Coastal Wetland Habitat Dynamics in Selected New South Wales Estuaries.

Volume 1

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Bachelor of Arts
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Abstract

Intertidal wetland habitats in southeastern Australia have changed significantly during the past sixty years. Mangrove habitats have expanded both seawards and landwards, the latter being at the expense of saltmarsh habitats. This relatively common phenomenon is generally suggested to be an outcome of sea-level rise. Several factors potentially responsible for this change are examined, including changes in mean sea-level during the past 50 to 100 years, changes in climate, population growth, catchment landuse, and estuary type.

A protocol for mapping estuarine habitats was developed and implemented, incorporating the application of geographic information systems. Spatial and temporal coastal wetland habitat changes at nine sites along the New South Wales coast are illustrated. These habitat dynamics were shown to not correlate between sites. The results demonstrate that sea-level rise in this region cannot solely account for the extent of change during the past sixty years. With the exception of one site (Careel Bay), there have been no correlations between contemporary mean sea-level rise and mangrove incursion of the saltmarsh habitats at the study sites, or with rainfall patterns, at the scale of observation in this study, which was largely decadal. The only correlations determined during this study have been between population growth and coastal wetland habitat dynamics in some sites.

In spite of saltmarsh habitat loss being a regional phenomenon, local factors appear to have a profound bearing on the rates of change. Neither contemporary mean sea-level rise, rainfall patterns, estuary type, catchment landuse, catchment natural cover nor population pressure can account solely for the patterns in the spatial and temporal dynamics of the coastal wetlands of New South Wales. It seems apparent that regional factors create preconditions favourable for mangrove incursion, but that localised conditions have been responsible for the extent of these incursions from site to site. That is, despite higher sea-level and greater rainfall, the extent of change has been determined by the unique characteristics of each site.
The results have important implications for current estuary management practices in the state of New South Wales. The lack of spatial and temporal trends in coastal wetland habitat dynamics point to the need for management to be conducted on a localised, rather than regional scale. Additionally, anthropogenic influences must be carefully managed, since the extent of mangrove habitat expansion into saltmarsh areas is unlikely to be a natural occurrence.
This thesis contains no material extracted in whole or in part from a thesis by which I have qualified for or been awarded another degree or diploma. Chapter 3: Habitat Mapping Protocols, written specifically for this thesis, has been reproduced in its original form as an Australian Catholic University Technical Report.

No other person’s work has been used without due acknowledgment in the main text of the thesis.

This thesis has not been submitted for the award of any degree or diploma in any other tertiary institution.

…………………..February 2002

Kylee M. Wilton date
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Chapter 1: Introduction and Literature Review

1.1 Introduction

1.1.1 Model of the response of saline coastal wetlands to changes in mean sea-level in New South Wales estuaries

It is hypothesised that the dynamics of saline coastal wetlands may be used as indicators of environmental change, due to their sensitivity to specific environmental factors. In particular, changes in the species distribution and community structure of saline coastal wetlands are hypothesised as being indicators of sea-level rise.

Estuarine ecosystems and saline coastal wetlands are dynamic, responding to a range of stresses, both natural and anthropogenic. Ecological indicators can be valuable tools for measuring and assessing the condition of coastal wetland habitats. An ecological indicator is any expression of the environment that provides quantitative information on ecological resources, and is frequently based on discrete pieces of information that reflect the condition of the habitat in question (Hunsaker and Carpenter, 1990). These include the condition of the habitat itself, the response of species in these habitats to changes in salinity, changes in inundation, changes in sedimentation patterns, and changes in the magnitude of these and other stressors.

Coastal wetlands are extremely sensitive and responsive to changed environmental conditions and anthropogenic stresses. As a result, they provide ideal research sites to study ecological structure and function, because they respond rapidly and dramatically to changed conditions. They can be used to quantify ecological responses to long-term variability and disturbances that influence spatial and temporal distribution of wetland species.

This project assesses the spatial and temporal patterns of change in saline
coastal wetland habitats in New South Wales estuaries, primarily in relation to regional changes in mean sea-level, and in relation to other parameters, including climate change and catchment landuse. Hence, the specific aims of this study are to assess:

1. changes in the areas of mangrove and saltmarsh habitats along the New South Wales coast.

2. whether changes in mangrove and saltmarsh habitat areas along the New South Wales coast have been uniform.

3. whether wetlands have exhibited both the landward and seaward expansion of mangrove habitats, or just landward mangrove habitat incursion into saltmarsh habitat.

4. if rising sea-level during contemporary time has been responsible for changes in coastal wetland habitats.

5. if factors other than rising sea-level (i.e. rainfall patterns, catchment landuse change and population pressure), or in combination with sea-level rise, may have been responsible for the habitat changes.

A working definition of a wetland is that defined in New South Wales legislation, being:

...an area with characteristics between terrestrial and aquatic environments that is flooded or waterlogged often enough to support aquatic or other wetland plants. Flooding may typically be to a depth of two metres.

Mossop (1992), p. 331

Alternatively, Sainty and Associates Pty. Ltd. (2000) recommended a definition based on that in the Environmental Planning and Assessment Act (1979) and the New South Wales Wetlands Management Policy (DLWC, 1996), being:
Natural wetlands – including marshes, mangroves, backwaters, billabongs, swamps, sedgelands, wet meadows, wet heathlands, seagrass and saltmarsh that form a waterbody when inundated cyclically, intermittently or permanently with fresh, brackish or salt water, and where the inundation determines the type and productivity of the soil and the plant and animal communities.


Such wetlands were identified in the 1997 State of the Marine Environment Report (Zann, 1997) as undervalued and locally threatened habitats in Australia, a fact supported by research by Mitchell and Adam (1989a) and Wilton (1998). Zann (1997) stated that loss of saltmarsh had been concentrated in southeastern Australia where development pressure has been greatest. This loss is of particular concern because saltmarsh habitats in New South Wales are relatively small compared to other states: as of 1985, New South Wales supported 57 km² of saltmarsh habitat, Queensland supported 5,322 km² and Victoria supported 125 km² (West et al., 1985).

A number of research papers from Australia and around the world have identified coastal wetlands as indicators of sea level change. Blasco et al. (1996) identified mangroves as indicators of coastal change, as their ecosystems are so specialised that any minor variation in their hydrological or tidal regime can force these habitats to change in structure. The same can be said of saltmarshes, as they too are located within only a particular spectrum of the tidal range. Similarly, Pethick (1991) highlighted the vulnerability of marshes and mangroves to sea-level rise, as did Woodroffe (1990, 1993, 1995, 1999). Gordon (1988) identified a tentative link between sea-level rise and coastal recession in New South Wales, touting coastal wetlands as an indicator of such change. Bird (1988) also suggested a link between contemporary sea-level rise and coastal change in Australia. Vanderzee (1988) observed that changes in saltmarsh communities might be an indicator of sea-level rise. This is because each saltmarsh species has a very limited set of environmental conditions within which it can survive and if any or all of these environmental conditions are altered by a rising sea-level, then the saltmarsh community itself will change in
The key changes brought about by sea-level rise can include changes in the salinity and in the microtopography of the intertidal substrate, thus changing the elevation of the saltmarsh zone within the local tidal range. In addition to sea-level rise being a catalyst to saline coastal wetland habitat change, French et al. (1995) identified human, ecological and physical factors to which these habitats are responsive.

This widespread decline in saltmarsh habitat, most often in response to mangrove incursion, has been noted in several studies, summarised by Wilton (1998) and Saintilan and Williams (1999, 2000). This thesis is limited to examining habitat changes during contemporary history, since the aerial photographic record is not appropriate for longer-term research into habitat dynamics. Additionally, one of the key reasons why it is important to assess saltmarsh loss to mangrove incursion in contemporary time is well established paradigm, developed following stratigraphic analyses, that saltmarsh has historically prograded seaward over mangrove habitats during longer-term timescales (i.e. during the past 500 to 1700 years) (Saintilan and Hashimoto, 1999). Current trends are therefore contrary to the longer-term pattern of vegetation change.

1.1.2 Mangrove and Saltmarsh Distribution and Ecology

Mangrove and saltmarsh habitats occupy the intertidal zone of many coasts around the world, primarily the low energy areas of estuaries. Mangroves tend to inhabit the intertidal zones of tropical and subtropical coastlines, while saltmarshes inhabit the intertidal zones of temperate coastlines (Clough, 1982). A typical cross section is shown in Figure 1.1 (Lynch and Burchmore, 1992, p. 1).
The coastal zone of Australia has been subject to many different classification schemes. Most recently, the Interim Marine and Coastal Regionalisation for Australia Technical Group (Thackway and Cresswell, 1998) devised an ecosystem-based classification for marine and coastal environments around Australia. Based on this classification, the coastal zone of New South Wales was divided into five meso-scale regions: the Tweed-Moreton continental shelf, the Manning Shelf, the Hawkesbury Shelf, the Batemans Shelf, and the Twofold Shelf. These environments are microtidal and characterised by high wave energies, relatively steep intertidal slopes and limited input of fine sediments. This complements the Interim Biogeographic Regionalisation for Australia (IBRA) previously devised by Thackway and Cresswell (1995). Both these schemes will be discussed further in Chapter 2.

1.1.2.1 Mangrove Habitats

Mangroves are halophyloous (salt-adapted) evergreen forests of trees and shrubs that inhabit the intertidal zone of sheltered saline to brackish environments in tropical and subtropical latitudes (Jones et al., 1990; Augustinus, 1995). Worldwide, Saenger (1983) identified six distinct mangrove regions, each with varying characteristics: Asia, Oceania (of which Australia is a part), the west coast of America, the east coast of America, the west coast of Africa and the east coast of Africa combined with the Middle East. In Australia, mangroves inhabit the coastline from tropical northern Queensland to temperate southern
Victoria (Clough, 1982). They occupy that part of the intertidal zone between mean sea-level and mean high water spring tide (Bird and Barson, 1982; Galloway, 1982; Sainty and Jacobs, 1981; Williams, 1990; Ellison and Stoddart, 1991).

In 1982 it was determined that Australia contained approximately 11 500 km² (nearly 3 million acres) of mangroves in its coastal zone, 95% of which existed in tropical Australia and 11% located on offshore islands within 1 km of the continent (Galloway, 1982). These figures have not been comprehensively updated since Galloway’s report. Mangrove habitats around Australia have colonised alluvial plains, tidal plains, barriers and lagoons, composite alluvial plains and barriers, drowned bedrock coasts, coral coasts, sheltered coasts, riverine forests, and exposed mudflats and coral debris (Lear and Turner, 1977; Galloway, 1982).

The distribution of mangroves is dependent upon a number of key environmental parameters and their variables (Lear and Turner, 1977; Galloway, 1982; Hutchings and Saenger, 1987; Duke et al., 1998), including temperature, substrate, exposure, salinity, tidal range, ocean and estuary currents, climate, dispersal and establishment of propagules, and estuary type and size. The subtropical area of northern New South Wales is colonised by five mangrove species, *Avicennia marina* (Forssk.) Vierh., *Aegiceras corniculatum* (L.) Blanco, *Rhizophora stylosa* Griff., *Exoecaria agallocha* L., and *Bruguiera gymnorrhiza* (L.) Lamk., with the temperate area of New South Wales to the south colonised by only the first two species.

Duke et al. (1998) discussed the distribution of mangroves in terms of four geographic scales: global, regional, estuarine and intertidal. Global distributional gradients of importance include temperature, suitable habitat and climate, dispersal and establishment of propagules, the global importance of continental drift and tectonic events. Of note is that no species occurs only in the subtropics, although (low) air and water temperature restricts most species to at least the tropical latitudes of the globe, with some exceptions, such as the
presence of *Avicennia marina* in temperate zones of the Australian coast. Duke *et al.* (1998) noted that areas of higher rainfall and fluvial input host mangrove communities of greater diversity, which is true in both a global and regional context. The importance of propagule characteristics in determining areas of mangrove establishment was also noted, such as the number of days a propagule can remain viable and buoyant for transportation, their proximity to currents to aid their transportation to established or new intertidal areas, and the availability of such suitable habitats for establishment.

These global distributional gradients for mangrove establishment observed by Duke *et al.* (1998) are also true largely for regional distribution, yet on a smaller scale. That is, rainfall and both water and air temperature gradients control species richness and establishment along the eastern Australian coast; for example, species richness is greater in tropical northeastern Australia than temperate southeastern Australia (Hutchings and Saenger, 1987; Duke, 1992; Ball, 1998; Duke *et al.*, 1998). In particular, rainfall falling within the catchment, and hence converted to runoff, can also affect species richness and establishment (Ewel *et al.*, 1998).

Estuarine distributional gradients of note identified by Duke *et al.* (1998) were divided into downstream, intermediate and upstream categories, each representing approximately a respective third of the estuary. Such delineations can control a number of key factors affecting mangrove establishment, including salinity (both in the water and the soil), exposure to tidal gradients, tidal range and tidal variation. For example, *Avicennia marina* thrives in intertidal areas near the mouth of an estuary, as it enjoys saline conditions equalling those of the ocean, whereas *Aegiceras corniculatum* is more likely to be found further upriver in an estuary, where salinity is not as high, and there is greater freshwater input and mixing from upriver. Duke *et al.* (1998) noted that salinity levels in Australian estuaries have been found to be a function of annual rainfall, tidal variation and catchment area during the wet season, or tidal variation and catchment area during the dry season, as these relationships can control freshwater input to the wetland.
Finally, Duke et al. (1998) discussed those gradients that affect zonation patterns within communities in the intertidal zone, such as physiological influences along tidal profiles, faunal associations, propagule characteristics, establishment, competition and light effects, and root structure. They noted that these factors would differ from species to species. One of the key factors controlling mangrove establishment in an intertidal zone is tidal range and thus tidal inundation, because mangrove habitats usually exist within that area bordered by mean sea level and the spring tide level. Fluctuations in this parameter have been identified as controlling zonation within the mangrove zone, in terms of both periods of inundation and the consequent exposure to changing water and soil salinities. Duke et al. (1998) noted that mangroves in fact favour conditions of lower salinity, and grow slower in conditions of higher salinity.

A diagrammatic example of Duke et al.’s (1998, p. 44) determination of the relationships between mangrove floristic, structural and functional diversity, and the range of factors chiefly influencing mangrove distribution globally, regionally, locally and intertidally is given in Figure 1.2.
Figure 1.2: Relationships between mangrove floristic, structural and functional diversity and the range of factors chiefly influencing mangrove distribution globally, regionally, locally and intertidally (Duke et al., 1998, p. 44).

1.1.2.2 Saltmarsh Habitats

Coastal saltmarshes were defined by Adam (1990) as areas vegetated by herbs, grasses or low shrubs bordering saline water bodies and which are subject to periodic, predominantly saline, inundation, usually from the adjacent waterbody, and are located within a very limited spectrum of the upper tidal range. At their lower limit in the intertidal zone, inundation limits saltmarsh growth, whereas at the upper limit both inundation and terrestrial species limit saltmarsh proliferation (Carter, 1988). In New South Wales, saltmarsh plains are dominated by *Sporobolus virginicus* L. Kunth, *Sarcocornia quinqueflora* (Bunge ex Ungen-Stemberg) A.J. Scott, and *Juncus kraussii* Hochst. (Adam et al., 1988; Allen and Pye, 1992). Saltmarshes are found in higher elevations in estuaries. As sediments become trapped by vegetation, the marsh increases in
height and inundation occurs only on the highest (spring) tides (Jones et al., 1990). These low energy zones are characterised by the accretion of fine-grained sediments, forming mudflats colonised by saltmarsh species (Pethick, 1992; Augustinus, 1995).

The location of saltmarsh communities is dependent primarily upon sediment supply, tidal regime, and the wind-wave climate, in addition to the factors listed as being important to mangroves (Adam et al., 1988; Allen and Pye, 1992):

*Under normal conditions, saltmarsh grows between the mean and maximum high tide levels and typically occurs immediately landward of mangroves. Saltmarshes usually exhibit vertical zonation, caused by a combination of biotic (e.g. inter-specific competition and grazing) and abiotic (e.g. salinity and degree of water logging) factors (Morrisey, 1995; Zedler et al., 1995). For example, samphire appears to be more tolerant of hypersalinity than is couch grass, although the latter species distributions vary, even within areas of similar tidal range (Morrisey, 1995, Zedler et al., 1995). On Kooragang Island (within the Hunter River estuary), for example, samphire has been found to dominate both the lower (adjacent to mangroves) and upper (adjacent to rushes) areas (Zedler et al., 1995).*

NSW Fisheries (2000), p. 33

1.1.2.3 The Mangrove and Saltmarsh Habitat Boundary

Mangroves primarily differ from saltmarshes in two ways: in vegetation structure (mangroves are trees or shrubs, rather than herbs as most saltmarsh species are), and in their position with respect to mean high-water level (Augustinus, 1995).

In New South Wales, and in particular the Sydney region, Pidgeon (1940), and Clarke and Hannon (1967, 1969, 1970, 1971) were amongst the first to study the relationship between saltmarsh and mangrove communities. Clarke and Hannon (1971) found that *Sarcocornia quinqueflora* rarely grew beneath mangroves, yet was not limited by salinity or in many situations by waterlogging, or by light where the canopy was open. Clarke and Hannon’s (1967, 1969, 1970, 1971)
work included the examination of the vegetation, soils and climate of Sydney’s mangroves and saltmarshes, the ‘holocoenotic’ complex with particular reference to physiography (i.e. plant zonation patterns in response to site-specific microtopography), plant growth in relation to salinity and waterlogging, and the significance of species interaction, referred to later in this study. Other ecological studies of mangrove and saltmarsh complexes have been conducted, such as those listed in Table 1.1. A number of other studies examined mangrove and saltmarsh habitat relationships, including Kratochvil et al. (1972), King (1981), and Pethick (1984).

1.1.3 The Nature of Change in Intertidal Habitats

Intertidal habitat change has been studied on a variety of scales throughout the world, with much of the research focussed on sites in North America and Australasia. These studies can be broadly classified as being of a theoretical, palaeo, or contemporary nature. For example, there are a variety of theories and models regarding the effect of various rates of sea level rise on intertidal wetlands and their substrates. Some of these papers assess this issue in terms of the survival of such wetlands, and others in terms of changes within these intertidal wetlands, such as mangrove incursion upon saltmarsh and other floristic community changes.

In this thesis, habitat change is quantified and examined as a function of processes that induce environmental change (Thom, 1982), including mean sea-level change, estuary types, catchment area, bioregions, rainfall, population change, and catchment landuse. This study assesses the spatial and temporal patterns of coastal wetland habitat dynamics, in relation to these factors. Due to their sensitivity to, or reliance upon, local tidal regimes, it has been hypothesised that intertidal and estuarine saline coastal wetlands are a physiographic, or ecological, indicator of change in mean sea-level. That is, it is predicted that a rise in mean sea-level will result in changes in the distribution of the habitats under examination (Vanderzee, 1988; Williams, 1990). Saenger (2001) noted that recent consensus is that such changes in mangrove habitats
appear to be more variable than the rate of sea-level rise itself.

1.1.3.1 Conceptual Models of Intertidal Habitat Response to Sea-level Change

A number of hypotheses have been presented regarding the effect of changes in mean sea-level on intertidal saline coastal wetlands. Orson et al. (1985) suggested that an intertidal wetland would only survive if sedimentation were equal to or greater than sea level rise. This was supported by Allen (1990a, 1990b), who stated that marsh vertical growth relative to a tidal frame was a function of the rates of minerogenic and organogenic sedimentation, the rate of change and tendency (i.e. direction) of relative sea level, and the rate of long-term sediment compaction. Allen (1990a, 1990b) concluded that a marsh builds to that level relative to the moving tidal frame at which the sedimentation upon it equals the rate of sea level rise (dynamic equilibrium), and that the equilibrium level is inversely proportional to the rate of sea level movement. Such theories regarding those mechanisms responsible for marsh accretion were supported in research by Vanderzee (1988), Ellison and Stoddart (1991), French (1991, 1993), Allen (1995), Ellison (1996), and Cahoon and Lynch (1997).

In summary of such mechanisms, Orson et al. (1985) listed the following three major responses a saltmarsh could have to rising sea-level:

1. *The marsh system could drown if rates of coastal submergence exceed the marshes’ ability to accrete vertically.*

2. *The marsh may remain stable if the input of sediments equals the rates of coastal submergence so that surface elevations are maintained.*

3. *The marsh can actively expand both vertically and laterally if accretion rates are higher than rates of submergence.*

Orson et al. (1985), p. 32

King (1981) examined the nature of both mangrove and saltmarsh communities on an individual basis and as wetland complexes. The changing nature of zone boundaries was attributed to changes in environmental factors such as those also discussed by Clarke and Hannon (1969), which included frequency of
inundation and its effects on soil salinity and aeration. It was suggested that it is unlikely that one or two factors alone could be responsible for boundary locations between mangrove and saltmarsh communities.

That changes in the extent of habitats in intertidal areas along the world’s coastlines will be a result of sea-level rise was recognised by Woodroffe (1990), who also noted that the local extent of change would be reliant upon local shoreline topography, sources of sediment, rates of sediment supply, and the rate of sea-level rise. This was based on a reconstruction by Woodroffe of the response of mangroves to past sea-level rise episodes, in order to forecast future changes.

Further to this, Woodroffe (1990, 1993, 1995) identified the extreme vulnerability of mangrove ecosystems around the world to the rapidly increasing rate of sea-level rise. Woodroffe noted in these papers that if sea-level rise and sedimentation keep in step with one another, then mangroves would keep pace within the tidal frame. But if sea-level rise is anywhere within the range previously suggested by Henderson-Sellers and Blong (1989) of between 20 and 140 cm by 2030 (and since revised by the IPCC, in Church et al., 2001), it was predicted that mangrove ecosystems throughout the world would almost certainly diminish in area in the absence of a corresponding rate of sedimentation.

In 1999 Woodroffe added to his body of work on the response of mangrove shorelines to sea-level change, stating that such a response would be a function of their overall sediment budget. Woodroffe (1999) used palaeo mangrove sediments to show that the habitats have not been land-builders, as suggested by Bird (1971), but instead have retreated in response to the sea-level rise of the past few thousand years. Further to this, Woodroffe (1999) stated that mangrove response to sea-level rise is different throughout the world, and is heavily reliant upon the degree to which mangrove organic matter contributes to the sediments in the intertidal zone. This in turn is reliant upon the local geomorphic setting, a concept originally developed by Thom (1982) and further developed by Woodroffe (1990, 1992, 1993) to incorporate the following settings: river-
dominated, (such as river deltas), tide-dominated, and carbonate. Mangrove response to sea-level rise can differ even within these settings.

Semeniuk (1994) made a number of predictions of the effect of sea-level rise on mangroves in general and in particular in northwestern Australia, more specifically with respect to the effects of global warming on mangroves. He stated that the three primary changes as a result of global warming that would alter mangrove habitats would be sea-level rise, increased storminess, and temperature rise. These factors would form a relationship with the key factors that determine the occurrence of mangroves along the coast, which are the geomorphic setting of the mangrove system, sedimentologic processes, the salinity structure of the groundwater/soilwater system, and the maintenance of strategies utilised by the mangrove populations. Following these relationships, based on the macrotidal coastal environment of Western Australia, Semeniuk (1994, pp. 1063-1064) listed those essential environmental factors that could influence the variable response of mangroves to rising sea-level (based on a rate of sea-level rise of 1 mm per year culminating in a net rise of 0.5 m by the year 2045):

1. Coastal geomorphology and habitat heterogeneity will determine: first, whether there will be a uniform or heterogeneous sedimentologic response at a small time scale, and second, whether mangroves will encroach onto a mosaic of substrate types or a relatively uniform substrate.

2. Tidal range will determine whether or not a sea-level rise will involve a significant proportion of the present mangrove habitat. For areas around Shark Bay and Exmouth Gulf, that are microtidal, a predicted rise of 50 cm will completely inundate the existing mangrove zone. For areas that are macrotidal, the predicted rise in sea level may be only 5-10\% of the total tidal range and will not have the same effect.

3. Whether the coast is eroding or accreting will determine whether a sea-level rise will encroach into an eroding coastal system and potentially exacerbate the situation, or into an accreting coast and potentially change the sedimentary budget of the system (e.g. by reworking sedimentary material currently stored out of reach of wave base).

4. The origin of the upper tidal flat (i.e. due to a falling or rising Holocene sea
level history) into which potential mangrove encroachment will occur determines whether the rising sea will encroach onto a shore with a small or large gradient. A very small rise in sea level onto a coast with a very low to negligible gradient will result in significant lateral incursion and flooding. A similar rise onto a coast that has a steeper gradient will involve a less dramatic incursion.

5. Whether the system occurs in an arid or humid climate will determine the structure of the salinity fields under the tidal flats, the extent of the salt flat landward of the mangroves, and the dominant type of mangrove population maintenance.

Semeniuk (1994), pp. 1063-1064

Stolper (1996, 1998) developed a computer simulation model to attempt to predict the response of estuarine mangroves to sea-level rise, and applied various scenarios to hypothetical and actual mangrove environments. The model was based on simulating the effect that sea-level rise, topography and sedimentation have on the profile of estuarine substrates, and consequently the intertidal zone. Stolper (1998, p. 16) gave the following example:

For example, imagine that, over a given time period, sea-level rises 10 cm and the maximum rate of [marsh elevational rise] is 12 cm. In this case, the rate of deposition at the seaward margin of the intertidal zone (which is reached by all high-tide events) will be greater than the rate of sea-level rise. This will shift the boundary in a seaward direction.

In contrast, at the landward edge of the intertidal zone the rate of sedimentation is lower than the rate of sea-level rise (because it is reached by a reduced number of high-tide events). This will lead to increased inundation and a landward migration of the intertidal margin. Thus the intertidal zone will be extended in both directions, accompanied by a flattening of the substrate. This conclusion is contrary to other recent studies, which have predicted that sea-level rise will lead to the devastation of mangrove environments.

Stolper (1998), p. 16

Stolper’s (1998) simulations suggested that sea-level rise would lead to an increase in the extent of the intertidal zone, as opposed to a decrease (aided by
development adjacent to many wetlands) of the intertidal zone, predicted by many.

### 1.1.3.2 Stratigraphic Palaeo-scale Models of Intertidal Habitat Change

A number of researchers have studied Holocene-scale intertidal habitat dynamics in response to past sea-level rise, in order to try to determine both contemporary and future responses.

Thom and Roy (1988) studied sea-level rise and climate change during the Holocene and their effect on coastal environments, with reference to the geometry of depositional units, sediment composition and ecological characteristics. This longer time scale of analysis prompted Thom and Roy (1988) to suggest that adjustments in coastal environments are not simply in response to changes in their boundary conditions, such as rates of sea level change and storminess, but also to internal sediment budget adjustments. They argued that caution must be taken with any predictions for the outcomes of sea-level rise on coastal and intertidal environments.

Ellison and Stoddart (1991) studied the ability of mangrove ecosystems to keep pace with various rates of sea-level rise based on a review of stratigraphic records during the Holocene, work which has since been widely criticised. They concluded that mangrove ecosystems would be able to keep pace with a rising sea-level of 8-9 cm/100 years, would be under stress at rates of rising sea-level of 9-12 cm/100 years, and would be completely removed at rates of rising sea-level greater than 12 cm/100 years. Based on their findings, they hypothesised that mangrove ecosystems of the world, and in particular Oceania, will be placed under extreme pressure if rates of rising sea-level will be in the 100-200 cm/100 years range as previously predicted before recent revisions.

Further to this, Ellison (1993) examined the retreat of the seaward edge of mangroves in Bermuda for the period 1932 to 1980 based fluctuating mean sea levels during that time. As mean sea level rose at a rate of 24 cm/100 years (Barnett, 1984) and $28 \pm 18$ cm/100 years (Pirazzoli, 1986), the study found that
the mangrove swamp kept pace with the relatively slow rate of sea-level rise during the past thousand years. The study also found that the rate of peat accretion for the past 100 years (8.5-10.6 cm/100 years) has been less than the rate of sea-level rise during the past few centuries (14.3 cm/100 years), and in particular less than the rate of sea-level rise for the past 100 years (28 ± 18 cm/100 years). The mangrove dieback at the seaward fringe was thus attributed to a greater degree of rising sea-level than sediment accumulation within the mangrove swamps of Bermuda.

Mangrove habitat dynamics and Holocene sea-level changes in the southwestern coast of Thailand were studied by Fujimoto et al. (1999), who used the relationships between ground level, sediments and present vegetation, spatial distribution of mangrove forest-floor deposits and their formative periods to clarify Holocene mangrove dynamics. They showed that as sea-level rose and fell cyclically during the Holocene, and when it fell approximately 2200 years BP, the site’s *Rhizophora apiculata* forest in fact expanded in a seaward direction, following the pattern of sea-level change. As sea-level rose during the last 2000 years at a rate of approximately 6.0 mm/yr, it was shown that the *Rhizophora* forest maintained its place in the intertidal zone by accumulating mangrove peat with the sea-level rise.

1.1.3.2 Contemporary Studies of Intertidal Habitat Change in Response to Sedimentation and Sea-level Change

Many studies of estuarine habitat change around the world have been focussed on saltmarsh habitats, at the exclusion of mangroves, which are less-commonly found at such research sites. Many of these studies were from the perspective of analysing the marsh substrate itself, in terms of sedimentation, accretion and erosion, which influence plant communities by controlling the local tidal regime, and in response to local rates of sea-level rise.

Stevenson et al. (1986) examined the relationship between sea-level rise and marsh accretion rates at fifteen U.S. Gulf and Atlantic coast marshes, and found no causal relationship. A significant positive correlation (r = 0.83) was found
between accretionary balance (accretion rate minus relative sea-level rise) and mean tidal range. From this, it was inferred that marsh sediment accumulation rates increase linearly with increasing tidal range.

The assumption that accretion rates of intertidal saltmarshes are approximately equal to rates of sea-level rise was tested by Bricker-Urso et al. (1989) at sites along the Rhode Island coast. This was in response to situations throughout Louisiana where a combination of subsidence and sea-level rise is leading to more open water areas rather than the maintenance of coastal wetlands, suggesting that these coastal wetlands cannot keep pace with relative sea-level rise. Bricker-Urso et al.’s (1989) study found that the coastal wetlands of the Rhode Island coast were in fact keeping pace with relative sea-level rise, in opposition to those in Louisiana. This was most likely due to a suitable level of sediment input to the Rhode Island marshes.

Sediment accumulation at a fringe marsh during transgression was assessed by Oertel et al. (1989) at a site in Virginia. It was determined that rates of sediment accumulation are governed by a combination of the surface elevation with respect to mid-tide elevation, the rate of sea-level rise, and outwash from the mainland. In the case of fringe marshes in particular, Oertel et al. (1989) concluded that inputs of coarse-grained sediments play a key role in the relationship between the rate of fringe marsh accretion and the rate of sea-level rise, though the variation between sedimentation and sea-level rise is progressively greater from mid-tide level to high-water level. From this, the fine-grained (tide-deposited) component of marsh deposition (at or below mid-tide level) may produce a constant rate that approaches the rate of sea-level rise.

Patterns of sediment accumulation were examined by Wood et al. (1989) over one year in the tidal marshes of Maine. This was carried out with reference to marsh morphology, local relative sea-level rise, mean tidal range, and ice rafting activity. Four different marsh types were examined (back-barrier, fluvial, bluff-toe, and transitional), all of which showed different rates of sedimentation. Wood et al. (1989) did not observe a causal relationship between the modern marsh sediment accumulation rate and the rate of relative sea-level rise.
Semeniuk (1994) attempted to predict the effect of sea-level rise on mangroves in northwestern Australia by examining background information such as the geomorphic setting, the sedimentologic processes along the coast, the salinity structure of the groundwater/soilwater system, the maintenance strategies utilised by the mangrove populations, coastal dynamics and erosion and industrial impacts. The study found that in King Sound, Western Australia, mangrove habitats had been advancing landwards at a rate equal to the local sheet erosion rate of 1-3 cm/year, which was attributed to rising sea-level. Parkinson et al. (1994) made a similar observation to Ellison and Stoddart’s (1991), that mangrove proliferation is largely controlled by rates of sea-level rise.

Kearney et al. (1994) supported the generalisation that vertical accretionary dynamics are the primary factor in determining long-term marsh stability, as tested by Bricker-Urso et al. (1989), but from their own studies showed that employing short-term estimates of site-specific vertical accretion rates could potentially over-estimate marsh accretionary response to future sea-level rise.

The impacts of sea-level rise on deltas in the Gulf of Mexico and the Mediterranean were examined by Day Jr. et al. (1995), who noted that rates of relative sea-level rise are great in sites such as these that are in fact also subsiding. They noted wetlands in these situations in danger of survival due to a combination of reduced sedimentation and increasing rates of relative sea-level rise, which can drown the wetland. They also stressed the importance of management strategies ensuring the delivery of sediment to these areas during strong pulsing events such as river floods and storms, to aid the wetlands’ survival.

Sedimentation and boundary changes of saltmarshes in Virginia, U.S.A., were examined by Kastler and Wiberg (1996). They determined that the island marsh lost 7.2% of its area in 8 years by overwash; that the lagoon marsh lost 10.6% of its area over 41 years by recession of marsh edges; and that the mainland marsh area increased by 8.2% over 50 years primarily by upland encroachment. Kastler and Wiberg (1996) attributed most of these changes to a reflection of the
moderation of oceanic processes, and to local shoreline submergence.

Accretion of a saltmarsh in Massachusetts, U.S.A., in response to inlet migration, storms, and sea-level rise was examined by Roman et al. (1997). They found that the marsh appeared to keep pace with the rate of local relative sea-level rise from 1921 to 1993 of 2.4 mm/year. That is, as sea level rose, the saltmarsh was able to migrate upwards and landwards in order to remain within the appropriate tidal range to sustain its inhabitation of this site. Roman et al. (1997) concluded that in the short term, storms were important for delivering pulses of sediment to the marsh surface, but in the long term factors such as proximity to major tidal channels and inlets and the character of the peat substrate were important determinants of marsh survival under rising sea-level.

Sea-level rise and rates of vertical accretion in a tidal saltmarsh in Connecticut, U.S.A. were interpreted by Orson et al. (1998). They found that this system was sensitive to changes in sea level and storm activity. They used peat records to reconstruct historic sea-level curves in the area. The results suggested that the relationship between the accretion deficit and plant community structure (Spartina patens in this case) is important, yet stressed that in systems where major vegetation changes are prominent over short periods of time of less than 50 years, interpretations of sea-level rise should be limited to the system in which they are developed unless careful vertical controls can be maintained and multiple datable horizons can be identified within the substrate. Orson et al. (1998) concluded that the most important determinants of rates of vertical accumulation in any wetland are sediment inputs, flooding regime, storm frequencies, microtopography, plant community structure and the time the plant communities require to adapt to the new flooding regime, and autocompaction of the substrate, and that these will vary from site to site.

A number of studies in the U.S.A. have been undertaken to specifically study changes in vegetation in response to marsh vertical accretion, by installing surface-elevation tables in marsh substrates in order to determine sedimentation, accretion and compaction, amongst other factors.
In their study of marsh vertical accretion in a Louisiana saltmarsh, Cahoon and Reed (1995) found that accretion was significantly related to duration of flooding of the marsh surface. Their data also indicated that accretion and accumulation varied temporally and spatially in relation to hydroperiod and suggested that accretion at lower marsh elevations is not always sufficient to maintain an equilibrium position in the intertidal zone. That is, accretion in the lower marsh will not necessarily enable marsh habitats to keep pace with a rising sea-level.

Later, marsh vertical accretion in the north arm of Tijuana estuary in southern California was measured by Cahoon et al. (1996), who found that accretion there during an 18 month period was related almost entirely to episodic storm-induced river flows during just three months of that time. Cahoon et al. (1996) also predicted that during times of drought the marsh surface elevation could be outpaced by predicted rates of rising sea-levels, hence outpacing the wetland habitats in the absence of freshwater and sediment delivery.

Vertical accretion and shallow subsidence in a mangrove forest in southwestern Florida was measured in another part of the U.S.A., by Cahoon and Lynch (1997). This study concluded that vertical development of mangrove soils was influenced by both surface and subsurface processes, noting that as sea-level has risen during the past 70 years the mangroves have remained stable, exhibiting no landward or seaward incursions.

Further to this, Cahoon et al. (1999) analysed various schools of thought regarding the influence of surface and shallow subsurface soil process on wetland elevation and hence wetland habitat change. They noted that the customary approach was based on the assumption that vertical accretion is equal to wetland elevation change, after comparing vertical accretion to relative sea-level rise and then calculating an accretion deficit. They stressed that their new approach would allows scientists: (a) to test the assumption directly; (b) to calculate an elevation deficit if accretion and elevation are not equal; (c) to calculate the amount of subsidence (accretion minus elevation change) in the top few metres of the substrate (i.e. shallow subsidence); and (d) to distinguish
the effects of surface (e.g. sediment deposition and erosion) versus subsurface processes (e.g. soil compaction, root growth/decomposition, pore water storage) on wetland elevation. This was achieved using sedimentation-erosion tables, originally devised by Boumans and Day, Jr. (1993). From this work, Cahoon et al. (1999) stressed that predicting the vertical buildup of coastal wetlands to sea-level rise always requires site-specific information, that generalisations cannot be made for wetlands everywhere.

Another study to employ the surface-elevation table was conducted by Day Jr. et al. (1999), who examined the soil accretionary dynamics, sea-level rise and the survival of wetlands in Venice Lagoon, Italy, based on the high rate of wetland loss in the lagoon. Their results suggested that reduced wave energy and increased sediment availability are needed to offset wetland loss due to storm activity, floods and sea-level rise at sites throughout the lagoon. They determined that changes in allogenic sediment deposition, decomposition and autogenic primary production were induced by changes in wetland elevation. Importantly, Day Jr. et al. (1999) noted that it is misleading to compare short-term rates of accretion to soil elevation change, and thus wetland habitat change, to rates of rising sea-level, because this ignores longer-term process such as compaction of the wetland substrate.

Hensel et al. (1999) examined the importance of riverine flooding with respect to wetland vertical accretion in the Rhône River Delta, France. They used a sedimentation-erosion table (originally developed by Boumans and Day, Jr. 1993) to measure riverine, marine and impounded wetlands habitats from 1992 to 1996. The importance of such a study is because as a wetland accretes vertically, the species that inhabit it may or may not also change in their distribution, in response to varying or new parameters within which they must attempt to thrive. Hensel et al. (1999) found that the riverine habitat accreted much faster than the impounded and marine habitats, which accreted at rates less than relative local sea-level rise. From this study, Hensel et al. (1999) concluded that the wetlands connected to the Rhône River can accrete rapidly from sediments deposited during floods, yet the wetlands in the impoundments (which they determined to be the most common ‘natural areas’ left in the delta)
were not keeping pace with relative sea-level rise and therefore vulnerable.

### 1.1.4 Mangrove Transgression in Australasia

Numerous studies have been undertaken in Australasia to quantify changes in mangrove and saltmarsh habitats (Table 1.1). These changes include mangrove seaward progradation and transgression (usually at the expense of saltmarsh habitats), a decline in saltmarsh habitats, and a seaward transgression of *Casuarina glauca* Sieb. Ex Spreng. habitats (also often at the expense of saltmarsh habitats).

Table 1.1: Studies reporting spatial and/or temporal changes in saltmarsh and mangrove habitats around the coast of Australia (after Saintilan and Williams, 1999, 2000).
<table>
<thead>
<tr>
<th>Location</th>
<th>Habitat Description</th>
<th>Scale (1954-1996)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kooragang Island, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>1:16 000, 1:32 000, 1:4000-1:1000, 1:10 000</td>
<td>Outhred &amp; Buckney (1983), Buckney (1987), Williams et al. (2000)</td>
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<td></td>
<td></td>
<td></td>
<td>MacDonal (1996)</td>
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<td>Lake Macquarie, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>1:16 000</td>
<td>Winning (1990)</td>
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<td>Brisbane Waters &amp; Hawkesbury River, NSW</td>
<td>mangrove, saltmarsh &amp; peripheral (casuarina)</td>
<td>1:100 000, 1:10 000, 1:16 000</td>
<td>Benson (1986), Department of Environment and Planning (1983), Harty (1999)</td>
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<td>Courangra Pt., Hawkesbury River, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>1:12 000</td>
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<td>Berowra-Marramarra Ck., NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>1:70 000</td>
<td>Williams and Watford (1997)</td>
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<td>Pittwater, Cowan Creek, Middle Harbour, NSW</td>
<td>mangrove</td>
<td>-</td>
<td>Blacker (1977)</td>
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<td>Careel Bay, Pittwater, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>1:10 000</td>
<td>Wilton (1997, 1998)</td>
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<td>Lane Cove River, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>various</td>
<td>McLoughlin (1987)</td>
</tr>
<tr>
<td>Parramatta River, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>various</td>
<td>McLoughlin (2000)</td>
</tr>
<tr>
<td>Homebush Bay, NSW</td>
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<td>-</td>
<td>Clarke and Benson (1988)</td>
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<td>Kurnell Peninsula, Botany Bay, NSW</td>
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<td>Evans (1997)</td>
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<td>Botany Bay, NSW</td>
<td>estuarine wetland</td>
<td>-</td>
<td>Watford &amp; Williams (1998)</td>
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<td>Towra Point, Botany Bay, NSW</td>
<td>mangrove &amp; saltmarsh</td>
<td>1:12000</td>
<td>Mitchell &amp; Adam (1989b)</td>
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<td>Fenech (1994), Odell (1994)</td>
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<td>Yassini (1985)</td>
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<td>Minnamurra River, NSW</td>
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<td>Carne (1989), Chafer (1998b)</td>
</tr>
<tr>
<td>Lake Illawarra, Shoalhaven River, St Georges Basin, Burrill Lake, Clyde River, Tomago River, Moruya River, NSW</td>
<td>mangrove, saltmarsh &amp; peripheral</td>
<td>1:10 000</td>
<td>Anderson et al. (1981)</td>
</tr>
<tr>
<td>Shoalhaven River, NSW</td>
<td>mangrove, saltmarsh &amp; peripheral</td>
<td>1:5000</td>
<td>Chafer (1998a)</td>
</tr>
<tr>
<td>Jervis Bay, NSW (Currambene Ck. &amp; Cararma Inlet)</td>
<td>mangrove, saltmarsh &amp; peripheral</td>
<td>1:4000</td>
<td>CSIRO (1989)</td>
</tr>
<tr>
<td>Clyde River, Coila Lake, Tuross River, Brou Lake, Wagonga River, Wallaga Lake, NSW</td>
<td>seagrass, mangrove &amp; saltmarsh</td>
<td>1:25 000</td>
<td>Briggs et al. (1980)</td>
</tr>
<tr>
<td>Merimbula Lake, NSW</td>
<td>seagrass, mangrove &amp; saltmarsh</td>
<td>1:5000</td>
<td>Meehan (1997)</td>
</tr>
</tbody>
</table>
1.1.4.1 Trends Throughout Australia

Saenger (2001) made an observation of the extent of mangrove habitats around the entire Australian coastline, noting that of the 30,266 km of coastline approximately 20% (6089 km) represents mangrove shoreline. Saenger (2001) made a further comment based on Galloway et al.’s 1982 work, that based on the total Australian mangrove area of 11,617 km², 70.5% of mangrove areas occur in estuaries, whilst 29.5% occur on open shorelines.

Cappo et al. (1998, p. 24) provided information on the estimated areas of saltmarsh and mangrove habitat alteration throughout Australia (Tables 1.2 and 1.3).

Table 1.2: Estimated areas (ha) of saltmarsh/saltpan habitat altered in Australian States (Cappo et al., 1998, p. 24).

<table>
<thead>
<tr>
<th>State</th>
<th>NSW</th>
<th>VIC</th>
<th>TAS</th>
<th>SA</th>
<th>WA</th>
<th>NT</th>
<th>QLD</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reported Instances</td>
<td>5</td>
<td>2</td>
<td>3</td>
<td>2</td>
<td>6</td>
<td>0</td>
<td>43</td>
<td>83</td>
</tr>
<tr>
<td>Area Lost</td>
<td>&gt;20</td>
<td>-</td>
<td>Loss</td>
<td>Loss</td>
<td>&gt;3,617</td>
<td>-</td>
<td>&gt;4,013</td>
<td>&gt;7,650</td>
</tr>
<tr>
<td>Area Gained</td>
<td>-</td>
<td>gain</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>slight gain</td>
</tr>
<tr>
<td>1989 Area</td>
<td>5,700</td>
<td>12,500</td>
<td>3,700</td>
<td>8,400</td>
<td>296,500</td>
<td>500,500</td>
<td>532,200</td>
<td>1,359,500</td>
</tr>
</tbody>
</table>
Table 1.3: Estimated areas (ha) of mangrove habitat altered in Australian States (Cappo et al., 1998, p. 24).

<table>
<thead>
<tr>
<th></th>
<th>NSW</th>
<th>VIC</th>
<th>TAS</th>
<th>SA</th>
<th>WA</th>
<th>NT</th>
<th>QLD</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reported Instances</td>
<td>10</td>
<td>1</td>
<td>-</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>68</td>
<td>83</td>
</tr>
<tr>
<td>Area Lost</td>
<td>&gt;42</td>
<td>loss</td>
<td>-</td>
<td>Loss</td>
<td>Loss</td>
<td>&gt;27</td>
<td>&gt;2,816</td>
<td>&gt;2,885</td>
</tr>
<tr>
<td>Area Gained</td>
<td>&gt;53</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>&gt;874</td>
<td>&gt;927</td>
</tr>
<tr>
<td>1989 Area</td>
<td>10,700</td>
<td>4,100</td>
<td>0</td>
<td>11,100</td>
<td>156,100</td>
<td>295,200</td>
<td>342,400</td>
<td>819,500</td>
</tr>
</tbody>
</table>

1.1.4.2 Trends in New South Wales

New South Wales Fisheries (2000) recently presented a document detailing the status of fisheries in the state, and made approximations of the total area (km²) of mangrove habitats in New South Wales regions (note: this data is based on the most recent state-wide assessment, which was 1981-1984, which was detailed in West et al., 1985):

- Far north coast: 16.3
- North coast: 13.9
- Sydney-Hunter: 65.4
- Illawarra: 5.8
- South coast: 4.7
- Far south coast: 2.0

Based on the previous data, New South Wales Fisheries (2000) noted that the most important mangrove habitats in New South Wales (in terms of overall proportion in the early 1980s) were the Port Stephens system (26%), the Hunter river (15%), the Hawkesbury River system (11%), the Botany Bay system (6%) and the Clarence, Macleay and Richmond Rivers (each 5%), most of which are either drowned river valley estuaries or barrier estuaries (i.e. coastal rivers).

The report by New South Wales Fisheries (2000, p. 26) noted that the overall area of mangrove habitat appears to be expanding, attributed to their observation that the area of suitable substrate appears to have increased because of
sedimentation.

New South Wales Fisheries (2000) also assessed the approximate total area saltmarsh area in New South Wales regions (note: this data is based on the most recent state-wide assessment, which was 1981-1984, which was detailed in West et al., 1985):

- Far north coast: 4.8 km$^2$
- North coast: 13.8 km$^2$
- Sydney-Hunter: 28.2 km$^2$
- Illawarra: 4.6 km$^2$
- South coast: 4.0 km$^2$
- Far south coast: 3.6 km$^2$

New South Wales Fisheries (2000) noted that of the 57 km$^2$ of saltmarsh in New South Wales estuaries in 1985, those estuaries with more than 2 km$^2$ of saltmarsh habitat were the Macleay River, Lake Innes/Lake Cathie, Wallis Lake, Karuah River, Port Stephens, Hunter River and Jervis Bay.

In their report, New South Wales Fisheries (2000) noted that many saltmarsh areas are degraded, and are threatened by a relatively poor public profile, reclamation, dumping, use of off-road vehicles, and the invasion by exotic plant species, and oil spills.

1.1.4.3 Previous Studies of Wetland Habitat Change in Eastern Australia

The benchmark study of West et al. (1985) assessed the extent of estuarine communities along the New South Wales coast, and is the only study to have provided a comprehensive inventory of these habitats to which many researchers have compared their subsequent results.

Although many of these studies have been successful in documenting and quantifying change in mangrove and saltmarsh habitats, they have not necessarily been successful in attributing these changes to one or more specific
factors. For example, various authors have shown that habitat change occurred at a certain time, as did change in one of the hypothetical variables being tested, but these two factors could not be conclusively linked to one another.

The spatial and temporal patterns of mangroves at Oyster Point Bay in South East Queensland were studied by McTainsh et al. (1986), for the period 1944 to 1983. They concluded that the mangrove area had increased by 110% during this period, with the greatest rate of increase occurring between 1964 and 1972. This increase was in a landward direction, with the seaward border of the mangroves remaining stable, and in the absence of a landward saltmarsh habitat. The authors studied a number of factors that may have been responsible for the increase in mangrove area, including mangrove zonation and stability, land use changes, sea level change, and climatic change. Despite arguing each of these factors, it was suggested that urbanisation of the catchment was not responsible for the expansion of the mangrove habitat, as the greatest degree of habitat change occurred prior to the majority of changes throughout the catchment. Alternatively, they identified sea-level rise as possibly being responsible for the landward expansion of mangroves at the site, in conjunction with the changing pattern of local rainfall, noting a positive correlation between increased rainfall during the period 1964 to 1972 with rapid mangrove expansion. The authors attributed this to a freshening of the wetland, though later periods of slower mangrove habitat expansion had weaker correlations with rainfall trends.

Research projects listed in Table 1.1 have assessed this issue of estuarine habitat change in Australasia by studying a variety of issues, such as the response of mangrove and saltmarsh habitats to rising sea level, to changes in catchment landuse and urbanisation, and in response to changes in climatic factors, such as changes in rainfall patterns.

Saintilan (1997a, 1998), Wilton (1998) and Saintilan and Williams (1999, 2000) examined changes in the mangrove and saltmarsh habitats in a number of estuaries, and suggested a number of hypotheses to explain the landward transgression of mangroves in south-east Australia (Saintilan, 1997a, p. 105), some examples of which are documented in the reports listed in Table 1.1:
Hypothesis 1: That higher rainfall in the second half of the century has decreased the salinity of the saltmarsh zone making it more suitable for mangroves.

Hypothesis 2: That higher nutrient levels within estuaries have allowed the colonisation of mangroves on flats which were previously nutrient depleted.

Hypothesis 3: That higher temperatures have increased the competitive ability of mangroves (which are essentially tropical) in saltmarsh environments.

Hypothesis 4: That the landward expansion of mangroves is evidence of sea-level change, either from land subsidence or absolute sea-level rise.

Saintilan (1997a), p. 105

Building on this, Saintilan and Williams (1999, 2000) assessed twenty-eight mapping surveys of estuarine habitat change in southeastern Australia for various periods ranging back to the 1930s (incorporated into Table 1.1), and noted that the issue of mangrove incursion into saltmarsh is a widespread phenomenon, taking place at the expense of saltmarsh in almost all instances. They presented a number of hypotheses for these changes: (a) a natural cyclicity in mangrove transgression, possibly in correspondence with patterns of rainfall and sea-level in the region; (b) human modifications of estuary; revegetation of areas previously cleared for agriculture; (c) altered tidal regimes or estuary water levels; (d) increased siltation and elevated nutrient levels in estuaries since and during human modification; and (e) competition between these communities along the intertidal gradient.

Some studies have attributed estuarine habitat change almost solely to changes in catchment landuse, whilst acknowledging that other factors, such as sea-level rise and increased rainfall, may play a small part.

Cappo et al. (1998, pp. 25-26) provided information based on recent assessments in Queensland (Table 1.4), which was able to both quantify and qualify changes in wetland areas in that state.
Table 1.4: Recent wetland vegetation assessments in Queensland, which highlight several factors associated with change (Cappo et al., 1998, pp. 25-26).

<table>
<thead>
<tr>
<th>Location</th>
<th>Wetland Type</th>
<th>1990s Area</th>
<th>Change Since Earlier</th>
<th>Change %</th>
<th>Change Rate ha/yr</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Johnstone River, 1951-1992 (Russell &amp; Hales, 1994)</td>
<td>mangrove freshwater</td>
<td>202 925</td>
<td>+26 -1,752</td>
<td>+15 -65</td>
<td>+0.6 -42.0</td>
<td>High rainfall area. Greatly altered by land clearing &amp; development.</td>
</tr>
<tr>
<td>Moresby River, 1951-1992 (Russel et al., 1996a)</td>
<td>mangrove freshwater</td>
<td>2,873 1,175</td>
<td>+640 -2,188</td>
<td>+29 -65</td>
<td>+15.6 -53.4</td>
<td>Moderate-high rainfall area. Greatly altered by land clearing &amp; development.</td>
</tr>
<tr>
<td>Hinchinbrook Channel Islands, 1943-1991 (Ebert, 1995)</td>
<td>mangrove saltmarsh/saltpan all intertidal</td>
<td>3,790 46</td>
<td>+208 -161</td>
<td>+5.8 -77.8</td>
<td>+4.3 -3.4</td>
<td>Moderate rainfall area. Not directly affected by development.</td>
</tr>
<tr>
<td>Moreton Region: Coolangatta to Caloundra, 1974-1987 (Hyland &amp; Butler, 1988)</td>
<td>mangrove saltmarsh/saltpan all intertidal freshwater artificial areas</td>
<td>14,457 5,101</td>
<td>+1,953 -591</td>
<td>+8.8 -10.6</td>
<td>+94.9 -45.5</td>
<td>Moderate rainfall areas. Greatly altered by reclamation, development &amp; construction of canal estates.</td>
</tr>
</tbody>
</table>

1.1.4.4 Northern New South Wales

The boundary stability between the mangrove and saltmarsh habitats of the Tweed River was assessed by Saintilan (1997b, 1998). Photogrammetric mapping showed that mangroves increased substantially in area in all parts of the estuary from 1930 to 1994, which was attributed to the progradation of intertidal flats in combination with a vertical accretion of the seaward edge of the intertidal zone. This progradation was in response to increased sediment delivery brought about by catchment landuse change, in conjunction with deposition of reworked marine sand with increased flood-tide velocities following dredging operations, and increased tidal amplitude following engineering (dredging) work, which was deemed to have promoted both the
landward and seaward expansion of the mangrove area throughout the estuary. Despite this, there was a net decline in mangrove area in the estuary between 1962 and 1971, attributed primarily to reclamation. Concurrently, it was shown that saltmarsh actually increased throughout the estuary from 1930 to 1971, and then declined until 1985, in response to the development of Greenbank Island, and documented to have remained stable from 1985 to 1994. Saintilan did note, though, instability in the mangrove-saltmarsh at some sites in the estuary, including Caddys Island, Chinderah Island, the Tweed Broadwater, and Ukerebagh Island, the last of which will be studied in detail in Chapter 4.

Saintilan’s (1997b, 1998) findings of saltmarsh habitat increase throughout the Tweed River catchment as a whole are significant, as the only other known study to have documented a similar increase in the extent of saltmarsh was that by Chafer (1998a), who studied estuarine communities on Commerong Island, Shoalhaven Heads, but this increase was in the absence of mangrove communities.

Of relevance to this thesis, Saintilan (1997b) reported instability of the mangrove-saltmarsh boundary on Ukerebagh Island’s western side, within which he mapped 34 hectares of new mangrove habitat, previously dominated by saltmarsh species. Landward transgression of mangroves into saltmarsh habitat was also noted by Saintilan in the central southern portion of the island. Saintilan’s findings will be compared to that of the present study in a later chapter.

1.1.4.5 Central New South Wales

Many studies of estuarine vegetation change on Kooragang Island, in the Hunter River, have been conducted. These have shown vegetation change was partly a response to catchment landuse change. Outhred and Buckney (1983) examined the nature of vegetation change on Kooragang Island using historical aerial photographs dating back to 1954. Analysis of these changes led them to the following results:
1. Mangrove areas are more extensive now [1983] than in 1954, mainly through transitions from saltmarsh. The newly established mangroves are away from the main drainage channels.

2. Many of the mangroves established since 1954 are now exhibiting stress symptoms.

3. The soil salinity in the areas occupied by these stressed mangroves is currently very high, considerably greater than that of sea water.

4. Large stumps in some saltmarsh areas indicate that mangrove-to-saltmarsh transitions have occurred in the more distant past.

Outhred and Buckney (1983), p. 68

Outhred and Buckney (1983) hypothesised that the following factors were responsible for these habitat changes, either individually or in combination: (a) changing climate patterns (which affect the salinity of the wetland, and thus the vigour of the habitats); (b) sediment deposition in mangrove areas, causing a transition to saltmarsh; (c) erosion, which would favour a transition from saltmarsh to mangroves; (d) construction of tidal channels, which would promote mangrove establishment higher in the tidal frame; and (e) flood mitigation schemes which may affect the mangrove/saltmarsh equilibrium.

Buckney (1987) furthered the studies of Kooragang Island, and noted that the rate of mangrove expansion into and at the expense of saltmarsh was most rapid between 1954 and 1979. This reinforced Outhred and Buckney’s (1983) findings that these changes were in response to both human modifications and reclamation of both the catchment and Kooragang Island itself during the 30 years in conjunction with long-term climatic change and engineering works within the Hunter River, which has altered the local tidal regime.

An extensive study of estuarine habitat change on Kooragang Island based on historical scientific and social data was also undertaken by Williams et al. (2000), who incorporated analysis of the whole island and those wetlands on the northern shore of the lower Hunter River, excluding Hexham Swamp. Four major periods of environmental change were observed: an early development stage (1796-1885), mid development stage (1896-1950), late development stage
(1951-1989) and rehabilitation stage (1990-2000). The study noted that during the early stages the island’s wetlands were dramatically altered and even extensively reclaimed. From the 1950s to the mid 1990s the mangrove habitat was observed to increase from 1310 ha to 1711 ha, an increase of 131%. Concurrently, saltmarsh decreased in area, from 2133 ha in 1954 to 1428 ha in 1994, a loss of 33%. Williams et al. (2000) attributed the nature of change in estuarine wetlands to changes in the amount of substrate that is tidally inundated, which in itself is controlled by catchment modifications (producing excess sediment) and engineering practices (such as dredging and redistribution of riverine sediments), which have in turn changed the extent of the intertidal zone.

MacDonald (1996) assessed the extent of estuarine vegetation on Kooragang Island, for the years 1993 and 1996. The advantage of this project was that MacDonald was able to identify some individual saltmarsh species communities during the mapping process, and hence assessed the required environmental conditions of individual species with reference to their mapped boundary changes. From 1993 to 1996, the *Juncus, Sarcocornia quinqueflora*, freshwater saltmarsh, and mangrove habitats increased in area, whilst mixed saltmarsh, non-specific upland trees, *Sporobolus virginicus*, upland pasture, bare earth and *Melaleuca quinquenervia* (Cav.) habitats decreased in area. The change from the original mangrove, saltmarsh and temperate vegetation habitats to primarily agricultural vegetation was attributed to European settlement and activities. Culvert construction throughout the island reduced areas that could be tidally inundated, hence reducing areas where mangrove and saltmarsh species could continue colonisation. MacDonald attributed the increase in mangrove and *Sarcocornia quinqueflora* areas is a reflection of increased rainfall during the period of study, stating that *Sarcocornia quinqueflora* can be a reflection of changing salinity levels. As such, the increased rainfall would have freshened the wetland, allowing both the mangrove species and this saltmarsh species to expand in area under the favourable conditions. MacDonald (1996) offered an extensive discussion of changes in the various wetland species present on Kooragang Island.
Winning (1990) conducted a mapping assessment of wetland habitat change in Lake Macquarie, and based on interpretation of aerial photographs of the area taken in 1961 and 1987, it was determined that there had been an overall decrease of estuarine wetland in the catchment of 5%, attributed primarily to clearing, reclamation and hydrological changes. During this time, saltmarsh was determined to have decreased with an almost equal amount of increase in mangrove habitat, half of which was at the expense of saltmarsh, and the other half the result of colonisation of adjacent mudflats.

The wetlands of Brisbane Water have perhaps been exposed to more catchment and direct landuse modification than most in New South Wales. The Department of Environment and Planning (1983) conducted an estuarine wetlands study of Brisbane Water. On an estuary-wide basis, mangroves increased in area by 114% from 1954 to 1979 (with encroachment in both landward and seaward directions), whilst saltmarsh habitats decreased by 58%. At sites studied in this thesis, mangroves increased by 113% on Pelican Island whilst saltmarsh decreased by 57%; on Rileys Island mangroves increased by 131% whilst saltmarsh decreased by 55%; and on St Huberts Island mangrove habitats were almost totally removed for canal estate reclamation (a reduction of 91%), whilst saltmarsh was totally removed. This catchment has experienced faster population growth rates than most local government areas in New South Wales, and it is catchment modification that has led to increased sediment and nutrient delivery to the estuary which has promoted most of the changes in its estuarine habitats, most of which have been directly significantly modified.

In order to assess the nature of boundary changes at Courangra Point, on the Hawkesbury River, Saintilan and Hashimoto (1999) examined the upper stratigraphy of the area to determine past boundary changes between mangrove and saltmarsh habitats. From this, they in fact determined that saltmarsh had transgressed upon mangrove habitats between 500 and 1700 years before the present, suggested to be in response to accretion of the intertidal flat. This data suggests that the contemporary phenomenon of mangrove encroachment of saltmarsh is in fact contrary to longer-term successional patterns, in response to greater rates of contemporary sedimentation, as opposed to during the period
500 to 1700 years ago.

1.1.4.6 The Sydney Metropolitan Region

Wetlands in metropolitan areas around Sydney have suffered greatly in response to extensive unnatural modification of the catchment. One such site is Homebush Bay, which has undergone widespread reclamation and disturbance since European colonisation. Clarke and Benson (1988) mapped estuarine vegetation change from 1930 to 1983, during which extensive reclamation of the vegetation occurred, yet there was also mangrove expansion into saltmarsh in some places during the 1940s in response to the breaching of man-made levee banks, which was halted in the 1950s by further reclamation. Following this reclamation, mangroves recolonised many areas, but saltmarsh species were not able to do so.

Another estuary which has undergone major catchment change is Botany Bay, within which are the Towra Point wetlands. The decline of saltmarsh in Botany Bay was attributed by Mitchell and Adam (1989b) primarily to changes in catchment landuse. This area has a history of significant and long-term impacts since, and closely related to, European colonisation over 200 years ago. The authors attributed some of the historical saltmarsh loss to reclamation, but focussed in their paper on saltmarsh loss to mangrove incursion. A freshening of the wetland was nominated as a possible cause for mangrove incursion into saltmarsh, but the possibility that the mangrove expansion is a form of recovery from intense local disturbance is hypothesised as a more likely reason. The authors concluded that further work should be undertaken if the disturbance was in fact local, or a component of regional change.

A number of university studies also assessed changes in the mangrove and saltmarsh habitats in this region. Fenech (1994) reviewed estuarine wetland change in the Sutherland Shire of Sydney for the period 1956 to 1994, during which the mangrove area remained relatively static, whilst the saltmarsh habitats decreased by 39%. Fenech stated these results were somewhat misleading, because as a whole, mangrove habitats throughout the Shire actually
decreased in area, largely due to reclamation, but the expansion of the mangrove habitat into the saltmarsh at Towra Point biased the results. The effects of urbanisation were concluded as being primarily responsible for wetland change in the shire. Similarly, Odell (1994) noted large-scale wetland loss to reclamation throughout the Shire, although some mangrove stands did actually increase in area.

Additionally, Evans (1997) calculated that mangrove habitats at Towra Point (referred to in his report as Kurnell Peninsula) have increased in area by 33% from 1956 to 1996, whilst saltmarsh decreased by 78%. Evans attributed this wetland loss, and degradation, to human impact, citing changes in water quality resulting from domestic and industrial pollution, and alterations in the patterns of wave action and currents in response to dredging and other developments around the bay as being potentially of greatest impact. Additionally, large-scale reclamation has been responsible for much of the saltmarsh loss.

Thorogood (1985) assessed changes in the distribution of mangroves in the Port Jackson-Parramatta River estuary from 1930 to 1985. It was found that the area of mangroves fluctuated, and in particular experienced a significant decrease during the period 1951 to 1970, attributed to reclamation and/or drainage of tidal land. Thorogood noted that reclamation in the estuary reached its peak between 1961 and 1970, when approximately 48 ha (24%) of the mangroves in the estuary were reclaimed. Thorogood inconclusively hypothesised about the reasons for the mangrove expansion between 1930 and 1951, and between 1970 and 1985, attributing these increases to increased sediment delivery to the estuarine waters as a result of increased clearing of the catchment for conversion to urban areas. This had the effect of increasing intertidal zones within the estuary. In particular, mangrove expansion in Homebush Bay may have been aided by the creation of drainage channels throughout the wetland, which increased local tidal inundation and hence promoted mangrove establishment.

McLoughlin (1987, 2000) comprehensively investigated changes in estuarine vegetation at sites throughout the Sydney region. Shoreline vegetation of the Middle Lane Cove River for the period spanning approximately 1780 to 1880
McLoughlin (1987) noted that the catchment of the Lane Cove River began to change substantially during the 1880s, from a catchment characterised by agricultural settlement to an increasingly suburbanised one. It was noted that with this suburbanisation of the catchment came further clearing of the native vegetation of the catchment, particularly on the slopes, closer suburban settlement and road building, and accelerated sedimentation of the river. With this also came an expansion of mangrove habitats along the river, primarily in a ‘seaward’ direction onto the newly accreted mudflats. By the 1890s, mangroves were increasingly colonising new areas of the river, wherever sediment had accreted to form new intertidal zones of a width of at least 1 to 2 metres. During the 1950s, the mangrove habitat along the river reached its peak in terms of area (28 ha), and by 1988 had plateaued to approximately 24 ha, following dredging, filling and reclamation along the river.

The increase in mangrove area along the river during the period of analysis was noted by McLoughlin (1987) to be similar in character to increases in other estuaries in the Sydney region, where mangrove colonisation had been associated with ‘continuing accretion on tidal flats eroded from either developing urban areas or bushland badly damaged by fire’. In her analysis of the reasons for the changes in shoreline vegetation, McLoughlin noted that mangrove colonisation did not occur on all accreting mudflats throughout Sydney, though she did not measure the height of the mangrove communities in relation to the Australian Height Datum. McLoughlin also hypothesised that the presence/input of nutrients in the form of suspended solids in run-off from suburbanised catchments that were subsequently delivered to particular mudflats
in fact aided mangrove colonisation.

McLoughlin (2000) furthered her theory of sedimentation following urbanisation as being the catalyst to increasing mangrove areas in her study of estuarine wetland distribution along the Parramatta River in Sydney for the period 1788 to 1940. Historical material suggested that until the 1870s mudflats and saltmarsh communities dominated the intertidal zones of the Parramatta River, and that mangroves began to colonise these areas in the later 19th century, and didn’t encroach upon saltmarsh communities until the 20th century.

1.1.4.7 Southern New South Wales

Yassini (1985) closely examined the zonal characteristics of the foreshore vegetation of Lake Illawarra, focussing on zonation within saltmarsh habitats, whilst noting the presence of mangroves around the lake, which increased in area between 1930 and 1985. Yassini noted the importance of the extent and duration of tidal inundation in controlling the spatial and temporal development of saltmarsh distribution, hypothesising that changes in tidal inundation will bring about changes in saltmarsh distribution. This theory relates closely to that of sea-level rise being responsible for changes in estuarine habitat distribution, which would have the effect of altering the inundation regime at affected sites.

Chafer (1998b) calculated the spatial and temporal changes in estuarine vegetation change in the Minnamurra River, south of Sydney, for the period 1938 to 1997. For the entire area studied, it was found that the *Casuarina* communities increased by 43%, consistently at the expense of saltmarsh. The area of mangrove increased by 70%, with a steady increase from 1938 to 1972, and an even greater rate of increase from 1972 to 1981. These increases were also at the expense of saltmarsh. From 1938 to 1997, saltmarsh decreased by 51%, at a reasonably constant rate, and usually at the hands of both the mangrove and *Casuarina* habitats. Chafer attributed these changes primarily to the changing landuse patterns of the Minnamurra River’s catchment. That is, parts of the catchment were cleared for grazing and agricultural landuse, others for residential and commercial development. It was suggested that clearing of
the catchment contributed to increased sediment delivery to the estuary, which promoted the natural creation of new intertidal areas which were subsequently colonised by mangroves, and also promoted the vitality of the mangrove community itself, purportedly in response to higher levels of phosphorus contained in the sediment eroded from the catchment. Chafer conceded that these changes might have worked in conjunction with sea-level rise to promote habitat change in some parts of the intertidal zone.

Chafer (1998a) examined the response of estuarine habitat change from 1949 to 1996 to major geomorphological change at the mouth of the Shoalhaven River, which has been documented to be cyclically opening and closing, which Chafer loosely related to strong positive and negative deviations in the Southern Oscillation Index. Chafer mapped the estuarine habitat change on Commerong Island. This was done in conjunction with an assessment of its geomorphological change, the El Niño Southern Oscillation, solar sunspot data, and a simplified categorical classification of the entrance condition. The saltmarsh was documented to have remained static from 1949 to 1970, and then increased by 442% from 1970 to 1996, with 83% of this increase occurring after 1981. Chafer hypothesised that this was in response to sedimentation on the island, with the establishment of a dune system.

The CSIRO (1989, 1991, 1994; Clarke, 1993) conducted a number of baseline studies of the wetland habitats of the wetland habitats of Jervis Bay, focussing on measuring environmental variables that were deemed to control estuarine vegetation distribution. The CSIRO (1991, 1994) found that there had been a net increase of 20 ha in the area of mangroves during the past 50 years in Jervis Bay, noting that much of this had been at the expense of saltmarsh. During the period 1944 to 1961 there had been a decrease of 5.1 ha of mangrove at Currambene Creek (though it was noted that the negative change might have been the result of measurement error due to poor resolution of canopy gaps in images from 1944). Yet, during the period 1961 to 1989 there had been an increase in mangrove area of 14.1 ha, most of which was noted to have been in the form of colonisation of the saltmarsh. Mangrove expansion in Caramba Inlet was noted to have been in primarily a seaward direction. The reports did not
offer any reasoning for these changes, but did note that they are part of a widespread phenomenon of change throughout southeastern Australia.

Meehan (1997) examined historical changes in seagrass, mangrove and saltmarsh communities between 1948 and 1994 in Merimbula Lake and Pambula Lake, both on the far south coast of New South Wales. The study found that both saltmarsh and seagrass habitats decreased in area at both sites, whilst mangrove habitats concurrently increased in area. It was noted that in Merimbula Lake the mangrove area increased by 121.7%, primarily at the expense of saltmarsh, which decreased by 29.5%. Mangrove expansion here was in both a landward and a seaward direction (suggested to be the result of increased sedimentation from the catchment). Meehan attributed the seaward expansion of mangroves to increased sediment delivery from the catchment, which created new intertidal areas that were colonised by mangroves, but was unable to determine a reason for mangrove incursion into saltmarsh. Saltmarsh loss was attributed to both reclamation and incursion by mangroves. In Pambula Lake the mangrove area increased by 83.5%, whilst the saltmarsh area decreased by 40%, almost entirely due to mangrove incursion and some seaward migration of the adjacent freshwater wetlands.

1.1.4.8 Southern Australia

Coleman (1998) examined changes in a mangrove/samphire community in North Arm Creek, South Australia, for the period 1979 to 1993. From 1979 to 1993 there was both landward and seaward migration of *Avicennia marina*; the landward migration was at the expense of two-thirds of the samphire community. Concurrently, samphire communities colonised unvegetated areas and some previously colonised by mangroves. Later, from 1985 and 1993, mangrove colonisation of the samphire community was reduced, attributed by Coleman to a combination of land subsidence and sea-level rise, changes to the water flows and tidal dynamics of the area.
The response of mangrove habitats to rising sea-level has been studied from a number of angles, some of which simply being the response to actual sea-level rise, and others in response to factors that artificially accelerate the rate of sea-level rise, such as land subsidence.

The mangrove habitats of South Australia have been noted as responding to a unique combination of rising sea level and land subsidence, in effect increasing the local rate of sea-level rise. Burton (1982a, 1982b) studied a number of mangrove stands north of Adelaide, each of which were responding to local variables in different ways. The study was confined to the period 1935 to 1982. In one instance, a mangrove stand was documented to have been expanding in a seaward direction at a rate of 18.2 m/year from 1949 to 1982, in the absence of human interference (this expansion was onto bare mudflat, supporting Burton’s belief that mangroves consolidate, rather than promote, progradation). In another case, bounded by coastal towns, there had been little or no increase in mangrove area. In a third case, there had been considerable, and locally conspicuous, landward expansion of the mangrove habitat, determined to have been at a rate of 17 m/year from 1935 to 1982, which was attributed to either a rise in sea level, subsidence, or a combination of the two.

1.1.4.9 New Zealand

Some studies of estuarine habitats have attributed boundary community changes to changes in the sedimentation of a system, which is often in response to the changing nature of the catchment.

Young and Harvey (1996) examined the spatial relationship between mangrove (Avicennia marina var. australasica) physiognomy and sediment accretion in the Hauraki Plains, New Zealand. Their experimental results indicated that accretion increases with pneumatophore density, and from that it was proposed that Avicennia marina grows pneumatophores at a density that balances the benefits of aeration with the drawbacks of sediment accretion. Evidence of this is the spread of the mangrove forest fringe 200-250 m across previously unvegetated mudflats in the Firth of Thames of the North Island, New Zealand,
during the past 49 years. From their results and observations, Young and Harvey (1996) hypothesised that the advance of *Avicennia marina* has been so rapid during the past 49 years that it may have in fact enhanced the background sedimentation rate during this period.

Other studies have attributed mangrove expansion to issues of classification during the mapping process.

For example, Morrisey *et al.* (1999) examined changes in the abundance and distribution of coastal and estuarine vegetation in the Auckland region from the late 1930s/early 1940s to 1999. They mapped estuarine habitat change in six estuaries in the region, and the degree of change varied at each site. At those sites that were determined to have experienced mangrove expansion, this was primarily attributed to the maturing of original mangroves into denser stands. An expansion of the mangrove habitat boundary itself, in a landward direction, rather than just an increase in the density of the original habitat, was observed in Puhinui Inlet, at the expense of saltmarsh. This was also observed in Whangapoua Harbour, yet the boundary moved in a seaward direction to inhabit the adjacent mudflat. The authors did not attribute these changes in an across-the-board manner to changes in catchment landuse; rather, they noted that the response of some of the mangrove habitats might be attributed to sediment input from the catchment, but noted that mangrove habitat expansion has also occurred in the absence of significant catchment change.

This compilation of studies shows that estuarine habitat changes can be attributed to a wide array of factors, as supported by Woodroffe (1999, p. 159), who stated that:

> *Future changes are to be expected on mangrove shorelines, but differentiating between natural changes, human-induced changes, or response to global environmental change will be difficult, and may differ between locations.*

Woodroffe (1999), p. 159

In summary, studies throughout Australia, and specifically along its east coast,
have shown that mangrove habitats have been increasing in area, usually at the expense of saltmarsh habitats. A wide variety of factors were hypothesised throughout these studies as causing these changes, such as rising sea-level, changing climate patterns, increased sedimentation, flood mitigation schemes, increased catchment urbanisation, the freshening of wetlands, changes in the amount of estuarine substrate that is tidally inundated, engineering practices and hydrological changes. This study recognises that each of these factors may have played a role in the changing nature of coastal wetland habitat dynamics, but contributes further to the issue by detailing the elevation of mangrove incursion in New South Wales estuaries. This study attempts to correlate both habitat changes and the elevation of mangrove incursion with mean sea-level, rainfall patterns and population growth.

1.2 Aims and Hypotheses

Aims

It has been hypothesised that the landward transgression of mangrove habitats in wetlands along the New South Wales coast is a function of sea-level rise. One of the main objectives of this research is to assess whether or not there is a tangible link between regional sea-level rise and changes in coastal wetland habitats.

If mangrove transgression into saltmarsh were a consequence of sea-level rise, one would expect the transgression to occur along elevational gradients. Areas preserved as saltmarsh would be of consistently higher elevation than those locations encountering mangrove incursion, and the elevation gradient crossed in the process of transgression would correspond to sea-level rise over the period.
Hypotheses

This study is based on three primary hypotheses, although both Chapter 4 (Wetland Habitat Mapping) and Chapter 5 (Site Contour Modelling and Sea-level Rise) contain their own subsets of hypotheses:

Hypothesis 1:
That changes in the areas of mangrove and saltmarsh habitats have been occurring along the New South Wales coast.

Hypothesis 2:
That changes in mangrove and saltmarsh habitat areas along the New South Wales coast have not been uniform. That is, changes have not been occurring in all appropriate habitats, and some wetlands have exhibited both landward and seaward expansion of mangrove habitats, or just landward mangrove habitat expansion.

Hypothesis 3:
That rising sea-level during contemporary time is consistent in timing and amplitude with changes in coastal wetland habitats.

Following from the analysis of these hypotheses is an assessment of other factors for consideration, which are climate change and catchment landuse.

1.3 Significance of the Study

Saltmarsh habitats in southeastern Australia have been observed as rare and declining habitats (Zann, 1997). They are one part of saline coastal wetlands, which are also comprised of mangrove habitats. Saltmarsh areas are important as fish nurseries and wader bird habitats. During the past 50 years, mangroves have been observed to be encroaching upon saltmarsh along the New South Wales coast (Thorogood, 1985; McLoughlin, 1987, 2000; Mitchell and Adam,
Chapter 1: Introduction and Literature Review

1.4 Thesis Structure

An assessment of the major factors to be assessed for their relationship with coastal wetland habitat change is made in Chapter 2, upon which the site selection is made and justified. These factors include consideration of estuary type, bioregion classification, rainfall change as an indicator of climate change, and population growth as an indicator of catchment landuse change.

Since mapping of the coastal wetland habitats was a prerequisite of the analysis of contributing factors, a protocol for mapping mangrove and saltmarsh habitats was devised, and presented in Chapter 3. This was done following a review of over 40 previous studies conducted throughout Australia, which were designed to map and quantify coastal wetland distributions. This chapter provides the justification for the decision rules applied to mapping in this study.

The maps of coastal wetland habitat distributions at the nine study sites, for five dates (roughly decadal) during contemporary history, and the showing areas and types of habitat change, were produced for Chapter 4. The earliest and the most recent maps of each site are displayed, as well as a map showing the summary of habitat changes during the entire period of mapping. The remainder of the dataset is provided in Appendix 3. From these maps, assessments of the nature of habitat change, specifically of saltmarsh habitat loss, are made.

The relationship between mean sea-level patterns of the past sixty years and the elevation of mangrove incursion at five representative sites (a subset of the nine sets mapped) is assessed in Chapter 5. This analysis involved creating digital
elevation models, based on data collected in the field, which were in conjunction with the maps of habitat change to determine the elevational ranges of habitat change, specifically of mangrove incursion into saltmarsh.

The relative influence of other factors is considered in Chapter 6, which assesses the importance of climate change (represented by rainfall patterns) and changes in catchment landuse (represented by population growth). This is because any environmental system is governed by a vast array of factors, and so the catalysts to coastal wetland habitat change cannot be limited to rising sea-level.

Relationships, conclusions, limitations and uncertainties from the analysis sections of Chapters Four, Five and Six are discussed in the final section, Chapter Seven. The major findings of the thesis are presented here, with a discussion of further research issues.
Chapter 2: Site Selection

2.1 Criteria for Site Selection

Although coastal wetland habitat dynamics are assessed in this research in relation to contemporary sea-level changes, a number of other factors are recognised as important considerations in influencing habitat dynamics. These considerations guided the selection of sites, to ensure the assessments were made in a broad range of geomorphic and geographical contexts. This helped to determine if these other considerations were of importance in habitat dynamics.

The New South Wales coast is diverse in nature. It consists of different terrestrial, marine and coastal biogeographic regions. It is home to population centres ranging from cities to small towns, to catchments of varying sizes, to different rainfall regimes, and to over 130 estuaries of differing type and size, which have been subject to differing levels of engineering works. Its diverse nature reflects the possibility that there are diverse factors affecting its coastal wetland habitat dynamics. However, mangrove encroachment on saltmarsh is occurring in all these settings (Saintilan and Williams, 2000).

The sites selected for this research were chosen to reflect this diverse nature. The first basis for their selection was the presence of coastal wetland habitat change during contemporary time. Following from this, sites were chosen to reflect the diverse nature of:

- estuary types and bioregions,
- rainfall, and
- population change and catchment landuse.

The following section will discuss the varying nature of each of these factors, followed by a justification of the site selections, and their background information.
2.1.1 New South Wales Estuaries

Estuaries have been defined in numerous ways, the most widely-accepted being Pritchard’s (1967) definition: ‘[an estuary is] a semi-enclosed coastal water body which has a free connection to the open sea and within which sea water is measurably diluted with freshwater derived from land drainage’. The NSW coast consists of over 130 estuaries of varying types. The coast has been described as an embayed bedrock coast with a narrow discontinuous coastal plain, subject to moderately high energy ocean waves and a microtidal (<2 m) tidal range (Roy, 1984).

There are a number of schools of thought in Australia with respect to estuary classification and characteristics, three of which are prominent. The first was originally developed by Roy (1984, 1994), further developed by Roy et al. (2001); the second by Bucher and Saenger (1989, 1991, 1994), further developed by Digby et al. (1999). A third, developed by Eyre (1998), was based on hydrology, dividing the country’s estuaries into the following five groups: Mediterranean, Temperate, Transitional, Arid Tropical and Subtropical, and Wet and Dry Tropical and Subtropical.

Roy et al.’s (2001) estuary classification scheme was based on the geology and morphology of each estuary. According to Roy et al. (2001), estuary type was considered as an important variable in site selection because different estuary types:

- reflect differing levels of habitat maturity (i.e. different degrees of sediment infilling, which influences depositional environments and thus habitat characteristics);
- have characteristic substratum conditions and hydrological regimes;
- contain habitat types which, within the varying estuaries, have characteristic assemblages of species, that can be assigned a value in terms of species richness or productivity;
- contain habitat assemblages which respond to the nature of its estuary’s evolutionary sequence;
• contain estuarine habitats that are vulnerable to anthropogenic activity and intrusion, that can subsequently affect water and sediment quality, thus possibly reducing the habitat’s functionality.

Seven main estuary types on the NSW coast were classified by Roy et al. (2001) (Table 2.1), based on (a) the inheritance of different coastal settings, which create distinct estuary types, and (b) differing rates of infilling that determine how far along their evolutionary continuum the present-day estuaries have progressed. The estuary types were grouped as follows:

Table 2.1: Roy et al.’s (2001) grouping of estuary types in eastern Australia.

<table>
<thead>
<tr>
<th>Group</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>I</td>
<td>Semi-enclosed bays, characterised by marine waters with little fresh water inflow, are transitional between true estuarine environments (Groups II-IV) and the coastal ocean.</td>
</tr>
<tr>
<td>II</td>
<td>Tide-dominated estuaries, that have the largest entrances, and tidal ranges similar to the open ocean, and are typified by Boyd et al.’s (1992) and Chappell and Woodroffe’s (1994) funnel-shaped macro-tidal estuaries from coasts with large tidal ranges, but also includes estuaries in micro-tidal estuaries, in which tides are relatively more important than waves in moving water and sediment. Drowned river valleys (e.g. the Hawkesbury River) and tidal basins (e.g. Moreton Bay) are included in this category.</td>
</tr>
<tr>
<td>III</td>
<td>Wave-dominated estuaries, whose tidal inlets are constricted by wave-deposited beach sand and flood tide deltas. Barrier estuaries (e.g. the Hunter River), barrier lagoons, and interbarrier estuaries (e.g. Tilligerry Creek in Port Stephens) are included in this category.</td>
</tr>
<tr>
<td>IV</td>
<td>Intermittent estuaries, which become isolated from the sea for periods of time, due to a combination of climatic and other reasons. Saline coastal lagoons (e.g. Dee Why Lagoon) and small coastal creeks are included in this category.</td>
</tr>
<tr>
<td>V</td>
<td>Fresh water estuaries, which include coastal water bodies that rarely, if ever, are brackish, but occasionally have some linkages to the sea.</td>
</tr>
</tbody>
</table>

This classification scheme was further elucidated by Roy et al. (2001) (Table 2.2).
Table 2.2: Roy et al.’s (2001) modified estuary classification scheme.

<table>
<thead>
<tr>
<th>Estuary Group</th>
<th>Estuary Type</th>
<th>Example</th>
<th>Mature Form</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. bays</td>
<td>1. ocean embayment</td>
<td>Botany Bay, NSW</td>
<td></td>
</tr>
<tr>
<td>II. tide-dominated estuaries</td>
<td>2. funnel-shaped macrotidal estuary</td>
<td>South Alligator River, NT</td>
<td>tidal estuary</td>
</tr>
<tr>
<td></td>
<td>3. drowned valley estuary</td>
<td>Hawkesbury River, NSW</td>
<td></td>
</tr>
<tr>
<td></td>
<td>4. tidal basin</td>
<td>Moreton Bay, Qld</td>
<td></td>
</tr>
<tr>
<td>III. wave-dominated estuaries</td>
<td>5. barrier estuary</td>
<td>Lake Macquarie, NSW</td>
<td>riverine estuary</td>
</tr>
<tr>
<td></td>
<td>6. barrier lagoon</td>
<td>The Broadwater/South Stradbroke Island, Qld</td>
<td></td>
</tr>
<tr>
<td></td>
<td>7. interbarrier estuary</td>
<td>Tilligerry Creek, NSW</td>
<td></td>
</tr>
<tr>
<td>IV. intermittent estuaries</td>
<td>8. saline coastal lagoon</td>
<td>Smiths Lake, NSW</td>
<td>saline creek</td>
</tr>
<tr>
<td></td>
<td>9. small coastal creek</td>
<td>Harbord Lagoon, Sydney, NSW</td>
<td></td>
</tr>
<tr>
<td></td>
<td>10. evaporative lagoon</td>
<td>The Coorong, SA</td>
<td></td>
</tr>
<tr>
<td>V. fresh water bodies</td>
<td>11. brackish barrier lake</td>
<td>Myall Lakes, NSW</td>
<td>terrestrial swamp</td>
</tr>
<tr>
<td></td>
<td>12. perched dune lake</td>
<td>Lake Hiawatha</td>
<td></td>
</tr>
<tr>
<td></td>
<td>13. backswamp</td>
<td>Everlasting Swamp, Clarence River, NSW</td>
<td></td>
</tr>
</tbody>
</table>

The sites chosen for this study were comprised of coastal embayments, drowned river valleys, and barrier estuaries.

Roy et al. (2001) defined coastal embayments as semi-enclosed bays that feature bodies of marine water partially enclosed by arms of mainland coast that provide some protection from the action of ocean waves. Examples in NSW include Broken Bay, Botany Bay, Bate Bay, Jervis Bay, Twofold Bay and Disaster Bay.

Drowned river valley estuaries were defined as palaeo-valleys occupied by drowned river valley estuaries which are deep, usually narrow with steep rocky sides (Roy, 1984, 1994; Roy et al., 2001). Examples in NSW include Port Stephens, the Hawkesbury River, Sydney Harbour (Port Jackson), and Port Hacking.

Barrier estuaries are characterised by narrow, elongated entrance channels within broad tidal and backbarrier sand flats that are part of the adjacent bay barrier. Mature barrier estuaries are infilled with so much sediment that they form riverine estuaries (Roy et al., 2001). Examples include the Clarence and Richmond Valleys, the Hunter River, Lake Macquarie, Tuggerah Lakes, Lake
Illawarra, the Shoalhaven River, and Wallis Lake.

Digby et al.’s (1999, p. 1) estuary classification scheme was based on quantifiable biologically important physical characteristics (Table 2.3).

Table 2.3: Digby et al.’s (1999, p. 1) criteria used for the classification of estuaries.

<table>
<thead>
<tr>
<th>Classification Type</th>
<th>Criteria</th>
</tr>
</thead>
<tbody>
<tr>
<td>Geomorphology</td>
<td>basin morphology and relief, catchment geology, basin origin, sediment dynamics and wave climate</td>
</tr>
<tr>
<td>Evolutionary stage</td>
<td>stage of infilling, channel development</td>
</tr>
<tr>
<td>Hydrological processes</td>
<td>freshwater input, tidal range and prism</td>
</tr>
<tr>
<td>Climate</td>
<td>annual rainfall, runoff, evaporation, seasonal variation, interannual variation</td>
</tr>
<tr>
<td>Water quality</td>
<td>suspended sediment, dissolved oxygen, nutrient status, heavy metal pollution, bacterial contamination, water temperature</td>
</tr>
<tr>
<td>Habitat</td>
<td>intertidal communities (mangrove, saltmarsh, seagrass etc.), intertidal area, substrate types</td>
</tr>
<tr>
<td>Land use</td>
<td>degree of disturbance (i.e. pristine vs. developed), types of landuse in catchment</td>
</tr>
<tr>
<td>Aesthetic</td>
<td>degree of ‘naturalness’ (or disturbance), water quality, exotic weed infestations</td>
</tr>
</tbody>
</table>

Both Roy et al.’s (2001) and Digby et al.’s (1999) schemes were taken into consideration during the site selection process. Nine wetlands were subsequently chosen for the various analyses of this study (Table 2.4).
Table 2.4: Coastal wetland habitats chosen for habitat dynamics assessment.

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Site</th>
<th>Estuary Group</th>
<th>Estuary Type</th>
<th>Morpho-Hydrological Category</th>
</tr>
</thead>
<tbody>
<tr>
<td>Botany Bay</td>
<td>Towra Point</td>
<td>Type I</td>
<td>coastal embayment</td>
<td>• (BB) single constricted mouth, unbranched bay with off-channel embayment (SCUBE)</td>
</tr>
<tr>
<td>Jervis Bay</td>
<td>Currambene Creek &amp; Carama Inlet (which themselves are a tributary rivers &amp; inlet, respectively)</td>
<td></td>
<td></td>
<td>• (JB) single unconstricted mouth, unbranched bay with no off-channel embayment (SUUBN)</td>
</tr>
<tr>
<td>Port Stephens</td>
<td>Tilligerry Creek</td>
<td>Type II</td>
<td>drowned river valley (tide dominated)</td>
<td>• (PS) single constricted mouth, branched bay with off-channel embayment (SCBBE)</td>
</tr>
<tr>
<td>Hawkesbury River</td>
<td>Courangra Point</td>
<td></td>
<td></td>
<td>• (HR) single unconstricted mouth, branched bay with off-channel embayment (SUBBE)</td>
</tr>
<tr>
<td>Pittwater</td>
<td>Careel Bay</td>
<td></td>
<td></td>
<td>• (P) single unconstricted mouth, branched channel with off-channel embayment (SUBCE)</td>
</tr>
<tr>
<td>Tweed River</td>
<td>Ukerebagh Island</td>
<td>Type III</td>
<td>barrier estuary with open entrance (wave dominated)</td>
<td>• (TR) single constricted mouth, branched channel with off-channel embayment (SCBCE)</td>
</tr>
<tr>
<td>Lake Macquarie</td>
<td>Black Neds Bay</td>
<td></td>
<td></td>
<td>• (LM) single constricted mouth, unbranched bay with off-channel embayment (SCUBE)</td>
</tr>
<tr>
<td>Brisbane Water</td>
<td>Pelican, Rileys &amp; St Huberts Islands</td>
<td></td>
<td></td>
<td>• (BW) single unconstricted mouth, branched channel with off-channel embayment (SUBCE)</td>
</tr>
</tbody>
</table>

A comparison of some of the main attributes of each site is provided in Table 2.5 (after DLWC, pers. comm.).

**Bioregions**

Whilst the NSW coast can be categorised according to estuary classification, it can also be classified according to bioregions. Two related classification schemes have been developed for the continent of Australia to reflect bioregional geographic differences. The first represents terrestrial systems, the second coastal systems. Since the habitats studied in this research are located
within the coastal zone, it is important to incorporate elements of both schemes.

An Interim Biogeographic Regionalisation for Australia (IBRA) (Thackway and Cresswell, 1995) was developed by the States and Territories as a framework for setting priorities in the National Reserves System Cooperative Program. A biographic region was defined by Thackway and Cresswell (1995) as “a complex land area composed of a cluster of interacting ecosystems that are repeated in similar form throughout. Region descriptions seek to describe the dominant landscape scale attributes of climate, lithology, geology, landforms and vegetation. Biogeographic regions vary in size with larger regions found where areas have more subdued terrain and arid and semi-arid climates.”

Eighty IBRA bioregions have been classified for Australia. Three regions apply to the coast of NSW (Figure 2.1): the NSW North Coast (NNC), the Sydney Basin (SB), and the South East Corner (SEC) (Thackway and Cresswell, 1995). Sites in this study were located in the first two categories.

![Figure 2.1: NSW IBRA bioregions (Thackway and Cresswell, 1995).](image)

The NSW North Coast bioregion is characterised by coastal plains and sand dunes, and inhabited by wetlands, heaths, and *Eucalypt* forests. It is composed
primarily of natural ecosystems that coexist with many industries, including agriculture, cropping, forest timber production and harvesting, grazing, pastoral activities, horticulture, mining, tourism, as well as varying degrees of urbanisation. The Sydney Basin’s geomorphology is characterised by Mesozoic sandstones and shales; dissected plateaus; forests, woodlands and heaths; skeletal soils, sands and podsolic soils. It also consists of natural ecosystems that compete with various industries, but also with the State’s largest cities, such as Sydney, Newcastle and Wollongong.

The second classification scheme is the Interim Marine and Coastal Regionalisation for Australia (Thackway and Cresswell, 1998), which is an ecosystem-based classification for these environments. Within the context of this classification scheme, a biogeographic region was defined as “a complex area (land/sea) composed of a cluster of interacting ecosystems that are repeated in similar form throughout. Region descriptions seek to describe the dominant land/sea scape in terms of a hierarchy of interacting biophysical attributes. Biogeographic regions vary in size, with larger regions found where areas have more subdued environmental gradients. These are defined and delineated at the meso-scale” (Thackway and Cresswell, 1998).

The NSW coast is comprised of four of these categories: the Tweed-Moreton Region (TMN), the Manning Shelf (MAN), the Hawkesbury Shelf (HAW), and the Batemans Shelf (BAT). The study sites in the research are located in the Tweed-Moreton Region, the Hawkesbury Shelf and the Batemans Shelf (Figure 2.2).
The Tweed-Moreton Region consists of subtropical coastal and estuarine waters from just north of Baffle Creek south to approximately Nambucca Heads. Most of its sediments are of a terrestrial origin, with large estuaries having formed behind sand islands. It is subject to a rainfall regime of between 1400 and 2000 mm a year. The Hawkesbury Shelf is also a warm temperate region, and is comprised of that area from Stockton south to Shellharbour. It is characterised primarily by drowned river valley estuaries. The Batemans Shelf is comprised of the area from Shellharbour south to Tathra. It is a moist cool temperate region and is dominated by smaller coastal lagoon estuaries (Thackway and Cresswell, 1998).

There is a relationship between estuary type and bioregion. The NSW coast can be more simply divided into three sections (Saintilan, 2001, unpublished):
- the northern bioregion, which is characterised by large, infilled river-dominated estuaries (mature riverine barrier estuaries). River salinities are dependent on freshwater delivery, and mangrove settings include point bars, back barrier settings (highly diverse in mangrove flora) and cut-off
embayments (Broadwaters). This region is subject to a subtropical climate, and seasonal rainfall.

- the central bioregion, which is dominated by drowned river valleys and large lagoon-type barrier estuaries. Its estuarine geomorphology is characterised by flood-tide deltas, tributary bay-head deltas, point bars on the main fluvial delta, and highly dynamic channel-fringing environments. It is subject to a temperate climate, with little seasonality in rainfall.
- the southern bioregion, which is dominated by small barrier estuaries, which are often intermittently open. Mangroves have colonised small bay-head deltas and sandy marine deltas.

2.1.2 Rainfall

Rainfall is not spatially or temporally uniform along the NSW coast. The 2000 NSW State of the Environment Report (EPA, 2000) made some observations of the pattern of rainfall along the NSW coast:

- the North Coast region receives an average of 1200 millimetres of rainfall a year, whilst the Central Coast averages 900mm and the South Coast 800mm;
- the northern region receives most rainfall in summer; and
- the central and southeastern regions have rainfall distributed evenly throughout the year.

Since freshwater delivery has been shown in some studies as being important in mangrove colonisation (Chapter 6), the assessment of rainfall patterns is important in determining if there is a relationship between habitat dynamics and changes in rainfall regimes. Site selection was guided by the intent to study habitats that have been subject to differing spatial and temporal trends in rainfall regimes. This was based on data from the Bureau of Meteorology (Figures 2.3 to 2.10).
Figure 2.3: Average annual rainfall at Tweed Heads (Tweed Heads Golf Club), 1887 to 1997 (data from the Bureau of Meteorology).

Figure 2.4: Average annual rainfall in the Tilligerry Creek region (Williamtown), 1944 to 1998 (data from the Bureau of Meteorology).
Figure 2.5: Average annual rainfall in the Black Neds Bay area (Newcastle), 1862 to 1998 (data from the Bureau of Meteorology).

Figure 2.6: Average annual rainfall in the Brisbane Water area (Gosford), 1916 to 1998 (data from the Bureau of Meteorology).
Figure 2.7: Average annual rainfall in the Courangra Point area (Wisemans Ferry), 1903 to 1998 (data from the Bureau of Meteorology).

Figure 2.8: Average annual rainfall in Pittwater (data from the Bureau of Meteorology). Rainfall History for Newport (1932 to 1992) and Avalon (1994 to 1996).
Figure 2.9: Average annual rainfall in the Towra Point area (Botany Bay, Sydney Airport), 1930 to 1998 (data from the Bureau of Meteorology).

Figure 2.10: Average annual rainfall in Jervis Bay, 1899 to 1998 (based on Bureau of Meteorology data).

Figures 2.3 to 2.10 depict the varying spatial and temporal nature of rainfall at each of the study sites.
2.1.3 Population Change

The majority of the population of NSW lives in the coastal zone. NSW has a population of approximately 6.5 million people, 20% of whom live in the Sydney Statistical Division on the eastern seaboard. The majority of the rest of the population lives in the coastal statistical divisions of Hunter, Illawarra, Richmond-Tweed, Mid North Coast and South Eastern. When combined with the Sydney division, these areas account for 14.7% of the NSW’s area, yet 88% of its population (ABS, 2000a). This places enormous pressure on the coastal zone of NSW and its resources.

The NSW coast has a history of increasing population levels, which are the result of people moving from cities to regional coastal towns. Much of this increase has been concentrated north of Sydney (Figure 2.11). Whilst the State’s average rate of population growth was 1.18% from 1976 to 1991, coastal statistical divisions such as Sydney, the Richmond-Tweed, and the Mid North Coast experienced rates of increase of approximately 4% during the 1970s.

The sites chosen for examination in this study all reflect increased population growth during the study period, yet to differing degrees (Figures 2.12 to 2.18), with the exception of Cararma Inlet, which served as a control site subject to virtually zero population growth. The average annual growth rate, \( r \), is calculated as a percentage using the formula:

\[
\left( \frac{P_n}{P_0} \right)^{1/n} - 1 \times 100
\]

where \( P_0 \) is the population at the start of the period, \( P_n \) is the population at the end of the period and \( n \) is the length of the period between \( P_n \) and \( P_0 \) in years (ABS, 1995).

The data provided in Figures 2.12 to 2.18 reflect only the population data for the statistical local government areas surrounding each site, since it is impossible to collect data on a catchment basis. Based on this, data for Courangra Point, on the Hawkesbury River, was not provided, since it was not possible to collect representative data.
Figure 2.11: Rates of population change, coastal statistical divisions, 1971 to 1998 (reproduced from EPA, NSW, 2000; incorporating data from ABS, 2000b).

Figure 2.12: Population levels in the Tweed Shire Council local government area, 1947 to 1999 (data from the Australian Bureau of Statistics).
Figure 2.13: Population levels in the Port Stephens local government area, 1947 to 1999 (data from the Australian Bureau of Statistics).

Figure 2.14: Population levels in the Lake Macquarie local government area, 1947 to 1999 (data from the Australian Bureau of Statistics).
Figure 2.15: Population levels in the Gosford local government area, 1947 to 1999 (data from the Australian Bureau of Statistics).

Figure 2.16: Population levels in the Pittwater area, 1947 to 1999 (data from the Australian Bureau of Statistics).
Figure 2.17: Population levels in the Botany (1947 to 1999) and Sutherland (1981 to 1999) local government areas (data from the Australian Bureau of Statistics).

Figure 2.18: Population levels in the Shoalhaven local government area (1947 to 1998) (data from the Australian Bureau of Statistics).

Again, these graphs indicate the population levels of local government statistical divisions, and not the catchments in which the sites are located.
Catchment Landuse

Catchment landuse can greatly affect coastal wetland habitat dynamics. For example, a pristine catchment is expected to be less affected by catchment landuse than a catchment in a heavily urbanised catchment like those in the Sydney Metropolitan area. The EPA (EPA, NSW, 2000) provided examples of both a natural catchment and an anthropogenically-influenced catchment, to show how each is impacted (Figures 2.19 and 2.20).

In urbanised areas, the catchment has been cleared, increasing sediment availability for delivery to the estuary. Urbanised areas also consist of more area of smooth surfaces than a natural catchment, which increases the accessibility and speed of delivery of sediment and runoff to the estuary (Leopold et al., 1964). Such point and non-point pollution can affect the functioning of a wetland, and hence its habitats.

Landuse is intrinsically linked to this issue. A natural catchment will host a more natural wetland than a catchment modified by urbanisation, or for agricultural purposes. These latter landuses can produce by-products which can affect habitat dynamics, such as increased nutrient input to the waterway, and consequently to the wetland.

The EPA (EPA, NSW, 2000) listed the main causes of biodiversity loss in marine and estuarine environments, which can also be inferred as affecting coastal wetland habitat dynamics: coastal development, pollution, unsustainable fisheries, shipping and boating activity, mineral and petroleum activities, and introduced species.

These pressures were translated into the following threatening processes (EPA, NSW, 2000): dredging and reclamation, waterfront developments (e.g. jetties, marinas, breakwalls), dams, weirs and structural flood mitigation works, bridges, roads, causeways and culverts, water pollution (e.g. pesticides, turbidity, thermal pollution, high nutrients, acidity), management of intermittently opening coastal lagoons, damage to or removal of marine and
Figure 2.19: The hydrologic cycle in a natural catchment (reproduced from EPA, NSW, 2000).

Figure 2.20: The hydrologic cycle in an anthropogenically-influenced catchment (reproduced from EPA, NSW, 2000).
estuarine habitats (e.g. mangroves and saltmarsh), and introduced and translocated species.

Catchment landuse also incorporates engineering works, which can considerably affect the evolution of an estuary and its habitats. For example, the evolution and dynamics of an estuary with a trained entrance, or groynes, or which has been subject to dredging works, can be quite different to that of one not modified. Some estuaries in this study have been subject to no engineering works whatsoever, whilst others have.

Incorporated into the site selection of Table 4.6 was a reflection of catchment condition, as assessed by the Australian Geological Survey Organisation (Heap et al., 2001). This assessment was a reflection of that percentage of the catchment’s natural cover that remained intact, defined in Appendix 1, Table A1.1.
2.2 Site Selection and Justification

Site selection was based on the previous considerations, the primary one being estuary type (Table 2.5). The sites are shown in Figure 2.21.

Figure 2.21: Study site selection along the New South Wales coast (the Brisbane Water and Pittwater catchments are incorporated in the Hawkesbury-Nepean River Catchment).

Table 2.5: Site selection for this study (following page). Data collected from the NSW Department of Land and Water Conservation (pers. comm.); the Bureau of Meteorology; and the Australian Bureau of Statistics.
<table>
<thead>
<tr>
<th>Wetland Site</th>
<th>Geomorphic Unit</th>
<th>Estuary</th>
<th>Estuary Type</th>
<th>Estuary Catchment Area</th>
<th>Estuary Waterway Area</th>
<th>Bioregions</th>
<th>Rainfall (mm)</th>
<th>Rate of Population Change (1947-1997)</th>
<th>Catchment Condition &amp; Remaining Natural Cover (%) (&lt;1999)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ukerebagh Island</td>
<td>marine delta</td>
<td>Tweed River</td>
<td>river with tributary lakes</td>
<td>1114 km²</td>
<td>23 km²</td>
<td>IBRA: NSW North Coast IMCRA: Tweed - Moreton Shelf</td>
<td>MEAN: 1685 min: 689 max: 3193</td>
<td>3.18% modified &lt; 65%</td>
<td></td>
</tr>
<tr>
<td>Tilligerry Creek</td>
<td>interbarrier depression</td>
<td>Port Stephens</td>
<td>barrier estuary in a drowned river valley</td>
<td>4950 km² TC: 82 km²</td>
<td>166 km² TC: 9 km²</td>
<td>IBRA: Sydney Basin IMCRA: Hawkesbury Shelf</td>
<td>MEAN: 1120 min: 541 max: 1793</td>
<td>4.42% largely unmodified &lt; 65-90%</td>
<td></td>
</tr>
<tr>
<td>Black Neds Bay</td>
<td>tidal inlet</td>
<td>Lake Macquarie</td>
<td>lake</td>
<td>700 km²</td>
<td>120 km²</td>
<td>IBRA: Sydney Basin IMCRA: Hawkesbury Shelf</td>
<td>MEAN: 1136 min: 597 max: 1919</td>
<td>2.87% severely modified &lt; 35%</td>
<td></td>
</tr>
<tr>
<td>Pelican, Rileys &amp; St Huberts Islands</td>
<td>flood tide delta</td>
<td>Brisbane Water</td>
<td>lake</td>
<td>170 km²</td>
<td>27.2 km²</td>
<td>IBRA: Sydney Basin IMCRA: Hawkesbury Shelf</td>
<td>MEAN: 1324 min: 630 max: 2232</td>
<td>4.21% severely modified &lt; 35%</td>
<td></td>
</tr>
<tr>
<td>Courangra Point</td>
<td>fluvial point bar</td>
<td>Hawkesbury River</td>
<td>drowned river valley</td>
<td>21500 km²</td>
<td>100 km²</td>
<td>IBRA: Sydney Basin IMCRA: Hawkesbury Shelf</td>
<td>MEAN: 845 min: 437 max: 1498</td>
<td>severely modified &lt; 35%</td>
<td></td>
</tr>
<tr>
<td>Careel Bay</td>
<td>tributary bay head delta</td>
<td>Pittwater</td>
<td>drowned river valley</td>
<td>P: 77 km² CB: 10.2 km²</td>
<td>17.3 km²</td>
<td>IBRA: Sydney Basin IMCRA: Hawkesbury Shelf</td>
<td>MEAN: 1212 min: 573 max: 2190</td>
<td>3.46% modified &lt; 65%</td>
<td></td>
</tr>
<tr>
<td>Towra Point</td>
<td>off-channel, back barrier embayment</td>
<td>Botany Bay</td>
<td>enclosed embayment</td>
<td>1100 km²</td>
<td>80 km²</td>
<td>IBRA: Sydney Basin IMCRA: Hawkesbury Shelf</td>
<td>MEAN: 1107 min: 523 max: 2025</td>
<td>severely modified &lt; 35%</td>
<td></td>
</tr>
<tr>
<td>Currambene Creek &amp; Carama Inlet</td>
<td>fluvial point bar, tidal inlet</td>
<td>Jervis Bay</td>
<td>semi-sheltered embayment with tributary creeks</td>
<td>410 km² CC: 165 km² CI: 25 km²</td>
<td>102 km² CC: 1.2 km² CC: 0.5 km²</td>
<td>IBRA: Sydney Basin IMCRA: Batemans Shelf</td>
<td>MEAN: 1104 min: 463 max: 2053</td>
<td>3.69% (Shoalhaven area; CI has no pop.) (1947-1999)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>CC: largely unmodified &lt; 65-90% CI: near pristine &gt; 90%</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
2.3 Site Descriptions

2.3.1 Ukerebagh Island, Tweed River

Ukerebagh Island is located in the Tweed River on the far north coast of NSW (153º33’E, 28º10’S) (Figure 2.22). It is a marine delta composed of silts and clays. Mangroves dominate vegetation on the island, and large areas of dredge spoil have been colonised by terrestrial species. There are a number of discrete saltmarsh habitats.

Figure 2.22: Ukerebagh Island in the mouth of the Tweed River.

The Tweed River incorporates the main river channel and two large cut-off embayments, Terranora and Cobaki Broadwaters, which join the river at Terranora Inlet. The tidal limit of the river is at approximately the township of Murwillumbah, approximately 26 km upstream of the mouth.
The Tweed River has a relatively small and compact catchment of 1,114 km\(^2\) (the Tweed Shire itself has an area of 1303 km\(^2\); Tweed Shire Council, 1998a). The catchment is largely comprised of alluvial floodplains and surrounded by three mountain ranges, the McPherson Range in the north, the Tweed Range in the west, and the Nightcap Range in the south, some of which are the remains of a Tertiary-age shield volcano (NSW National Parks and Wildlife Service, 1999, p. 5). The NSW Department of Public Works (1991) noted that the small size of the catchment makes it more susceptible to urban and catchment pressures and even more liable to shoaling from marine sands transported to the river mouth by wave action. Its IBRA classification is the NSW North Coast, and its IMCRA classification locates it on the Tweed-Moreton Shelf.

Ukerebagh Island itself is a designated Historic Site and Nature Reserve, of particular significance to the local Goori people. The Minjungbal Trading Company and the NSW National Parks and Wildlife Service manage the island cooperatively. The Ukerebagh Island Nature Reserve consists of a total area of 125 hectares (NSW NPWS, 1999).

**Climate**

The Tweed River catchment is subject to a humid, sub-tropical climate, which is characterised by an average annual rainfall ranging from 689 mm to 3193 mm between 1887 and 1997, and a mean annual rainfall during that time of 1685 mm (Figure 2.3).

**Geomorphology**

The Tweed River has undergone a series of morphological changes, both natural and human-induced (Tweed River Management Plan Advisory Committee, 1998b). These have particularly been focussed in the lower estuary, within which Ukerebagh Island lies. The estuary works of the past 100 years have served to alter the tidal range, predominantly in the lower estuary, which was subject to a rise of 16 cm in mean spring tide between 1960 and 1991 (Druery and Curedale, 1979). A primary mode of contemporary morphological change
has been bank erosion, which threatens both public and private land, including the northern shore of Ukerebagh Island. Much of this bank erosion is the result of short period waves created by boating activity and increased tidal range.

Druery and Curedale (1979) mapped the sediment of the lower Tweed floodplain, during which it was found that the Tweed River flows through a dual barrier system of marine sands. Ukerebagh Island lies between a Pleistocene barrier which itself lies between Chinderah and the eastern shoreline of Terranora and Cobaki Broadwaters, and the Holocene Outer Barrier of the active beaches, dune ridges and marine delta sand flats of the present day shoreline. Their study determined that behind this dual barrier system, the Tweed River has in fact filled its palaeo estuarine basin and is now prograding across the dual barrier as a low gradient, leveed channel and floodplain. The floodplain itself consists of a thin mantle of alluvial deposits on top of estuarine clays and muds (i.e. estuarine basin infill).

Extensive dredging of the river and its tributaries has further confounded the natural changes in the shoals of the river and its entrance condition. Druery and Curedale (1979) determined that the mouth of river is not in its historical position, and that movements in the mouth, increased sediment input, and the disruption of dredging practices have all served to constantly change the river’s natural regime. Since the river has not yet reached a mature state of geomorphology, more changes can be expected.

Ukerebagh Island has been subject to large-scale dredge spoil deposits (marine sand). This is a consequence of dredging of the river and its tributaries in the late 1950s and early 1960s. Areas of higher ground are most likely due to sand deposition from dredging works (Druery and Curedale, 1979). Substantial amounts of marine sand have been deposited on the northwestern tip of the island, the northern shore and the northeastern tip, which terrestrial species such as *Casuarina glauca* have colonised.

The NSW National Parks and Wildlife Service (1999, p. 8) noted that rubble walls on parts of the eastern shore of the island have partially collapsed, and
foreshore erosion has been occurring in several places, particularly on the mideastern shore of the island. The long-term erosion of the northern foreshore of the island has resulted in the recession of the dredge spoils on the island (Patterson Britton & Partners, 1997, pp. 11-12). A further indication of the long-term erosion of the northeastern foreshore of the island was the exposure of a peat layer in the late 1990s that was full of mangrove rootlets and dead mangrove pneumatophores (Patterson Britton & Partners, 1997, pp. 20-21).

**Estuarine Habitats**

Ukerebagh Island represents just one portion of the estuarine habitats of the Tweed River system. Three studies of estuarine vegetation in the system produced slightly differing results. In its Lower Tweed Estuary Management Plan, the NSW Department of Public Works (1991) reported that mangroves occupied approximately 300 ha in the Tweed River estuary, with saltmarsh occupying 25 ha. A study in the early 1980s by West *et al.* (1985) reported that mangroves occupy approximately 309 ha in the Tweed River estuary, with saltmarsh occupying 21.3 ha and seagrass 33.1 ha.

The mangrove habitats of the area consist primarily of two species, *Rhizophora stylosa* and *Avicennia marina*, with some occurrences of *Aegiceras corniculatum*, *Exoecaria agallocha*, *Bruguiera gymnorrhiza*, and *Ceriops tagal var. australis* (Patterson Britton & Partners, 1997; Saintilan, 1997b, 1998). The mangrove stands on Ukerebagh Island are of biogeographic significance as they are the only mangrove stand in NSW to contain all five species of mangroves (Pressey and Griffith, 1987).

The saltmarsh communities on Ukerebagh Island are composed primarily of *Sporobolus virginicus*, *Sarcocornia quinqueflora*, *Baumea juncea* (R. Br.) Palla and *Juncus kraussii* (Patterson Britton & Partners, 1997), inhabiting parts of both the southeastern and northwestern quarters of the island.

Terrestrial glycophytes are located almost completely on the eastern and northeastern shores and also the northwestern shore, and appear to inhabit
former dredge spoil sites where waste material dredged from the surrounding watercourses was probably deposited during previous decades (Druery and Curedale, 1979; NSW NPWS, 1999). These include fringing open swamp forest, open forest and rainforest.

**Population**

The Tweed Shire is one of the fastest growing local government areas in NSW, with a growth rate of 3.18% during the period 1947 to 1997, well above the average growth rate for NSW of 1.18% for that period (NSW Department of Public Works, 1991) (Figure 2.12). In 1998 its population registered at 68,816 people.

**Catchment Modifications and Land Use**

The river has been substantially impacted since European settlement approximately 150 years ago. This has been the result of extensive land clearing for rural and urban purposes, and of extensive dredging and other engineering works in the river itself. Pressey and Griffith (1987) calculated that 90% of the coastal lowland vegetation in the coastal lowland of Tweed Shire had been cleared for urban development, leaving only 10% remnant vegetation.

The catchment was defined by Heap *et al.* (2001) as modified (< 65% remaining natural cover). The largely rural nature of catchment landuses has contributed to extensive point and non-point pollution of the waterway. The NSW Department of Public Works (1991) identified a number of catchment-sourced pollutants to the Tweed River. These included pesticides and fertiliser residues having originated from the sugar can industry in the upper catchment, effluent from sewage treatment plants, industrial pollution from industries on the banks of the river, contaminated runoff associated with flooding, and fine sediments and nutrients from urban stormwater runoff.

The Tweed River has undergone extensive change at the hands of engineers since its European colonisation. These practices began in earnest in the late
1870s with dredging of the river to allow steamers and watercraft to safely navigate the waters around the growing shoals. In order to improve the potential of the river as a shipping port, a training wall (entrance breakwater) was constructed at the river mouth in 1904. From 1960 to the present time this training wall has been continually maintained, and is now subject to the Tweed River Sand Bypassing Project. Changes in the entrance conditions of the estuary have affected the fluvial dynamics of the river, and the channels surrounding Ukerebagh Island. This has been exhibited in the form of new accretion and erosion patterns in the channels surrounding the island. Consequently, the island has been subject to bank erosion and undercutting.

### 2.3.2 Tilligerry Creek, Port Stephens

Port Stephens (152°12′E, 32°43′S) (within which Tilligerry Creek is located) is a wave-dominated drowned river valley with a catchment of 4,950 km², a waterway area of 166 km², and its entrance is open and untrained. It incorporates the tidal waters and intertidal wetlands of the Port Stephens estuary, the Myall River to Myall Lakes National Park, the Karuah River to Karuah, and all of Tilligerry Creek and Twelve Mile Creek. Port Stephens is the largest estuarine waterway in NSW, comprised of 23.26 km² of mangroves and 7.719 km² of saltmarsh (West et al., 1985). Port Stephens is located in the Sydney Basin IBRA category, and the Hawkesbury Shelf IMCRA category.

Tilligerry Creek itself is a riverine tributary of the drowned river valley of Port Stephens (Figure 2.23). It has a catchment area of 82 km², a waterway area of 9 km², is 22 km in length, its tidal reaches maintain a width of 500 to 900 metres, and its entrance is open and untrained. The creek enters the mid-southern shore of Port Stephens between the towns of Soldiers Point and Lemon Tree Passage (NSW Department of Land and Water Conservation, 2000a). The creek occupies the long narrow depression between the Pleistocene and Holocene sandy barriers of Newcastle Bight (Stockton Beach), and has very restricted water circulation. Its primary catchment landuse is agriculture and grazing (Manly Hydraulics Laboratory, 1999; Umwelt, 2000).
Mapping of the coastal wetland habitats in this study was limited to the area in the mouth of the creek only. The study site is bounded by Wallis Creek to the east, the northern border of Fenninghams Island, the land just to the west of Bobs Farm Creek, and the wetland boundary adjacent to the road joining Nelson Bay and Salt Ash to the south (Figure 2.23). This area incorporates Fenninghams Island Creek and Jessies Island. The area has been extensively channelised and flood gated, with drains running throughout the southeastern corner of the site.

**Climate**

Port Stephens has a slightly winter-dominated rainfall regime, with a minimum of 541 mm and a maximum of 1793 mm between 1942 and 1998. During that
same period, the average annual rainfall has been 1120 mm (Figure 2.4).

**Geomorphology**

The evolution of the Pleistocene inner barrier (now the Myall Lakes basin) 120,000 years ago during a period of global sea-level rise (and subsequent fall), followed by the creation of the Holocene outer barrier during the sea-level rise of the Holocene, have combined to create the depression which Tilligerry Creek occupies. During the past 10,000 years, muds have continued to accumulate throughout the estuary, creating extensive intertidal areas for colonisation by coastal wetland species (Manly Hydraulics Laboratory, 1999).

The estuarine sediments, which the wetland species have colonised, are comprised predominantly of quartzose sands and zones of sediment comprised of varying degrees of sand, muddy sands, sandy mud, shelly sands and clean sands. There is also an element of lithic sediment composition (Manly Hydraulics Laboratory, 1999). Manly Hydraulics Laboratory (1999) contended that the tidal flats at the margins of Tilligerry Creek are actively forming where fluvial and tidal processes intermix, and are subsequently being colonised by mangrove and saltmarsh species.

**Estuarine Habitats**

The waterways of Port Stephens and Myall Lakes support the largest stand of mangroves in NSW and approximately 18% of the remaining saltmarsh in the state. One hundred and thirty-seven wetlands in the catchment are protected by SEPP 14 legislation, and some are listed in the *Directory of Important Wetlands in Australia* (Australian Nature Conservation Agency, 1996; Manly Hydraulics Laboratory, 1999).

*Avicennia marina* and *Aegiceras corniculatum* mangrove species are present in the area. The saltmarsh community is dominated by the herb *Paspalum paspalodes* (Michx.) Scribn., *Sarcocornia quinqueflora* and *Suaeda australis* (R. Br.) Moq. and there are also *Juncus kraussii* and *Sporobolus virginicus*
communities. Further landward of this is the swamp forest of *Casuarina glauca* and *Eucalyptus robusta* Sm., often with an understorey of *Juncus kraussii* and *Phragmites australis* (Cav.) Trin. (Manly Hydraulics Laboratory, 1999).

**Population**

The Port Stephens Local Government Area was subject to a growth rate of 4.42% per annum from 1947 to 1997 (Figure 2.13), with a population of 55,971 in 1999.

**Catchment Modifications and Land Use**

Occupation of the Port Stephens area began in 1816 when Governor Macquarie granted James Smith the right to cut and transport cedar to Sydney. The timber industry has remained active since this time, under the administration of State Forests. Rural grazing of cleared land began in 1826, following clearing of these areas by the timber industry. Townships are located throughout the catchment, and in recent years there has been a rapid expansion of urban and rural residential areas surrounding these townships. Settlement on the Tilligerry Peninsula did not begin until 1831 when a land grant of 50 acres was made to Caswell, and settlement of Lemon Tree Passage did not begin in earnest until the 1920s (Manly Hydraulics Laboratory, 1999).

Bucher and Saenger (1989) assessed the Port Stephens catchment as being moderately developed (25-50%). Further to this, Tilligerry Creek itself is largely unmodified (approximately 65-90% of the natural cover is remaining) (Heap *et al*., 2001). Approximately 65% of its catchment still contains native vegetation, most of which is open forest/woodland. Estuarine habitats within the Port Stephens catchment comprise only 3% of total vegetation. The remaining 35% of the catchment has been cleared for residential development, transport infrastructure, recreational facilities, agricultural activities (which accounts for the majority of the catchments’ landuse), commercial developments, industrial developments and military facilities (MHL, 1999a).
2.3.3 Black Neds Bay, Swansea Channel, Lake Macquarie

Lake Macquarie (151°40'E, 33°06'S) is the largest coastal lake in eastern Australia. It has a catchment of 700 km$^2$, a waterway of 120 km$^2$, and its entrance is open with twin training breakwaters (Lake Macquarie City Council, 1997a; NSW DLWC, 2000d). AGSO classified it as a severely modified wave-dominated estuary (AGSO, 2001c), separated from the Pacific Ocean by a breached narrow sand barrier. It incorporates nineteen sub-catchments draining directly to the Lake or to creeks entering the Lake, seven coastal catchments draining to the ocean, and one lagoon catchment (Lake Macquarie City Council, 1999a). Lake Macquarie is located in the Sydney Basin IBRA category, and the Hawkesbury Shelf IMCRA category.

Black Neds Bay is a tidal wetland composed of marine sand on the south side of Swansea Channel (Figure 2.24a, 2.24b), which itself is a tidal inlet that connects Lake Macquarie to the Pacific Ocean. Swansea Channel is 4 km in length and has a spring tidal range of 1.25 m at the entrance (NSW Department of Public Works, 1992).

The NSW Department of Public Works (1976) described Black Neds Bay as a marginal lagoon consisting of a shallow depression in medium to fine sand. It was once a section of the main waterway of Swansea Channel, but was cut off when the channel changed its meander pattern. Its southern and western shores have subsequently been stabilised by the encroachment of residential development, whilst its northern and eastern shores are characterised by the wetland containing mangrove and saltmarsh species.

Climate

Rainfall data for more than ten years was not available for Swansea, so data for Newcastle, the site’s closest regional centre, is provided for Black Neds Bay. For the period 1862 to 1998, Newcastle was subject to a minimum annual rainfall of 597 mm and a maximum of 1919 mm, and a mean of 1136 mm (Figure 2.5).
Figure 2.24a: Black Neds Bay, in Swansea Channel, the mouth of Lake Macquarie.

Figure 24b: Black Neds Bay.
The NSW Department of Public Works (1976) observed that Lake Macquarie and Swansea Channel are comparatively free from flooding due to rainfall, which they attributed to the fact that the ratio of the Lake Macquarie catchment area to the storage area of the waterway is low, and hence the effect of rainfall alone on the water level is minimal. Additionally, it was determined that the annual inflow of freshwater was less than 10% of the volume of the Lake.

**Geomorphology**

Lake Macquarie is the result of sea incursion during the Holocene sea-level rise of a coastal valley, and is now an embayment of an almost closed, low-lying sandy barrier estuary. Swansea Channel is its coastal lagoon entrance, and consists primarily of quartzose sand.

There have been some key sedimentation and erosion problems within the entrance channel to Lake Macquarie that have implications for the wetlands of Black Neds Bay. Shoaling is taking place in the bay, particularly in the western channel and at its entrance, and there is foreshore retreat of Salts Bay, the northern bank of Black Neds Bay, which is threatening the wetland (Lake Macquarie City Council, 1997b, 1997c).

From aerial photographs, the NSW Department of Public Works (1974) calculated that shoreline recession had accounted for the loss of approximately 8 ha of land in Black Neds Bay from 1959 to 1974. The Department later calculated (NSW Department of Public Works, 1976) that from 1940 to 1976 the shoreline had receded by up to 400 m. A further assessment was made by the NSW Department of Public Works in 1992, in which they assessed shoreline erosion in Salts Bay. They determined that at that time approximately 85% of the shoreline along Salts Bay was undergoing erosion of between 0.5 to 3.0 metres/year, and that the remaining 15% at the western end of Mats Point was undergoing accretion of up to 1.5 metres/year. The conclusion was that the combination of the sometimes-destructive forces generated by wind, waves and tidal currents had contributed to the erosion of Salt’s Bay. From aerial photographs, they estimated that approximately 8 hectares of land had been

Australian Water And Coastal Studies (1995) furthered this course of study, and predicted the increased continuation of erosion and foreshore recession in Salts Bay in response to sea-level rise. They predicted that a rise in sea-level of 0.3 m would result in a 15 m recession of the Salts Bay foreshore, which would result in a further reduction in the extent of saltmarsh in Black Neds Bay.

The entrance channel to the estuary has changed considerably since European settlement of the area. In the late 1800s entrance training works were completed, to alleviate the extensive sand spits and shoals of the natural entrance. AWACS (1995) suggested that this would have resulted in an increase in the tidal range and prism throughout the estuary, and increased wave penetration to Salts Bay.

In an attempt to stabilise Salts Bay, Lucy’s Groyne was constructed on the southern breakwater and a groyne was constructed approximately in the centre of the shoreline of Salts Bay, and a training wall was extended to Mats Point. In addition to this, approximately 10,000 m$^3$ of sand has been dredged from Black Neds Bay since 1959 and placed on the foreshore immediately to the east of Mats Point. The dredge spoils littered throughout Black Neds Bay are prominent in aerial photographs, and have since been colonised by mangroves. By July 1992, another 6000 m$^3$ of sand from the dredging works of Black Neds Bay Mooring Basin was placed on the foreshore dune of Salts Bay (NSW Department of Public Works, 1992; AWACS, 1995). A ventilation dredging project is currently underway, which may have adverse impacts on the wetlands of the bay.

**Estuarine Habitats**

Black Neds Bay is uniformly composed of the mangrove *Avicennia marina* (Fam. Verbenaceae) (AWACS, 1995), with patches of terrestrial vegetation in the middle of the two largest saltmarsh communities. The primary saltmarsh species are *Juncus kraussii* and *Scirpus nodosus* Rottb., which almost
completely fill two large fields. There are instances of small, scattered communities of *Sporobolus virginicus* and *Sarcocornia quinqueflora*. Within the saltmarsh zones there some ponds of water. Surprisingly, there were also instances of freshwater Buffalo grass. The northern sand dunes of the wetland have been colonised by bitou bush and coastal wattle.

*West et al.* (1985) approximated that Lake Macquarie as a whole consisted of 99.8 ha (0.998 km\(^2\)) of mangrove and 1339.1 ha (13.391 km\(^2\)) of saltmarsh. On the other hand, the Australian Geological Survey Organisation (2001a) reported that Lake Macquarie had 54 ha (0.54 km\(^2\)) of mangrove and 164 ha (1.64 km\(^2\)) of saltmarsh. These results are based on vastly different mapping schemes.

**Population**

Lake Macquarie, with a catchment area of 700 km\(^2\), had a population growth rate of 2.87% for the period 1947 to 1997 (Figure 2.14). Most of this has been within in urban areas, and in 1998 the total population was 180,826.

**Catchment Modifications and Land Use**

The Lake Macquarie catchment has been severely modified (< 35% catchment natural cover remaining) (*Heap et al.*, 2001), by a combination of residential and industrial development. Its major industries and commercial operations include power stations, a smelter, mining, extractive industries and light industries (*Lake Macquarie City Council*, 1999a).

The Council (*Lake Macquarie City Council*, 1999) delineated the catchment’s landuses, with 383 km\(^2\) being forested land, 97 km\(^2\) is urban area, and 126 km\(^2\) has either been cleared or classed as rural.

This increased urban landuse has been identified by the Council as involving the removal of vegetation cover, alteration of drainage lines, modification of runoff characteristics, and loss of soil through erosion. These have combined to create a serious sedimentation problem for the lake.
Although the Lake’s water quality has generally been good, nutrient and suspended solids concentrations have been observed to be increasing since the 1950s, with nutrient levels approaching the upper acceptable limits recommended by the Australian and New Zealand Environment and Conservation Council (1999). Some other water quality parameters have exceeded the recommended limits, including the mean concentration of orthophosphorus, mean chlorophyll $a$, and ammonia. These have the potential to impact on the wetland community.

In order to provide a water quality guideline for preparation of the Lake Macquarie Estuary Management Study, WBM (1997, cited in Lake Macquarie City Council, 1997c, p. 14) conducted water quality sampling and modelling of the rate of nutrient release from bed sediments in order to verify the accuracy of water quality modelling for the Lake. Conceptual modelling based on this data found that urban runoff was the largest source of nutrients to the lake, having contributed more than 80% although only accounting for 35% of the total catchment area.

### 2.3.4 Pelican, Rileys and St Huberts Islands, Brisbane Water

Brisbane Water (151°20'E, 33°32'S) was defined by NSW DLWC (2000b) as a broad, shallow lake, with depths of up to only 5 to 6 m in the main body of the estuary (Figure 2.25). On the other hand, AGSO (2000b) has classified Brisbane Water as a severely modified wave-dominated estuary. Its entrance is open and untrained, and enters into Broken Bay, at the mouth of the Hawkesbury River. Brisbane Water has a waterway area of 27.2 km$^2$ (2,768 ha), drains a catchment of 185 km$^2$, and has a number of significant estuarine tributaries, including Woy Woy Inlet, Narara Creek, The Broadwater, and Kincumber Broadwater (NSW DLWC, 2000b). The islands that are the subject of this study (Pelican, Rileys and St Huberts) occupy a flood tide delta composed of marine sand, situated between the estuary entrance and the Brisbane Water Broadwater.

The city of Gosford is located on its foreshore. Its IBRA classification places it
in the Sydney Basin, and its IMCRA classification places it on the Hawkesbury Shelf.

![Figure 2.25: The study islands in Brisbane Water.](image)

**Climate**

From 1916 to 1998, Gosford has been subject to a mean annual rainfall of 1324 mm, with a minimum of 630 mm, and a maximum of 2232 mm (Figure 2.6).

**Geomorphology**

Gosford City Council (1995) determined from comparisons of bathymetric data from 1901 and 1993 that there had been little change during the 92-year period, although some areas did have some accretion. These areas tended to be at the mouths of local creeks, and were often subsequently colonised by mangroves.
Estuarine Habitats

West et al. (1985) reported that Brisbane Water in its entirety had 163.5 ha (1.635 km²) of mangrove and 91.8 ha (0.918 km²) of saltmarsh.

The Brisbane Water area has been colonised by the two sympatric mangrove species of the region, *Avicennia marina* and *Aegiceras corniculatum*. The saltmarsh communities consist of *Sarcocornia quinqueflora*, *Sporobolus virginicus*, *Suaeda australis*, *Samolus repens* (Forst. Et f.) Pers., *Triglochin striata* Ruiz et Pav., and *Juncus kraussii*. Swamp forest and fringing vegetation are present on Rileys Island (Harty, 1999, pp. 31-32).

The NSW Department of Environment and Planning (1983, pp. 13-14) noted that although Rileys Island had been comprised primarily of various combinations of mangrove and saltmarsh species during the 1980s, the vegetation had been modified in the eastern half by grazing, clearing and construction during its settlement in 1830s. This settlement has since been abandoned, and this area of the island is slowly returning to its previous state.

Similarly, the NSW Department of Environment and Planning (1983, p. 15) observed that Pelican Island was different in character from Rileys Island, in that is was colonised predominantly by mangroves, with a closed scrub of *Avicennia* and individual *Aegiceras* trees, and little saltmarsh area. The Department determined that the island supported the largest single stand of mangroves (43 ha) in Brisbane Water at the time (1983).

The NSW Department of Environment and Planning (1983, p. 17) suggested that prior to its complete development during the 1970s, the wetland habitats of St Huberts Island were similar in to those on Rileys Island. The only wetland remnants left following the development of canal estates were individual mangroves along the northern channel and the southern tip of the island.
Population

The rate of population growth between 1947 and 1997 was 4.21%, compared to the state growth rate of 1.1%, and the national growth rate of 1.4% (Figure 2.15). In 1999 it was 158,172.

Catchment Modifications and Land Use

The Brisbane Water catchment has been severely modified (Heap et al., 2001). Resulting from its extensive catchment modifications, Brisbane Water has been subject to sedimentation and high nutrient loads, and in some instances canal urbanisation. The region’s landuses include commercial; industrial; low, medium and high density residential; rural residential; rural; open space; national parks and nature reserves; conservation and scenic protection; amongst other special uses (Gosford City Council, 1999). As with many areas along the NSW coast, it is within the coastal fringe that much of this development has occurred.

Despite an ever-increasing population, the agricultural areas surrounding Gosford, incorporating Brisbane Water, have not suffered from significant land degradation, apart from suggestions that the soil fertility is relatively low. Additionally, the NSW DLWC (2000b) suggested that soil erosion in the area is of little significance, and any that has been occurring is in fact slowing, due to the rate of urbanisation and Gosford City Council’s Erosion and Sediment Control Policy and Code of Practice (Gosford City Council, 1999).

2.3.5 Courangra Point, Hawkesbury River

The Hawkesbury River (151°15’E, 33°35’S) is a drowned river valley located on the northern boundary of the Sydney Metropolitan area (Figure 2.26). It has a vast catchment area of 21,500 km², a waterway of 100 km², and its entrance is open and untrained, entering into the Pacific Ocean through the mouth of Broken Bay. Brisbane Water to Broken Bay’s north and Pittwater to the south
are both tributaries of the Hawkesbury River.

Most of the northern and western areas of the Hawkesbury Valley are heavily timbered and comprised of deeply incised dissected Triassic sandstone plateaus. It has many tributaries, which include Brisbane Water and Pittwater (NSW DLWC, 2000c). AGSO (2001c) classified the Hawkesbury River as a severely modified tide-dominated estuary. Its IBRA classification is the Sydney Basin, and its IMCRA classification is the Hawkesbury Shelf. Courangra Point itself is located at approximately 151°07’E and 33°27’S on the Hawkesbury River, between Haycock Reach and Gentlemans Halt.

![Map of Hawkesbury River Catchment](image)

Figure 2.26: Courangra Point, on the Hawkesbury River.
Climate

Wisemans Ferry, which is home to Courangra Point’s closest rainfall gauge, has been subject to a mean rainfall of 845 mm from 1903 to 1998. Its lowest annual rainfall was 437 mm, whilst its maximum was 1498 mm (Figure 2.7).

Geomorphology

The Hawkesbury River estuary is a mature, deeply incised, bedrock-confined, drowned river valley that developed during the post-glacial marine transgression (PMT), between approximately 17 000 and 6000 years ago, during the late Pleistocene and early Holocene, and is the longest estuary in NSW (Roy, 1984). During the PMT, continental shelf sediment migrated landwards to infill the valley with sediment, resulting in a succession of fluvial to estuarine to marine sedimentation. Accretion of the Hawkesbury River estuary during the PMT was at the very high rate of about 10-15 mm/yr (Roy, 1994). The central section of the basin has remained relatively infilled by the sediment that was delivered to it during the PMT. Its maturation has followed the stages described by Roy (1984), creating new intertidal areas for potential colonisation by mangrove and saltmarsh species.

Mud sedimentation in the lower part of the estuary was relatively brief, occurring between 10 000 and 7000 years ago. From radiocarbon dates on shell and charcoal from scattered drill hole samples throughout Broken Bay, Roy (1994) found that the tidal delta in Pittwater was in place by 7000 years ago, with the delta front still accreting into the Pittwater basin. Approximately $2 \times 10^8$ m$^3$ of marine sand had accumulated in the Hawkesbury River mouth and Broken Bay by this time.

The Courangra Point wetland occupies a subaerial alluvial plain that is traversed by a leveed channel (Saintilan and Hashimoto, 1999). This point bar is a component of an upper bay-head delta that began prograding approximately 6000 years ago, which has been traversed by the leveed channel of the Hawkesbury River (Saintilan and Hashimoto, 1999).
Estuarine Habitats

Both West et al. (1985) and AGSO (2000e) calculated that the entire catchment contained 1065 ha (10.654 km\(^2\)) of mangrove and 112.6 ha (1.126 km\(^2\)) of saltmarsh.


Although the ecosystem health of the western (upper) section of the river and the middle section is poor, the health of the eastern (lower, estuarine) section of the river is relatively healthier and less likely to be impacted than those ecosystems upstream (Healthy Rivers Commission, 1998, p. 61). It is possible to infer from this that mangrove habitats have flourished in these healthier conditions, at the expense of saltmarsh habitats.

Population

Population for this site was impossible to determine, since the Hawkesbury River basin incorporates several very large catchments, and data was only available for local government statistical divisions, which did not coincide with the catchment boundaries. The population would be in the order of several hundred thousands.
Catchment Modifications and Land Use

According to the Healthy Rivers Commission (1998, p. 2), catchment landuse is divided into 68% forested, 25% agricultural and less than 7% is urbanised. It has thus been categorised as severely modified (Heap et al., 2001). The river faces a number of major use-issues, including deteriorating water quality; expanding urbanisation of the catchment and river frontage; land and river bank degradation and erosion; flood mitigation combined with severe flooding; recreational use (including a wide variety of watercraft); impacts of non-urban landuses; sand and gravel extraction (supplying 80% of Sydney’s industrial sand); agricultural use (2 800 ha of land irrigated from the waterway); storage and water supply; groundwater resources; riverine ecosystem degradation; and the river economy (such as oyster leases and tourism) (Hawkesbury-Nepean Task Force, 1991).

The Healthy Rivers Commission (1998, pp. 2-3) summarised adverse effects on the catchment:

- removal of riverside vegetation (associated with both agricultural activities and urban development);
- dams and weirs which reduce downstream flows and inhibit fish passage;
- water abstraction for irrigation, town water supply, stock and domestic use;
- effluent disposal from sewage treatment plants, on-site disposal systems and boats;
- extractive industries both past and present; and
- an array of recreational pursuits.

The waterways of the Hawkesbury River are of very poor quality, incorporating even (blue-green) algal blooms in the lower reaches of the river (from Penrith to Wisemans Ferry), which indicate excessive amounts of plant nutrients in the water or sediments available to the algae (Healthy Rivers Commission, 1998).

commented that in the more natural areas, higher levels of phosphorus and nitrogen do not appear to lead to excessive algal or macrophyte growth, which they attributed to generally faster velocities and greater shading. They listed the major sources of excessive nutrients entering the catchment waterways as being: sewage treatment plants operated by Sydney Water and several country towns; septic tanks adjacent to waterways; in rural areas, contributions from stock grazing in or adjacent to waterways, animal wastes from high intensity dairy activities and erosion of nutrient-rich soils (principally from unstable gullies and unvegetated streambanks); and urban stormwater runoff.

2.3.6 Careel Bay, Pittwater

Careel Bay is a bay of Pittwater (151°19’E, 33°36’S), which itself is a tributary of the drowned river valley of the Hawkesbury River (Figure 2.27). The bay’s wetlands have colonised its tributary bay-head delta, which is comprised of fluvial sands derived from the catchment. Pittwater is located 40 kilometres north of Sydney. Pittwater has a surface area is 125 km², has an average depth of 15 to 17 m, a volume of 250 km³, and is bounded by 77 km of foreshore (AWACS, 1996b). Its IBRA classification is the Sydney Basin, and its IMCRA classification is the Hawkesbury Shelf.

Careel Bay lies between Sand Point and Stripe Point on the eastern side of Pittwater. It was once almost totally a saltmarsh, and now is covered mostly by mangroves, with almost no pure saltmarsh habitat remaining. At the narrowest point, Careel Bay is divided by land from the Pacific Ocean by just over 200 metres of dune sand, swamp and bedrock. Careel Bay’s catchment is 10.18 km² (Pittwater Council, 1997). The location of Careel Creek, combined with Pittwater’s and Careel Bay’s tidal movements and wind and wave regime, is important when observing the natural functioning of Careel Bay. The bay has a tidal range of 2 metres, and during the past 100 years approximately 1.3 metres of sediment has reportedly been deposited within it. A bar of sediment is located at the entrance to Careel Bay, preventing further significant deposition from Pittwater (Hutchings and Recher, 1974).
Figure 2.27: Careel Bay, Pittwater.

Climate

Mean annual rainfall in the eastern portion of Pittwater (i.e. Newport and Avalon) between 1932 and 1936 was in the order of 1212 mm, whilst its minimum was 573 mm, and its maximum 2190 mm (Figure 2.8).

Geomorphology

Pittwater’s geological history is incorporated into that of the Hawkesbury River (Section 2.2.5). In opposition to the primary mode of sediment movement in Broken Bay’s north arm (Brisbane Water) of open-ocean waves moving in from the southeast, Pittwater’s sediment movement is primarily driven by tidal
currents. Accretion rates fell to 1-2 mm/yr on the channel bed towards the end of the PMT, characterised more by sandy sediment rather than muddy sediment (Roy, 1984, 1994). In this mature stage of the estuary, sediment movement is predominantly seaward. This rate of historical advance is considered to be relatively compared to Roy’s (1994) rate of mud accumulation in NSW estuaries of 0.1 to 3.0 mm/yr. The basement geology of the basin consists of the Triassic Narrabeen group sandstones and shales. Humic gley soils are present in the swamps.

Triassic sandstones and shales of the Newport Formation and Garie Formation and Quaternary Alluvium underlie Careel Bay’s surrounds. On the surface, the bay is surrounded by estuarine, erosional, colluvial and disturbed terrain (Hitchcock Park, in Careel Bay’s south-east, is constructed of man-made fill). The bay itself contains deep waterlogged calcareous sands, fluvial mud and sandy mud (>50% mud) and marine muddy sand (10-50% mud), upon which lives seagrass, mangroves and saltmarshes.

Much of Careel Bay’s sediment (sand) was deposited at the head of the bay by erosion from the catchment. The sediment within the bay remains there, controlled by the action of waves brought on by winds blowing from northern and westerly directions. These winds produce active inwards drift of the beach sand on the eastern littoral. The growth of seagrasses and mangroves in the littoral and sub-tidal areas of the eastern shore of Careel Bay also serve to trap the sediment (Hattersley et al., 1973).

**Estuarine Habitats**

West et al. (1985) calculated that Pittwater in its entirety contained 18 ha (0.18 km²) of mangrove and 2.6 ha (0.026 km²) of saltmarsh.

The mangrove flora of Careel Bay consists of *Avicennia marina* and *Aegiceras corniculatum*. The saltmarshes in Careel Bay represent a rare and declining habitat. Species include *Sarcocornia quinqueflora*, *Sporobolus virginicus*, *Samolus repens*, *Triglochin striata*, *Suaeda australis*, *Cyperus polystachyos*.
Rottb., and *Juncus kraussii*. Terrestrial communities of *Casuarina* and *Eucalyptus* species surround the wetland.

Pittwater Council have nominated the Careel Bay wetlands as a regionally significant estuarine wetland and the changes in their dynamics are of concern (Hattersley *et al.*, 1973; Pittwater Council, 1995; AWACS, 1996b). Their areal extent in Careel Bay has been documented to be declining since the 1970s (Wilton, 1998, 2001).

**Population**

The Pittwater region (Pittwater and Warringah) have been subject to a population growth rate of 3.46% for the period 1947 to 1997, and in 1999 was 189,914. Careel Bay itself is surrounded by a densely urbanised catchment, most of which occurred in the late 1960s, early 1970s.

**Catchment Modifications and Land Use**

The Pittwater catchment was classified by Heap *et al.* (2002) as modified (it contains < 65% of its natural cover).

European settlers explored the Barrenjoey Peninsula in the late 19th century. By the 1950s, settlement of the land was on a permanent occupation basis, and infrastructure was limited to lower ground near the water (McDonald McPhee, 1989). As a result, activity on the foreshores and the water increased significantly, placing a new stress upon the environment. Land was cleared, altering the natural pathways of the food chain and changing the environment, by removing some native vegetation and introducing exotic flora and fauna.

The catchments feeding Pittwater consist of a mixture of urban areas and Ku-Ring-Gai Chase National Park. These surroundings are responsible for a combination of pressures placed upon the waterway. On the one hand, from the National Park, there is little known pollution running into the water, and a minimum of sediment runoff being introduced, except after bushfire events.
when sediment is mobilised. On the other hand, an array of pollution and user-
group pressures are placed upon the waterway from developed areas. Pollution
inputs are both point and non-point source and consist of sediment, phosphorus,
nitrates, heavy metals, urban waste and litter, wastes from domestic animals,
and weed infestation (Pittwater Council, 1995). The eastern shore of Pittwater,
within which Careel Bay is located, has been largely urbanised, and includes
residential, industrial and commercial development, but also contains many
parks and recreational areas.

Contaminated sites affect Careel Bay. Hitchcock Park, on its eastern shore, is a
former landfill site that has been converted to sports fields. Soil erosion
(primarily as a result of construction of residential estates) and degradation of
original vegetation zones has been occurring.

There are a number of key events which have contributed to the changing
ecological character of Careel Bay, including:

- the construction of a sports ground/landfill site in the south-east corner of
  Careel Bay, adjacent to Careel Creek, in approximately 1965;
- the extension of the waste site/landfill along the eastern and northern
  perimeters of Careel Bay in approximately 1972;
- the conversion of the landfill to the sports fields of Hitchcock Park;
- urbanisation of the land immediate surrounding Careel Bay. Land which
  has been cleared for construction loses sediment to the bay during the
  construction phase; and
- bushfire activity around Careel Bay and further afield around Pittwater,
  such as the January 1994 fires which burnt through most of Ku-Ring-Gai
  National Park, on the western perimeter of Pittwater.
2.3.7 Towra Point, Botany Bay

Botany Bay (151°14’E, 34°00’S) is an oval shaped enclosed embayment on the southern perimeter of the Sydney metropolitan area, at the mouth of the Georges River (Figure 2.28). It is approximately 7 km long and 5.5 km wide, with a relatively shallow depth at low water of approximately 5.5 m. It has a catchment area of 1100km², a waterway area of 80 km², and a deep and open entrance with all weather access (NSW DLWC, 2000d). The IBRA classification of Botany Bay is the Sydney Basin, and its IMCRA classification is the Hawkesbury Shelf.

Figure 2.28: Towra Point, Botany Bay.

Towra Point itself is situated on the Kurnell Peninsula, which is an 8 km long promontory on the southern side of Botany Bay, between Woolooware and Quibray Bays (Australian Heritage Commission, 2001). It is a back barrier wetland composed of marine sand.
Most of the land at Towra Point (281.7 ha) was acquired by the Federal Government in 1975, in order to meet the obligations of a number of international treaties protecting the migratory bird habitats of the area. In 1982 the land was transferred to the NSW State government, and gazetted as the Towra Point Nature Reserve under the NSW National Parks and Wildlife Act, 1974. The Nature Reserve later had more land added, and now comprises 386.4 ha, which includes the beds and foreshores of Weeney Bay and land at Quibray Bay. Further to its designation as a Nature Reserve, in 1984 Towra Point Nature Reserve was placed on the list of Wetlands of International Importance. In 1987 Towra Point Aquatic Reserve was established flanking the Nature Reserve, under the NSW Fisheries and Oyster Farms Act, 1935, and in 1989 its environmental importance was recognised with the development of the Sydney Regional Environmental Plan (REP) No. 17: Kurnell Peninsula (Towra Point Steering Committee, 1999).

The wetlands of Towra Point are protected by the National Parks and Wildlife Act, 1974, the Threatened Species Act, 1995, the Environmental Planning and Assessment Act, 1979, and the Fisheries Management Act, 1994. The wetlands are also the subject of three international treaties, for which the government is required to meet certain conservational obligations: The Japan-Australia Migratory Bird Agreement (JAMBA), the China-Australia Migratory Bird Agreement, and the Ramsar Convention on Wetlands of International Importance (Towra Point Steering Committee, 1999).

Climate

From 1930 to 1998, Towra Point was subject to a mean annual rainfall of 1107 mm, whilst its minimum annual rainfall was 523 mm, and its maximum 2025 mm (Figure 2.10).

Geomorphology

The most recent assessment of the evolution of Botany Bay suggested that it has been a large trap for marine sands transported northward by littoral drift during
the past 2 million years (Healthy Rivers Commission, 2000, p. 148). This has resulted in the extensive dune system that Botany Bay is a part. This dunal system was most likely responsible for maintaining the area currently known as Botany Bay as an estuarine lagoon approximately 5000 to 6000 years ago. When the dune system began to erode approximately 4000 years ago, Botany Bay’s entrance was exposed to the ocean. This began a long series of geomorphic instability, which is continuing today. It has been manifested in constant changes in the bay’s morphology and its shorelines (Healthy Rivers Commission, 2000, p. 148).

The Commission (Healthy Rivers Commission, 2000, p. 148) also noted that Botany Bay is particularly susceptible to storms due to its geographic orientation. These too can result in temporary, yet considerable, changes in the physical processes of the bay, such as large-scale beach erosion. Climatic events have helped to create changes in the shoreline throughout both geological and contemporary time, in conjunction with events such as dredging and the creation of infrastructure in contemporary time.

Major development projects in the bay and its shorelines in contemporary time were assessed by the Healthy Rivers Commission (2000, pp. 44-46). The development of major infrastructure between 1970 and 1994 has served to create large-scale disruptions in the bay’s morphological character, which has resulted in ongoing shoreline adjustments. It has been ascertained that the bay has most recently increased its rate of readjustment. This was attributed to two factors: (a) the increased wave energy along the southern shore induced by dredging, and (b) the instability caused by the groynes on Silver Beach to Bonna Point, which reduced the amount of sand available to the Towra Point area for a period lasting two decades. The assessment determined that the most complex and vulnerable area of the bay is immediately downstream of either side of the Georges River, which incorporates the Towra point wetlands.
Estuarine Habitats

The entire Georges River - Botany Bay catchment contains approximately 870 ha of estuarine wetland, nearly 70% of which is located in Towra Point, Quibray Bay and Woolooware Bay (Healthy Rivers Commission, 2000). Both West et al. (1985) and AGSO (2000d) reported that Botany Bay has 399.6 ha (3.996 km²) of mangrove and 160.1 ha (1.601 km²) of saltmarsh. It is estimated that Towra Point contains 50% of mangroves remaining in the Sydney region (Towra Point Steering Committee, 1999).

The mangrove habitat at Towra Point consists of primarily *Avicennia marina* with some instances of *Aegiceras corniculatum*. The saltmarsh zone consists of *Sarcocornia quinqueflora*, *Scirpus nodosus*, *Suaeda australis*, *Triglochin striata*, *Hydrocotyle bonariensis* Lam., *Juncus kraussii* and *Phragmites australis* (Cav.) Trin. Throughout the saltmarsh are instances of *Chrysanthemoides monilifera* and *Lantana camara*. Large communities of terrestrial vegetation are present, limited to the northern sections of the main wetland (Fenech, 1994; Australian Heritage Commission, 2001).

The Healthy Rivers Commission (2000) determined that there were approximately 870 ha of estuarine wetlands in the Georges River – Botany Bay catchment, with nearly 70% found on the southern shore of Botany Bay around Towra Point, Quibray Bay and Woolooware Bay.

Population

The entire Georges River/Botany Bay catchment contains more than 1.5 million residents (Healthy Rivers Commission, 2000, p. 3) (Figure 2.17). As with the Courangra Point catchment population, it is impossible to accurately determine the population size of the Towra Point catchment.
Catchment Modifications and Land Use

The Botany Bay region has been subject to some of the most intense urbanisation in the Sydney region since European settlement, and as such has been categorised as severely modified (Heap et al., 2001). It has been suggested (Mitchell and Adam, 1989b) that earlier settlers used the natural resources of the wetlands to sustain their new colony. The land clearing associated with this urbanisation has led to erosion of the catchment and increased sediment delivery to the estuarine embayment, which has provided large intertidal areas for mangrove colonisation. Mitchell and Adam (1989b) suggested there has been three phases of direct disturbance in Towra Point: the soap industry, agriculture, and the oyster industry.

The Kurnell Peninsula itself has been subject to large-scale landuse changes, and has produced subsequent pollution. The Kurnell Refinery was constructed in 1955, which incorporated extensive dredging and wharfage works. An oil spill that took place in 1979 is thought to have been responsible for large-scale mangrove dieback in Quibray Bay in the mid-1980s (Evans and Williams, 2001).

The Healthy Rivers Commission (2000, pp. 152-155) concluded that the engineering interventions in Botany Bay during the past 200 years had impacted the bay’s system in four major ways: (1) alteration of ocean wave direction; (2) changes in wave energy; (3) changes in sand movement; and (4) modification of shoreline morphology.

2.3.8 Currambene Creek and Cararma Inlet, Jervis Bay

Jervis Bay (150°48’E, 35°07’S) is a semi-sheltered, tide-dominated embayment with tributary creeks and deep, all-weather access (Figure 2.29). It has a catchment area of 410 km², and a waterway area of 102 km² (NSW DLWC 2000e). It is approximately 180 km south of Sydney. Its tributaries are Currambene Creek, Cararma Inlet, Callala Creek, Bib Bib Creek, and Moona
Moona Creek. Jervis Bay’s IBRA classification is the Sydney Basin, and its IMCRA classification is the Batemans Shelf.

Currambene Creek is a fluvial tributary of the larger Jervis Bay, located in the northwestern corner (Figure 2.29). It has a catchment area of 165 km² and a waterway area of 1.2 km² (NSW DLWC 2000e). HMAS Albatross in the northwest, Braidwood and Turpentine Roads in the southwest and Jervis Bay at Huskisson border the catchment in the east. Its catchment comprises 60% of the entire catchment of Jervis Bay (Shoalhaven Catchment Management Committee, 1999).

Cararma Inlet (Figure 2.29) is also a tributary of the Jervis Bay, located in its northeastern corner, adjacent to Hare Bay. It is a tidal inlet composed of marine sand. Cararma Creek runs through the centre of the inlet. The mapped wetlands of Cararma Inlet are bordered by Chinamans Beach to the south of the mouth, and Hare Point to the north of the mouth, Currarong Road to the east and north, and Cabbage Tree Creek and its associated terrestrial habitats to the west (Saintilan and Wilton, 2001).

Climate

Jervis Bay is located in a temperate maritime region of NSW, and is subject to warm summers and mild winters. It has a rainfall range of between 463 mm and 2053 mm, with 1104 mm being its mean annual rainfall (Figure 2.10).

Geomorphology

The Australian Nature Conservation Agency (1995, pp. 3–4) described Jervis Bay as comprising five main geological units, all but one belonging to the Permian Shoalhaven group. This geological group is part of the sedimentary rock formation on the southern edge of the extensive Sydney Basin system. Less extensive geological units are also found in the bay. The group of geological units of Permian age (c. 280-225 million years before present) that make up Jervis Bay include the Currambene Dolerite, a volcanic sill that is barely
evident in the Currambene Creek area; the ‘Snapper Point Formation’ of quartzose sandstone that covers most of the Bherwerre and Beecroft Peninsulas except where there are sandstones; the ‘Wandrawandian Siltstone’ formation that is composed of a mix of siltstone to sandstone cover the northern and eastern parts of Jervis Bay; the ‘Nowra Sandstone’ of quartzose sandstone that covers a large part of the upper reaches of the Currambene Creek catchment area; and the ‘Berry Siltstone’ to the north of Currambene Creek, composed of siltstone and fine sandstone (ANCA, 1995, pp. 3-4).

The tidal wetlands of Currambene Creek and Carama Inlet are characterised by a flat muddy landscape across which meandering tidal channels traverse and sandbars prograde, behind which mangroves and saltmarsh habitats have colonised (ANCA, 1995). The Agency determined that the tidal flats are
composed of fine-grained materials ranging from fine-grained sand to clay, with muds predominating. These areas are frequently inundated by brackish or salt tidal water and in many cases covered by swampy vegetation. The sediments are generally rich in organic matter and dark-coloured. Many of the tidal/lagoon deposits also contain pyrite and when exposed are readily oxidized which results in a dramatic increasing acidity or lowering pH of the sediments (ANCA, 1995, p. 45). Currambene Creek also consists of alluvial deposits particularly along its middle reaches (ANCA, 1995, p. 45).

Currambene Creek has flowed through considerably different locations, since at least 1832, when surveys of the areas were first made. In particular, just over a hundred years ago, the meander at Woollamia was cut off, and a new channel was formed. High rates of erosion have been recorded along the banks of the estuary during last century. A rock groyne was constructed by Council and NSW Public Works in the 1980s after the creek attempted to break a new channel through to the bay.

**Estuarine Habitats**

The wetlands of Jervis Bay (which is composed of Jervis Bay, 30 000 ha, and Beecroft Peninsula, 4044 ha, Lake and Peninsula, 7000 ha) are included in the *Directory of Important Wetlands in Australia* (ANCA, 1996, pp. 94-96). West *et al.* (1985) determined that in 1984 Jervis Bay contained 125 ha of mangrove and 230 ha of saltmarsh. Of the total 30 000 ha area, 1552 ha are protected by SEPP 14 legislation, including Currambene Creek. The mangroves of Jervis Bay cover 125 ha (1.25 km²), whilst the saltmarsh covers 233 ha (2.33 km²) (West *et al.*, 1985; AGSO 2001e).

Both *Avicennia marina* and *Aegiceras corniculatum* are present in Jervis Bay. Saltmarsh communities consist of various combinations of *Sarcocornia quinqueflora*, *Sporobolus virginicus*, *Sclerostegia arbuscula*, *Samolus repens*, *Wilsonia backhousei*, *Triglochin striata*, *Suaeda australis*, *Gahnia filum*, *Halosarcia*, *Juncus kraussii*, *Baumea juncea* and *Selleria radicans* Cav. Stands of terrestrial communities are scattered throughout the various wetlands.
Parts of the Currambene Creek wetlands are protected as part of the Jervis Bay Marine Park (gazetted in 1998), which has jurisdiction up to mean high water. This effectively means that only the mangroves, and not the saltmarsh habitats, are protected by this park. Currambene Creek itself contains eight separate wetlands defined and protected by State Environmental Planning Policy Number 14.

**Population**

The 1996 Census showed that the Currambene Creek catchment itself was home to 2280 people, 1800 of who are classed as rural population (Shoalhaven Catchment Management Committee, 1999) (Figure 2.18). Cararma Inlet has virtually nil residents. For the greater Shoalhaven region, population growth was 3.69% from 1947 to 1997.

**Catchment Modifications and Land Use**

Currambene Creek has been largely unmodified (Heap *et al.*, 2001), although much of it has been converted for agricultural use. Most of Currambene Creek’s landcover in the western half (the catchment is divided in half by the Princes Highway) consists of grazing land in the north and native bushland in the south of which there is almost a 1:1 ratio. The land in the east is primarily rural residential with native vegetation throughout (Shoalhaven Catchment Management Committee, 1999). The wetlands are a sink for point and non-point runoff, since the rural population of 1800 in the catchment resides higher in the catchment than the wetlands. The Cararma Inlet catchment is near pristine (> 90% of the catchment’s natural cover remains) (Heap *et al.*, 2001).

The upper catchment has been deemed to have a high rate of runoff, due to a combination of landcover and soil type, resulting in high creek flows during rain periods, and low creek flows during dry periods (Shoalhaven Catchment Management Committee, 1999).
2.4 Summary

In summary, the sites chosen reflect a variety of estuary types, bioregions, rainfall conditions, and catchment landuse. The sites reflect a range of estuary types, from the drowned river valleys of the Hawkesbury River, to barrier estuaries like Tilligerry Creek, to coastal lakes like Brisbane Water, to open and enclosed coastal embayments like Jervis Bay and Botany Bay. A range of bioregions are represented, from the northern coast of New South Wales to the southern coast. The sites reflect a range of regional rainfall characteristics, from the high rainfall catchment of the Tweed River to the low rainfall area of Courangra Point. Catchments which have been subject to various degrees of modification are represented, from highly developed, to near pristine.