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Managing Estuaries for Resilience Under Climate Change: Integrating Socio-Economic and Ecological Goals and Proposing Appropriate Strategies

by
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Abstract

Australian estuaries should be able to meet the needs and aspirations of society and ecological integrity in the face of future change, while continuing to support integrated human and ecological values. This report presents a new approach to the challenging task of making decisions regarding estuaries with respect to climate change adaptation through the introduction of estuarine focused adaptation strategies.

The range of communities likely to be present in estuaries, and their associated values have been summarised in the report, as well as ecological and socio-economic goals for these communities. This will better enable those making management decisions for estuaries to consider the full range of estuarine communities. Adaptation strategies, and examples, are presented to provide a framework approach for decision making. Case studies of seven Australian estuaries; Towra Point, Georges River Estuary, Wilson Inlet, the Mary River, Gippsland Lakes, Tomago Wetland and the Richmond River Estuary, are used to illustrate past estuarine management successes and failures, and provide examples of estuarine goals and associated strategies.

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1. Introduction

The challenge for the 21st century, then, is to understand the vulnerabilities and resilience of ecosystems so that we can find ways to reconcile the demands of human development with the tolerances of nature (World Resources Institute, 2001).

The most critical task facing humanity today is the creation of a shared vision of a sustainable and desirable society (Costanza, 2000).

Estuaries are extremely valuable ecosystems, sitting between rivers, land and the open ocean. They provide numerous anthropocentric and ecocentric values (Pendleton, 2008). These values include transport, waste disposal, recreation/aesthetics, fisheries and habitat for hundreds of species, some of which are endemic to estuaries (Perkins, 1974). Despite providing so much, estuaries are also recognised as being vulnerable to human pressures (Kennish, 2002) and, consequently, need to be managed in order to sustain desired goods and services.

In addition to existing pressures, climate change is clearly a major threat to coastal communities and estuarine ecosystems (Harley *et al.*, 2006; IPCC, 2007; Byrne, 2011; Gillanders *et al.*, 2011; USEPA, 2011). Appropriate future management must recognise and anticipate changes and plan accordingly using combinations for mitigation and adaptation. *Climate Change Adaptation* accepts climate change and addresses unavoidable impacts with various options (Klein *et al.*, 2001). Unfortunately, this is an extremely difficult management challenge, especially if socio-economic and ecological aspects are considered together in a systems approach. These complex social ecological systems (SESs) are populated by numerous human and ecological stakeholders; the goals for which are sometimes in conflict. Nonetheless, it is essential to adopt an SES approach in order to evade two common false assumptions in natural resource policy. The first assumption is that human and natural systems can be treated independently. The second is that ecosystem responses to human pressures are linear, predictable and controllable (Folke *et al.*, 2002). In addition to these assumptions, a SES approach to climate change adaptation also decreases the risk of maladaptation, whereby the adaptation action has negative consequences or perverse outcomes elsewhere in the system.

This SES approach has been termed “resilience thinking” (Walker and Salt, 2006) and is both inclusive and complex. It is inclusive because it links economic, social and ecological systems and also accommodates panarchy, the cross-scale, dynamic interactions between human and natural systems. It is complex because it takes a holistic, dynamic systems view, considers spatial and temporal scales, and the synergisms of multiple pressures. Importantly, it recognises the possibility of non-linear, discontinuous ecological responses that create alternative states when thresholds have been exceeded (Carpenter *et al.*, 2011). An estuarine example would be when excessive nutrients cause a clear-water, seagrass-based estuary to flip to a turbid, phytoplankton system (Harris, 1999). Although the SES approach is in its infancy, it has particular relevance to estuaries since they are of immense importance for recreation, fisheries and urban development (the socio-economic dimension of resilience) and they provide several critical ecosystem services dependent on intact ecosystems (the ecological dimension of resilience) (Costanza *et al.*, 1997).

Estuaries are particularly threatened by climate change by virtue of the fact that changes over the land and in the world’s oceans will manifest themselves in the coastal ecosystems. In light of these threats, the sustained condition of estuarine ecosystems will depend on both their ability to adapt to climate change and the responses of humans that live, work and play in the coastal zone. This is particularly true within a single estuarine system, but there may also be consequences of management actions on adjacent ecosystems, settlements and infrastructure. To this end, a SES approach that integrates the connectivity both within and between estuarine ecosystems and across social, economic and environmental sectors is the best approach through which climate change adaptation decisions can be made.

Regularly management can be ineffective without stated desired objectives (Segan *et al.*, 2010; Wintle *et al.*, 2011). It is now common practice to focus management through the development of a vision statement. Accompanying this vision statement are goals consistent with the vision, strategies for reaching these goals, monitoring to assess progress and an active adaptive capacity to be engaged if the strategies are failing. While planning documents for estuaries have existed for years (e.g. NSW Government, 1992), the threat of climate change demands fresh thinking. Consequently, for estuarine SESs, this paper proposes (a) a vision statement that is consistent with public policy, the principles of ecologically sustainable development (ESD) and both ecocentric and anthropocentric ethics, (b) a range of ecological and socio-economic goals and (c) strategies to achieve the goals. As well, some case studies are used to illustrate past management successes and failures in estuaries.

1.1 Our Vision Statement

Moving forward, Australian estuaries should be able to meet the needs and aspirations of society and ecological integrity in the face of change, while continuing to support integrated human and ecological values. Recognising the co-dependencies between human and non-human systems heightens the opportunity for altering the trajectory of change in Australian estuaries. Essentially, healthy estuarine ecosystems and flourishing human communities are envisaged for the future.

1.2 Our Approach

Our approach to climate change adaptation decision making is based on the premise that in order to adapt to climate change threats (and non-climatic threats too), it is necessary to first assess the values of the estuary in question and the goals that all of the component parts of the estuary have with respect to maintaining or improving their current conditions. Once this process has been undertaken, adaptation strategies can be considered to achieve these goals (and maintain the values). The remainder of this paper is dedicated to articulating (a) the approach to evaluating values and goals, (b) the range of adaptation strategies to choose from to achieve the stated goals, and (c) how to apply (a) and(b) in case study examples.

2. Background

The people of Australia have a history with a coastal focus: *"The experience of most Australians – before and after 1788 – has not been in the outback, the broad plains and the arid interior, but on the coastal fringe"* Coltheart (1997, p. xiii). Around 85 % of the Australian population live in the coastal region (DCC, 2009) and the coasts and estuaries are part of the Australian culture, livelihood and daily life. Three main areas of estuarine planning and management have traditionally surrounded ports and harbours, water quality and environmental flows. Each of these areas has different implications for management and planning when considering estuaries as an entirety.

Following the arrival of the First Fleet, ports and harbours were being built in estuaries along the Australian Coast. Signs of British occupation were public works, in a maritime age the most important of these being port and harbour facilities. It has been suggested that in the early days of British settlement, ships were signs of hope, bringing fresh supplies, news from home and for some convicts, a chance of escape (Coltheart, 1997). Coltheart (1997) documents the early days of exploration of coastal NSW and the discoveries of major river systems and natural harbours, such the Hunter River and Jervis Bay. Permanent entrances were very important for ships, with frustration experienced of the intermittent opening and closing of estuaries such as the Shoalhaven. The rawness of the newly colonised Australia contrasts with the major ports of Australia today and their specialised cargo facitilites, dominating sky and shorelines. The expense of modernising shipping faciltites has resulted in a concentration of works in a few major ports around Australia (Coltheart, 1997). Natural estuarine processes are regularly upset with changes to estuaries through port development, such as the construction of entrance training walls for entrance stabilisation and alterations to the estuarine floor. Within a decade of the arrival of the first container ship in New South Wales, Botany Bay was dredged, creating the deepest port in Australia, for a five-berth container terminal (Coltheart, 1997). An additional example of this alteration is rock being removed from the Newcastle Harbour entrance channel and three places within the harbour to achieve the long sought-after depth of 36 feet.

Water quality, like environmental flows, is not specific to the estuarine environment. Water quality in estuaries can be altered by both the quality of the freshwater entering the estuarine system as well as activities on and around the estuary. The former Department of the Environment, Water, Heritage and the Arts (DEWHA, 2002) estimated that around 80 % of coastal and marine water quality impairment world-wide is caused by broad scale land use activities. Land-based activities can contribute to suspended sediment, nutrients, pathogens, heavy metals and other pollutants to the environment, having significant effects on marine and estuarine water quality (DEWHA, 2002). Australia is committed to protecting Australian water through the development and implementation of the National Water Quality Management Strategy, the key objective of which is: "*to achieve sustainable use of the nation's water resources by protecting and enhancing their quality while maintaining economic and social development"* (DEWHA, 2002, p. 4). Subsequently, the Framework for Marine and Estuarine Water Quality Protection was developed to protect the nation's marine environment from the adverse effects of land-based activities (DEWHA, 2002). The Coastal Catchments Initiative, through the use of Water Quality Improvement Plans, seeks to deliver significant reductions in the discharge of pollutants to coastal and urban quality hotspots (Natural Resource Management Ministerial Council, 2006), improving coastal and estuarine health. Generally, management of coastal and estuarine waters has improved greatly in Australia during the past decade, with some high-profile programs to improve river and estuarine health in several metropolitan areas (Hatton *et al.*, 2011).

Environmental flows are required to maintain the health and biodiversity of estuaries (Peirson *et al.*, 2002). This water is a flow maintained solely for environmental reasons. Upstream catchments supply estuaries with fresh water, which then mixes with salt water entering from the ocean. Division of administrative responsibility regularly results in the management of estuaries being separated from their upstream catchments (Peirson *et al.*, 2002). Subsequently, catchment management authorities have traditionally had little responsibility for management of their downstream estuaries. Environmental flows are essential to minimise the negative influence on estuarine health resulting from changes in flow regime. Peirson *et al.* (2002) present and explain a method of determining appropriate levels of environmental flows in Australian estuaries. However, it was recognised significant knowledge gaps existed in aspects of assessing the environmental flow requirements of estuaries, implementation and management of environmental flows effectively (Peirson *et al.*, 2002; Gippel, 2002). Gippel *et al.* (2009) explore the gaps, constraints and opportunities for development of a standardised and integrated approach towards assessing, implementing and governing environmental flows for estuaries at the national, state and territory levels.

All Australian state capital cities lie on estuaries, and the continued expansion of these centres has the potential for detrimental effects on estuarine health and processes. Indeed, the most recent Australian State of the Environment report highlights that the three main drivers of environmental change: climate change, population growth and economic growth, can, and are, resulting in a range of pressures on the Australian coastal environment (Hatton *et al.*, 2011). With the predicted changing climate, adaptive planning for the future should be at the fore. Planning for sea level rise and other climate change impacts is important, as they begin to affect our populated coastal and estuarine regions.

Traditionally, Australian planning has not considered adaptation strategies. Estuarine planning in Australia predominately falls to the responsibility of the states, with each taking different approaches to management and planning. The majority of the state planning policies and guidelines referring to the coastal regions are centred around the open coast, with estuaries included to varying degrees in different states. Additionally, there have been several attempts to create national unity in estuarine and coastal planning. When considering climate change, planning for climate change and potential climate change risks, again the procedure and process varies from state to state. The following paragraphs discuss some of the different estuarine management strategies implemented in each of the states, and the aforementioned national attempts at unity.

In NSW local Councils are responsible for preparing and implementing detailed management plans for estuaries in their jurisdiction under the NSW Estuary Management Policy. The Estuary Management Manual (EMM) (NSW Government, 1992) was produced to assist local Councils in developing and implementing plans for estuarine management. The EMM discusses climate change and relevant, related issues to NSW. These include the possibility of tropical cyclones penetrating further south along the coast, potential sea level rise and discussions of the impacts of weather and water level changes on the habitat and ecosystems of estuaries. Physical consequences in estuaries suggested include saline intrusion, increased flooding of wetlands and inland migration of estuarine habitats and ecosystems (NSW Government, 1992).

In order to assist the NSW Councils with development of estuarine management plans, grants were available (under the NSW Government’s coastal and estuary management programs) to provide a subsidy of up to 50 % to local government to:

- Prepare (or update) coastal zone management plans and associated technical studies (including estuary health and coastal hazard assessments); and
- Undertake actions to manage the risks associated with coastal hazards and to protect or improve coastal environments and estuary health (OEH, 2012).

In Victoria the State Government, in accordance with the Coastal Management Act 1995, appoints a Victorian Coastal Council. The Council's requirements include state-wide strategic coastal planning, facilitating the operation of Regional Coastal Boards and coordinating the implementation of the Victorian Coastal Strategy and Coastal Action Plans. Three estuarine coastal action plans exist in Victoria for the South West Estuaries, Central West Estuaries and Gippsland Estuaries respectively (Harty, 2002; Western Coastal Board, 2005; Gippsland Coastal Board, 2006).

The National Estuaries Network (NEN), established in 2000, is a mechanism for linking scientists and estuarine decisions makers at a national level. The fundamental aim of the NEN is improving natural resource management and environmental conservation of Australia's estuaries (OzCoasts, 2012). The network is comprised of estuary managers from each Australian state and territory and selected researchers. They meet twice yearly to discuss emerging and pressing estuary issues and learn from estuarine specialists (OzCoasts, 2012). As noted by Peirson *et al.* (2012), there are a broad range of stakeholders and representative professional bodies with interests in estuaries. Communication between these groups could be improved.

In South Australia, the *Living Coast Strategy* outlines the State Government's environmental policy directions for sustainable management of South Australia's coastal, estuarine and marine environments (Natural and Cultural Heritage, 2004). South Australia has also drafted an Estuaries Policy & Action Plan however this, drafted in 2005, remains a draft to date. The vision of the draft Estuaries Policy and Action Plan is – 'healthy estuaries for the benefit of present and future generations' (DEH, 2005). This vision was to be achieved by addressing three underlying problems:

- Poor coordination and integration of management and planning for estuaries;
- Lack of knowledge (both science and management) about South Australia's estuaries; and
- Low level of awareness in the general community of estuaries as important natural environments that need protection.

These problems are not necessary limited to South Australian estuaries, and may be applicable nationwide.

In 2006 the framework and implementation plan for a "National Cooperative Approach to Integrated Coastal Zone Management" was introduced. This framework and implementation plan set the scene for national cooperation in management of the coastal zone, aiming to help achieve ecologically sustainable development (Natural Resource Management Ministerial Council, 2006).

At the state and territorial level, progress on climate change mitigation and adaptation policy and legislation is at different stages of development (Gurran *et al.*, 2011). Gurran *et al.* (2011, p. 22) summarise the state policy and law for climate change adaptation planning relevant to coastal Australia.

Human response to the predicted changes in climate (including and especially migration), and consequential impacts such as land use changes, are also unknown. One approach has been documented by Short *et al.* (2012) relevant to the current major climate change adaptation challenges facing the water supply industry, particularly in an Australian context, highlighting the critical knowledge gaps and strategies required to assist in the formulation of adaptation responses to the range of potential impacts on water infrastructure and future water security. They recommend a modified adaptive management approach integrated into the conventional corporate business cycle of major water utilities.

The following discussion aims to highlight potential change to estuaries under climate change and assist in the formulation of adaptation responses to these.

3. Estuarine Communities

As estuaries provide many uses/values for a range of anthropocentric and ecocentric communities, there are potential goals for each combination of use and community. These goals have been summarised in Table 1 where the anthropocentric communities are aggregated for convenience into Community, Private property and Public infrastructure and the ecocentric communities into Ecosystems. We have not attempted to decompose the ecological communities further but the former can be classified under the typology of communities of place and communities of interest. This separation differentiates the regular users, those whom interact with the estuary on a regular basis, with those, while concerned about the estuary, do not have as regular, personal connection with it. Communities of interest may be limited to users such as regional residents, conservation groups, upstream land users and regulatory bodies. People under the 'communities of place' banner may include the following:

- Recreational and commercial fishermen (including aquaculture);
- Boat users and marina authorities;
- Ports;
- Residents;
- Conservation agencies and managers;
- Tourists and those involved in the tourism industry;
- Indigenous people;
- Property/asset owners and investors;
- Local governments;
- Utilities providers; and
- Farmers.

Communities will not necessarily be the same from estuary to estuary, and it must be remembered that while people can be assigned to 'groups' such as farmers, their individual commercial interests, their ideology, use of the estuary and what they envisage for the future can be different. Successful climate change adaptive planning for estuaries must include interdisciplinary and inter-community heterogeneity in understanding, working cooperatively to create a better future for their estuary as outlined by the goals in Table 1. In the estuarine environment, some goals may be in conflict. For example, it may be impossible to maximise biodiversity and the protection of human infrastructure simultaneously.

Table 1 summaries the uses and non-uses, the different communities and values of estuaries and how the values are reflected in the goals for ecosystems, private and public property and the community. Aspects of Table 1 have been outlined in the case studies in Section 5. While all the different uses and non-uses have not necessarily been directly highlighted in Table 1 in the related discussion, the majority of the uses and non-uses are relevant to at least one, usually more, of the case studies.

Table 1: Community Values and Goals for Estuaries

Uses & Non-uses	Ecosystems	Private property	Public infrastructure	Community
Recreational fishing	Continued abundance of target species Absence of nuisance species	Protection of marinas/boats	Maintain access, boat ramps, car parking	Right to fish (no prohibitions), lifestyle, tourism, leisure opportunity, happiness & wellbeing
Bird watching	High bird diversity, diversity of bird habitats, breeding		Access to bird habitat, boardwalks, bird hides, car parking	Lifestyle, tourism, leisure opportunity
Commercial fishing	Continued abundance of target species Absence of nuisance species	Protection of boats, nets, marinas	Access, Protection and maintenance of wharves, roads	Permission to fish (via social license), Employment
Ports and shipping	Absence of nuisance species	Protection, maintenance and expansion of wharves, cranes, cargo, transport infrastructure, etc	Access, Protection and maintenance of wharves, roads	Employment
Recreational surface water use	Absence of nuisance species Good water quality	Protection of boats, other vessels	Maintain access, boat ramps, roads, car parking	Happiness & wellbeing, tourism, leisure opportunity
Recreational shore use	Absence of nuisance species Good water quality Shoreline stability		Maintain access, roads, car parking, foot paths, playgrounds, bbqs, picnic facilities, public toilets	Happiness & wellbeing
Residential use	Absence of nuisance species Good water quality Shoreline stability	Protection of shoreline properties (houses and gardens), vehicles and other	Boat ramps, roads	Happiness & wellbeing

Uses & Non-uses	Ecosystems	Private property assets	Public infrastructure	Community
Tourism	Abundance of charismatic species Absence of nuisance species Healthy ecosystem Good water quality	Protection of commercial buildings with tourist orientation, accommodation, restaurants, resorts etc.	Maintain access, roads, car parking, foot paths, playgrounds, BBQs, picnic facilities, public toilets	Happiness & wellbeing Employment
Agriculture	Absence of nuisance species Good water quality (low salinity) Shoreline stability	Protection of land, private farm infrastructure and machinery, buildings, e.g. pump stations, fences	Protection of roads and rail	Happiness & wellbeing Employment Lifestyle
Aquaculture	Absence of nuisance species and disease Good water quality Primary productivity	Protection of private aquaculture infrastructure, buildings, e.g. pump stations etc.	Protection of roads, access to boat ramps and shoreline frontage	Employment, Food production
Water supply	Good water quality	Protection and maintenance of water supply infrastructure, access to shoreline	Protection and maintenance of water supply infrastructure, access to shoreline	Settlement resilience
Power generation and supply	Cooling water		Protection and maintenance of power supply infrastructure, access to shoreline	Settlement resilience
Communications			Protection and maintenance of communications infrastructure, e.g. submarine cables, foreshore access	Settlement resilience
Conservation	Healthy ecosystems,			Happiness & wellbeing

Uses & Non-uses	Ecosystems	Private property	Public infrastructure	Community
	Biodiversity, Functional ecosystems, Abundance of target species			
Roads, railways and bridges			Protection and maintenance of transport infrastructure	Transport system resilience
Maritime infrastructure	Absence of nuisance species	Protection and maintenance of maritime infrastructure	Protection and maintenance of maritime infrastructure	Transport system resilience
Existence	Healthy ecosystem Biodiversity Functional ecosystem	Visual access	Protection of heritage structures, e.g. lighthouses, jetties etc.	Happiness & wellbeing
Research	Presence of study objects	Protection of experimental apparatus (e.g. data loggers)	Maintenance of access, roads, boat ramps etc.	Future decision making
Diving	Biodiversity Good water quality (especially clarity)	Protection of boats	Maintenance of access, roads, boat ramps etc.	Lifestyle, Tourism, Leisure opportunity
Sand extraction	Presence of substrate		Maintenance of access, roads, protection of machinery & private infrastructure, buildings	Settlement development, Navigation
Commercial enterprise (place based)		Protection of buildings, vehicles	Maintenance of access, roads,	Employment
Discharge (stormwater, wastewater)	Ability of ecosystem to absorb and decompose/dilute discharge	Protection of residences from pollution	Protection and maintenance of water disposal infrastructure	Settlement resilience, health, Happiness and Wellbeing

4. Adaptation Strategies

Traditional literature on adaptation strategies has been centred around three classes of adaptive management: retreat, accommodate and protect (IPCC, 2001). These traditional classes need to be developed and expanded to include different variations applicable to the SES context. Table 2 displays eight different adaptive management strategies and how they can potentially be applied to the estuarine environment. These adaptive management strategies provide more positive options for communities in the face of climate change.

Abandon relates to completely abandoning the site, this may include removing any preventive infrastructure such as tidal barrages and then relocating to a different location.

Retreat involves relocating to a less threatened position on the same area. In the process of doing this it may be necessary to relocate defensive infrastructure to protect the new location.

The *defend* strategy involves the community remaining in place and imposing different strategies to prevent the climate change impacts from having an effect on the region. Substantial on-ground works are likely to be required when undertaking the defend strategy.

Do nothing involves not acting. However if nothing is done it is likely a threshold point is reached where it is necessary to do something. At this threshold some of the other adaption strategies may not be possible due to the severity of the situation.

Wait and see is another option in which essentially nothing major is done, but a great deal of monitoring may be undertaken. Like in the *do nothing* option, a threshold at which it is necessary to take major action may be reached.

The *accommodate* strategy involves staying in place, but taking the appropriate measures, such as elevating infrastructure, to enable continued comfortable living in the same location.

Improve involves minimising existing stressors through upgrades/retrofitting of structures and more effective management of the coastal, estuarine and catchment environments as a whole.

Hedge involves not ruling out future options. This may include capitalising on existing infrastructure for new functionality and education of the community about the other options that may have to occur in the future.

Acceptance and effective decision making surrounding these different options relies on the way in which the stakeholders are approached and included in the adaptive management process. Different ways in which to manage and include the stakeholders are: facilitate, educate, prohibit, legislate and regulate, change values, manage expectations, research, empower and debate.

Table 2: Climate Change Adaptation Strategies

Strategy	Exemplary On-ground actions	Governance
1. Abandon	Remove barrages	Prohibit further use
2. Retreat	Remove armouring Relocate barrages upstream	Provide recipient areas for relocation Incentivise (push & pull) Legislate Educate Research
3. Defend	Beach nourishment Dredging Species husbanding (e.g. pygmy perch in Coorong) Minimise existing stressors	Educate Research
4. Do nothing – Liberate	Allow nature to take its course – autonomous adaptation	Decide on action during/after extreme events
5. Wait and see	Monitoring?	Educate Research
6. Accommodate	Elevate infrastructure	Manage expectations (i.e. get used to getting wet) Educate Research
7. Improve	Retrofitting hard structures (e.g. fish ladders, introduce rough habitats, minimise shading by structures, reduce floating and vertical surfaces) Species husbanding (e.g. pygmy perch in Coorong) Catchment management Minimise existing stressors	Educate Research
8. Hedge (not ruling out future options – opportunities)	Capitalise on existing infrastructure for new function (e.g. energy generation from barrages, Coorong water pumping for multiple uses) Desalinisation plants	Educate Research Institutional reform – flexible, responsive institutions

5. Case Studies

The case studies included in the following discussion have been selected to provide an overview of a range of different Australian estuarine systems. Each case study, and indeed estuary, is different with the potential for a range of future issues under predicted climate change conditions. Some of the case studies are relatively pristine estuaries, such as the Mary River, NT, while others, the Georges River, NSW for example, are highly modified. Figure 1 presents the locations of the case study estuaries.

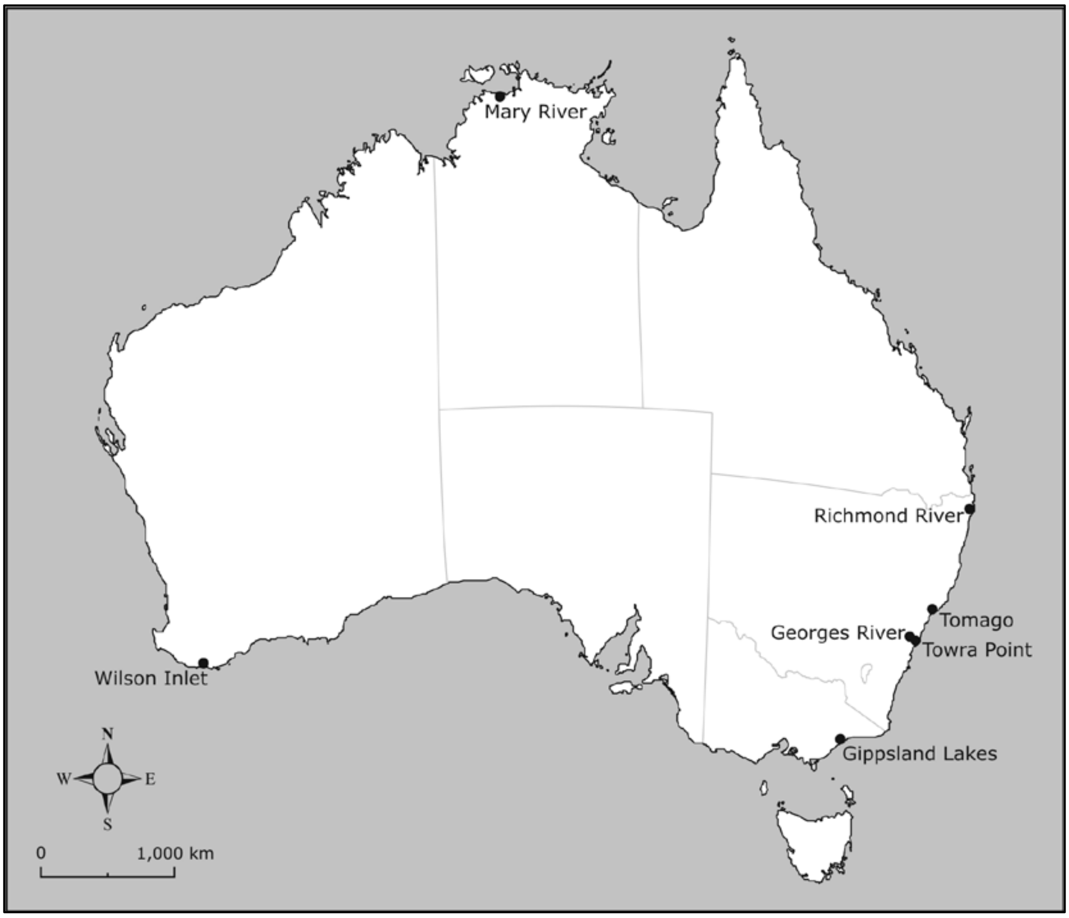

Figure 1: Case Study Locations

5.1 Towra Point – New South Wales

Near the mouth of the Georges River, in Botany Bay, lies Towra Point. Towra Point is a relatively pristine estuarine system in comparison with the highly modified and urbanised Georges River. Towra Peninsula arose due to coastal processes that accreted alluvial sand forming large sand bars about 4,000 – 7,000 years ago. These coalesced and formed the existing peninsula. Geologically, Towra Point consists of deep, unconsolidated alluvial sands. Nutrient rich organic muds and muddy sands have deposited in low energy depositional environments, such as Quibray, Weeney and Towra Bays (NSW NPWS, 2001).

Towra Point is notable for its conservation values. It is not only the most important wetland in the Sydney region but also has national and international significance, particularly for migrating seabirds. In consequence, Towra Point was officially made a nature reserve in 1982 and declared a RAMSAR site (or wetland of international importance) in 1984. In 1987, the Towra Point Aquatic Reserve was created, covering 1400 ha of the surrounding waterways. These reserves also attempt to meet the Federal government's obligations to the China–Australia Migratory Bird Agreement, which came into force in 1988. Together, they form the largest and most diverse estuarine wetland complex remaining in the Sydney region (NSW NPWS, 2013).

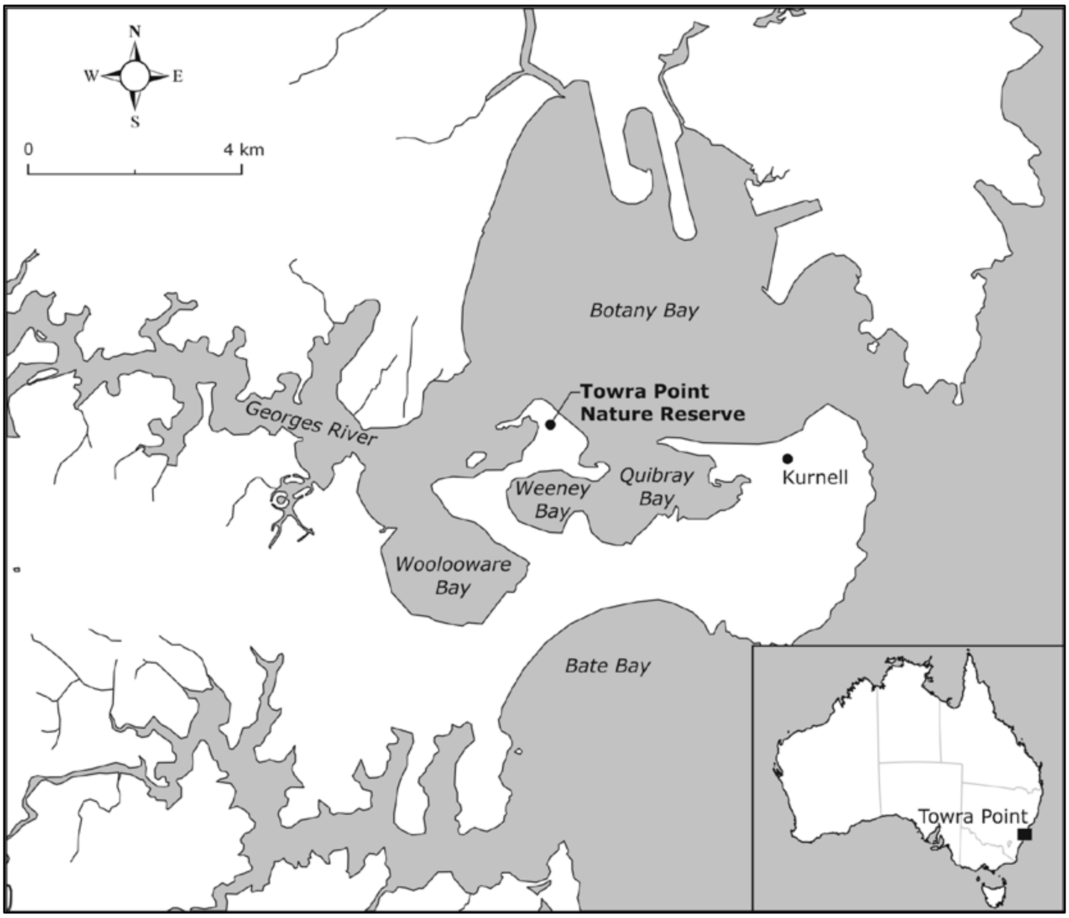

Figure 2: Towra Point

The range of ecological habitats include the freshwater Captain Cook lagoon, clean sandy beaches and most of Botany Bay's seagrasses, mangroves, saltmarshes and tidal mudflats. More than 230 species of fish have been recorded in the reserve which also supports numerous migratory and threatened bird species such as the little tern (NSW NPWS, 2001). A Plan of Management divides the reserve into an outer refuge area and an inner sanctuary area of 500 ha. Recreational fishing is prohibited in the sanctuary zone but permitted within the refuge area. Commercial fishing in Botany Bay ceased in 2002.

The area provides numerous ecological services as listed by the NSW Department of Environment, Climate Change and Water. These include: provisioning services (fisheries production and trophic relay), regulating services (maintenance of hydrological regimes, shoreline stabilisation and storm protection, biological control of pests and disease and pollution control), cultural services (recreation and tourism, science and education, aesthetic amenity, heritage - both Aboriginal and non-Aboriginal), supporting services (hydrological processes, food webs, physical habitat, nutrient cycling, primary production, sediment trapping stabilisation, biodiversity, special ecological, physical or geomorphic features, threatened wetland species, habitats and ecosystems, priority wetland species and ecological connectivity) (DECCW, 2010).

Towra Point became a conservation issue due to wetland losses elsewhere in the Sydney region, anthropogenic losses of local mangroves, oil pollution, and because of erosion at Towra Beach since 1973 (TEL & AMBS, 2003). This erosion (and changed patterns of sand deposition), have been accelerated by changes in wave energy and water movements in Botany Bay due to dredging a shipping channel combined with the hard structures of Sydney Airport runway and Port Botany. In consequence, Towra Beach has narrowed, especially at Towra Point where trees have died (Figure 3) and Captain Cook lagoon has become brackish due to seawater intrusion. Also, sand deposition has covered some sea grass beds. An intensification of these impacts is expected with rising sea levels and altered weather patterns in the future.

Adaptation strategies suggested to address the erosion problem include offshore breakwaters and sand nourishment (Defend). For the present, the Waterways Authority and National Parks and Wildlife Service decided to nourish the beach (Defend). Eroded sand was replaced with sand that had accumulated downdrift within the same littoral drift transport system. A total of about 60,000 m^3 of sand was recycled from downdrift borrow areas and placed along Towra Beach, thereby feeding the process of natural sand transport (Jones *et al.*, 2008). This sand was then shaped and stabilised to form an appropriate beach profile. Thus the beach was raised and extended seaward. The effects of the nourishment on the abundance of intertidal amphipods and the area of seagrasses were studied as part of the project.

Figure 3: Erosion at Towra Point (Photo: AJ)

The issue of erosion is of particular relevance to Towra Point since the present erosive forces remain and these will be enhanced by raised sea levels and greater storm surges expected as a result of climate change. While the current 'Defend' option may be suitable in the short term, either nourishment will have to be repeated at intervals (say 5-10 years) to provide continued protection or else the beach will migrate inland with the loss of the lagoon and wetlands. Retreat is not a valid adaptation option for management of the Towra Point ecosystem, however the site could be abandoned and/or the 'do nothing' approach applied to the system.

5.2 Georges River Estuary – New South Wales

The Georges River estuary, located in Southern Sydney, flows into Botany Bay near Towra Point. It drains an area of approximately 900 km^2, 86 % of Botany Bay's total catchment area (Sydney Water, 1997). The tidal influence extends over 40 km from the mouth of the Georges River, at Botany Bay, to the Liverpool Weir (MHL, 1993).

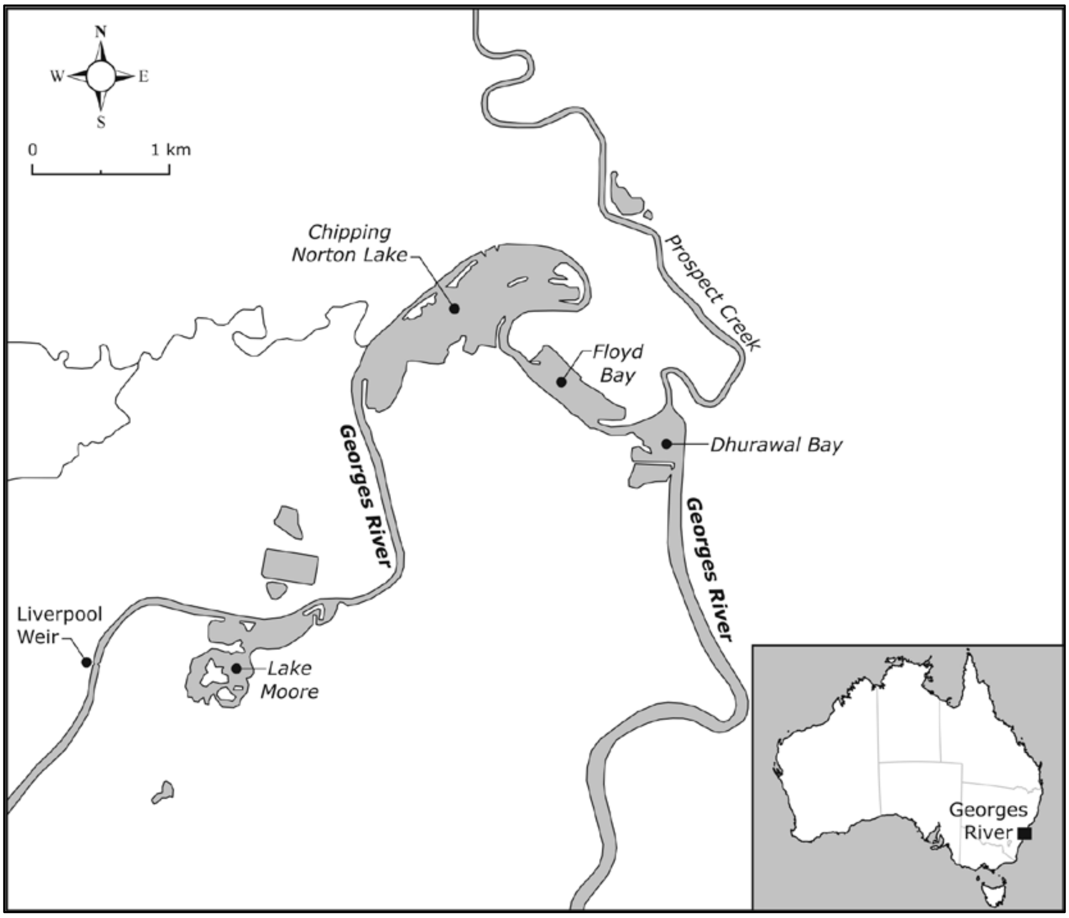

Figure 4: Georges River

The Georges River is a highly modified system, particularly due to widespread dredging of the river to provide sand for Sydney's building and construction industries. The dredging began prior to the Second World War (Warner and Pickup, 1973) and continued at a large scale into the 1950s and 1960s. By the early 1970s the Chipping Norton region posed a major environmental problem. The decision was made to undertake an environmental restoration project, comprising the construction of a series of recreational lakes in the dredged areas. Subsequently the Chipping Norton Lakes Scheme was initiated (PWD, 1990).

The Chipping Norton Lakes Scheme, including the main Chipping Norton Lakes and the off-channel lake at Moorebank, encompasses approximately 3 km^2 (Anderson, 1982). The unnatural shapes of the Lakes Scheme can be seen in Figure 4. The average depth of the lakes is 8 m below low tide (Anderson, 1982). The aim of undertaking an environmental restoration project has been partially fulfilled, with the once destroyed area now providing a recreational haven for the community. However, both past and present studies have demonstrated the lakes scheme has had negative impacts in regions of the river not directly impacted by the original dredge activities.

Munro *et al.* (1967) examined the effect of construction of a proposed lakes scheme at Chipping Norton on the Georges River. Significant erosion of the banks downstream was predicted if the downstream river was not also dredged and artificially widened. It appears that the additional lake at Moorebank was not proposed at that time. The significant increase in tidal prism, including construction of the lake at Moorebank, would be anticipated to lead to the observed increase in bank erosion downstream (Peirson *et al.*, 2001). Insufficient mitigation measures to prevent erosion were undertaken and significant riverbank erosion has occurred between Liverpool and East Hills in the subsequent years (Lawless, 2005; Fullagar, 2007).

Hydraulic modelling has been conducted on the Georges River (Lawless, 2005; Fullagar, 2007), with results confirming the Chipping Norton Lakes have had a major impact on the hydraulic behaviour of the Georges River estuary. Most of the changes are attributed to the increased upstream storage volume resulting from the construction of the lakes and associated increased in tidal flows and velocities. These impacts include:

- Increased tidal range within the estuary;
- Increased erosion potential downstream of Prospect Creek under tidal conditions;
- Increased siltation potential within each of the dredged lake areas;
- Increased erosion potential upstream of Lake Moore, between Lake Moore and Chipping Norton Lake and between Floyd and Dhurawal bays under flood conditions; and
- Reduced flood levels and gradients, particularly upstream of the Prospect Creek junction.

The Georges River provides an example of a past estuarine management failure, highlighting the importance of adoption of the available understanding of estuarine processes, and the potential impacts associated with large scale ecosystem alterations. It can also be used as an example of the uncertainty surrounding estuaries and climate change. Predicted sea level rise, and likely associated increased tidal area in the estuarine system, are likely to lead to additional changes in flushing of the estuary. A consequence of a changing climate is that if tidal inundation changes significantly with rising sea levels, substantial estuary channel erosion may occur. Furthermore, the Georges River floodplain accommodates many of Sydney's residents. The bank erosion associated with the Chipping Norton Lakes Scheme has impinged on infrastructure in the floodplains already, however with sea level rise, and the potential inundation associated, a major problem arises for future management of the estuarine region.

The 'defend' option has been employed already for the Georges River region to combat erosion, with substantial bank works, particularly in vulnerable regions such as surrounding bridges. Another adaptation strategy currently employed in the Georges River region is 'retreat' as a range of houses have been purchased by the local Councils to reduce community risk during flood events. Increased rainfall intensity combined with sea level rise may lead to more frequent flooding of the Georges River, a region already under threat during large scale flood events. The 'improve' option could also be employed in the region, with potential upgrades possible for facilities such as the Liverpool Sewage Treatment Plant to manage water quality better, particularly during high flow events. It may also be possible up 'improve' facilities such as these to provide tertiary treatment of waste and additional potable water for the community.

5.3 Wilson Inlet – Western Australia

Wilson Inlet is a shallow, seasonally open estuary on the south coast of Western Australia. It opens into Ratcliffe Bay. Wilson Inlet is 14 km long and 4 km wide at the widest section, creating a surface area of approximately 48 km^2. The central area is deeper than 3 m and extensive shallows exist around the edges (Ranasinghe and Pattiaratchi, 1999a). The regional towns of Denmark and Mt Barker are within the catchment area, as well as several smaller communities. The region houses a range of different industries, including horticulture, viticulture, dairies, mixed grazing and forestry (WICC, 2011).

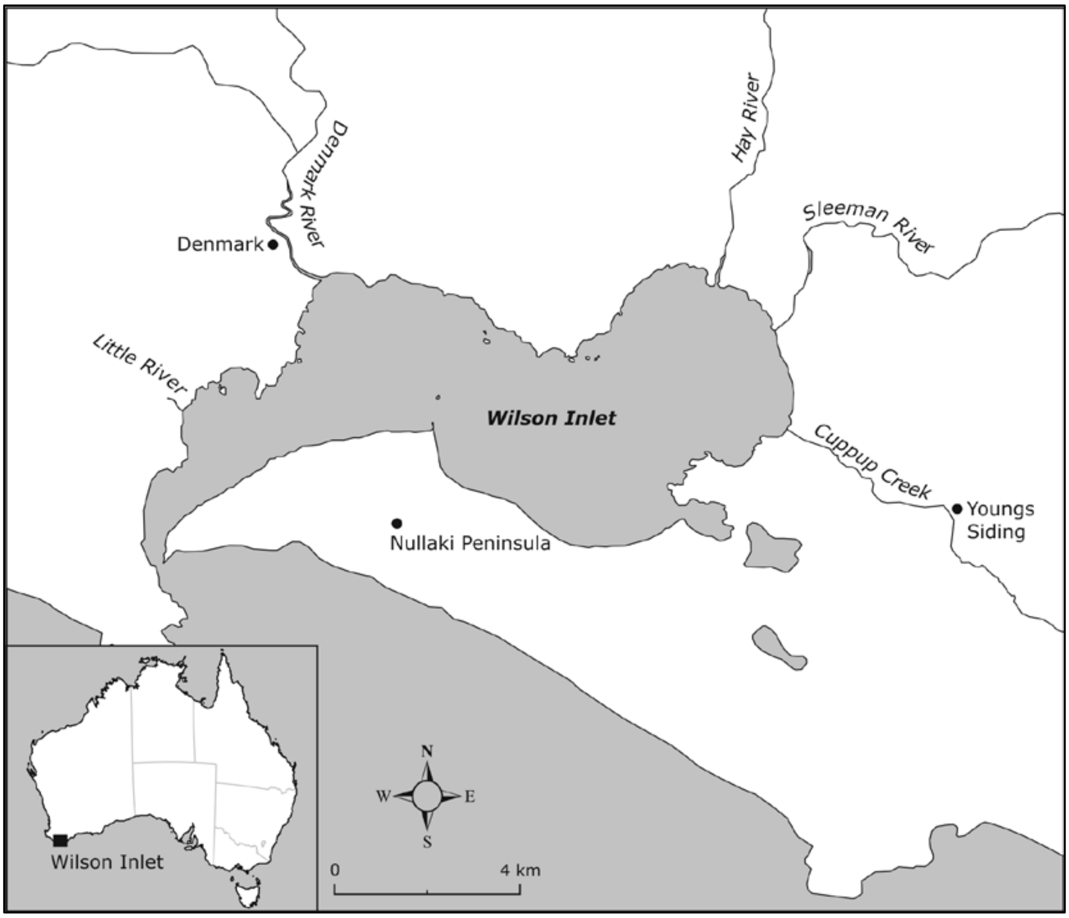

Figure 5: Wilson Inlet

For 6 – 7 months of the year, the entrance to Wilson Inlet is generally blocked by a sand bar. The entrance bar is breached artificially in winter to prevent the flooding of low-lying farmland (Ranasinghe and Pattiaratchi, 1999a). Official records note the estuary has been open each year since 1955, excluding 2007 and 2010 (Stewart, 2011) with anecdotal evidence suggesting the bar breaching practice dates to the early 1900s (Chuwen *et al.*, 2009). The entrance bar was not opened in 2007 and 2010 due to low water levels in the Inlet (Chuwen *et al.*, 2009; Stewart, 2011).

The climate of the region is temperate; cool, wet winters and dry warm summers (Master, 2009). There is evident seasonality, with substantial rainfall during winter, and limited rainfall during spring and summer (Ranasinghe and Pattiaratchi, 1999a). The average annual rainfall in the region is 1120 mm, of which approximately 75 % falls during winter. Average monthly rainfall exceeds average evaporation from May to September (Hodgkin and Clark, 1988). The major rivers discharging into Wilson Inlet; the Hay, Denmark and Sleeman rivers account for 65 %, 25 % and 10 % of the mean annual streamflow respectively (Ranasinghe and Pattiaratchi, 1999a).

Climate change predictions suggest that Western Australia's south-west region is likely to become drier in the next century (Smith *et al.*, 2009). Smith *et al.* (2009) analysed the response of a hydrologic model of the Denmark River catchment to general circulation model downscaled rainfall for three emissions scenarios. It was found that predicted rainfall deductions could result in reductions of mean streamflow over the year of 8 % for a low emissions scenario and 32.5 % for a high emissions scenario. A slight shift in seasonality was observed, however there was no change in peak rainfall month. Reduced rainfall would result in low river flows into the Inlet. Peirson *et al.* (2002) present and discuss a checklist of major ecological processes by which reduced estuary inflows may cause impacts on estuarine ecosystems and the adjacent marine environment, including elevated salinities, diminished flushing frequencies and aggravation of pollution problems. Furthermore, alteration to flow regimes represents an important disturbance influencing the health and sustainability of flow dependant ecosystems (Close, 2005). This is especially true in temperate ecosystems such as estuaries on the south-west coast of Australia, where aquatic ecosystems function in an environment influenced by highly variable rainfall and runoff (Close, 2005).

At Young's Siding, to the west of the Inlet, the mean maximum summer temperature is 24.5°C, while the mean minimum summer temperature is 14°C. The average winter temperatures range between 8°C and 17°C. It has been suggested that by 2030 the average daily temperature could increase by 0.7 to 0.8°C, with increases predicted to continue in the future (Master, 2009). Climate change, and the associated warming of air temperatures, is likely to lead to increases in water temperatures and an increase in nutrient cycling. Environmentally, Wilson Inlet is already of concern, with large inputs of nutrients from agricultural land (Radke *et al.*, 2004). This additional nutrient load can account for considerable increases in the amount of the seagrass (*Ruppia megacarpa*) in Wilson Inlet (Carruthers *et al.*, 1999) and substantial growth of epiphytic algae and phytoplankton (Dudley *et al.*, 2001; Twomey and Thompson, 2001). Wilson Inlet has previously been classed as mesotrophic (Lukatelich *et al.*, 1987) and additional nutrients have the potential to create a eutrophic state. The Wilson Inlet Nutrient Reduction Action Plan was prepared to reduce algae growth in the Inlet through the reduction of excess nutrient inputs form the catchment (WINRAP, 2003). Nutrient blooms are a major problem in many estuaries around Australia, particularly in Western Australia. There is concern in the community around Wilson Inlet that continued deterioration may result in Wilson Inlet beginning to emulate the Peel-Harvey Estuarine system prior to the construction of the Dawesville Channel (Water and Rivers Commission, 2008). The Peel-Harvey estuarine system is known for its large scale eutrophication problems (Rose, 2003). The estuary experienced macroalgal blooms in the Peel Inlet since the late 1960s, and after 1978 toxic blue-green algae blooms occured in the Harvey Estuary and tributary rivers (Rose, 2003).

Furthermore, increased water temperatures in estuarine systems, through temperature increases due to climate change, has the potential to stimuate growth of cyanobacteria blooms. Research has shown optimum growth rates for many types of cyanobacteria are usually at 25°C (Robarts and Zohary, 1987). Median temperatures in Wilson Inlet during summer are usually 22°C, with a range of 18°C to 26°C (Water and Rivers Commission, 2003). Additionally, high freshwater flow conditions can lead to large blooms of cyanobacteria, such as those observed by Robson and Hamilton (2003) in the Swan River, WA, in January 2000.

Chuwen *et al.* (2009) measured salinity, water temperature and dissolved oxygen concentration at the surface, and bottom, of the water column at sites throughout Wilson Inlet for over two years. Mean salinities in deeper waters of the inlet were almost the same at the surface and bed, and salinities in the nearshore waters of the basin did not differ significantly from those at the surface of the water column in the nearby offshore waters. Pronounced seasonal changes were not observed in Wilson Inlet, with the salinity remaining below that of full strength seawater during the monitoring period, between 15 and 26. Mean seasonal dissolved oxygen concentrations in the basins and rivers underwent consistent seasonal changes with maxima occurring during winter and minima during summer. Increasing salinities in nearby, normally closed, estuaries were observed following subsequent years of very low annual rainfall. This could potentially be the case for the future of Wilson Inlet, if reduced rainfall and streamflow results in less regular bar openings. Hoeksema *et al.* (2009) investigated fish fauna in Wilson Inlet between Summer 2006 and Spring 2007. A marked decline in species richness was noted in 2007 when the bar was not artificially breached. This was due, in part, to a decline in the number of marine species. The unopened bar in 2007 meant a lack of access to these estuaries for the new recruits of marine-estuarine species. Again this may be somewhat indicative of the future of Wilson Inlet.

Not opening the sand bar in 2007 and 2010 resulted in an increased length of time and height of water inundation for those years. There has been no long-term monitoring of vegetation in the inlet, however a vegetation survey was conducted in 2011. Stewart (2011) observed that in plots where the water depth, at the time of surveying, was deeper than 0.4 m no native plants were alive, but weed species survived at deeper depths. Increased length of inundation, coupled with potential changed in salinity and likely increased water temperatures in the future may further impact native vegetation growth.

Ranasinghe and Pattiaratchi (1999b) attribute the sediment transport mechanism governing the seasonal closure of the inlet to be a result of persistent swell wave conditions. Storm or high wave energy conditions generally extend the 'open duration' of the inlet. Increased storminess has the potential to result in longer entrance opening times. However, high streamflow events which occur during the closing process result in the inlet being open for longer (Ranasinghe and Pattiaratchi, 1999b). The lower flows predicted for the future may hasten the closure process.

Additionally, Ranasinghe and Pattiaratchi (1998) determined streamflow is the major influence governing the flushing of the estuary, while tidal exchange, wind and entrance channel location have a minimal affect. Under high streamflow conditions sea water does not propagate into the deeper parts of the estuary as the intrusion that forces into the estuary during flood periods is flushed out by the following ebb period. However, when the streamflow is less than the mean annual streamflow seawater propagates into the deeper parts of the estuary and is not flushed out by the ebbing tides. Reduced streamflow may result in more tidal influence and increased salinity in Wilson Inlet during 'open' conditions.

It is evident that Wilson Inlet is a dynamic system that is vulnerable to climate change, in particular, predicted reduced rainfall and subsequent decreased streamflow. This will lead to a more eutrophic environment. Furthermore, the predicted reduced streamflow is likely to lead to fewer instances of artificial bar breaching, as the requirements are not met. The some impacts of which have been already observed in 2007 with a reduction in fish species when the bar was not breached. Moreover, changes suggest that with reduced streamflow the entrance may not remain open as long. As discussed, hydrological and temperature changes may also result in increased cyanobacterial blooms and changes in estuarine salinity. The future of Wilson Inlet is heavily reliant on streamflow and the opening of the bar system.

A range of estuarine communities discussed in Table 1 are present within the Wilson Inlet system. Commercial practices in the Inlet pertain to fishing, aquaculture and tourism. Both commercial and recreational fishing are permitted within the Inlet, since the early 1900s professional fisherman have fished in the Wilson Inlet (WICC, 2012). In 2010-11, fish caught commercially in Wilson Inlet were a major part of commercial fishing in the South Coast Bioregion (Fisheries Research Division, 2011). Wilson Inlet has historically produced the vast majority of South Coast Bioregion landings of cobbler, and in 2010-11 was also the key contributing estuary in the South Coast Bioregion to commercial landings of sea mullet (Fisheries

Research Division, 2011). Currently there are 25 fishing licences giving access to Wilson Inlet for commercial fishing purposes (WICC, 2012). Aquaculture is also present in the Inlet, albeit a smaller commercial sector. Currently there are two licences held for growing and farming mussels and oysters (WICC, 2012). These enterprises require a significant influx of sea water for the maturation and spawning of the mussels and oysters.

Tourism is a thriving industry in Denmark with peak periods occurring during Easter and Christmas holidays. It is estimated that 114,000 tourists visit the area annually, spending approximately $40 million p.a. (Wilson Inlet Review Steering Group, 2009). Wilson Inlet is central to the tourism industry in Denmark and also contributes to regional tourism. Two caravan parks and other holiday accommodation are located on or adjacent to the Inlet foreshore and tourism operators offer tours that include Wilson Inlet and/or its foreshore, taking in the scenery, fauna and flora (Green Skills, 2008). Visitors also bring and hire canoes, kayaks, boats and other craft, using both the Inlet and the lower sections of the Denmark and Hay Rivers. Maintaining the ecosystem health of the Inlet is essential to keeping not only the local tourism industry viable but also maintaining healthy fish stocks and other aquatic fauna in the Inlet.

A range of adaptation strategies can be applied to the Inlet to manage different communities in different manners. Abandon may be an option for some low lying places with predicted sea level rise, but establishment of a permanent entrance may provide flood relief for these areas (Defend). Retreat may be an option for some farming communities, businesses and residences. Currently the threshold for opening the entrance bar is water level, however this could be changed to allow for management of the bar, water level and water quality in a manner that may potentially provide a greater range of benefits for the communities.

5.4 The Mary River – Northern Territory

The Mary River is located 300 km east of Darwin, with a catchment area of approximately 7,700 km^2. The catchment experiences high rainfall over the wet season (generally November to March) and very low rainfall over the remainder of the year (Wyllie *et al.*, 1997). This seasonality creates a temporal and spatial mosaic of habitats that are important for wildlife, abundant in the Mary River wetlands and floodplain. They are home to breeding populations of many waterbirds, and house key barramundi fishing spots (Woodroffe and Mulrennan, 1993). The area borders Kakadu National Park, offering many tourism opportunities (Sterling, 1992). Tourism in the Mary River, including recreational fishing and hunting, has been estimated to exceed $AU 2 million (Ypma and Zylstra, 2006).

Saltwater intrusion on the Mary River is a major problem (Sterling, 1992). The Lower Mary only 50 years ago was a large coastal freshwater wetland with minimal channel formation but is now a meso tidal estuary (Williams, 2010). The two major tidal creeks (Sampan and Tommycut Creeks) have expanded since the 1940s to accommodate larger tidal flows, as well as the gradual extension of tidal influence along prior and existing channels, the expansion of small tidal channels and the formation of new channels into freshwater areas (Woodroffe and Mulrennan, 1993). Surveys by Williams (1996) show the channel dimensions have increased by an order of magnitude, as well have the tidal amplitudes. At a point 10 kilometres upstream of the mouth the channel width has increased from 25 metres wide and 2.5 metres deep with tidal amplitude of 0.1 – 0.3 metres to 100 metres wide, 10 metres deep and tidal amplitude of 3.5 metres. The widening and deepening of the channels has resulted in the wetland being more efficiently drained during wet season flow events (Williams, 2010).

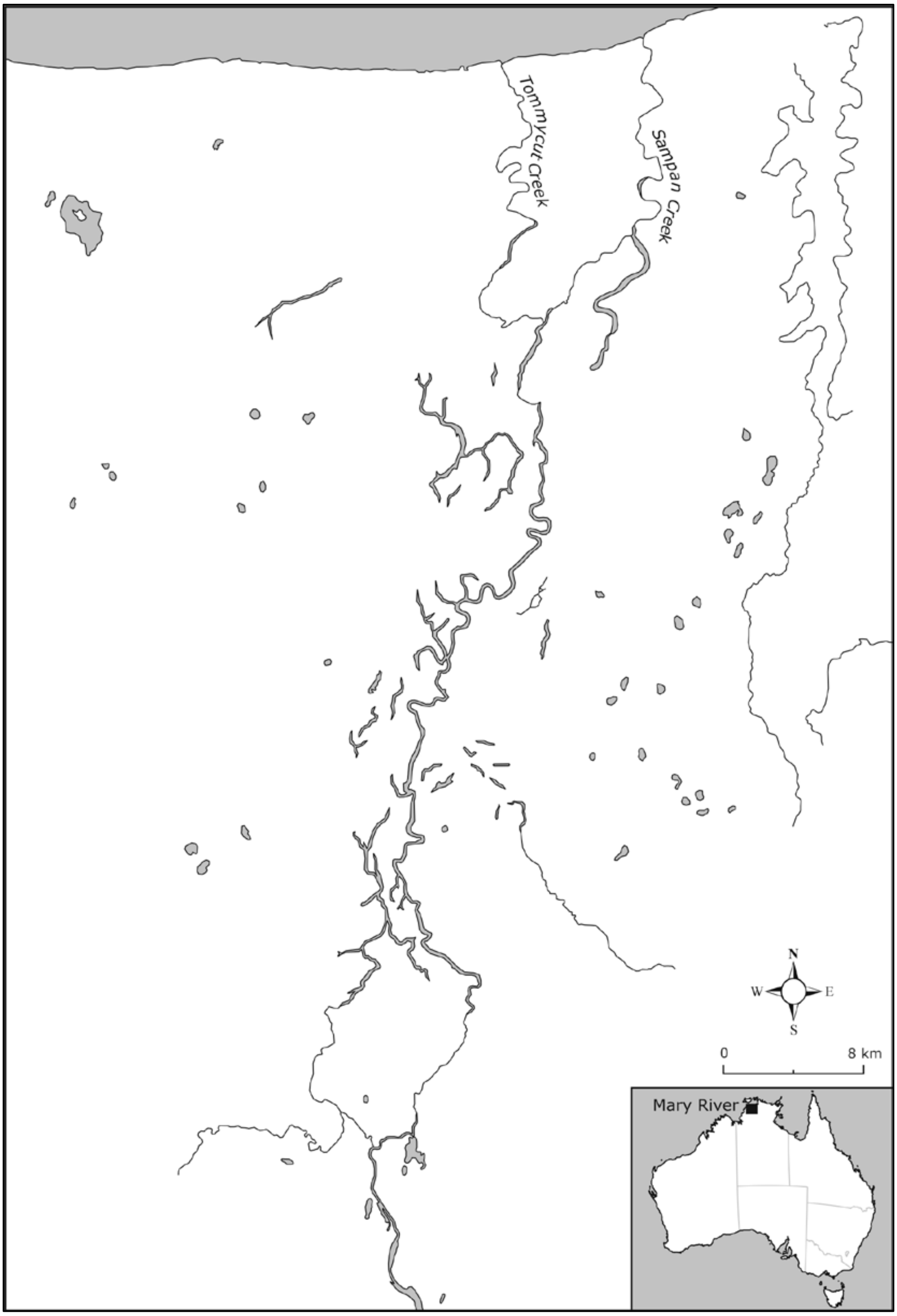

Figure 6: Mary River

The penetration of saltwater into the plains has stressed the native vegetation leading to: extensive dieback, the destruction of freshwater communities in swamps and billabongs, filling of billabongs with tidal sediments, tidal flooding and accretion of sediment on the floodplains adjacent to the tidal channels (Finlayson *et al.*, 1988). In 1990 it was estimated that over 19 % of the total floodplain and wetland area had been destroyed by saltwater intrusion, with an additional 35 – 40 % immediately threatened (Applegate, 1990). Bach and Hosking (2002) report on a remote sensing and ground data collection scheme in the Mary River. Several sites studied on the river are successfully undergoing regeneration following the construction of barrages undertaken since 1988 in attempts to limit saltwater intrusion (Defend).

Maximum elevations of the coastal floodplains of the Mary River are less than five metres. The floodplain elevations are commonly close to spring high tide levels, approximately 3 metres above mean sea level. Large areas of the coastal plains are at elevations below this (Woodroffe and Mulrennan, 1993). Many of the remote backwater plains lie at, or below, the elevation reached by the highest tides, however they are protected from tidal inundation by the slightly higher elevation of levee-like features adjacent to the river channels (Knighton *et al.*, 1991).

Despite extensive research conducted on floodplains of the Mary River, no single explanation has been identified to account for the extension of the tidal influences and salt water intrusion over the past 50 years (Cobb *et al.*, 2007). Woodroffe and Mulrennan (1993) highlight the Lower Mary River plains are particularly prone to saltwater intrusion due to a combination of the large tidal range, the small elevation differences across the plains, and the existence and distribution of paleochannels. The distinct palaeochannels recognisable within the Mary River region are remnant tidal channels that were active during the mid-Holocene. They have since been partially or completely in-filled by the deposition of tidal mud and sediments (Woodroffe *et al.*, 1986; Woodroffe and Mulrennan, 1993).

Six possible explanations for the large scale saline intrusion observed in the Mary River are discussed by Woodroffe and Mulrennan (1993): sea level change, rainfall variability, direct human effects, buffalo impact, consolidation and compaction of the plains, and a natural cycle of change. While these authors claim that the changes cannot be linked to a specific cause, these changes have revealed the vulnerability of the extensive areas of these, and similar, floodplains. In 1995 regional property owners, with the support of the Northern Territory Government, attempted to close the entrance to Tommycut Creek in an endeavour to reduce the salt water inundation in the area (Wyllie *et al.*, 1997). This attempt was not successful.

Cobb *et al.* (2007) suggest the rapidity with which the networks of tidal creeks has expanded and intensified during the past 50 years on the Mary River is indicative of either a trigger mechanism, moving the floodplain system towards a new morphological state, or short-term fluctuations in atmospheric, fluvial and oceanographic processes. This highlights the uncertainty surrounding what may happen to the Mary River, and other estuarine systems, with predicted sea level rise and climate change in the future.

Furthermore, the possibility of increased storminess and cyclonic activity in the region has the potential to have major impacts. High flow events impacting on an already vulnerable region have the potential for substantial channel erosion and scour. Additionally, despite the large tidal range at the Mary River, changes in sea level and associated changes in tidal prism have the potential to increase this scour and further enhance saline intrusion on the river.

At the Mary River, a large scale ecosystem and tourism, are the main communities under threat. Previous attempts at blocking off the river mouth (defend) have not succeeded and other adaptation strategies need to be introduced to the region. As the Mary River is such a large region it may become necessary to protect specific areas with high conservation and/or tourism values, essentially employing ‘defend’ strategies at some locations and potentially ‘do nothing’ at others. Adaptation strategies such as 'Wait and See' may need to be employed with caution to such large scale systems as irreversible thresholds may be reached that trigger entirely new physical and biological states.

5.5 Gippsland Lakes – Victoria

The Gippsland Lakes are a series of large, coastal lakes in Eastern Victoria. The system is approximately 69 km in length, and 10 km wide at the widest point (Webster *et al.*, 2001). The Lakes are connected to the ocean by a narrow, man-made channel at the eastern end (Lakes Entrance), constructed in 1889. Prior to the construction of the navigation channel the Lakes were most likely usually fresh, but would have experienced salinity when the barrier was intermittently breached (Collett, 1987).

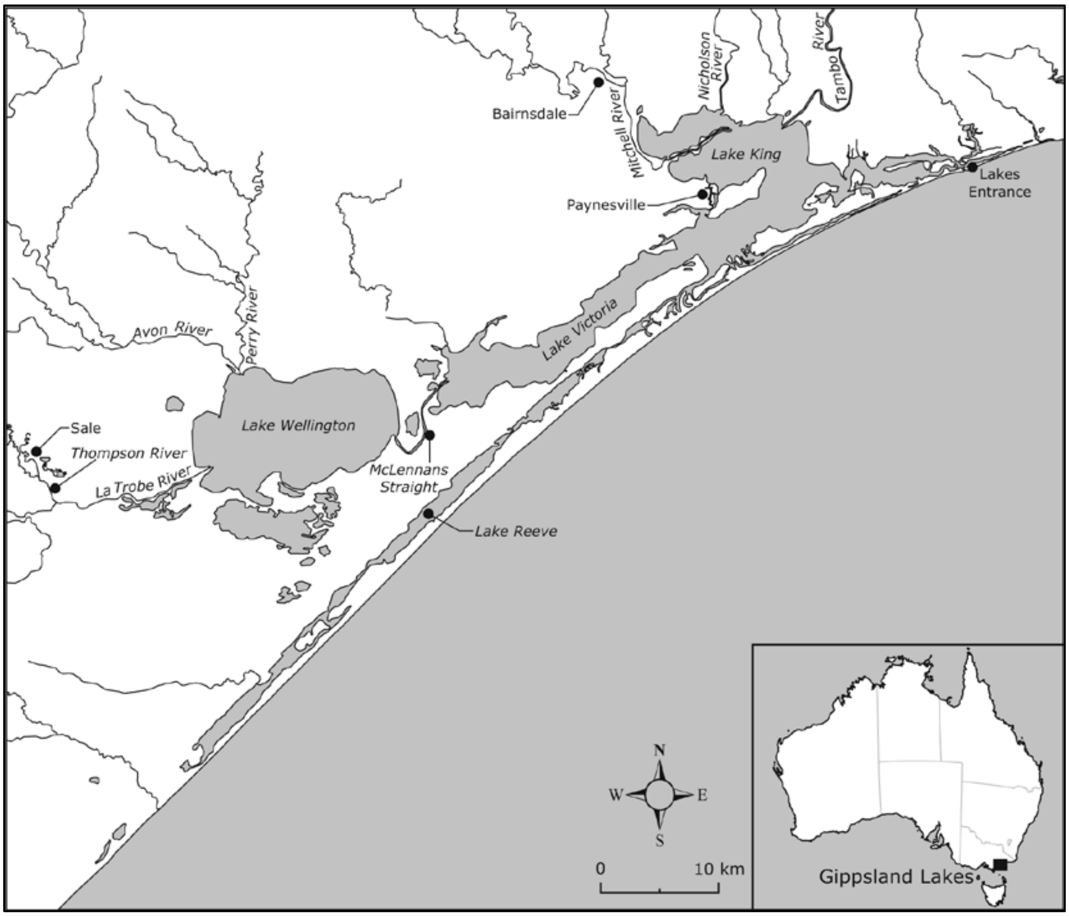

Figure 7: Gippsland Lakes

The Gippsland Lakes system has an area of 364 km^2, with Lake Wellington, Lake Victoria and Lake King being the three main water bodies. Lake Reeve, along the coastal barrier, is usually dry, except during times of high rainfall (Webster *et al.*, 2001). Lakes Victoria and King, in the east, are more closely connected to the ocean than Lake Wellington. The eastern lakes have mean depths of 4.8 m and 5.4 m respectively and often stratify in salinity (Webster *et al.*, 2001). Lake Wellington, in the west, is connected to Lake Victoria by McLennans Strait, a 9.7 km long channel. The restricted connection between Lake Wellington and the other lakes causes Lake Wellington to be less saline than the others. Lake Wellington is a flat bottomed basin of a uniform shallow depth (mean depth 2.6 m) (Webster *et al.*, 2001).

Five major rivers (the La Trobe, Avon, Mitchell, Nicholson and Tambo Rivers) flow into the Gippsland Lakes, draining an area of 20,600 km^2 (Webster *et al.*, 2001). The majority of flow is delivered by the La Trobe River (43 % of mean annual flows) and Mitchell River (36 %) while the Tambo (11 %), the Avon (8 %) and the Nicholson Rivers (2 %) comprise the remaining flows (Ecos, 2008b). Significant seasonal trends in stream flow occur, with higher flows in winter and spring and lower flows present during summer months. The eastern rivers are characterised by relatively steep slopes and are forested in their upper catchment, whilst the western catchments have a higher proportion of farming, industry and urban development in the riverine corridor (Ecos, 2008b). Land use in the Gippsland Lakes catchment has significantly changed since colonisation. Extensive areas have been cleared for pasture, combined with mining in the upper catchments, have affected the water quality in the rivers. These land use changes, combined with dam construction, have also resulted in alteration of the hydrological regime of the waterways (Ecos, 2008b).

Approximately 590 km^2 of the Gippsland Lakes is incorporated within the Gippsland Lakes RAMAR Site. Three wetland types, under the RAMAR Convention classifications, are present in this region: coastal brackish/saline lagoons, permanent saline/brackish pools, and permanent freshwater marshes (DSE, 2003). The Gippsland Lakes RAMAR site is characterised by its diverse and large waterbird populations, the most important species of which are strongly freshwater dependent. (Ecos, 2008a) identified 48 species of waterbird as notable or important. Additionally, it was recommended that seven of these were considered as significant in supporting the Gippsland Lakes listing under the RAMAR Convention on Wetlands of International Significance.

The RAMSAR criteria fulfilled by the Gippsland Lakes region when listed in 1982 were:

- It is a particularly good representative of natural or near-natural wetland characteristic of the appropriate biogeographical region;
- It regularly supports 20,000 waterbirds;
- It regularly supports substantial numbers from particular groups of waterfowl, indicative of wetland values, productivity or diversity; and
- It regularly supports 1 % of individuals in a population of one species or subspecies of waterfowl (DSE, 2003).

The eastern lakes are important breeding and nursery grounds for some species of fish, such as the commercially important Black Bream and the rare Australian Bass. These species breed in the lakes or the inflowing rivers whenever suitable temperatures and salinities occur (DSE, 1999). Many other species of fish also make use of the lakes to breed, grow or feed. Permanent changes in the conditions of the estuarine waters, in particular the temperature and salinity, may result in the relation of breeding and nursery grounds for some fish species.

Additionally, there have been regular algal blooms in the Lakes in recent years. Major blooms have substantial effects on the biological systems of the lakes and on the tourism industry and local economy generally (DSE, 1999). The main lakes of the Gippsland Lakes are highly sensitive to eutrophication due to several factors, including:

- They are shallow, subsequently loads per unit area translate into high loads per unit volume of water;
- They experience episodic periods of very high nutrient loads from the modified catchment, large enough to markedly increase nutrient concentrations in the water column;

- The water column stratifies vertically in some of the lagoons due to the differences in salt concentrations; and
- Submerged macrophytes may compete with the algae for nutrients cover little of the sediment area (Webster *et al.*, 2001).

A major impact of predicted sea level rise in the Gippsland Lakes will be the increased frequency and duration of inundation of fringing swamps and wetlands. Some low lying areas are likely to become permanently inundated. Additionally, there is the possibly of increased frequency of dune breaches. Multiple breaches of the barrier dunes, and permanent flooding of much of the low lying areas, has the potential to turn the lakes system into a shallow marine embayment with saline wetlands and saltmarsh behind a sandy beach (Ecos, 2008b). Changes such as these would be accompanied by profound changes in the biota of the lakes system. It is possible that an altered system would no longer retain the required attributes for listing as a wetland of International Significance under the RAMSAR Convention.

Several problems maybe faced by stakeholders in the Gippsland Lakes region, both now and into the future, with predicted climate change and associated sea level rise. Ecos (2008b) predict potentially catastrophic decreases in flow, with “climate change expected to *more than halve* the median annual inflows to the lakes compared with the ‘natural’ flow regime (53 % reduction, 1520 GL per year)” (Ecos, 2008b, p.243).

Anecdotal evidence suggests that currently, periods of low fresher inflows result in increased water salinities in the main lakes (Ecos, 2008b) leading a gradual transition to a more marine environment in the Gippsland Lakes. Reduced rainfall is predicted for the region in the future and the associated lower riverflows may lead to increased salinities in the Lakes. The effects of the increasing intrusion of marine salinity into the Gippsland Lakes are likely to include; depletion of shoreline vegetation, increased stress from wind-borne salt on vegetation near to the shoreline and above water level, wetland habitat degradation and loss through vegetation change, loss of breeding habitat for fish, restriction of the available habitat for a number of bird species and formation of 'halocline stratification' producing a layer of deoxygenated water at the bottom of the lakes (DSE, 1999).

Climate change decreased rainfall and corresponding riverflows, combined with saline intrusion and the possibility of increased evaporation are predicted to result in the region experiencing much more saline, possibly hypersaline conditions (Ecos, 2008b). Decreased flow and saline intrusion are likely to become a major issue for many Australian river and estuarine systems. The Gippsland Region, is region of high agriculture production, and requires water for irrigation and survival of the industries. Decreased rainfall in these regions is likely to lead to an increase in the water required by the farmers, resulting in even less reaching the lakes system. Furthermore, reduced flows and associated saline intrusion are likely to have a major impact on the flora and fauna of the region, with associated affects for many stakeholders and potentially, changes in the underlying factors that resulted in the inclusion of the site as a RAMSAR wetland.

The Gippsland Lakes are a large, vulnerable system that encompass the majority of communities outlined in Table 1. Stressors on the Gippsland Lakes system are numerous and complex. The complexity of this system means that it is likely that several of the adaptation strategies outlined in Table 2 are needed to reduce community vulnerability to climate change effects.

5.6 Tomago – New South Wales

Many coastal wetlands have been modified by industrial and urban development (Williams *et al.*, 2000), with urban development occupying over 25 % of the NSW coastline (Beeton *et al.*, 2006). The impact of urban development is particularly evident in the southern states, with 17 % of mangroves and 21 % of saltmarshes in NSW and Victoria destroyed by coastal development (Turner *et al.*, 2004). Pressey and Middleton (1982) investigated the impacts of flood mitigation works on coastal wetlands in NSW and estimated that approximately 38 % (40 000 ha) of coastal wetlands had been destroyed. It was estimated, in 1995, that 60 % of coastal wetlands in NSW were degraded or destroyed (Bowen *et al.*, 1995). Flood mitigation works, dredging and river channelisation works have been responsible for a 41 % decrease in the area of saltmarsh within the Hunter Estuary Wetlands since the RAMSAR listing of the site in 1984 (Hydro Tasmania Consulting, 2010).

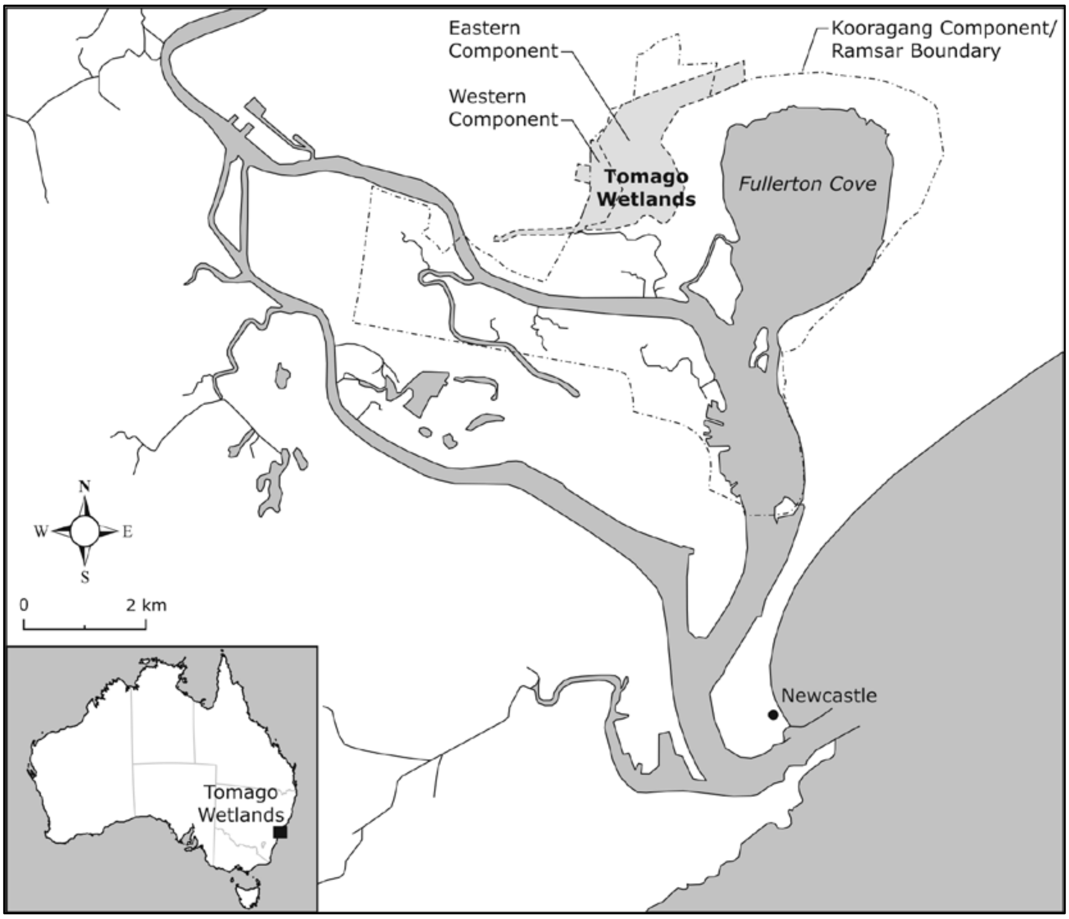

Figure 8: Tomago and the Lower Hunter Estuary

The Hunter Estuary Wetlands RAMSAR site is located in the Lower Hunter River estuary, on the central coast of New South Wales (NSW). The site is comprised of two components, the Shortland Wetlands and the Kooragang Wetlands. The Kooragang Wetland component includes parts of Kooragang Island, the bed of Fullerton Cove and the eastern section of Tomago Wetland. Tomago Wetland lies immediately to the west of Fullerton Cove. Works to restore tidal flushing and restore this drained wetland within the Kooragang Nature Reserve are presently underway (Glamore *et al.*, 2005).

Extensive works have been conducted in the vicinity of the Hunter Estuary RAMSAR site. Between 1913 and 1928 a levee and internal drainage system was constructed around Fullerton Cove, including an internal ring drain following the levee bank. The Lower Hunter Valley Flood Mitigation Scheme began in 1956. The aims of the scheme were to:

- Reduce the frequency of flooding;
- Reduce the time floodwaters lie on land after the flood has passed; and
- Control the direction and velocity of floodwaters to reduce damage to farmlands and property.

By 1980, the Public Works Department had completed 160 km of levees and spillways, 175 floodgates, 111 km of flood canals, 14 km of bank protection works and 40 km of control and diversion banks (PWD, 1980). These works covered the majority of the Hunter River between Maitland and Hexham, as well as the Williams River downstream of Seaham. A levee bank also extends from the Tomago Wetland to the opposite side of Fullerton Cove. The levee bank, ring drain and other drains within the Tomago Wetland were enlarged by the Public Works Department between 1968 and 1980 (MacDonald *et al.*, 1997). These engineering works, including the installation of floodgates at the tidal boundary, ensured that tidal waters were excluded from wetland (i.e. the site drains via one-way tidal floodgates). The main objective of the levee and culverts was to provide a flood detention basin to offset flooding in the Hunter River (the ring levee overtops during a 2 % Annual Exceedance Probability). During non-flood periods agriculture was promoted on the site.

The drainage and exclusion of tidal waters within Tomago Wetland degraded the salt marsh ecosystem, fostering the growth of other species. Winning (1996) demonstrated that the previously dominant salt marsh species had been replaced with saline pasture species. Lowering of the groundwater table also oxidised sub-surface soils causing soil acidification and poor water quality. Grazing and other uses of the site further degraded the ecosystem and reduced the migratory wading bird habitat.

The Kooragang Nature Reserve (more recently named the Hunter Wetland National Park), including the Tomago Wetland, was gazetted in 1983 and is under the management of the NSW National Parks and Wildlife Service. The Kooragang Wetland Rehabilitation Project was formed in 1993 to rehabilitate the coastal wetlands of the Hunter estuary. As part of this project, tidal restoration works were proposed at Tomago Wetland.

The restoration of tidal exchange to the western section of Tomago Wetland was largely designed as compensatory habitat for migratory wading birds and, as such, numerical modelling was undertaken to ensure that the correct hydraulic regime would be reinstated (Glamore *et al.*, 2005). Furthermore, Rayner and Glamore (2010) determined the impact of restoring tidal exchange at the eastern floodgates of Tomago Wetland. Using two-dimensional (2-D) numerical modelling hydrodynamic tools were used to simulate the reintroduction of tidal exchange at the site and to determine the optimal configuration of the on-ground structures.

It has been shown that salt marsh habitat can be fostered if the tidal inundation depth is limited to 0.3 m (Howe, 2008; Howe *et al.*, 2010). As such, the on-ground works at the Tomago site have been designed to limit the extent of tidal inundation to this level. Restoration of the additional sections of Tomago Wetland need to be designed to ensure that tidal inundation improves hydrologic conditions for the existing salt marsh habitat while ensuring that upland stakeholders are not negatively impacted (Rayner and Glamore, 2010).

The on-ground works were designed to ensure that initially only the western portion of Tomago Wetland was restored. Following several years of preparatory works, the main set of western floodgates were modified in August 2007 to permit tidal exchange. The modified gates allow water to enter the wetlands until a predetermined water depth is reached. The total restored salt marsh habitat in the western section (Stage 1) is approximately 2.5 km^2. The eastern and western components of Tomago Wetland are connected via a main ring drain extending along the southern and eastern boundary of the site. A flap gate restricting exchange between either side of the site was installed on the ring drain as part of the 2007 on-ground works. The eastern component of the wetlands was further divided into two (Stage 2 and Stage 3). Stage 2 (adjacent to the western, Stage 1, component) was also fitted with modified flood gates and opened to tidal flushing in November 2011. The far eastern component of Tomago Wetland remains in an unrestored state, however planning has been undertaken to allow tidal flushing in this area.

Other on-ground works to support the restoration included:

- Construction of a 1.8 km levee across the upstream boundary;
- Installation of floodgate flaps and culverts to direct the tidal water;
- Clearing of exotic and undesired species; and
- Installation of floating booms to minimise mangrove colonisation of the restored floodplain (Rayner and Glamore, 2010).

The floating booms, coupled with hand removal of mangrove seedlings on site, are particularly important to establishing the salt marsh ecosystem. The expansion of mangrove habitat in estuaries in southeast Australia since colonisation is a well-established trend (Saintilan and Williams, 1999; McLoughlin, 2000; Harty, 2004). Corresponding saltmarsh decline throughout southeast Australia is well documented, with most estuaries losing over 25% of their saltmarsh in the past five decades and as much as 80% in some estuaries (Saintilan and Williams, 1999). Some of the decline in saltmarsh can been attributed to land reclamation, however the trend of mangrove encroachment into saltmarsh has been the predominant cause of saltmarsh decline (Saintilan and Williams, 2000).

Sea level rise poses a major threat to the Tomago site. As it currently stands, the rehabilitated sites will be protected until overtopping of the levee banks occurs, at which time the site will become regularly inundated and optimum water levels for saltmarsh growth will be exceeded. This highlights the need for the consideration of climate change when implementing rehabilitation schemes and planning for future change.

Recent modelling of the Hunter Estuary, based on current rates of sea level rise and the intertidal elevation currently supporting mangroves and saltmarsh, predicted an increase of future areas within the elevation range suitable to support mangrove and saltmarsh communities (Rogers *et al.*, 2012). On the basis of these results, Rogers *et al.* (2012) suggest planning for sea-level rise should be directed towards facilitating wetland adaptation by promoting tidal exchange to mangrove and saltmarsh, and providing land for wetland migration. However, these suggestions do not take into consideration the other communities in the estuarine region, such as the agricultural industry, that may be affected by natural wetland migration. Potential conflicts such as these highlight the importance of integrated future planning for estuarine ecosystems and communities.

The adaptation strategy currently in place at the Tomago Wetland is 'improve'. Works have been undertaken that not only protect the area from sea-level rise (as normal floodgates would), but enhance the ecosystem, by controlling the hydrological regime, encouraging salt marsh, not mangrove, growth. Future adaptation options for the Tomago Wetland would involve additional ‘improvements’, building on the work already undertaken to further assist inland wetland migration as hydrological conditions change.

5.7 The Richmond River Estuary – New South Wales

The Richmond River is a major coastal river system in northern NSW, with a catchment area of approximately 6,850 km^2. The Richmond River has a large floodplain (approximately 1,000 km^2) relative to catchment area and a small water surface area of 19 km^2 (WBM, 2006). The estuarine region of the Richmond River is comprised of three main arms: Richmond River (main), Bungawalbin Creek and Wilsons River.

70 % of the land within the estuary has been cleared and the remnants of native vegetation are generally restricted to steep slopes or heathlands. Mapping of the estuary conducted in 1996 determined the following land use percentages (ABER, 2011):

- Grazing or grasslands - 54 %
- Forested lands – 26 %
- Cropping – 11.2 %
- Water bodies 5.3 %
- Urban – 1.8 %

The Richmond River floodplain has been extensively drained via a network of drainage channels and floodgates. 391 floodgates comprise the Richmond River Community Flood Mitigation System as well as additional private gates (ABER, 2011). The majority of the cleared and drained lands are used for cattle grazing or sugar cane production. Much of the lower estuary, including the entrance, has been rock lined to stabilise channels and maintain navigation (ABER, 2011).

The Richmond River Catchment Stream Health Assessment Report, partially reported by (Bird, 1997), presented conclusions of findings for riparian vegetation, at the regional level, ranged from the highest to lowest grades (‘very good’ to ‘very poor’). Aquatic vegetation was classed as ‘poor’ to ‘very poor’ (Dalby-Ball *et al.*, 1999). The Bungawalbin arm of the estuary has been found to be in the best condition (Bishop, 1999) with good aquatic vegetation for fish (Day, 1994). Very limited field observations of the lower main channel of the estuary indicate that mangroves dominate the edge vegetation to approximately 35 km upstream where *Phragmites australis* becomes dominant (Dalby-Ball *et al.*, 1999).

Saline intrusion has been investigated in detail in the Richmond River. During periods of low freshwater inflow to an estuary, saline waters enter from the ocean through the estuary mouth. These waters enter as density currents, or as a result of tidal mixing. During periods of high freshwater inflow from the catchment, salt water is flushed from the estuary. The hydrological nature of the Australian climate means that significantly different saline structures can be observed in any given estuary depending on the antecedent rainfall (Peirson *et al.*, 1999). Large pools of freshwater can remain in the upstream reaches of an estuary, such as the Richmond River, during long periods of low freshwater inflow. These have been termed ‘tidal pools’.

Substantial changes to aquatic vegetation and fauna have been recorded in estuaries due to salt incursion (e.g. Holm and Sasser, 2001; Bornman *et al.*, 2002; Alexander and Dunton, 2002). Gillanders and Kingsford (2002) indicated that salinity changes may alter the distribution and abundance of saltmarsh, mangroves and seagrass species. When freshwater input is reduced, the salinity may become more stable and salt tolerant species may colonise upper estuarine areas. Conversely, brackish and freshwater macrophytes would be displaced resulting in reduced species diversity (Wortmann *et al.*, 1998). However, if a substantial flood event follows a drier period, and associated saline intrusion, it is likely to destroy the upstream communities that have now become adapted to the saline conditions.

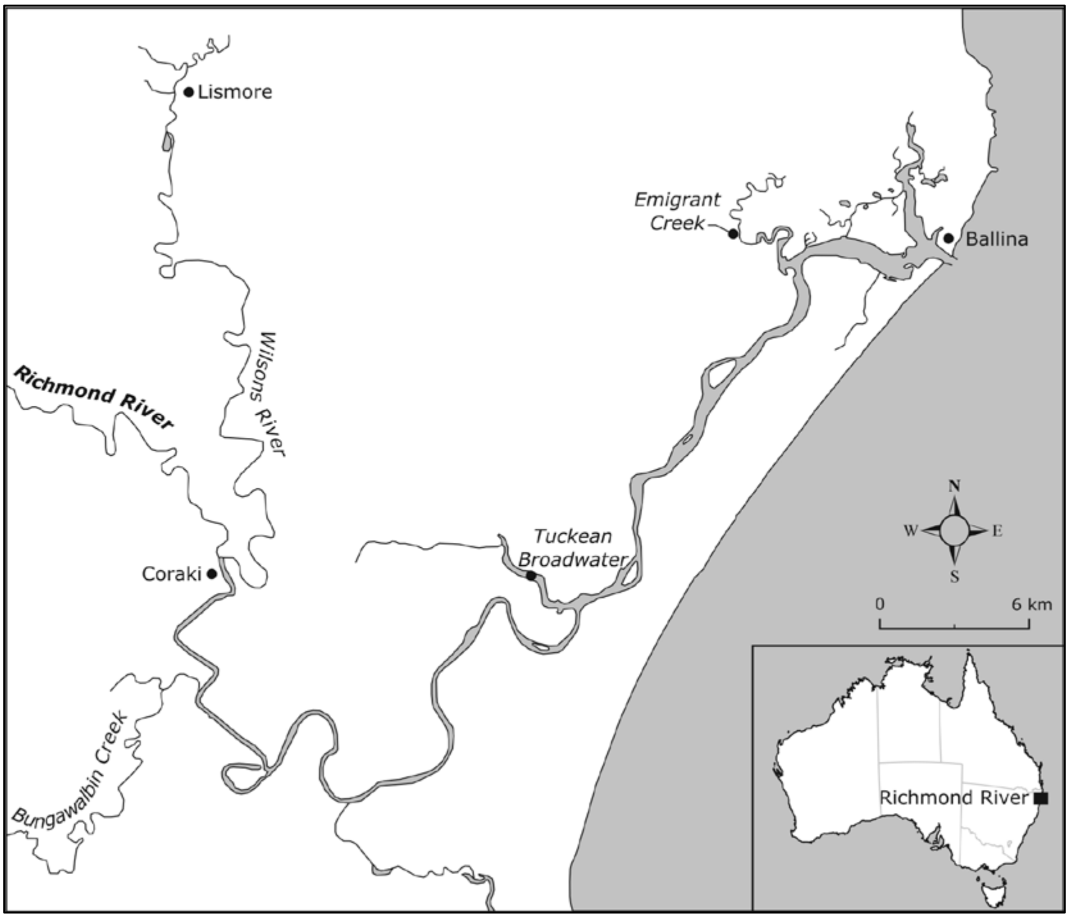

Figure 9: Richmond River Estuary

A review of salinity tolerances of aquatic and riparian vegetation revealed high variability in tolerance limits for some species (Peirson *et al.*, 1999). Salinity limits determined from experimental studies and field observations vary greatly between and within species. Tolerance limits of plants are suggested to be greater in systems where salinity fluctuates due to influences such as freshwater inputs and tides.

Peirson *et al.* (1999) conducted an investigation in the Richmond River estuary to determine if irrigators below the tidal limits of the estuaries should be treated differently from their counterparts, extracting riverine water upstream. The fundamental objective of the investigations were to assess the impact of freshwater extractions from the tidal pool on the estuarine behaviour and aquatic ecology of the Richmond River system, in particular for irrigation. Miller *et al.* (2004) also assessed the impact of freshwater extractions from the Richmond River tidal pool, on the estuarine behaviour and aquatic ecology of the river system.

The focus of this study was the Wilsons River, as Rous Water was considering supplementing the water supply for the region by extracting freshwater from the Wilsons River, near Lismore.

Peirson *et al.* (1999) calibrated models against available data, using it to estimate saline excursion into the estuary between 1940 and 1997. The numerical modelling of the movement of the freshwater/saltwater interface highlighted the large excursions in its motion in response to rainfall and sustained periods of dry weather. Movement of freshwater and saltwater were mapped, along with the impact of increases in the volumes being withdrawn from the river. The magnitude and frequency of saline intrusion into the estuary arms were found to increase (in comparison to the existing conditions) when freshwater inflows to the estuary were reduced by 30 %. However, the magnitude of the intrusion was found to be generally less than that produced by increasing the irrigation rates beyond a factor of two, this was more predominant during the drier, low flow periods (Peirson *et al.*, 1999).

Results in both investigations were discussed with relevance to key indicator species. Using the historical analysis of saltwater intrusion, parallel investigations, targeting vegetation, fish and platypus were undertaken. The use of indicator species was necessary to gather and analyse pertinent information necessary for assessing the potential impacts of predicted shifts in the estuary's salinity structure (Miller *et al.*, 2004). However, there are considerable uncertainties regarding the response of many freshwater biota to increased salinity levels. In particular, soil salinity is a critical issue for estuarine riparian vegetation, and there is little information available on the flow of saline estuarine waters into bed sediments.

Key species in the Richmond River estuary included two listed threatened fish species (the Eastern Freshwater Cod and the Oxleyan Pygmy Perch) and a range of freshwater associated species. Within specific species likely shifts in salt sensitivity with stages of the life cycle should be considered, e.g. greater sensitivity associated with breeding for species such as the Australia Bass. It is also important to consider likely impacts on estuarine invertebrates, in particular the commercially important prawn species and the Sydney Rock Oyster. Platypus, present in the upper estuary, are also a high-value aspect of the region, because of their high public appreciation and their conservation value (Miller *et al.*, 2004).

Peirson *et al.* (1999) determined any changes in water extraction rate on the Bungawalbin Creek arm would be reflected in changes in ecosystem risk. Bungawalbin Creek was already at risk and Peirson *et al.* (1999) noted any existing water usage should be reviewed. This is particularly important as this arm was identified as containing extensive areas of high-value physical habitat. Existing irrigation rates were determined to have not significantly increased the ecosystem risk within the Richmond River arm, however if this rate is doubled, the risk does increase. While existing irrigation rates are acceptable, they should only be increased with caution and after detailed investigation (Peirson *et al.*, 1999). Slight increases in ecosystem risk were identified between the no irrigation and existing conditions on the Wilsons River arm. Ecosystem risk was noticeable only during extremely dry periods. Existing irrigation rates are probably acceptable, but should only be increased if more detailed investigations show that this does not have an adverse ecological impact. Consideration should be given to restricting irrigation during extremely dry periods (Peirson *et al.*, 1999).

Saline intrusion has the potential to be a major problem in the Richmond River in the future, as it is expected to increase marginally due to sea level rise (Peirson *et al.*, 1999). In January 2003 the Manly Hydraulics Laboratory recorded water with salinity ranging from 1 to 2 ppt entering the upper-most 10 km of the estuary, such salinities would be expected to impact beds of aquatic macrophytes normally associated with freshwater (Miller *et al.*, 2004). If occurrences increase in frequency then there may be associated impacts. *Phragmites australis*, a prominent species in the upper estuary, is an edge species that may die-off from the lower reaches of the river if salinity levels rise above these plants tolerance levels (reported limits are mostly 5-25ppt) (Dalby-Ball *et al.*, 1999).

Predicted sea level rise is likely to also have impacts on other areas of the estuary, including: shoreline recession, inundation of low lying ecosystems, implications for drainage and flooding in urban and agricultural areas, and increased salt penetration into tidal pools or freshwater wetland systems. Furthermore, like many of Australia's estuaries, the Richmond River region continues to feel the impacts of urban expansion. While urban areas currently account for only 1.8 % of the land use surrounding the Richmond River estuary, the urban growth rate is rapidly increasing (ABER, 2011). Future urban expansion and the need for additional freshwater, is likely to place additional stress on the ecosystem. Additionally, increased saline intrusion from sea level rise, combined with the effects of decreases in rainfall are likely to have a major influence on the available water for agriculture and have substantial impacts on ecological communities.

6. Conclusions and Recommendations

Articulation of the values, goals and possible adaptation strategies for estuaries and/or parts of estuaries provide a framework approach to the challenging task of making decisions with respect to climate change adaptation. In this report, we present values in non-economic terms, but we do recognise that work in this area is progressing and represents perhaps the best way to compare and trade-off the environmental and human values of estuaries, especially including the ecosystems services these environments provide.

This report presents a new approach to the challenging task of making decisions regarding estuaries with respect to climate change adaptation through the introduction of estuarine focused adaptation strategies. The range of communities likely to be present in estuaries, and their associated values have been summarised in the report, as well as ecological and socio-economic goals for these communities. This will better enable those making management decisions for estuaries to consider the full range of estuarine communities. Adaptation strategies, and examples, are presented to provide a framework approach for decision making. Case studies of seven Australian estuaries; Towra Point, Georges River Estuary, Wilson Inlet, the Mary River, Gippsland Lakes, Tomago Wetland and the Richmond River Estuary, are used to illustrate past estuarine management successes and failures, and provide examples of estuarine goals and associated strategies.

Real climate change adaptation will be complex, so we advocate a cautious and holistic approach to climate change adaptation decision making. There are several reasons for this caution. First, the risk of maladaptation, also known as unforeseen or sometimes perverse outcomes, of any management intervention must be carefully considered prior to its implementation. Indeed, in cases where a variety of adaptation strategies may be feasible, decisions should then be based not solely on economic, social or environmental grounds, but rather with respect to the flow on consequences of the adaptation action across all three sectors. The outcome of these deliberations will therefore be determined very much by local context, the spatial and temporal scales of consideration, and the trading of values between the interested parties. Ultimately there is almost certainly no single adaptation strategy that will please everyone, but if a thorough examination is undertaken of the likely success of the action, in terms of maintaining the values and attaining the goals for the system as well as the flow on consequences in a highly connected landscape, then the likelihood of maladaptation will be significantly reduced. As demonstrated in our case studies, local context and local knowledge are critical both in terms of identifying the components of the systems and the values attributed to them, as well as nominating the adaptation strategies that show the greatest promise in terms of achieving our vision of estuaries that sustain environmental, economic and social values.

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