australis*), dusky flathead (*Platycephalus fuscus*), sand whiting (*Sillago ciliate*), several mullets species (e.g. sea mullet, *Mugil cephalus*), garfish (*Arrhamphus sclerolepis*), mulloway (*Argyrosomus japonicus*]) eels and many crustaceans such as mud crabs (*Scylla serrate*) and

prawns (e.g. School prawn, *Metapenaeus macleayi*; Banana prawn, *Fenneropenaeus merguiensis*; and western school prawns, *Metapenaeus dalli*) (Daly 2013, Creighton 2013).

The multiple habitats and mosaic patterns of seascapes function as nursery grounds and unique feeding and habitat opportunities for larvae, several species of threatened micro-bats and birds such as migratory shorebirds (Saintilan and Rogers, 2013). Birds are diverse including foraging rails, crakes, plovers, stilts, avocets, ibis, egrets and ducks, and roosting swans, cormorants and pelicans (e.g. Land Conservation Council 1993 as cited by Victorian Saltmarsh Study 2011). Migratory birds protected under federal legislation and international treaties (e.g. China Australia Migratory Bird Agreement or CAMBA and Japan Australia Migratory Bird Agreement or JAMBA) roost and feed in the multiple micro-habitats provided.

Shoreline protection

Seascapes protect shorelines from erosion by buffering wave action and trapping sediments. They reduce flooding by slowing and absorbing rainwater and protect water quality by filtering runoff, and by metabolising excess nutrients (e.g. <u>http://oceanservice.noaa.gov/facts/saltmarsh.html</u>). Seascapes protect estuary foreshores by dissipating the energy of wind and wave action and providing a natural buffer that helps minimise erosion (e.g. Moller et al., 1996). With predicted increases in storm surge intensity and rising sea levels associated with a changing climate, these landscapes will become increasingly important in protecting estuary foreshores (Creighton 2014).

1.3 Habitat loss in seascapes

Seascapes are threatened by virtue of their coastal location and close proximity to urban areas. Their flat profile, apart from the various gutters and tidal channels, makes them amenable to being filled and drained, and converted to sports fields, houses, and canal- or industrial estates. Saltmarshes have furthermore been drained, sometimes filled and levees constructed to exclude tidal inundation within coastal floodplain environments, as part of land development for agriculture (e.g. cane lands) or grazing (e.g. ponded pastures)

Loss of seascapes is one of the key-contributing agents to the loss of amenity and condition of our coastal resources. An Australia-wide assessment of 1000 estuaries and embayments undertaken by the National Land and Water Resources Audit of 1997-2002 (National Land and Water Resources Audit 2002) indicated that 30% were modified to some degree. The most highly degraded were in New South Wales, where 40% were classified as 'extensively modified' and 10% were 'near pristine'. Since that review, urban populations have continued to grow rapidly, and increasing pressures for industrial and agricultural development in the coastal zone have resulted in ongoing degradation of Australia's estuaries and embayments. This degradation has had serious effects on biodiversity, carbon sequestration (e.g. Lawrence et al 2012) and commercial and recreational fishing (Creighton et al 2015).

Specific quantitative information on the loss of critical habitat is available from a number of habitat- or region-specific studies to expand upon the National Land and Water Resources Audit's (2002) Australiawide assessment. Saintilan and Williams (2000), for example, reviewed the record of loss of coastal saltmarsh in eastern Australia since World War 2, and reported losses as 100% for parts of Botany Bay, New South Wales over the period 1950-1994 and 67% for the Hunter River (excluding Hexham) from 1954-1994. Harty and Cheng (2003) reported a loss of 78% of saltmarshes in Brisbane Water, near Gosford, New South Wales, between 1954 and 1995. Sinclair and Boon (2012) showed that the state-wide loss of coastal wetlands (mainly mangroves and seascapes) in Victoria since European colonisation has been variously 5-20% by area across the state, with the greatest losses occurring in heavily urbanised areas such as around Port Phillip Bay (~50% loss) and in agriculturally developed regions such as Gippsland (e.g. 60% loss from Anderson Inlet in South Gippsland).

As noted under the National Land and Water Resources Audit (2002) and the National Vegetation Information System, there is no consistent and functionally Australia-wide map of saltmarsh landscapes,

either pre 1770 or current. This was detailed in Creighton et al. (2015; Phase I of this project) when attempts were made to collate areas of prior and current saltmarsh in the case study States.

Historically, vegetation mapping has usually been undertaken at a State or regional scale and has varied in attributes State by State. Often mapping has been strictly from a botanical perspective, nominating key plant species or sometimes complexes of species. Areas such as fringing mangroves, tidal gutters and mud flats that are functionally part of the seascapes have been omitted from the prior mapping. From a net primary productivity or biodiversity perspective virtually all this prior mapping is only marginally useful. Indeed the recognition of ecological function and biodiversity outcomes is key to our rationale for this project to explore these coastal systems from a seascapes perspective.



Human development can alter saltmarshes and mangroves. Photo credit: Paul Boon

Shifting baselines of productivity for estuaries

It follows that with the losses of habitat, connectivity, tidal flows and changes to catchment hydrology there have been substantial reductions in estuary productivity. Creighton (1984) explores this concept for the Camden Haven and proposes substantial investment in repair actions (Creighton 2015).

Pauly (1995) contends shifting baselines are key to understanding how fisheries management has failed. Essentially this syndrome according to Pauly has arisen because each generation of management accepts as a baseline the stock size and species composition as known early in their careers and uses these perceived baselines to evaluate management actions. With ongoing stock and species decline the result is a gradual shift of the baseline, an accommodation by management of the disappearance of species and acceptance of the reduction of net primary productivity. This then adversely impacts on management decisions and most importantly perceptions and motivations for repair of fisheries habitats especially essential nursery areas and conditions such as seascapes and tidal flows and fluxes.

1.4 Links to Government policy objectives

Australian and state Government policy and regulations have recognized the importance of these coastal landscapes. Examples include:

- EPBC Act Subtropical and Temperate Coastal Saltmarsh Threatened Community Recovery Plan;
- Capricorn (Dawson) subspecies of the yellow chat (*Epthianura crocea macgregori*) is listed as "Critically Endangered". Salt marsh in the Fitzroy River Delta and Torilla Plains is critical for breeding, feeding and shelter for this species.
- Various State-based habitat retention and management legislation and policies

1.5 Phase 1 NESP Findings

For Australia's seascape landscapes information is readily available on vegetation types and the mix of species that make up the mosaic of wetlands from mangroves through the tidal height spectrum of intertidal, to supra-tidal and highest astronomic tides to the fresh water systems of sedges, reeds, and *Melaleuca* dominated wetlands. For most states, these vegetation types are reliably mapped and spatial data sets are readily available. Likewise, for most states the areas of wetland loss can be broadly estimated and the cause of loss detailed. Similarly, current threats to wetland integrity are well documented. In states such as New South Wales and Queensland, this information has also led to various mechanisms to protect remaining seascapes.

For all states, key sites for repair can be at least broadly identified. Likewise, the role of seascapes in marine net primary production, crustacean and fish nursery phases, carbon sequestration, nutrient assimilation, coastal foreshore protection, bird habitat and related ecological functions is broadly known and generally parallels that found globally. This synopsis also found that Australia cannot yet categorically ascribe levels of net primary production to seascapes or indeed estimate the biomass of high value, fecund, annual and high profile species such as prawns, let alone quantify food chains and the equally complex issue of quantifying their role in fish populations.

In brief, the findings of Phase 1 NESP project were:

- Salt marshes and seascapes generally are undervalued by the Australian community, as demonstrated by their alteration and by the general lack of understanding as to their role in estuary productivity such as prawn biomass;
- Substantial areas have been lost, especially to agriculture;
- In some areas urban development planning (often infrastructure such as road and rail) has not accounted for the need for tidal flows to ensure ongoing function and connectivity;
- The level of legal protection varies across states from substantial (e.g. Queensland and NSW) to nil (e.g. Tasmania);
- Mapping and spatial data sets have often focused on vegetation type rather than function. Definitions of vegetation type and wetland type are not always consistent across states;
- Changes such as percent loss of salt marshes since European settlement are well defined for some states such as Queensland, but not for others such as NSW;
- Ecosystem services will differ depending on the complex, the geography and estuary type;
- Likewise, articulating the key benefits as an input to business cases for repair will need to vary across states and localities;
- Therefore a 'one size fits all' approach communicating the value in terms of broad ecosystem services is unlikely to work;
- There are many sites for repair for each state and there is value in continuing to develop benefit statements than can be used as part of the decision-making processes towards protection and repair;
- This work is probably best done at the seascape level recognising the inter-dependence and inter-relationships of the various wetland vegetation types in estuary productivity from sub-tidal such as seagrasses to tidal such as mangroves, saltmarshes and mud flats to the fresh to brackish back swamps.

Given this assessment, key knowledge gaps and priorities for research and management identified in Phase 1 were:

- Build a more seascapes-level scale of knowledge and appreciation on the role, function and benefits of coastal wetlands;
- Improve understanding of the different benefits provided by these seascapes in different regions (e.g. quantifying fish production, shoreline protection, carbon sequestration);

- Develop tailored communication media based on the benefits provided by these seascapes with their different suites of vegetation communities comprising the mosaic commonly known as coastal wetlands;
- Focus on benefits and functions that are directly relevant to coastal stakeholders such as
 recreational and commercial fishers. For example, promotion of their role in fish production will
 most likely resonate with fishers and can be used to potentially engage these groups in seascapes
 conservation and protection. This also opens up resourcing opportunities to form public-private
 partnerships such as the various recreational fishing trust funds;
- Prioritise restoration sites in each state and undertake case studies on these priorities to foster an increase in repair investment;
- Develop other benefit statements as potential markets emerge (e.g. models for carbon sequestration that could be used in carbon markets).

Certainly, if a business case is to be devised for investment in seascape repair then quantification of their potential value and returns to the health and productivity of coastal and marine systems is essential. There are many competing demands on both public and private investment streams. Quantification of potential benefits using simple readily understood indices is essential so that the costs and benefits of investment in repair can be compared to other possible investment activities.

Readily identifiable and high value indicators of biomass such as prawns could be useful as an initial indicator of value. At a broader scale, quantification of all ecosystem services that these seascapes provide should be prioritised. However, this broader-scale quantification will require substantial resources. Accordingly, this second phase of the project will focus first and foremost on quantifying high value and readily understood indicators of benefits.

Phase 1 Outputs

Reports:

- Creighton C, Gillies CL and McLeod IM (Eds) (2015) Saltmarsh habitats: A synopsis to underpin the repair and conservation of Australia's environmentally, socially and economically important bays and estuaries. TropWATER Report 15/59, James Cook University, Townsville, 88 pp
- Five Regional Reports (GBR region, New South Wales, Victoria, Tasmania, South Australia and Western Australia)
- All reports available for download from: www.saltmarshrestoration.org.au
- •

Website

- www.saltmarshrestoration.org.au
- This website includes background information relevant to the project and an information Hub where people can download the NESP reports and other relevant documents.

Phase 2 Tasks

The primary objective is to provide information, facts and figures that can be used by those advocating the protection and repair of these important coastal systems (Table 1).

This involves

- 1. Estimating the benefits of seascape repair for an easily publicly understood indicator e.g. key prawn and fish species.
- 2. Undertaking this work in three case studies, wherever possible in parallel with potential repair works so that very concrete case studies are available to demonstrate the benefits of repair:

- Tropical Qld Dry Tropics, probably Burdekin / Cape Bowling Green region with its extensive seascapes, especially existing good condition seascapes to demonstrate likely repair benefits of areas that have been lost to ponded pastures and/or tidal impoundments
- Sutropical NSW's largest estuary the Clarence focusing on Lake Wooloweyah and both existing seascape landscapes such as parts of Micalo Island and the potential for repair of flood mitigated landscapes such as the extensive back swamp of Lake Wooloweyah's south western foreshore adjacent to Yuragir National Park
- Temperate Tasmania and probably involving natural areas as well as areas for potential repair such as Pitt Water.

Table 1: Summary of Activities

Activities		
Activities		
1. Quantify the benefits of saltm	arsh repair using praw	ns and fish as simple indicators
Collect data to address the knowledge gaps on the key ecosystem benefits provided by seascapes by quantifying examples of productivity	Tropical	Focus on banana prawns as an example of a key commercial and recreational species. Work involved evaluating the quality of productivity estimates that were possible and validating food web links that support productivity.
	Subtropical	Quantitative sampling of School Prawn within a restored wetland, to identify the potential recruitment subsidy that may be derived from similar habitat repair elsewhere. Apply this data in a fishery model to evaluate a habitat repair scenario for Lake Wooloweyah in the Clarence River, to quantify the potential economic outcomes that can be derived through the associated recruitment subsidy for School Prawn.
	Temperate	Fish sampled in large numbers (>80 fish per 100 m ²) in "altered" saltmarshes, including those behind naturally breached levees. This indicates potential fisheries benefits from planned breaching of further levees, in negotiation with land managers, to expand fish habitat and broader seascape productivity.
2. Build a summary Business Cas	e that articulates bene	efits and identifies opportunities for repair

Collate findings from these 3 case studies and integrate with proposed repair investments to assess the economic and environmental benefits of repair.

Rationale for the site selection of the 3 case studies

With limited resources in the NESP project, research was concentrated on Australia's east coast while still attempting to cover the range of ecological productivity, social and policy issues that will impact upon repair motivations within the community. The matrix summarizes the underpinning rationale to case study selection (Table 2).

Climate Zone	Case Study	Biophysical Productivity Knowledge	Community Awareness	Policy Framework
Tropical	Bowling Green,	Sufficient to compile	Ponded pastures and the	Saltmarshes, mangroves and
	North Qld	opportunities for	benefits of their removal	tidal channels protected
		productivity	already demonstrated [e.g.	under Fisheries Act. Major
		improvements [e.g.	Cape Bowling Green	investment in repair

		banana prawns]. The banana prawn example allows assessment of what productivity estimates are possible given current knowledge and available techniques. Other species yet to be well quantified in terms of productivity returns and will require more extensive data collection	National Park Biodiversity Fund project]. Local angling groups aware of the value of wetlands as are commercial fishers.	proposed under Reef Plan 2050.
Subtropical	Clarence, NSW	Limited knowledge of biological productivity and changes to net primary productivity of estuaries. Generally well developed knowledge of species life history, both crustacean and fin fish so that nursery habitat needs are well documented.	School Prawn are a highly valued commodity by both professional and recreational fishers. Momentum for seascape repair building and investment sources such as Recreational Fishing Funds increasingly interested in seascape repair.	Coastal saltmarsh habitat and associated ecological community is listed as an "endangered ecological community" under NSW Threatened Species Conservation Act. State Fisheries agencies identified "restore key habitat areas to enhance natural productivity within aquatic ecosystems" as a Key Result Area in strategic plan.
Temperate	Northern coast, Tasmania	Very limited knowledge of saltmarshes in relation to the seascapes – only recently mapped and documented. Almost no information available on fish use of saltmarshes.	Building awareness of saltmarshes and their seascapes from a biodiversity perspective. Improving awareness of their value for fisheries. Starting a conversation on saltmarsh and seascape management from a fisheries perspective.	No recognition of saltmarshes and their seascapes and their values within State legislation. Some protection afforded under the State planning regime, subject to enforcement

Outcomes sought and their application

- 1) Improved understanding of prawn / fisheries productivity and broader ecosystem benefits of seascapes in three contrasting communities (temperate, sub-tropical and tropical);
- 2) Quantification of easily understandable, ecosystem benefits (e.g. prawn production) provided by seascapes in which to communicate their value in terms of benefits to coastal stakeholders (e.g. recreational fishers) and the broader Australian public.
- 3) Communication resources that simply articulate the value of seascapes and their need for protection, conservation and repair.

This investment will provide base information required to inform and scope large-scale repair investment opportunities for Australia's most threatened coastal marine habitats. Equally importantly, it will be paralleled by investments in hands-on repair in a number of small to medium scale projects.

Estuary habitat repair is a developing area of community interest, especially in those states with recreational fishing license fees, as habitat improvements generally rank at the top of the investment priorities of recreational fishing communities. Importantly, 'habitat' is also the common ground between the recreational and commercial fishing sectors and environmental groups.

The practical outcomes this project will deliver include:

- 1. Improved understanding of the location, ecology and functional role of salt marshes and their role in supporting the ecological health and productivity of estuaries (documented in journal papers and in media);
- 2. Summary evidence of the productivity and economic benefits that are expected to be generated with repair;
- 3. Framework for data collection, analysis and reporting to accompany/evaluate any repair investment and best-practice restoration;
- 4. Provide an easily understandable demonstration of the benefits of seascapes and their repair to the community (e.g. prawn productivity).

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CHAPTER 2 - DEVELOPING ACHIEVABLE MEASURES OF FISHERIES VALUES FOR NORTHERN AUSTRALIA'S COASTAL WETLANDS AND ESTUARIES

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2.1 Executive Summary

Healthy estuaries and coastal wetlands (ECWs) and their habitats play vital roles in supporting coastal food webs and fisheries production, acting as critical feeding, nursery and reproductive areas for many important species. However, Queensland's ECWs are severely degraded due to impacts of a diversity of anthropogenic stressors. As a consequence, ECW function has been compromised by substantial losses of some of Queensland's most productive of aquatic habitats. Careful management and repair and revitalisation actions are therefore urgently needed. These actions need to be well targeted, carefully prioritised, and their success evaluated. Fundamental to this is the need to be able to **value** ECWs and their habitats. However, a number of gaps in the current scientific knowledge of northern Australian wetlands limit our ability to assess wetland value. These include the lack of understanding of the exact way wetlands and their habitats support important fisheries species, and how the values of wetlands and their habitats can be measured in robust and valid ways, so that fisheries benefits can be validly linked to wetland habitat function.

This study investigates how the value of coastal wetlands and estuaries can be measured in robust, valid and meaningful ways. These measures need to be relevant at the scale of unit or outcome to be evaluated, broadly meaningful and easy to communicate to end-users. Quality estimates of the **production** of exploited species are of particular value in a fisheries context. However, a substantial body of data are needed for the calculation of production estimates. **Biomass** estimates are more achievable and, as long as their limitations are understood, can provide useful measures of estuary or coastal wetland habitat value that are easily understood and easily communicated.

Most common sampling approaches are unsuitable for estimating density, the most fundamental component of fisheries biomass and production estimates. However, cast nets and beam trawls have proven effective for providing suitable data on penaeid prawns and bait fish in north Queensland estuaries, and have the potential to be developed into useful estimates of production per area of tropical estuary or coastal wetland habitat. Substantial data-sets of these types exist but additional research and development are required before such data can usefully be related to specific areas of estuary or coastal wetland. Because samples from methods suitable for larger species cannot be related to an area fished they cannot provide spatially explicit estimates but only estimates relative to the effort needed to catch the fish. If we are to fully account for the value of the different ECW habitats to fisheries, it is also important to understand the ecological context around the species-productivity and species-habitat linkages, and to consider all the variables that influence these linkages.

Substantial additional studies are required to produce workable and valid estimates of biomass and production that are truly representative.

2.2 Introduction

Background

Estuaries and Coastal Wetlands (ECWs) provide humans with a range of goods (e.g. food, construction materials), services (e.g. tourism, recreational) and cultural benefits (Barbier 2007). Importantly, they play vital roles in supporting coastal food-webs and fisheries production (Weinstein & Litvin 2016). For instance, in northern Australia, important fisheries species such as barramundi (*Lates calcarifer*), mangrove jack (*Lutjanus argentimaculatus*), banana prawns (*Fenneropenaeus merguiensis*) and mud crabs (*Scylla serrata*) are profoundly estuary dependent. This dependence results from the reliance of critical life history phases on habitats such as mangroves (Robertson & Duke 1987, Sheaves et al. 2007b), seagrass (Coles et al. 1987, Watson et al. 1993) and saltmarshes (Russell & Garrett 1983); on the occurrence of suitable environmental conditions in those habitats (Sheaves 1996); and on the primary production (Hughes et al. 2009) and integration of external or allochthonous nutrient subsidies (Abrantes & Sheaves 2008) that occurs there. Consequently, maintaining and improving ECW function and quality is critical to ensuring continued fisheries productivity (Walker et al. 2004).



Freshwater wetland in Giru, Queensland. Photo credit: Carla Wegscheidl

Queensland's ECWs are severely degraded and impacted by a diversity of anthropogenic stressors including expanding agriculture, development of coastal commercial activities and ports, and increasingly urbanisation (Grech et al. 2011). Much of Queensland's original lowland forest (Moore et al. 2007), and large areas of freshwater wetlands and brackish swamps (Russell et al. 2011, Saintilan & Rogers 2013) have been converted to agricultural land over the last 100 years, and ca. 8.5% of the total area of estuaries in the Great Barrier Reef region has been lost since European settlement (Sheaves et al. 2014). The historical wetland and riparian loss continues today (Sheaves et al. 2014). Much of this deterioration is the result of loss of tidal wetland area, including mangroves and saltmarsh, and this is compounded by the large areas from which fisheries species are excluded by barriers (e.g. weirs, tidal exclusion bunds, sand dams, and road and rail crossings). Connectivity is further reduced by inefficient culverts and crossings, and macrophyte chokes. The different anthropogenic impacts generate a complexity of consequences and outcomes, including for example increasing sediment loads (Alongi & McKinnon 2005), declining estuarine water quality (Cox et al. 2005), increasing exposure to acid sulphate soils and blackwater events (Powell & Martens 2005, Wong et al. 2010, Hladyz et al. 2011), and toxic cyanobacteria blooms (Albert et al. 2005). All of these pose risks for the condition of coastal biotic assemblages and their habitats (Fabricius et al. 2005).

The consequences are far reaching. ECW function has been compromised by substantial losses of some of Queensland's most productive of aquatic habitats (Boys et al. 2012, Heatherington & Bishop 2012). These impacts are compounded by impeded hydrological and biological connectivity (Sheaves & Johnston 2008)

that interrupts the delivery of allochthonous nutrients and limits access for fauna to highly productive wetland areas, compromising nursery ground value (Sheaves et al. 2014).

The widespread damage to Queensland's ECWs means there is an urgent need for their remediation (Sheaves et al. 2014, Creighton et al. 2015). Repairing these key ecosystems can lead to a raft of benefits: increased fisheries output and ecosystem resilience, enhanced food security and livelihoods, and the protection of ecological assets of national and global significance (Sheaves et al. 2014). Works are underway to repair and revitalise wetlands and estuaries along the Great Barrier Reef (GBR) coast. For the success of these repair initiatives, it is imperative that they are well targeted, carefully prioritised and their success adequately evaluated. Fundamental to this is the need to be able to **value** ECW services and ensure that outcomes are measureable in meaningful ways (Wegscheidl et al. 2017). However, a number of gaps in the scientific knowledge of northern Australian wetlands limit our ability to assess wetland value. Two of the key gaps are: a lack of understanding of (1) the exact way that wetlands and their habitats support important fisheries species, and (2) how the values of wetlands and their habitats can be measured in robust and valid ways, so that fisheries benefits (in terms of production) can be linked to wetland habitat function.

Objectives

The aim of this study is to investigate how the value of northern Australia's coastal wetlands and estuaries can be measured in robust, valid and meaningful ways. Consequently, we investigate the pre-requisites for the development of achievable measures of fisheries benefits that can be attributed to ECWs. In doing this we (a) examine the need for measures of ECW value and what form appropriate estimates should take, (b) assess appropriate methods for collecting and analysing necessary data, and (c) determine the additional studies needed to convert the available data into useable measures of fisheries benefits.

2.3 Measures of Estuary and Coastal Wetland Value

Background

The services provided by ecosystems are critical to human societies and contribute directly (e.g. food security) and indirectly to human welfare and economies (Costanza et al. 1997). ECWs are particularly important because of the diversity of services they provide (e.g. fisheries, nursery grounds, filtering and detoxification, blue carbon) (Barbier 2000). Despite arguments that wetlands and estuaries should be protected purely on grounds of their intrinsic ecological value, there is still a need to attribute a value to these systems, both because there are equally valid moral arguments related to the potential food security values stemming from altering wetlands (Costanza et al. 1997) and because arguments about intrinsic ecological value are difficult for decision makers to evaluate when balanced against tangible economic gains (Freeman 1991). In fact, the decisions society makes about ecosystems imply valuation (Costanza et al. 1997), and as long as we are forced to make choices we are intrinsically basing those choices on some measures of value (Costanza & Folke 1997).

Accurate, robust and valid measures of the value of ECWs are critical for many reasons. For instance, comprehensive estimates are needed to ensure the values of ECWs are given appropriate weight in policy and management decisions (Costanza et al. 1997), and that offsets and ecosystem repair can be prioritised and their outcomes measured (Sheaves et al. 2014, Creighton et al. 2015). However, and despite their widely recognised importance, many of the benefits from healthy CWEs are undervalued due to the high complexity of valuation methods (Brander et al. 2006, Barbier 2012). Indeed, many factors make it difficult to estimate the value of ecosystem services of ECWs because their values are multifaceted and interact in complex ways (Costanza et al. 1997), with high levels of connectivity among components, meaning management of the entire seascape will usually be necessary to preserve synergistic effects (Barbier 2000). The importance of ECWs can be both physical (habitat) and trophic, and depends on various factors including for example landscape and hydrological conditions, meaning that their value to fisheries production can greatly differ even among similar systems (Rozas & Zimmerman 2000, Minello et al. 2008,

Minello et al. 2012a). For example, in the southeast USA, estimated values of saltmarshes for recreational fishing vary between 981 Int\$ ha⁻¹ y⁻¹ and 6,471 Int\$ ha⁻¹ y⁻¹ (1984-value) for the west and east coast of Florida respectively (Bell 1997), a significant difference¹. This means that it is often not possible to translate values from one region/ecosystem to another (Brander et al. 2006), and estimating habitat value using data from other systems is not a valid or appropriate approach. In another example, based on penaeid landing data and estimates of saltmarsh in each state of the Gulf of Mexico, Engle (2011) calculated that catches attributed to saltmarsh varied between 57 and 1,660 (mean = 241) kg ha⁻¹ y⁻¹ depending on the geographic area, a difference of two orders of magnitude. Although only rough estimates, because the methods used assumed that the whole catches could be attributed to saltmarsh area (which is not the case since penaeids also use other habitats), results give an indication of the geographical variability in importance of saltmarsh wetlands to penaeid prawns. Furthermore, the economic value of the various services provided by ECW systems also depends on factors such socioeconomic conditions, management regime and policies, the balance of artisanal/subsistence versus commercial exploitation, and the particular uses of the different products by local people, which vary among countries (Freeman 1991, Smith 2007, Vo et al. 2012). Thus, scaling the contributions of wetlands to fisheries production needs to take into account both the trophic, landscape and socio-economic settings of the systems (Kneib 2003); emphasising that it will rarely be valid to predict the value of a wetland based on data from other systems (Woodward & Wui 2001, Kneib 2003).

Research on habitat valuation is led by the USA, particularly on saltmarsh habitats, which support some of the largest and most valuable fisheries (Zimmerman et al. 2002). Given the extent of saltmarsh loss in several US regions, many restoration projects have been implemented in the last decades and much work related to how fisheries communities respond to restoration projects (e.g. Minello & Webb Jr 1997, Minello 2000), on the best approaches to take in habitat restoration projects (e.g. Thom et al. 2004, Rozas et al. 2005, Reed et al. 2007) and on the economic benefits from restoration (e.g. Minello et al. 2012b) has been done. However, and despite the high economic importance and research effort put into those habitats over the last few decades, the complexity of biological and biophysical interactions means that estimating the production and monetary contribution of ECWs to adjacent fisheries remains a complex and challenging problem. Even the most advanced models have important limitations, and differences in settings, climates and contexts of different ECWs mean that approaches to deriving estimates will often need to be location-specific.

The following section describes examples of the most updated valuation methods for two types of commonly encountered coastal wetlands: estuarine saltmarshes and subtidal seagrass meadows.

Methods used in the valuation of wetland habitats

Example 1. Estuarine saltmarshes in Galveston Bay, Gulf of Mexico

Most studies on habitat-production relationship have been conducted in the saltmarshes of the USA, mostly in the Gulf of Mexico, where much research on the importance of saltmarsh habitats to fisheries species such as the penaeids *Farfantepenaeus aztecus, Farfantepenaeus duorarum* and *Litopenaeus setiferus* and the blue crabs *Callinectes sapidus* has been done (Zimmerman et al. 2002). These species are highly valuable and most of their US fisheries productivity comes from the Gulf of Mexico area, explaining the focus of research in that region over the last few decades. Available studies include detailed mapping and in-depth analysis of tides and inundation patterns (e.g. Minello et al. 2012a), broad- and fine-scale analysis of habitat use and density (e.g. Rozas & Zimmerman 2000, Minello & Rozas 2002, Rozas et al. 2007), estimating natural mortality and growth rates (e.g. Minello et al. 1989, Mace III & Rozas 2015),

¹ The Geary Khamis dollar, also known as International dollar or Int\$ is a hypothetical dollar widely used by economists to facilitate comparisons between currencies. It is based on a 'purchasing power parity value' with the US\$ at the time of comparison, by incorporating exchange rates and average prices of commodities. Refer to: https://unstats.un.org/unsd/methods/icp/ipco_htm.htm

identification of the trophic importance of saltmarsh and of marsh flooding to these species (e.g. Baker et al. 2013), etc. Catch data in the Gulf of Mexico has also been methodically recorded for decades (e.g. Hart 2012, Hart & Nance 2013). All this prior information is needed to obtain accurate estimates of the importance of the different habitats to fisheries production (Minello et al. 2008, Minello et al. 2012b).

Minello et al. (2008) provides the most comprehensive production estimates for a fishery species from a natural saltmarsh wetland published to date. They used new data as well as prior information on topography, hydrology, habitat use, growth and mortality rates, etc. obtained over almost 3 decades of research to produce production estimates of penaeid (F. aztecus, L. setiferus) and blue crab (C. sapidus) juveniles in the regularly flooded saltmarsh wetlands of lower Galveston Bay, Texas. Briefly, their method consists on using fine-scale data on juvenile **densities** in different habitats, including at the marsh edge and at different distances from edge both towards the marsh interior and towards open water, to produce small-scale distribution models. These data were then combined with detailed habitat, topography and bathymetry GIS mapping of the area to calculate the total area of each habitat and sub-habitat category, so that the **population size** in the overall area could be estimated. Animal sizes were also measured to calculate length frequencies and species-specific length-weight relationships, to be used to produce an estimate of biomass (based on the estimate of population size). Subsequently, daily increases in biomass for each size class were projected using estimated growth rates, and the values for all sizes averaged to estimate the mean potential daily increase in mass per individual. This value was then multiplied by the previously calculated population size to estimate daily **production** in that area, producing base data that could then be transformed into annual production per hectare.

Using this method, Minello et al. (2008) estimated a production from the lower Galveston Bay marsh complex of 128 kg ha⁻¹ y⁻¹ for *F. aztecus*, 109 kg ha⁻¹ y⁻¹ for *L. setiferus* and 170 kg ha⁻¹ y⁻¹ for *C. sapidus*, which was 3.1, 2.5 and 8.8 times higher than in shallow open water, respectively. They also found that although only 15% of the available habitat was saltmarsh, this habitat supported 45% of brown shrimp, 34% of white shrimp and 53% of the blue crab population. Therefore, using this method, it is possible not only to calculate the overall production in an area, but also to the production that can be attributed to particular habitats within the system.



Coomera saltmarsh, South Queensland. Photo Credit: Norm Duke

This method would be the most appropriate to be use in northern Australia's mangrove and saltmarsh wetlands. However, much of the basic research in that region is still needed before we can make accurate predictions of habitat value. Australian saltmarshes are also very different to those in the USA, as they are higher in the intertidal, typically landward of mangrove forests, and have shorter and less frequent inundation periods (Connolly 2009, Davis et al. 2012). In the northern hemisphere, however, saltmarshes extend into the mid-intertidal zone and mangrove forests are less extensive and less dense, or absent. Therefore, the functional importance of saltmarsh either as habitat provider or a source of nutrition is likely to be different to that in the northern hemisphere, and because of their higher position in the intertidal this functionality is more difficult to ascertain. This is also true for mangrove wetlands, which are not accessible for much of the tidal cycle and are difficult to effectively and quantitatively sample using the available sampling methods.

Example 2. Seagrass meadows in Southern Australia

For low intertidal and subtidal wetlands such as seagrass meadows, the most advanced valuation studies use a seagrass residency index (SRI) to estimate the proportion of fish landings that can be attributed to seagrass area (e.g. McArthur & Boland 2006, Jackson et al. 2015). SRI values for each species are based on published data and expert opinion on habitat use, specifically on the proportion of time each life-stage spends in the seagrass habitat (McArthur et al. 2000).

In southern Australia, McArthur and Boland (2006) used a model based on SRI and 16 years of comprehensive spatial and temporal catch-per-unit-effort (CPUE) data to estimate the contribution of seagrass habitats and impacts of seagrass loss on various fisheries species. Briefly, their study area was divided into a ~1°latitude by ~1°longitude grid of fishing blocks, and multiple linear regressions between catch (kg of live weight), effort (boat days) and seagrass area (determined by mapping) for all non-zero effort fishing blocks were run for all species with SRI > 0.30, i.e. for species perceived to have at least some dependence on seagrass habitats. Using this model, they estimated that South Australian seagrass meadows had an economic contribution of \sim \$A114 M y⁻¹ to fisheries. This method is however reliant on broad and untested assumptions, is very general and has important limitations. For example, the information used on proportion of time that the different life stages reside in the different habitats is limited for most species and geographical regions. Additionally, using this approach it is difficult to incorporate spatial and temporal variation in habitat use (Jackson et al. 2015). Further limitations are that estimates did not take into account differences among seagrass communities or densities across the region, and that effort was measured in number of boat days, irrespective of boat size or number of fishers in the boat. Overall, while these estimates provide some general useful insights, the range of limitations mean that the estimates produced cannot be considered as reliable quantification of actual fisheries value.

In another approach, also used for oyster reefs (Peterson et al. 2003, zu Ermgassen et al. 2016), Blandon and zu Ermgassen (2014) estimated the enhancement of juvenile fish abundance provided by the presence of seagrass habitats by conducting a meta-analysis of juvenile fish abundance in seagrass vs. unvegetated sites in southern Australia. There, the average enhancement for each species was calculated based on density data for each species and each study using the equation: Enhancement = $\rho_{seagrass} - \rho_{unvegetated}$ (ρ = density of 0.5 yr old fish, in ind m⁻²). The potential annual production attributable to seagrass was then calculated based on the average enhancement as well as on species-specific growth models, age-specific weight (using published length-weight relationships) and species-specific natural mortality rates (taken from Fishbase (Froese & Pauly 2017) or from published studies), to estimate the per-unit-area augmentation of production that resulted from the presence of seagrass to commercial fish biomass and, subsequently, to the economy. Results from that study suggest that seagrass enhances the biomass of the 12 commercial fisheries considered by 0.98 kg m⁻² y⁻¹, corresponding to ~\$A230,000 ha⁻¹ y⁻¹.

As with the approach based on SRI, this method also has limitations. Firstly, the studies considered in the meta-analysis encompassed areas with different climates and ecology, but did not address the resulting differences in seagrass communities, even though it is known that the degree of enhancement of fisheries production resulting from a habitat can differ among regions (zu Ermgassen et al. 2016). Moreover, sampling effort and sampling methods also varied among studies, and most studies used methods such as beach seines and push nets (and of different mesh sizes), which are not appropriate for providing estimates of biomass density per area, and also can not provide comparable data because they do not have similar catching characteristics. Also, studies assumed that all individuals captured were juveniles of similar age, and used length-weight relationships and growth and natural mortality rates from similar species as proxies when species-specific parameters were unavailable, despite that the use of inaccurate parameters can greatly affect production estimates (zu Ermgassen et al. 2016).

Clearly, it is unlikely that either method could produce precise estimates. Indeed, the two methods led to very different results for the same species, even though they were conducted in the same broad region. For example, for garfish (*Hyporhamphus melanochir*), Blandon and zu Ermgassen (2014) estimated an enhancement of 0.02×10^{-5} kg ha⁻¹, worth \$A0.06 ha⁻¹, much lower than the values of 0.68 kg ha⁻¹ (\$A3.9 ha⁻¹) estimated by McArthur and Boland (2006). In contrast, for King George whiting (*Sillaginodes punctata*), the estimated enhancement was higher using the method of Blandon and zu Ermgassen (2014) (4.64×10^{-3} kg ha⁻¹; \$A824 ha⁻¹) than when using the method of McArthur and Boland (2006) (0.49 kg ha⁻¹; \$A6 ha⁻¹). This illustrates the limitations of currently available valuation methods for seagrass habitats, and underlines that care must be taken when extrapolating results from one region to another, and when comparing the value of habitats using results from different studies.

Pre-requisites for the development of achievable, robust and valid measures of ECW value for northern Australia

As clear from the previous examples of habitat valuation, whether the final output is a complex ecologicaleconomic model (Barbier 2007) or an estimate of the value of a particular habitat or area to be managed or repaired, there are two crucial pre-requisites for estimating the value of a ECW:

- 1. obtaining **estimates of the areal extent** of the units of interest and the habitats that comprise them, and
- 2. using high quality measures of the value of the particular units (habitats, estuary reaches etc.).

For northern Queensland estuaries, the first prerequisite is partially satisfied because recent extensive and detailed GIS topographic mapping (e.g. by the Queensland Wetland Program) means that appropriate high quality mapping is available for the intertidal and subtidal part. However, detailed bathymetry is lacking for most estuaries meaning estimates relating to subtidal areas will be less precise than those relating to the intertidal components.

In the case of high quality measures, the value of the different habitat units should be (1) measureable at the scale of the unit or outcome to be evaluated, (2) broadly meaningful and (3) easy to communicate to end-users. High quality estimates of *production* (the expected increase in biomass over time for a population (Chapman 1978)) *of exploited species* are of particular value in a fisheries context (e.g. McArthur & Boland 2006, Barbier 2007) because they provide detailed information on the value of a habitat unit, by detailing the amount of biomass produced from that unit over a specific time period. Not only is production per unit area of fisheries species directly relevant to end users, and so easy to communicate, but it provides the added advantage of integrating across complex factors such as connectivity and nursery ground provision, that are often hard to assign a defensible values to (Costanza et al. 2006).

However, even if these two basic pre-requisites are satisfied there is the further requirement that:

 the spatio-temporal nature of organismal utilisation of units needs to be such that it allows unambiguous allocation of a specific component of production to the particular unit in question. This condition is very difficult to satisfy, particularly in the case of mobile organism that move between different units and different types of units over time, or when the same life stage occupies a number of different habitats at one time.

2.4 Estimating Fisheries Production

Production is classically defined as the "total elaboration of fish tissue during any given time interval Δt , including what is formed by individuals that do not survive to the end of Δt " (Ivlev 1966). It is expressed in units of quantity per area per time, typically kg ha⁻¹ y⁻¹ for fishery populations. Its calculation therefore requires information on changes in population size and biomass through relatively short periods of time, along with recruitment, growth and mortality rates, so that annual production can be estimated by adding up production over all short intervals to make up a year. Therefore, estimating production relies on a large amount of long-term data obtained by time-consuming studies, and most methods of production calculation rely on a number of assumptions about for example recruitment, changes in mortality with size (age) and migration.

Standing stock biomass (the biomass of a species in a defined area at a point in time (Rozas et al. 2005)) is the most basic parameter needed to estimate production and is typically calculated using abundance-bylength data and length-weight relationships. The first step needed to calculate biomass involves estimating the **densities** of the different sizes in the different habitats, so that the abundance-by-size can be estimated. These values are then multiplied by the respective mean weights of the different size classes (based on length-weight relationships) to estimate biomass. For many species, length-weight relationships are available from the literature and/or on Fishbase (Froese & Pauly 2017) but for more accurate results these parameters need to be calculated directly from the studied populations. **Growth** and **mortality** estimates of fisheries species are typically obtained from mark-recapture (Pine et al. 2003), lengthfrequency analysis (Pauly & Morgan 1987) or the analysis of periodic markings on growing structures such as otoliths and scales in teleosts and cartilage in elasmobranchs (e.g. Newman et al. 1996, Russ et al. 1998, Walker et al. 1998).

Other factors also need to be taken into account when estimating production. For example, for systems including mangroves (Baker et al. 2015) and saltmarshes (Connolly 1999, Minello et al. 2012a, Baker et al. 2013), there can be substantial spatial and temporal variations of accessibility (both physical and trophic) to the different habitats, related to the spatial arrangement of habitats and to differences in topography, hydrology, and consequent differences in flooding patters (area, duration, frequency and depth) and connectivity (Zimmerman et al. 2002, Baker et al. 2015). This leads to spatial/temporal differences in functional values provided by the different habitats and different parts of the habitats (e.g. edge vs. inside of saltmarsh/mangrove forest) (Roth et al. 2008, Minello et al. 2012a, Baker et al. 2015). These factors should also be considered when estimating the contributions of the different habitats to fisheries species, but they are difficult to identify and quantify.

In summary, a comprehensive body of data are needed for the calculation of production estimates for a species from a wetland (Figure 1), including data on:

- (i) the extent of each habitat type,
- (ii) replicate small-scale estimates of density within each habitat,
- (iii) spatial arrangement of habitats,
- (iv) size frequency of the species,
- (v) size-weight relationships, and
- (vi) growth and mortality rates.

Data on (i) and (ii) allow **population abundance** to be estimated, while (iv) and (v) allow abundance to be converted to **biomass**. Sampling biomass over time and combining it with growth and mortality rates (vi) allows abundance estimates to be converted to estimates of **biomass production** over a period of time (e.g. annually) (Minello et al. 2008).



Figure 1. Steps needed for the calculation of production estimates for a species from a single habitat.

2.5 Sampling Methods for Estimating Density and Production

Background

As explained above, the most fundamental component of fisheries biomass and production estimates are measures of **density**. There are many methods and gears to sample fisheries species that provide catchper-unit-effort (CPUE) data (e.g. Table 1). However, few of the available gears sample a definable volume of water or area of wetland, a basic requirement to enable the conversion of CPUE into a measure of density. Of those that do provide area-based estimates, many can only be deployed in a few specific situations, restricting their usefulness for comparisons among habitat types (Rozas & Minello 1997, Connolly 1999, Baker & Minello 2011). Even those that have been successfully used to provide estimates of density (e.g. drop samplers (Minello et al. 2008), pop nets (Serafy et al. 1988), cast nets (Sheaves et al. 2016a)) have important limitations. Drop samplers and pop nets are limited to shallow water applications and, because operators need to enter the water to harvest catches, they are unsuitable in areas, such as tropical Australia, where estuarine crocodiles occur. *Cast nets* can be used without the need to enter the water but have the limitations of being less effective on large fish, which may be able to escape as the net sinks, and in not providing a completely consistent sampling area. Beam trawls have also proved effective in providing estimates of density per unit area, particularly for sampling deeper open bottom habitats including seagrass beds (Watson et al. 1993).

Details of appropriate gears

Cast nets are particularly useful for sampling shrimps and prawns because their escape response tends to be tactile rather than visual (Watson et al. 1992), meaning they show little response until the net covers and captures them, and even if they are alarmed their escape direction is random (Watson et al. 1992, Xiao & Greenwood 1993). Cast nets are particularly useful in the structurally complex habitats of tropical estuaries, where submerged timber, or 'snags', are common (Sheaves 1992), because they can be used across most habitats (Sheaves et al. 2007a, Johnston & Sheaves 2008). In fact, if visibility is good they can even be deployed directly adjacent to snags, something not possible with most other netting approaches. The consistency of the area sampled by cast nets can be improved by the use of experienced operators

(Johnston & Sheaves 2007) and they have proved successful for estimating densities of smaller fisheries species in tropical estuaries (Sheaves et al. 2007b). Thus, on balance, cast nets provide a simple way to estimate density of shrimps and prawns in many tropical estuary and coastal wetlands (Sheaves et al. 2016a). Although not as reliably effective on fish, because the possibility of avoidance is higher, cast nets are still one of the most effective ways of sampling smaller fish (e.g. herrings and silver biddies) in tropical estuaries (Sheaves et al. 2007b) and so provide some of the better estimates of density for baitfish species.

Table 1. Comparison of the effectiveness of some gears commonly used to sample estuary and wetland fisheries species.

Gear	Measure of area?	Sampling habitats	Comments	
Beam trawl	Swept area	Smooth unstructured bottoms	Suitable for prawns because of random escape response. Do not enclose so inefficient for mobile fish species. Difficult to deploy in shallow water.	
Cast nets	Enclosed radius	Many habitats; can be used close to structure	Can be used across many habitats except for heavily structured ones. Area sampled can however vary and more mobile species can escape. Most suitable for prawns because of random escape.	
Drop sampler	Enclosed radius	Open areas and light vegetation	Accurate sample once deployed but vessel needs to be deployed close to the sampling location potentially causing fish to move away. Only usable in very shallow water.	
Electrofishing	No	Most habitats	Only effective in very low salinities.	
Fish traps	No	Most habitats	Attract fish with bait so unsuitable for density estimates	
Fyke nets	No	Blocking drains	Used to block channels draining areas of wetland so difficult to define area sampled.	
Gill nets	No	Unstructured open water	Designed to intercept moving fish so can not be used to relate catch to area. Efficiency dependent on day-to-day behaviour.	
Lift /pop nets	Enclosed radius	Open areas and aquatic vegetation	Need to be set on the substrate prior to sampling so may bias samples. Only useable in very shallow water. Operators need to enter water so unsuitable in crocodile/hippopotamus risk areas.	
Seine nets	Swept area	Smooth unstructured bottoms	Only useable on smooth bottoms with consolidated sediments. Also, need to be deployed adjacent to a shoreline.	
Video (baited)	No	Most habitats	Area 'fished' difficult to define because bait is used to attract.	
Video (unbaited)	No	Most habitats	Main limitations are water clarity and difficulty in defining area sampled. Most useful for detecting presence in a habitat.	

Beam trawls can provide estimates of density via the swept-area method but suffer the restrictions that they are difficult to operate in very shallow water and can only be used in areas lacking hard structures such as snags or rocks. However, beam trawls have proven useful in estimating densities of seagrass-associated prawns such as *Penaeus esculentus*, *P. semisulcatus* and *Metapenaeus endeavouri* (e.g. Watson

et al. 1993, Loneragan et al. 1995). Consequently, samples of penaeid prawns and baitfish captured with both cast nets and beam trawls combined have the potential to be developed into useful estimates of production per area of ECW habitat.

Estimating the production of larger species such as barramundi (*Lates calcarifer*) is more difficult, and tends to rely on CPUE rather than density per unit area, making it difficult to meaningfully translate estimates to variables such as the area of wetland. Traditionally, stocks of species like barramundi have been assessed on the basis of commercial catches from the gill net fishery (e.g. Staunton-Smith et al. 2004). These data can provide indices of abundance but, because of the diversity of factors affecting gill net catches and because gill net catches cannot be related to a specific fished area (Table 1), such indices are only suitable for estimates of relative rather than absolute production. Consequently, because of the ability to obtain estimates of biomass per unit area, measures of the production of prawns and baitfish provide the greatest opportunity for development as fisheries-based indices of value for northern Australia's ECWs.

2.6 Considerations for the Development of Appropriate Measures of Value for Northern

Australian ECWs

Using **production** of species of commercial and recreational importance as indicators of the productivity of ECW habitats has the substantial advantages of being broadly meaningful and easy to communicate to end-users. However, there are three key considerations that still need to be taken into account to more accurately value ECWs and their component habitats:

- (i) linking estimates of biomass density to areas of habitat in a meaningful and valid way,
- (ii) taking into account the ecological context of the species and its link to productivity,
- (iii) considering the state of understanding of species and community ecology.

Meaningfully linking estimates of biomass density to areas of habitat

As mentioned above, sampling methods such as cast nets and beam trawls can provide reliable estimates of density per unit area for the habitats in which they can be deployed. However, a number of steps are needed to convert these into biomass estimates. The problem is relatively simple if the unit of interest is a single habitat type where biomass density can be assumed to be, on average, homogeneous. The steps are then straightforward (Figure 2):

- 1. Collect sufficient biomass density samples to ensure that:
 - a. The pattern of within habitat variability is well understood. This will allow evaluation of the extent to which the assumption that the species biomass density is homogenous across the habitat type, and so whether it is reasonable to use an average value (e.g. the mean biomass) as an estimate for the whole habitat;
 - b. the mean biomass is accurately and precisely estimated; and
- 2. use this mean biomass as an estimate of the biomass per unit area for the habitat, and the estimated variability to provide a measure of uncertainty about the estimate.

However, because ECWs are composed of mosaics of habitats, the problem will usually be more complex. Take for instance the problem of estimating standing stock for an estuary reach. The reach will (i) comprise a number of different habitats, each with intrinsically different densities of the target species, and (ii) include both habitats that are efficiently sampled using the particular gear and those that aren't.

Additional steps are therefore necessary (Figure 2):

- 1. Areas in which the target species are well sampled can be treated as in (1) above.
- 2. Comparable estimates will need to be made for areas in which sampling with the standard gear is inefficient. This will often be difficult. For instance, although cast nets are inefficient for structurally complex habitats like fallen timber, other sampling methods are also unsuitable. While there is no perfect approach to solving this problem there are workable solutions. One is to use a technique such as unbaited video (Meynecke et al. 2008, Kimball & Able 2012, Sheaves et al. 2016b) to determine the extent to which the species utilises the difficult-to-sample habitat and use this information to construct approximate biomass density estimates for those habitats, together with measures of the uncertainty involved in the estimates.
- 3. Once the total area of each habitat type is known, standing stock estimates for the whole estuary can be constructed.



Figure 2. Steps needed to convert biomass per unit area to an estimate of standing stock.

Although estimates will never be perfect, this protocol can provide useful approximations of biomass, providing well founded estimates with a defined level of uncertainty. However, the issue of the spatio-temporal nature of organismal utilisation of units needs to considered, but this condition is very difficult to satisfy for mobile organisms (see above).

The ecological context of the species and its link to productivity

Not only are there readily available methods for sampling biomass per unit area for species such as penaeid prawns, that can provide valid data for estimating biomass, but the ecological context of these species makes them good candidates for linking their productivity to ECW habitat area. The food webs leading to penaeids, such as banana prawns *Fenneropenaeus merguiensis*, are relatively simple and well understood (Loneragan et al. 1997, Abrantes & Sheaves 2009a, 2010, Abrantes et al. 2015). These food webs are short (see Figure 3), enabling direct links to be made between the prawns and the ECW resources that support their productivity. In contrast, not only it is difficult to obtain density data for large fish predators such as barramundi (*Lates calcarifer*), but the food webs leading to high trophic level species are typically much more complex (e.g. Figure 4), making it much more difficult to relate their biomass to particular resources. In fact, highly mobile species like barramundi are likely to depend on a complex mosaic of interlinked habitats throughout their life history (Nagelkerken et al. 2015) meaning more integrated measures of the value of wetlands to higher level predators are needed.



Figure 3. Simplified schematic food web leading to *Fenneropenaeus merguiensis* at the mangrovedominated Hinchinbrook Channel, North Queensland (adapted from Abrantes and Sheaves (2009b)), showing the main trophic pathways. 1 – mangrove detritus; 2 – microphytobenthos; 3 – green filamentous algae; 4 – seagrass, seagrass detritus and seagrass epiphytes; 5 – phytoplankton; 6 – upper intertidal benthic fauna; 7 – lower intertidal and subtidal benthic fauna; 8 – zooplankton. Figures and symbols courtesy of the Integration and Application Network, University of Maryland Centre for Environmental Science (ian.umces.edu/imagelibrary).



Figure 4. Simplified schematic the food web leading to barramundi *Lates calcarifer* at the mangrovedominated Hinchinbrook Channel, North Queensland (adapted from Abrantes and Sheaves (2009b)), showing the main trophic pathways. 1 – mangroves; 2 – mangrove detritus; 3 – microphytobenthos; 4 – green filamentous algae; 5 – seagrass (including seagrass detritus and seagrass epiphytes); 6 – phytoplankton; 7 – terrestrial insects; 8 – sesarmid crabs and mangrove snails; 9 – upper intertidal benthic

fauna; 10 – phytodetritivorous fish; 11 – lower intertidal and subtidal benthic fauna; 12 – zooplankton; 13 – macrobenthic carnivores (fish); 14 – plantivorous fish. Figures and symbols courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/imagelibrary).

One approach to overcome the problems with obtaining meaningful estimates for large predators is to use the density of easily measured surrogate species (e.g. Lewandowski et al. 2010, Mellin et al. 2011, Fontaine et al. 2015) as relevant indicators of the support that ECWs provide for these predators. Measuring the productivity of penaeid prawns is one obvious option because these are key prey of many commercially important predators such as barramundi, trevallies (Caranx spp.), queenfish (Scomberoides spp.), threadfin salmons (Eleutheronema tetradactylum and Polydactylus spp.) and snappers (Lutjanus spp.) (Robertson 1988, Salini et al. 1990, Salini et al. 1998, Baker & Sheaves 2005). For example, Fujiwara et al. (2016) investigated the importance of penaeids in supporting commercially important fish in the Gulf of Mexico using data collected over 28 years to analyse the relationships between CPUE of fish predators and those of their penaeid prey in different areas and seasons. Results from that study could then be used in conjunction with those from studies on the importance of coastal wetland habitats to penaeids to extrapolate the importance of those habitats to the commercial fish predators. A second alternative is to measure the density of small planktivorous fish such as herring (Herklostichthys spp.), also key components of food webs linking primary productivity to high order predators (e.g. Salini et al. 1990, Baker & Sheaves 2005). Estimates of planktivore and penaeid biomass can be obtained simultaneously by cast netting (Sheaves et al. 2016a), making this process even more valuable as a tool to measure ECW productivity. However, on a cautionary note, many of these species demonstrate substantial spatiotemporal variability in density and biomass (Sheaves & Johnston 2010).

The state of understanding of species and community ecology

Even for well-studied species there is often a deficit in the information needed to effectively make the species-productivity link. This limits the reliability with which biomass production can be linked to particular habitat units. For example, even though the issues seem reasonably straightforward for species like F. merguiensis, this is not necessarily the case. Commercial fisheries for F. merguiensis occur in offshore waters but their juveniles are strongly associated with mangrove estuaries (Vance et al. 1990), meaning there is an apparent link with mangrove wetlands. Indeed, offshore catches of adult prawns are correlated with the extent of mangrove forests (Manson et al. 2005). However, the extent to which the apparent relationship between juvenile penaeids and mangroves reflects specific utilisation of mangroves, or just the use of shallow, organically rich, muddy habitats has been questioned (Lee 2004). For example, a study focussing on juvenile F. merguiensis within 30 mangrove estuaries spanning 650 km of the coast of north-eastern Australia (Sheaves et al. 2012) assessed the prawn-mangrove relationship among and within estuaries. The study indicated that (i) at the among-estuaries scale mangrove extent appeared to influence CPUE but was extensively confounded with the effects of two non-mangrove variables: intertidal extent and substrate type, (ii) connectivity with mangrove forests was not influential, pointing to the likely importance of the non-mangrove variables rather than mangrove extent, and (iii) at the within-estuary scale CPUE showed no correlation with mangrove variables but rather correlated with the extent of shallow water, again implicating the role of a complex of ECW habitats in supporting juvenile F. merguiensis populations. This idea is strengthened by studies that indicate that wetlands where mangroves are not the dominant vegetation are also important habitats for juvenile F. merguiensis (Sheaves et al. 2007b). Consequently, there is a clear need to develop a more explicit understanding of the ways in which coastal wetlands support even species such as F. merguiensis that are well recognised as having strong links to mangroves. Developing a more sophisticated knowledge of the specific ways that ECW habitats influence fisheries populations is clearly critical if we are to fully account for the value of these habitats to fisheries.

Therefore, although the basic information on biomass density is available for key species such as *F. merguiensis* in north Queensland estuaries, substantial research is still needed before these can be converted to valid estimates of biomass and production. This includes both the careful and comprehensive sampling needed to provide estimates for all the habitats well-sampled by the sampling gear employed, detailed estimates of the extent of each habitat type, and extensive studies to develop the best possible estimates for habitats that cannot be sampled using conventional gears. There is also a critical need to develop a more detailed knowledge of the exact ways ECW habitats influence fisheries populations, to fully account for the value of these habitats to fisheries

Conclusion

Estimates of biomass density over time provide a full picture of the time-integrated production from a habitat or area, and are therefore the most comprehensive way to assess habitat value. However, in all but a few cases many of the necessary key data are unavailable, either because it is logistically difficult to collect such data, or because the dynamics of species populations relative to individual target habitats prevents the development of valid estimates (Table 2). Particularly where there are logical difficulties in producing valid production estimates, it is likely to be most profitable to focus on lower value estimates that can also provide useful information. For instance, initial work in northern Australia's estuaries could be directed to producing high quality estimates of **biomass**. Although not integrated over time, and so not providing direct information on the increase in biomass in a unit of time, biomass estimates are readily achievable and can provide useful relative measures of estuary or coastal wetland habitat value that are easily understood and easily communicated, as long as their limitations as snapshots in time are recognised. For a particular point in time, standing stock biomass provides a well-established and valid basis for evaluating the contributions from ECW habitats and a basic measure of how those contributions are likely to change under different scenarios.

	Biomass	Production
Data needed	 Areal extent of the units of interest and the habitats that comprise them, at appropriate scales Density within each habitat, at appropriate scales Size-frequency in the different habitats Size-weight relationship Also needs to consider factors such as spatial arrangement of habitats, including details on accessibility/ connectivity 	In addition to the data needed to estimate biomass, production estimates require:
		 Biomass estimates over time Growth rates Size/stage-specific mortality rates Recruitment rates Migration information

Table 2. Data needed to produce estimates of biomass and production for north Australian estuaries.

What can be validly interpreted from data collected?	 Can provide easily understood relative measures of habitat value Easy to communicate to end- users. Can be used as a basic measure of how contributions from different habitats are likely to change under different scenarios 	 Most comprehensive way to assess habitat value Easy to understand and to communicate to end-users. Broadly meaningful Most useful to assess how contributions from different habitats are likely to change under different scenarios
Limitations	 Spatio-temporal variations in habitat use, habitat availability and functional value need to be considered but are difficult to identify and quantify Most available gears are not useful to provide adequate biomass per unit area estimates Difficult for some mobile organisms due to use of multiple habitats 	 Additional to the limitations in estimating biomass density, production estimates rely a large amount of long-term data obtained by time-consuming and often logistically challenging studies Most key data are unavailable, so estimates are often based on a number of assumptions (e.g. on recruitment, size-related mortality, migration), limiting precision
Can necessary data be collected validly?	Yes, in many cases.	Very difficult. Only possible in few cases.
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CHAPTER 3 - IMPROVING OUR ABILITY TO ESTIMATE THE VALUE OF ESTUARINE NURSERY HABITATS TO JUVENILE BANANA PRAWNS (*FENNEROPENAEUS MERGUIENSIS*) IN NORTH QUEENSLAND ESTUARIES

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3.1 Abstract

Estimates of the value of different habitats can provide an objective basis for the prioritisation of conservation and restoration actions. The fisheries production that can be attributed to a particular unit (e.g. a wetland, a mangrove forest or a whole estuary) is one obvious measure, but one that has proven difficult to estimate effectively. We used the case study of the use of a tropical estuary by juvenile banana prawns, Fenneropenaeus merguiensis, to assess the potential to produce valuable production estimates, in terms of logical constrains and estimated error structure, at three spatio-conceptual scales: (i) the estuary reach, (ii) the whole estuary, and (iii) the contribution of the estuary to the exploited stock. Because habitats with low occupancy can have high trophic value, we also used stable isotope analysis to assess the importance of mangroves and saltmarshes as ultimate sources of nutrition to F. merguiensis in four North Queensland estuaries. Estimates of production showed high spatial and temporal variability. This result was not unexpected; evaluating patterns of abundance of mobile organisms is always difficult because, rather than a few prominent parameters determining how many individuals occur in a particular place at a particular point in time, local abundance is the result of the interaction of a wide array of factors, what has been described as a causal thicket. Of the three conceptual scales investigated, estimates for the whole estuary were the most viable. Estimates for individual estuary reaches or habitat types require the unreasonable assumption that prawns remain in the one area during their time in the estuary, while estimates of contribution of an estuary to the offshore fishery require difficult to obtain information, such as the proportion that different estuaries contribute to offshore stock(s). Stable isotope analysis indicated that F. merquiensis juveniles used a variety of primary producers as sources of nutrition, suggesting that no particular type of habitat is of critical nutritive importance, adding weight to the idea that productivity outcomes for F. merguiensis are likely to be most usefully and validly assessed at the whole-of-estuary scale. Interest in repair investments is high. So too are expectations by these investors that science can provide accurate quantification of potential productivity benefit. Indeed the systems and interactions are too complex for accurate quantification. We provide very conservative estimates of productivity benefit using Banana Prawns as an index of likely productivity change. Even these conservative estimates demonstrate the likely outstanding benefits of seascape repair.

3.2 Introduction

Penaeid prawns are commercially important throughout their distribution due to their high densities, very high reproductive output, and fast growth rates that allow harvesting within a year. Their typical life-cycle involves adults that occur in near- and offshore waters, and larvae that migrate to estuarine wetland habitats, which are used as nursery areas. Due to their high economic importance, penaeid biology and fisheries are some of the most well-studied. However, even within a region, landings can vary substantially both among and within years due to the complex range of factors that affect recruitment, growth and survival, including both biological and climatic predictors (Robins et al. 2005, Diop et al. 2007). Typically, the abundance of commercial-sized penaeid prawns is predicted using one of three approaches: 1) adult abundance is related to biological factors such as the abundance of post-larvae or juveniles in related habitats, 2) adult abundance is related to environmental parameters (e.g. rainfall, habitat type and availability, temperature), or 3) stock-recruitment models are used to relate the abundance of penaeids at

the reproductive stage in one generation to the numbers entering the fishery in the following generation. Models have also been developed that combine the three approaches (Diop et al. 2007). It is thus clear that the abundance of early life-history stages and the ecological conditions in adjacent coastal nursery habitats are critical factors affecting adult abundance and, therefore, potential fisheries output. In face of the worldwide increase in coastal habitat degradation due to various anthropogenic actions (e.g. agriculture, urbanisation), it is imperative to identify and attribute a value to the most important habitats supporting the early life-history stages of penaeid prawns, so that these can be prioritised in conservation and/or restoration projects, ensuring continuing profitable fisheries (Sheaves et al. 2014).

The banana prawn, *Fenneropenaeus merguiensis,* is an important fishery species in northern Australia, where there has been extensive research on the species' biology and ecology (e.g. Vance et al. 1990, Vance et al. 1996, Loneragan et al. 1997, Vance et al. 1998, Kenyon et al. 2004). Juvenile *F. merguiensis* are particularly associated with mangrove estuaries (Vance et al. 1990, Kenyon et al. 2004), and *F. merguiensis* catches have been found to be correlated to the extent of mangrove forests along adjacent coasts (Manson et al. 2005), making this species an ideal model to study habitat-production relationships. However, because several factors synergistically affect prawn catches (Vance et al. 1985, Diop et al. 2007), because many factors other than the presences of mangroves drive abundances (Sheaves et al. 2012a), and because catches typically occur in offshore habitats that are distant and different to the coastal juvenile habitats, it is very difficult to attribute a monetary value per area to the different juvenile habitats, i.e. to estimate the value for fisheries production that arises from the preservation or restoration of a unit of each of the different juvenile habitats (Sheaves & Abrantes 2017).

Production estimates

The first step in addressing habitat valuation in terms of fisheries output would be to determine the importance of the different juvenile habitats. Measures of production (the increase in biomass over time for a population (Chapman 1978)) are particularly useful in habitat valuation (e.g. McArthur & Boland 2006, Barbier 2007) because they attribute a value to a habitat unit by detailing the amount of biomass produced from that unit over a specific time period. Although basic information on biomass density is available for *F. merguiensis* for several North Queensland (Australia) estuaries (Sheaves et al. 2012a; Sheaves, Abrantes, unpubl. data), substantial research is still needed before this can be converted to valid estimates of total biomass and/or production (Sheaves & Abrantes 2017), particularly if the purpose is to link production to specific habitat units (e.g. specific areas of intertidal wetland). Indeed, biomass density needs to be estimated for the different habitats at finer scales, and the results then combined with detailed habitat mapping (including topography and bathymetry), and growth rates, size-related mortality rates, recruitment and emigration rates need to be estimated (Minello et al. 2008, Minello et al. 2012). Studies are also needed to develop the best possible estimates of biomass density for habitats that can not be sampled using conventional gears (Sheaves & Abrantes 2017).

Although production estimates provide the most complete picture of the value of a habitat or area, in many cases the necessary base data are still unavailable, both because it is time-consuming and logistically difficult to collect, and because the dynamics of species populations relative to individual target habitats prevents the development of valid estimates (Sheaves & Abrantes 2017). Consequently, it will often be more reasonable and profitable to direct work towards producing high quality estimates of *biomass density* (i.e. biomass per unit area) for the different habitat units. Although not integrated over time, and so not providing direct information on the increase in biomass per time, biomass density estimates are usually more achievable and can provide useful relative measures of habitat value that are easily understood and easily communicated, as long as their limitations as snapshots in time are recognised (Sheaves & Abrantes 2017). For a particular point in time, standing stock biomass density provides a well-established and valid basis for evaluating the contributions from estuary and coastal wetland habitats and a basic measure of how those contributions are likely to change under different scenarios.

Sources of Nutrition

Fenneropenaeus merguiensis are key components of well-known food webs (e.g. Figure 1). They are of low trophic level and feed mostly on detritus of various origins (up to 75%) and small benthic invertebrates such as crustaceans and gastropods (Chong & Sasekumar 1981, Robertson 1988). Stable isotope-based studies have been used to identify the ultimate sources of nutrition for *F. merguiensis* juveniles in several North Queensland systems including rivers (Loneragan et al. 1997), estuarine creeks (Abrantes et al. 2015), floodplain pools (Sheaves et al. 2007a, Abrantes & Sheaves 2009b) and semi-enclosed coastal channels (Abrantes & Sheaves 2009a). Although those studies suggest that mangroves can be important contributors to juvenile nutrition, but no study has confirmed if this source is of critical importance or if F. merguiensis juveniles can exclusively rely on other sources when mangroves are not present. Stable isotope analysis of carbon (δ^{13} C) and nitrogen (δ^{15} N) are useful in coastal food web studies because (1) they often differ among different types of primary producers (e.g. seagrass, mangroves, microalgae; particularly for δ^{13} C) (France 1995, France 1996) and (2) because they change predictability as they are passed on from food source to consumer (DeNiro & Epstein 1978, 1981, McCutchan et al. 2003). Stable isotope analysis can therefore be used to determine if mangroves and/or saltmarsh are crucial sources of nutrition to F. merguiensis juveniles in North Queensland estuaries, so that the trophic importance of these habitats can be evaluated.



Figure 1. Simplified schematic food web leading to *F. merguiensis* at the mangrove-dominated Hinchinbrook Channel, North Queensland (adapted from Abrantes and Sheaves (2009a)). 1 – mangrove detritus; 2 – microphytobenthos; 3 – green filamentous algae; 4 – seagrass, seagrass detritus and seagrass epiphytes; 5 – phytoplankton; 6 – upper intertidal benthic fauna; 7 – lower intertidal and subtidal benthic fauna; 8 – zooplankton. Figures and symbols courtesy of the Integration and Application Network, University of Maryland Centre for Environmental Science (ian.umces.edu/imagelibrary)

In the current study, we **(a)** use measurements of biomass density in a typical North Queensland estuarine system to assess the suitability of the currently available methods of biomass/production/productivity estimation for the particular case of the banana prawns *F. merguiensis* in this region. In particular, we evaluate the potential to produce valuable estimates, in terms of logical constrains and estimated error structure, at three spatio-conceptual scales: (i) estuary reach, (ii) the whole estuary seascape, and (iii) the contribution of the estuary to the offshore exploited stock. Note that even if/when high quality estimates of biomass production are available, more information is likely to be required to truly assess productivity. This is because even habitats with low occupancy can have a trophic importance through the provision of essential nutrients needed for growth and development. Consequently, to identify the main sources of nutrition supporting *F. merguiensis* juveniles, we also **(b)** used stable isotope analysis of carbon (δ^{13} C) and nitrogen (δ^{15} N) to investigate the importance of mangroves and saltmarshes as ultimate sources of nutrition to *F. merguiensis* in four North Queensland estuaries.



Saltmarsh and mangrove habitat near Cairns, North Queensland. Photo credit: Ross Johnston

3.3 Methods

Evaluating F. merguiensis productivity

The productivity part of this study was conducted in Alligator Creek (Figure 2), a short and narrow (maximum width ~150 m) creek typical of Australia's wet-dry tropics. The wet-dry tropical climate is characterised by distinct and short wet seasons and long dry seasons when very little rainfall occurs. Alligator Creek is mostly bordered by mangrove forest, particularly in the downstream reach, and saltmarsh also occurs in some areas (Table 1). Tides are semi-diurnal (maximum range ~4 m). Sampling was conducted in three estuarine reaches: downstream, mid-estuary and upstream, each sampled over a length of ~1.5 km. The downstream reach was the area immediately inside the estuary mouth, the upstream reach was the limit of navigation of our 4.3 m boat, and the mid-estuary was approximately half-way between the downstream and upstream reaches.

Table 3. Proportion of edge habitat (mean ± SD) in the downstream, mid-estuary and upstream reaches ofAlligator Creek.

	Proportion of edge habitat (%)	
Reach	Mangrove	Saltmarsh
Downstream	94 ± 7	6 ± 7
Mid-estuary	66 ± 13	34 ± 13
Upstream	79 ± 16	21 ± 16



Figure 2. Map showing the locations of the study sites in North Queensland.

Sampling was conducted over the 2015-2016 wet season (December 2015 to April 2016), to encompass the period of high *F. merguiensis* juvenile abundance in northern Australia (Staples & Vance 1986, 1987). Prawns were captured during 11 trips with a 5 mm mesh monofilament draw-string cast net (sampling diameter: ~2.4m; sampling area: ~4.5m²) deployed from a small boat by an experienced operator. Sampling was done during the lower part of the tides when animals are forced out of the edge habitats and into the main channel. During each trip, 15, 20 or 30 cast net replicates were taken along the creek banks (\leq ~2m from creek margin) at each of the three reaches. All prawns collected were stored in ice. In the laboratory, prawns were identified and *F. merguiensis* counted, measured and weighed. Numerical density (in ind.net⁻¹) and biomass density (in g.net⁻¹) were then calculated for each reach, and used to estimate total abundance and total biomass per reach. Here, measured densities per net were used as representative of the area along the 2m wide creek margins, as *F. merguiensis* typically occur along shallow water creek edges (Johnston & Sheaves 2007). The remaining estuarine area (i.e. the middle of the creek, beyond the 2m wide edges) was considered to contain few prawns, i.e. densities were considered to be close to zero (R. Johnston pers. com.). The length-weight relationship (Figure 3), a crucial parameter for production estimates was also estimated, and relative abundance-by-length was analysed for each reach and trip to identify movements among reaches and recruitment/emigration events.



Figure 3. Length-weight relationship for *F. merguiensis* collected at Alligator Creek: $W = 2E-0.6 \times L3.2$, $R^2 = 0.98$ (n = 1006).

Sources of nutrition

Fenneropenaeus merguiensis juveniles and a range of available primary producers were collected from four representative systems in North Queensland: Deluge Inlet, Cocoa Creek, Doughboy Creek, and two un-vegetated semi-isolated floodplain pools in the Ross River estuary, one surrounded by mangrove vegetation and one by saltmarsh, with only a few small mangrove trees present (Figure 2). Deluge Inlet is a mangrove-dominated system that flows into the Hinchinbrook Channel. Seagrass beds occur at the mouth of the inlet and in the Channel. Cocoa and Doughboy creeks are short and narrow mangrove-fringed systems typical of the North Queensland coast. Mangrove forest is more abundant at Doughboy Creek, with a percentage cover of 50% compared to 19% at Cocoa Creek (percentage cover calculated for the area within 1km from the creek margins) (Abrantes et al. 2015). Saltmarsh occurs landward of the mangrove fringe and seagrass is also present at the mouth of Cocoa Creek. The two Ross River floodplain pools are relatively small and shallow (<1m at low tide) and are intermittently connected at spring tides to other pools and to the main estuary through narrow channels. These different systems were considered as they encompass the range of systems and habitats available to *F. merguiensis* juveniles in North Queensland.

Juvenile sizes ranged between 30 and 45 mm TL, with exception of the Hinchinbrook Channel, where it was only possible to collect larger (50-55 mm TL) juveniles. Details of animal collections and stable isotope sample processing and analysis can be found in Abrantes and Sheaves (2009a, b) and Abrantes et al. (2015).

To quantify the importance of the different primary producers to *F. merguiensis* juveniles, stable isotopebased Bayesian mixing models were run, using the package simmr (Stable Isotope Mixing Model in R v.3; Parnell et al. 2013) in R (R Development Core Team 2013). Details of this model can be found in Parnell et al. (2010, 2013). Trophic discrimination factors of $1.0 \pm 0.5\%$ for δ^{13} C and $2.8 \pm 0.5\%$ for δ^{15} N were used, as appropriate for non-acid treated muscle tissue (McCutchan et al. 2003), and juveniles were considered to be of trophic level of 2.5 (Abrantes & Sheaves 2009b). Results on the importance of mangrove and saltmarsh were considered as indicators of the likely trophic value of these habitats to *F. merguiensis* juveniles.

3.4 Results

Evaluating F. merguiensis productivity

Because *F. merguiensis* juveniles occur mostly along the shallow, 2m wide, edges (Johnston & Sheaves 2007), with few prawns occurring beyond this area (R. Johnston pers. com.), we focus our results and assessments on densities (per net) within the 2 m creek margins, rather than average densities over the overall creek area.

Both average numerical density and biomass density varied greatly among trips and reaches (Figure 4). In general, few prawns were captured in the first two trips (mid-December 2015), but catches increased from the third trip onwards (Figure 4). Catches were typically much higher in the downstream reach than in the mid-estuary and the upstream reaches. In the upstream reach, average numerical and biomass density were very low, generally <5 ind.net⁻¹ and <3 g.net⁻¹ respectively (Figure 4), corresponding to <3,000 juveniles, weighing ≤ 4 kg, for the overall upstream reach area. Only in trips 8 and 9 was upstream abundance (15,136.2 and 7,693.5 individuals respectively) and biomass (5.3 and 7.2 kg respectively) higher, suggesting a peak in abundance and biomass prior to emigration from the estuary.



Figure 4. Average density (top) and biomass density (bottom) of *F. merguiensis* in Alligator Creek (for the 2m wide edge area) over the 2015-2016 wet season, and rainfall recorded for Townsville (the closest weather station, ~27 km away) for the same period (Bureau of Meteorology, www.bom.au, accessed 09/02/17). For rainfall, values correspond to the total rain (mm) that fell between that date and the previous sampling trip, apart from Trip 1, for which rainfall of the previous 15 days is indicated. Trip numbers are indicated in the top panel.

For the overall Alligator Creek estuary, and based on the mean abundance/biomass values for each reach, we estimated a maximum abundance of $154,044 \pm 1,898$ SE individuals (for Trip 9) and a maximum biomass of 369.3 ± 1.3 kg (Trip 8).

Despite the large number of replicates taken at each reach/time (up to 30; see Figure 6), there was very high variability in numerical and biomass density among replicate samples (Figure 5). For example, in Trip 9 (08/03/16), the trip with highest downstream numerical density (see Figure 4), downstream catches ranged from 0 to 185 ind.net⁻¹ (Figure 5) even though the different replicate samples were collected from the same type of habitat. This maximum density, if considered as representative of the overall downstream edge area, would lead to an estimate of total abundance of 782,229 individuals, with a biomass of 2,011 kg for this area, showing the substantial variation in spatial distribution and the importance of using accurate values for the estimation of total abundance and biomass.



Figure 5. Box and whisker plots showing the median (line within the boxes), interquartile range (boxes), 10th and 90th percentiles (whiskers) and outliers (x) of numeric density (number of individuals per net; top) and biomass density (weight per net; bottom) of *F. merguiensis* collected in Alligator Creek (for the 2 m wide edge area only) over the 2015-2016 wet season, illustrating the variability in catches among sampling trips (Trip 1 to 11; see trip dates in Figure 3), estuary reaches (D = downstream; M = mid-estuary; U = upstream reach), and among replicates.

In the downstream reach, numerical density increased towards the peak of the rainy season (March 2016), up to a maximum of an average 27 ind.net⁻¹ (Figure 5). This led to an estimated abundance of 113,881 individuals for the overall downstream area, weighing 250 kg. This average numerical density (per net) corresponds to 41.1 ind.m² for the 2m band along the creek edges, but only to 1.8 ind.m⁻² for the overall downstream reach area, as very few F. merguiensis occur in the middle of the creeks (R. Johnston pers. com.). As with numerical density, biomass density also increased from very low average values (<5 g.net⁻¹) at the end of 2015 to a maximum of ~80 g.net⁻¹ (~337.4 kg for the whole reach) in February 2016 (Trip 8; Figure 4), likely because of an increase in prawn abundance combined with growth of individuals. This average biomass density of 80 g.net⁻¹ corresponds to a density of 17.8 g.m⁻² along the margins, but to only 0.8 g.m⁻² if considering the overall downstream reach area. This illustrates the importance of the use of adequate methodological approaches and adequate results presentation/interpretation. For example, if the cross-creek distribution of F. merguiensis juveniles was not known, replicate samples would likely be taken from different distances from the edge, including in mid-creek, and average values would be calculated based on samples from all areas, leading to erroneous density, abundance and biomass estimates. This also shows the importance of creek width and edge convolution on these numerical/biomass density estimates.

After the peaks in density, values decreased sharply (Figure 4), probably because of emigration of larger juveniles out of the estuary, a likely response to heavy rainfall, a known driver of *F. merguiensis* emigration from estuaries (Staples & Vance 1986, 1987). Although this emigration resulted in a decrease in biomass, continuing recruitment resulted in a further increase in numerical density, beyond the date of peak biomass density. This explains the fact that the peak of biomass density occurred before the peak in numerical density (Figure 4). Indeed, it can be seen in Figure 6 that in Trip 8 (17/02/16) two distinguishable cohorts were present, but the largest (>100 mm total length (TL)) cohort was almost completely absent on the following trip (Trip 9, 08/03/16) (Figure 6), implying that many larger individuals moved out of the estuary with the rains that fell in the first part of March (see Figure 4), leading to a decrease in biomass density (Figure 4).



Aerial photo of the Alligator Creek estuary, Townsville, Queensland. Photo credit: Carla Wegscheidl/DAF



Figure 6. Size frequency distribution of *F. merguiensis* captured in the downstream, mid-estuary and upstream reaches of Alligator Creek, between December 2015 and April 2016 (i.e. over the wet season). Sample size, in brackets, is also indicated for each trip, and corresponds to the number of replicates taken from each of the three reaches.



Figure 6 (contd.) Size frequency distribution of *F. merguiensis* captured in the downstream, mid-estuary and upstream reaches of Alligator Creek, between December 2015 and April 2016 (i.e. over the wet season). Sample size, in brackets, is also indicated for each trip, and corresponds to the number of replicates taken from each of the three reaches.

There was also a smaller peak of rainfall at the end of December 2015 that again coincided with a small peak in *F. merguiensis* numerical (average 9 ind.net⁻¹) and biomass density (average 20 g.net⁻¹) for the downstream reach (Figs. 4). This also coincided with small dips in the mid-estuary reach, suggesting that animals moved from the mid-estuary to the downstream reach with the rainfall, before some moving out of the estuary and others dispersing again through the more upstream areas. Indeed, it can be seen in Figure 6 that while in Trips 2 (22/12/15) and Trip 4 (05/01/16) prawns were present in the mid-estuary and upstream reaches in substantial density, in Trip 3 (30/12/15) very few individuals were captured in those reaches (Figs. 4-6). This was likely due to the reduction in salinity in the more upstream parts of the estuary due to the rainfall that occurred between Trips 2 and 3 (see Figure 4). Note that salinity changes due to rainfall are less pronounced at the downstream areas due to effective tidal mixing. In Trip 4, it can also be seen that the largest sizes (>~65 mm TL) that were present in Trip 3 are no longer present in abundance in any of the three reaches (Figure 6), suggesting emigration of those larger individuals out of the estuary after the rains.

Both numerical density and biomass density in the mid-estuary were typically much lower than in the downstream reach, but a similar pattern was present, with both parameters increasing towards the peak of the wet season, and decreasing after the main rains (Figure 3).

Sources of nutrition

Stable isotope results show that *F. merguiensis* juveniles rely on a range of available sources (Figs. 7-9). The 95% credibility intervals (CI) of mangrove contribution included 0% for all estuaries, meaning that it was not possible to positively identify a critical contribution of mangroves. However, mangroves could be important contributors particularly for Doughboy and Cocoa Creeks, where the upper limits of the 95% CI were relatively high (67% and 57% respectively; Figure 8). For Deluge Inlet (Figure 7) and for the mangrove-lined pool in the Ross River floodplain (Figure 9), mangrove contribution was limited, with 95% CIs of only 2-26% and 0-17% respectively.



Figure 7. Posterior density curves of the proportional source contributions to *F. merguiensis* juveniles at Deluge Inlet, based on stable-isotope based Bayesian mixing model.



Figure 8. Posterior density curves of the proportional source contributions to *F. merguiensis* juveniles at Cocoa Creek (left panel) and Doughboy Creek (right panel), based on stable-isotope based Bayesian mixing models.



Figure 9. Posterior density curves of the proportional source contributions to *F. merguiensis* juveniles at a mangrove-lined (top) and a saltmarsh-lined (bottom) floodplain pool of the Ross River floodplain, Townsville, based on stable-isotope based Bayesian mixing models.

Saltmarsh and its epiphytes were also important in the saltmarsh-dominated floodplain pool of the Ross River estuary (15-82%; Figure 9), but its importance was limited at Cocoa and Doughboy Creeks (1-42% and

1-43% respectively; Figure 8) where saltmarsh availability is much lower. Similarly, seagrass has some importance in Deluge Inlet (13-55%; Figure 7), where extensive seagrass meadows occur (Lee Long et al. 1998), but the importance of this source was limited in Cocoa Creek (2-42%; Figure 8), where seagrass cover is sparse and limited to the creek mouth.

3.5 Discussion

Suitability of available methods of biomass/production/productivity estimation

Sources of variability in biomass density estimates

Overall, there was high spatial and temporal variability in F. merguiensis juvenile numerical density and biomass density in the Alligator Creek estuary. The reasons for this variability are complex, but not unexpected. Evaluating patterns of abundance of mobile organisms is always difficult because, rather than a few prominent parameters determining how many individuals occur in a particular place at a particular point in time, local abundance is the result of the interaction of a wide array of factors, what has been described as a causal thicket (Harris & Heathwaite 2012). Causal thickets are characterised by the presence of many interacting and synergistic drivers that change in their relative importance in determining the location and abundance of individuals from time to time and place to place, and that are often strongly aliased and invariably include substantial indeterminacy (Harris & Heathwaite 2005). Some of the complicating drivers are obvious, e.g. numbers and biomass are simultaneously influenced by emigration, recruitment, within-system movements and mortality, as well as changes in the physical environment such as rainfall-driven salinity depression. The influences of these factors are clear in the present case study. For instance, simultaneous growth and emigration of individuals already in the population, coincident with recruitment of new individuals resulted in changes in biomass density apparently decoupled from changes in numeric density per net. Other drivers are likely to be important over longer time scales. For instance, this study was conducted during an El Niño event, during which rainfall in North Queensland was well below average (Bureau of Meteorology, www.bom.gov.au, accessed 09/02/17), so results presented here are unlikely to be representative of those under different climate phases. Indeed, the structure of nekton assemblages in north eastern Australian estuaries (Sheaves et al. 2007b) and the sources of nutrition that support them (Abrantes & Sheaves 2010) are fundamentally different during extended wet versus extended dry periods.

Substantial variations in *F. merguiensis* distribution and abundance have been reported by several previous studies. Although juveniles are known to mostly occur along mangrove-lined banks (Vance et al. 1990, Kenyon et al. 2004), there are considerable variations in density both among and within estuaries (Sheaves et al. 2012b). For example, within a system, densities in smaller creeks are often higher than those in the main river (Vance et al. 1998, Vance et al. 2002, Kenyon et al. 2004). Depth, bank slope and the proximity to mangroves are also important factors affecting densities (Vance et al. 1990, Vance et al. 2002). Moreover, juveniles can move up to 200 m into mangrove forests at high tide (Vance et al. 1996, Rönnbäck et al. 1999, Vance et al. 2002) and use the adjacent banks during the low-tide periods when waters are mostly outside of the forests (Robertson 1988; Vance et al. 1996, 2002), meaning that densities vary through the tidal cycles. Despite their willingness to move into mangroves, at high tides, densities of *F. merguiensis* have been found to be higher at creek edges than in the inside of mangrove forests (Meager et al. 2003) and there are also spatial variations in density within the mangrove forest, at different distances from the mangrove edge (Vance et al. 2002).

Evaluating the importance of particular habitat types to prawn nutrition

There is no evidence that mangroves, saltmarsh or other particular wetland habitat types are of critical trophic importance to *F. merguiensis* juveniles. The stable isotope analysis in the current study underlines that *F. merguiensis* juveniles can use a range of primary producers as sources of nutrition, and that the importance of the different sources depends on the relative availability, as previously reported for other estuarine species (e.g. Bouillon et al. 2004, Abrantes et al. 2013). Indeed, the importance of mangrove-derived carbon for *F. merguiensis* has been found to depend on the extent of mangrove forest in the

estuaries, and this importance is significant only in systems where more productive habitats such as seagrass beds are not present (Abrantes et al. 2015). In systems with seagrass beds, even if extensive mangrove forests are present, the importance of mangrove carbon is limited and *F. merguiensis* juveniles typically rely mostly on a range of aquatic sources (Loneragan et al. 1997, Abrantes & Sheaves 2009a, Abrantes et al. 2015).

The lack of evidence that any particular habitat type is critically important to the nutrition of *F. merguiensis* juveniles means that the value of the habitats used (e.g. mangroves) is most likely primarily related to physical conditions, rather than to a trophic function. These physical conditions could be a function of the presence of mangroves themselves and/or to the provisioning of vast intertidal banks and/or shallow water habitat, factors that have been reported as important predictors of *F. merguiensis* catches in North Queensland estuaries (Sheaves et al. 2012b). The lack of a specific trophic relationship between particular wetland habitat types and *F. merguiensis* juveniles adds weight to the argument that productivity outcomes for *F. merguiensis* are likely to be most usefully and validly assessed at the whole-of-estuary scale, and that it will rarely be possible to link biomass to specific wetland units

Evaluating the value and validity of productivity estimates at different scales

It is clear that quantifying numerical density, population numbers, biomass density and total biomass is complex, even if an adequate sampling design is combined with fine-scale mapping. Consequently, although the methodological approach used here was useful to identify important habitat use patterns, the high variability among replicate samples and the logical issues involved mean that for most tropical estuary situations it will be impossible to obtain numerical or biomass density estimates with low variability and for which the particular regulating factors are unambiguously known.

The potential for mobile fauna such as *F. merguiensis* to move longitudinally in an estuary over short periods of time creates an important additional problem if the aim is to link increase in biomass to particular areas of the estuary or to particular habitat units. Since biomass density estimates are a critical to calculate productivity, and a detailed understanding of movement is necessary to allow that productivity to be attributed to specific habitats, there are likely to be very few cases where biomass increase or productivity can be reliably attributed at the within-estuary scale. It is however important to note that many of the sources of indeterminate variability are related to among-habitat within-estuary scales. This implies that carefully collected data that cover the whole estuary are likely to provide a means for understanding, addressing and integrating variability, for instance due to movement among reaches. Thus, it is possible that in many cases estimates of whole-of-estuary productivity i.e., estimates that integrate across the whole estuary are achievable.

A diversity of factors need to be taken into account when estimating biomass and production for *F. merguiensis'* estuarine populations, in evaluating the contribution of particular tidal wetlands to biomass production, and ultimately in linking particular habitat units to the productivity of offshore fisheries stocks (Figure 10). Firstly, a detailed sampling of the available habitats at different temporal and spatial scales is critical, but this is logistically difficult and time consuming. Indeed, penaeid densities (Minello 1999, Shervette & Gelwick 2008), biomass and production (Minello et al. 2008) vary among habitats and depend on a range of factors such as flooding patterns (duration, frequency and depth), the assemblage of habitats, habitat fragmentation (which influences amount of marsh/water edge), overall arrangement of habitats in the coastal seascape, salinity and temperature (Rozas & Minello 1999, Minello et al. 2008, Roth et al. 2008, Baker et al. 2015). Therefore, not only must the area of the different habitats be adequately measured, but the access to these habitats also needs to be considered, and this is influenced by topography, hydrology, and connectivity between the different habitats (Zimmerman et al. 2002). Different scales of habitat should also be taken into account (e.g. edge vs. different distances from the edge). All these factors can affect habitat availability and habitat use and value, and need to be considered in biomass and production estimation models.



Figure 10: The complexity of estimating (a) production and value for one habitat, (b) production for one system (estuary), (c) fisheries value of one system (measured as contribution to offshore fisheries stock), and (d) indicative cost-benefit function model showing curves for changes in certainty of meaningful interpretation, cumulative error of estimates and cumulative difficulty of collecting valid data, versus the likelihood of a valid estimate. The approximate potential range of estimates (a), (b) and (c) are indicated below the figure. *Note*: The curves are illustrative only and not intended to represent specific functions. They are based on the logic that (i) the certainty of meaningful interpretation is likely to be approximately sigmoidal in shape and to decrease, while (ii) the cumulative error and difficulty of collecting valid data are both likely continually increase, as the number and complexity of parameters to be estimated increases.

Growth rates and natural mortality rates are other key parameters needed to estimate penaeid production. Growth rates can be estimated using methods such as mark-recapture (e.g. Knudsen et al. 1996, Webb & Kneib 2004, Braccini et al. 2013), length-frequency analysis and modal progression (e.g. Staples 1980b, Haywood & Staples 1993, Watson et al. 1993), as well as mesocosm (e.g. Rozas & Minello 2009, 2011) and caging experiments (e.g. Shervette & Gelwick 2008, Baker & Minello 2010). Mortality estimates can also be derived using mark-recapture studies (e.g. Knudsen et al. 1996, Montgomery et al. 2012, Mace III & Rozas 2015) or catch-curve or cohort progression analyses (Wang & Haywood 1999, Baker & Minello 2010, Mace III & Rozas 2015).

These biological parameters are difficult to estimate for juveniles in coastal wetlands. For example, cohort analysis is difficult to apply due to long periods of continuous recruitment and when there are multiple waves of recruitment and/or immigration (Staples 1980a, Haywood & Staples 1993, Baker & Minello 2010). Also, penaeid juveniles only use estuarine nurseries for relatively short periods of time, and typically use different parts of the estuary as they grow, so different sized juveniles occur in different areas (Vance et al. 1998, Vance et al. 2002, Kenyon et al. 2004) making it difficult to follow cohorts or to relate increase in biomass to nutrition derived from particular habitat units. Tag-recapture methods are also difficult to use due to the small size of prawn juveniles, and because of their high abundances and very high natural mortality rates. Moreover, penaeid fecundity (Crocos & Van der Velde 1995) and recruitment (Garcia 1985, Vance et al. 1985, Vance et al. 1998, Zhou et al. 2009) often vary seasonally and among years; growth rates can vary with temperature (Staples 1980a), salinity and food availability (Rozas & Minello 2011); juvenile densities not only vary seasonally but also through day-night and tidal cycles (Vance & Staples 1992, Griffiths 1999), and mortality rates vary with penaeid sizes and among years, habitat type and complexity, and with predator density (Minello et al. 1989, Macia et al. 2003). It is therefore clear that significant research is needed before we can accurately estimate F. merguiensis production for North Queensland estuaries, before we can validly attribute an economic value to the different juvenile habitats, and before we can estimate the likely benefits from habitat preservation or repair.

All these and a range of other considerations make estimating production and value a complex issue. Using our *F. merguiensis* example, we could consider three different types or stages of estimates (Figure 10): (i) estimates of production and value of just one estuarine habitat (Figure 10a), (ii) estimates of production from one (estuarine) system (Figure 10b), and (iii) estimates of fisheries values of one system to the fishery (i.e. the offshore stock) (Figure 10c). Each of these is dependent on having a range of data and on the quality and validity of those data. Estimating production from a single habitat requires a diversity of data (Figure 10a). Much of it is difficult to collect but potentially estimable (e.g. extent of habitat, species size frequency). Other necessary data (e.g. growth, mortality, recruitment and emigration rates, proportion of time spent in a habitat) are virtually impossible to collect with any available technology. Moreover, each of the estimates needed to calculate production is associated to a certain level of uncertainty (error), which is often substantial (note for instance the substantial variability in the density estimates (Figure 4)), each adding to the uncertainty of the combined estimate. The complexity and uncertainty mean that both the cumulative difficulty in collecting valid data and the cumulative error of the estimates is high, leading to a low likelihood of a valid estimate and low certainty of meaningful interpretation (Figure 10d).

Estimating the production from one whole system (Figure 10b) is likely to involve a lower level of uncertainty (assuming that prawns remain within the overall estuary for their juvenile phase) because some of the more problematic estimates (e.g. proportion of time spent in each habitat, the growth advantage due to refuge value) are not required, and the additional estimates (extent of each reach) will usually be viable to estimate. Consequently, the certainty of meaningful interpretation and likelihood of a valid estimate are likely to be higher than for estimates of the value of individual habitats (Figure 10d). However, while estimating the production from a whole system is likely to be more feasible than assessing production for individual habitats, such estimates are sensitively dependent on the quality of data and on the assumptions underpinning the various estimates. Consequently, both these provisos need to be

exhaustively assessed and the value and reliability of estimates critically appraised before any quantitative use could be made of them.

The final stage of estimates is to evaluate the contribution of prawns from one system to the fisheries stock (for F. merguiensis this would be in the offshore adult population) (Figure 10c). This is a potentially useful estimate because it allows the attribution of a dollar value to a particular system. However, these estimates require the calculation of some problematic values. For instance, juveniles from several estuaries usually contribute to the same offshore fishery. Moreover, recruitment can vary greatly even among adjacent estuarine systems due to the hydrological pattern and the spatial arrangement of habitats within systems, and systems within the area (Staples 1979, Staples & Vance 1987, Kenyon et al. 2004), meaning that different estuaries will likely contribute differently to the offshore fishery. Within the same system, different cohorts can also contribute differently to the offshore fishery, and this can vary among years (Haywood & Staples 1993). Moreover, the same species can have differences in recruitment patterns in different fisheries, and can enter the fisheries at different ages and growth phases depending on the system (Watson et al. 1996). As a result, it is vital to know the proportion of the stock contributed by each system. Some biochemical techniques (e.g. stable isotope analysis) have potential to assist with this question but these are only useful where there are measurable differences in biochemical composition among the potential source estuaries. Even more difficult to estimate is the loss of individuals (and therefore biomass) during migration to the adult grounds and prior to sampling/fishing. The substantial difficulty of estimating these parameters probably makes the likelihood of a valid estimate at least as unlikely as obtaining a valid estimate for a single habitat (Figure 10d).

Added to the difficulties associated with valid data collection and parameter estimation, are the critical limits placed on estimation by the presence of pervasive complexity that puts limits on what is possible to predict (Harris & Heathwaite 2005). For instance, it is now understood that cause and effect are rarely linearly related but rather under the influence of causal thickets (Harris & Heathwaite 2012) and extensively aliased (Harris & Heathwaite 2005). Additionally, even in well-studied systems there may be pervasive irreducible uncertainty (Lo & Mueller 2010). Thus, it is likely that in many case detailed levels of estimation will be unachievable.

In conclusion, even for a well-studied species such as *F. merguiensis* there is still a deficit in the information needed to make a precise species-productivity link, and substantial research is still needed before the available broad-scale biomass estimates can be converted to valid estimates of total biomass and production, and before these can usefully be related to particular habitats. Being able to estimate productivity and attribute it sensitively to particular units (a wetland or an estuary) is a very important aspirational goal. However, with current technologies, and technologies likely to come on line in the short to medium term, for most situations it is probably unattainable at an acceptable level of certainty.

3.6 References

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CHAPTER 4 - UTILISATION OF A RESTORED WETLAND BY A COMMERCIALLY IMPORTANT SPECIES OF PENAEID SHRIMP

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4.1 Abstract

Penaeid shrimp represent an important group of valuable exploited species that are known to either directly utilise saltmarsh habitats, or utilise saltmarsh-derived productivity. Consequently, both areal coverage and primary productivity of saltmarsh habitat has direct consequences for the productivity of these important fisheries, and they are likely to be key beneficiaries of habitat repair. This study aimed to establish quantitative estimates of abundance of School Prawn, *Metapenaeus macleayi*, across a recovering wetland system; Hexham wetland in the Hunter River. Six surveys were conducted across the wetland using a specialized benthic sled, and absolute abundance of School Prawn was estimated. School Prawn were consistently more abundant in certain areas of the wetland (the highest abundance site supported 1,017 prawns per 100 m²), and the average density across the wetland was 244 prawns per 100 m². All areas of the wetland (except the area closest to the wetland mouth) supported the full range of size classes, and multiple cohorts of prawns moved through the system during the sampling program. The asymmetry observed in the distribution of prawns across the wetland is likely due to a combination of water quality and inter-specific interactions. These results show that the recovering wetland is supporting a high abundance of School Prawn, and our estimates of recruitment for the species will be useful in gauging the potential fisheries productivity benefits of habitat repair.



Research vessel travelling through a marsh channel on the hunter river estuary. Photo Credit: Matt Taylor

4.2 Introduction

Estuaries contain a mosaic of different habitats, and are productive environments that provide food and shelter for a diverse assemblage of fishes (Beck et al. 2001). Many species rely on estuarine habitats for some or all of their life cycle, and this association may be either opportunistic or dependent (Elliott et al. 2007). In most estuarine systems, this includes a number of commercially and recreationally exploited species, which are either harvested directly from these habitats as adults or reside in these habitats as juveniles and are harvested elsewhere (Taylor et al. 2017a). As such, these estuarine habitats provide important ecosystem services with tangible economic outcomes obtained through recreational and commercial fisheries for both fish and crustaceans (Lenanton and Potter 1987).

Saltmarsh habitats include a diversity or producers that support considerable levels of estuarine primary production (e.g. Groenendijk 1984). These habitats can also fulfil a dual role of providing food (e.g. Paterson and Whitfield 1997) and refuge (e.g. Paterson and Whitfield 2000) for juvenile fishes, direct foraging habitats for predators (Rozas and Odum 1988), and supporting a diverse range of consumers that can link primary producers with higher trophic levels (e.g. Becker and Taylor 2017; Mazumder et al. 2011). Penaeid shrimp represent an important group of exploited species that are known to either directly utilise saltmarsh habitats (e.g. Fry 2008), or utilise saltmarsh-derived productivity (Taylor et al. in review). In some systems, fishery productivity of these species is inextricably linked to salt marsh habitats (e.g. Gulf of Mexico, Zimmerman et al. 2000). Consequently, both areal coverage and primary productivity of saltmarsh habitat has direct consequences for the productivity of these important fisheries.

Cumulative anthropogenic impacts have adversely affected the diversity and productivity of estuarine systems (Lotze et al. 2006). Much of this damage has occurred through the loss of saltmarsh, or the alteration of wetland ecosystem function. Saltmarsh loss can be attributed to various factors which includes the disruption of natural depositional processes through activities such as dredging and flood mitigation (Turner 1997). Loss of saltmarsh ecosystem function can occur through disruption of tidal connectivity and associated drainage of wetlands to support other land uses (such as grazing, Rogers et al. 2015). Not surprisingly, recent case studies have implicated the historic loss of these habitats in the declining productivity of inshore fisheries, including penaeid fisheries (e.g. Barbier and Strand 1998; Creighton et al. 2015). It follows that efforts to repair these habitats are likely to produce concomitant positive impacts on the productivity of penaeid species (Rozas et al. 2005).

School Prawn *Metapenaeus macleayi* are a penaeid prawn (=shrimp) species that supports a large estuarine and inshore fishery off eastern Australia. Productivity of the species has been associated with both high freshwater flows (Glaister 1978), and estuarine wetland extent (Creighton et al. 2015; Saintilan and Wen 2012). Across the species range, fishing effort for the species is generally greatest within the Clarence River estuary, the Hawkesbury River estuary and the Hunter River estuary (Figure 1). The Hunter River estuary is significant in that it has been the subject of concerted efforts to repair degraded estuarine wetlands over the last 20 years. The impact of these habitat repair efforts for School Prawn are unclear, with recent studies in this system presenting conflicting findings with respect to the utilisation of these wetlands by School Prawn (Boys and Williams 2012; Taylor et al. 2017b). Thus, there is a need to better understand usage of these repaired wetlands in the Hunter River, and use this information to both assess the impacts of repair efforts and refine future work. This study presents the results of a sampling program aimed at establishing quantitative estimates of School Prawn abundance across the recently restored Hexham wetland, and evaluating potential drivers of these patterns.



Figure 1 Map of the Hunter River estuary and the Hexham wetland system. Inset a. shows the location of the Hunter River estuary on the Australian east coast, and inset b. shows the Hunter River estuary, including mangrove (green polygons) and saltmarsh (brown polygons) habitats. Inset c. shows the Hexham wetland, with red circles indicating samples undertaken in this study.

4.3 Materials and Methods

Study area

The Hunter River estuary is a large, wave-dominated barrier estuary located in New South Wales, Australia (**32°54'S 151°46'E**). The lower estuary is heavily urbanised, but contains extensive mangrove and saltmarsh habitats concentrated within three major wetland systems: 1) Tomago wetland; 2) Kooragang wetland; and 3) Hexham wetland (Figure 1). There is no longer seagrass present within the estuary. The estuary supports a significant population of School Prawn and Eastern King Prawn (*Penaeus plebejus*), with the former supporting a 50-70 tonne per annum estuarine fishery (Taylor et al. 2017b). There is also considerable harvest of various finfish and crab species (Taylor and Johnson 2016).

During the early-mid 20th century the wetland systems in the estuary were significantly degraded, primarily through flood mitigation barrages severing connectivity with the estuary, and the installation of dykes, drains and drying of the wetlands. Rehabilitation projects have now been carried out on these systems, initially targeting the Kooragang wetland (undertaken from 1990-1996, Williams et al. 2000), followed by the Tomago (undertaken from 2007-2011, Rayner and Glamore 2010) and Hexham wetlands (from 2008-2013, Boys 2016). The most recent studies on School Prawn in this estuary (Taylor et al. 2017b; Taylor et al. 2016) have shown that all three wetlands represent effective juvenile habitat (Dahlgren et al. 2006) for School Prawn, and also that the downstream areas of Hexham wetland are an important habitat for juvenile Eastern King Prawn.

Sampling design and approach

Sampling involved a quantitative assessment of juvenile School Prawn abundance across the Hexham wetland (Figure 1). Six sampling sites were selected to cover the wetland system, and were surveyed monthly over the sampling period (Spring 2016 to Autumn 2017). Each site was sampled during each time point using 4 replicate tows of a 26B-6C sled net, which yielded a total of 144 samples over the study period. Sampling commenced after dawn, and a GPS waypoint was marked at the start and finish of each tow (to calculate tow-length). Depth and water quality (salinity, pH, turbidity [NTU], dissolved oxygen [mg L⁻¹] and temperature [°C]) were recorded at each site during each sampling period.

Sample processing and data analysis

Following landing, sled samples were immediately placed on ice and then frozen for later processing in the laboratory. Following thawing of samples, all organisms within the samples were sorted, identified and counted. All penaeid prawns were measured for carapace length (CL, mm) and weight (g). The tow length (m) was calculated using a Euclidean formula, and this was used with the gear dimensions and a gear efficiency estimate to standardise abundance estimates to School-Prawn-per-hundred-square-metres (SP 100 m⁻²). Patterns in School Prawn abundance were analysed using a two-factor ANOVA, comparing standardised abundance by Site (fixed, 6 levels) and Month (fixed, 6 levels). Carapace length measurements (mm) were also expressed as length-frequency distributions to examine evidence for differences in size structure among sites and months.

4.4 Results

School Prawn abundance estimates were highly variable among sites and months (Figure 2). Site 1 consistently had the greatest abundance, which averaged 1017 SP 100 m⁻², whereas sites 2-5 had much smaller average densities (~50 SP 100 m⁻²), and Site 6 had negligible densities of School Prawn throughout the study period (Figure 2). ANOVA indicated that most variation was due to factor Site ($F_{5,108}$ = 135.01; P << 0.01), but differences between months were also significant ($F_{5,108}$ = 5.59; P << 0.01). The relationship among sites was not consistent as evidenced in a significant Site by Month interaction ($F_{25,108}$ = 3.19; P << 0.01). While Site 1 had consistently higher abundance estimates (usually by one order of magnitude) over other sites, the source of the interaction was likely due to differences in the relationships between sites 2-6 among months (Figure 2). There were few obvious differences in water quality among sites throughout the study period (Figure 3). Sites 1 and 2 usually had lower salinity, and salinity levels at these two sites tracked relatively closely throughout the season. Site 1 and 2 also had lower pH and lower dissolved oxygen from October – December, however these values increased and were similar to other sites from December through to the end of the study period.



School Prawns captured during the sampling in the Hunter River Estuary. Photo Credit: Matt Taylor



Figure 2 Grouped bar plot (mean \pm SE) showing the absolute abundance of School Prawn present in sites and months sampled during the study. Site numbers correspond with those shown in Figure 1, and errors bars are standard.



Figure 3 Temporal trends in water quality measured throughout the study period, including (from the top) salinity, pH, turbidity, dissolved oxygen and temperature. Coloured lines correspond to sites shown in Figure 1, as indicated in the legend.

Due to the low numbers of prawns captured, monthly length frequencies could only be calculated for School Prawn at Site 1. When all months were considered together, there were no obvious differences in the distribution of lengths among size-categories at difference sites (Figure 4). Prawn generally ranged from 3mm to about 20mm carapace length (CL; mm) at all sites, with the distributions dominated by prawns in the 5-10mm size classes (Figure 4). At Site 1, there was some evidence for multiple cohorts present throughout the months sampled (Figure 5). In October, soon after recruitment commenced, the population was dominated by smaller prawns. In the following months, proportional abundance of larger sizes classes increased, likely reflecting growth of these prawns as the season progressed (Figure 5). Distributions appeared bi-modal during December, January and February, indicating that multiple cohorts moved through (recruited and emigrated) this site throughout these warmer months (Figure 5).



Figure 4 Length-frequency distributions for School Prawn among sites, for all samples pooled across the study period. The number of prawns represented in the distribution is indicated in the top-left of each panel. Site names correspond to Figure 1. All *x*-axes and *y*-axes are on the same scale.



Figure 5 Length-frequency distributions for School Prawn captured at Site 1, for each month of sampling. The number of prawns represented in the distribution is indicated in the top-left of each panel. Site names correspond to Figure 1. All *x*-axes and *y*-axes are on the same scale.
4.5 Discussion

This study presents the first quantitative (i.e. targeted sampling standardised for gear efficiency) estimates of juvenile School Prawn abundance in estuarine habitats. While previous studies have reported relative densities, these have principally used commercial gear targeted at larger sizes, and/or not corrected estimates for efficiency of the gear (e.g. Coles and Greenwood 1983; Rotherham et al. 2008; Ruello 1973a; Ruello 1973b). Consequently, there are few studies that report absolute abundances of School Prawn with which to compare our results. School Prawn has many aspects of its behaviour, physiology and life history in common with Brown Shrimp (*Farfantepenaeus aztecus*), which has been extensively studied in the context of wetland repair. One of the most significant papers published on the species in this context showed abundance of Brown Shrimp between 22,246 – 37,748 ha⁻¹, which equates to 222 – 378 Brown Shrimp 100 m⁻² (Rozas et al. 2005). This is similar to the average density across months and sites in the Hexham wetland, which equated to 244 SP 100 m⁻².

Patterns in abundance

School Prawn abundance estimates were highly asymmetric across the Hexham wetland system. Average abundance of School Prawn at Site 1 was 1016 SP 100 m⁻², which is the greatest abundance yet reported in the literature for the species. In contrast, abundance at sites 2-5 were lower, and abundance at Site 6 was negligible, and these patterns were relatively consistent across the sampling period. Furthermore, Site 1 contained a broad cross section of size classes, and multiple cohorts, so did not simply reflect a high number of newly recruited postlarvae. The specific factors driving high abundance of School Prawn at this site are not clear. Water quality data shows that while the salinity was marginally lower at this site (juvenile School Prawn readily recruit to brackish water, Ruello 1971), other water quality variables were within reasonable limits across all sites. That being said, water quality was measured at a relatively low temporal resolution, and always at the same point in the tidal cycle, and did not capture any of the tidal variability or other irregular variation caused by inflow from the catchment. Salinity alone is unlikely to be driving the patterns, as Site 1 and 2 had similar salinity, but Site 2 supported a much lower abundance of School Prawn.

Pinto and Maheshwari (2012) show responses in School Prawn populations to various water quality variables, including temperature and dissolved oxygen. Variation in response to temperature largely describes seasonal variation (with abundance greatest in spring/summer). Dissolved oxygen also varies in relation to temperature, but the low dissolved oxygen affecting patterns in School Prawn abundance in Pinto and Maheshwari (2012) was hypothesised to be driven by anaerobic fermentation. Prawns can be highly sensitive low hypoxic water (e.g. Brown Shrimp, Renaud 1986), and this may have contributed to the patterns we observed in Hexham wetland. Following reinstatement of full tidal connectivity in 2013, Hexham wetland has been in a state of flux as the freshwater aquatic vegetation (e.g. *Phragmites*) dies and is gradually replaced with saltwater species (e.g. *Avicennia marina., Sporobolous* sp., and *Sueda* sp.). This die off could be contributing excessive amounts of organic matter to different regions of the wetland, which could contribute to low dissolved oxygen and pH (Pinto and Maheshwari 2011). This is supported by the observation that Sites 2-5 lie within channels which drain a much larger portion of the wetland than Site 1.

If these water quality dynamics are present, they would be most obvious during the low tide when the influence of oceanic water is at its lowest. The patterns were certainly not evident in our data, as our measurements were always taken during the daytime, and never on the ebb tide. While Site 1 and 2 showed marginally lower pH and dissolved oxygen at certain times during the study period, this is likely reflecting the lesser influence of fresh inflowing seawater at these sites as they were furthest from the wetland entrance. At any rate, resolution of the above patterns and their role in driving School Prawn abundance will require further research, and water quality monitoring at a much finer resolution.

As Site 6 was the most downstream it was most exposed to clean inflowing seawater, so the patterns outlined above are least likely to affect this site. A potential explanation of the negligible School Prawn

abundance at this location may be found in recent work in the lower Hunter River estuary. Patterns in School Prawn abundance in this system appear to be inversely related to abundance of Eastern King Prawn; and lower Ironbark Creek (the main tributary draining Hexham wetland) supports some of the highest densities of Eastern King Prawn reported in the literature (Taylor et al. 2017b). While the proximal mechanism affecting this relationship is not known, it is possibly being driven by avoidance of areas by School Prawn where Eastern King Prawn are abundant. Partitioning of space between similar species has been shown previously in other crustaceans (e.g. mysid shrimp, Taylor 2008), but detailed reports for prawn species are rare. Wild juvenile Eastern King Prawn have been shown to outcompete other animals for space (e.g. Ochwada-Doyle et al. 2012), co-occurring Brown Shrimp and White Shrimp (*Farfantepenaeus setiferus*) have been shown to select different habitats (Minello and Zimmerman 1985). If interspecific interactions between School Prawn and Eastern King Prawn are occurring, this could potentially explain the absence of School Prawn from this site. Detection of these patterns would require adjacent sampling of Eastern King Prawn and School Prawn during both the night and day respectively, as these species have opposing diel activity.

Implications for fisheries productivity and future habitat rehabilitation

Hunter River School Prawn populations are supported by a range of different nurseries across the estuary (including multiple wetland habitats, Taylor et al. 2017b), however harvest of the species is concentrated within the north arm. Recent work has shown that when fishing pressure is high, the majority of prawns emigrating to spawn originate from the south arm, and likely fed from the Hexham wetland (Taylor et al. 2017b). Consequently, in years where harvest from the north arm is high, prawns originating from Hexham wetland likely underpin a significant proportion of the next generation of recruits. Therefore, a strong population of prawns in this wetland is important for ongoing recruitment to the stock

School Prawn are not the only species utilising the wetland system, with other commercially exploited species such as Yellowfin Bream *Acanthopagrus australis*, Sea Mullet *Mugil cephalus* and Dusky Flathead *Platycephalus fuscus* also using the recovering wetland (Boys 2016). Recent work has shown that outwelled saltmarsh productivity supports a considerable proportion of fisheries productivity in the Hunter River (Raoult et al. in prep.). As the wetland continues to recover and is increasingly recolonised by saltmarsh plants, the benefits of repair to species across the wetland and the south arm is likely to continue to increase. As a final comment, quantitative estimates of abundance such as those presented here will ultimately be useful for determining potential benefits of habitat repair elsewhere.

Conclusions and future work

This study presents the first targeted investigation of juvenile School Prawn abundance within a coastal wetland. By applying sampling gear specifically designed for the species, which has a known efficiency for capture of prawns, we were able to derive absolute estimates of School Prawn abundance with minimal replicate error. We show that even a relatively recently restored wetland can support considerable numbers of prawns. The factors driving variation among different areas of the wetland require further investigation. While our study reflects the recruitment subsidy that might be derived from repaired wetlands, we did not account for further trophic subsidies that may benefit animals elsewhere in the system (Taylor et al. in review). This would be a good area to target for future studies, which should examine patterns in the food web across the wetland, as well as monitoring water quality at a higher level of resolution.

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CHAPTER 5 - THE POTENTIAL BENEFITS FOR SCHOOL PRAWN (*METAPENAEUS MACLEAYI*) FROM TARGETED HABITAT REPAIR IN THE LOWER CLARENCE RIVER ESTUARY

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5.1 Abstract

Development along Australia's subtropical estuaries has led to degradation of tidal wetlands and alteration of tidal flows, which has had concomitant impacts on fishery productivity and seafood production. Contemporary management seeks to lessen land use impacts on aquatic environments and restore estuarine ecosystem services, and knowledge of potential benefits of repair will inform investment and assist in galvanizing community action. School Prawn Metapenaeus macleayi, being an annual, highly fecund stock with high commercial and community value, provide a useful index of the potential benefits of seascape repair. A conventional coupled population-fishery model was used to assess the potential impact of returning connectivity to degraded areas within Lake Wooloweyah on the Clarence River estuary. Simulations showed potential benefits of restoring 27.6 ha of subtidal channels in this area could yield a recruitment subsidy which contributes up to 2,578 kg y^{-1} of additional School Prawn harvest, generating additional revenue of around AUD24,078 y⁻¹ (gross value of product) and associated economic output of AUD142,336 y^{-1} . These estimates are conservative, not accounting for the economic outcomes likely from other species directly utilising the additional habitat, or the outwelling of additional saltmarshderived productivity to support productivity of School Prawn other commercial species within other regions of the estuary. These values are considered in the context of habitat repair within the Clarence River estuary, and other estuarine systems.



Inundated saltmarsh and mangrove habitats in subtropical New South Wales. Saltmarsh habitats support the productivity of many exploited species. Photo credit: Matt Taylor

5.2 Introduction

Suitable habitat is a fundamental requirement for the early life history stages of most exploited fishes (Beck et al. 2001). Estuaries are remarkably productive systems, and many species rely on estuarine habitats to support fast growth through these life history stages where they are most vulnerable. Many Australian commercial and recreational target species retain this reliance on estuaries throughout their entire life (Elliott et al. 2007). Pollard (1984) highlighted the concept of estuary dependence and the imperative to ensure healthy estuaries and inshore waters. Many of the ecosystem services provided by estuaries are derived from subtidal, intertidal and supratidal ecosystems and related substrates, such as seagrass meadows, mangrove forests and channels, sandspits and mudflats, saltmarshes, salt flats, sedgelands and swamp forests. The key primary producers that dominate these habitats support aquatic food webs, but can also provide and shelter for fishes, and positively impact water quality in the system (Caraco et al. 2006). Consequently, estuarine habitats are key to both healthy estuary function and productive fisheries.

Over the previous two centuries, the quality of estuarine habitats has dramatically decreased (e.g. Albert 1988; Rabalais et al. 2007). Much of this is attributed to poor land use practices in catchments, especially the clearing, draining and filling of these habitats, the disruption of important processes such as tidal flow and connectivity, and the transport of excessive sediments or chemicals into the estuarine environment (Creighton 2013; Dauer et al. 2000; Harris 2001). However, the loss of aquatic vegetation that dominate these habitats exacerbates the effects of these land-based impacts (Lotze et al. 2006). The reduction in habitat function alters the conditions that fish and crustacea have evolved to rely on, reduces estuary biodiversity and most importantly estuary productivity. This ultimately limits the productivity of stocks with flow-on implications for resource sharing across conservation, recreational and commercial fishing sectors.

Productivity of penaeid prawns shows strong associations with estuarine habitats, and thereby provides useful signals for changes in estuary health and potential improvements in productivity if estuary habitats and processes are repaired. At broader-scales, fishery productivity has been clearly linked to the areal coverage of intertidal vegetation (Turner 1977). At finer scales, it is evident that a number of attributes of the habitat mosaic support these broad-scale relationships. Some examples include the location of habitats with respect to current flow (e.g. Taylor et al. 2017a), the physicochemical attributes of the aquatic environment surrounding these habitats (e.g. Taylor et al. 2017b), microhabitats created by vegetation (e.g. edge habitats, Browder et al. 1989; Rozas and Reed 1993), and the trophic food webs supported by primary producers (e.g. Melville and Connolly 2003; Melville and Connolly 2005). While these attributes are important on their own, it is likely to be the combination of these attributes in a seascape which supports the productivity of these species (Nagelkerken et al. 2015).

Efforts to repair habitats such as estuarine wetlands are steadily gathering momentum, with progress stimulated by the potential benefits that may be derived for exploited species (including penaeid prawns), among other ecosystem services (Creighton et al. 2015). This is well recognised at the fisheries management level, with an increasing number of programs aimed at repairing habitats across multiple jurisdictions, funded by a diverse range of conservation, commercial fishing and recreational fishing funding sources. In Australia, recent examples have demonstrated substantial recruitment of exploited species to rehabilitated wetland habitats (e.g. Boys and Pease 2016; Boys and Williams 2012), which have in turn been shown to link with exploited components of the stock (Taylor et al. 2017b; Taylor et al. 2016). Valuation of the potential outcomes that can be derived from habitat repair will assist with making an economic case in support of future work (Taylor 2016). Enhancement of fisheries productivity is likely to be one of the first benefits realised after repair is undertaken, especially for fecund, short-lived species like penaeid prawns. Other outcomes over the longer term will include carbon capture and storage, recreational amenity, biodiversity productivity (such as increased bird populations and species) and improvements in water quality.

This paper applies a simple framework to value the potential benefits of habitat repair for School Prawn (*Metapenaeus macleayi*) in the lower Clarence River, a large estuary in New South Wales, Australia. School Prawn are a valuable species in eastern Australia primarily harvested within the Estuary Prawn Trawl fishery, and mature barrier estuaries such as the Clarence River estuary generally support large wetland habitats that positively correlate with commercial harvest of this species (Saintilan 2004). We employed the length-based simulation model which Watson et al. (1993) applied to value commercial productivity for Australian penaeid species supported by tropical seagrass habitats, and parameterised the model using novel data on School Prawn densities and length frequencies from a rehabilitated wetland, alongside habitat mapping and fishery stock assessment data (growth and mortality) derived from the Clarence River estuary (Montgomery et al. 2010a; Montgomery et al. 2010b). The model was implemented in a stochastic framework to establish the potential economic values of yields derived from the reinstatement of connectivity with tidal creeks draining former saltmarsh habitat around the mouth of Lake Wooloweyah in the lower Clarence River estuary (described below).

5.3 Materials and Methods

Study area

The Clarence River estuary (29°43'S, 153°37'E) is the largest estuarine system in New South Wales, Australia (Figure 1), and also drains the largest coastal catchment. Historically, the Clarence floodplain had well in excess of 100 islands containing extensive wetland habitats. Development (primarily for agriculture) has seen substantial areas of floodplain and wetlands drained and isolated from the estuary. Lake Wooloweyah is an important feature of the lower estuary, and is an expansive shallow lake connected to the main estuary by four key subtidal channels, which intersperse deltaic islands formerly dominated by saltmarsh and mangrove habitats the majority of which is now reclaimed or degraded. Modification to the four main entrance channels through road construction and sedimentation has led to greatly reduced tidal fluxes within the lake (Figure 1). Palmers Channel connects with the main river ~10 km from the mouth, and the remaining three key channels are closer to the ocean (within 3 km of the mouth) with much larger tidal prisms. Further degradation of habitats has followed reduced tidal flows and heights, and has included construction of dyke's and floodgates that isolated mangroves, low-lying saltmarsh, tidal and supratidal channels and brackish to fresh back swamps from the estuary. With no tidal inundation or connectivity, all ecosystem function and productivity from these wetlands is lost to the estuary. Lake Wooloweyah represents an important nursery and trawling area for School Prawn.



Figure 1 Map of the Clarence River system, showing major features of the estuary and an inset showing the detail of the model region, the deltaic islands between the mouth of Lake Wooloweyah and the main river channel.

Habitat mapping

Historic (1942) aerial photography and current (2009) satellite imagery stored in the NSW Department of Primary Industries Aquatic Habitat Database was used to establish some of the former areal extent of tidal creeks and former saltmarsh and mangrove habitats in the model region (Figure 2). Historic imagery was digitised and orthorectified, and habitat polygons for macrophytes and subtidal creeks manually constructed in ArcGIS as previously described (see Williams and Thiebaud 2007). The difference between the areal coverage of subtidal creeks draining former saltmarsh habitats within the model region at the two time points was calculated to determine the gain in subtidal habitats that could be utilised by School Prawn following repair by reinstating connectivity with the estuary.

Determination of recruitment subsidy from habitat repair

The model of Watson et al. (1993) used the standing stock of prawns in the habitat of interest (seagrass in Cairns Harbour) to estimate potential value of that habitat derived through fisheries harvest following recruitment to the fishery. In our model, we used a similar concept but since standing stock cannot be measured prior to repair (as there is no recruitment), we calculated an expected annual recruitment subsidy (N_{sub} , y^{-1}) to the exploited stock derived from areal estimates of habitat extent and expected School Prawn densities in this habitat.

To estimate expected School Prawn densities, School Prawn were measured in the Hexham wetland system in the Hunter River, New South Wales by Hart et al. (in review). The Hexham wetland system is a recently (2013) restored saltmarsh system in an estuary of identical geomorphological classification to the Clarence River, that experiences a similar salinity regime and is a similar distance to sea as the model region. Given the summer growth rates of School Prawn and the estimation of modal groups in Montgomery et al. (2010b) for both the Hunter and Clarence Rivers, we assumed a progression of two cohorts would move through the repaired habitat in a particular year. This was likely a conservative estimate, but ensures that the estimates of value derived from our simulations remain conservative. To establish the potential impact of habitat repair, the annual recruitment subsidy was calculated using $N_{sub} = A \cdot D \cdot N$, where A (ha) is the area of new habitat available for colonisation by School Prawn, D (# ha⁻¹) is the expected density of School Prawn in the repaired habitat, and N (y⁻¹) is the number of cohorts moving through the new habitat in a year. Two scenarios were modelled, one reflecting average recruitment (D_{avg}) and one reflecting good recruitment (D_{good}). Average recruitment reflected the average abundance detected across all sites sampled in the wetland system, whereas good recruitment reflected the average of the highest abundance site in the system (as estimated from field data as described above).

Simulation model

The approach of Watson et al. (1993) was reproduced in R v. 3.3.2 (R Development Core Team 2016) within a stochastic model framework reflecting a Monte-Carlo Analysis of Uncertainty (MCAoU). This allowed us to assess the value of potential harvest derived from School Prawn using subtidal channels within the repaired saltmarsh system in the model region and incorporate uncertainty in model parameters. This approach employed traditional relationships used to model population dynamics in exploited species, and draws on some established approaches for evaluating the economic impact of fisheries harvest in the literature. All relationships used a weekly time-step, and model parameters were provided as distributions where possible (see Table 2; note that a and b were provided as a bivariate normal distribution fitted from empirical data for the estuary), and these distributions were randomly and independently sampled in each of 10,000 simulations. All economic values were expressed in Australian dollars (AUD).



Figure 2 Detail map showing the deltaic islands between the mouth of Lake Wooloweyah and the main river channel, and the extent of mangrove and saltmarsh habitat in 1942 (saltmarsh shaded brown, and mangrove shaded green) and 2009 (remaining habitat from 1942 is overlaid with grey hatching). The subtidal creek areas that would be available for colonisation following repair are indicated as black polygons.

Growth of School Prawn (L_t) across the period t_0 to t_{max} , was modelled using case 5 of the Schnute (1981) model as suggested by Montgomery et al. (2010a) for the Clarence River system (equivalent to the von Bertalanffy growth function), and the parameters described for this system in Montgomery et al. (2010a, Table 2). Consequently, $L_t = L_{\infty}(1 - e^{-k(t-t_0)})$, where L_{∞} was the asymptotic length (hereafter all references to length refer to carapace length; CL, mm), k was the rate of approach to L_{∞} , and t_0 was the theoretical age at which L=0 (which was set to zero given that early life stages were dealt with in the model). Weight was estimated from length using $W_t = aL_t^{-b}$ where W_t reflected weight at time t, and a and b were the weight coefficient and exponent respectively. These relationships produced three master vectors of time (age), length, and weight, from the period since hatching to the conclusion of the model period (t_{max}). Sub-vectors were then extracted for the period between the age of population measurement (t_m) and the age at the conclusion of the model period (t_{max}).

We used a length-based relationship which scales natural mortality with prawn size and has previously been applied to small prawns (Loneragan et al. 2003), whereby $M_t = \alpha e^{-\beta L_t}$ and L_t was the carapace length at time t, and α and β were constants describing how natural mortality scales with animal size. Fishing mortality (F_t) was expressed as the product of the fishing mortality (F) estimated for the Clarence River by Montgomery et al. (2010a) and a length-based logistic selectivity curve $F_t = F \cdot \frac{e^{(n+p \cdot L_t)}}{1+e^{(n+p \cdot L_t)}}$ where *n* and *p* are calculated based on length at 25%, 50% and 75% retention (L_{25} , L_{50} and L_{75} respectively) using $n = -L_{50} \cdot p$ and $p = \frac{2.197}{L_{75} - L_{25}}$. The total weekly mortality rate (Z_t , wk⁻¹) was expressed as the sum of M_t and Ft: The abundance of the enhanced population was estimated over the period from the mean age for N_{sub} (t_m) until the end of the model period (t_{max}), where $t_m = -\frac{1}{k} \cdot \log(1 - \frac{l_{N_{sub}}}{L_{\infty}})$ and $l_{N_{sub}}$ (mm) is the mean carapace length of prawns used to determine N_{sub}. A standard negative exponential model was used to calculate abundance as a function of time and mortality ($N_{t+1} = N_t e^{-Z_t}$) where N_t is the abundance of the enhanced stock at time t, and N_t at time $t = t_m$ is the recruitment subsidy derived from habitat repair (N_{sub}) . Instantaneous biomass harvested (H_t) was calculated as a function of time for the period $t_m \le t \le t_{max}$ using $H_t = \frac{F_t}{Z_t} \cdot N_t (1 - e^{-Z_t}) \cdot W_t$ which was summed across the model period to estimate the total biomass harvested (H). Harvest was converted to economic value by multiplying by the associated market price (P_{SFM} , AUD) for the species, which was estimated from the CPI-corrected (consumer price index) average Sydney Fish Market values across the period 2005/06 – 2014/15. Two scenarios were considered. The first was a measure of Gross Value of Product (GVP), which is reflected by the simple multiplication mentioned above ($GVP = H \cdot P_{SFM}$). The second represented an extrapolation of GVP to account for expected flow-on economic values from product harvested on the broader economy using a multiplier (m, derived from the data reported in Voyer et al. 2016, Table 2), thus reflecting a Total Economic Output from the habitat repair ($TO = H \cdot Mp \cdot m$).

5.4 Results

Over the period 1942 – 2009 up to 66% of saltmarsh habitat has been lost from the model region (Table 2, and Figure 2), however there has been a modest gain in areal coverage of mangrove. This has included the loss of 27.6 ha of shallow subtidal creeks which drain the saltmarsh and mangrove habitats described above, and which would be available for colonisation following repair of the saltmarsh habitat.

 Table 2 Change in saltmarsh and mangrove habitat within the model region between the period

	1942	2009	Percent change	
Mangrove	405 ha	415 ha	2%	
Saltmarsh	666 ha	223 ha	-66%	

Distributions of key management indicators for the modelling are provided in Figure 3 and Figure 4. These distributions give some indication of the variability in their prediction, given the level of uncertainty in some model parameters. While the tails of these distributions represent relatively unlikely outcomes, the mean values (as indicated in the figures) represent the most likely outcomes under average conditions. Under a scenario of good recruitment, repair of habitat and subsequent colonisation of subtidal creek habitat by School Prawn is most likely to yield around 2,578 kg of product, which equates to a gross value of around AUD24,078 and total output of around AUD142,336 per year (Figure 3). If we consider the average recruitment scenario, however, repair of habitat and subsequent colonisation of subtidal creek habitat by School Prawn is most likely to yield around 623 kg of product per year, which equates to a gross value of around AUD5,800 and total output of around AUD34,258 (Figure 4).



Saltmarshes and estuaries provided critical habitat for School Prawns. Photo Credit Matt Taylor



Figure 3 Potential outcomes from re-instatement of connectivity with marsh habitats in the model region given good recruitment, showing potential harvest impacts (top panel), and associated annual Gross Value of Product (GVP, middle panel) and Total Economic Output (TO, lower panel), given the assumption and parameters described in the text. Vertical lines indicate the average of estimated impacts.



Figure 4 Potential outcomes from re-instatement of connectivity with marsh habitats in the model region given average recruitment, showing potential harvest impacts (top panel), and associated annual Gross Value of Product (GVP, middle panel) and Total Economic Output (TO, lower panel), given the assumption and parameters described in the text. Vertical lines indicate the average of estimated impacts.

5.4 Discussion

Our estimates add to the growing body of literature reporting the potential fisheries harvest benefits that can be derived from habitat repair. Population models such as this provide an oversimplified view of many complex and inter-linked ecological benefits, and by just focusing on a single species also markedly underestimate potential economic benefits. Nevertheless such models provide a useful approach for the broader community to understand the importance of estuarine habitat, and the opportunities that habitat repair represent. Models in which habitat is linked to fisheries in a quantitative fashion have the advantage of relying on a small number of variables and a small suite of conservatively realistic assumptions.

There are several examples that apply such an approach in the literature. For example, Blandon and zu Ermgassen (2014) used a similar population and fisheries model to evaluate fisheries enhancement arising from repair of seagrass habitats, and derived a cumulative value (across 12 species) of ~AUD230,000 ha⁻¹ y⁻¹, although most species ranged from AUD2 – AUD6,500 ha⁻¹ y⁻¹. Watson et al. (1993), using the same model employed in the current study, estimated values of AUD72 – AUD11,084 ha⁻¹ y⁻¹ (converted to 2015 dollars) from prawn harvest derived from the standing stock of juveniles in seagrass in northern Australia.

For the 27.6 ha restoration scenario considered here, our estimates broadly align with these previous studies, and provide an indication of the potential value that can be obtained from habitat repair. Simulation studies such as these contribute to the consideration of benefits alongside the costs of habitat repair.

Broader benefits derived from repair

As mentioned above, the benefits of habitat repair are by no means limited to the values presented here. Saltmarsh habitats can make substantial contributions to the exploited biomass harvested from estuarine systems (Taylor et al. in review). Firstly, the re-connected subtidal channels arising from the repair scenario addressed here will provide habitat to directly support species other than School Prawn. For example, subtidal channels draining saltmarsh habitats have been shown to contain exploited species such Mud Crab (*Scylla serrata*), Dusky Flathead (*Platycephalus fuscus*), Yellowfin Bream (*Acanthpagrus australis*), Luderick (*Girella tricuspidata*) and Sea Mullet (*Mugil cephalus*) (Mazumder 2009; Morton et al. 1987; Webley et al. 2009). Direct support of adults and/or juveniles of these exploited species will produce fishery benefits that contribute additional value from habitat repair.

Secondly, and probably most importantly, are the potential gains in primary productivity to the estuary that arise from re-instatement of connectivity with these habitats. Saltmarsh is one of the most productive habitats in Australian estuarine systems, and is the major contributor to the biomass of a cross-section of exploited species in seagrass limited systems (Raoult et al. in prep.). When connected to the broader estuary, primary production in saltmarsh habitats is outwelled to other areas across the estuarine system. This can occur through a number of mechanisms including the transport of particulate organic carbon (POC), transport of dissolved organic carbon (DOC), or consumption of marsh plants by small nekton on the marsh surface (when inundated), and subsequent movement throughout the estuary. These additional benefits for School Prawn (or other species) are not captured in the analysis presented here, but the fishery value of saltmarsh productivity through the trophic subsidy described above has been shown to be as high as AUD18,352 ha⁻¹ y⁻¹ in the Clarence River system (Taylor et al. in review).

Implications for repair

Our simulations reflected the total gains from across the model region only, but these average values can be converted back to per-hectare estimates. For the average recruitment scenario, estimated harvest would be approximately 23 kg ha⁻¹ y⁻¹, and the associated GVP and TO would be AUD210 ha⁻¹ y⁻¹ and AUD1,241 ha⁻¹ y⁻¹ respectively. Most of the former saltmarsh habitat within the model area is not used productively for any other economic output, as they are gleyed clay and poorly drained soils that might occasionally be used for rough cattle grazing. Essentially these seascapes have been lost to the estuary simply as a result of the design of the flood/tidal levees constructed in the 1960's and 1970's. In developing the model and its application it has been assumed that tidal prisms remain constant; this is important for supply of recruits (Taylor et al. 2017b) and outwelling of saltmarsh-derived nutrition. Yet for Lake Wooloweyah (as indeed for most Australian subtropical estuaries), tidal flow velocity, fluxes and total tidal prism have been constrained by training walls, bridge approaches, causeways and barrages. Lake Wooloweyah is connected to the main estuary by four tidal channels (Figure 1). Palmers Channel is some 10 km by river upstream from the mouth, with a subtidal reverse delta into Lake Wooloweyah that has reduced tidal fluxes throughout the system. The remaining three key channels are within three kilometres of the mouth, and hence experience larger tidal prisms, but road construction across Romiaka, Oyster and Shallow Channel (Figure 1) and subsequent sedimentation has likely contributed to reduced tidal fluxes. No estimate of changed tidal fluxes, the reduced height of the tidal prism within Lake Wooloweyah or the subsequent reduction in the area of wetland inundated during high tides is available, but these will all ultimately affect the realised benefit from repair. Any strategic assessment of opportunities for enhancing Lake Wooloweyah and overall Clarence River fishery should include an analysis of opportunities to enhance tidal flows.

Outside of the model area, similar leveed sites suitable for repair exist for Lake Wooloweyah foreshores including one contiguous prior wetland in the southwest sector of the lake. This complex of subtidal channels, mangroves and saltmarsh is well in excess of 500 hectares, which could contribute considerable value to the fishery if connectivity with the estuary is reinstated (certainly well in excess of its value for alternate land uses). In addition, there are many other sites suitable for repair in the lower Clarence River estuary, and our estimates of School Prawn productivity from the model area contribute to a strategic assessment of opportunities for enhancing fishery productivity. Such an analysis would need to be multiobjective, simultaneously accounting for enhancement of fishery productivity alongside benefits through improved flood management, enhanced recreational access and good quality agricultural land and water management. Ultimately this would provide a useful guide for repair across other large estuaries on the Australian coast.



A cow feeds on saltmarsh grass in the Wooloweyah wetland. Photo credit: Matt Taylor

Parameter	Estimate	Units	Source
k	N(0.035,0.00357)	wk ⁻¹	Montgomery et al. (2010a)
L∞	N(36.6, 2.3737)	mm CL	Montgomery et al. (2010a)
а	fitted	-	Length-weight series
b	fitted	-	Length-weight series
t_0	0	wks	
α	N(0.5,0.005)	-	Loneragan et al. (2003)
в	N(0.1,0.001)	-	Loneragan et al. (2003)
A	27.6	ha	Mapping data
D_{good}	100,881	# ha ⁻¹	Hart et al. (in review)
D_{avg}	24,366	# ha ⁻¹	Hart et al. (in review)
Ν	2	-	Estimated
$l_{N_{sub}}$	7.697	mm CL	Hart et al. (in review)
F	U(0.0014, 0.0358)	wk⁻¹	Montgomery et al. (2010b)
L ₂₅	10.5	mm	Commercial fishery data
L ₅₀	12.0	mm	Commercial fishery data
L ₇₅	13.5	mm	Commercial fishery data
t _{max}	75	wks	
P _{SFM}	N(9.27,0.89)	AUD	Sydney Fish Market
т	N(5.89,0.14)	-	Voyer et al. (2016)

Table 1 Parameter values, distributions, and associated sources for model simulations.

Comparison of scenarios

The simulations presented here reflect a network of simple relationships between carrying capacity, growth, mortality, harvest and market value. While most parameters used in the model were actually derived from studies in the Clarence River (Table 1), the density estimates employed were based on prior work conducted in a repaired wetland in the Hunter River. Consequently, we modelled two scenarios based on this data, which included an average recruitment and a good recruitment scenario. The Clarence River represents the largest School Prawn fishery in New South Wales, and this is due to a combination of factors that enhance recruitment and productivity. While the average recruitment scenario probably represents a conservative estimate of benefits from repair in the Clarence River, the good recruitment

scenario may be more relevant to this estuary for a number of reasons. Spawning and recruitment of School Prawn is dependent on freshwater inflow to the estuary (Glaister 1978a; Glaister 1978b; Ruello 1973), and the Clarence River experiences the largest flows of any estuary that supports the species and likely leads to enhanced reproduction and recruitment. Secondly, Clarence River is over 3 degrees of latitude to the north of the Hunter River, and warmer water temperatures in this area likely have a positive impact on the fecundity of adults (for example, see Penn 1980), as well as the growth and survival of larvae (Preston 1985) and juveniles (Montgomery et al. 2010b). Considering these factors, it is likely that recruitment is greater in the Clarence River estuary, and this is evidenced in the magnitude of catches obtained from this estuary (Ives et al. 2009). Thus, the good recruitment scenario may present a more realistic indication of the benefits of repair.

Conclusions

These benefits of the restoration scenario outlined in this study will likely be relevant to both commercial and recreational fishers alike. The restoration of connectivity to these areas will open up the habitat for direct usage by a broad cross-section of other recreationally targeted species, and these species will similarly benefit from any associated trophic subsidy. Such indicative benefits would be markedly increased should a major repair initiative for the Clarence River estuary be implemented, as there are many similar tidally isolated areas of wetland across the system that could be repaired relatively easily. More importantly, benefits of restoration and a healthier estuary in general would be shared across the broader community through increased food production, recreational fishing opportunities, tourism, and community lifestyles. Any multi-objective repair initiative would also deliver improvements to flood management, road infrastructure and agricultural land management.

Many science challenges remain as estuary repair strategies are developed. Key knowledge gaps worthy of further investigation include further understanding the ecological productivity implications of restoring tidal flows, modelling the multi-species productivity benefits of repair including finfish, other crustaceans and avifauna, and determining the likely interactions and ecological responses of outwelling of additional saltmarsh-derived productivity as a contribution to enhanced estuary health.

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CHAPTER 6 - FISH USE OF TASMANIAN SALTMARSH WETLANDS: BENEFITS FOR FISH FROM TIDAL RESTORATION AND COASTAL REHABILITATION

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6.1 Abstract

Fish use of coastal saltmarsh wetlands have been documented for many parts of Australia with a notable exception of Tasmania. Our study primarily aimed to document the diversity, density and patterns of fish use in the Circular Head coast saltmarshes, north west Tasmania. We also explored any effect of saltmarsh habitat fragmentation on fish species diversity and density by sampling concurrently at nearby paired sites of predominantly unaltered and altered saltmarshes. Three site pairs were selected 2.5-10 km apart for each other, on the basis of the extent of habitat fragmentation caused by earthworks that was representative of the study localities. Fish were caught using buoyant floorless pop nets at the six paired locations over successive neap and spring tide cycles. A total of 851 fish from 11 species were caught in 37 of the 48 net releases at a mean density of > 72 fish per 100 m⁻². Three of these species, *Aldrichetta forsteri, Arripis truttaceous* and *Rhombosolea tapirina*, are targeted by recreational and commercial fishers and contributed close to 20 % of the total catch numbers. Although there were minor differences in fish assemblages between locations, there was no significant difference between the unaltered and altered marshes, including those areas behind naturally breached levees. This provides an indication that any rehabilitation of saltmarshes by restoring tidal flows will deliver benefits for fish productivity through expanded habitat.



Buoyant pop nets (5 m x 5 m) in action on Tasmanian saltmarsh. Photo credit: Vishnu Prahalad

6.2 Introduction

Coastal saltmarsh wetlands are increasingly recognised as fish nurseries with a growing literature documenting the importance of these habitats for itinerant fish use (e.g. Connolly 2009; Raposa and Talley 2012). The general expectation is that saltmarshes and their associated tidal creeks provide secure and productive habitat for fish at varying spatial and temporal scales as part of the seascape (e.g. Kneib 1997; Deegan et al. 2000; Valiela et al. 2000). In Australia, there is increasing evidence of fish utilisation of food resources found in saltmarshes (Crinall and Hindell 2004; Hollingsworth and Connolly 2006; Mazumder et al. 2006a; Mazumder et al. 2011; McPhee et al. 2015; Platell and Freewater 2009). As elsewhere, Australian saltmarshes have been documented to produce organic materials (plant and animal matter) that are exported to coastal waters through tides, thus improving seascape fisheries productivity (Melville and Connolly 2003; Svensson et al. 2007).

While more research is being undertaken in Australia, the majority of research on saltmarsh fish has been focused elsewhere in the world, particularly in North America. A review conducted by Connolly (1999) indicates that, of literature published before 2000, 90% of studies were from North America, 7% from Europe and 3% from the southern hemisphere including Australia (although further work has since been published). Differences exist in habitat type between Australian and North American saltmarshes and mangroves, including differences in typical elevation, water depth and plant assemblages (Connolly 2009), making comparisons between international studies problematic. Within the Australian literature, studies have primarily been reported from temperate, subtropical and tropical waters in South Australia, Victoria, New South Wales and Queensland (Connolly et al. 1997; Crinall and Hindell 2004; Davis 1988; Mazumder et al. 2006b; Thomas and Connolly 2001).

Australian literature reporting on the use of temperate saltmarshes by Australian fish species record up to 35 species with densities of up to 56 fish per 100m⁻² (Connolly 2009; Wegscheidl et al. 2017). In terms of patterns of fish use of saltmarshes, Australian literature describe spatial and temporal differences between regions, including varying effects of seasonality, tide regime, water depth, diel time, temperature and salinity on fish assemblages (Connolly et al. 1997; Crinall and Hindell 2004; Davis 1988; Mazumder et al. 2005a; Morton et al. 1987; Thomas and Connolly 2001). Although a major focus of research reported from North America has been on differences in fish use between varying saltmarsh condition (e.g. Raposa and Talley 2012), few published studies have dealt with this partially in Australia (Connolly 2005; Mazumder et al. 2006b), and none directly (Connolly 1999).

A conspicuous omission from the Australian literature has been of fish use of Tasmanian saltmarshes, with no previous record of fish species diversity, density, patterns of use and preference between varying habitat conditions. As both saltmarshes and mangroves have been found to host many fish species (Mazumder et al. 2005a; Saintilan et al. 2007), and given the lack of mangroves in Tasmania, measuring the diversity, density and patterns of fish use of saltmarshes (where no adjoining mangrove habitat is present) is important. As well as lacking mangroves, Tasmania's saltmarshes differ in context to those found in mainland Australia. In comparison, they are situated slightly lower on the tidal frame (thus being subject to different flooding regimes) and contain different saltmarsh plant assemblages compared to many of their mainland counterparts (Mount et al. 2010).

Saltmarshes of temperate and subtropical Australia are listed as an endangered ecological community under the Australian Federal *Environment Protection and Biodiversity Conservation Act 1999* to highlight their historic and ongoing loss and degradation (Threatened Species Scientific Committee 2013). In the context of Tasmania, loss and degradation of saltmarshes has occurred most extensively in the north-west part of the State, in the Circular Head region (Prahalad 2014). Previous management interventions focussed heavily on conserving shorebirds and had struggled to raise the profile of saltmarshes has not been a major focus of efforts to conserve and repair saltmarshes, due in part to the lack of documented studies. Our aim is to document fish use of the saltmarshes in the Circular Head area by addressing the following

questions: (1) what is the diversity and density of fish in the saltmarshes of our study area during our sampling season? (2) are there any observable patterns of fish use relative to sampling location, tide regime, water depth, diel time, temperature and salinity? (3) is there difference in fish use between saltmarshes of varying condition? and, (4) what are implications for management and further research?

6.3 Materials and methods

Study area

The Circular Head coastal area is located in the far north-west of Tasmania, between the small town of Stanley and Woolnorth Point (Figure 1). The area is almost entirely sheltered from the high-energy wave climate of Bass Strait and forms an expansive seascape matrix of tidal flats, seagrass beds, saltmarshes and *Melaleuca ericifolia* swamp forests on the landward margin (Figure 2, Mount et al. 2010). The Circular Head area is home to almost a quarter of all saltmarsh mapped across Tasmania, occupying 1326 ha in 23 distinct clusters, each associated with a river/creek mouth, embayment, sheltered passage or tidal island (Prahalad 2016). The study area has a mesotidal range of up to 3.1 m, the largest on the Tasmanian coast (Donaldson et al. 2012), with a semi-diurnal tidal cycle. Within the tidal frame, saltmarshes occupy a narrow niche of about 0.5 m elevation range (Mount et al. 2010). The low marsh is characterised by a succulent mat of *Sarcocornia quinqueflora* often co-occurring with *Samolus repens*, ranging in elevation between 10-20 cm (when flooded). The high marsh and back marsh areas are dominated by the succulent shrub *Tecticornia arbuscula* often mixed with grasses and sedges. Saltmarshes are flooded partially during neap tides and almost fully during spring tides.



Figure 1 Study area and three saltmarsh locations used in the Circular Head coastal area of north-west Tasmania. Base imagery from SPOT5 satellite, dated 2009.



Figure 2 Circular Head coastal seascape matrix of tidal flats, seagrass beds, saltmarshes and *Melaleuca* ericifolia swamp forests. Illustration used with permission from Mount et al. (2010).

The landward boundary of the saltmarsh coincides roughly with the storm tide extent (Mount et al. 2010). *M. ericifolia* dominates the landward margins of the tidal frame competing with saltmarsh and extending onto nearby low lying coastal floodplain areas. A large part of *M. ericifolia* and the adjoining saltmarsh has, however, been cleared for agricultural use, with over 25km's of levees built along the shoreline to restrict tidal flooding (Prahalad 2014). Earliest evidence of levee building in saltmarsh was observed from old aerial imagery from the late 1960s, while the most extensive period of clearing and draining was during the 1980s. The estimated absolute loss of saltmarsh between 1952 and 2006 is 219 ha (16%), with 752 ha (65%) of the remaining saltmarshes subject to impacts including clearing, ditching, grazing and buffer zone removal (Prahalad 2014). Levee building continues, with a 2 ha area of saltmarsh lost between 2013 and 2016 (unpublished data). The Circular Head coastal area has been selected for this study for having the greatest potential for saltmarsh rehabilitation in the state. The area is also of significant importance to recreational and commercial fisheries, with the local saying that: 'if you are not catching a flathead in Smithton, you are not trying'. There are also active oyster farms in the area which depend on good water quality.

Sampling design

Methods used to sample fish in saltmarshes include block nets, flume nets, flume weir, fyke nets, lift nets, pop nets, drop samplers, traps, dip nets and hand trawls, and also poisoning (Connolly 1999, 2009). Of these, pop nets are used in Australia now more than other techniques (Connolly 2009), due to their easy portability allowing for sampling replications and their ability to provide a density measure (fish per m⁻²) that is comparable to other studies (Wegscheidl et al. 2017). Various authors have used different pop net types and sampling regimes, with the general tendency to use a larger sample area (~25 m⁻²) to avoid small scale patchiness, a fine mesh size (~2 mm) to catch juvenile fish and a remotely controlled release. In this study, we employed four custom made buoyant floorless pop nets, each covering an area of 25 m^{-2} (with 5 m long x 1 m high walls) and with a fine mesh size of 2 mm. The bottom of the net walls had a lead-core rope that was tucked under the saltmarsh substrate forming a shallow depression and pegged down by 10-12 weed mat pins on each side. This helped avoid trenching and excessive soil disturbance (cf. Connolly 2005). The top of the net walls had a sleeve suitable for a 20 mm PVC pipe that was inserted in-situ and sealed for floatation. The net was folded under the top sleeve containing the PVC pipe so that the net sat flat on the marsh surface as much as possible. Weights were placed on the PVC pipe to keep it depressed with the incoming tide until the nets were ready to be popped. The installation was done during low tide and took 60 mins per net with two people working in tandem.

The four nets were used concurrently at nearby paired sites of unaltered and altered saltmarshes, located in Robbins Passage, Big Bay and Perkins Passage (see Figure 1). The three locations was 2.5-10 km apart for each other and selected on the basis of being representative of the saltmarshes of the Circular Head region. Unaltered saltmarshes had no hydrological alterations due to nearby levees or other notable human impacts (such as ditches, clearing, grazing), were surrounded by a contiguous buffer zone of native vegetation, and were relatively unfragmented being part of a larger cluster or matrix of saltmarsh. Altered

marshes had significant hydrological alterations due to nearby levees and other human impacts (such as ditches, clearing, grazing), had a little to no native buffer vegetation being juxtaposed to agricultural land used mainly for cattle grazing, and belonged to highly fragmented clusters of variable size (Table 1.).

Site	Condition o	lass and variable	5		
			2	Saltmarsh	Saltmarsh
Location	Class	Levees ¹	Buffer zone ²	fragmentation ³	area ⁴
Robbins Passage	Unaltered	Absent	Present	Absent	12.1 ha
	Altered	Broken levees	Present but limited	Medium	35.5 ha
Perkins Passage	Unaltered	Absent	Present	Medium	13.5 ha
	Altered	Broken levees	Absent	High	18.9 ha
Big Bay	Unaltered	Absent	Present but limited	Medium	15 ha
	Altered	Intact levees	Absent	High	1.7 ha

Table 1. Condition of saltmarshes used in the study.

¹Broken levees are regularly breached by incoming tide.

²Buffer zone, e.g. *Melaleuca ericifolia* swamp forest.

³Degree of fragmentation of marsh and associated tidal creeks by levees since 1960's.

⁴Area of saltmarsh, contiguous but spread along the coast with a high marsh area to edge ratio.

Sampling procedure

At slack high tide, the fully installed nets were released remotely (10-15 m) by two field personnel pulling the strings connected to the weights at the same time. The nets popped instantaneously (~1 second) and were then surveyed for entrapped fish, mostly at the downstream side(s) into which they were channelled as the tide receded. Fish were collected at regular intervals using hand-held dip nets to mitigate loss due to predation by birds and crabs inside the net. Depending on the tide height, it took between 1-2 hrs for the flood tide to recede fully from the marsh surface. On a couple of occasions during spring tide, the crab holes in the marsh were still holding water well after the marsh surface had drained and hiding fish of the family Gobiidae. A thorough final inspection was made before concluding each sampling effort by checking all four walls of the net and tiny depressions for camouflaged species. Collected fish were identified in the field, recorded and released. Representative samples of each species were taken to confirm field identification by fish experts (following Gomon et al. 2008). Fish were terminally anesthetised in the field using a lethal dose of AQUI-S^{*}, a commercially available derivative of clove oil. Specimens were preserved immediately into a solution of 95% ethanol.

Fish were sampled concurrently in both unaltered and altered sites during successive neap and spring tide cycles in the months of April and May 2017. Sampling was done during both high tides (night and day) of semi-diurnal tidal cycle. Each sampling effort involved two replicates placed in the saltmarsh at the paired locations, located randomly on the marsh flats and spaced no further than 25 m apart (*cf.* Thomas and Connolly 2001). The neap tidal cycle samples were located on the seaward edge of the marsh expecting lower water levels and the spring tidal cycle samples were located slightly higher on the marsh platform expecting higher water levels (with distance to seaward edge proportional to the paired unaltered and altered marshes). Water temperature, salinity and time of net release (diel time) were recorded at each sampling location. Water depth was recorded at each net as the mean of maximum and minimum depth, as the marsh surface was often sloped.

Data analysis

Summary statistics were used to gain an overall impression of the fish community. To gauge the completeness of the sampling, a species accumulation curve (collector's curve) was produced using *specaccum* in the *vegan* library (Oksanen et al. 2011). Samples taken when the maximum water depth was

less than 5 cm (mean water depth <3 cm) yielded no fish and were excluded from further analysis. To explore the relationship between the environmental variables and fish species abundance within the overall assemblage, we used a permutational MANOVA (the *adonis* function from the *vegan* library) which fits linear models to distance matrices and uses a permutation test (n = 999) with pseudo F-ratios. We related four variables - fish species richness per sample, fish catch per sample and the abundance of the more two most common species - to a suite of predictor environmental variables - location, condition status, tide cycle, diel phase (night vs. day), water salinity and mean water depth. Since the response variables were based on count data, Poisson or quasi-Poisson models with a log link function were applied as appropriate.

The multiple response permutation procedure (MRPP) in *vegan* was used to test for any significant difference between the unaltered and altered sites based upon their fish assemblages. The Bray–Curtis dissimilarity measure and 999 permutations were employed. The MRPP statistic delta is the overall weighted mean of within-group means of the pairwise dissimilarities among the sampling units. A is a chance-corrected estimate of the proportion of the distances explained by group identity, a value analogous to a coefficient of determination in a linear model (Oksanen et al. 2011). The degree to which the fish assemblages varied between unaltered and altered sites was assessed using nMDS ordination based on the Bray Curtis dissimilarity measure (Clarke and Warwick 2001). Fish counts were not transformed since the range of values was not extreme. The stress level of 0.1909 in 2 dimensions was acceptable (Quinn and Keogh 2002). Analyses were carried out in the R statistical environment (R Foundation for Statistical Computing, Vienna, Austria).

6.4 Results

A total of 851 fish of 11 species from 8 families were caught (Table 2.). The species accumulation curve (Figure 3) suggests that the total number of species present at the sampling sites is about 12 and that the number of samples was satisfactory to reveal most of the fish taxa present at the sites. The family Atherinidae contributed 3 species and 74% of the total catch numbers, of which *Atherinosoma microstoma* and *Leptatherina presbyteroides* were most abundant (57% and 16% respectively, Figure 4). Two members of the family Gobiidae, *Pseudogobius* sp. and *Nesogobius maccullochi*, contributed 3% and 2% to numbers respectively. Three species, *Aldrichetta forsteri* (Mugilidae), *Arripis truttaceous* (Arripidae) and *Rhombosolea tapirina* (Pleuronectidae) are targeted by recreational and commercial fishers (Lyle et al. 2014), and these taxa contributed almost 20% of the total catch numbers. Of these, *A. forsteri* was both common and numerically dominant, present in 24 (65%) of the 37 nets that caught fish and made up 19% of the total catch. All the specimens caught were identified as either juveniles or sub-adults. *Palaemon* sp. (palaemonid shrimp) was observed in most of the nets, sometimes in large numbers (~200) but not censused as the study was restricted to finfish. Crabs were also observed in all of the nets and have been previously documented for this area by Richardson et al. (1997). Some of the larger and more active individuals were evicted from the nets to avoid predation of fish when the water levels were low.

Table 2. Fish caught using buoyant floorless pop nets on saltmarsh flats on the Circular Head coast, north-west Tasmania, during April-May 2017.

Family	Genus/species	Common name	Contri	bution t	to catch	ı										
			Robbiı	ns Passa	nge		Perkin	s Passa	ge		Big Ba	y			Total	
			Unalte	ered	Alter	ed	Unalte	red	Altere	d	Unalte	red	Altere	d		
			Total	%	Total	%	Total	%	Total	%	Total	%	Total	%	Total	%
Atherinidae	Atherinosoma microstoma (Günther, 1861)	Smallmouth Hardyhead	37	55.2	129	64.2	21	34.4	47	34.8	146	63.5	102	65.0	482	56.6
	Kestratherina esox (Klunzinger, 1872)	Pikehead Hardyhead	0	0	3	1.5	3	4.9	0	0	0	0	6	3.8	12	1.4
	Leptatherina presb yteroides (Richardson, 1843)	Silver Fish	7	10.4	50	24.9	6	9.8	15	11.1	39	17.0	18	11.5	135	15.9
Gobiidae	<i>Nesogobius maccullochi</i> (Hoese and Larson, 2006)	Girdled Goby	2	3.0	2	1.0	2	3.3	7	5.2	5	2.2	0	0	18	2.1
	Pseudogobius sp.	Eastern Bluespot Goby	10	14.9	7	3.5	1	1.6	4	3.0	0	0	6	3.8	28	3.3
Mugilidae	Aldrichetta forsteri (Valenciennes, 1836)	Yellow-eye Mullet*	10	14.9	10	5.0	27	44.3	50	37.0	40	17.4	23	14.6	160	18.8
Pleuronecti dae	<i>Rhombosolea tapirina</i> (Günther, 1862)	Greenback Flounder*	0	0	0	0	0	0	1	0.7	0	0	0	0	1	0.1
Pseudaphri tidae	Pseudaphritis urvillii	Congolli	0	0	0	0	0	0	5	3.7	0	0	1	0.6	6	0.7

	(Valenciennes, 1832)															
Tetrarogida e	Gymnapistes marmoratus (Cuvier, 1829)	Soldier	0	0	0	0	0	0	0	0	0	0	1	0.6	1	0.1
Arripidae	<i>Arripis truttaceus</i> (Cuvier, 1829)	Australian Salmon*	1	1.5	0	0	0	0	0	0	0	0	0	0	1	0.1
Galaxiidae	Galaxias maculatus (Jenyns, 1842)	Common Galaxias	0	0	0	0	1	1.6	6	4.4	0	0	0	0	7	0.8
Total cate	ch per sample type		67		201		61		135		230		157		851	
Fish de	nsity per 100m ⁻²		38.3		100 .5		30.5		67.5		115		78.5		72.4	
Fish density	per 100m ⁻² (excluding than 5 cm water depth	nets with less ı)	44.7		100 .5		40.7		67.5		115		104. 7		83.0	
The asterisk and com	 (*) indicates species or mercial interest (Lyle et 	f recreational t al. 2014).														



Figure 3. Species accumulation curve (with SD) for fish species sampled at three saltmarshes on the Circular Head coast, north-west Tasmania, during April-May 2017.



Figure 4. Boxplot of the fish taxa at three saltmarshes on the Circular Head coast, north-west Tasmania. Common name codes used are YEM: Yellow-eye Mullet, SMH: Smallmouth Hardydhead, SLF: Silver Fish, SF: Soldierfish, PHH: Pikehead Hardyhead, GG: Girdled Goby, GBF: Greenback Flounder, EBG: Eastern Bluespot Goby, CON: Congolli, CGA: Common Galaxias, AS: Australian Salmon.

The pop nets were very effective at catching fish with 37 of the 48 net releases returning between 3-69 fish per net. One of the nets failed in the Robbins Passage unaltered saltmarsh during the neap tide night-time sample. The mean density of fish caught in the remaining 47 nets was 72.4 fish per 100 m⁻² (Table 2.). This figure is a lower end estimate given the maximum water depth was less than 5 cm (average water depth <3 cm) on 5 occasions where the high tide mark did not fully extend to the area covered by the nets. When corrected for these 5 samples, this mean density goes up to 83 fish per 100 m⁻². In addition, it is

likely that Gobiidae were undersampled on a couple of occasions where they were hiding in crab holes well after the marsh flat had drained after the spring high tide. We therefore consider the mean density of fish caught to be > 72 fish per 100 m⁻².

The mean catch and species richness \pm SE per net/sample was 18.11 \pm 2.58 individual fish and 2.60 \pm 0.22 taxa respectively. Both catch (r = 0.6113, p < 0.01) and species richness (r = 0.5131, p < 0.01) were positively correlated with mean water depth. However, there was no correlation between water salinity and either catch (r = 0.0842, p > 0.05) or species richness (r = -0.0249, p > 0.05). The range in salinity level was modest across the samples (33.1 to 36.6 ppt). Only two of the environmental variables measured were significant in the generalised linear models (Table 3). Fewer fish and slightly lower species richness were apparent in the daylight relative to night-time of the diel phase. In contrast, mean water depth had a strong positive effect on all four response variables. Permutational MANOVA revealed that location and the tide cycle (neap vs. spring) were the two most important influences (p < 0.001) on the fish community (Table 4).

Table 3. Coefficients for GLMs relating fish species richness, catch and the abundance of the two most common fish species to environmental variables. Values have not been exponentiated. The model for species richness uses Poisson regression, the other response variables follow a quasi-Poisson distribution. Significance levels are indicated as: *** p < 0.001, ** p < 0.01, * p < 0.05, . p < 0.10.

	Estimate	SE	t value	Pr(> t)	signif.
Species Richness					
(Intercept)	-1.7124	8.7227	-0.196	0.8444	
Location: Perkins Passage	0.0653	0.3400	0.192	0.8476	
Location: Robbins Passage	0.0750	0.3618	0.207	0.8357	
Status: Unaltered	-0.0613	0.1858	-0.330	0.7415	
Tide: Spring	-0.0782	0.4972	-0.157	0.8751	
Phase: Light	-0.6144	0.2487	-2.471	0.0135	*
Salinity	0.0660	0.2369	0.279	0.7806	
WDmean	0.0204	0.0077	2.639	0.0083	**
Catch Numbers					
(Intercept)	-7.5104	7.9607	-0.943	0.3510	
Location: Perkins Passage	0.0757	0.3289	0.230	0.8190	
Location: Robbins Passage	0.1768	0.3317	0.533	0.5970	
Status: Unaltered	-0.1332	0.1735	-0.768	0.4470	
Tide: Spring	0.0933	0.4700	0.198	0.8440	
Phase: Light	-1.4914	0.2993	-4.983	0.0000	* * *
Salinity	0.2643	0.2158	1.225	0.2280	
WDmean	0.0458	0.0080	5.719	0.0000	* * *
Smallmouth Hardyhead					
(Intercept)	-5.5752	9.0677	-0.615	0.5420	
Location: Perkins Passage	-0.5936	0.4059	-1.462	0.1520	
Location: Robbins Passage	-0.0056	0.3742	-0.015	0.9880	
Status: Unaltered	-0.1959	0.2030	-0.965	0.3410	
Tide: Spring	-0.2365	0.5553	-0.426	0.6730	
Phase: Light	-1.5162	0.3424	-4.429	0.0001	* * *
Salinity	0.1989	0.2461	0.808	0.4240	
WDmean	0.0521	0.0097	5.395	0.0000	* * *
Yellow-eye Mullet					

(Intercept)	-10.0165	18.8957	-0.530	0.5991	
Location: Perkins Passage	1.4328	0.7216	1.985	0.0542	
Location: Robbins Passage	0.1393	0.9812	0.142	0.8878	
Status: Unaltered	0.3963	0.3192	1.242	0.2218	
Tide: Spring	1.1728	0.9923	1.182	0.2444	
Phase: Light	-3.5222	1.3271	-2.654	0.0114	*
Salinity	0.2367	0.5092	0.465	0.6446	
WDmean	0.0576	0.0146	3.946	0.0003	* * *

Table 4. Permutational Multivariate Analysis of Variance (*adonis*) relating the overall fish assemblage to environmental variables (added sequentially first to last). Wtemp: water temperature, WDmean: mean water depth. Number of permutations = 999. Significance codes: *** p < 0.001, * p < 0.05.

Variable	df	SS	MS	F.Model	R ²	Pr(>F)	Significance
Location	2	0.7512	0.3756	3.922	0.1491	0.001	* * *
Status	1	0.0778	0.0777	0.8122	0.0154	0.53	
Tide	1	0.5761	0.576	6.0152	0.1143	0.001	* * *
Phase	1	0.2607	0.2607	2.7226	0.0517	0.04	*
Salinity	1	0.0606	0.0605	0.6327	0.012	0.641	
Wtemp	1	0.1317	0.1316	1.3747	0.0261	0.228	
WDmean	1	0.2044	0.2044	2.1347	0.0405	0.08	
Atherinidae	1	0.1586	0.1586	1.6563	0.0314	0.155	
Gobiidae	1	0.3256	0.3256	3.3999	0.0646	0.015	*
Residuals	26	2.49	0.0957		0.4943		
Total	36	5.0367			1		

MRPP showed there was no significant difference between the altered and unaltered sites based upon their fish assemblages (chance corrected within-group agreement A = -0.01202, based on observed delta = 0.6353 and expected delta = 0.6278, the significance of delta = 0.917. Ordination results further showed that there were almost no differences in the sampled fish assemblages between unaltered and altered sites (Figure 5).



Figure 5. nMDS ordination of the pop net samples based upon their fish communities. Stress in 2D = 0.1909. Samples from unaltered sites by closed circles, altered sites are represented by open circles. Status labels are plotted at their respective centroids.



A smallmouth hardyhead (Atherinosoma microstoma) caught in the sampling of Tasmanias saltmarshes. Photo credit: Vishnu Prahalad

6.5 Discussion

Fish species composition and density

This is the first study to document fish use of Tasmanian saltmarsh wetlands and lays the foundation for future work. The fish species encountered in this study overlap with those reported from other temperate Australian saltmarshes (Connolly 2009), dominated by species from the three families of Atherinidae, Gobiidae and Mugilidae. Of the other common fish families reported in temperate Victorian (Crinall and Hindell 2004) and New South Wales saltmarshes (Mazumder et al. 2005a, Mazumder et al. 2006b), species from Ambassidae, Gerridae and Sparidae were absent likely due to their geographic range not extending to Tasmania. Endemic species of the families Sillaginidae and Tetraodontidae were also not captured in our nets, though the presence of Tetractenos glaber (Smooth Toadfish) was observed adjacent to the nets in Big Bay and Robbins Passage. Notably, this study provided a rare record in Australian saltmarshes of a member of the recreationally and commercially valuable family Arripidae. The two other species of recreational and commercial interest recorded, A. forsteri and R. tapirina have been frequently reported from other temperate Australian saltmarshes. The relative abundance of *A. forsteri* in our total catch (19%) is, however, comparatively much higher for a member of the family Mugilidae (cf. <4% of total catch by Crinall and Hindell 2004 in Victoria and by Mazumder et al. 2005a and Mazumder et al. 2006b in New South Wales). This could not be fully attributed to the season of the sampling coinciding with the breeding period of A. forsteri given the variable sizes (~4-20 cm total length) of samples caught (Chubb et al. 1981). In terms of species richness, the 11 species recorded in our single season of sampling compares well with other temperate Australian studies where reports range from 10 species in Victoria (Crinall and Hindell 2004) to 14 species in New South Wales (Mazumder et al. 2005a).

Our observed density of >72 fish per 100 m^{-2} is higher than from other Australian saltmarshes, including from subtropical studies (Table 5.). The high density recorded in this study may be a seasonal artefact where sampling in autumn returned high fish catches. A study by Jin et al. (2007) of fish in Yangtze River saltmarsh in China found strong seasonal variation in mean abundance, although suggested that abundance was significantly higher during northern temperate spring and summer seasons. There is little evidence so far in Australia for significant seasonal variations in fish on temperate saltmarshes. Mazumder et al. (2005a) showed seasonal variation in fish abundance in mangroves, peaking in summer, but not in the case of saltmarshes near Sydney. Bloomfield and Gillanders (2005) also reported no significant differences in fish abundance and richness in saltmarshes from South Australia between months. A plausible explanation for the high fish density reported in this study could be the unique position of Tasmanian saltmarshes as part of seascapes where mangroves are absent. Consequently, Tasmanian saltmarshes are situated lower on the tidal frame and, in the case of our study area with a high tidal range, are partially flooded even during neap tides unlike most mainland Australian saltmarshes which get flooded only in spring tides (Bloomfield and Gillanders 2005, Connolly 2009). It is therefore valid to hypothesise that saltmarshes in our studied seascape are likely to provide higher habitat value for fish compared to Australian mainland marshes, due both to longer availability of flooded habitat and a lack of complementary habitat offered by adjoining mangroves.

Patterns of fish use and implications for coastal rehabilitation

It is expected that saltmarsh rehabilitation through restoring tidal flows bestows potential benefits for fish through expanded habitat (Roman et al. 2002; Raposa and Talley 2012). Two main findings in our study of patterns of fish use support this expectation. The first one relates to the strong effect of water depth on fish density and richness found in this and some other studies (Thomas and Connolly 2001; Connolly 2005). When currently tide-restricted areas are open to flooding, they will be accessed by spring tides and can accommodate the spread of a given volume of water (entering the embayment or sheltered passage) over a greater surface area. This opens up more shallow, sheltered environments, rich in food sources, preferred by juvenile and sub-adult fish species (e.g. *A. forsteri* and *Mugil cephalus*, Sea Mullet: Chubb et al. 1981). The second related finding from our study indicates that there was only a minor effect of the sampling location on both fish density and richness. Given that the study area has over 25 kms of levees

spanning multiple private properties, coastal rehabilitation works can be initiated wherever opportunities arise with likely benefits for local fish use. While the saltmarsh area already lost to clearing was 221 ha, a further 629 ha (55% of current extent) is affected by impaired tidal flows (Prahalad 2014) and can benefit from simpler on ground works (e.g. levee breaching) aimed at tidal restoration.

Difference in fish use between unaltered and altered saltmarshes

This study is also the first in Australia to document the difference in fish use between paired unaltered and altered saltmarshes (Connolly 1999, 2009). Our findings indicate that altered marshes can support high densities of fish and of comparable species richness to unaltered marshes. One of the reasons for high fish numbers in our altered marshes could be due to the greater marsh to edge ratio, a product of habitat fragmentation, allowing greater access to fish (Minello et al. 1994). A more substantive reason, however, could be just that altered marshes can provide similar habitat function for fish use if they are subject to the natural tidal regime comparable to its unaltered counterparts. There is considerable evidence from elsewhere, such as the Atlantic coast of temperate North America, of restoring saltmarshes having similar fish habitat value to reference sites (Raposa and Talley 2012). Further, restoration of tide-restricted marshes (i.e. into restoring ones) has been shown to return fish density and richness to levels comparable to unaltered ones within one year (Roman et al. 2002). Indeed, our spring tide samples from Robbins Passage and Perkins Passage altered sites were both located immediately behind the breached levees, and returned high fish density and species richness. It must be noted though that these altered sites had comparable tidal regime, plant composition and crab activity to their paired unaltered sites (an indication of some functional equivalence). A comparative study by Mazumder et al. (2006b) of three saltmarshes of the Sydney region indicated that one of the marshes reclaimed from dredge spoil had significantly lower abundance and diversity of fish, possibly due to lack of functional equivalence. The contrasting results from these two studies indicate an unexplored threshold effect in saltmarsh condition, likely context specific (e.g. with and without mangroves), which can help explain fish habitat value and guide tidal restoration efforts.

			No of		Diversity (number	Mean density	Temporal context		Water depth
Region	State	Reference	pop net releases	Total fish caught	of species)	, (fish per 100 ⁻²)	(sampling month)	Spatial context (with mangroves etc.)	(proxy for volume)
Subtropical	QLD	Thomas and Connolly 2001	134	577	23	17.2	August, January	Flats	4-72 cm
		Connolly 2005	88	1073	19	48.8	May, December	Flats, adjacent runnels and mangrove-lined creeks	6-48 cm
Temperate	NSW	Mazumder et al. 2005b	48	766	15	52.8	Year round (monthly)	Flats, adjacent mangroves	Not reported
		Saintilan et al. 2007	36	~568	14	52.2	Year round (monthly)	Flats, adjacent mangroves and seagrass	Not reported
	SA	Connolly et al. 1997	48	19	2	4.4*	April-July	Flats with creeks, adjacent mangroves and seagrass	10-30 cm
		Bloomfield and Gillanders 2005	30	1	1	0.4*	July, August, December- February	Flats, adjacent mangroves and seagrass	> 70 cm
	TAS	Present study	48	851	11	72.4**	April, May	Flats	0-70 cm

Table 5. Compilation of fish data from existing literature that report using pop nets on saltmarsh flats in Australia (cf. Connolly 2009; Wegscheidl et al. 2017).

*Pop nets were of 3 x 3 m size, covering an area of 9 m^{-2} ; all other studies listed covered an area of 25 m^{-2} or more.

**Mean density calculated for 47 releases as one of the nets failed.
Value of saltmarsh for fish and seascape food web

The high density of fish recorded in our study has implications for better understanding the contribution of saltmarsh to fish and the broader seascape food web. Saltmarshes are primarily regarded for their shallow and sheltered environment suitable for juvenile and sub-adult fish to shelter and feed, and also for trophic relay to the greater seascape food web (Kneib 1997; McPhee et al. 2015). Gut content analysis of *A. microstoma* from New South Wales found that they fed exclusively on crab zoeae found abundantly in the saltmarsh. The study also reported Gobiidae and Mugilidae feeding on polychaetes, insects, insect larvae and fine detritus, all resources which can be found in abundance on the saltmarsh (Platell and Freewater 2009, also see Mazumder et al. 2006a). Further, with respect to trophic relay, Mazumder et al. (2009) reported the highest density of zooplankton from saltmarsh, compared to adjacent mangrove, seagrass and open water during a high tide event at a New South Wales estuary. In addition, Svensson et al. (2007) demonstrated detrital pathways where estuarine fish were found to be feeding on detritus derived from temperate saltmarshes in Western Australia. These studies underscore an important role played by Australian saltmarshes in providing organic matter into marine food webs (a role now well recognised, e.g. Deegan et al. 2000; Mazumder et al. 2011).

In a broader context, an ongoing debate on seascape fisheries corresponds to the relative importance of different habitat types, including saltmarshes (see Figure 2), to the seascape food web. While few studies from Australia have simultaneously compared saltmarsh with other nearby habitats (mangroves, seagrass and unvegetated/open water), they differ in their valuation of saltmarsh relative to other habitats with respect to fish use. Bloomfield and Gillanders (2005) noted that saltmarsh had the least number of fish (a solitary A. microstoma for a saltmarsh area of 270 m⁻²), compared to mangroves, seagrass and unvegetated habitats of a South Australian estuary. Similarly, Saintilan et al. (2007) reported lower numbers of fish in a New South Wales saltmarsh compared to nearby mangrove and seagrass. The latter study however showed that fish moved between these habitats and that saltmarsh plays both a complementary role in terms of added food resource and also a refuge role for smaller fish during spring tides (when the seagrass habitat is 'exposed' to larger predatory fish). Mazumder et al. (2005a) also reported more fish in the mangrove relative to saltmarsh in the same New South Wales saltmarsh. However, they noted that fish density was higher in saltmarsh when corrected for water volume. A common emphasis of these and other overseas literature sources (e.g. Valiela et al. 2000) has been on the role of a functioning seascape matrix of habitats for fish to access at varying timescales. The value of saltmarsh for fish and seascape food web is very likely higher in Tasmania with the absence of mangroves at high latitudes.

Management implications

The priority for management in the context of our study area and other similar locales is primarily the protection of existing saltmarshes and their tidal connectivity, both for unaltered and altered sites with breached levees. Of particular importance are the Boullanger Bay and Robbins Passage areas which are least affected by levees and other associated clearing and drainage activities (see Figure 1 and Prahalad 2014). Further, targeted tidal restoration can be undertaken in areas of Big Bay, Perkins Passage, and other nearby areas of Duck Bay and West Inlet. In addition, activities aimed at the rehabilitation of the buffering *M. ericifolia* swamp forests could benefit the broader functioning of the seascape, through enhanced detrital pathways (e.g. Svensson et al. 2007) or reduced nutrient stress on the seascape from the nearby cattle farms (Holz 2009). A further aspect for management would be to focus on science communication of the links to recreational and commercial fisheries, including oyster farming. North-west Tasmania, as with many other areas of Tasmania and elsewhere, is renowned for its fishing culture, and fish may well be an important and locally unexploited avenue for community engagement towards seascape rehabilitation and management (Wegscheidl et al. 2017).

This study provides the first documentation of the fish species present in Tasmanian saltmarshes, their densities and patterns of use. The temporal scale of this study is a known limitation, and sampling during different seasonal periods is an important area for future research. A central aim of this study was to evaluate the difference in fish use between unaltered and altered saltmarshes spread across the larger Circular Head coastal area. There was no significant effect of altered saltmarsh habitat on fish assemblages as long as they are tidally connected. There was also no strong effect of sampling location of fish density and richness. This indicates that restoring tidal flows to marshes cut off due to the levees would provide habitat suitable for fish use despite its historically 'altered' condition owing to levee works and related impacts such as ditching, clearing and grazing. The findings reveal a hitherto unrecognised aspect of Tasmanian saltmarshes and provide a foundation for further research coupled with rehabilitation efforts. While the importance of saltmarshes is being recognised for a range of factors, fish remain a compelling subject with broad resonance. Improving our understanding of fish use of saltmarshes could raise much needed public awareness and material support for saltmarsh rehabilitation.

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CHAPTER 7 - BUILDING A BUSINESS CASE FOR SEASCAPE REPAIR AND PROTECTION

7.1 - Seascapes – a checklist of their ecosystem services

Key components of any ecosystem services inventory for seascapes include:

✓ Commercial and recreational food value

A diverse community of finfish, crustaceans and shellfish use seascapes for food, shelter, spawning, nursery areas and refuge from predators. These ecological functions have substantial economic value. A large proportion of Australia's commercial or recreationally caught fish and shellfish species depend on estuaries and coastal wetland habitat at some point in their life cycles, including most prawn species, many crab species, and many of the recreational and commercial target fish species such as the mullets, flatheads, bream, whiting, barramundi and mangrove jack.

✓ Erosion control and storm surge protection

Vegetation helps shield the upland from erosion by waves and currents. In addition, seascapes can absorb and moderate much of the impact of coastal storm surge and floods.

✓ Water quality and quantity maintenance

As seascapes slow and retain water, these ecosystems also filter pollutants, sediments and nutrients. Both nitrogen and phosphorous can be absorbed by plants and form part of the coastal food web. Nutrients e.g. phosphorous can also be bound in sediments, and then deposited. Nitrogen can be lost as gas, leached as nitrate or exported in the form of animal or bird tissue.

✓ Carbon sequestration

✓ Seascapes along with mangroves and seagrasses are collectively known as "blue carbon" and sequester more carbon per hectare than any terrestrial ecosystems.

✓ Coastal biodiversity

Seascapes as part of the coastal wetland continuum provide the habitat for our coastal biodiversity.

✓ Support of coastal food webs

Coastal wetlands perform a critical role in uploading terrestrial nutrients to underpin the productivity of coastal marine food webs.

✓ Australia's coastal lifestyle

Seascapes provide exceptional sites for access to recreational fishing sites, crabbing in their tidal channels, and enriches the visual landscape for painting, photography and bird watching.

These have been listed in order from fully quantifiable as readily accessible even flow market-based values [e.g. food value] to event dominated characterised by uneven flows [e.g. erosion and water quality] through to intangible non-market values [e.g. biodiversity and lifestyle]. Carbon sequestration in time may become readily quantifiable. This will occur once Australia has developed and adopted a policy framework and measurement protocols for assessing and accounting for "blue carbon". For the more intangible benefits economic valuation tools such as stated preference techniques (Hanley et al. 2001) need to be applied to provide some estimate of \$ value.

This project has as its central objective to provide valuation information for repairing seascapes that is well accepted in decision-making frameworks from senior policy maker through to local community advocate. If ecosystem repair is to overcome social impediments to broadscale adoption, simple, readily understood messages must dominate. It follows that only simple economic valuation tools can be applied.

A suite of criteria to meet the objective of the valuations being readily accepted and applied in decisionmaking processes include:

- Clear, simple and readily understood calculations with these calculations explicit and well documented in the analysis to determine dollar-value
- Valuations that are well founded and preferably based on Australia's existing market economy
- Conservative and generally under-estimates of value, with only selected, usually single benefit streams used in the valuation process
- Benefit streams are accompanied by lists of assumptions that clearly demonstrate that the values are conservative
- Many other benefits are listed rather than valued thereby also demonstrating the conservative nature of the results.

The only metric from the checklist of ecosystem services provided by seascapes that currently meets the criteria is "Commercial and recreational food value', based on the economic worth in the market place of seafood. Choosing prawns and potential changes in prawn biomass as the basis for valuation calculations has added advantages. Prawns are annual highly fecund stocks that will rapidly expand in population size by exploiting repaired habitat. Prawns are also highly visible seafood products, generally in demand and well understood as a potential benefit stream from policy maker to community advocate.



Mangroves and saltmarshes can provide important nursery habitat for a multitude of species. Photo credit: Paul Boon.

7.2 Valuing the benefits of repair to seascapes

East Coast Tropical

Banana prawns, *Fenneropenaeus merguiensis*, use tropical estuaries as nursery grounds. Banana prawns were chosen as a study species because Banana prawns are:

- (a) highly fecund and an annual stock that will recruit rapidly to repaired environments,
- (b) a commercially important food species,
- (c) important targets of recreational fishers throughout north Queensland estuaries,

(d) vital prey of other high profile commercial/recreational species such as barramundi, and

(e) known to exhibit a random escape response which means that an assumption of minimal undersampling is valid.

Fish are highly variable in escape-patterns to sampling by most types of sampling gears that can provide accurate estimates of density. Compared to fish, banana prawns are ideal targets, particularly for cast net sampling, a gear type that is particularly suitable for small mangrove lined estuaries and this gear type provides for accuracy in sampling through generally high replication (Johnston & Sheaves 2007). Banana prawns as a highly valued stock for both commercial and recreational catch also makes them an ideal species to use in any broad estimates of the benefits of seascape repair.

The tropical study and prior work found that estimates of productivity of individual components of the estuary were highly variable and depended on a number of difficult to validate assumptions (Minello et al. 2008, Rönnbäck et al. 1999, Rozas & Minello 2007). Estimates at the whole-of-estuary level at the "seascape" approach in comparison required a relatively low number of assumptions and produced estimates with relatively low variability. The study Sheaves et al. (in review) found as a conservative estimate a maximum juvenile prawn biomass of 6.5 g/m² for the 2m wide band along the estuary edge where prawns are found. For the estuary studied, with an edge area of 5.6 ha and 11.5km total length, the conservative total biomass of juvenile pawns was 0.37 tonnes.

The actual estuary productivity would be much higher because this estimate only relates to the maximum juvenile stock for a sampling occasion and doesn't take account continual movements of prawns to offshore adult habitat once they reach a sufficient size. To more precisely calculate estuary productivity for Banana Prawns information would be needed on patterns of recruitment, growth rates, mortality, predation and emigration. Furthermore, estuaries are not mono-specific and similar information would also be required for all other crustacean and finfish as well as the net primary productivity that is exported from the estuary. Suffice it to say an estimate of Banana Prawn productivity of 0.37 tonnes for an estuary of 11.5 km total length is probably orders of magnitude below total estuary productivity. Nevertheless it provides a baseline estimate that can be used to demonstrate the potential benefits of seascape repair.

Many studies for the Great Barrier Reef coast have catalogued the decline in areas of wetland / seascapes, the loss of connectivity, the deterioration of water quality, loss of key sub-tidal habitat such as seagrasses, the more rapid flow of runoff, the reduced stream flow persistence following rain events and the constriction of tidal flows (Wegscheidl et a. 2017). All of these changes adversely impact on estuary productivity.

The next phase of research investment most likely to yield valuable management orientated information is to couple the multiple GIS-based data sets on Reef coastal wetlands with a suite of best available predictions for productivity opportunities, key fish and crustacean species. This would yield estimates of total likely productivity benefits should investment be available to repair seascapes, especially such as connectivity. Worked case studies of particular high profile opportunities could be used to illustrate the complete suite of likely benefits.

East Coast Subtropical

School Prawns [*Metapenaeus macleayi*] was chosen for the east-coast subtropical case study for several reasons. First, School Prawns are highly reliant on estuarine nursery habitat, and primary productivity derived from estuarine habitats for rapid growth through their early life history stages. Second, the species is important to both commercial and recreational fisheries. Third the species is highly fecund, and given reasonable freshwater inflow to estuaries it is unlikely to experiment stock-related limitations to recruitment. Finally, School Prawn is both fast growing and an annual species. Notwithstanding the time taken for any repaired habitat to recover, any associated benefits of repair are likely to be evident in this species over a very short time frame.

While School Prawn is mostly commercially harvested, this commercial harvest provides product for both consumption and also represents the most widely used bait product for recreational fisheries. Locally sourced product for both bait and human consumption is already becoming more important in light of the recent white-spot (WSSV) outbreaks in Queensland. This has directly led to 1) restrictions on imported prawn meat for consumption; 2) disposal of significant local grown aquaculture product which has led to supply problems for local markets; and 3) biosecurity-related restrictions on the movement of uncooked prawns across most Australian borders to prevent transmission of the disease among jurisdictions.

Noting the assumptions outlined in Chapter 4, our estimates indicate that reinstatement of connectivity of 27.6ha of shallow sub-tidal creeks and subsequent colonization by School Prawns (under good recruitment) could yield ~2,500 kg of product equating to a gross value of ~AUD24,000 and associated total output of ~AUD140,000 per year. When converted back to a per-hectare estimate, these values equate to ~AUD900 ha⁻¹ y⁻¹ and ~AUD5,000 ha⁻¹ y⁻¹ respectively for marsh channel habitat. If the total amount of habitat loss in NSW was known, these values could conceivably be applied to other areas to determine the value derived from habitat repair at a statewide level.

The benefits of habitat repair are not limited to the values estimated from direct usage of the habitat for School Prawn. Saltmarsh habitats contain important primary producers that contribute to the overall productivity of the estuary, and consequently they make substantial contributions to the exploited biomass harvested from estuarine systems. Potential gains in primary productivity when these habitats are reconnected to the broader estuary will be outwelled to other areas across the estuarine system. This can occur through a number of mechanisms including the transport of particulate organic carbon (POC), transport of dissolved organic carbon (DOC), or consumption of marsh plants by small nekton on the marsh surface (when inundated), and subsequent movement throughout the estuary. These additional benefits are not captured in the analysis presented here, but could contribute to a fishery-derived value of up AUD20,000 ha⁻¹ y⁻¹ of areal saltmarsh that is reconnected to the estuary in the Clarence River system.



A housing estate abutting a saltmarsh. Photo credit: Paul Boon

In addition, any re-connected subtidal channels arising from repair will also provide habitat to directly support other target species such Mud Crab (*Scylla serrata*), Dusky Flathead (*Platycephalus fuscus*), Yellowfin Bream (*Acanthpagrus australis*), Luderick (*Girella tricuspidata*) and Sea Mullet (*Mugil cephalus*) (Mazumder 2009; Morton et al. 1987; Webley et al. 2009). Direct support of adults and/or juveniles of

these exploited species will produce fishery benefits that contribute additional value from habitat repair. Both these factors will see flow on benefits for recreational and commercial fisheries alike.

The next phase in science enquiry to support marine biodiversity repair will be to take these estimates of likely productivity benefit and apply it to the entire Clarence estuary. Building a visionary scenario for estuary repair that could maximize community benefit requires the biological information from this study to be coupled with companion investigations of flood management, catchment hydrology, a full assessment of economic costs and benefits, engineering works and an understanding of social feasibility. An opportunity exists to develop a full multi-disciplinary program of science enquiry closely linked to and with the support of key management agencies and beneficiaries.

East Coast Temperate

Saltmarshes of the Circular Head region of north-west Tasmania were found to support 11 fish species with a high mean density of > 72 fish per 100 m⁻² (sample data from April-May 2017). The family Atherinidae contributed 3 species and 74% of the total catch numbers, of which *Atherinosoma microstoma* and *Leptatherina presbyteroides* were most abundant (57% and 16% respectively). Two members of the family Gobiidae, *Pseudogobius* sp. and *Nesogobius maccullochi*, contributed 3% and 2% to numbers respectively. Commercial and recreational species that utilize these saltmarshes in north-west Tasmanian seascapes include: *Aldrichetta forsteri* (yellow-eye mullet), *Arripis truttaceuos* (Australian salmon) and *Rhombosolea tapirina* (greenback flounder). These three species contributed close to 20% of the total catch numbers. Of these *A. forsteri* was most abundant and common, present in 24 (65%) of the 37 nets that caught fish and made up 19% of the total catch. Extended sampling throughout the year may reveal further species using saltmarshes.

A. forsteri, *A. truttaceuos* and *R. tapirina* are among the seven key species targeted by recreational fishers in Tasmania (Lyle et al. 2014). Notably, *A. forsteri* and *A. truttaceuos* help underpin recreational fisheries in the north-west region of Tasmania, with by far the greatest proportion of mullets and salmon (74% and 23% of recreational catch in 2012-13) being caught from this region (Lyle et al. 2014). In the north-west region, recreational fishing effort is almost entirely (93%) by local residents, with *A. truttaceuos* and *A. forsteri* being the second and third most caught after flathead (Platycephalidae) species (Lyle et al. 2014).

Fishes of the family Atherinidae and Gobiidae are not targeted directly by commercial and recreational fishers in Tasmania. They provide an abundant food source for other piscivorous fish that are targeted by commercial and recreational fishers (cf. Mazumder et al. 2011). Most importantly, these are part of the suite of species that contribute to overall marine biodiversity and productivity. These seascapes contribute more broadly to the marine food web via export of plant and animal matter to coastal waters, thus improving seascape fisheries productivity (Melville and Connolly 2003; Svensson et al. 2007). Construction of food webs encompassing all aspects of seascape productivity – e.g. carbon, diatoms, phytoplankton, fish, crustaceans, insects, birds and mammals would prove a useful way to demonstrate the multiple benefits the community enjoys from healthy seascapes.

The priority for management in north-west Tasmania and other similar locales is primarily the protection of existing saltmarshes and their tidal connectivity, both for unaltered and altered sites with breached levees. About 629 ha (55% of current extent) of saltmarsh in the Circular Head coast is affected by impaired tidal flows (Prahalad 2014) and can benefit from on ground works (e.g. levee breaching) aimed at tidal restoration. Samples taken from saltmarshes heavily altered due to levees indicate similar fish assemblages to nearby unaltered marshes, indicating that tidal restoration is likely to return immediate benefits for fish use through expanded habitat and food resource (cf. Roman et al. 2002; Raposa and Talley 2012). Table 1 illustrates the potential benefits for just one species, *A. forsteri* of improved protection and management just in this case study area:

Table 1: Benefits of seascape restoration to A. forsteri

Estimate of total Circular Head coast saltmarsh area that can benefit from	629 ha
tidal restoration and saltmarsh rehabilitation	
Estimate of fish density of <i>A. forsteri</i> per 100m ⁻² averaged across the study	13.6 fish
sites	
Estimated gain in A. forsteri with tidal restoration of 100 ha saltmarsh area	136,000
	fish

The next phase of science enquiry for these temperate systems is similar to that for subtropical Australia – to undertake management orientated investigations that clearly identify both protection and repair opportunities. Food webs that illustrate the overall likely community benefit will play a key role in fostering improved community and agency understanding and hopefully investment in securing an enhanced future for Tasmania's coastal marine biodiversity.

7.3 A framework for informing future investment in seascape repair and learning

While acknowledging a suite of ecosystem services associated with repair, this report has emphasized benefits stemming from increased harvest for human consumption of a subset of species. If these benefits are estimated to be greater than the costs of implementation, then a prospective repair project has a benefit-cost ratio of *at least* 1.

Our biological understanding of the magnitude of stock increase associated with any specified repair action remains rudimentary. Predicting the payoff of investment in repair projects is clearly difficult (see chapter 2). But risk and uncertainty are ubiquitous features of many kinds of investment. Delaying a decision until uncertainty is more or less entirely resolved carries the cost of foregone benefits. It also ignores the benefits of learning via implementation through adaptive management (Walters 1986). Here we outline a decision support framework for considering investment in repair under uncertainty.

A primary source of uncertainty is the size of the increase in yield or quota a repair project might bring. For example, in chapter 4 the key uncertain variable was recruitment subsidy associated with restoration of a discrete area of habitat within the Clarence River estuary, and its implications for School Prawn biomass and harvest.

Assume that we are considering repair for three hypothetical candidate sites, A, B and C, all of which are motivated primarily by an increase in prawn abundance and availability. Although we may not know the true magnitude of the recruitment subsidy, we can use expert judgment to estimate the probability of a discrete set of possibilities and estimate associated improvements in quotas (Table 2).

Considering a single candidate project

Considering Site A first, the *risk-neutral* approach is to calculate the *expected* benefit using the probability weighted difference between estimates with and without repair:

0.25 × (250 - 100) + 0.50 × (700 - 300) + 0.25 × (950 - 400) = 375 kg/yr.

That is, our risk-neutral estimate of the pay-off for repair at Site A is an additional harvest of 375 kg/yr, on average.

with repair			without repair				
	pessimistic	best estimate	optimistic		pessimistic	best estimate	optimistic
	p = 0.25	p = 0.50	p = 0.25		p = 0.25	p = 0.50	p = 0.25
site A	250	700	950		100	300	400
site B	400	900	1200		200	550	700
site C	200	600	800		150	400	500

Table 2. Estimated annual harvest quotas (kg per year) for three hypothetical candidate repair sites.

Now if the market price for prawns is \$10 per kg, we can estimate the present value, PV, of the benefit,

$$PV = \left(\frac{A}{r}\right) \cdot \left(1 - \frac{1}{(1+r)^h}\right),$$

where A is the annual benefit, r is the discount rate and h is the time horizon (in years) over which the repair project is to be assessed. For A = \$3,750, r = 4% or 0.04, and h = 30 years, PV = \$64,845. If the (discounted) costs of implementing the project are less than \$64,845 then the risk-neutral decision-maker will proceed with implementation, knowing that the expected ratio of benefit to cost exceeds 1.

Of course, not all decision-makers are risk-neutral. Should the pessimistic scenario play out, a lower bound on the present value of the (250 - 100) = 150 kg/yr benefit is just \$25,938. For the optimistic estimates, the upper bound on present value is \$95,106.

So depending on risk attitude, a decision-maker may be prepared to incur project costs up to AUD\$25,938 for the risk-averse, \$64,845 for those that are risk-neutral, and \$95,106 for those that are risk-seeking.

If the actual costs are estimated to be below the break-even point for the risk-averse, the project is clearly worth undertaking. If costs are above the break-even point for the risk-seeking, the project is clearly non-viable. If costs are in the interval [\$25 938, \$95 106] the decision-maker needs to consider their attitude to risk. In addition, the prospects for transferring learning outcomes to other speculative projects and investments may be very much worth considering, as we outline below.

Considering multiple candidate projects.

Prioritization when we have no interest in learning

Returning now to the full set of three candidate sites, A- C, for which (uncertain) benefits are shown in Table 2. After applying the calculations shown above to Sites B and C we report best estimates and plausible bounds for the present value of the benefit of repair at each of the three sites in Table 3.

Table 3. Best estimates and plausible bounds for the present value of benefits for each of three candidate repair projects.

Present value of benefit	site A	site B	site C
lower bound	\$25 938	\$34 584	\$8 646
best estimate	\$64 845	\$60 522	\$32 423
upper bound	\$95 106	\$86 460	\$51 876

The estimated costs of repair are shown in Table 4 below. Up-front costs include capital works and compensatory payments to landholders for inundation of otherwise productive land, among other possible impacts. Ongoing costs are to be incurred for maintenance. Using the same formula above for calculating the present value of maintenance costs (again with a 30 year time horizon and a 4% discount rate), we obtain total costs for each candidate project.

	site A	site B	site C
costs of capital works	\$8,000	\$7,000	\$10,000
costs of landholder compensation	\$10,000	\$25,000	\$20,000
annual cost of ongoing maintenance	\$1,500	\$500	\$1,000
Present value of total costs	\$43,938	\$40,646	\$47,292

Table 4. Costs for each of three candidate repair projects.

There are now two ways to summarise the merit of the candidate projects. In Table 5 we report net monetary value (the present value of benefits – the present value of costs). In Figure 1 we graph the benefit to cost ratio.

Table 5. Net monetary value (present value of benefits – present value of costs) for three candidate repair projects.

net monetary value	site A	site B	site C
lower bound	-\$18,000	-\$6,062	-\$38,646
best estimate	\$20,907	\$19,876	-\$14,869
upper bound	\$51,168	\$45,814	\$4,584



Figure 1. Benefit-cost ratios of the three candidate repair projects, with plausible bounds.

The risk-neutral decision maker focusses on best estimates. Risk-averse decision-makers focus on lower bounds, and risk seekers on upper bounds. The priority order of the three projects depends on risk attitude where B is (weakly) preferred to A, and C is non-viable if you are risk-neutral; A is (weakly) preferred to B, and B is preferred to C if you are risk seeking; and none of the projects may appeal if you are risk-averse, with B being the best of a bad lot!

Prioritization when we are able to learn

Figure 1 shows the lower bounds for all three sites are below 1 (or equivalently all lower bounds of net monetary value shown in Table 5 are negative), so the most risk-averse of decision-makers is unlikely to progress implementation of any of the three candidate projects. But this outlook ignores the potential gains to be made through learning about the true pay-off of repair via implementation and monitoring of outcomes. The information might lead to more astute investments in future.

What is the expected value of information? In Table 6 we show for Site A the net monetary value (present value of benefits – present value of costs) for the choice between do nothing and implementation under each uncertain state for recruitment subsidy.

 Table 6
 Pay-off of do nothing versus implementation for candidate project A under each uncertain state .

		Recruitment subsidy	
decision	pessimistic	best estimate	optimistic
	p = 0.25	p = 0.50	P = 0.25
do nothing	\$0	\$0	\$0
implement	-\$18,000	\$25,230	\$51,168

If we knew for certain the recruitment subsidy was low, consistent with the pessimistic estimate, we would do nothing rather than incur an \$18,000 loss. If we knew the recruitment subsidy was *not* low, we would proceed with implementation. If we had perfect information the pay-off would be zero, or \$25,230, or \$51,168. Now the *expected* value *with* perfect information is

EV | PI = 0.25 × \$0 + 0.50 × \$25,230 + 0.25 × \$51,168 = \$25,407.

The expected value of the two options under uncertainty is:

EV | U = 0.25 × \$0 + 0.50 × \$0 + 0.25 × \$0 = \$0, for do nothing, and EV | U = 0.25 × -\$18,000 + 0.50 × \$25,230 + 0.25 × \$51,168 = \$20,907, for implementation.

If we have to make a decision under uncertainty (and are risk-neutral) we would opt for implementation.

Now the expected value of perfect information is the difference between these two quantities (Pratt et al. 1995). That is,

EVPI = EV|PI - max EV|U

EVPI = \$25,407 - \$20,907 = \$4,500.

So if we were able to pay someone to tell us what the true recruitment subsidy was to going to be if we were to implement repair at Site A then a fair price for that service would be \$4,500. Of course, such a service in (credible!) clairvoyance is unavailable. We would have to implement the project and monitor outcomes. But having implemented the project, the outcomes of monitoring are uninformative at Site A. The benefits of learning are realized if those insights are transportable to Sites B and C.

Results of the equivalent calculation of EVPI for sites B and C are shown in Figure 2, together with Site A. Assuming monitoring outcomes are transportable, an upper bound on the learning benefit of implementation at Site A is the sum of EVPI for sites B and C. That is if we implement at,

Site A, there is a \$1,515 + \$1,146 = \$2,661 learning benefit for Sites B and C, and at Site B, there is a \$4,500 + \$1,146 = \$5,646 learning benefit for Sites A and C, and at Site C, there is a \$4,500 + \$1,515 = \$6,015 learning benefit for Sites B and C.

Of course we would also need to consider the costs of learning via monitoring. Nevertheless, looking at the outcomes in Figures 1 and 2 we are now in a much better position to choose a project that (a) has good prospects for outcomes at the site of implementation, and (b) enables an improved basis for future decisions at other candidate sites. Site A looks like a good bet for immediate implementation if we are only interested in outcomes at Site A, but it has only modest learning benefits, with a combined value of information of \$2,661for improved post-learning decision-making at Sites B and C. The opposite is true for Site C. Implementation at Site B best satisfies our two objectives – there are strong prospects for positive outcomes at Site B and solid learning insights to be gained for subsequent decision-making at Sites A and C.



Figure 2. Expected value of perfect information for each of the three candidate projects.

The framework outlined here shows how to make effective restoration decisions despite uncertainty. It can be readily extended to include continuous probabilistic judgments and additional sources of uncertainty, including for example, the rate and intensity of climate change (IPCC 2014). We note that expert judgment need not be a critical bottleneck. There are simple and accessible protocols available for eliciting the kinds of judgments used in our hypothetical example here (Burgman et al. 2011).

The framework argues against use of uncertainty as an excuse for procrastination. It emphasizes that even where uncertainty makes the stand-alone merit of a candidate repair project unclear, the benefits to be gained from learning through implementation and subsequent monitoring may make implementation worthwhile. In the early stages of a repair and restoration enterprise, where there are many candidate sites for repair, there will be many highly speculative investments but many opportunities for deriving benefits through learning. As implementation progresses and uncertainty is reduced, the list of viable candidates

should become progressively smaller and their site-specific benefits (and costs) appreciated with much greater clarity.

Concluding Comments

The three case studies have demonstrated the substantial benefits to accrue from seascape protection, management and where possible, repair. While only indicator species were used, the benefits are multiple. Equally importantly, even with just the value of the indicator species such as school prawns is used the economic argument is compelling – with the value of school prawns exceeding current land values for the case study site of the lower Clarence.

Under the Environmental Protection and Biodiversity Conservation Act (EPBC Act) subtropical and temperate coastal saltmarshes were listed as vulnerable in 2013 and that a Recovery Plan is required. The following suggestions build on this need and may assist in defining the scope and content of the proposed Recovery Plan.

Firstly, this project has demonstrated the likely dual biodiversity and economic benefits of repair. Most importantly this brings into play another key user group additional to conservation groups and agencies that will support and in some instances, may be able to provide resources for repair. Most states have in place or are moving to put in place policies and procedures to allocate funds to repair habitat. For example, WA, NSW and Vic already provide via their recreational fishing license programs guaranteed funds available off consolidated revenue. These opportunities for multi-beneficiary outcomes would strength any implementation activities for the Recovery Plan.

Recognising that saltmarsh systems are also listed under NSW legislation as endangered and that recreational fisheries groups in NSW are very active in advocating habitat repair, in the formulation of the Recovery Plan NSW may be the highest priority for repair. Probably the best estuary to initiate such work would be the Clarence. The Clarence is the largest estuary in NSW, was the subject of this research so the facts and figures are available, and has very active commercial fishing and recreational fishing groups. The Clarence is also bounded by large areas of National Park, Bundulung and Yuragir, both of which would benefit by extension into repaired estuary environments. As a signature species that will benefit from repair, the school prawn is also well recognised as synonymous with the Clarence Estuary. The Clarence Estuary is the biggest and most productive estuary in NSW and worthy of detailed multi-disciplinary research across ecology, hydrology, flood management, economics and social issues to develop a draft blueprint for repair at the whole-of-estuary scale. Such a proactive multi-disciplinary research project would both inform the Recovery Plan and extend the outputs of research under the Marine Biodiversity Hub into the outcomes of securing long-term enhanced estuary biodiversity.

The EPBC listing and proposed Recovery Plan does not extend to tropical environments such as the Great Barrier Reef catchments. Parallel processes for coastal tropical landscapes of the Great Barrier Reef are underway via the Reef Plan 2050. Reef Plan 2050 already calls for the repair where possible of these coastal marine seascapes. The next task for Great Barrier Reef seascapes is to bring together the multiple layers of GIS-based mapping on coastal seascapes and determine which subset of these are the most prospective for repair.

The third case study area for this project was temperate systems in Tasmania. The first task for restoring Tasmanian temperate seascapes is most probably legislative - to gain formal recognition and protection of existing remnants. In parallel, especially for areas such as the case study, repair works could start immediately, benefitting coastal marine biodiversity at no cost to other land uses. Building into the Recovery Plan support for companion legislation in both Tasmania and Victoria is recommended.

Much of the repair activities are simple from a works perspective – generally removing small bunds to reinstate tidal connectivity. These repair activities are also inexpensive. Future work needs to build on the prioritization framework outlined here to identify the best sites for immediate investment, and those that offer the best prospects for learning.

Together with greater clarity and rigor in identification of suitable repair sites, a key challenge in fostering enhanced coastal marine biodiversity will be to address the social issues, to build community understanding of the opportunities to optimize benefits from their coastal landscapes and to overcome resistance to change.

7.4 References

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