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An integrated approach to assess the vulnerability of mangrove and saltmarsh to sea-level rise at Minnamurra NSW

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An integrated approach to assess the vulnerability of mangrove and saltmarsh to sea-level rise at Minnamurra NSW

Abstract

Mangrove and saltmarsh communities are essential components of estuarine ecosystems on the NSW south coast. Globally, projected sea level rise under IPCC climate scenarios pose a significant threat to the survival of mangrove and saltmarsh ecosystems. The resilience of mangrove and saltmarsh to the impacts of sea-level rise is dependent upon the maintenance of their elevation with respect to rising water levels and their ability to migrate landwards and requires further analysis. An integrated vulnerability assessment of the mangrove and saltmarsh at Minnamurra River, NSW, was undertaken to assess the resilience of mangrove and saltmarsh to the impacts of sea-level rise and recommend future adaptation policy and management strategies to preserve these communities.

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Comments

Mangrove and saltmarsh communities are essential components of estuarine ecosystems on the NSW south coast. Globally, projected sea level rise under IPCC climate scenarios pose a significant threat to the survival of mangrove and saltmarsh ecosystems. The resilience of mangrove and saltmarsh to the impacts of sea-level rise is dependent upon the maintenance of their elevation with respect to rising water levels and their ability to migrate landwards and requires further analysis. An integrated vulnerability assessment of the mangrove and saltmarsh at Minnamurra River, NSW, was undertaken to assess the resilience of mangrove and saltmarsh to the impacts of sea-level rise and recommend future adaptation policy and management strategies to preserve these communities.

Surface Elevation Tables (SET) and Marker Horizons (MH) were established within mangrove and saltmarsh in 2001 to establish the elevation and accretion dynamics and trends of the study site. Current surface elevation trajectories were much lower than average local sea-level rise over the study period and compared to longer-term local sealevel trends. Surface elevation appeared to be associated with fluctuations in groundwater levels and hence linked with El Niño and La Niña patterns. In accordance with models of sea-level rise impacts on mangrove and saltmarsh communities, photogrammetric mapping of the study site for the period 1938 to 2011 indicated mangrove encroachment of saltmarsh was evident and in recent years consolidation of mixed zones by mangroves.

A Digital Elevation Model (DEM) was constructed using LiDAR and high precision GPS data collected from the site. This was used as the basis for the creation of statistical and spatial models exploring the relationship between surface elevation dynamics, water level changes and mangrove encroachment of saltmarsh. These models projected wetland surface elevation and vegetation distributions in accordance with IPCC projections of sealevel rise. The models indicated that coastal wetlands at Minnamurra are

highly vulnerable to future sea-level rise. Using the highest IPCC sea-level rise scenario, the models showed a significant loss of saltmarsh in the next 40 years and loss of mangrove communities by the end of the century. It is recommended that future adaptation policy and management focuses on groundwater regulation in the catchment and the introduction or extension of buffer zones.

INTRODUCTION

This project examines the response of mangroves and saltmarsh at the Minnamurra River, to sea-level rise in order to assess their vulnerability and provide a firm basis on which to develop climate change adaptation policy and management for the future. To achieve this aim, the following integrated methodology will be used: measuring of surface elevation dynamics and vertical accretion, spatio-temporal mapping of vegetation zones and statistical and spatial modelling using sea level projections provided by the Intergovernmental Panel on Climate Change (IPCC).

CHAPTER 1: BACKGROUND TO THE STUDY

1.1 INTRODUCTION

Understanding mangrove and saltmarsh surface elevation dynamics, mapping their changing distributions and modeling future changes is essential if we are to understand the response of mangrove and saltmarsh to future threats such as sea-level rise. The preservation of mangrove and saltmarsh communities is essential to climate change adaptation as they are integral parts of estuaries and interact with broader marine systems providing essential ecosystem services to communities and society.

Mangrove and saltmarsh provide habitat for many juvenile fish, water birds and invertebrates such as prawns, crabs and oysters. Therefore they are vital to maintaining sustainable long-term commercial and recreational fishing industries. Wetland communities also sequester carbon and therefore have the potential to reduce carbon emissions if their current distribution is preserved (Gilman et al. 2008). They also act as a natural barrier and play a major role attenuating wave energy and protecting the hinterland against severe storm surges and tsunamis as seen recently in the 2004 Indian Ocean Tsunami (Dahdouh-Guebas et al. 2005).

Preserving mangrove and saltmarsh is therefore critical for our society in adapting to a warming climate. Investigating the vulnerability of mangroves and saltmarsh to sea-level rise is the first step towards adaptation. Such investigations may change attitudes and challenge the current regulatory environment, promoting adaptation options and management frameworks that address key issues and concerns for these communities.

1.2 SEA-LEVEL RISE PROJECTIONS

1.2.1 Global projections

Sea-level projections published by the (IPCC) range from 0.18 – 0.59 metres by 2100 (Intergovernmental Panel on Climate Change 2007). While there appears to be a wide range for these projections this is largely due to uncertainty in our understanding of earth's natural systems especially the role of ice sheets such as the Greenland and West Antarctic ice sheet (Church et al. 2008; Nicholls & Cazenave 2010) and the strength of various feedback mechanisms such as clouds and sea-ice albedo (Intergovernmental Panel on Climate Change 2007).

1.2.2 Sea-level rise on the NSW south coast

IPCC sea-level projections are global averages will not affect areas uniformly across the globe (FitzGerald et al. 2008). Sea-level rise will not be at the same rate in all places because of vertical land movements such as isostatic adjustment and changes in ocean surface dynamics (Nicholls & Cazenave 2010). Therefore it is necessary to understand local factors which control sea-level such as local subsidence, local sea temperature as well as tidal patterns and regimes. The NSW east coast of Australia appears to have higher than global average rates of sea level rise (Church et al. 2008). Sea level at Port Kembla, south of Wollongong, has risen at an average rate of 2.1mm.yr⁻¹ since 1991 (Bureau of Meteorology 2010 p.37), although there is conjecture over this value due to uncertainty in the magnitude of barometric corrections (C. Woodroffe, personal communication). Sea-level rise at Fort Denison, Sydney, where the longest sea-level records exist for the Australian east coast, is 0.86 mm.yr⁻¹. However, since 1990 rates have been substantially higher and there has been an increase in the frequency of extreme sea-level events (Church, McInnes, & Hunter 2006). Watson (2011) also notes sea-level rise acceleration in the most recent portion (>1990) of the sea-level record.

1.3 ASSESSING MANGROVE AND SALTMARSH VULNERABILITY TO SEA-LEVEL RISE

1.3.1 Introduction

The vulnerability of mangrove and saltmarsh to sea-level rise varies according to a number of controls: (1) shoreline topography, (2) rate of sediment supply, (3) rate of surface elevation change, (4) rate of sea-level rise, (5) tidal range (Kirwan &

Guntenspergen 2010; Morris et al. 2002; Reed 1990; Woodroffe 1990). Techniques to assess the vulnerability and attempts to quantify possible future changes of mangrove and saltmarsh distribution include measuring surface elevation dynamics and vertical accretion, spatio-temporal mapping, and modeling of coastal wetlands using various statistical and spatial approaches.

Northern hemisphere studies examining coastal marsh vulnerability to sea-level rise differ substantially in their conclusions. Yet all agree that there will be some loss of mangrove and saltmarsh communities, and that their response will be either migration or accretion involving the complex interaction of many geomorphological and ecological factors and feedbacks (Craft, Clough, et al. 2009; Kirwan & Temmerman 2009; Morris et al. 2002).

Studies in Australia which directly assess the vulnerability of mangrove and saltmarsh to sea-level rise are limited; however, work by Rogers, Wilton, & Saintilan (2006); Rogers & Saintilan (2009) in particular shows that local factors are important in controlling surface elevation and hence the vulnerability of Australian wetlands on the southeast coast.

Suspended sediment concentrations, which would likely be very different to European and American sites due to different climate and river morphologies, has not yet been integrated into assessments of mangrove and saltmarsh vulnerability.

1.3.2 Surface elevation and accretion dynamics

1.3.2.1 Surface elevation and sea level

Mangrove and saltmarsh occupy specific regions within the tidal range and elevation and sea level control their position on the shoreline (Kirwan & Guntenspergen 2010). In response to sea-level rise, mangrove and saltmarsh must either migrate landwards or adjust their elevation relative to water levels. Measuring surface elevation changes and vertical accretion in mangroves and saltmarsh and comparing these values to projected sea-level rise provides insight into the extent of possible future changes in their distribution. However, rather than uniform changes in mangrove and saltmarsh zonation, distribution or succession, mangroves may respond opportunistically to changing microtopography and hence tidal flooding patterns may be most significant when considering sea-level threats (Thom 1967). Indeed the zonation of different mangrove types is complicated and based on a range of factors relating to their position in the tidal prism, including sedimentation and salinity (Woodroffe 1990).

Similar zonation based on surface elevation and sea level occurs in saltmarsh communities where even individual species have elevation ranges (Adam 2002; Hickey & Bruce 2010; Singh Chauhan 2009). Spatial variation of specific saltmarsh species is also based on microtopography within the marsh environment (Adam 2002; Hickey & Bruce 2010). Therefore, in assessing the vulnerability of mangrove and saltmarsh to sea-level rise, studies which examine individual wetlands may provide important insight about the response of mangrove and saltmarsh microtopography to increasing water levels.

1.3.2.2 Processes and controls

Sediment processes in mangrove and saltmarsh environments are complex and involve the co-adjustment of form and process. There have been relatively few studies which have examined the processes, controls and feedbacks within these environments, the majority of which have been concerned with Northern Hemisphere settings. Therefore it is appropriate to undertake field-based studies in southeastern Australia to adequately quantify and account for such differences between estuarine settings and sea-level rise in Northern and Southern Hemispheres.

Differences in mangrove and saltmarsh form and function is based on external controls and processes such as climate, groundwater, sediment supply, primary production, decomposition rates, subsidence and autocompaction (Cahoon et al. 2006). Short perturbations such as storms also cause changes to sediment processes due to: high winds causing large volumes of organic matter such as leaves to fall to the ground, fresh-water flushing of the estuary causing soil swelling (Cahoon 2006), and changing sediment supply leading to morphodynamic adjustment of subsurface processes (Whelan et al. 2009). Longer term changes in weather patterns such as drought can also lead to large elevation changes, up to five times greater magnitude than long term trends due to groundwater level (Cahoon et al. 2011).

Cahoon et al. (2006) proposes that processes operating in mangrove and saltmarsh sediment may be categorised into biotic and abiotic and subsequently divides biotic processes into indirect, processes that modify natural deposition and erosion, for example aerial roots trapping sediment (Krauss 2003; Stokes, Healy, & Cooke 2010), and direct, processes that directly alter sediment volume or strength, for example, accumulation of decaying organic matter or the subsurface growth of mangrove roots (McKee, Cahoon, & Feller 2007; Nyman et al. 2006). Kumara et al. (2010) suggests that higher density of

mangrove trees enhances sediment accretion and surface elevation change and does not compromise mangrove growth and survival. This could be an example of either indirect, through roots trapping sediment or directly from greater volumes of organic material supplied to the sediment surface.

Abiotic controls such as estuarine suspended sediment concentration, which is determined by surrounding geomorphology, geology and river hydrology, also influence wetland sediment accumulation and hence surface elevation change (Krauss et al. 2010).

Groundwater has been identified as another significant control influencing mangrove and saltmarsh in Australian settings. Due to the relationship between rainfall and groundwater height, surface elevation has been correlated with El Nino-Southern Oscillation (ENSO) events (Rogers, Saintilan, & Cahoon 2005; Rogers & Saintilan 2008).

Mangrove and saltmarsh sediment can be regarded as being mature or immature (Cahoon et al. 2006). Immature wetland sediment which is weak or deteriorated is more likely to experience shallow subsidence (Cahoon, Day, & Reed 1999). Sediment accretion is an important indicator of progress in wetland restoration for immature wetlands (Takekawa et al. 2010). Shallow subsidence and surface elevation deficits, where amounts of accretion exceed amounts of surface elevation gain, may be large at immature wetlands (Cahoon et al. 1999). In contrast, the contribution of accretion along with subsurface processes to overall surface elevation gain or maintenance is more balanced in mature coastal marshes (Howe, Rodríguez, & Saco 2009).

1.3.2.3 Australian studies

Coastal wetland form and function varies with space and time and across different regions throughout the world. Southeast Australian mangrove and saltmarsh environments vary greatly from those of the Northern Hemisphere and this may be reflected in the vastly different rates of surface elevation change and surface elevation trajectories (Howe et al. 2009; Temmerman et al. 2004). Such differences may be a product of differences in the biotic and abiotic controls outlined above, but require further analysis.

Analysis of surface elevation in southeastern Australia has focused on a number of estuaries along the east coast of NSW including the Tweed River, Hunter River, Hawkesbury River, Parramatta River, Minnamurra River, Jervis Bay, and also Western Port Bay in Victoria (Rogers et al. 2006). For each of these locations an inventory of

SET's and marker horizons has been established and information collated regarding estuarine setting adopting the classification of Roy et al. (2001), catchment areas, temperature and rainfall ranges, long term sea level changes and dominant mangrove and saltmarsh species (Rogers et al. 2006). The results of this inventory suggest that local subsurface processes such as groundwater may be an important control, along with vertical accretion, on surface elevation dynamics and hence long-term elevation trajectories must incorporate short term perturbations such as droughts and *El Nino* fluctuations (Rogers et al. 2006; Rogers, Saintilan, & Cahoon 2005; Rogers, Saintilan, & Heijnis 2005; Rogers & Saintilan 2008). Work by Howe et al. (2009) on the Hunter River considered the difference between disturbed and undisturbed wetlands and concluded that while they showed comparable rates of surface elevation change, this was driven mostly by vertical accretion in the disturbed site, and equal amounts of vertical accretion and subsurface processes in the undisturbed site. The varying contributions of subsurface processes and vertical accretion reflect site maturity, where more mature sites appear to have equal surface elevation contributions from subsurface processes and vertical accretion (Howe et al. 2009). However, rates of surface elevation change at both sites were lower than sea-level projections and therefore migration of mangrove and saltmarsh will be essential for future survival of wetlands at this site (Howe et al. 2009).

1.3.3 Spatio-temporal coastal vegetation mapping

It is clear that mangrove and saltmarsh communities are dynamic systems. Spatio-temporal mapping of vegetation distributions indicates that many coastal wetland systems have undergone significant changes over the past 50 years (Clarke & Benson 1988; Saintilan & Williams 1999). Saltmarsh decline has been up to 80% in some estuaries in NSW and most estuaries in NSW have lost more than 25% as a result of mangrove encroachment (Saintilan & Williams 1999). Rogers et al. (2006) observed saltmarsh decline at all their sites from the Tweed River in the northeast of NSW, Hunter River, Hawkesbury River, Parramatta River, Minnamurra River, Jervis Bay and Western Port Bay in Victoria.

There is circumstantial evidence to suggest that sea-level rise may be an important factor influencing this pattern of vegetation change (Chafer 1998), a suggestion that has also been proposed by (Saintilan & Hashimoto 1999). Rogers et al. (2006) presents compelling evidence for linkages between mangrove encroachment of saltmarsh, surface elevation

trends and sea-level rise. Other hypotheses have also been proposed to explain this trend such as: higher precipitation, changing land use and agricultural practices, altered tidal regimes due to estuary entrance modification and changing sedimentation and nutrient inputs (Saintilan & Williams 1999).

1.3.4 Modeling coastal wetland response to sea-level rise

Modeling represents an important approach to forecasting how wetlands will respond to sea-level rise. Modeling coastal wetlands will be of increasing importance in future years as wider coverage of LiDAR data sets becomes available to coastal researchers and managers (Rogers & Saintilan 2009). Models such as Sea Level Affecting Marshes Model (SLAMM) has taken Digital Elevation Models (DEM) and produced future outcomes for marshes in South Carolina, USA, and along the Georgia coast (Craft, Clough, et al. 2009). The SLAMM model simulates the dominant processes in coastal marshes including shoreline changes, salinity, tidal processes and wave action-erosion under sea-level rise scenarios (Craft, Clough, et al. 2009). In Australian marshes, the SLAMM model has been applied to three estuaries on the north NSW coast with results indicating that there will be a net saltmarsh loss of around 26% by the end of the century (Akumu et al. 2010).

The application of the SLAMM model by Craft, Clough, et al. (2009) has received criticism by Kirwan & Guntenspergen (2009) who question the usefulness of the model based on the notion that accretion rates are modeled to decline, an assumption which they strongly refute, arguing that positive feedbacks result in ‘dynamic accretion’ which co-adjusts with sea-level rise. Craft, Pennings, et al. (2009) respond to this criticism by arguing that while positive feedbacks relating to accretion are significant and need to be further accounted for, “SLAMM represents a trade-off of mechanistic detail and broad spatial coverage” and that surface elevation is also a product of below ground processes which are also coupled with sea level.

Other models have been developed to examine the vulnerability of coastal marshes such as the Wetland Change module within DIVA (Dynamic Interactive Vulnerability Assessment) used by McFadden, Spencer, & Nicholls (2007) for Louisiana, USA. High resolution LiDAR for the southeast coast of Australia is being combined with 10 year surface elevation and accretion data, from which dynamic elevation models will be developed to establish the impacts of future sea-level rise on Australian marshes (Rogers & Saintilan 2009).

Incorporating feedback mechanisms into models of wetland response to sea-level rise is extremely complex (Craft, Clough, et al. 2009). Feedback mechanisms such as: increasing sediment supply to marshes as tidal inundation increases with sea-level rise thus promoting elevation increase (Kirwan & Guntenspergen 2009); the promotion of biomass growth by higher sea levels leading in turn to higher elevations (Kirwan & Guntenspergen 2010). The quantification of such feedback mechanisms is essential to understanding the response of coastal marshes to sea-level rise. Kirwan et al. (2010) suggests that non-linear feedbacks in inundation, plant growth, organic matter deposition and sediment deposition would allow marshes to survive medium to low sea-level projections providing that suspended sediment concentrations were greater than approximately 20 mg L^{-1} . Evidence from coastal marshes in Belgium supports this conclusion indicating that marshes could keep pace with rising sea levels as long as suspended sediment concentrations remained at present levels (Temmerman 2003; Temmerman et al. 2004).

Short-term changes in climate, sea level and sediment dynamics introduces variability into marsh systems and feedback mechanisms (Kolker et al. 2009; van Wijnen & Bakker 2001). This must be accounted for in long-term modeling which must also consider larger scale and long-term controls on wetland communities (Craft, Pennings, et al. 2009). Therefore long-term records of wetland elevation and accretion dynamics and their response to climate variability and sea level perturbations provide the best information for assessing the vulnerability of mangroves and saltmarsh to sea-level rise and accurately modeling future changes.

1.4 MINNAMURRA RIVER

1.4.1 Geomorphology and Holocene evolution

The Minnamurra River is located on the southeast coast of NSW between Kiama and Shellharbour. Unlike many of the estuaries along the east coast of Australia, the Minnamurra River is a river-dominated estuary under the Roy et al. (2001) classification, this being the mature form of wave-dominated barrier estuaries. There are significant areas of mangrove and saltmarsh, 0.94 km^2 and 0.53 km^2 respectively (Ozcoasts 2011). The climate of this region is temperate, with an average annual temperature range between 9.8°C and 24.4°C and annual rainfall of 1278 mm yr^{-1} (Bureau of Meteorology 2011).

There have been five stages of estuary evolution recognised for Minnamurra as it has evolved from a wave-dominated system 8500 years ago to its present river-dominated form (Panayotou 2004). Throughout these five stages the estuary has infilled with marine and fluvial sediment according to the Roy et al. (2001) model, although episodes of sporadic and rapid deposition of marine sediment, possibly from a tsunami or very large storm event, is also found in the sedimentary record (Haslett et al. 2010). As a result, what was once a mud basin was transformed into extensive swamp accompanied with an increase in floodplain area and a meandering fluvial channel (Panayotou et al., 2007). In the late 1800's and early 1900's a portion of the floodplain was drained and part of the river channel was straightened (Haslett et al. 2010; Panayotou 2004). Interestingly foraminiferal evidence suggests that mangrove and saltmarsh environments were not present through most of the Minnamurra River's Holocene development, perhaps due to the rapidity and sporadic infilling of the estuary (Haslett et al. 2010).

The presence of enigmatic and sedimentologically unique sand sheets seen at Minnamurra River, Dunmore embayment and Killalea Lagoon (both to the north), has been proposed as evidence for a tsunami by a number of authors (Haslett et al. 2010; Switzer & Jones 2008; Switzer et al. 2005). At Minnamurra River these sand sheet overwash deposits drape back-barrier sequences and their sedimentological characteristics suggest a tsunami or large storm event as opposed to a large flood (Haslett et al. 2010). However, for all these sand sheet deposits, a storm event seems unlikely as it would have to be exceptionally large, several orders of magnitude larger than any observed in recorded history along this stretch of coastline (Switzer et al. 2005).

1.4.2 Estuary status and current management

Estuaries are primarily managed under state legislation. The NSW Office of Water (2011) lists the 'Estuary Value' of Minnamurra River as 'high' in their report card for the Minnamurra River Management Zone, due to the number and variety of species present in the catchment. The former Department of Natural Resources provided an inventory of general characteristics of the Minnamurra River including physical features, management issues, tidal and hydrological characteristics and vegetation types and distribution. This information is now managed by the NSW Office of Environment and Heritage. The NSW Office of Environment and Heritage administers legislation relating to NSW estuaries including the Catchment Management Authorities Act (2003), Threatened Species Act

(1995) and the Coastal Protection Act (1979). Other relevant legislation includes the NSW Environmental Planning and Assessment Act (1979) and associated planning tools including State Environmental Planning Policy 14 Coastal Wetlands, and the NSW Fisheries Management Act (1994).

The Minnamurra River Estuary Management Plan was one of the first to be prepared in NSW and has thus undergone review in 2003. In this review an action to “assess the status of wetlands with possibility of extending protected wetland areas and provide fringe buffer zones” was listed as a ‘medium’ priority. The management of wetland and other vegetation types has been recurring though both the previous and subsequent management plans and concern has been focused on increases in bank erosion along the Minnamurra River, especially along Riverside Drive which runs parallel to the estuary on the eastern side opposite the study site. In light of the fact much of the recommended bank stabilisation has been completed, and that the redirection of the Princes Highway over Terrigong Swamp has also been completed, another review of the Minnamurra River Estuary Management Plan seems required.

1.5 Aim

This project aims to examine the response of mangroves and saltmarsh at the Minnamurra River, to sea-level rise in order to assess their vulnerability and provide a firm basis on which to develop climate change adaptation policy and management for the future.

CHAPTER 2: METHODS

2.1 INTRODUCTION

Assessing the vulnerability of coastal wetlands to sea-level rise requires measurements of wetland surface elevation dynamics and mapping of changes in vegetation distribution. This information can then be incorporated in models, which may be developed using various techniques providing insight into the future response of wetlands to projected sea-level rise. This chapter outlines the methods used to measure wetland surface elevation dynamics, map changing vegetation zones and develop a model to forecast the response of wetlands on the Minnamurra River to sea-level rise.

2.2 STUDY SITE

The Minnamurra River (34°38'S, 150°52'E) is located on the south-east coast of NSW between Bass Point and Kiama, approximately 90km south of Sydney (Fig. 2.2.1). It has reached the mature stage of a wave-dominated barrier estuary in the geomorphological classification by Peter Roy (Roy et al. 2001). Tidal influence extends upstream as far as Terragong Swamp and there are significant areas of mangrove and saltmarsh, 0.94 km² and 0.53 km² respectively (Ozcoasts 2011), fringing the lower estuary.

The climate of this region is temperate, with an average annual average minimum temperature of 9.8°C and annual average maximum of 24.4°C. Average annual rainfall is 1278 mm (Bureau of Meteorology 2011). The catchment area is relatively small (110km²) and includes the tributaries Rocklow and Wrights Creek. Much of the upper catchment is used for agriculture but a significant portion also lies within the Budderoo National Park. Significant areas of residential land exist in the lower catchment although there are areas of natural coastal vegetation. A waste disposal plant also lies adjacent to Rocklow Creek.

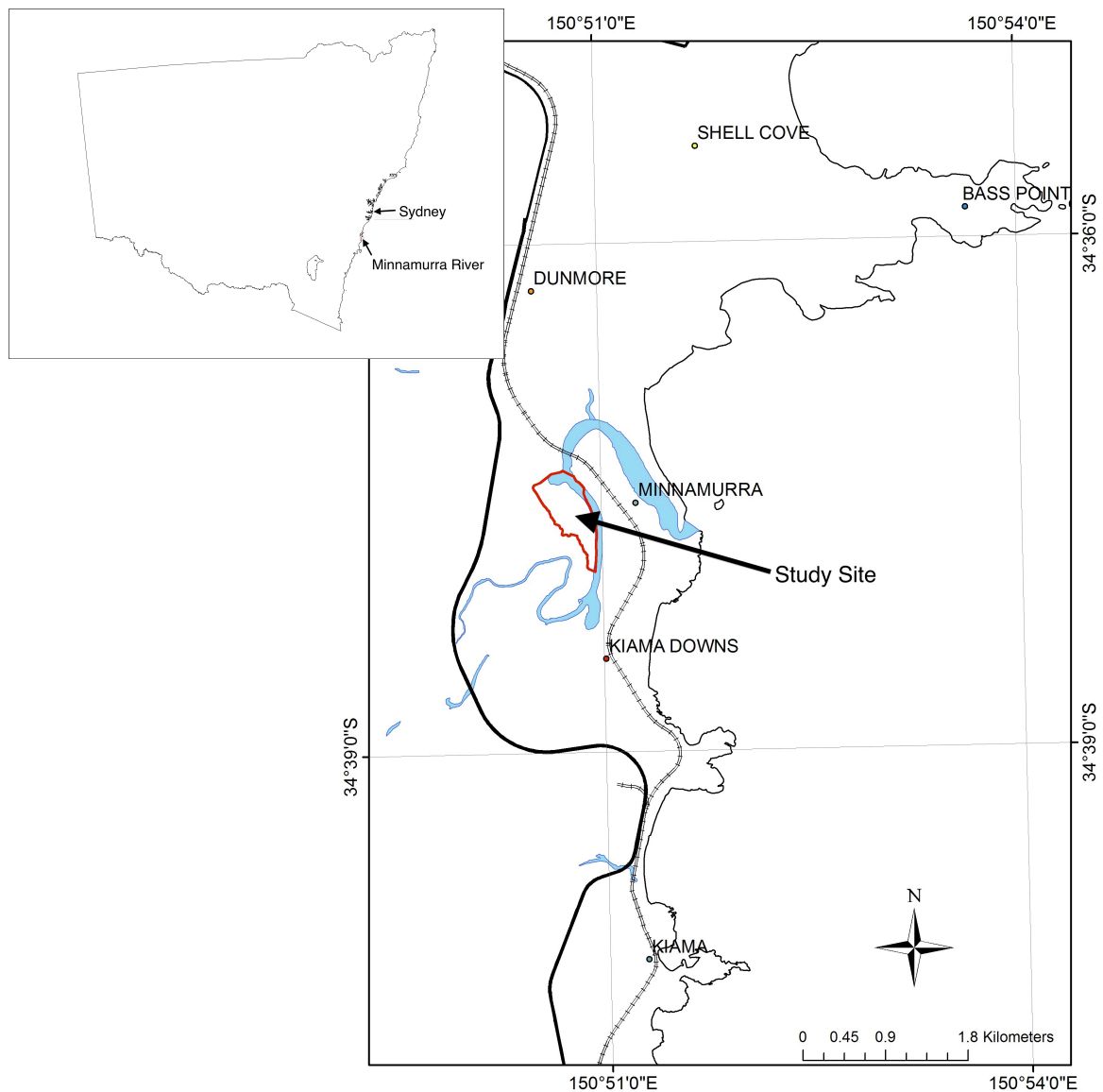


Figure 2.2.1 Map of Minnamurra River and surrounding locations showing the position of the study site. Inset map shows Minnamurra River in relation to Sydney.

The study site lies on the western bank of the river just upstream from the railway bridge (Fig. 2.2.1). It extends, along the westward shore of the river for about one kilometre upstream of the vehicle bridge (Fig.2.2.2). Westward the site extends to the farmland in the north, and in the south to the transition from *Casuarina* to sclerophyll forest. The site shows a typical landward succession of mangrove, saltmarsh and casuarina with the exception of the southeastern edge where a stand of *Casuarina* exists on a sand levee bank (Fig.2.2.2).

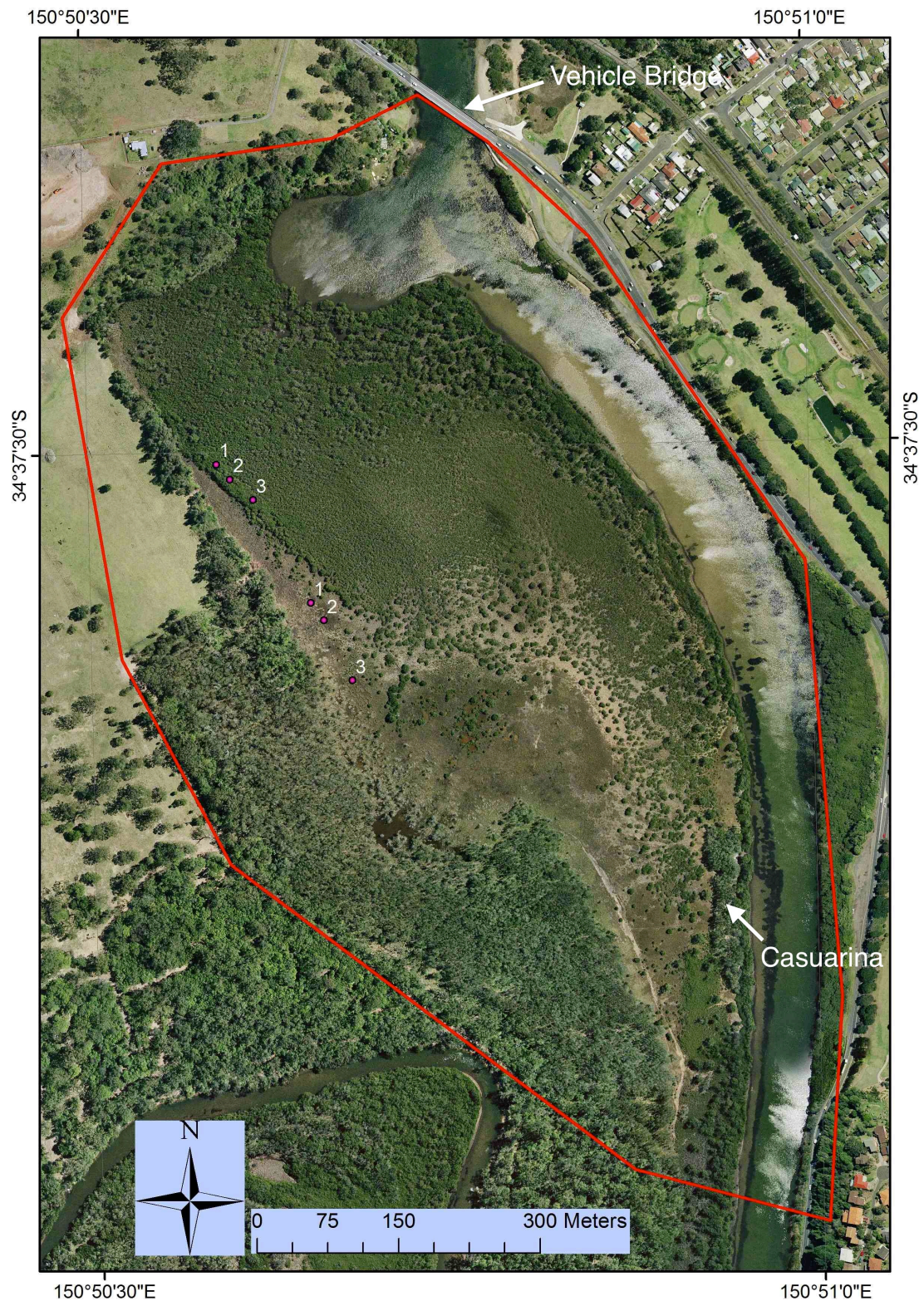


Figure 2.2.2 Detailed aerial photo of the Minnamurra River showing the position of the mangrove and saltmarsh SET sites (numbered 1 to 3 in the mangrove and saltmarsh) in the context of the study site which is defined by the red boundary line.

2.3 MEASURING SURFACE ELEVATION AND ACCRETION DYNAMICS

2.3.1 Surface Elevation Table technique

The Surface Elevation Table (SET) is apparatus from which it is possible to measure surface elevation changes through time (Fig.2.3.1). Version 4 of the SET (Cahoon et al. 2002) used in this study, consists of a number of components some of which remain *in situ* and others which are carried to the site. The *in situ* component of the SET consists of an aluminum pipe which is driven 3-6m into the marsh surface and the ‘base support pipe’ which is cemented into this aluminum pipe. These *in situ* elements were constructed when the SET plot was created by (Rogers 2004) and are not adjusted during the sampling period and thus they occupy a fixed position in space. This base support pipe then accommodates the rest of the apparatus which is brought to the site.

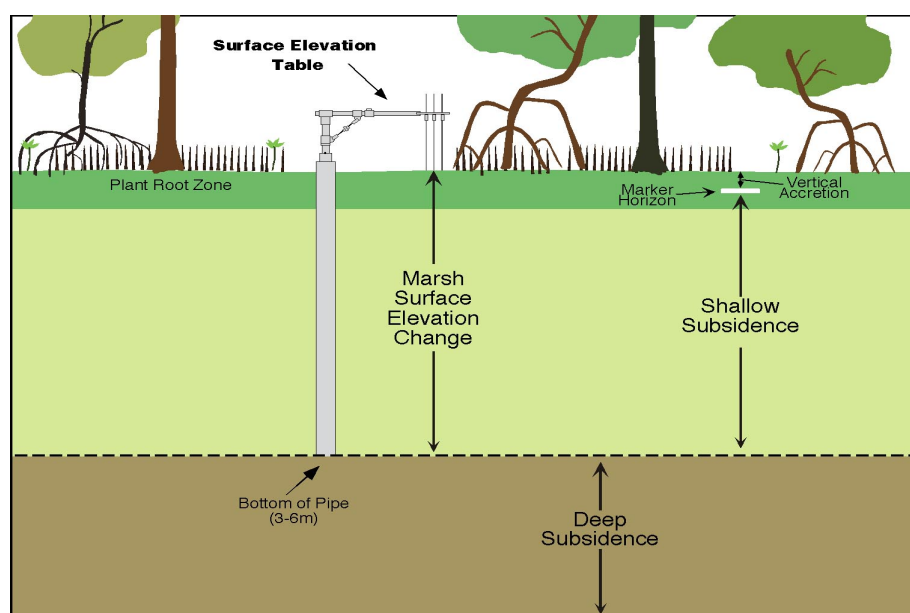


Figure: 2.3.1 Surface Elevation Table (SET) showing the elements of the apparatus including the core pipe, horizontal arm and pins.

The horizontal arm with the plate attached, nine fiberglass pins (5mm diameter and numbered), nine clips and a ruler are carried to the site to complete a measurement. The same equipment was used throughout the study period to avoid any error that may be introduced by changing equipment. The horizontal arm slots into a series of grooves in the base support pipe in different directions allowing different directions to be measured at each SET site. This arm is leveled with a spirit level in preparation for a measurement. The plate attached to the end of the arm has nine numbered holes into which the nine

numbered fiberglass pins are inserted until they touch the marsh surface at which point they are clipped into position so that the weight of the pin does not compress the marsh sediment surface. The height of the pin above the plate is then recorded. Consecutive measures at each SET site allow the relative change in the height of the marsh sediment surface to be examined.



Figure 2.3.2 Image showing the operation of the SET. The pins are being inserted into the plate and clipped to position. (Image courtesy of K. Rogers).

Errors involved with SET technique have been documented extensively (Cahoon et al., 2002). In the latest version of the SET, Version 4 (used in this study), Cahoon et al., (2002) reported a 95% confidence interval of ± 1.4 mm (± 0.7 mm to ± 1.9 mm) and a significant reduction in measurement error from previous versions. As error can be introduced when measurements are taken in different tide and weather conditions care was taken to only measure at low tide and in fair weather.

The sampling design for this site has elements of a nested design common in ecology. At the study site there are six SET sites: three in the mangrove and three in the saltmarsh (Fig. 2.2.2). For each site four directions are measured and for each direction there are nine pins. Thus a total of thirty six measurements are taken for each SET. There being three SET in the saltmarsh and three in the mangrove this gives a total of 216 measures for

each recording day. Thus the sampling design incorporates fine scale measurements with many replicates over a broad area.

Six SET's were established at this site in 2001 by Rogers (2004). Measurements were taken in September 2001, October 2002 and August 2003 (Rogers 2004). Follow up measurements were then taken in September 2009 and April 2010 by K. Rogers. As part of the subject MARE393, the author took measurements in August, September and October 2010. Measurements continued as part of this project in November and December 2010 and then February, March, April, May and June of 2011. All this data has been collated and analysed for this project.

2.3.2 Marker Horizons technique

Marker Horizons (MH) are a simple technique to measure vertical accretion in mangroves and saltmarsh that has shown to be a viable alternative to more complex isotope dating (Cahoon & Turner 1989). The technique is simple and involves placing a layer of feldspar, a white powdery substance that is distinguishable from the soil, in a marked 25cm² plot. This must be done at low tide and under favorable conditions to avoid re-suspension and removal by tides. Upon returning to the site, a small core is removed from the plot and the volume of sediment above the marker is measured with a ruler (Fig.2.3.3).



Figure 2.3.3 Image of a core from a MH. The volume of sediment above the marker is being measured. (Image courtesy of K. Rogers).

For this project, three MH were situated around each SET site so that a total of 18 plots were established at Minnamurra River. In this way the SET and MH are coupled, allowing better comparison. However as reported in Cahoon & Turner (1989) success at finding a clear and distinct horizon declines over time, in some cases to less than 60% success after 12 months. A similar problem also occurred for our MH plots at Minnamurra River. In the saltmarsh success declined below 50% and in some measures two or only one horizon could be found out of nine. Reasons for this significant decline in success are most likely a combination of bioturbation by crabs and bivalves and missing the horizon during coring operations (Cahoon & Turner 1989) even though at least three attempts were made in each plot to find the feldspar layer. The feldspar was re-laid in September 2009, as little if any remained from 2001. For the saltmarsh plots, although discovering the horizons was difficult, there is enough data to present meaningful results.

In the mangrove plots even less success was had. When the new feldspar horizon was laid in September 2009 none could be found in any of the plots in April 2010. Another attempt was not made to relay the feldspar. Thus the MH data remains incomplete for mangrove.

2.3.3 Data Analysis

SET and MH data from Minnamurra from 2001 to 2003 has been published (Rogers et al. 2006; Rogers 2004) and the same method of data analysis has been adopted for this study. For the SET data, incremental (change from the previous measure) and accumulative (sum the increments giving surface elevation relative to the baseline) data was extracted from the raw values. Values outside two standard deviations of the mean were regarded as outliers and removed during the calculation of incremental data. These outliers were largely attributed to pins being placed on crab mounds or burrows or upon mangrove roots. Rates of change between measurements are calculated by standardising the incremental data against time. Overall rates of change in mm yr^{-1} are calculated from the slope of the surface elevation and vertical accretion trend line.

MH data analysis involves formatting the raw data to remove outliers (± 2 standard deviations of the mean) and then standardizing these values to get rates of vertical accretion in mm yr^{-1} . A Repeated Measures ANOVA was conducted in JMP on rates of change for surface elevation and vertical accretion to see if these varied significantly over the study period. Surface elevation change was compared between each SET in the

mangrove and saltmarsh. Surface elevation change was also plotted against accretion to assess any correlation between these measures.

2.4 SPATIO-TEMPORAL COASTAL VEGETATION MAPPING

2.4.1 Chafer's work continued

Chafer (1998) completed photogrammetric mapping at Minnamurra River using historical aerial photography from 1938-1997. Chafer (1998) demonstrated mangrove encroachment into saltmarsh at this site and proposed sea-level rise as a driver of these changes. An aim of this project was to continue this mapping of vegetation zones from aerial photography flown in 2003 and 2011, using the mapping protocols outlined by Wilton (2002), in order to examine more recent changes in mangrove and saltmarsh distribution and to further investigate proposed links with sea-level rise. With the advances of camera technology, aerial photography is now at much greater resolution and can be processed and rectified to a much higher spatial accuracy. Mapping from aerial photography provides the most accurate assessment of coastal vegetation patterns (Manson et al. 2001).

Chafer (1998) classified mangrove and saltmarsh based on area, whereby a polygon was drawn and the boundary between mangrove and saltmarsh, or between saltmarsh and *Casuarina*, was discriminated based on whether it contained >90% of the particular habitat. This leads does not allow differentiation of ecotones between vegetation types. However with the acquisition of much higher resolution aerial photography, a 'mixed' zone can be delineated and is more appropriate for accurately defining such ecotones. Thus for 2003 and 2011, imagery a mixed zone was defined in addition to mangrove and saltmarsh. These vegetation classes were defined according to the wetland habitat mapping protocols out lined by Wilton (2002).

Wilton (2002) outlined six protocols, two of which are directly applicable to this study:

- 1:5000 scale or larger should be used to delineate mangrove and saltmarsh boundaries in the ecotone,
- (2) The following classification be used to define mangrove and saltmarsh habitats: Mangrove 0-10m canopy gaps, Mixed 10-20m canopy gaps, Saltmarsh >20m canopy gaps. 'Canopy gap' refers to the distance between mangrove trees. With high-resolution imagery that was utilized, mangrove trees as small as $\approx 1\text{m}$

high could be discerned and therefore a ‘mangrove tree’ was defined as any mangrove tree over $\approx 1\text{m}$.

2.4.2 Mapping in ArcGIS

Aerial photography for 2003 and 2011 were acquired with a spatial resolution of approximately $20\text{cm} \times 20\text{cm}$ and $6.4\text{cm} \times 6.4\text{cm}$ pixel size respectively, the later being acquired from Nearmap™. Both images were already preprocessed to very high accuracy ($<1\text{m}$) and georectified to MGA Zone 56. ArcGIS was used to view the aerial photography and to digitize polygons defining each wetland community.

All mapping was done at a larger scale than 1:5000, however the high-resolution imagery allowed viewing of the image at $<1:1000$ which assisted in delineating boundaries. During the polygon digitisation process a $10\text{m} \times 10\text{m}$ grid was overlaid in order to make adherence to the second recommendation by Wilton (2002) easier and less time consuming.

2.5 MODELING WETLAND RESPONSE TO SEA-LEVEL RISE

Developing a model of coastal wetland response to sea-level rise included a number of steps. Initially a Digital Elevation Model (DEM) was developed, followed by the construction of models calculating wetland elevation response to sea-level rise through time.

2.5.1 Making the DEM

A DEM is an interpolated elevation surface on which the model will calculate and define the interaction between sea-level rise, surface elevation change and mangrove and saltmarsh distribution through time.

GPS points were collected at the study site on the Minnamurra River in April 2011. This involved the use of Real Time Kinematic (RTK) Global Positioning System (GPS) equipment which has a mean accuracy of 1.7cm vertical and 1cm horizontal. Over 300 points were collected over a period of approximately 5 hours. The RTK GPS is referenced to the Survey Control Information Management System (SCIMS) where SCIMS ‘marks’ have known coordinates thus checking the accuracy of the device. The ‘Base Station’ component of the RTK GPS is set up on top of a SCIMS mark and the survey was started and ended on other SCIMS marks in the local area to ensure the accuracy of the survey

and reference GPS points to AHD (sea level). These points were displayed in ArcGIS and a DEM was constructed using the Triangulated Irregular Network (TIN) method, which calculates elevations across triangular facets that are interpolated between points.

A DEM can also be made from optical remote sensing technology such as airborne Light Detecting And Ranging LiDAR, which is an excellent source of topographic data even in low relief areas such as coastal wetlands (Rayburg, Thoms, & Neave 2009). LiDAR was acquired for this site at Minnamurra from the Land and Property Management Authority (LPMA). Unfortunately the data was flown so recently that it had not been processed so as to extract ground level from intermediate levels where the laser was reflected off vegetation. To make use of this data therefore, LiDAR points were selected that could either be seen to be ground returns when compared with the aerial photo imagery, or where they corresponded to known ground elevation ranges within the site based on the RTK GPS data. The Root Mean Squared (RMS) error between the GPS and the selected LiDAR points was 9.3 cm.

From the combination of RTK GPS data and LiDAR data the Inverse Distance Weighted (IDW) interpolation was used to create another DEM. Thus there were two surfaces on which wetland elevation models were built. The first created from the RTK GPS data alone using the TIN method, and the second created from the combination of the RTK GPS data and selected LiDAR data using the IDW method.

These surfaces were exported as raster data files ready to be used for modeling. Note that differences in the shape site are simply the result of different clips of the TIN or IDW grids and are related to the use of different data inputs.

2.5.2 Sea-level rise scenarios used for modeling

Two IPCC sea-level rise scenarios were used to develop wetland elevation change models for this site. These were selected from the IPCC Fourth Assessment Report: Climate Change 2007 (AR4). The lowest scenario (B1 5% Confidence Interval (CI)) and the highest scenario (A1FI 95% CI) were chosen (Fig.2.5.1) to capture the range of sea-level rise projections and examine their effect on coastal wetlands at the study site.

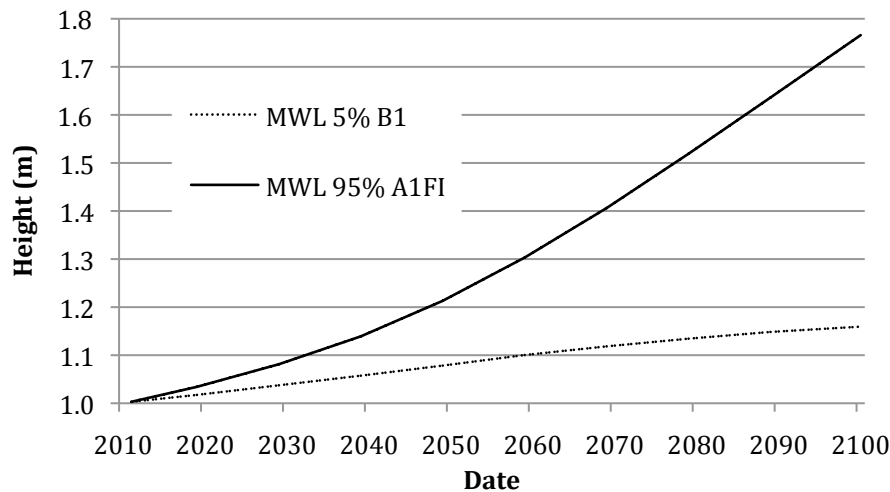


Figure 2.5.1: IPCC sea level rise scenarios used for modeling. A1FI at the 95% confidence interval (CI) and B1 at the 5% CI. Mean Water Level (MWL) starting point is 1.003m.

The starting value from which these scenarios progress was the mean sea level in April 2011 at the Port Kembla tide gauge (1.003m). This was the starting date for all the models because the GPS data was collected in April 2011 and thus the model calculates elevation change from the absolute elevation in April 2011.

2.5.3 Simple Vegetation Model

As a ‘pilot study’, for the purpose of comparison, and to understand the interaction of model variables in ArcGIS, a Simple Vegetation Model was developed. This model used a fixed rate of elevation change for each vegetation type: mangrove, mixed, saltmarsh and *Casuarina* (See Table 2.5.1). These values were chosen to reflect observed SET elevation trajectories at this site and based on the assumption that surface elevation change is greater in areas that are more frequently inundated.

Each vegetation type was considered to occur only within a specific elevation range based on the 2011 mapping polygons (Tab. 2.5.1). The elevation range was defined as the mean \pm 1 standard deviation. Where upper or lower bounds overlapped an average was used as the break height. However to define the upper boundary of the saltmarsh zone no average was taken even though this value overlapped with the lower *Casuarina* boundary. This decision was made because it could be clearly seen that saltmarsh occurred up to this height from the 2011 mapping.

Table 2.5.1 Vegetation types showing fixed elevation change used for the Simple Vegetation Model, and upper and lower elevation break values used for the classification of vegetation of the DEM in all models.

Vegetation Type	Fixed elevation change	Lower break value	Upper break value
Water	0mm/yr	-	0m
Mangrove	8.2mm/yr	0m	0.567m
Mixed	5mm/yr	0.567m	0.684m
Saltmarsh	1.8mm/yr	0.684m	0.877m
Casuarina	0mm/yr	0.877m	-

This model was run in ten-year increments up to 2100 in ArcGIS ©ESRI. The DEM created from the RTK GPS data using the TIN method provided the initial 2011 elevation surface, and the IPCC A1FI 95% CI sea-level rise scenario was used (Fig. 2.5.1). Initially the distribution of vegetation over the DEM was classified based on elevation as shown in Table 2.5.1. Sedimentation was added to each pixel/cell based on the vegetation types (using rates in Table 2.5.1), which created a new elevation model for 2020. This was then reclassified in terms of each vegetation type, based on the new elevation range, defined as the initial elevation break value for each vegetation class, plus the sea-level rise modeled to occur between 2011 and 2020. Thus the upper bound of the mangrove zone which was 0.567m in 2011, becomes $0.567\text{m} + 0.0359\text{m}$ (0.6029m) for the 2020 classification because the sea level has risen 0.0359m between 2011 and 2020 according to the IPCC A1FI 95% CI sea-level rise scenario. This process was repeated in decadal increments up to 2100.

2.5.4 Factorial analysis model

Factorial analysis using JMP statistical software was applied to a range of possible factors that may influence wetland surface elevation changes at Minnamurra River. Factors that were used within the analysis included vertical accretion and surface elevation as well as a range of environmental factors (Tab. 2.5.2). Some of these environmental factors incorporate a time lag, as it has been suggested that such a lag exists between elevation and accretion response and environmental controls (McFadden et al. 2007).

Table 2.5.2 Factors used in the construction of a factorial analysis model in JMP.

Vertical Accretion: Observed accretion from the MH for each month.
Absolute Elevation: The elevation of each SET as recorded from the RTK GPS survey from which the SET data is added and subtracted through time forwards and backwards from the GPS survey date.
Surface Elevation: The accumulative elevation change over time as recorded by the SET.
Rainfall Index 1 (RFI1): Average rainfall for the previous month from four weather stations (Blackbutt (Tarmma Place) to the north, Albion Park Post Office to the north west, Jambaroo (The Ridge) to the west within the catchment, and Kiama Bowling Club to the south.
Rainfall Index 2 (RFI2): Average rainfall for the previous three months from the same stations mentioned above.
Average Annual Rainfall (AAR): Annual rainfall for the previous year also from the stations above.
Southern Oscillation Index 1 (SOI1): SOI value for the previous month
Southern Oscillation Index 2 (SOI2): SOI average for the previous 3 months
Southern Oscillation Index 3 (SOI3): SOI value for the previous 6 months
Mean Water Level (MWL): Average water level for the previous month from the Port Kembla Tide gauge.
6 Month Mean Water Level (6MWL): Average water level for the previous 6 months from the Port Kembla Tide gauge.
Mean Maximum Water Level (MMWL): Average maximum water level for the previous month from the Port Kembla Tide Gauge.
6 Month Mean Maximum Water Level (6MMWL): Average maximum water level for the previous 6 months from the Port Kembla Tide Gauge.
Distance to Shore (DTS): The distance from each SET MH site to the shoreline (edge of the mangrove).
Time: Time in days since the first SET MH measure at the site.

Initially, stepwise factorial analysis was undertaken to identify those factors that best contributed to the observed rates of vertical accretion and surface elevation change. A Standard Least Squares model and a prediction equation were developed using these factors. The prediction equation defines the relationship between these factors, such as water level, rainfall and SOI, and observed vertical accretion and surface elevation at this site. Statistical analysis of the association between vertical accretion and surface elevation, based on the SET MH data, was completed to determine whether there was a statistical difference between accretion and elevation gain at Minnamurra River. This would establish whether two separate equations, one for vertical accretion and one surface elevation, were needed, or if one equation could be used to accurately model wetland elevation change at this site.

2.5.5 Factorial analysis model applied in ArcGIS – spatial models

Following the successful application of the simple vegetation model, the statistical model was applied in ArcGIS to develop spatial models of future wetland surface elevation change. While the simple vegetation model assumed a constant rate of surface elevation change within each vegetation zone, applying the factorial analysis model in ArcGIS to create a spatial model calculates surface elevation change based on defined spatial and temporal variations of environmental factors, enabling a more complex and spatially-resolved model of potential future changes at Minnamurra.

The equations developed from the factorial analysis model were used to create models of wetland elevation change at Minnamurra in response to IPCC sea-level scenarios. The equations calculated the amount of vertical accretion and surface elevation change for each cell of the DEM in 1-year or 10 yearly increments up to 2100. Two spatial models were developed (named FA 1 and FA 2), which allowed comparison of the effects of altering variables, such as the distance to the shoreline and the number of model iterations, and using different DEM surfaces and sea-level rise scenarios (Tab. 2.5.3).

Table 2.5.3: Comparison of the three models: the surfaces to which they were applied, the yearly increment in which they were calculated, the sea-level scenario used and whether distance to shore changed in the model. FA 1 is Factorial Analysis model 1 and FA 2 is Factorial Analysis model 2.

Model	Surface applied	Yearly increment	Distance to shore	IPCC Sea-level rise Scenario
Simple Vegetation model	TIN of GPS points	10 years	n/a (based on vegetation)	A1FI 95% CI
FA 1	TIN of GPS points	1 year	Not changing	A1FI 95% CI
FA 2	IDW GPS & LiDAR	10 years	Changing according to SLR	A1FI 95% CI and B1 5% CI

In order to aid in the development of this model it was assumed that vertical accretion at the back of the saltmarsh zone was equal to zero. This assumption was deemed valid as surface elevation change in this area of the marsh is almost exclusively from *in situ* production of organic material from plant growth.

The application of the factorial analysis model to a spatial model predicting wetland response to sea-level rise, required the factors used for the factorial analysis model to be defined for the future (Tab 2.5.4). As this model is run, a new DEM is created which is the

calculated response of the wetland to the defined value or future value of the factors (Tab. 2.5.4). To map the changing distribution of vegetation, the DEM was reclassified according to the same vegetation classification used for the simple vegetation model, which was delineated from the spatio-temporal mapping (Tab. 2.5.1). The break elevation values of this classification also changed according to the sea-level rise scenario used for each particular model. Prediction maps of vegetation distribution were developed and analysed to understand projected wetland changes at the study site.

Table 2.5.4 The values for factors used for the application of the factorial analysis model in ArcGIS.

Time: in days since the first SET MH measure at the site.
RFI1: kept constant as the average value for April 2011 when the GPS survey was done
6MMWL: increased according to the A1FI 95% CI or B1 5% CI IPCC sea-level rise scenario shown in Figure 2.5.1, although with a starting value of 1.9720m – the Port Kembla mean maximum for April 2011
AAR: kept constant as the average value from all weather stations used in the factorial analysis model
DTS: The calculated distance of every cell in the grid to the shore where shore is defined as the position of 0m AHD sea level in 2011. For FA 1 this was constant through time. For FA 2 distance to shore was recalculated according to sea-level rise for each time increment.
SOI2: kept constant as the overall average SOI value.
MWL: changing through time according the IPCC 95% A1FI or the 5% B1 sea-level scenario (see Fig. 2.5.3)

CHAPTER 3: RESULTS

3.1 INTRODUCTION

This section outlines the results corresponding to the same headings as outlined in the Methods (Chapter 2) section: surface elevation and accretion dynamics, spatio-temporal mapping, and modeling wetland response to sea-level rise. The models integrate the results of the surface elevation and accretion results and the patterns observed in mapping vegetation changes at this site.

3.2 SURFACE ELEVATION AND ACCRETION DYNAMICS

In general terms patterns of mangrove and saltmarsh surface elevation (SE) and vertical accretion (VA) have overall positive trajectories, and over the past 2 years these have become more rapid and pronounced.

3.2.1 Mangrove surface elevation and vertical accretion patterns

Ten-year surface elevation trajectories in the mangrove zone were positive, increasing at a rate of 0.62 mm yr^{-1} (Tab 3.2.1). Incomplete vertical accretion data does not allow comparisons with surface elevation trends. Since September 2009, surface elevation shows a much steeper trajectory: 9.7 mm yr^{-1} (Tab. 3.2.1). Rates of surface elevation change in the mangrove varied significantly through time ($P = 0.0029$).

Inter-annual variability was evident when sampling was undertaken at monthly intervals, as opposed to annual intervals (Fig. 3.2.1). From September 2009 onwards when measurements are more frequent there significant variability is observed. Such variability could not be captured with a yearly measuring program. If the yearly measurement happened to be taken on the 1st of December 2010 (the low value seen in Fig. 3.2.1), this would misrepresent the actual patterns of SE at this site.

Table 3.2.1 Comparison of rates of surface elevation change and accretion across zones and time.

Zone	Surface Elevation (2001-2011)	Surface Elevation (Sep 2009-June 2011)	Vertical Accretion (2001-2011)
Mangrove	0.62 mm yr ⁻¹	9.7 mm yr ⁻¹	-
Saltmarsh	0.29 mm yr ⁻¹	1.9 mm yr ⁻¹	1.4 mm yr ⁻¹

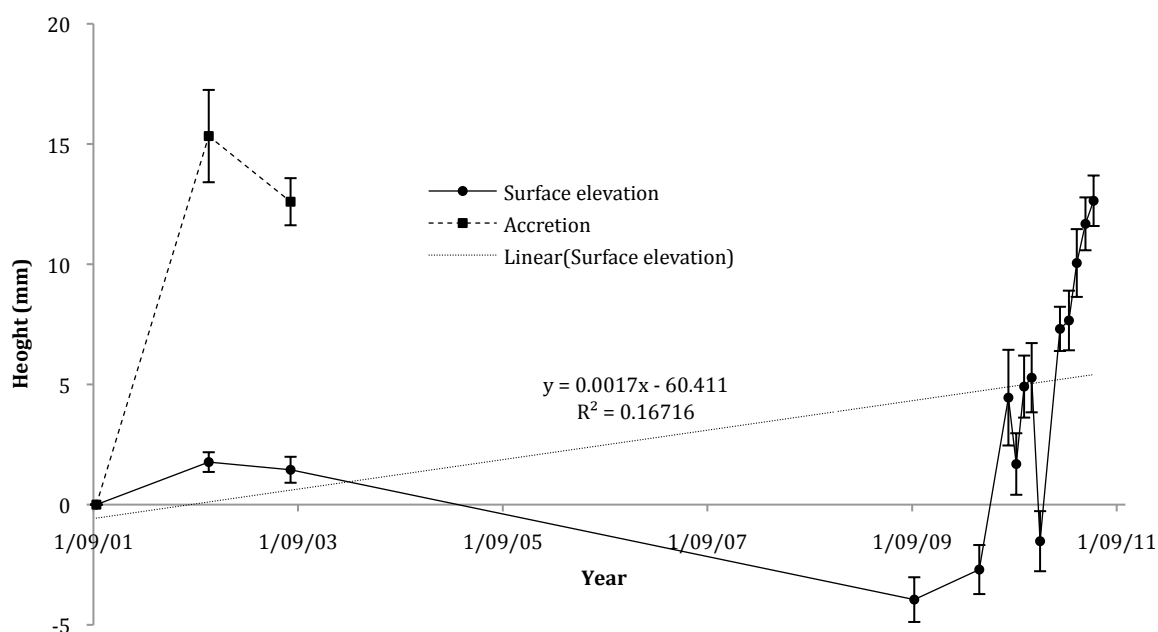


Figure 3.2.1: Mangrove surface elevation change and vertical accretion in millimeters from September 2001 to June 2011. Error bars are ± 1 standard error.

Negative values such as the value September 2009 (-3.95 mm) were observed in the surface elevation record, indicating that relative to the 2001 baseline, when the SET plots were created, mangrove surface elevation trajectory up till 2009 was negative. Following this surface elevation jumps back up higher than that of 2003 and from then onwards there is a clear upward trajectory with the exception of the measurement in December 2010. The last five values from February to June show an unbroken and steep upward trend (Fig. 3.2.2.).

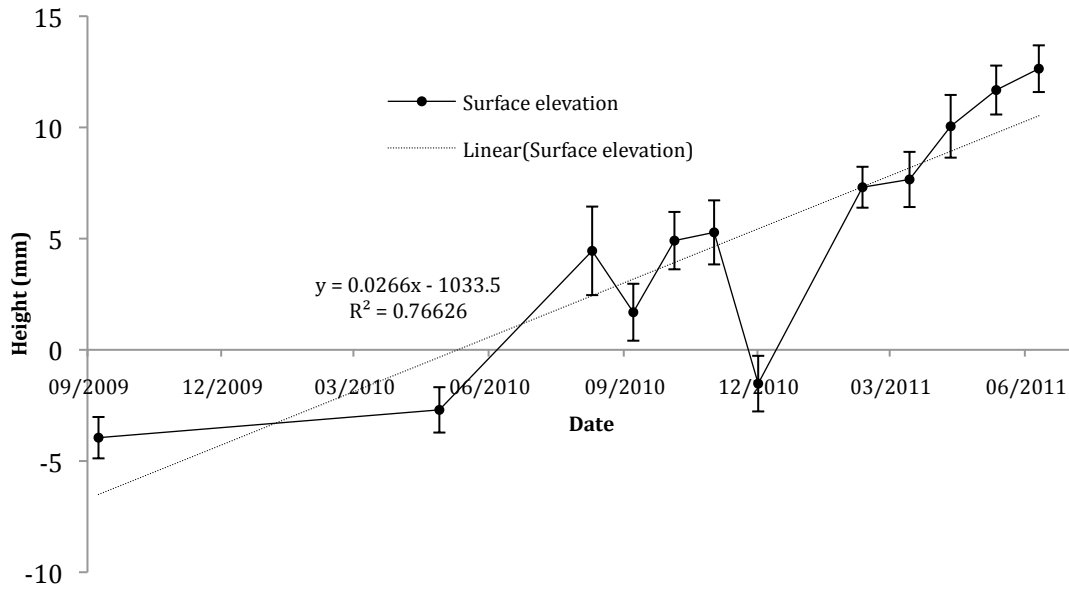


Figure 3.2.2: Mangrove surface elevation in millimeters since September 2009. Error bars are ± 1 standard error.

Comparing the surface elevation patterns of each individual SET in the mangrove zone revealed that values were remarkably similar (Fig. 3.2.5). The significant upward trend seen after September 2009 (when measurement frequency increases) has been observed in all 3 SETs. SET 1 has much higher values than SET 2 and 3 in early 2010 (Fig. 3.2.5).

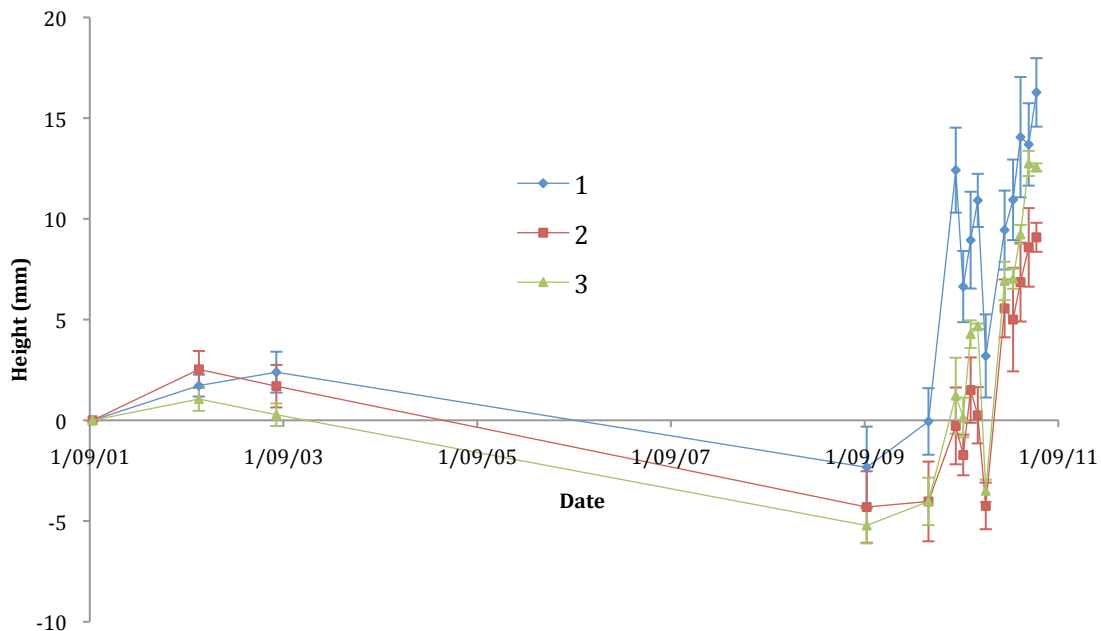


Figure 3.2.5: Mangrove surface elevation change for each individual SET between 2001 and 2011.

3.2.2 Saltmarsh surface elevation and vertical accretion patterns

The ten-year surface elevation trajectory in the saltmarsh showed a positive trend increasing at 0.29 mm yr^{-1} (Tab. 3.2.1). Ten-year vertical accretion trajectory in the saltmarsh had a value of 1.4 mm yr^{-1} . Accretion values stayed well above surface elevation values throughout the study period. Between September 2009 and June 2011 higher rates of surface elevation in the saltmarsh 1.9 mm yr^{-1} were observed (Tab. 3.2.1). Rates of surface elevation change in the saltmarsh varied significantly through time ($P = 0.0029$) as did rates of vertical accretion ($P = 0.0215$). In addition, rates of surface elevation and vertical accretion within the saltmarsh were significantly different ($p = 0.00242$).

Saltmarsh surface elevation showed significant variability when measuring was conducted at monthly intervals, although unlike surface elevation in the mangroves, there are no negative values (cf. Fig. 3.2.1 & 3.2.3). While there is a steeper positive surface elevation trajectory after September 2009 compared to the overall trajectory, this is not of the magnitude of difference seen in the mangrove (Tab. 3.2.1). Vertical accretion in the saltmarsh also shows variability though time (Fig. 3.2.3 & 3.2.4). After September 2009, vertical accretion values vary greatly, more than surface elevation, yet they still have a positive trajectory and never dip below the surface elevation values (Fig. 3.2.3).

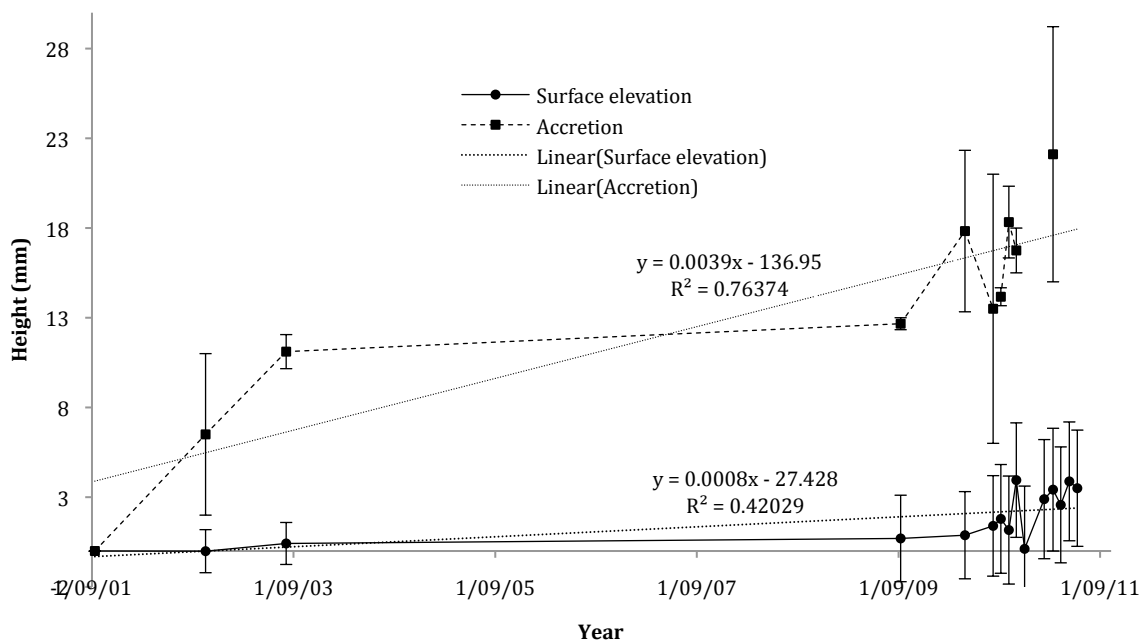


Figure 3.2.3: Saltmarsh surface elevation change and vertical accretion in millimeters from September 2001 to June 2011. Error bars are ± 1 standard error.

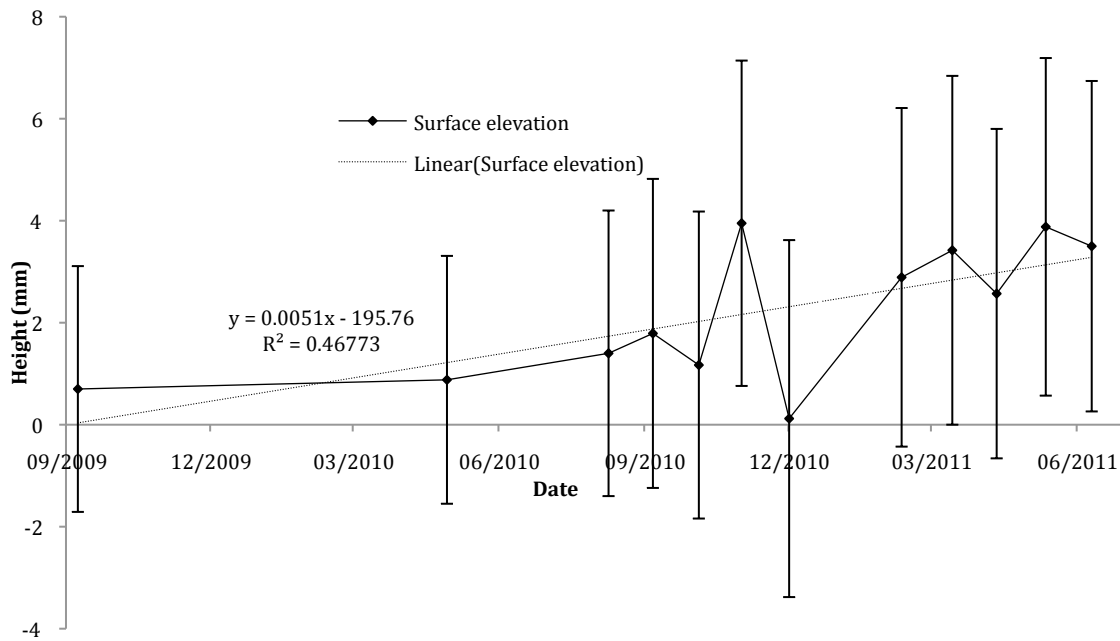


Figure 3.2.4: Saltmarsh surface elevation change in millimeters since September 2009. Error bars are ± 1 standard error.

It must be noted that September 2009 has no significance as a tipping point or threshold in time, rather from this point onwards, there was much greater frequency of measuring. VA and SE values between September 2003 and September 2009 are unknown.

There was no significant relationship between vertical accretion and surface elevation change at the study site at Minnamurra ($p = 0.00242$). However, when comparing surface elevation change and vertical accretion in the saltmarsh, both appear to have a positive trend. The lack of vertical accretion data in the mangrove prevents a similar comparison. Thus this result suggests that values of vertical accretion are not necessarily accompanied by similar changes in surface elevation.

When surface elevation patterns of each SET in the saltmarsh are compared significant differences in trajectory are observed. SET 1 has an upward trend after September 2009 and has values of approximately 5-10mm higher than SET 2 and 3 (see Fig.3.2.6). SET 2 and 3 have flat or slightly negative trajectories. This difference in trajectory is not evident between 2001 and 2003. The overall trajectory of saltmarsh SE in Figure 3.1.3 is thus a product of averaging the three saltmarsh SET. This has also resulted in such a high standard error as can be seen in Figure 3.2.4.

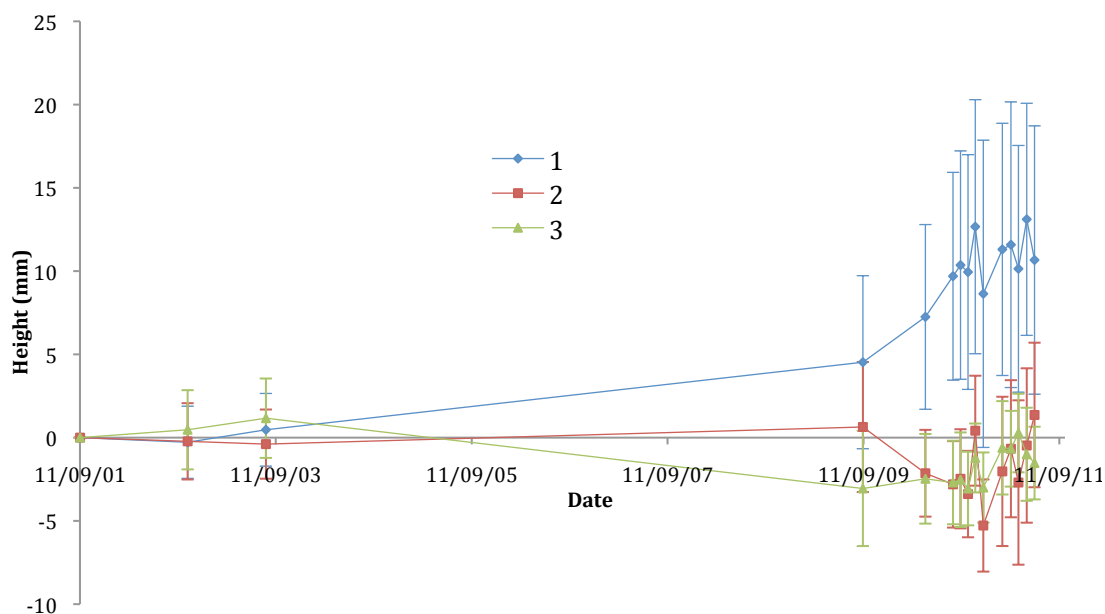


Figure 3.2.6: Saltmarsh surface elevation change for each individual SET between 2001 and 2011.

3.3 SPTIO-TEMPORAL COASTAL VEGETAITON MAPPING

The most obvious trend when comparing the distribution of vegetation zones between 2003 and 2011 is that there is consolidation of mangrove trees resulting in mixed zone in 2003 being classified as mangrove in 2011 (Fig 3.3.1). This trend is most pronounced in the middle of the study site where a large area of mangrove has invaded the saltmarsh zone. The decline of the mixed zone occurred at a rate of $-7.1 \% \text{ yr}^{-1}$ between 2003 and 2011. The area of mangrove increased at a rate of $1.8 \% \text{ yr}^{-1}$ over the same period. Interestingly the area of saltmarsh increased a $0.4 \% \text{ yr}^{-1}$, however as the mixed zone contains mangrove and saltmarsh, the net result has been a loss of saltmarsh at the site.

In 2003 almost all areas of saltmarsh have a narrow mixed zone bordering the mangrove, here referred to as a 'buffer'. However, this is not the case in 2011 where in many places mangrove and saltmarsh abut (Fig. 3.3.1). A similar pattern is also seen at the southern end of the study site where in 2003 there is a thin strip of mixed zone between the mangrove and *Casuarina*. However in 2011 this small strip has disappeared and mangrove abuts the *Casuarina* zone (Fig. 3.3.1). Significant expansion of mangrove was also observed between 2003 and 2011 at the southern end of the study site between the upland *Casuarina* forest and a small band of *Casuarina* that has remained on a levee bank (Fig. 3.3.1).

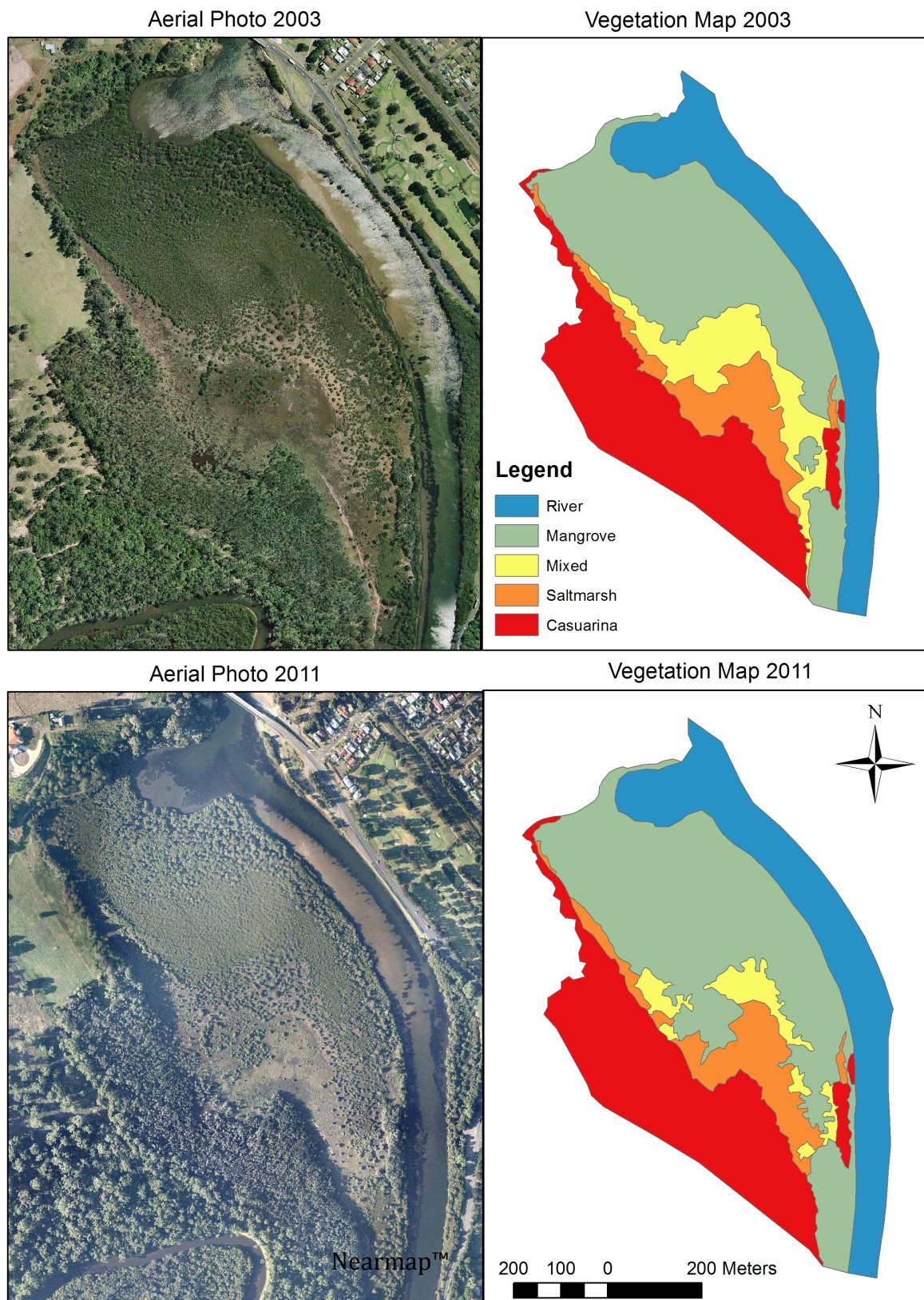


Figure 3.3.1: Vegetation map from the aerial photography of the study site on the Minnamurra River showing water, mangrove, mixed, saltmarsh and *Casuarina* in 2003 and 2011.

3.4 MODELING WETLAND RESPONSE TO SEA-LEVEL RISE

3.4.1 Simple vegetation model

Based on the results of the simple vegetation model (Fig. 3.4.1 & 3.4.2) which used the A1FI 95% CI IPCC sea-level scenario (Fig.2.5.1) it is evident that saltmarsh distribution declines quickly from 2011 through to 2050 after which it has all but disappeared. It is only maintained from 2050 onwards on the southeastern edge of the site where there are small sandy ridges. Likewise *Casuarina* quickly disappears from the southwest of the site and is only maintained at the southeastern edge after 2030.

The mixed zone has an interesting pattern of change through time showing expansion until 2040. This expansion is most pronounced in the middle of the study site and especially in the region of higher elevation which extends north eastward. Mixed zone dominates this area up to 2040. Interestingly this pattern of initial expansion of the mixed zone is not accompanied by an expansion of the saltmarsh zone. After 2040 this expansion is reversed and from 2050 onwards it then shrinks rapidly in a westward direction.

The mangrove zone retreats as the mixed zone expands until 2040. The mangrove zone then expands rapidly from 2050 until 2100. The first region of expansion after 2040 is into the areas in which the mixed zone expanded. In particular, in 2050 the mangrove recovers in the region northeast of the middle of the study site and also expands into the region in between the stand of *Casuarina* near the river in the southeast and the saltmarsh at the westward boundary of the study site. Interestingly the shoreline does not change until 2100 even though mangrove is expanding rapidly from 2050 onwards.

Examining vegetation change trends in two dimensions, the pattern of expansion of the mixed zone and contraction of the saltmarsh until 2040 clearly observed (Fig.3.4.2). After this point, as also demonstrated in Figure 3.4.1 mangrove expansion is rapid and mixed zone decline is inversely so. The upward trend of mangrove flattens after 2070 but has increased by more than 40% between 2050 and 2070.

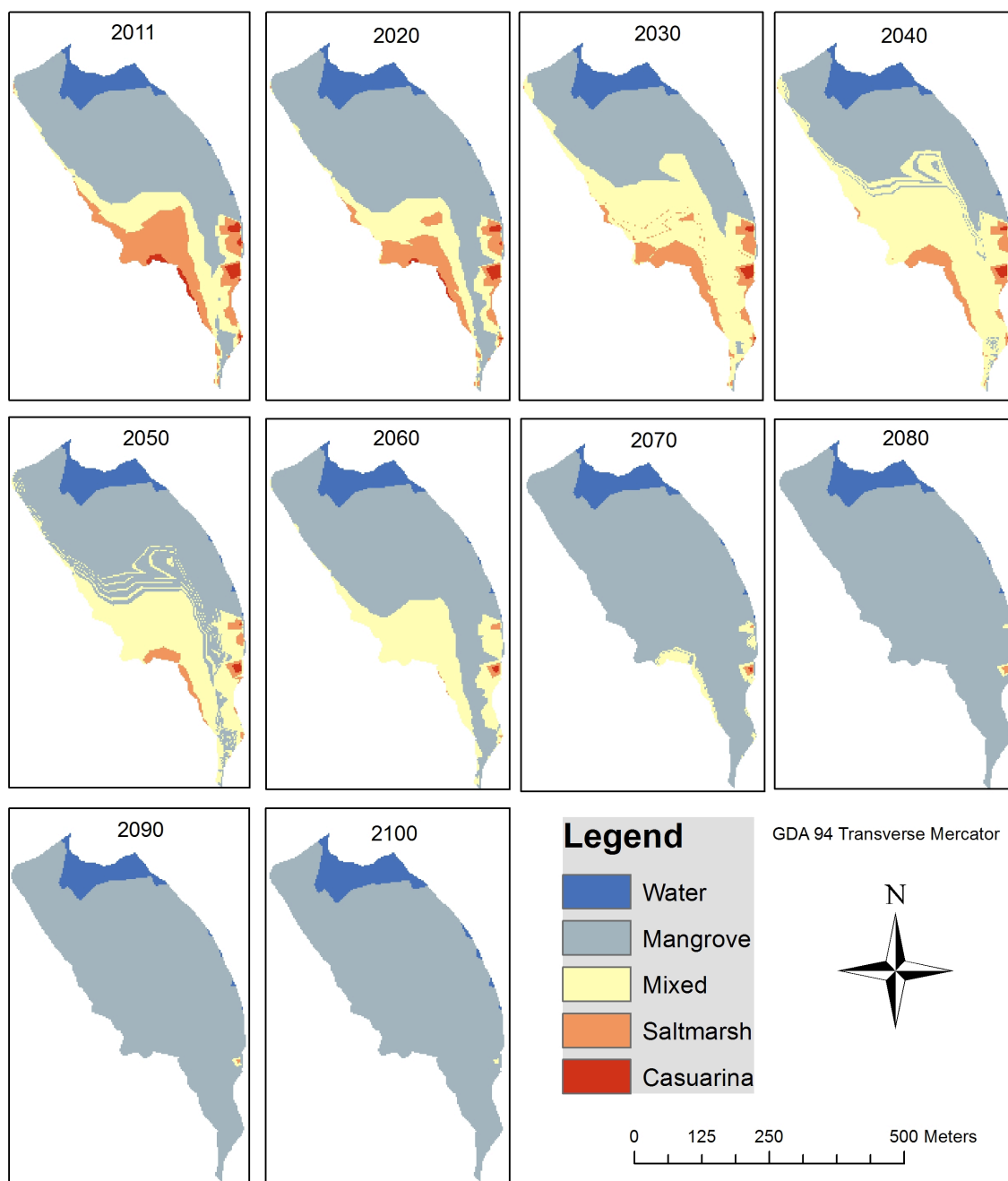


Figure 3.4.1: Simple vegetation model showing the distribution of vegetation zones through time from 2011 to 2100 using the A1FI 95% CI sea-level scenario.

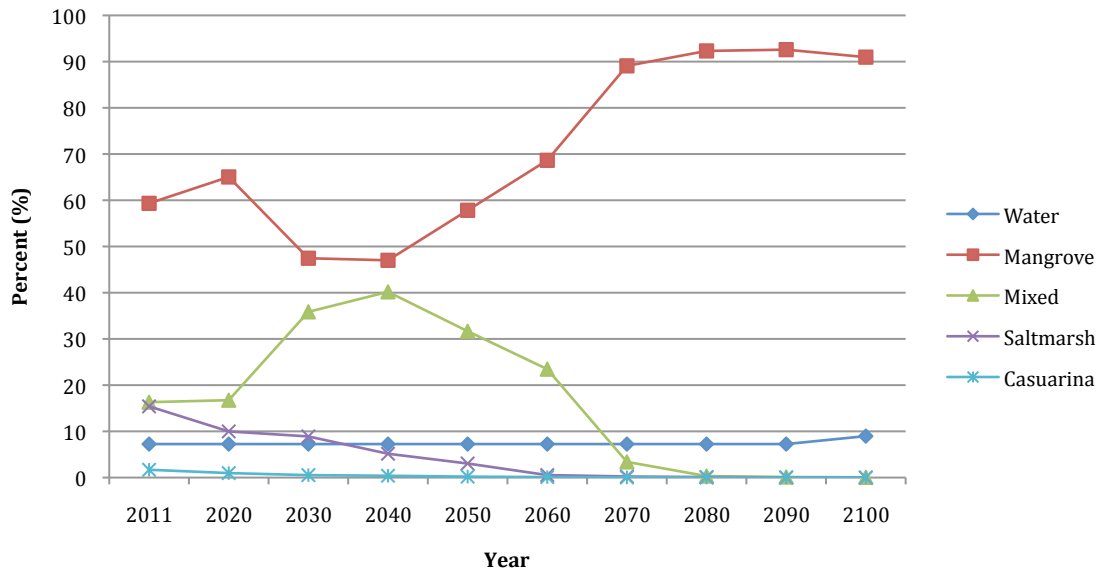


Figure 3.4.2: Percentage area of each vegetation zone at the study site at 10-year increments for the simple vegetation model using the AIFI 95% CI sea-level scenario.

3.4.2 Factorial analysis model

The results of the factorial analysis model in JMP are presented as two prediction equations, which define the relationship between surface elevation change and vertical accretion with the most influential factors that were selected by the factorial analysis. As surface elevation and vertical accretion were not significantly associated two equations were needed to adequately model the wetland response to sea-level rise at this site.

The equations are:

Accretion ($R^2 = 0.73$)

$$\begin{aligned}
 &0.17352121734876 \\
 &+ -0.0178892698368 * \text{Absolute Elevation} \\
 &+ 0.00000490846704 * \text{Time} \\
 &+ -0.000054523569 * \text{RFI1} \\
 &+ -0.0794492957302 * 6\text{MMWL}
 \end{aligned}$$

Elevation ($R^2 = 0.68$)

$$\begin{aligned}
 &0.00295789064853 \\
 &+ -0.019818453413 * \text{Accretion} \\
 &+ 0.0000051543649 * \text{AAR} \\
 &+ -0.0000085746375 * \text{DTS} \\
 &+ -0.0000245006673 * \text{SOI2} \\
 &+ -0.0067155128499 * \text{MWL}
 \end{aligned}$$

3.4.3 Factorial analysis model applied in ArcGIS – spatial models

3.4.3.1 Spatial model: FA 1

The results of this model are presented in Figures 3.4.3 and 3.4.4. The first observation is that saltmarsh and *Casuarina* retreat rapidly southwestward in the first 40 years of the model. After 2050 saltmarsh is only found on the sand levee bank on the eastern edge of the study site (Fig.3.4.4). The mixed zone also declines in a similar manner to the saltmarsh and *Casuarina* although at the south west edge of the marsh it is maintained until 2070. The decline of the mixed zone however is not immediate (as the saltmarsh and casuarina is) and its maintains its percentage area until 2030 after which time it begins its decline (Fig. 3.4.3).

The mangrove zone maintains a consistent extent until 2060, occupying 62% of the study area and then declines rapidly to only 19% by 2100, a loss of 43% (Fig. 3.4.3 & 3.4.4). Until 2060 even though mangrove encroaches westward the net result is neither gain nor loss in percentage area because approximately the same percentage of mangrove area being gained in expansion is being lost at the eastern edge as sea level rises.

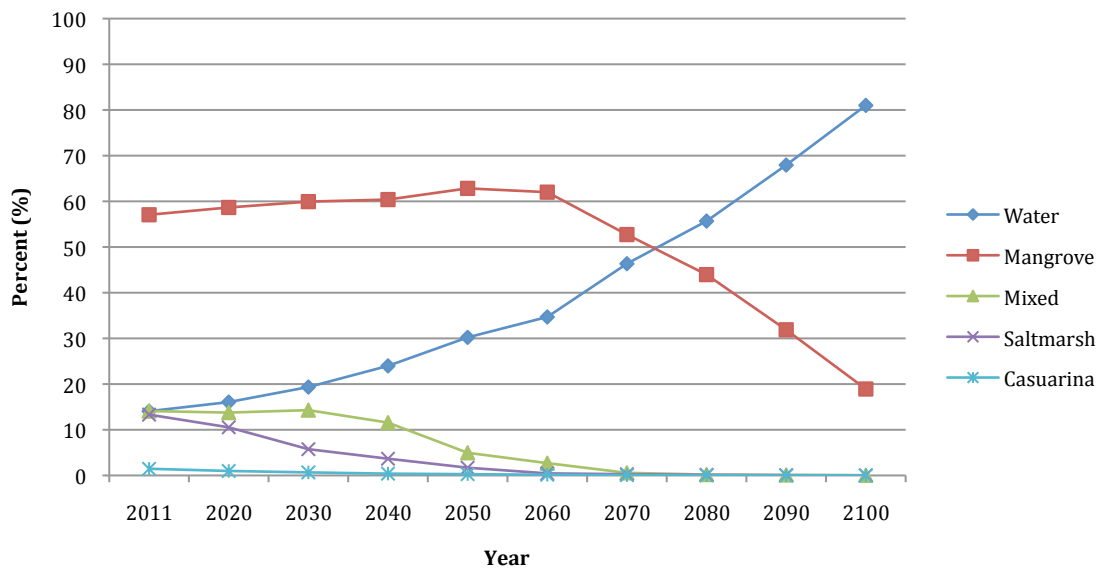


Figure 3.4.3: Percentage area of each vegetation zone at the study site at 10-year increments for Spatial model: FA 1.

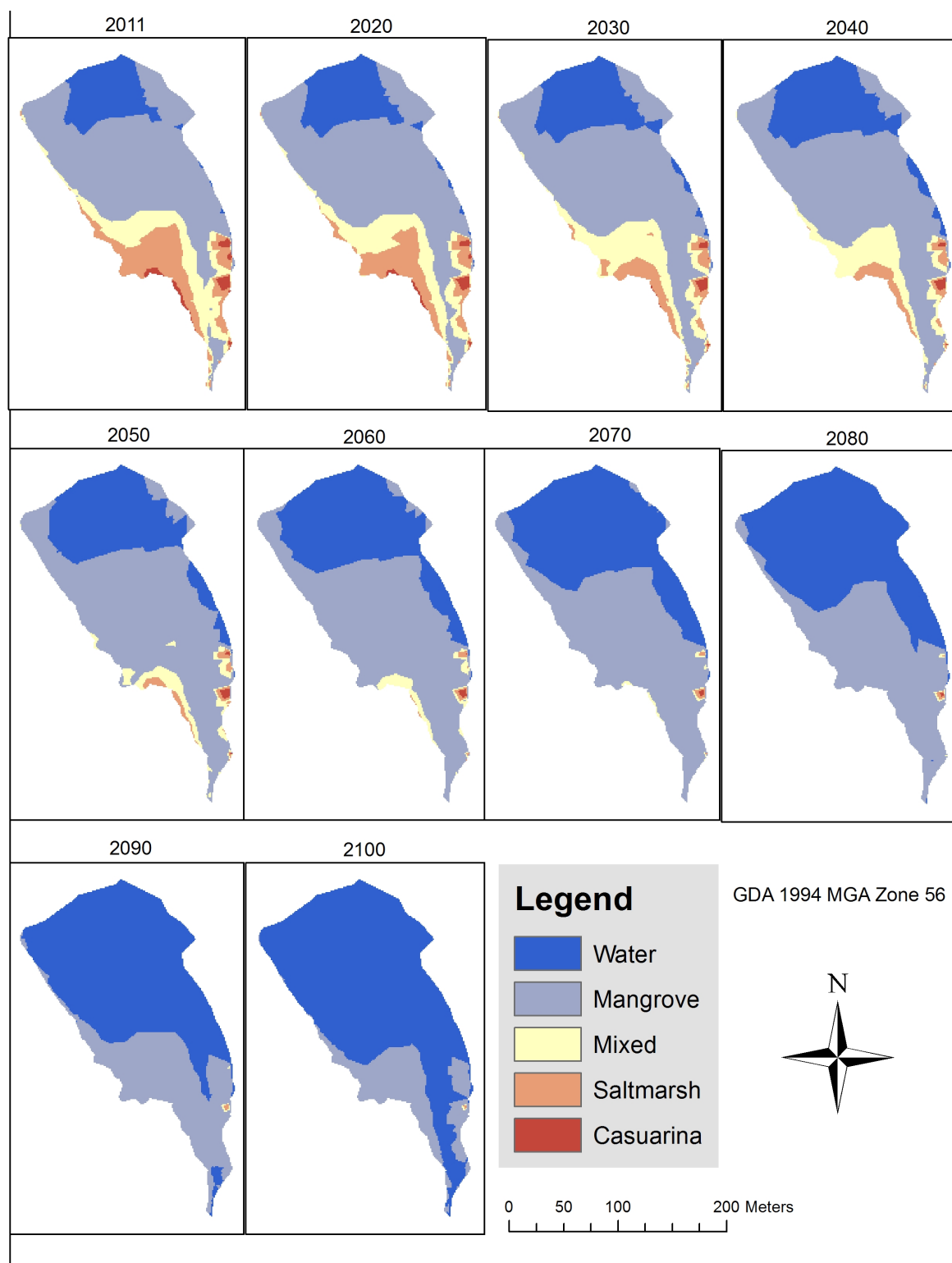


Figure 3.4.4: Spatial model: FA 1 showing the distribution of vegetation zones through time from 2011 to 2100 using the A1FI 95% CI sea-level scenario.

The percentage area of water at the site increases rapidly through time. The point at which water and mangrove are at 50% each is at around 2073. The point at which the water level area curve steepens is at around 2060 which is the point at which the mangrove line drops away rapidly. The spatial pattern of water increase is first of all into the north western portion of the site and then in from the eastern edge of the middle portion of the site. Eventually from 2080 onwards it begins to move southwards in between the sand bank levee on the south eastern side and the westward edge of the site. This same region between the levee bank and the higher ground to the west is also the first area to become mangrove between 2011 and 2040.

3.4.3.2 Spatial model FA 2

This model was run with two sea-level scenarios IPCC A1FI 95% CI and B1 5% CI (Fig. 3.4.5, 3.4.6, 3.4.7 & 3.4.8).

A1FI 95% CI sea-level rise scenario (Fig.3.4.5 & 3.4.6)

Focusing on the water and mangrove zones it is seen that once again, mangrove firstly encroaches into the area between the sand levee bank and the higher ground to the west (Fig. 3.4.5). Unlike the simple vegetation model and FA 1 however, this area is not dominated by mangrove until 2050. Mangrove also begins to move into back marsh area after 2040 and by 2070 dominates this region (Fig. 3.4.5).

Looking at patterns of mangrove area change in two dimensions it is seen that mangrove expands at an increasing rate until reaching a high of 70% in 2070 (Fig. 3.4.6). It then decreases rapidly again to a level approximately 10% higher than its 2011 starting percentage (Fig. 3.4.6). The area of water expands slowly up to 2050 and then far more rapidly up until 2100. The main region of expansion of water after 2050 is in the middle of the site, seen especially in 2080, and then expansion begins rapidly from the middle of the eastern side (Fig. 3.4.5).

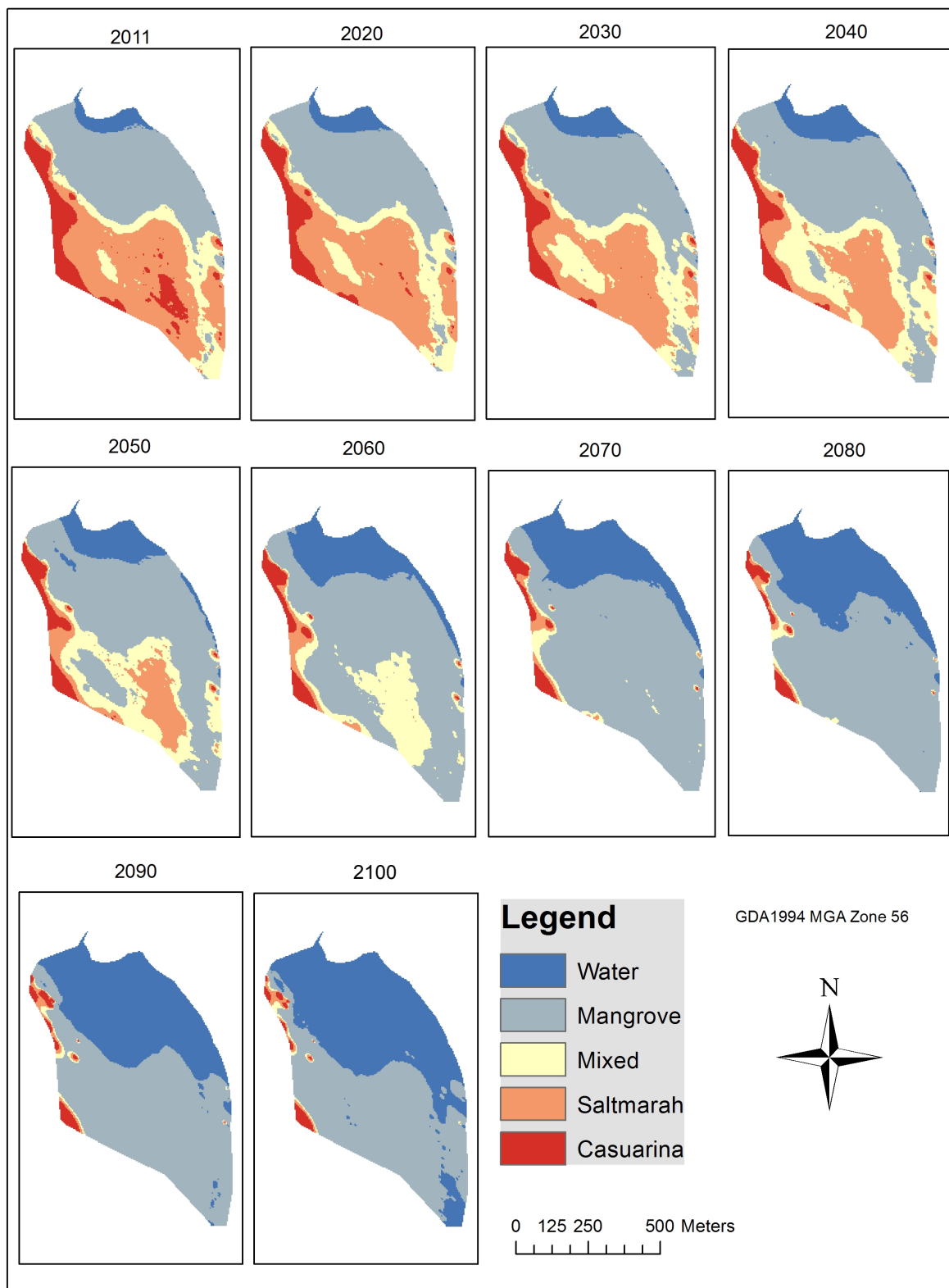


Figure 3.4.5: Spatial model: FA 2 showing the distribution of vegetation zones through time from 2011 to 2100 using the A1FI 95% CI sea-level scenario.

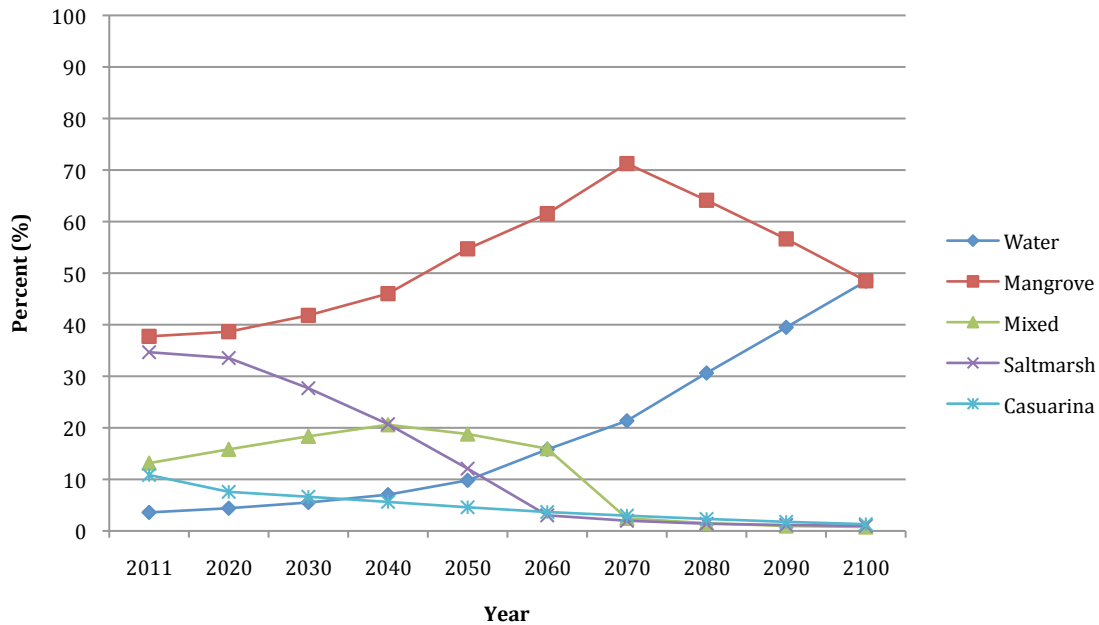


Figure 3.4.6: Percentage area of each vegetation zone at the study site at 10-year increments for Spatial model: FA 2 using the A1FI 95% CI sea-level scenario.

The mixed zone maintains a ‘band’ between the mangrove and saltmarsh through to 2050. Examining percentage area, the model indicates that there is an expansion of the mixed zone until 2040, after which it begins to drop off gradually (Fig. 3.4.6). Then between 2060 and 2070 there is a sharp decline in the mixed zone area (Fig. 3.4.6). Comparing the vegetation maps of 2060 and 2070 (Fig.3.4.5) this loss clearly observed. From 2070 onwards the mixed zone occupied only a thin zone bordering the saltmarsh and *Casuarina* in the northwest of the study site.

Saltmarsh disappears from the middle and south of the site by 2060 after which it fringes the *Casuarina* in the northwest (Fig. 3.4.5). Between 2011 and 2020 there is only a small decrease in saltmarsh area but from then until 2060 it falls steeply (Fig. 3.4.6). *Casuarina* gradually declines and from 2040 onwards is only maintained in the northwest of the site (Fig. 3.4.5 & 3.4.6).

B1 5% CI sea-level rise scenario (Fig. 3.4.7 & 3.4.8)

The overall pattern of vegetation change for this model using the B1 5% CI sea-level rise scenario, is a small loss of saltmarsh and expansion of mixed zone (Fig. 3.4.7 & 3.4.8). In the southeastern corner of the study site there a portion of the mixed region that becomes mangrove. *Casuarina* is lost from the middle southeast of the study site. In comparison with this same model using the 95% A1FI IPCC sea-level scenario there is much less drastic changes in vegetation distributions. In the FA 2 A1FI model, mangrove and water dominated the study site by 2100 (Fig. 3.4.7 & 3.4.8). However in this model vegetation changes are much less pronounced. Differences between the 2011 vegetation map and the 2100 vegetation map are noticeable, but not as drastic compared to the difference between the 2011 and 2100 vegetation maps for the FA 2 A1FI model (Fig.3.4.5).

There is an increase of both mangrove and mixed zone and a decrease in saltmarsh and *Casuarina* (Fig. 3.4.7). Indeed the lines through time for each vegetation type appear to be flattening to a new equilibrium from 2080 onwards.

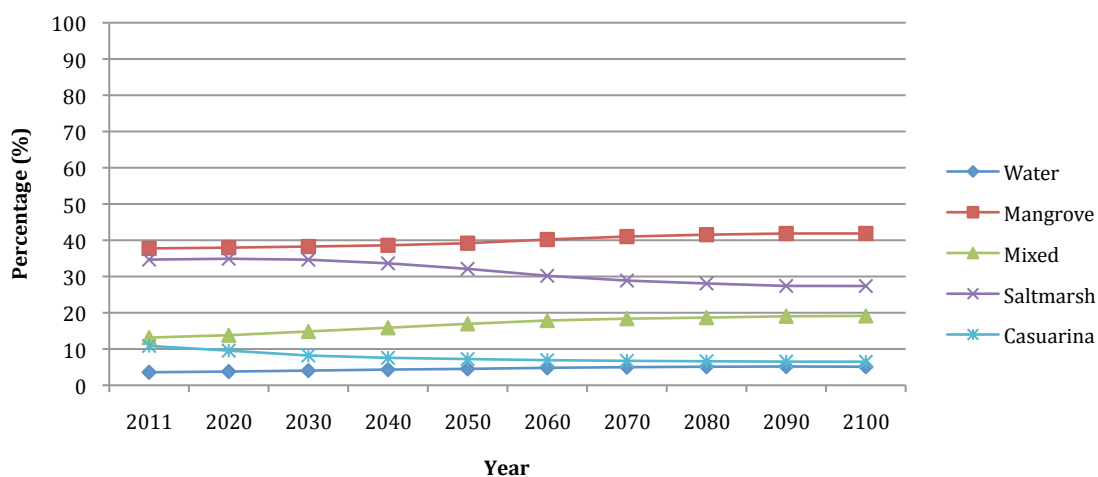


Figure 3.4.7 Percentage area of each vegetation zone at the study site at 10-year increments for Spatial model: FA 2 using the B1 5% CI sea-level scenario

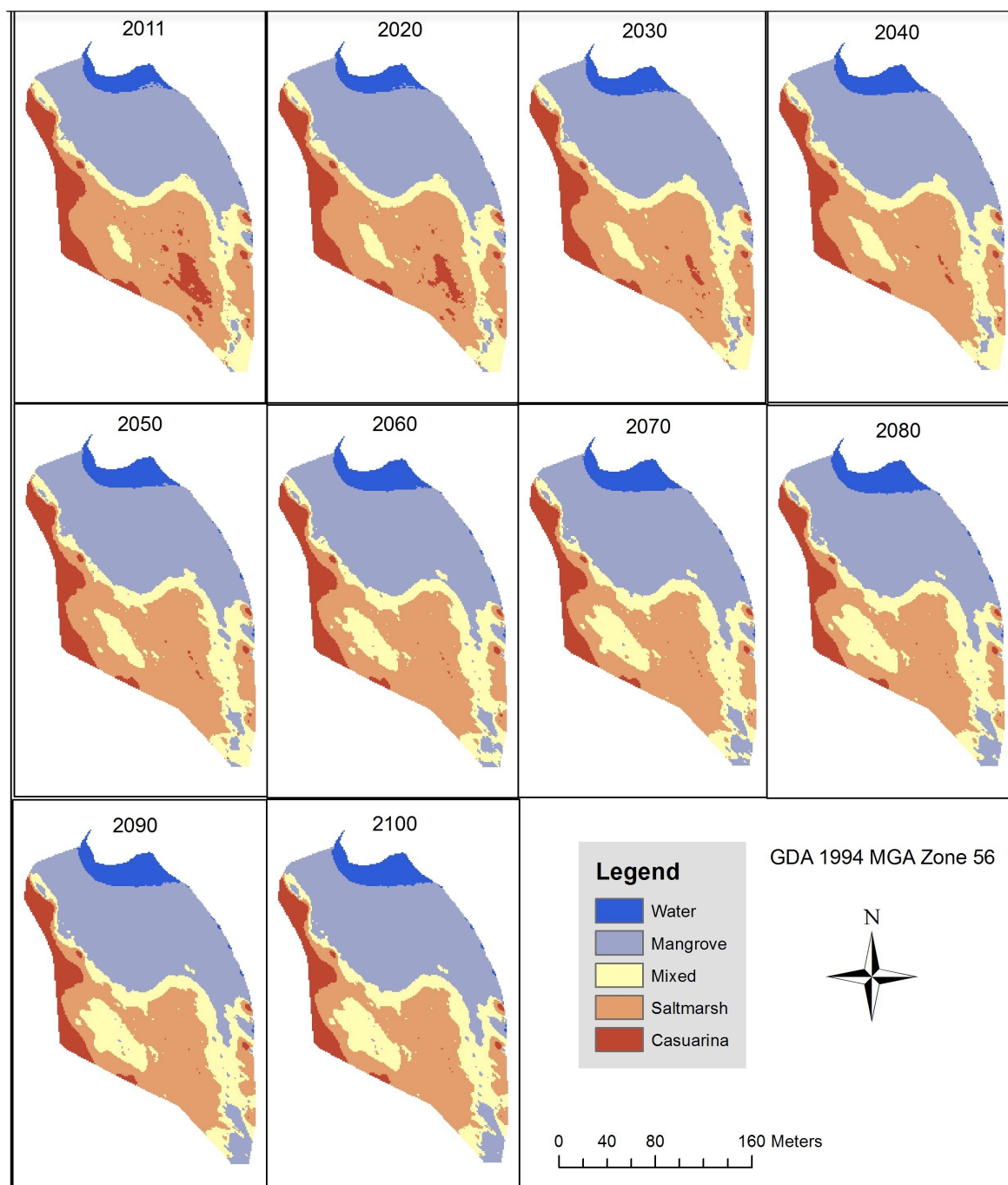


Figure 3.4.8 Spatial model: FA 2 showing the distribution of vegetation zones through time from 2011 to 2100 using the B1 5% CI sea-level scenario.

CHAPTER 4: DISCUSSION

4.1 INTRODUCTION

This chapter discusses the context, significance and implications of the results presented above also addressing any drawbacks or shortcomings. It is presented in three sections: surface elevation and accretion dynamics, spatio-temporal coastal vegetation mapping and modeling wetland response to sea-level rise, following the same sequence adopted in the Methods (Chapter 2) and Results (Chapter 3) sections.

4.2 SURFACE ELEVATION AND ACCRETION DYNAMICS

4.2.1 Surface elevation trajectories

The ten-year surface elevation trajectories reported at this site for mangrove and saltmarsh are 0.62 mm yr^{-1} and 0.29 mm yr^{-1} respectively. Rogers et al. (2006) reported very similar values using the first 3 years of this data 0.61 mm yr^{-1} for the mangrove and 0.26 mm yr^{-1} for saltmarsh. Other areas close by at Jervis Bay have similar surface elevation trajectories: Cararma Inlet, mangrove -0.81 mm yr^{-1} , saltmarsh 3.25 mm yr^{-1} ; Currambene Creek, mangrove 0.29 mm yr^{-1} , saltmarsh 0.14 mm yr^{-1} (Rogers et al. 2006). To the north, mangroves in Homebush Bay mangroves had a surface elevation trajectory of 7.2 mm yr^{-1} (Rogers, Saintilan, & Cahoon 2005). However, these surface elevation trajectories are from data over a short time period, between 2001 to 2003. There is no more recent surface elevation data at those sites for further comparisons. Compared to surface elevation trajectories from Europe (e.g. $>4.3 \text{ mm yr}^{-1}$ (Temmerman et al. 2004)) and America (e.g. 3.6 mm yr^{-1} (Cahoon et al. 2011)), Australian sites show much lower trajectories. This could be a product of differences in ecological geomorphological setting such as lower suspended sediment concentrations. Vertical accretion trajectories in the saltmarsh at Minnamurra were 1.42 mm/yr . Minnamurra appears to have very similar trajectories compared to Cararma Inlet (1.27 mm yr^{-1}) and Currambene Creek (1.37 mm yr^{-1}) in Jervis Bay (Rogers et al. 2006).

The surface elevation trajectories presented above vary through time and according to vegetation zone. For the mangrove zone, a ten-year trend of 0.6 mm/yr and a trend since 2009 of 9.7 mm/yr are remarkably different, reinforcing the inter-annual variability of surface elevation patterns at this site. This variability is confirmed by the statistical results.

Furthermore, isolated values such as seen for the 1st of December 2010, highlight the need for long-term datasets that average results and minimize the influence of outlying values on elevation trajectories. It is therefore recommended that future SET measuring programs measure at more frequent intervals than yearly to account for such inter-annual variability. This study also demonstrates the advantage of long-term datasets for extracting surface elevation and vertical accretion trajectories.

As vertical accretion and surface elevation were not correlated at Minnamurra River, this study demonstrates and supports the growing weight of evidence that vertical accretion cannot be assumed to be a proxy of surface elevation change in Australian mangrove and saltmarsh environments (Cahoon et al. 1999; Lovelock et al. 2011; Rogers et al. 2006). The contribution of vertical accretion to surface elevation change appeared to be small, as vertical accretion did not correspond to similar surface elevation values (Fig 3.2.3). This is evidence of upper-level autocompaction at this site (Rogers et al. 2006). Thus the control of vertical accretion on surface elevation change appears to be minimal and other factors must be examined to explain these patterns.

Groundwater fluctuations can influence wetland surface elevation by changing sediment volume due to swelling when wetland sediments are water saturated (Cahoon et al. 2011). Rogers & Saintilan (2008) present compelling evidence of the relationship between surface elevation and groundwater, stating that “groundwater inputs... exert a direct influence on surface elevation in the mangrove forests of southeast Australia” and that surface elevation was observed to correlate with the Southern Oscillation Index (SOI). SOI is an indicator of El Niño and La Niña conditions. Sustained negative values (indicating El Niño conditions where NSW receives less rainfall) occurred between 2001 and 2007 (Fig. 4.2.1). Two weak positive periods were observed in 2007 and 2008 which affected only the north of Australia to any significant extent (BOM 2011a) (Fig.4.2.1). However the sustained positive values in 2010 (Fig.4.2.1) coincided with a strong La Niña phase that resulted in the wettest year in 50 years for NSW (BOM 2011b). This corresponds with the higher mangrove and saltmarsh surface elevation values observed in 2010 and 2011. Cahoon et al. (1999) argue strongly that compaction in the saltmarsh is controlled by seasonal changes in rainfall. From this basis it is concluded groundwater inputs may have had a significant control on surface elevation at Minnamurra.

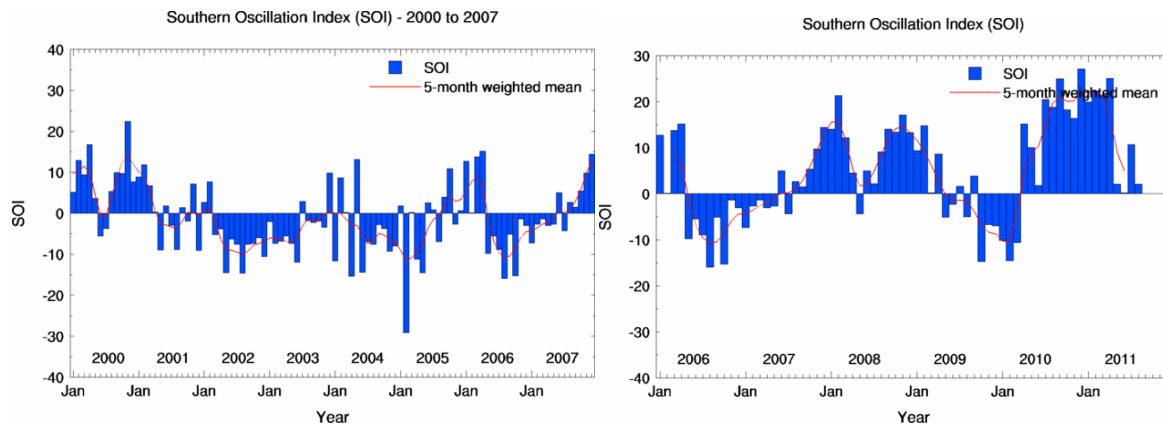


Figure 4.2.1 Patterns of SOI from 2000 to 2011 from the Australian Bureau of Meteorology. Strong positive values indicate a La Niña phase and strong negative values indicate an El Niño phase.

In view of the variability and fluctuation of surface elevation long-term measuring programs are essential to account for such perturbations and accurately measure longer-term surface elevation trajectories. Despite the similarity of measures of surface elevation taken over the 3 year study period reported by Rogers *et al.* (2006) and this study, the variability in surface elevation from this study demonstrate the need for long sampling periods.

4.2.2 Surface elevation and sea-level rise

Comparing sea-level rise at Port Kembla over the study period with observed surface elevation trajectories over the same period, there is considerable discrepancy. Sea level has risen at a rate of 1.8 mm yr^{-1} between 2001 and 2011. Surface elevation trajectories at Minnamurra were 0.61 mm yr^{-1} and 0.26 mm yr^{-1} for mangrove and saltmarsh respectively; considerably lower than sea-level rise. Average sea-level rise at Port Kembla since the installation of the tide gauge in July 1991 was 2.6 mm yr^{-1} , which is even higher than observed surface elevation trajectories. Compared to Fort Denison (Sydney) where sea level has risen at an average rate of 0.86 mm yr^{-1} for the past 85 years, Minnamurra surface elevation trajectories are still below sea-level rise.

Given this discrepancy between surface elevation trajectories and current local sea-level rise, global IPCC sea-level projections should cause considerable concern for the future of coastal wetlands at Minnamurra. Indeed the lowest sea-level rise scenario, B1 has rates of sea-level rise between 1.5 and 3.9 mm yr^{-1} (Tab. 4.2.1). This is comparable to current local rates of sea-level rise, and surface elevation trajectories at Minnamurra are currently much lower than these rates.

Table 4.2.1 IPCC Sea-level rise scenarios from the Fourth Assessment Report released in 2007. The top row is the 5% and 95% confidence interval (CI) for the sea level increase in meters and the bottom row is the 5% and 95% CI for the rate of sea level rise.

		B1		B2		A1B		A1T		A2		A1FI	
Sea level rise	m	0.18	0.38	0.20	0.43	0.21	0.48	0.20	0.45	0.23	0.51	0.26	0.59
	mm yr ⁻¹	1.5	3.9	2.1	5.6	2.1	6.0	1.7	4.7	3.0	8.5	3.0	9.7

The highest sea-level scenario, A1FI, has rates of sea-level rise between 3.0 mm yr⁻¹ and 9.7 mm yr⁻¹ (Tab 4.2.1). Considering current surface elevation trajectories such rates of sea-level rise warn of the severe consequences of rapid sea-level rise to coastal wetlands at Minnamurra.

4.3 SPATIO-TEMPORAL COASTAL VEGETATION MAPPING

The most obvious pattern observed from the mapping of vegetation zones at the study site from 2003 and 2011 is the conversion of mixed zones to mangrove zones. Mangrove now abuts saltmarsh in many areas of the site with no mixed zone ‘buffer’ in between. This trend of mangrove expansion at this site has been documented by Chafer (1998) between 1938 and 1997 (Fig. 4.3.1). The results from this study show a continuation of this expansion trend. It is speculated from the evidence of consolidation of mangroves between 2003 and 2011, that colonisation followed by consolidation is the mechanism by which mangroves have encroached landwards at this site. Unfortunately spatial resolution constraints prevented Chafer (1998) from mapping a mixed zone. There is evidence of mangrove colonisation in the 1993 and 1997 Chafer (1998) maps in the middle of the site (Fig.4.3.1). Our classification for 2003 shows this same area as mixed zone. This is a product of different classification systems, but further examination of the original aerial photographs reveals that this new ‘island’ of mangrove is in fact colonising the saltmarsh zone. In the 2011 map (Fig. 3.3.1) it is classified as mangrove rather than mixed supporting the colonisation consolidation mechanism.

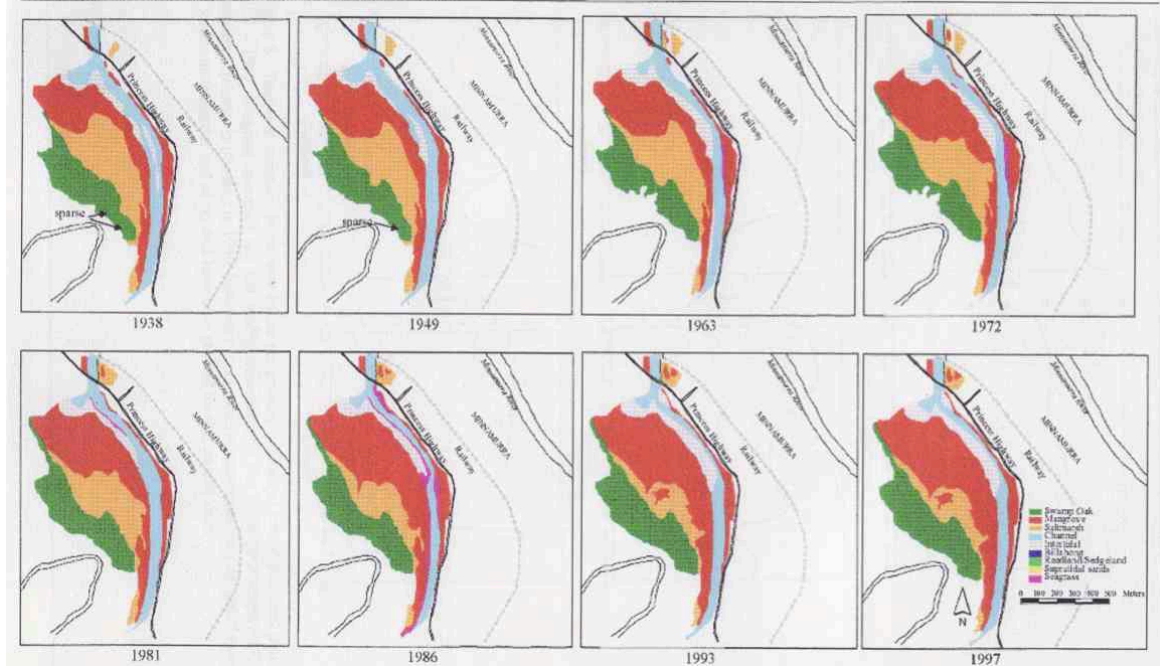


Figure 4.3.1 Distribution of vegetation zones from 1938 to 1997 by Chafer (1998). Mangrove is red, saltmarsh is yellow and casuarina is green.

Other authors have observed similar trends of mangrove expansion in southeast Australia (Saintilan 2009). (Saintilan & Williams 1999) present compelling evidence for mangrove encroachment along the NSW coast and predict a similar pattern to continue into the future considering SLR predictions. Rogers et al. (2006) found mangrove expansion at all their 10 sites along the southeast coast of NSW. Rogers et al. (2006) reported a mangrove expansion of $1.17 \% \text{ yr}^{-1}$ at Minnamurra. This study observed a rate of mangrove expansion between 2003 and 2011 of $1.8 \% \text{ yr}^{-1}$ which is slightly higher.

Saltmarsh decline at Minnamurra was $-0.86 \%/\text{yr}$ (Rogers et al. 2006). Interestingly the results of this study show an increase of saltmarsh area at a rate of $0.4 \% \text{ yr}^{-1}$. However, there was a significant decline in the area of mixed zone ($-7.1 \% \text{ yr}^{-1}$) and thus the net result has been a loss of saltmarsh at this site between 2003 and 2011. These patterns suggest that mangrove encroachment and saltmarsh decline is a well established trend at this site and others along the southeast coast of NSW.

The cause of this broader trend is still debatable. However, at this site, according to these findings, it is proposed the cause is a product of sea-level rise outpacing surface elevation trajectories and resulting in upslope migration of mangroves due to increasing frequency of tidal inundation. This suggestion is supported by the conclusions of Rogers et al. (2006)

who also noted that sites where mangrove encroachment was high were those where vertical accretion was not equivalent to surface elevation change (Rogers et al. 2006). Minnamurra River shows this pattern in the saltmarsh where vertical accretion is not transferred into surface elevation gains (Fig. 3.2.3).

As groundwater has been proposed as an important control on surface elevation at this site by impacting sediment volume, it follows that El Niño phases during which rainfall is generally lower may promote mangrove encroachment as sea-level rise continues. Rogers et al. (2006) attributed higher amounts of autocompaction to periods of El Niño and reduced rainfall, which in turn amplified mangrove encroachment. In contrast, results from Moreton Bay, Queensland suggest that higher rather than lower amounts of rainfall promote greater rates of mangrove encroachment (Eslami-Andargoli et al. 2009). However, the mechanisms of mangrove establishment may be different in Moreton Bay, as research there indicates that mangroves are responding to lower salinity levels in the saltmarsh zone (Eslami-Andargoli et al. 2009). The conclusion of Rogers et al. (2006) seems most appropriate considering the obvious strong association between SOI and surface elevation.

It is suggested at this site that mangrove encroachment is related to autocompaction and sea-level rise, with autocompaction being strongly controlled by El Niño phase. Periods of El Niño which promote higher autocompaction lead to upslope migration of mangrove as sea-level increases and tidal flooding extends further inland.

Once mangroves have established in an area the surface elevation trajectory is dramatically changed. Evidence for this is seen when the results of the vegetation mapping and an examination of the surface elevation trajectories of the individual SETs are combined. Considering the position of the SETs at the site (Fig. 2.2.2), the patterns of vegetation change (Fig.3.3.1), and the patterns of SE change for individual SETs (Fig. 3.2.6 & 3.2.7), an interesting result emerges. In section 3.2 it was demonstrated that in the saltmarsh zone, SET 1 had a very different surface elevation trajectory to SET 2 and 3. While SET 1 & 2 remained almost flat, SET 1 showed a clear positive trend. This was not the case in 2003 where the value for SET 1 falls in between SET 2 and 3.

If we consider where SET 1 is located in relation to the mapped vegetation zones (Fig. 4.3.2), in 2003 SET 1 is on the edge of the mixed and saltmarsh zone. However in 2011 the mangrove zone has encroached landwards and SET 1 is now on the edge of the

saltmarsh and mangrove. In fact, having visited the site many times to conduct measurements the author observed a mangrove tree growing in the plot of SET 1. This is compelling evidence that elevation trajectory is strongly associated with vegetation type. Such a change in surface elevation trajectory may be due to many factors such as the influence of aerial mangrove roots trapping sediment (Krauss 2003; Kumara et al. 2010) or subsurface growth of mangrove roots (Nyman et al. 2006).

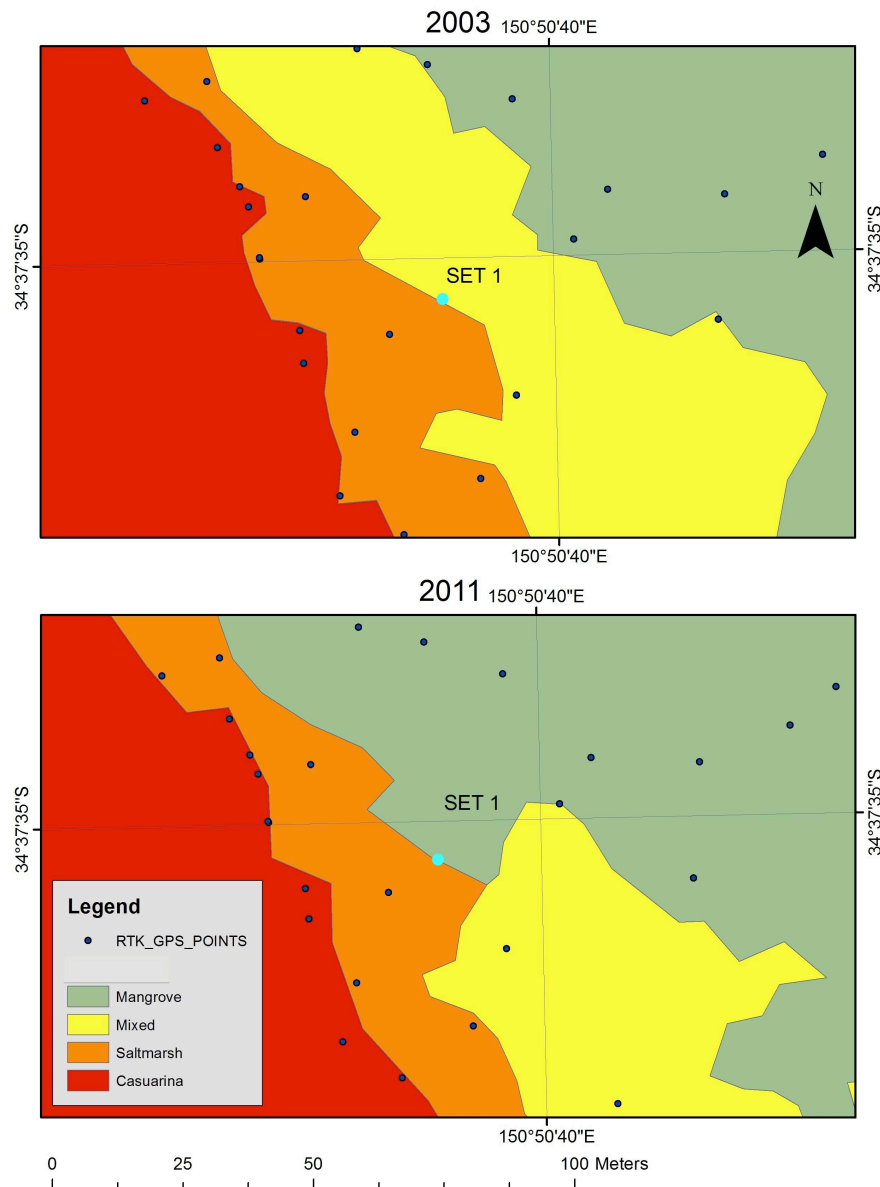


Figure 4.3.2: Comparison of the position of SET 1 (highlighted light blue) in relation to the mapped vegetation patterns from 2003 and 2011. Other points are from the RTK GPS survey from April 2011.

4.4 MODELING WETLAND RESPONSE TO SEA-LEVEL RISE

4.4.1 *Comparing the models*

To understand and interpret the results of the modeling presented in Chapter 3 it is essential to compare and contrast their different inputs and variables and examine how they affect the final output. General differences between the models are presented in Table 2.5.2 and their effects are discussed below.

The simple vegetation model applied a block increase of surface elevation change according to vegetation type, which was defined on the basis of elevation classes. This model is different from the other two models (based on factorial analysis) in which vertical accretion and surface elevation are calculated based on individual elevation values for each cell rather than elevation classes. Thus the factorial analysis models (FA 1 and FA 2) calculate elevation changes at a greater spatial resolution compared to the simple vegetation model.

However, calculating surface elevation on the basis of vegetation zone is also an advantage of the simple vegetation model over the FA models. Even though vegetation class can be defined by absolute elevation, the difference in the surface elevation trajectory between mangrove and saltmarsh is greater than can be accounted for by the elevation difference. This is demonstrated most clearly in Figures 3.2.7 and 3.3.3 where a vastly different elevation trajectory is observed after mangroves invade a saltmarsh SET plot, more than can be accounted for by this point's difference in absolute elevation before and after the mangrove invasion. The FA models do not account for elevation and accretion differences due to vegetation type.

The other main difference between the simple vegetation model and the FA models is how sea-level rise is applied in the model. For the simple vegetation model, sea-level rise is added to the vegetation classification values (heights of elevation within the DEM) each year and thus the block surface elevation gain is actually applied to a new group of cells each year. In contrast to this the FA models incorporate sea-level rise into the equation as it is a factor in the calculation the new elevation for each cell in the DEM. Just as elevation change is generalized for the simple vegetation model as explained above, so also the effect sea-level rise is generalized in the model. These generalizations must be kept in mind when interpreting the results of the simple vegetation model.

Interesting comparisons may also be made between FA 1 and FA 2. Firstly comparing DEM surfaces, for FA 1 a TIN of the RTK GPS points was used. The strength of this surface is that the RTK GPS points have the highest accuracy of all the data available. The disadvantage is that the points are not evenly spaced throughout the study site and there are areas of the site where there are few elevation points. FA 2 used an IDW of the RTK GPS points and selected LiDAR points. However while this reduced the error introduced by the wide spacing of the RTK GPS points, it introduced new error, because RMS error between the elevations of the selected LiDAR points and the RTK GPS points was 9.3cm.

Applying calculations of millimeters of vertical accretion and surface elevation change on such a surface thus seems inappropriate. However it must be remembered that even the RTK GPS points have a vertical accuracy of ± 1.7 cm. The differences between these two surfaces and their effects on each of the FA models can be summarized by saying that they will produce different patterns of elevation change as the model is run (corresponding to how elevation has been calculated at a given point on the surface) and that the creation of an accurate DEM at this site was a trade-off between point spacing and RMS error.

The second difference between FA 1 and FA 2 is the application of the model over one year or ten year increments. The finer the time scale at which accretion and elevation is applied the greater the accuracy of the calculation. However, the more times the model is run in ArcGIS the greater the compounding of spatial error within the model. There is thus a trade-off between statistical accuracy and spatial accuracy within the model.

Another difference between FA 1 and FA 2 is the effect of changing distance to shore according to sea-level rise. In FA 1 when distance to shore remains constant, the assumption is that although sea-level has risen by almost 80cm by 2100 there has been no change in the position of the shoreline. In comparison, in FA 2 distance to shore is recalculated with each run of the model. Thus surface elevation responds to sea-level rise spatially as well as statistically in the model.

The effect of applying two different sea-level rise scenarios to FA 2 produced the most drastic differences between all the models. When the A1FI 95% scenario is applied saltmarsh and casuarina are rapidly removed from the site, whereas when the B1 5% scenario is applied the resulting vegetation changes are much less obvious. It is highly significant that the effect of applying a different sea-level rise scenario produces such a

significant difference between the outputs, indeed a difference of a similar magnitude to the difference between the sea-level rise scenarios themselves. Such a result demonstrates the importance of accurately projecting sea-level rise, particularly its magnitude and rate, in determining future outcomes for mangrove and saltmarsh ecosystems.

4.4.2 Context of these models

Small-scale coastal wetland modeling is a developing field. So far there is no peer reviewed publications of such work, although work has been published in State Government reports (Rogers & Saintilan 2011). Models such as SLAMM, which operate at larger scales, were not considered for this project because of the emphasis on wave action-erosion as well as the effect of freshwater inputs to the marsh system (Craft et al. 2009). These factors may be important for marshes in the northern hemisphere but are inappropriate for Australian geomorphological settings where fetches are often small within our confined barrier estuaries (K. Rogers, personal communication). Much of the focus of these models is also on suspended sediment concentrations (French 2006; Temmerman 2003; Temmerman et al. 2004). Including this factor in surface elevation modeling would be essential for larger catchments in Australia with higher discharges and more extensive coastal marsh than the Minnamurra River.

The results of modeling in European countries such as Belgium on the Scheldt Estuary show that tidal marshes will be able to keep pace with rising water level (modeled as the current rate and 1.5 times the current rate which was $>3\text{mm/yr}$ depending on estuary position) provided that suspended sediment concentrations (SSC) did not decrease (Temmerman et al. 2004). Similarly at North Inlet, South Carolina, high sediment loads allow marshes to tolerate sea-level rise 3.5 times current long-term rates for the region (currently 3.4mm/yr) (Morris et al. 2002). Although these studies have not used IPCC sea-level projections it appears that the capacity of these systems to cope with sea-level rise is much greater than Australian estuaries such as the Minnamurra River, where sediment supply to the intertidal zone is minimal and continues to dwindle as the estuary matures (Panayotou et al. 2007; Roy et al. 2001).

4.4.3 Interpreting the results

While the three models show very different changes for each of the vegetation zones there are some patterns common to all the models which emerge: (1) the area of saltmarsh and *Casuarina* at this site is significantly reduced in all three models, (2) mangroves initially maintain or expand in area but towards the end of the model begin to decline, (3) for FA 1 and FA 2 water dominates the study site by 2100. The overall conclusion to be drawn therefore is that under IPCC sea level projections, the current relationship between wetland surface elevation change and sea-level rise, is unable to ensure these communities maintain their current distributions and proportions.

Also of significance to this conclusion is that when the B1 5% sea-level rise scenario was applied to FA 2, a loss of saltmarsh and casuarina was still observed. The observed sea-level rise for Port Kembla since the installation of the gauge (2.6 mm yr^{-1}), is already above the highest rate for the B1 scenario (2.2 mm yr^{-1}), so the fact that loss of saltmarsh and casuarina still occurred warns that we should expect future changes to be at least as significant, but likely more, as the FA 2 B1 5% model. Indeed, given the evidence from the mapping by Chafer (1998) and its continuation in this project, the changes in vegetation distribution already observed are far more drastic than those shown by the FA 2 B1 5% model.

Examining the implications of results of the FA 1 and FA 2 model using A1FI 95% sea-level rise scenario, the vegetation distribution changes to this site are extremely drastic. The rate of sea-level rise for the last decade of the model under this scenario is 11.8 mm/yr . While this may seem extreme, other authors present model projections which reach even higher (Church et al. 2008; Nicholls & Cazenave 2010). Therefore, the results of these models, although they cannot be assumed to be an exact predictor of future changes, must be a firm baseline and an indicator of the likely impacts of sea-level rise for coastal wetlands.

4.4.4 Implications for adaptation and management policy

This study demonstrates that there is an increasing need for firm scientifically-based adaptation and management policy for the Minnamurra River and other estuaries along the NSW coast. The results and conclusions presented above reinforce the need for greater measures to be taken to ensure that coastal wetlands are preserved into the future including allowing these communities to self-adjust, by migration and accretion, to external changes such as sea-level rise. There are three policy points raised by this study that need to be addressed promptly: the regulation of groundwater use, the introduction of buffer zones and allowing greater volumes of sediment to reach the lower parts of estuaries.

Groundwater appears to be correlated with surface elevation change at this site on the Minnamurra River. By mitigating the severity of groundwater depletion in El Niño periods, the opportunity for mangroves to encroach landwards may be prevented and surface elevation trajectories may more closely match sea-level rise. Adaptation policy must therefore focus on integrating decisions regarding groundwater usage with estuary management in order to protect, in particular, saltmarsh ecosystems.

Buffer zones will be immensely important at this site for maintaining intertidal vegetation such as saltmarsh into the future. Assessing the possibility of providing or extending fringe buffer zones was listed as a medium priority in the Minnamurra Estuary Management Plan published in 2003. This study puts weight behind this management strategy by demonstrating from modeling that the encroachment of mangrove into saltmarsh communities in the next 20-40 years will be significant unless buffer zones allow landward migration of saltmarsh. Evidence from the modeling above shows that under the most severe IPCC sea-level rise scenario changes to the Minnamurra River and its intertidal wetland zones will be drastic with extensive losses of wetland habitat. Policy must therefore also focus on community consultation and education.

In northern hemisphere settings, adequate suspended sediment concentrations (SSC) have been identified as vital to allowing coastal wetlands to adapt to future sea-level rise (Temmerman et al. 2004). Australian estuaries generally have considerably lower SSC due to smaller catchment sizes and difference in estuary geomorphology (Roy et al. 2001). It has been argued that SSC should be minimised in southeast Australian estuaries to promote the maintenance of seagrass beds, which are important habitat for many popular

adult and juvenile fish species. However, higher SSC would advantage saltmarsh and mangrove ecosystems in adapting to sea-level rise and the importance of mangrove and saltmarsh in maintaining healthy estuary ecology has often been undervalued. Therefore it is recommended that SSC be increased in lower estuaries by restoring natural flows in upper catchments, so that sediment is available for intertidal wetlands in responding to sea-level rise. However, this must not be at the expense of losing large areas of sea grass and therefore coastal management must focus on identifying minimum thresholds of SSC for wetlands and maximum thresholds of SSC for seagrass.

CHAPTER 5: CONCLUSIONS

This project has examined the response and assessed the vulnerability of mangroves and saltmarsh at the Minnamurra River, to sea-level rise and suggests some directions in which to develop climate change adaptation policy and management for the future. Specifically the examination has lead to the following conclusions.

Vertical accretion cannot be assumed to be a proxy for surface elevation change and therefore future research must take into account the other factors such as groundwater when extracting surface elevation trajectories. Surface elevation trajectories on the Minnamurra River and on the southeast Australian coast are much lower than those reported for Northern Hemisphere studies and even results from northern parts of Australia. Therefore wetland vulnerability must be assessed locally and integrate local ecological and geomorphological factors.

SET measuring programs are recommended to be long term and to involve frequent measuring to capture inter-annual variability and short-term perturbations in surface elevation. Surface elevation trends at this site appear to be strongly controlled by groundwater and thus connected to patterns of El Niño and La Niña reflected in the Southern Oscillation Index. Surface elevation trajectories at Minnamurra are considerably lower than observed sea-level trends over the study period, over the past 85 years according to Fort Denison, Sydney, and compared to all future IPCC projections. Therefore coastal wetlands at Minnamurra are highly vulnerable to future sea-level rise.

Mangroves have continued to encroach landwards by colonisation followed by consolidation into the saltmarsh zone between 2003 and 2011, continuing the pattern observed by Chafer (1998). This has resulted in loss of saltmarsh habitat which is detrimental to the ecological and geomorphological health of the Minnamurra River. Considering observed surface elevation trajectories and sea-level trends the cause of this encroachment is likely to be greater inundation frequency of the saltmarsh zone as sea-level rise has outpaced surface elevation change. Saltmarsh are particularly sensitive to greater inundation frequency, thus future sea-level rise will be more detrimental to saltmarsh than mangrove. El Niño periods where rainfall and hence groundwater level are lower promote mangrove encroachment due to sediment shrinking and upper level autocompaction of wetland sediments.

The initial loss of saltmarsh and the expansion of mangrove in the modeling demonstrates that future sea-level rise will particularly impact saltmarsh communities in the next 40 years. After 2050 both mangrove and saltmarsh are projected to rapidly decline. Even in the FA 2 model when the IPCC B1 5% sea-level scenario is used there is still a loss of saltmarsh by 2100. The highest rate of sea-level rise used in the B1 scenario is less than the current rate of sea-level rise for Port Kembla since 1991. Thus future changes will be at least as significant as shown by the FA 2 B1 model.

There have been several warnings that rates of sea-level rise may exceed those projected by the IPCC and thus the changes observed for FA 1 and FA 2 using the IPCC A1FI 95% scenario, while seeming very drastic, are plausible and indicate the range of impacts of rapid sea-level rise for intertidal wetlands in NSW. Adaptation policy and estuary management policy should focus on groundwater regulation and monitoring in the catchment, the introduction of buffer zones and the regulation of suspended sediment concentrations.

Intertidal wetlands at this site on the Minnamurra River, NSW Australia are extremely vulnerable to the threat of future sea-level rise. Considering observed local sea-level rise, current surface elevation trajectories have not been able to prevent significant geomorphological and ecological changes at this site, evidenced by the spatio-temporal mapping. The consequences of projected sea-level rise for the Minnamurra River, reflected in the modeling, indicate significant future changes to mangrove and saltmarsh distribution. Climate change policy and management will need to address the causes of sea-level rise and minimise its effects if the future loss of mangrove and saltmarsh ecosystems is to be prevented.

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