Mangrove and Saltmarsh Threat Analysis in the Sydney Coastal Councils Region

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EXECUTIVE SUMMARY

The Sydney Coastal Councils Group, as part of its Sydney's Salty Communities Program, engaged the University of Wollongong, in partnership with Macquarie University, to undertake a Mangrove and Saltmarsh threat analysis. The aim of the project was to provide knowledge of the potential impacts of sea-level rise on mangrove and saltmarsh, the response of mangrove and saltmarsh to sea-level rise, and policy and management actions that can be implemented to improve ecosystem outcomes for mangrove and saltmarsh in the Sydney Region. Key elements of this project include:

- Review of the pressures and impacts on mangrove and saltmarsh in the Sydney Region, indicating the consequences of current management activities on the provision of ecosystem services;
- Preparation of a first pass assessment of the vulnerability of mangrove and saltmarsh to inundation and erosion related to sea-level rise and other climate related processes;
- Higher resolution assessment of the vulnerability of mangrove and saltmarsh to sea-level rise using spatial modelling approaches integrated with available datasets; and
- Overview of adaptation barriers and opportunities, including strategic planning activities that can be implemented to improve the ecosystem services provided by mangrove and saltmarsh in the Sydney Region.

Mangrove and saltmarsh wetlands occur within the intertidal zone of low energy shorelines. They provide a range of ecosystem services including fisheries and biodiversity habitat, shoreline protection, carbon sequestration and visual amenity. These ecosystems are subject to numerous climatic and non-climatic pressures and impacts. Non-climatic pressures include human activities that alter the hydrology, erosion and sedimentation dynamics and cause physical or biochemical disturbance to mangrove and saltmarsh. Climatic factors and sea-level influence the distribution and character of these ecosystems. These factors may favour increases in mangrove extent relative to saltmarsh. Analysis of historic records, including aerial photographs, has shown substantial change in the extent of mangrove and saltmarsh in the Sydney region, with declines related to land use change, but also shifts towards a dominance of mangrove area relative to saltmarsh. While there is evidence that mangrove and saltmarsh have some capacity to increase elevations over time, sea-level rise may alter the position of mangrove and saltmarsh within the tidal frame; it remains unknown as to whether the degree of elevation change within mangrove and saltmarsh will off-set changes in tidal position following sea-level rise.

First-pass assessment of vulnerability of mangrove and saltmarsh

A first-pass assessment was undertaken for the local government areas of the Sydney Coastal Councils Group to: 1) identify exposure, sensitivity and adaptive capacity of mangrove and saltmarsh to inundation and erosion associated with climate change; and 2) to integrate these maps to provide an indication of the overall vulnerability of mangrove and saltmarsh to climate change. This assessment incorporated a range of data sources and demonstrated the potential for increases in estuarine vegetation extent throughout the Sydney region, assuming that all upslope areas could be managed to allow for ecosystem migration.

Importantly, the first-pass approach does not account for possible changes in the distribution of vegetation communities over time within the Sydney Coastal Councils area, and only provides an indication of vulnerability based on current vegetation distribution within the tidal frame. High resolution modelling, focussed on areas where the vulnerability assessment indicated higher vulnerability, would provide a more robust indication of the vulnerability of coastal wetlands to sea-level rise.

Projecting sea-level rise threats to mangrove and saltmarsh: Wolli Creek pilot study

High resolution assessment of mangrove and saltmarsh vulnerability to sea level rise was derived using the Sea Level Affecting Marshes Model (SLAMM) for the Cooks River catchment. The model projected mangrove increases between approximately 280-560% under the various sea-level rise scenarios when wetlands were able to maintain their position in the tidal frame (i.e. when built-up areas, classified as urban areas having land use consistent with residential, commercial and industrial purposes, were able to convert to other vegetation classes). Critically, however, significant declines in the extent of mangrove (and to a lesser extent saltmarsh) were projected when built-up areas were not able to convert to other land classes. This effect was most pronounced under the high sea-level rise scenario, further highlighting the vulnerability of coastal wetlands to sea-level rise when there is limited or no opportunity for vegetation migration. SLAMM also demonstrated the restricted nature of saltmarsh throughout the study region and its reduced capacity to adjust to sea-level rise due to its distribution within a narrow tidal range positioned between the highly adaptive vegetation communities dominated by mangrove and *Casuarina*.

Recommendations for sea-level rise adaptation and strategic planning

Findings of the review of pressures (Chapter 2) and modelling exercises (Chapters 3 & 4) identified two specific adaptation needs for mangrove and saltmarsh of the Sydney region. These are 1) the importance of management strategies which accommodate vegetation migration; and 2) the importance of management strategies which specifically preserve and accommodate saltmarsh. Based upon these needs, four types of adaptation option have been recommended:

1. Eliminate or reduce non-climate stressors to enhance resilience of existing ecosystems
It has been demonstrated healthy wetlands have greater resilience to the impacts of sea-level rise (through their elevation-building capacity) than degraded ecosystems. Therefore, management actions which improve wetland health through the elimination or reduction of non-climate stressors is an important first adaptation option.

2. Protection through engineering options (including modification of existing structures)
There are multiple locations throughout the Sydney region where existing engineering structures influence the extent and adaptation potential of mangrove and saltmarsh. Modification of existing structures, or creation of new structures might be used to improve ecosystem health and resilience, or to introduce tidal exchange to potential wetland migration locations. Ecologically-sensitive structures, including living shorelines and use of natural materials such as oyster shells and seagrass wrack can be incorporated to enhance habitat and ecosystem service provision.

3. Elevation maintenance through sediment nourishment;
Sediment nourishment within saltmarsh and Casuarina may provide the most viable and cost effective means of maintaining the extent and position within the tidal frame for these vegetation communities. This approach would benefit from trial studies incorporating detailed investigation of elevation and sediment dynamics as well as potential ecological impacts of sediment nourishment.

4. Planning for living shoreline establishment and migration
Our modelling (SLAMM results) suggests that utilisation of existing open spaces and development areas may provide the most feasible opportunities for supporting future wetland establishment and migration. In suitable locations the establishment of new tidal wetlands or rehabilitating existing wetlands will act to attenuates wave action, minimises erosion and offer shoreline resilience to sea-level rise. Living shorelines will also provide a range of co-benefits for ecosystems, but require consideration of land use and planning constraints.

Priorities identified in the current planning documents by the Greater Sydney Commission for the Sydney region highlight a planning emphasis upon population density increases in areas near Sydney's estuarine waterways, and the improvement of waterways and foreshore areas including importance of public access and amenity. These priorities present both opportunities and challenges for the implementation of mangrove and saltmarsh adaptation options. Creative options, such as the creation of living shorelines along the planned 'Green Grid' of the Cooks River and Wolli Creek Open Space Corridors, incorporating wetland adaptation in the upcoming repurposing of industrial foreshores and utilisation of market based approaches can be used to maximise the extent and adaptation of mangrove and saltmarsh to sea-level rise.

Improvements in data coverage, modelling products and the incorporation of social and planning constraints will be useful in informing mangrove and saltmarsh adaptation planning across the broader Sydney region. To this end, there is a specific need for validation of DEMs, tidal plane modelling and SLAMM modelling; as well as expansion of the SET-MH network in the Sydney region. Other research needs include the study of impacts of mangrove removal and cutting, and experimental trials of sediment nourishment. The establishment of a targeted monitoring and evaluation program that monitors the effectiveness of adaptation options and feeds into adaptive management is central to the broader implementation of the recommended adaptation strategies.

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Chapter One: Introduction

Climate change and its effects on sea level, temperature, ocean currents, storms, rainfall and run-off, and hydrodynamics along coastal margins is of particular concern for managers tasked with protecting both natural and built assets. In Australia where more than 85% of the population lives within 50 km of the coast (Trewin, 2004), climate change is and will continue to be a key threatening process. This risk was quantified in The First Pass National Coastal Risk Assessment (DCC, 2009), with up to $63 billion of existing residential buildings estimated to potentially be at risk from a 1.1 m sea-level rise. The effect of climate change, particularly on the built environment, is likely to be more acute near population centres, such as cities. With the exception of Canberra, all of Australia's capital cities are coastal cities, but none is larger than Sydney – with a population approaching 5 million people (ABS).

As cities are dominated by buildings and associated infrastructure, research attention has largely focussed on the effects of climate change on built assets. However, natural assets also have significant environmental, social and economic value. The most recent estimate of the value of global ecosystem services indicated that tidal marsh and mangrove provide an economic value of $193 843 USD per hectare per year for functions such as climate regulation, nutrient cycling, erosion and sediment retention, and refuge for commercially important and non-commercially important fauna (Costanza *et al.*, 2014). In addition, urban ecosystem services will differ from those occurring in regional areas due in large part to greater use by a larger population, and their contribution to public health and quality of life for urban populations (Bolund & Hunhammar, 1999). Spatial planning has a significant influence on the delivery of ecosystem services in urban landscapes, and planning for the effects of climate change on natural assets will be essential for the maintenance of urban ecosystem services.

Sydney – Australia's most densely populated city – provides a unique case study for exploring the effects of climate change on urban ecosystems and the services they provide. With a large population clustered along the coastline and estuarine shorelines that spill westward, urban ecosystems are restricted in their distribution. This is particularly the case for coastal ecosystems, such as mangrove and saltmarsh, whose natural distribution is already restricted between mean sea level at the seaward margin and highest astronomical tide at the landward margin. Like other urban areas, the distribution of mangrove and saltmarsh is contained within the margins of urbanisation, with significant losses associated with land-cover conversion from wetland to other land uses (such as land fill, parks and industrial), and developments at wetland margins that restrict landward expansion of wetlands. However, unlike other Australian cities, the distribution of mangrove and saltmarsh is further constrained by factors inherited from the geology of the Sydney Basin. In this regard, sediment delivery during the Quaternary period to the deeply incised drowned river valleys of the Sydney Basin was not high enough to result in the development of expansive depositional environments suitable for the establishment of large mangrove forests and saltmarsh plains. Consequently, mangrove and saltmarsh development within Sydney Harbour, Port Hacking and the Hawkesbury River is restricted to narrow margins along these estuaries or within small fluvial delta regions where bedrock depth is shallow and hydrodynamic energy is low enough to facilitate sediment deposition and vegetation establishment. The ocean embayment of Botany Bay, which is dominated by marine processes and has relatively little freshwater inflow (Roy *et al.*, 2001) supports the most expansive areas of mangrove and saltmarsh in the Sydney region at Towra Point (Mitchell & Adam, 1989); here coastal wetland development is facilitated by the development of large areas of Aeolian and alluvial sands.

In recognition of the value of urban ecosystems and the high relative carbon-sequestration and storage provided by so-called Blue Carbon ecosystems, as well as the unique geomorphic setting in which mangrove and saltmarsh have evolved within the Sydney Region, the Sydney Coastal Councils Group, through the federally funded Sydney's Salty Communities Program, are supporting a mangrove and saltmarsh threat analysis. This report is structured as follows:

Chapter Two provides an overview of the pressures, and impacts on mangroves of the Sydney Region and potential implications for ecosystem services. We have argued for the significance of potential climate change impacts in driving further changes in the extent of mangrove and saltmarsh, in ways profoundly impacting on the ecosystem services provided. These wetlands are the remnant of a previously extensive intertidal wetland resource. Direct and diffuse alterations have in recent decades favoured mangrove growth at the expense of saltmarsh, prompting targeted saltmarsh restoration projects in several locations.

Chapter Three provides a first pass assessment of the vulnerability of mangrove and saltmarsh to inundation and erosion related to sea-level rise and other climate related processes. Indicator maps of the exposure, sensitivity and adaptive capacity of mangrove and saltmarsh to inundation and erosion associated with climate change are presented, with the findings integrated to provide an assessment of the overall vulnerability of mangrove and saltmarsh to climate change. The study focusses on mangrove and saltmarsh within the local government areas of the councils that form the Sydney Coastal Councils Group regional organisation of councils.

Chapter Four provides focussed, high-resolution modelling of the vulnerability of coastal wetland vegetation within the Cooks River under a variety of sea level rise scenarios (IPCC high-RCP8.5; intermediate-RCP4.5; and low-RCP-2.6). Detailed consideration is given to changes within Wolli Creek. This chapter presents the spatial and statistical results of SLAMM (Sea-Level Affecting Marshes Model), a complex, non-hydrodynamic model. The modelling was informed by vertical accretion measures from the nearest Surface Elevation Tables (those at Homebush Bay) and tidal levels recorded along the Cooks River. We model the impact of coastal squeeze by comparing the projected extent of tidal flat, mangrove, saltmarsh and *Casuarina* while (i) allowing for encroachment of vegetation on built areas and (ii) prohibiting encroachment of vegetation into built areas (but allowing colonisation of parkland and reserves).

Chapter Five provides an overview of sea-level rise adaptation approaches for coastal shorelines, including mangrove and saltmarsh. It then incorporates the findings of Chapters 2-4 to identify the specific adaptation needs of mangrove and saltmarsh in the Sydney region. The principles and practices related to recommended adaptation options are discussed as well as suggestion of suitable locations where such options might be implemented. Opportunities for strategic planning and land use change are presented, knowledge gaps identified and recommendations future investigation are discussed.

Chapter Two: Review of pressures and impacts on mangrove and saltmarsh in the Sydney Region

2.1 Introduction

The chapter provides an overview of the ecosystem services provided by mangrove and saltmarsh in the Sydney region and a discussion of the changes in extent observed since European colonisation.

2.1.1 ECOSYSTEM SERVICES PROVIDED BY MANGROVE AND SALTMARSH

The ecosystem services provided by estuarine wetlands in the Sydney region have been clarified through studies conducted within the region over the past few decades. These services may be broadly summarised as fisheries provision; biodiversity habitat provision; carbon sequestration; shoreline protection; and visual amenity, as summarised briefly below:

Fisheries provision: Research at Towra Point and Homebush Bay has demonstrated the importance of intertidal mangrove and saltmarsh as a habitat for estuarine fish, including species of commercial significance (Mazumder *et al.*, 2006). The release of crab larvae from these wetlands into the spring tide provides an important source of nutrition for zooplanktivorous fishes (principally the Port Jackson Glassfish *Ambassis jacksoniensis*) but also juveniles of commercially important species (eg the Flat Tailed Mullet *Liza argentea*). These species, the most numerically important in the estuary, in turn serve as prey for higher order carnivores, such as the bream *Acanthropagrus australis*

Biodiversity Habitat Provision: Coastal saltmarsh is an important roosting habitat for migratory shorebirds. Migratory birds inhabiting estuaries feed on nutrient rich mudflats exposed by the low tide, but at high tide select roosting habitats providing protection from predators and, if possible, secondary feeding opportunities. A study by Lawler (1998) of 134 roosting sites found 83% to be more than 30 m distant from 5m tall trees, including mangroves. While mangroves were used as roosts by some species, these were primarily those immediately adjacent to the estuary. Spencer (2010) working in the Hunter estuary found waders preferentially using saltmarsh as a night time roost, and feeding on chironomids within saltmarsh pools to supplement their diet.

Carbon sequestration: Mangroves and saltmarsh are highly effective at capturing and storing atmospheric carbon dioxide, a characteristic captured by the term “Blue Carbon”. Saintilan *et al.* (2013) demonstrated relatively high carbon stores within mangrove and saltmarsh in the Hawkesbury River to the north of the region, with higher carbon in the saltmarsh rush *Juncus kraussii* than in the succulent and grass-dominated saltmarsh communities. Kelleway *et al.* (2016) demonstrated that mangrove encroachment of saltmarsh at two locations on the Georges River (Half Moon Bay and Towra Point) over 70 years led to an increase in carbon store, primarily after 40 years. Carbon stores are lower in sandy environments characteristic of the flood-tide deltaic mouths of estuaries and higher in the riverine silts and muds of the fluvial delta and mud basin environments (Kelleway *et al.*, 2016, Saintilan *et al.*, 2013).

Shoreline protection: The root systems of mangrove and saltmarsh protect estuarine shorelines in two important ways. First, they add to the cohesive strength of soils by providing a connected matrix resistant to erosion (Krauss *et al.*, 2014). Second, they promote vertical accretion of sediment, both by interrupting flow and thereby promoting soil deposition, but also more directly through root mass increase over time. In this way the presence of mangrove and saltmarsh improves the capacity of the wetland to increase elevation in relation to sea-level rise. At Homebush Bay, mangrove surface elevation gain has matched the rate of sea-level rise since 2000 (Saintilan and Rogers in prep.), though saltmarsh has been less effective in this regard.

Visual amenity: Shoreline vegetation is an important component of the visual landscape of the urban estuary. Efforts to provide a more aesthetically pleasing environment in relation to concreted channels have been one driver for the reintroduction of coastal saltmarsh in such programs as the Cooks River Urban Water

Initiative. The issue of mangroves growth interrupting views to the estuary is one to which we will return in chapter 4.

2.1.2 TRENDS IN MANGROVE AND SALTMARSH EXTENT

Reviews of mapping conducted in preparation for the NSW Coastal Reforms have highlighted the need for better inventory of coastal wetlands in the Sydney Region, hitherto excluded from SEPP14 (Allan Young pers. comm. 2016). The most recent estimate of the extent of mangrove and saltmarsh in the Sydney region is provided by the NSW Comprehensive Coastal Assessment, and incorporated into the databases maintained for the Estuaries Monitoring, Evaluation and Reporting (MER) strategy (NSW Government Unpublished Data). These data, correct as of 2006, are provided in Table 2.1.

Table 2.1: Area of mangrove and saltmarsh in the estuaries of the Sydney Region. Source: Comprehensive Coastal Assessment, data provided by P. Scanes, OEH.

Estuary	**Mangrove (ha)**	**Saltmarsh (ha)**
Pittwater	17.5	2.7
Narrabeen Lagoon	0.07	0.8
Dee Why lagoon	0	6.3
Manly Lagoon	0.02	0
Port Jackson	184.7	9.4
Cooks River	10.8	0.3
Georges River	382.4	84.0
Botany Bay	229.6	76.2
Port Hacking	29.9	12.8

The mangrove and saltmarsh extent listed in Table 2.1 is a highly reduced remnant of the pre-European extent. It is difficult to estimate the original area of mangrove and saltmarsh across the region, but significant developments up until the 1970's saw the reclamation of large areas of mangrove and saltmarsh for commercial, residential and recreational facilities, and Municipal tips, in the Georges River, Cooks River, and Parramatta River and Lane Cove River. Interrogation of early plans and documents by Lynne McLoughlin has demonstrated the effect of sedimentation providing fresh habitat for mangroves headward of their pre-20th century extent in the Lane Cove and Parramatta rivers and the replacement of saltmarsh by mangrove across the range of estuarine environments since earliest European settlement (McLoughlin, 1987).

Comparison of historic and aerial photography has demonstrated a consistent increase in mangrove extent and subsequent saltmarsh decline in intertidal flats across the region (Table 2.2) (Saintilan & Williams, 1999, Saintilan & Williams, 2000). The decline has prompted the listing of Coastal Saltmarsh as an Endangered Ecological Community in three NSW Bioregions, including the Sydney Basin bioregion. The cause of mangrove encroachment into saltmarsh across SE Australia is being investigated, and best available

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science for the region suggests changes in relative sea-level are likely to have been an important driver (Rogers *et al.*, 2006), and is consistent with a global trend of mangrove proliferation at poleward limits of mangrove range (Saintilan *et al.*, 2014).

Table 2.2: Increase in mangrove and decline of saltmarsh in the Sydney Region- results of API surveys

Location	Saltmarsh Loss (%)	Mangrove increase (%)	Period	Source
Berowra/Marramarra Creeks, Hawkesbury River	25	30	1941-1994	Williams and Watford 1997
Careel Bay, Pittwater	92	551	1938-1994	Wilton 1997
Couranga Point, Hawkesbury River	30	30	1954-1994	Saintilan and Hashimoto 1998
Lane Cove River	Yes, not quantified	Yes, not quantified	1930-1986	McLoughlin 1987
Homebush Bay, Parramatta River	>80	65	1930-1986	Clarke and Benson 1988
Quibray Bay, Botany Bay	70	33	1950-1997	Evans 1997, Evans and Williams 2001
Weeney Bay, Botany Bay	100		1950-1994	Fenech 1994;
Woolooware Bay, Botany Bay	63		1950-1994	Fenech 1994; Hughes 1998
Towra Point, Botany Bay	30	36	1942-1997	Mitchell and Adam 1989

2.2 Contemporary Pressures and Impacts

2.2.1 CLIMATE CHANGE IMPACTS

Mangroves and saltmarshes are sentinels of climate change, due to their situation within narrowly defined elevations in relation to sea-level, and their sensitivity to changes in atmospheric carbon dioxide, temperature and rainfall. This sensitivity is evident in palaeo-stratigraphic studies of mangrove and saltmarsh extent through the Holocene, which show transitions in distribution and community composition in relation to changing atmospheric, climatic and hydrological conditions (Saintilan & Hashimoto, 1999). The following section details the basis for predictions of impacts resulting from alterations in sea-level, temperature, atmospheric carbon dioxide concentrations, and rainfall in the Sydney region.

Sea-level Rise

The capacity of a wetland to remain stable is determined by the relationship between the rate of sea-level rise and the rate at which the marsh surface elevation is able to increase. As mentioned above, mangrove and saltmarshes promote vertical elevation gain by trapping sediment and building root mass. For example, Figure 2.1 shows the rate of surface elevation gain of mangrove, saltmarsh and mixed mangrove-saltmarsh habitat at Homebush Bay in the Parramatta River. Mangrove surface elevation gain has matched the rate of sea-level rise in the Port Jackson estuary over the period, suggesting that in this site at least, the wetland is capable of remaining stable under current rates of sea-level rise (circa 3.4 mm per annum). However, the adjacent saltmarsh has not gained elevation over the 10-year period, and the lower capacity of saltmarsh to gain elevation is characteristic of saltmarshes across the region. These results have indicated that sea-level rise is a factor in the observed mangrove encroachment of saltmarsh, and that this trend is therefore likely to continue as sea-level continues to rise. These results indicate that there is insufficient sediment supply to the saltmarsh to offset the increased inundation frequency that is likely to occur with sea-level rise.

The replacement of saltmarsh by mangrove may therefore be a response of the wetland to sea-level rise, providing a negative feedback whereby the shifting character of the vegetation counterbalances the effect of increased inundation frequency. For this reason, several authors have argued that mangrove removal in this context may block a natural response building long-term resilience of the wetland to sea-level rise (Kelleway *et al.* in prep.).

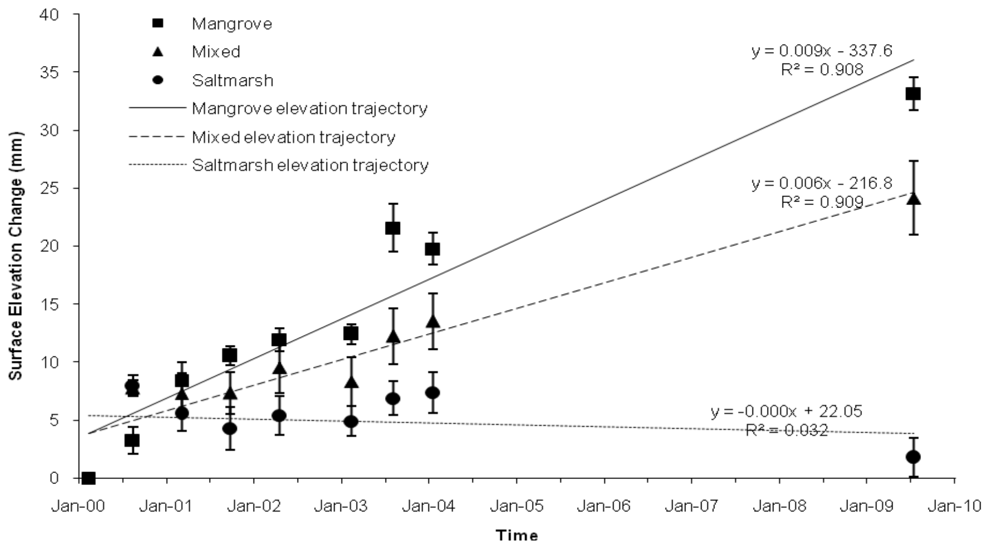

Figure 2.1: Surface elevation trajectories in mangrove, mixed mangrove and saltmarsh and saltmarsh habitats at Homebush Bay, Parramatta River (Source: Rogers, 2010).

The limited capacity of the upper intertidal environment to gain surface elevation may also allow the landward expansion of the wetland over time. Under low to moderate rates of sea-level rise, it is possible for mangrove and saltmarsh wetlands to increase in extent if unimpeded by barriers to dispersal (Rogers *et al.*, 2013b).

However, palaeo-stratigraphic records of the post-glacial marine transgression suggest upper limits to the capacity of wetlands to gain elevation with sea-level rise. These upper limits are poorly documented, but are thought to be lower than the high-end IPCC projections of ~10mm per annum sea-level rise. If this eventuates, the prospect of coastal squeeze (the erosion of the wetland seaward edge effaced with barriers to landward transgression) becomes an important restriction on wetland survival in urbanised estuaries.

Increased Temperature

The diversity of mangroves consistently decreases with increasing latitude on the east Australian coast, and only the two most cold-resistant species occupy the Sydney region: grey mangroves (*Avicennia marina*) and river mangroves (*Aegiceras corniculatum*). A recent study has noted the expansion in the range of the tropical species *Rhizophora stylosa* on the north NSW coast, with the extent of expansion (100 km) consistent with temperature increases over the past century (Wilson & Saintilan, 2012). A further 1.5-2ºC increase in mean minimum temperature projected as probable by 2100 would see this species, and others within the Rhizophoraceae (eg. black mangrove, *Bruguiera gymnorrhiza*) capable of surviving within the Sydney region.

Perhaps more concerning is the possible influence of elevated temperature on the reproductive capacity of several saltmarsh species. Saltmarsh diversity increases in direct correlation with mean minimum temperature on the Australian east coast (Saintilan, 2009). Extrapolating from this trend, saltmarsh diversity in the Sydney region would be expected to decrease from 50 species to approximately 40 if mean minimum temperatures rise 1.3°C (i.e. to resemble the northern NSW and SE Queensland assemblages), possibly due to inhibitive effects of temperature on germination (though this requires further study). The following species are particularly vulnerable, not being located in latitudes characterised by these temperatures and find their northern limit in the Sydney bioregion: *Rhagodia baccata; Austrostipa stipoides; Distichlis distichophylla; Gahnia filum; Lampranthus tegens; Wilsonia backhausii* (endangered); and *Wilsonia humilis*.

Elevated CO_2 and Rainfall

These two factors have been grouped together because of their influence over mangrove survival in saltmarsh environments. Both elevated CO_2 and rainfall potentially act to decrease the physiological aridity effect of the upper intertidal environment. Regional climate modelling under the NARCLIM program (Fita *et al.*, 2012) has projected possible increases in rainfall in the Sydney region of up to 30 percent by 2040. Changes in rainfall regime towards wetter conditions have been implicated in mangrove encroachment of saltmarsh in SE Queensland (Eslami-Andargoli *et al.*, 2009) and may therefore contribute to continued mangrove encroachment in the Sydney region. Elevated atmospheric carbon dioxide contributed to “carbon fertilisation” of plants globally. However, plants utilising the C4 photosynthetic pathway, including the saltmarsh grass *Sporobolus virginicus*, are saturated in relation to CO_2 uptake at pre-industrial concentrations and are therefore immune from this benefit. The increase in atmospheric CO_2 is therefore one factor favouring the growth of mangrove over saltmarsh in the Sydney region. The effect of high ambient CO_2 may be particularly advantageous in the upper intertidal environment where mangrove and saltmarsh interact, because the presence of salt and generally dryer conditions at this elevation create an environment of water stress that enhanced CO_2 partially overcomes (Ball *et al.*, 1997, Saintilan & Rogers, 2015).

2.2.2 OTHER ANTHROPOGENIC IMPACTS

Proximity to the urban environment exposes mangrove and saltmarsh wetlands to a range of direct and indirect anthropogenic impacts in addition to climate change. These can be summarised as follows:

- *Waste and spills*: the proximity of Towra Point shipping and refinery operations presents a degree of exposure to petrochemical spills, a pollutant known to smother mangrove pneumatophores leading to asphyxiation.
- *Trampling and cutting*: saltmarshes are easily damaged by off-road vehicles (Kelleway, 2005). Recovery of chenopod saltmarsh vegetation following vehicle or foot trampling can be slow- up to 5-10 years (Laegdsgaard, 2006; Saintilan unpublished data). Previous research has identified at least five location on the Georges River that have sustained substantial vehicle damage (Kelleway, 2005), though this impact may be widespread. Illegal cutting of mangroves is also common, and is discussed in Chapter 4.

- *Reclamation*. The rate of mangrove and saltmarsh reclamation has slowed considerably over the past several decades, with the introduction of the Fisheries Management Act 1994 (with reference to the protection of mangrove and other marine vegetation); the listing of Coastal Saltmarsh as vulnerable under the NSW Threatened Species Conservation Act and the Commonwealth Environment Protection and Biodiversity Conservation Act; and, outside of the Sydney Region, the protection of coastal wetlands under SEPP14.
- *Ferry and boat wake erosion*: Erosion has been occurring in sections of the Parramatta River associated with increased hydrodynamic energy, wave action and wakes from regular ferry and boat traffic (Kelleway *et al.*, 2007). This is likely to occur in other locations in association with other boating activity.
- *Eutrophication*: the estuaries of the Sydney region have been subject to elevated nutrient concentrations associated with population increase and urbanisation. Nitrogen concentrations in estuarine waters have increased substantially on modelled pre-European levels (Table 2.3). Given that nitrogen limitation is common in intertidal wetlands (Feller *et al.*, 2003) the increase in Nitrogen is expected to promote growth of both mangrove and saltmarsh, and potentially contribute to mangrove colonisation of upper intertidal environments. There is also evidence that nitrogen loading may truncate estuarine wetland food webs (Mazumder *et al.*, 2015).
- *Stormwater discharge*: Stormwater outfalls are commonly located at the landward fringe of saltmarshes in the Sydney region. Anecdotal observations from Sydney Olympic Park and Careel Bay suggest that discharge points are associated with concentrated mangrove recruitment into saltmarsh, possibly a consequence of wetter and less saline conditions associated with point-source discharge. While wetlands potentially play an important role in "polishing" water prior to entry into the estuary, this should be concentrated on less sensitive wetlands (mangrove rather than saltmarsh).
- *Shoreline modification and wave climate*: The alteration of shoreline configuration and the imposition of hard surfaces may alter the wave climate of an estuary, leading to changes in erosion and sedimentation patterns more broadly. For example, shoreline alterations associated with Sydney Airport and Port Botany are likely to have influenced patterns of erosion and sedimentation at Towra Point, the largest contiguous area of mangrove and saltmarsh within the region.

Table 2.3: Catchment modification in the Sydney region and nutrient loading in the Sydney region (Mazumder et al. 2015)..

	Georges River	***Botany Bay***	***Parramatta River***	***Brisbane Water***
Catchment area (km^2)	450940	18566	67918	39025
% Catchment cleared	59	61.5	86.1	52.1
Population	851544	110990	652176	100969
Population density (no/km^2)	900.3	1210.5	2460	644.4
TN ($t\ yr^{-1}$)	450940	18466	67918	39025
% increase TN on pre-development	870.6	824.7	397	188
Flushing time	62.5	39.9	17.3	24.6

2.3 Consequences of current management activities on the provision of ecosystem services

Mangroves and saltmarshes are subject to strong legislative and policy protection in the Sydney region, in spite of their historic exclusion from State Environmental Planning Policy (SEPP) 14 Coastal Wetlands. The cutting of mangrove is prohibited under the *NSW Fisheries Management Act* (1994) and coastal saltmarsh is protected as a vulnerable ecological community under the *NSW Threatened Species Conservation Ac*t (1995) and as an endangered ecological community under the *Commonwealth Environment Protection and Biodiversity Conservation Act* (1999).

Impacts of management activities on intertidal wetland ecosystem services are primarily occurring as the interacting effects of diffuse pressures. Table 2.4 provides a summary of the pressures and impacts of a range of catchment and estuary modifications, the risk they pose, and the possible mitigation. Management of the movement of people in sensitive ecosystems is particularly challenging in urban environments, but an important element of saltmarsh and mangrove management. Boardwalks in saltmarshes may be an effective way of reducing foot traffic and trampling. Vehicle exclusions have been effectively applied to minimise off-road-vehicle impacts in saltmarsh across the region (Kelleway, 2005). Speed restrictions on boat traffic in the vicinity of intertidal wetlands has been the primary mechanism for controlling erosion from wake, but this may need to be supplemented by ecologically sensitive shoreline armoury, such as the imposition of oyster reefs at the sea-ward edge of wetlands.

Mangroves have flourished in the estuaries of the Sydney region in spite of elevated levels of nutrients and contaminants. The removal of heavy industry on harbour foreshores and improvements to wastewater and urban runoff treatment has lessened the threat of ongoing contaminant and nutrient impacts in recent decades, although the legacy of contaminated sediments remains in certain locations (Barry *et al.* 2001; Birch *et al.* 2008). While mangroves are susceptible to oil spills, the likelihood of this being a significant threat into the future is low, following the removal of oil refining facilities in Sydney Harbour and Botany Bay. The effect of elevated nutrients on broader ecosystem function in the aquatic environment is less well understood. Mazumder *et al.* (2015) demonstrated that nitrogen derived from effluent is a major source of nitrogen in mangrove-based ecosystems in the Sydney region, but that this did not deleteriously effect the linkage between primary production, zooplankton production and the feeding of zooplanktivorous fishes. However, they provide some preliminary evidence that the total food-chain length (from primary producer to estuarine fish) may be lower in estuaries more heavily impacted by nutrient input (Sydney Harbour and Botany Bay, when compared to Brisbane Waters).

The catchments of the estuaries of the Sydney region are amongst the most urbanised in Australia. Management of sediment, nutrient and water flows from catchment to estuary has the potential to influence mangrove and saltmarsh extent and condition. Increased sedimentation resulting from the clearance of vegetation during urbanisation has contributed to sedimentation and the deposition of new intertidal flats for mangrove colonisation (Haworth, 2002, McLoughlin, 1987). This has built “elevation capital” in intertidal wetlands, providing additional buffering in relation to sea-level rise. However, the hardening of urban surfaces and the paradigm of trapping sediment in urban tributaries may reduce sediment delivery to intertidal wetlands in the future.

It is noteworthy how many of the pressures listed in Table 2.4 promote the growth of mangrove, and in particular mangrove growth in upper intertidal saltmarsh environments. For example, elevated nutrient levels, and alterations to estuary entrance conditions both promote mangrove colonisation, compounding the effect of elevated sea-level, temperature, CO_2 and rainfall in facilitating mangrove colonisation of formerly saltmarsh environments. Modelling of outcomes for mangrove, saltmarsh and casuarina under a range of sea-level rise scenarios is presented in Chapter 4. The chapter provides a strong case that continued expansion of mangrove is to be expected, partly at the expense of saltmarsh and casuarina under most sea-level rise scenarios.

Several small-scale saltmarsh restoration projects have been completed the Sydney region, for example in Cooks River under the Cooks River Urban Water Initiative, and in the Sydney Olympic Park. Prioritising saltmarsh restoration over mangrove has been justified by the vulnerability of saltmarsh to existing and projected pressures, and the significance of saltmarsh to migratory shorebirds in particular. These areas are likely to be converted to mangrove in the absence of ongoing maintenance. We argue in Chapter 5 for a re-consideration of mangrove management along the lines suggested in the proposed urban mangrove policy, allowing ecological and social objectives to be set at the catchment scale.

Table 2.4: Summary of the pressures and impacts of a range of catchment and estuary modifications, the risk they pose, and the possible mitigation.

Pressure	Impact	Consequence	Likelihood	Confidence	Outcome	Mitigation
Sea-level rise	Drown-out of existing intertidal wetland. Landward expansion	High	High	Moderate	Loss of most mangrove and saltmarsh, with associated ecosystem services	Sediment nourishment. Accommodation of landward retreat
Temperature rise	Introduction of mangrove species. Loss of saltmarsh diversity	Moderate	High	Low	Alterations to floristic structure.	Removal of invasive native mangroves if justified
Elevated rainfall and CO_2	Further mangrove encroachment of saltmarsh	Moderate	Moderate	Low	Loss of saltmarsh, and associated migratory bird habitat	Accommodation of landward saltmarsh retreat. Mangrove exclusion
Petrochemical Spills	Dieback of mangroves	High	Low	High	Visual impact of dead mangroves, loss of carbon	Rapid response to contain spills
Vehicle and trampling effects	Loss of saltmarsh vegetation	Moderate	Moderate	High	Loss of saltmarsh cover. Disturbance to migratory bird roosting	Public education and signage, vehicle gates
Reclamation	Loss of habitat	Moderate	Low	High	Loss of all associated ecosystem services	Appropriate land-use planning
Boat wake erosion	Loss of habitat	Moderate	High	High	Retreat of shoreline	Boating restrictions, ecosystem sensitive shoreline armouring
Eutrophication	Increased vegetative growth	Low	Moderate	Moderate	Competitive advantage of mangrove compared to saltmarsh	Water sensitive urban design, wastewater treatment, catchment scale nutrient management
Stormwater run-off	Introduction of potential pollutants, modification of salinity and hydrology regimes	Low	High	High	Localised change in vegetation composition	Relocation of stormwater outfalls away from sensitive wetlands
Hardening of shorelines	Redistribution of wave energy	Moderate	High	High	Erosion of intertidal shorelines, and altered sedimentation patterns and tidal regimes	Ecosystem sensitive shoreline armouring
Dredging of ICOLL entrances	Alteration of tidal regime	Moderate	High	High	Alteration of vegetation distribution and composition	Optimise entrance opening frequency

Chapter Three: First pass assessment of the vulnerability of mangrove and saltmarsh in the Sydney Region

3.1 Introduction

This chapter provides a first pass assessment of the vulnerability of mangrove and saltmarsh to inundation and erosion related to sea-level rise and other climate related processes. First pass assessments have been advocated as a means for prioritising investment in high resolution models of climate change vulnerability at locations where risks are high (Rogers & Woodroffe, 2016), and in doing so are a useful tool for prioritising investment in spatial planning to improve ecosystem outcomes in urban areas. The approach used in this study is to create choropleth indicator maps of the exposure, sensitivity and adaptive capacity of mangrove and saltmarsh to inundation and erosion associated with climate change, and to integrate these maps to provide an indication of the overall vulnerability of mangrove and saltmarsh to climate change. The study focusses on mangrove and saltmarsh within the local government areas of the councils that are part of the Sydney Coastal Councils Group regional organisation of councils. Additional components of this study include a review of impacts and threats to mangrove and saltmarsh in the Sydney region, higher resolution modelling of changes in mangrove and saltmarsh distribution at Wolli Creek under a range of possible future sea-level rise scenarios, and discussion of possible adaptation options, including strategic land use planning.

3.2 Method

3.2.1 STUDY AREA

The Sydney metropolitan area lies within the Sydney Basin and is flanked by the New England Fold Belt and the Lachlan Fold Belt. The geology of the region is dominated by the resistant Hawkesbury Sandstone that deposited in the basin during the Permian-Triassic period. Over this period rivers incised deep valleys into the bedrock of the Sydney Basin. During the last marine transgression, commencing approximately 20 ka, sea levels rose and flooded the incised river valleys. This inherited geology creates that landscape upon which mangrove and saltmarsh have evolved within the Sydney Basin (Roy, 1984) (Figure 3.1a).

The drowned river valleys of Sydney Harbour, Hawkesbury River and Port Hacking, once the deeply incised river valleys prior to the last marine transgression, are uniquely restricted to the Sydney Basin of NSW. In these estuaries, mangrove and saltmarsh development is typically restricted to shorelines and fluvial deltas on tributaries where bedrock depth is shallow and hydrodynamic energy facilitates the accumulation of sediments over time (Figure 3.1a). These accumulated sediments provide the surface upon which mangrove and saltmarsh establish and grow. The drowned river valleys of the Sydney Basin, together with the oceanic embayment of Botany Bay, are the primary estuaries within the Sydney Basin. The ocean embayment of Botany Bay, which is dominated by marine processes and has relatively little freshwater inflow (Roy *et al.*, 2001), supports the most expansive areas of mangrove and saltmarsh in the Sydney region at Towra Point (Mitchell & Adam, 1989); here coastal wetland development is facilitated by the development of large areas of Aeolian and alluvial sands.

Long-shore drift along the coastline has facilitated the accumulation of barrier sands between headlands, constraining run-off from catchments through small lakes whose entrances intermittently close due to the dominating influence of long-shore drift on sediment deposition within the entrances of the small lakes. These lakes, including Narrabeen Lagoon, Dee Why Lagoon and Manly Lagoon, are commonly referred to as intermittently closed and open lakes and lagoons or ICOLLs. Over time, ICOLLs have infilled with sediment enabling the development of intertidal surfaces that support estuarine vegetation establishment and growth. Prevailing inundation regimes within ICOLLs often limit mangrove establishment and/or survival and saltmarsh vegetation is more typical in these estuaries (Haines *et al.*, 2006).

This vulnerability assessment was restricted to saline coastal wetland vegetation occurring within the local government areas of the Sydney Coastal Councils Group (SCCG) regional organisation of councils (Figure 1b). To identify potential pathways for longitudinal expansion of coastal wetlands along estuaries, the study area was expanded to include all coastal wetland vegetation associated with the SCCG as well as low-lying land associated with the Wolli Creek. This study focussed on low-lying sediments that have accumulated over the Quaternary period and upon which mangrove and saltmarsh develop (Figure 1c).

3.2.2 DEFINING VULNERABILITY

The method employed in this study focusses on biophysical aspects of vulnerability and builds upon the approach of Rogers and Woodroffe (2016) who used geomorphology as an indicator of the vulnerability of estuaries in southern NSW to climate change. In this regard the approach relies upon the definition of vulnerability advocated in the fourth assessment report of the IPCC (Parry *et al.*, 2007), whereby:

> *Vulnerability is the degree to which a system is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate change and variation to which a system is exposed, its sensitivity, and its adaptive capacity.*

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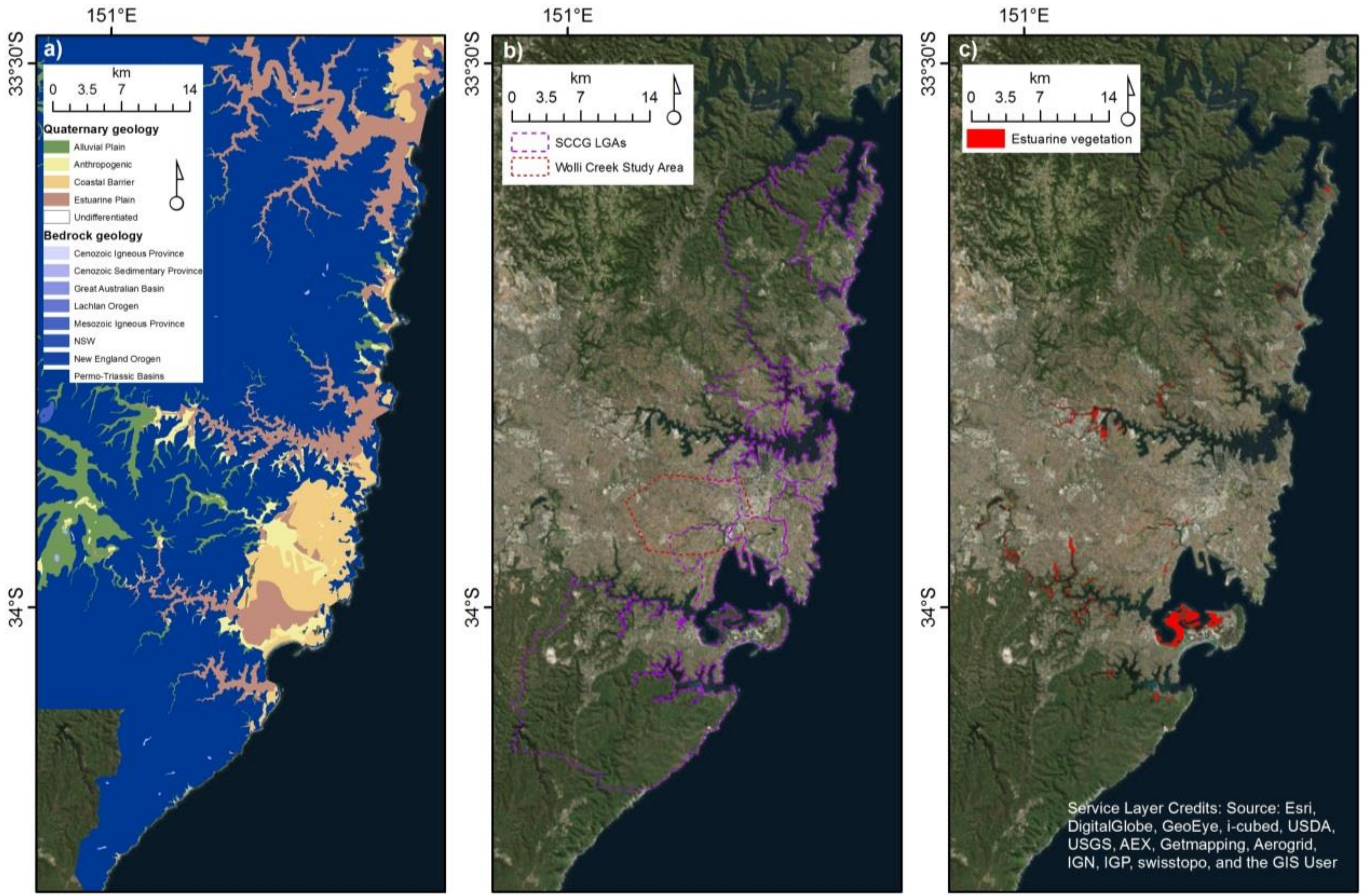

Figure 3.1: Location of study site in the Sydney Basin. **a)** bedrock and Quaternary geology of the Sydney Basin, **b)** extent of the Sydney Coastal Councils Group regional organisation of councils and the Wolli Creek study area, and **c)** extent of estuarine vegetation within the Sydney Basin.

As per Rogers and Woodroffe (2016), the approach is dependent upon some caveats:

- The approach focusses solely upon biophysical factors. While few studies adequately integrate biophysical and socio-economic aspects within vulnerability assessments, we have chosen to incorporate socio-economic factors through post-hoc analyses of the effect of socio-economic factors on the biophysical vulnerability of mangrove and saltmarsh. Socio-economic factors are considered in later chapters.
- The approach focuses on hazards and climate change related effects associated with inundation and erosion of coastal wetlands. Other stressors that may affect coastal wetlands, such as water temperature, acidification, and water quality have not been incorporated in the analysis.
- Coastal geomorphology provides a useful foundation from which to assess the vulnerability of coastal landforms to climate change (Pethick & Crooks, 2000, Rogers *et al.*, 2015, Rogers & Woodroffe, 2016). However, coastal ecosystems are also defined on the basis of biotic factors, particularly the in situ vegetation. Therefore, this assessment incorporates the influence of climate change on both geomorphology and vegetation.
- The approach focusses on capturing the exposure, sensitivity and adaptive capacity of coastal wetlands within the study area. These factors relate not only to the in situ vegetation but also the depositional environments in which they establish.
- Exposure is defined as 'the character, magnitude and rate of change in climate drivers operating on a system' (Parry *et al.* 2007). In high resolution assessments exposure would be defined on the basis of hydrodynamic models of sea-level rise, wave height and wave energy, however in this indicator-based assessment spatial variation in these processes was characterised on the basis of the relatively simple relationship between exposure of landforms and associated vegetation to these processes and their elevation. More specifically, lower elevations were characterised as being more exposed to inundation and erosion effects than higher elevations.
- Sensitivity is defined as 'the degree to which a system is affected by climate variability or change' (Parry *et al.* 2007) and was defined on the basis of the characteristics of the sediments upon which mangrove and saltmarsh form and their sensitivity to erosion and inundation.
- Adaptive capacity is defined as 'the ability of a system to adjust to climate change' (Parry *et al.* 2007) and in our study was characterised on the basis of the vegetative response of mangrove and saltmarsh to changes in coastal processes and did not include other adaptation mechanisms (planned, facilitated, anticipatory). In this study autonomous adaptation largely constitutes building land elevation through natural processes of sediment accretion or plant productivity; and binding of sediments due to below-ground biomass additions and complex root structures. Autonomous adaptation contrasts with planned adaptation such as creating living shorelines, or establishing seawalls and buffers.
- Scale is an important consideration in vulnerability assessments. First order assessments typically cover large spatial scales and use lower resolution data; this trade-off balances assessment outcomes against computing resource requirements. While higher resolution digital elevation data is available for the Sydney region (i.e. Lidar-derived digital elevation models), this assessment emphasises the spatial scale, rather than spatial resolution. Accordingly, this first pass assessment utilises lower resolution elevation data derived from the Shuttle Radar Transfer Mission, as opposed to higher resolution LIDAR-derived digital elevation models so as to maximise the spatial scale of the assessment. The Lidar-derived DEM has been used for demonstration purposes in a high resolution assessment of changing mangrove and saltmarsh distribution at Wolli Creek using as high resolution modelling approach (i.e. Sea Level Affecting Marshes Model).

3.2.3 ASSESSMENT APPROACH

The assessment used a raster-based approach within GIS using the ArcGIS spatial analyst extension. Input datasets were used as proxy indicators of coastal wetland exposure, sensitivity and adaptive capacity.

- *Shuttle Radar Topography Mission-derived 1 second Digital Elevation Model (SRTM-DEM)* was the primary dataset for characterising exposure of coastal landforms to the effects of inundation and erosion. The SRTM-DEM dataset was also used to derive the slope of landforms to indicate the sensitivity of landforms to inundation. The Slope function in ArcGIS was used to generate the *Slope* surface; the generated slope surface is a function of the spatial resolution of the input dataset. The SRTM-DEM is a raster surface with a cell size of approximately 30 m and elevation to the nearest metre.
- *Coastal Quaternary geology mapping* was the primary dataset for characterising the sensitivity of coastal landforms to inundation and erosion. The NSW Coastal Quaternary Geology, which was prepared by the NSW Department of Mineral Resources as part of the Comprehensive Coastal Assessment of Southern NSW (Troedson *et al.*, 2004), was recently revised to incorporate the Sydney metropolitan area (Troedson, 2015), and constitutes a significant increase in the coverage of coastal Quaternary geology mapping of NSW. This mapping differentiates lithified (bedrock) from unlithified Quaternary sediments that are more sensitive to erosion. Quaternary sediments can be further classified on the basis of the depositional environment in which they were deposited; this enables discrimination of the finer fluvial sediments from the coarser marine sediments that deposit under high energy conditions and may be less sensitive to erosion due to their mass.
- *Sydney's salty vegetation mapping* was the primary dataset for characterising the adaptive capacity of coastal ecosystems to inundation and erosion. This dataset was originally prepared by OEH (2013a) and a subset of 53 vegetation communities were selected by the Working Group of the Sydney's Salty Communities program following a series of workshops with member councils within the Sydney Coastal Councils Group.. Vegetation communities pertinent for this assessment included i) Coastal Swamp Paperbark-Swamp Oak Scrub (S_FoW12), ii) Estuarine Swamp Oak Forest (S_FoW08), iii) Estuarine Saltmarsh (S_SW02), iv) Estuarine Reedland (S_FrW06), and v) Estuarine Mangrove Forest (S_SW01). These communities were used to differentiate salinity tolerance, capacity for building elevation, and capacity for binding sediments and buffering wave action. This data set was used to indicate the capacity of ecosystems to adapt to increasing exposure to coastal processes that cause inundation and erosion.

All input datasets were prepared for the analysis. The cell position and size of SRTM-DEM was used as the template for converting the vector layers of Coastal Quaternary geology and Sydney's salty vegetation mapping to raster surfaces.

Two composite choropleth maps were prepared that provided a relative indication of the vulnerability of coastal wetlands in the study region to inundation and erosion. To generate these maps, input raster surfaces were processed according to the geomorphic and biotic criteria detailed in Table 3.1. The extract function was used to extract relevant cells from the input datasets. These cells were then reclassified and assigned a value of 1-3 depending on whether the extracted cells were indicative of high, moderate (mod.) or low exposure, sensitivity or adaptive capacity. Only 3 cell values were employed to simplify processing of raster surfaces and to ensure that equal weighting was given to each component of vulnerability. The composite choropleth maps characterising inundation and erosion vulnerability were generated by adding raster surfaces that characterised the exposure, sensitivity and adaptive capacity of cells using the raster calculator tool, with cell scores ranging between 1-9 for both choropleth maps; low cell scores indicated lower vulnerability to either inundation or erosion. To aid visual presentation in prepared choropleth maps, erosion and inundation raster surfaces were stretched to show a graduation of cells from low to high vulnerability.

A final raster surface of the vulnerability of cells to both inundation and erosion was compiled by adding the original cell scores for inundation and erosion using the raster calculator tool to create an integrated raster surface with cell scores ranging between 0 and 18. As the approach considered both biotic and geomorphic factors contributing to the vulnerability of mangrove and saltmarsh, the assessment broadly applied to depositional areas, as defined by Quaternary geology mapping, and was not limited to areas that currently support mangrove and saltmarsh. This approach was founded upon evidence that mangrove and saltmarsh are spatially dynamic, and alterations in their spatial position over time are a natural adaptive response to changing environmental conditions (Rogers *et al.*, 2016). However, in some instances overlying land-cover and land-use may supplant the capacity for a natural adaptive response to changing environmental conditions; this effect is commonly termed coastal squeeze and refers to the limitation of landward migration of coastal ecosystems as an adaptive response due to barriers such as buildings and infrastructure. The analysis therefore needed to be confined to areas conducive to supporting mangrove and saltmarsh now or in the future; that is areas where landward migration of mangrove and saltmarsh was not limited by existing buildings and infrastructure.

In consultation with Sydney Coastal Councils Group (Duncan Webb *pers comm*.), the integrated vulnerability assessment was constrained, or masked, using numerous approaches.

- *Masking on the basis of SCCG LGAs*: the vulnerability assessment was initially masked to areas that were within the study area defined by the boundaries of the SCCG LGAs and the Wolli Creek study area. As described above, this approach assumed all landforms could support mangrove and saltmarsh and ignored the influence of existing buildings and infrastructure on wetland distribution. This was achieved by converting the study area extent to a raster dataset and using the extract by mask tool in ARCGIS to extract cells that occurred within the study area boundary. This step was also applied to the two choropleth maps for inundation and erosion vulnerability.
- *Masking on the basis of SCCG LGAs and built up areas*: The vulnerability assessment was also masked so that that assessment only applied to non-built areas where mangrove and saltmarsh could feasibly occur. In doing so, it was presumed that all non-built areas could support mangrove and saltmarsh now and in the future should hydrodynamic conditions be favourable. This approach did not account for any future strategic planning or planned adaptation responses that could alter the area suitable for mangrove and saltmarsh in the future, and also assumed that cleared land that was not built-up (e.g. parkland, golf courses, airports) could potentially support mangrove and saltmarsh in the future. This step was achieved using the *BuiltUpAreas* shapefile contained within the Quaternary Geology base maps, which was converted to a raster dataset and reclassified to identify areas that were non-built up areas. The extract by mask tool in ArcGIS was used to extract cells that coincided with non-built up areas. This step was also applied to the two choropleth maps indicating inundation and erosion vulnerability.
- *Masking on the basis of the distribution of estuarine vegetation:* To quantify the vulnerability of in situ vegetation, the vulnerability assessment was masked on the basis of the current distribution of Coastal Swamp Paperbark-Swamp Oak Scrub, Estuarine Swamp Oak Forest, Estuarine Saltmarsh, Estuarine Reedland, and Estuarine Mangrove Forest from *Sydney's salty vegetation mapping*, thereby quantifying the vulnerability of in situ vegetation. This technique did not account for any changes in the distribution of mangrove and saltmarsh over time and only applied to the current vegetation distribution.
- *Masking on the basis of 90cm water level projections and distribution of estuarine vegetation* (McInnes *et al*., 2012)*:* To account for potential changes in the distribution of mangrove and saltmarsh with sea-level rise, modelling of +90cm water levels (provided by SCCG) was integrated with the distribution of estuarine vegetation to provide an indication of vulnerability of mangrove and saltmarsh both now and in the future.

Table 3.1: Approach to assessing the vulnerability of coastal wetland vegetation to climate change effects causing inundation and erosion. Indicators for the different vulnerability components were established and input datasets that spatially represent those indicators were identified. Cell scores were assigned to raster surfaces on the basis of the indictors and their relationship to various components of vulnerability. Choropleth maps for each vulnerability component were developed on the basis of cell scores.

Effect	*Component*	*Input data*	*Explanation*	*Cell label*	*Cell description*
Inundation (I)	Exposure (IE)	Elevation (SRTM-DEM)	Lower elevations more exposed to inundation by proximity, higher elevation less exposed. Based on simple bath-fill approaches that have been widely used as an indicator of vulnerability; see for example The First Pass National Coastal Risk Assessment (DCC, 2009). Rogers *et al.* (2012) demonstrates why bath-fill approaches do not constitute a high resolution, quantitative assessment.	High (3)	Elevation 0-1 m
				Mod. (2)	Elevation 1-2 m
				Low (1)	Elevation 2-5 m
				Nil (0)	Elevation > 5 m
	Sensitivity (IS)	Quaternary geology (NSW Coastal Quaternary Geology)	Low slopes indicative of well-developed Quaternary sediments that create ideal habitat for coastal wetlands now and in the future. Low slopes indicative of greater sediment deposition and wetland development, and less sensitivity to sediment movement. This is synonymous with the capacity of mature estuaries exhibiting greater mangrove and saltmarsh development than immature estuaries, as discussed by Roy *et al.* (2001), though applied at a smaller spatial scale.	High (3)	Quaternary geology + Slope>5°
				Mod. (2)	Quaternary geology +Slope 2-5°
				Low (1)	Quaternary geology +Slope <2°
		Slope (SRTM-DEM derived)		Nil (0)	Bedrock geology
	Adaptive capacity (IAC)	Vegetation (Sydney's Salty Veg)	Mangrove, typically at lower elevations, is less sensitive to changes in inundation as it can build elevation, through accretion and plant productivity, at higher rates than saltmarsh. Based on relationships between elevation gain and vegetation within southeastern Australia (Rogers *et al.*, 2005, Rogers *et al.*, 2006). Adjoining upland vegetation sensitive to salinity changes (Greenwood & MacFarlane, 2006).	Low (3)	Coastal Swamp Paperbark-Swamp Oak Scrub (S_FoW12), Estuarine Swamp Oak Forest (S_FoW08)
				Mod. (2)	Saltmarsh (S_SW02), Estuarine reedland (S_FrW06)
				High (1)	Mangrove (S_SW01)
				Nil (0)	Other Veg
Erosion (E1)	Exposure (EE)	Elevation	Lower elevations more exposed to erosive wave action.	High (3)	Elevation 0-1 m or Marine sediments
				Mod. (2)	Elevation 1-2 m
				Low (1)	Elevation 2-5 m
				Nil (0)	Elevation > 5 m
	Sensitivity (ES)	Quaternary geology	Quaternary sediments more sensitive to erosion, particularly fine material associated with fluvial deposits, than bedrock geology.	High (3)	Fluvial sediments
				Mod. (2)	Estuarine sediments
				Low (1)	Marine sediments (undiff, anthro)
				Nil (0)	Bedrock geology
	Adaptive capacity (EAC)	Vegetation	Mangrove has greater capacity to buffer wave action and accumulate sediments than saltmarsh.	Low (3)	Coastal Swamp Paperbark-Swamp Oak Scrub (S_FoW12), Estuarine Swamp Oak Forest (S_FoW08)
				Mod. (2)	Saltmarsh (S_SW02), Estuarine reedland (S_FrW06)
				High (1)	Mangrove (S_SW01)
				Nil (0)	Other Vegetation

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3.3 Results and Discussion

3.3.1 MASKING EFFECT

Due to the nature of the assessment approach, the initial assessment provides an indication of vulnerability for all cells that had a score, including within estuaries, built-up areas, and areas beyond the study areas (Figure 3.2). Consequently there was a need to define the extent to which the vulnerability assessment applied. Multiple approaches were used to define the extent. It is anticipated that this will provide appropriate information to guide discussion about defining the scope of the assessment.

The first mask that was applied limited the assessment to the study area (Figure 3.3), but this approach did not account for the effect of coastal squeeze on the distribution of coastal wetlands now and in the future. The areas included in this assessment likely indicate past distribution of coastal wetlands prior to land cover change and have been included in the assessment on the basis of their existing geomorphic characteristics rather than biotic characteristics. This approach has been used elsewhere to identify changes in coastal wetland distribution following European colonisation in Australia (Rogers *et al.*, 2015). Without significant land use change and restoration of prior wetland areas, this assessment will not provide a reasonable indication of the vulnerability of mangrove and saltmarsh in the region as it does not adequately reflect current distribution or future vegetation distribution.

To account for the influence of current land-use on the distribution of in situ vegetation, the assessment was masked to exclude areas from the assessment where existing buildings and infrastructure (indicated by hatching in Figure 3.4) currently limit the distribution of estuarine vegetation; this was achieved by constraining the assessment to non-built areas. This approach produced maps covering areas that will not support mangrove and saltmarsh both now and in the future without significant management interventions, such as the Sydney Airport precinct, Centennial Park, Sydney Cricket Ground, and golf courses (Bonnie Doon The Lakes, The Australian, Moore Park, The Royal Sydney, Manly and Long Reef) and sporting precincts. Most of these areas are not plausible locations for future mangrove and saltmarsh; however, the mapping does indicate the need to explore options where existing land use may be compatible with the provision of mangrove and salt marsh in the future. This could occur at the margins of sporting fields and golf courses; or may be achieved by conversion of these land uses to natural habitat.

Using the current distribution of estuarine vegetation, the assessment was further constrained (Figure 3.5). This masking approach at Botany Bay (Figure 3.6) illustrates that this approach provided an improved indication of the vulnerability of in situ vegetation to climate change. However, this mask did not adequately account for possible expansion of estuarine vegetation along tributaries, such as Wolli Creek. Landward expansion of coastal wetlands, and mangrove in particular, has been identified as a regional (Saintilan & Hashimoto, 1999) and global trend (Saintilan *et al.*, 2014), that has been linked to multiple climate changes drivers, including sea-level rise.

While future sedimentation within mangrove and saltmarsh will improve the adaptive capacity of coastal wetlands, as indicated in the Inundation Adaptive Capacity (IAC) component of the model, some landward expansion may still occur. To account for landward expansion of coastal wetlands, the 90cm inundation modelling with the current distribution of wetland vegetation was used as a mask (Figure 3.7). The assessment at Botany Bay (Figure 3.8) illustrates that this mask approach may provide the most realistic indication of the vulnerability of mangrove and saltmarsh to climate change. This masking demonstrated the potential capacity for mangrove to migrate landward and longitudinally along estuaries when barriers to the occurrence were limited and accommodation space was available.

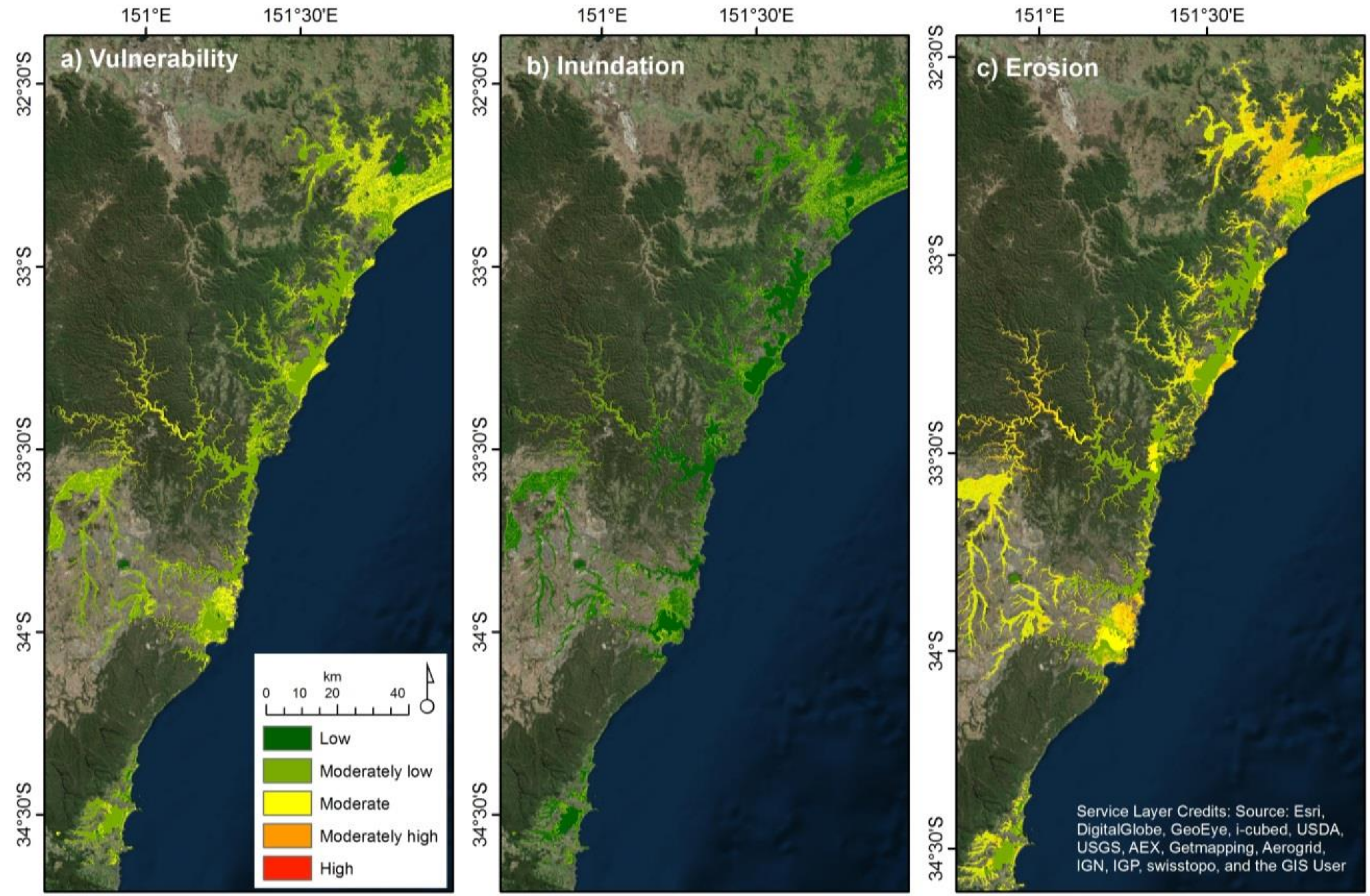

Figure 3.2: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment without mask applied.

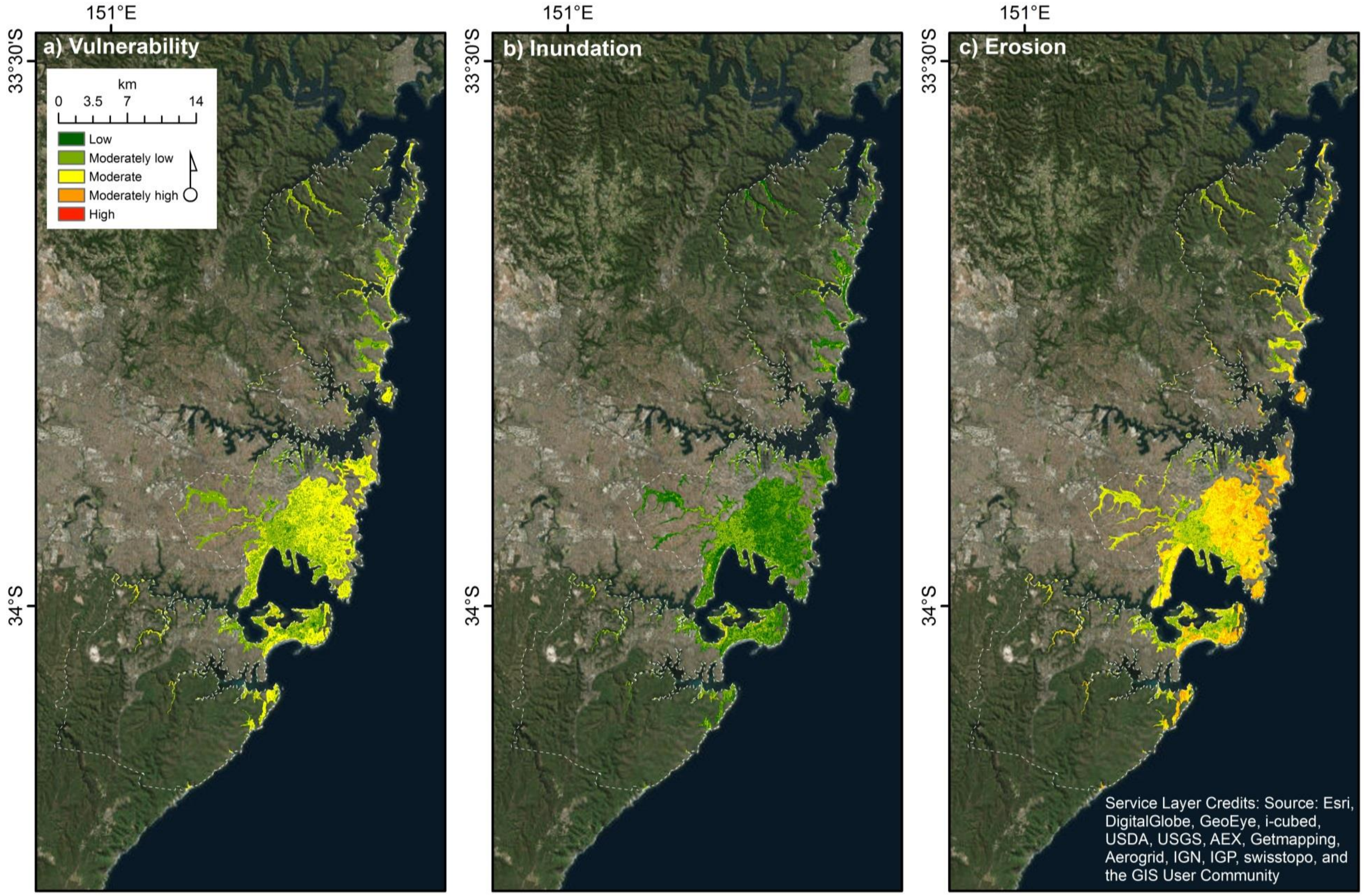

Figure 3.3: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment with study area mask applied.

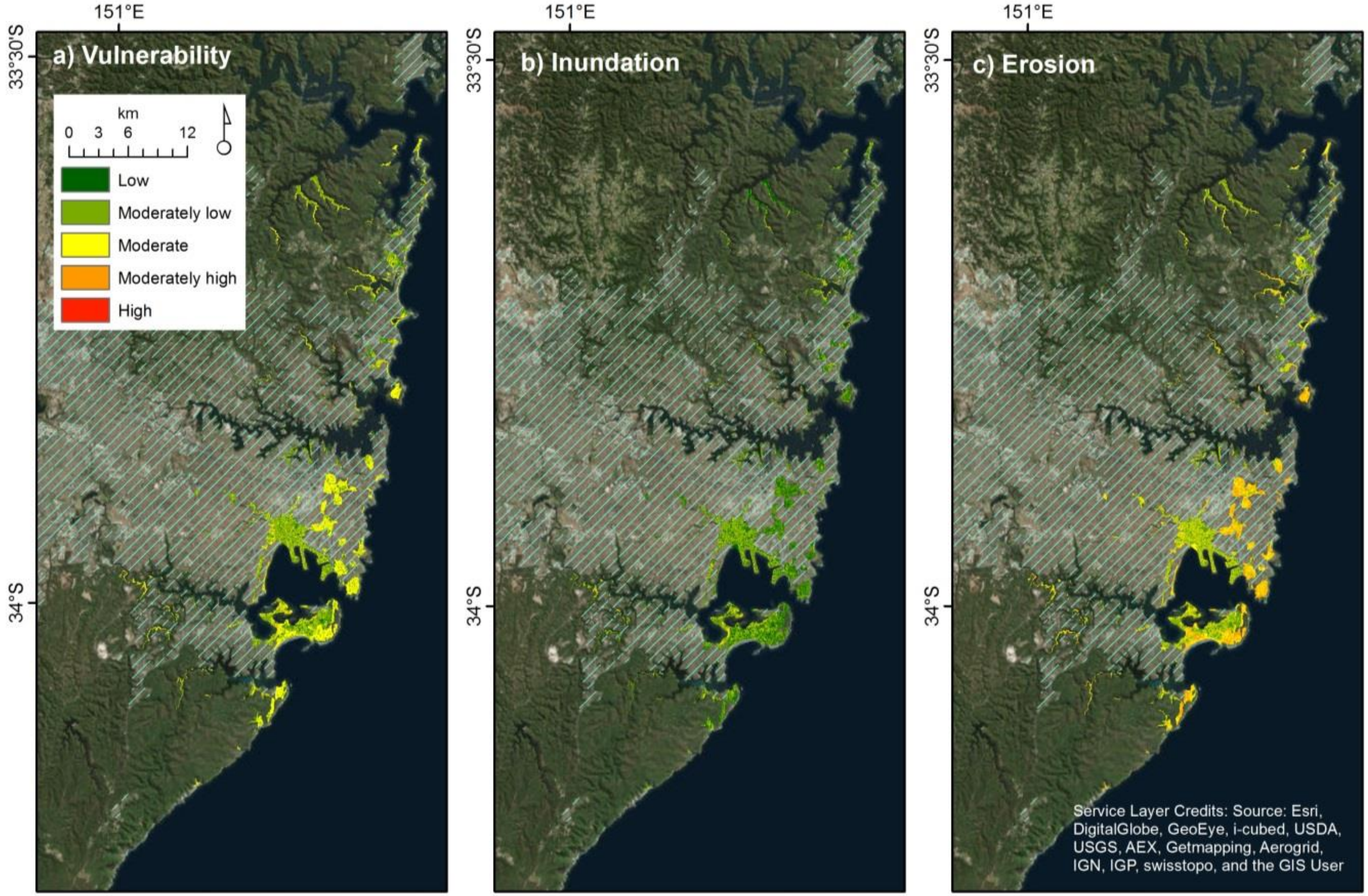

Figure 3.4: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment with study area and built-up area mask applied. Built-up areas shown with blue hatching.

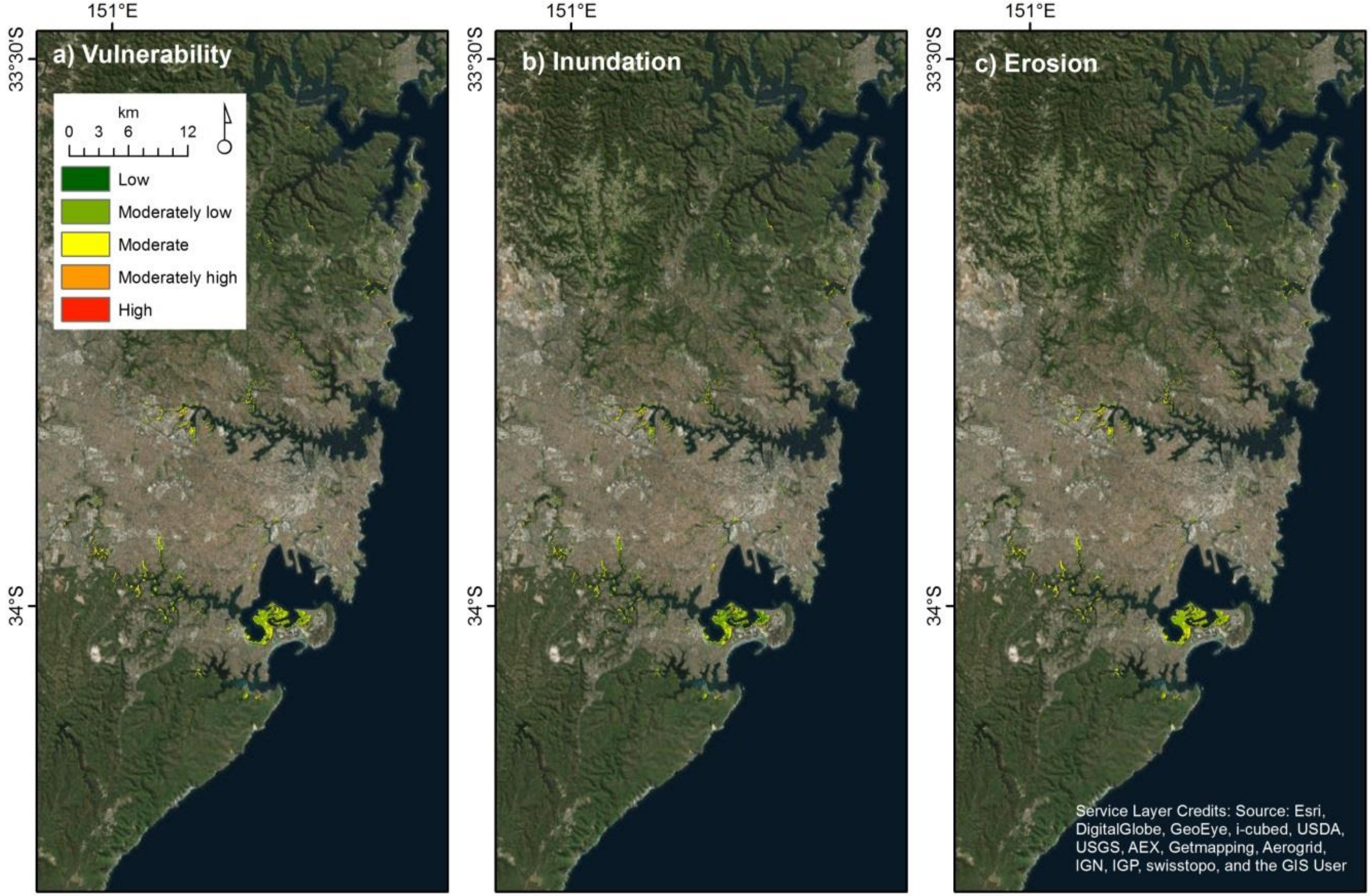

Figure 3.5: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment with estuarine vegetation mask applied.

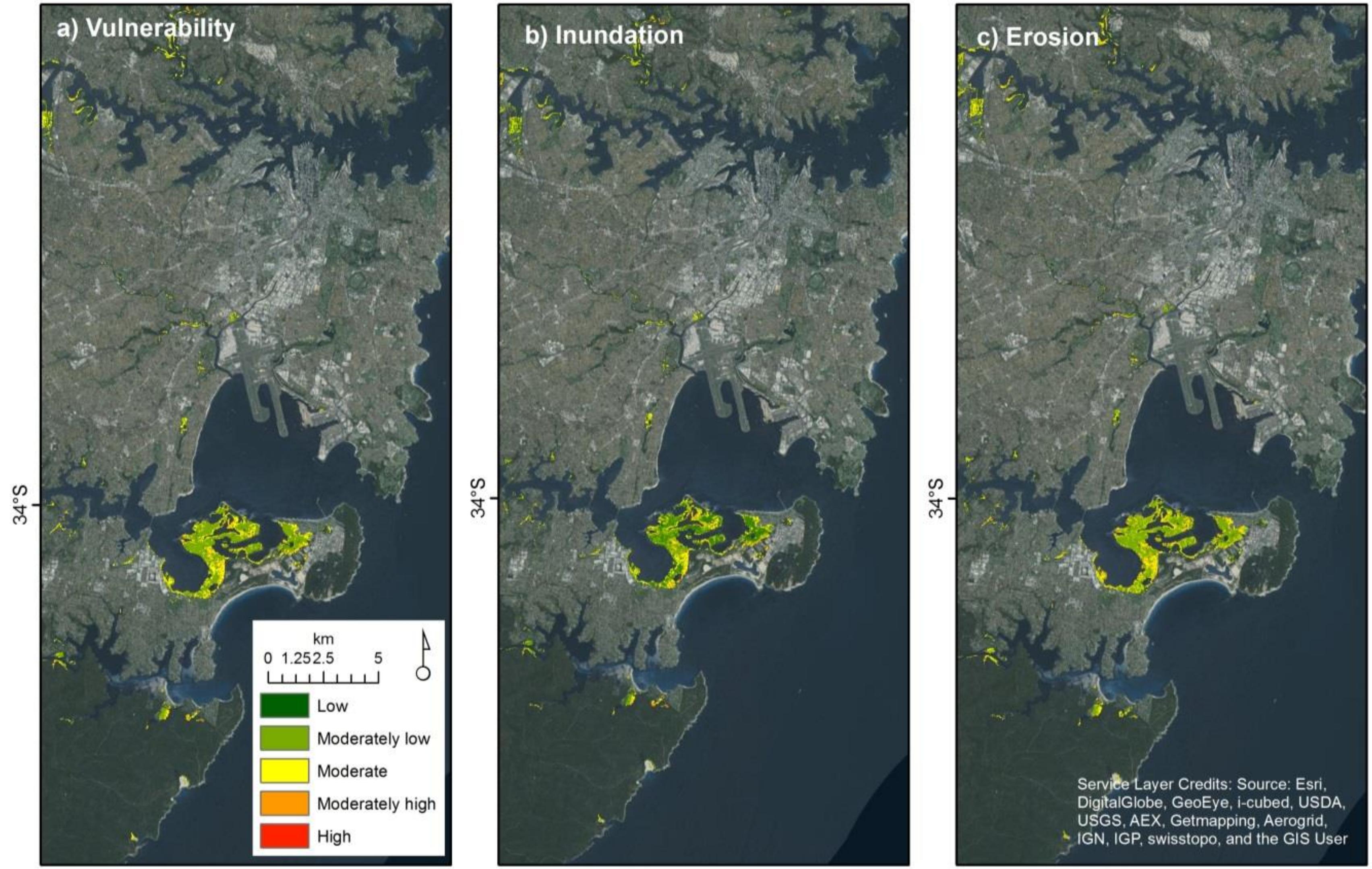

Figure 3.6: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment at Botany Bay with estuarine vegetation mask applied.

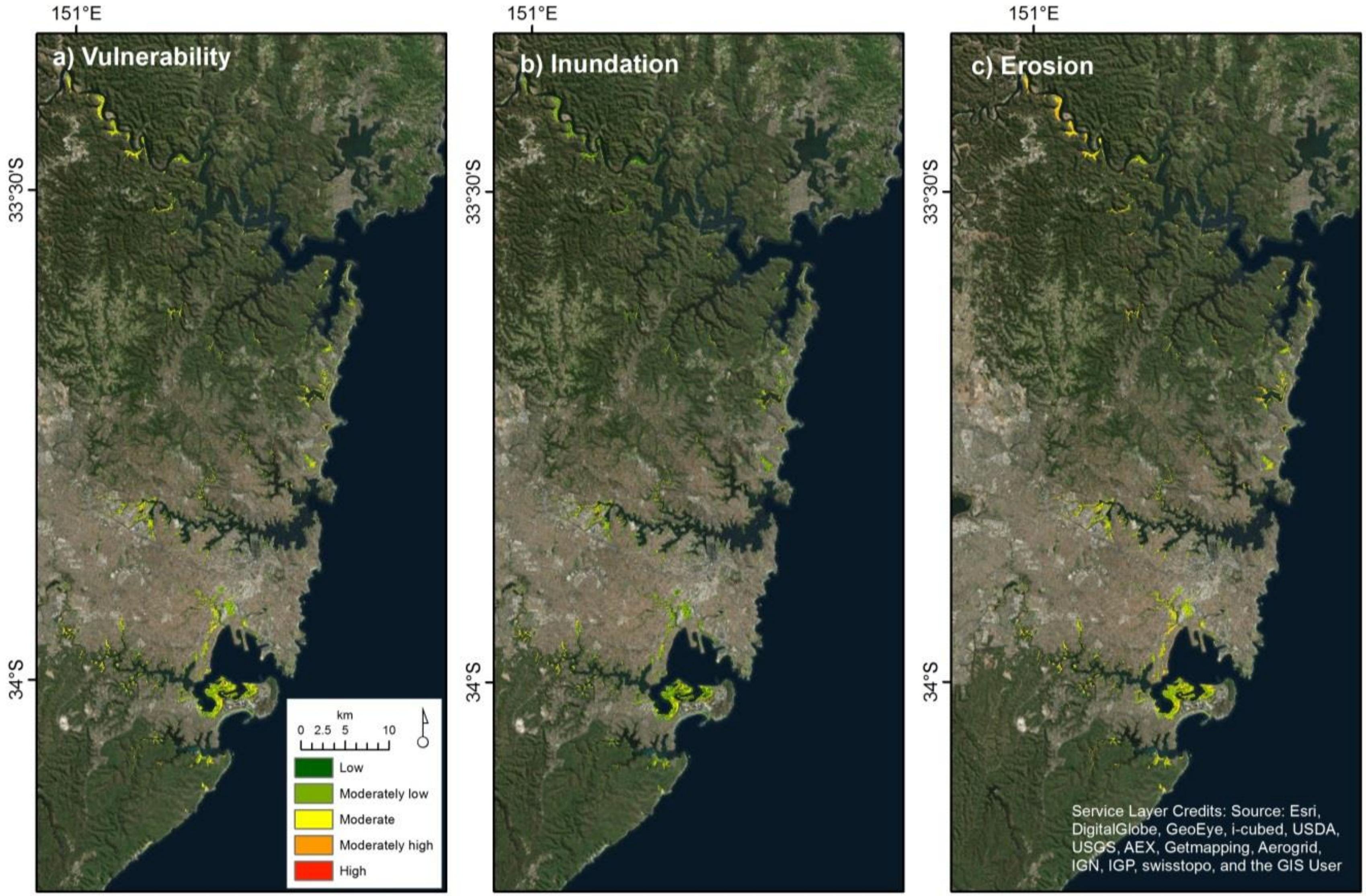

Figure 3.7: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment with 90cm SLR projection and vegetation mask applied.

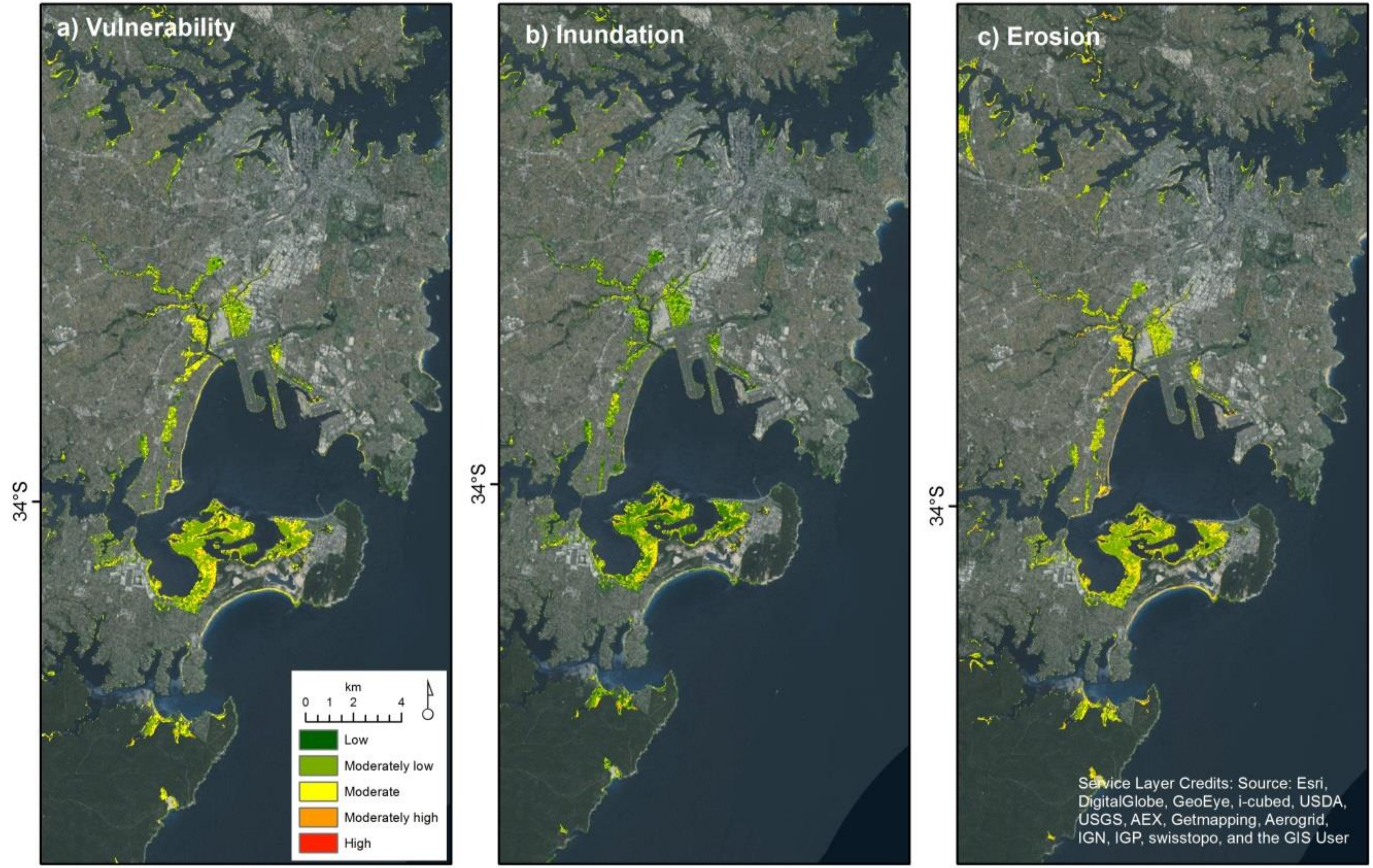

Figure 3.8: **a)** Vulnerability assessment and **b)** inundation and **c)** erosion assessment at Botany Bay with 90cm SLR projection and vegetation mask applied.

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3.3.2 VULNERABLE AREAS

The final two masking approaches were used to provide an indication of the vulnerability of in situ vegetation, and potential future vegetation distributions. To assist with reporting of vulnerability, scores ranging between 0 and 18 were reclassified into five classes ranging from low to high vulnerability; the purposes of reclassification was to provide a relative indication of vulnerability that was qualitative and without bias, but did not provide an absolute or quantitative indication of vulnerability. The vulnerable area of current and future estuarine vegetation (i.e. estuarine vegetation with 90 cm SLR projection) is indicated in Table 3.2. On the basis of this assessment, the proportion of wetland exhibiting moderately high to high vulnerability was less than 13% of existing vegetation area and less than 8% of the existing and potential future vegetation area. The proportion of wetland area assigned moderately high to high vulnerability with the 90cm SLR projection was lower as future wetlands areas that have migrated landward are parameterised to have lower vulnerability, thereby resulting in a larger proportion of the future wetland area having moderate to low vulnerability.

Table 3.2: Area (ha), and proportion of total area (%) with low to high vulnerability. Estimates are based on the current distribution of estuarine vegetation, and possible vegetation distribution under a 90cm sea level rise projection.

Vulnerability Class	***Cell Score***	***Estuarine vegetation***		***Estuarine vegetation with 90cm SLR projection***	
		Area (ha)	Proportion of Area (%)	Area (ha)	Proportion of Area (%)
Low	1-3	52.45	3.29	267.66	6.07
Moderately low	4-7	682.34	42.79	2044.66	46.36
Moderate	8-11	658.91	41.32	1787.75	40.54
Moderately high	12-15	191.61	12.02	300.70	6.82
High	16-18	9.18	0.58	9.35	0.21
	Totals	1594	100	4410	100

When considering the area identified as moderately high to high vulnerability, there is a substantial increase in area of more than 50% when the 90cm sea-level rise projection was incorporated into the assessment (i.e. an increase from 200ha to 310ha). The vulnerability assessment at Botany Bay illustrates that these areas of moderately high to high vulnerability are primarily located at the seaward and landward margins of the existing wetland distribution (Figure 3.6 and Figure 3.8). This is not unexpected as lower elevations exhibit the greatest exposure to sea-level rise and higher elevations exhibit lower adaptive capacity. The interior of wetlands tend to exhibit lower vulnerability to climate change, and will act as an important refuge for wetlands exposed to sea-level rise. Planning to accommodate these refuge areas will be essential.

Conceptual models of the response of mangrove and saltmarsh to sea-level rise typically indicate landward incursions of vegetation zones as the sea rises (Rogers *et al.*, 2016, Vanderzee, 1988). Comparisons of the vulnerability assessments with and without the 90cm sea level rise projection support this hypothesis, with both landward and longitudinal expansion along estuaries evident, particularly along the Wolli Creek tributary (Figures 3.6 and 3.8). On the basis of this response, vulnerable landward margins may change vegetation cover to communities that exhibit greater adaptive capacity (i.e. transition from saltmarsh to mangrove), thereby reducing the vulnerability of landward zones over time. The capacity for mangrove to build elevation at higher rates than saltmarsh through sediment addition and below-ground biomass additions is now well-established in southeastern Australia (Kelleway *et al.*, 2016, Rogers *et al.*, 2006), and it is feasible that a transition of vegetation cover from saltmarsh to mangrove may improve the adaptive capacity.

However, the approach employed in this study does not account for possible changes in the distribution of vegetation communities over time, and only provides an indication of vulnerability based on current vegetation distribution. High resolution modelling, focussed on areas where the vulnerability assessment indicated higher vulnerability, would provide a more robust indication of the vulnerability of coastal wetlands to sea-level rise. Indeed, application of first pass assessments for prioritisation of high resolution modelling is the primary purpose for undertaking a first pass assessment (Rogers & Woodroffe, 2016). High resolution modelling approaches are typically data intensive and require detailed information about the evolution and productivity of in situ vegetation. Models have been prepared for southeastern Australian wetlands using high resolution LiDAR data and measurements of surface elevation change over time (Oliver *et al.*, 2012, Rogers *et al.*, 2012, Rogers *et al.*, 2013a). This approach is beyond the scope of this project, but may be applied in future assessments.

Visual inspection of vulnerability maps indicated that highly vulnerable areas were widely distributed throughout the study area. The distribution of highly vulnerable areas corresponds to the broad distribution of wetlands along the margins of estuaries and the geomorphic setting in which the wetlands have formed. With the exception of the large wetlands at Botany Bay, which exhibit a high proportion of cells with lower vulnerability, all wetland areas have formed within drowned river valleys where wetland development is limited to areas where the bedrock depth is low and hydrodynamic energy facilitates sediment deposition. The high proportion of cells with lower vulnerability in Botany Bay reflects the enhanced capacity for sediment deposition in this embayment.

3.3.3 RECOMMENDATIONS

Application of masks for this assessment had a significant impact on the area of focus and the indication of vulnerability. It is recommended that the latter two masking approaches—by both current estuarine vegetation extent and by 90cm sea-level rise—be used to indicate wetland vulnerability, with the former indicating the vulnerability of existing vegetation and the latter providing an indication of future vulnerability under a projected sea-level rise scenario.

The capacity to undertake high resolution modelling of wetland vulnerability is dependent upon high resolution and high accuracy digital elevation data, typically derived from LiDAR sensors, and accurate measurements of wetland surface elevation trajectories over time. Wetland surface elevation trajectories have been measured at Homebush Bay for 16 years using a network of surface elevation tables (Rogers *et al.* in prep, Rogers *et al.*, 2005) and now provide a robust indication of the capacity of wetland surfaces to increase elevation over time. This data could feasibly be incorporated into a high resolution model of wetland response to sea-level rise to provide a more robust indication of wetland vulnerability. Preliminary modelling by Bowie (2015) at Homebush Bay, using site specific wetland surface elevation trajectories and the readily available Sea Level Affecting Marshes Model (SLAMM) identified issues with this modelling tool set. We therefore recommend that readily available modelling approaches, such as SLAMM, be used with caution; alternatively, individual site-based models that account for unique geomorphic characteristics are likely to provide the best indication of wetland vulnerability to sea-level rise. Stage II of this project will apply surface elevation trajectories from Homebush Bay to a LiDAR-derived digital elevation model of Cooks River as a demonstration of the sea level affecting marshes modelling environment and its application towards assessing adaptation options within a highly developed catchment.

Chapter Four: Projecting sea-level rise threats to mangrove and saltmarsh: A pilot study at Wolli Creek using SLAMM

4.1 Introduction

In recognition of the value of urban ecosystems and the high relative carbon-sequestration and storage provided by Blue Carbon ecosystems, as well as the unique geomorphic setting in which mangrove and saltmarsh have evolved within the Sydney Region, the Sydney Coastal Councils Group, through the federally funded Sydney's Salty Communities Program, are supporting a mangrove and saltmarsh threat analysis. This project will provide a review of the pressures and impacts on mangrove and saltmarsh, a broad-scale assessment of their vulnerability to sea-level rise, and an overview of adaptation options, including strategic planning that will improve or maintain the ecosystems services provided by mangrove and saltmarsh in the Sydney Region in the 21st century. This chapter provides a high resolution assessment of mangrove and saltmarsh vulnerability to sea level rise derived using the Sea Level Affecting Marshes Model (SLAMM).

SLAMM is one of the most widely used mangrove and saltmarsh-focused spatial landscape models and was originally developed for North American coastal wetlands. The model, created in 1986 and revised numerous times, is a complex, non-hydrodynamic model that simulates six primary processes affecting the survival of coastal wetlands with long term sea-level rise. These processes include inundation, erosion, overwash, saturation, salinity and accretion, of which inundation and accretion are most frequently implemented. Though used primarily to simulate the changes in wetland boundaries and shoreline modifications with increasing sea level (Clough *et al.*, 2015, Galbraith *et al.*, 2002, Glick & Clough, 2006, Linhoss *et al.*, 2015), SLAMM has been used to simulate the effect of sea-level rise on the natural services and regulatory processes offered by coastal wetlands, such as species habitats and carbon sequestration and denitrification (Craft *et al.*, 2009a, Glick *et al.*, 2007, Naughton, 2007). In addition, SLAMM has been coupled with ecological models to examine the potential effect of sea-level rise on wetland dependent and threatened species (O'Mara, 2012, Traill *et al.*, 2011).

The ability of SLAMM to adequately describe the response of wetlands to sea-level rise has been questioned previously (Craft *et al.*, 2009a, Craft *et al.*, 2009b, Kirwan & Temmerman, 2009, Kirwan & Guntenspergen, 2009); however, due to ease of access, it is increasingly used for landscape-scale models and planning purposes. Fortunately, with further funding, advancements in computer science and renewed model development, successive versions of SLAMM, versions 6.2 and 6.7, have addressed some concerns, particularly incorporation of an accretion feedback component. Applying the SLAMM to coastal wetlands, such as those of southeastern Australia, where the geomorphology and ecology differs from that of North America, throws into question the efficacy and reliability of the final output. SLAMM has been rarely applied to wetlands in Australia, with studies currently limited to northeastern NSW (Akumu *et al.*, 2011) and southeastern Queensland (Traill *et al.*, 2010, Traill *et al.*, 2011). Recently, Mogensen (2016) attempted to verify and validate SLAMM through comparison with other modelling approaches at Minnamurra River, NSW, finding that SLAMM offered the greatest predictive power over decadal timescales, but indicating that flaws in the model regarding vegetation succession, processes influencing wetland surface elevation change and simulation of tidal water levels decrease the predictive ability and increase model uncertainty. Further model refinement is essential for application of SLAMM in the Australian context – model refinement has been incorporated into this study; however, there remains significant scope to improve high resolution landscape scale models of mangrove and saltmarsh vulnerability in Australia and the Sydney Coastal Councils Group region.

Application of SLAMM in this chapter focuses on mangrove and saltmarsh within the Cooks River catchment and Wolli Creek catchment area, a small mature catchment within the Sydney Coastal Councils Group region that supports mangrove and saltmarsh and which has a highly developed catchment with multiple threats and stressors. Application of SLAMM to mangrove and saltmarsh at Wolli Creek has provided an indication of possible changes to mangrove and saltmarsh within the Sydney Coastal Councils Group region and have been used to explore possible adaptation and planning actions that will improve ecosystem services provided by mangrove and saltmarsh in the 21st century. This chapter provides a higher resolution extension of the first pass assessment of mangrove and saltmarsh vulnerability provided in Chapter 3 and includes the provision of high resolution vulnerability assessment maps. Detailed analysis of impacts and threats to mangrove and saltmarsh in the Sydney region, and discussion of possible adaptation options, including strategic management and land use planning are provided in Chapter 5.

4.2 Method

4.2.1 CATCHMENT OVERVIEW

The Cooks River catchment (Figure 4.1) is located to the south-west of Sydney and covers an area of approximately 102 km^2, incorporating 13 local government areas. The Cooks River begins as a small watercourse near Graf Park, Bankstown, before flowing in an easterly direction and discharging into Botany Bay, south of Sydney Airport.

The two main tributaries of Cooks River are Alexander Canal and Wolli Creek. Bardwell Creek is a tributary of Wolli Creek. Other tributaries of Cooks River include Muddy Creek, Cup and Saucer Creek and Cox's Creek. Many unnamed stormwater channels also discharge into Cooks River. The majority of the banks of the Cooks River are concrete or steel sheet lined. The natural bed of certain reaches of the Cooks River is also concrete-lined.

The catchment area has undergone significant anthropogenic development since European settlement, with the Cooks River and its tributaries being altered and degraded by a wide variety of activities (Ecological, 2010). Extensive urban and industrial areas now dominate the Cooks River catchment. Despite the high percentage of built-up areas present within the catchment, the Cooks River is surrounded by parkland, open spaces and remnant bushland along much of its length (Ecological, 2010).

Vegetation within the Cooks River catchment is concentrated around the Cooks River and its tributaries. Mangroves fringe much of Cooks River and Wolli Creek, whilst concrete or iron walls stabilise the banks of the rivers where mangroves have been removed. *Avicennia marina,* or the grey mangrove, dominates the mangrove species along the Cooks River. Few saltmarsh communities are found within the catchment. The largest areas of saltmarsh are situated in the intertidal zone of Wolli Creek whilst patches of *Sarcocornia quinqueflora* are found along the banks of the Cooks River. Historically, the catchment contained extensive mangrove and saltmarsh areas (Clouston, 1997). The filling and re-routing of the river as well as the development of the Sydney airport resulted in a considerable reduction of wetland areas to their present distribution with the Cooks River catchment.

4.2.2 MODEL OVERVIEW

A non-hydrodynamic vegetation model was developed for the Cooks River catchment using the SLAMM. SLAMM is a complex, non-hydrodynamic model that attempts to simulate the response of wetlands to sea-level rise (Clough *et al.*, 2012). The model has previously been applied primarily to natural and urban wetland systems of North America. Recently, however, the model has been used to examine the effect of sea level on Australian wetlands (Akumu *et al.*, 2011, Mogensen, 2016, O'Mara, 2012, Traill *et al.*, 2011).

The model simulates inundation and conversion of vegetation over time based upon six primary processes; inundation, erosion, overwash, soil saturation, accretion and salinity. Of these, certain processes are optionally incorporated in simulations, such as overwash and soil saturation. The basic conceptual model of SLAMM is premised on the assumption that wetland categories only inhabit a certain elevation range that is a function of tidal range or salinity. The model can simulate changes in 25 different land-cover categories under rising sea levels. Wetland categories are based on the National Wetland Inventory (NWI) prepared for the United States of America (Cowardin *et al.*, 1979). Details of how SLAMM was utilised to develop a high-resolution model of the Cooks River catchment are presented below.

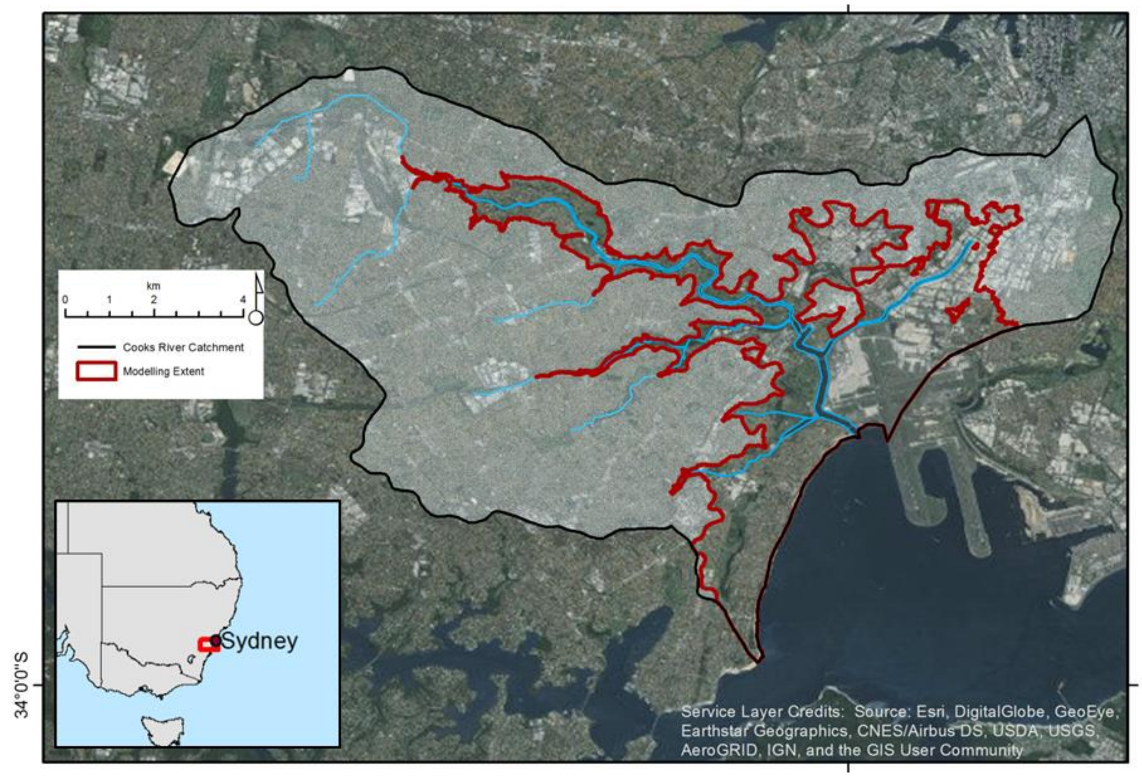

Figure 4.1: Cooks River catchment, and area of interest for model application.

4.2.3 MODEL SET-UP AND PARAMETERS

Model extent

SLAMM version 6.7 (Clough *et al.*, 2016) was utilised in this study to model the effect of sea-level rise on the Cooks River catchment. To maximise the efficiency of the model whilst maintaining high resolution modelling of relevant regions within the Cooks River catchment, all areas less than or equal to 10 m (AHD) were delineated from a digital elevation model and utilised as the model extent. The 10m elevation boundary was selected as an area beyond which it was almost certain any conservative rise in sea level would not have an effect. The final extent of the model is illustrated in Figure 4.1.

Elevation

Elevation data is a fundamental parameter within SLAMM. Topographic data in the form of LiDAR point files and derived digital elevation models (DEM) for the area were acquired under licence from the University of Wollongong. Tiled DEMs were acquired at a 1 metre and 5 metre resolution. Due to technical difficulties in utilising the 1 metre DEM within SLAMM, the 5 metre DEM was evaluated to provide the greatest efficiency at the highest resolution for this study. The LiDAR survey was flown in 2013 and has a reported vertical RMSE of 1 m. The individual DEM tiles were mosaicked to develop a 5 m DEM that covered the entire Cooks River catchment. The elevation model was then clipped to the model extent and converted to text, as required for SLAMM.

Vegetation and land use

The model can simulate changes in 25 different land-cover categories under rising sea levels. Wetland categories are based on the National Wetland Inventory (NWI), originally prepared for applications in the United States of America (Cowardin *et al.*, 1979). The vegetation layer used in this study as input into SLAMM was developed from the digital mapping of the *Native Vegetation Communities of the Sydney Metropolitan Area* (OEH, 2013b). The Australian vegetation communities from this layer were converted to NWI categories as per Table 4.1, and based on their corresponding characteristics to the NWI categories or the function of the NWI category within SLAMM. Areas categorised as Undifferentiated Regenerating Scrubs (UD_Reg_Shr), Urban Exotic/Native (Urban_E/N), Water (Water), Artificial Wetland (Art_WL) and Exotic Species (Weed_Ex) within the acquired OEH vegetation layer were individually examined and assigned a SLAMM category based upon interpretation of 2013 ADS40 aerial imagery of the area, elevation and position within the tidal frame.

In addition to vegetation classes, developed and undeveloped areas were delineated in preparation for SLAMM. Developed areas were defined as built-up areas, including urban, industrial and commercial zones. Developed areas were delineated from the *Standard Instrument Local Environmental Plan - Land Zoning (LZN)* (DPE, 2013) and *Inner Sydney Regional Cadastral Survey* 2012 acquired from the Sydney Coastal Councils Group. Developed areas were selected based upon attributes from the *Land Zoning* mapping. Areas not included in the selection were considered undeveloped areas, that is, natural areas that contain no built-up zones. To ensure accuracy of mapped undeveloped areas, each polygon assigned this land use category was examined. Polygons that contained greater than 20% developed areas were added to the developed areas selection. Where *Land Zoning* did not cover the Cooks River Catchment, the cadastral information was utilised to derive developed and undeveloped areas following the same method. The cadastral and zoning-derived developed and undeveloped layers were combined to create one layer of the two land uses across the entire catchment.

Developed and undeveloped zones were combined with the vegetation data, resulting in a vegetation and land use spatial layer (map) for the Cooks River catchment. To ensure the merging of the two layers, vegetation and land use, occurred correctly, the data was quality assessed and the geometry of the polygons checked for self-intersections. The final, quality-assessed layer was clipped to the modelling extent before being converted to a text file, as required for input into SLAMM.

Vegetation elevation ranges

Within SLAMM, each vegetation class is assigned a certain elevation range within which it will exist. The lower boundary of the vegetation class elevation range determines the conversion of one class to another during simulations. For this study, elevation ranges for each vegetation and land use category were assigned with respect to the tidal range. See below for further information on the development of tidal range values. Elevation ranges were analysed per subsite (see below for development of subsites), with statistics of elevation being assigned to each class within each subsite. In some cases, the minimum elevation value of a class from the statistical analysis was below the feasible elevation at which the vegetation class existed. In such case, a value two standard deviations below the mean was instead adopted. Final vegetation elevation ranges are presented in Table 4.2.

Table 4.1: Vegetation map unit codes, community description and dominant species (OEH, 2013b) corresponding to SLAMM categories and NWI classes used within modelling.

Map Code	Community description	Dominant species	SLAMM ID	SLAMM Category
S_DSF01	Dry Schlerophyll Forests; shrubby open forest	*M.decora/M.nodosascattered Eucalypts*	2	Undeveloped Land
S_DSF02	Dry Schlerophyll Forests	*E.fibrosa/E.molucanna/M.decora/E.longifolia*	2	Undeveloped Land
S_DSF04	Dry Schlerophyll Forests	*E.punctata/A.costata/E.piperita/ C.gummifera*	2	Undeveloped Land
S_DSF06	Dry Schlerophyll Forests	*A.costata/E.piperita/C.gummifera/S.glomulifera/E.resinifera*	7	Transitional Marsh
S_DSF09	Dry Schlerophyll Forests	*A.costata/E.piperita/C.gummifera/S.glomulifera/E.resinifera*	2	Undeveloped Land
S_DSF15	Dry Schlerophyll Forests	*E.racemosa(haemastoma)/C.gummifera/A.costata*	2	Undeveloped Land
S_DSF17	Dry Schlerophyll Forests	*E.pilularis/E.punctata/A.costataE.racemosa*	2	Undeveloped Land
S_FoW01	Forested Wetlands; floodplains, flats and estuaries	*E.botryoides*	7	Transitional Marsh
S_FoW02	Forested Wetlands; floodplains, flats and estuaries	*E.robusta*	7	Transitional Marsh
S_FoW03	Forested Wetlands; floodplains, flats and estuaries	*C.glauca*	7	Transitional Marsh
S_FoW05	Forested Wetlands; floodplains, flats and estuaries	*M.stypheliodes/M.linariifolia*	7	Transitional Marsh
S_FoW06	Forested Wetlands; floodplains, flats and estuaries	*E.tereticornis/E.baueriana/A.floribunda*	7	Transitional Marsh
S_FoW08	Forested Wetlands; floodplains, flats and estuaries	*C.glauca*	7	Transitional Marsh
S_FoW12	Forested Wetlands; floodplains, flats and estuaries	*C.glauca/M.ericifolia*	7	Transitional Marsh
S_FrW03	Freshwater Wetlands; coastal freshwater lagoon	*T.orientalis/freshwater sedges*	23	Tidal Swamp
S_FrW06	Freshwater Wetlands; coastal freshwater lagoon	*P.australis/B.juncea*	23	Tidal Swamp
S_GL01	Grasslands	*S.sericea/C.glaucescens*	26	Backshore
S_GW03	Grassy Woodlands	*E.tereticornis/E.molucannaE.crebra/E.eugenioides*	2	Undeveloped Land
S_HL03	Heathlands	*L.laviegatum/B.integrifolia/Exotics/urban scrub*	26	Backshore
S_HL05	Heathlands	*B.integrifolia/A.smithii/L.laevigatum/G.ferdinandiiC.anarchardiodes*	2	Undeveloped Land
S_HL08	Heathlands	*B.ericifolia/Kunzea spp/A.distyla*	2	Undeveloped Land
S_RF02	Rainforest	*C.apetalum/T.laurina/C.serratifolia*	7	Transitional Marsh
S_SW01	Mangrove	Mangroves	8	Regularly Flooded Marsh
S_SW02	Saltmarsh	*S.repens/S.quinqueflora/S.virginicusJ.krausii*	20	Irreregularly Flooded Marsh
S_SW03	Seagrass	Seagrass (DPI)	n/a	n/a
S_WSF02	Wet Schlerophyll Forest	*A.costata/E.piperita/C.gummifera/S.glomulifera/E.resinifera*	7	Transitional Marsh
S_WSF06	Wet Schlerophyll Forest	*E.resinifera/S.glomulifera/C.gummifera*	2	Undeveloped Land
S_WSF09	Wet Schlerophyll Forest	*E.pilularis/S.glomuliferaA.costata/E.resinifera*	2	Undeveloped Land
UD_Reg_Shr		Undifferentiated Regenerating Scrubs	2	Undeveloped Land
Urban_E/N		Urban Exotic/Native	2	Undeveloped Land
Water		Water	15-17	Inland Open Water/ Estuarine Open Water
Weed_E		Exotic Species >90%cover	2	Undeveloped Dry Land

Table 4.2: Elevation boundaries for the major vegetation and land use categories modelled in SLAMM. HTU is equivalent to half the tidal range. As the tidal range varied throughout the catchment (see below), the value of the HTU changed based on the tidal ranges within a subsite.

Vegetation Type	**Min (HTU)**	**Max (HTU)**
Developed Land	1.47	3.39
Undeveloped Land	1.05	3.39
Casuarina	1.00	2.67
Saltmarsh	0.10	1.33
Reedlands	0.56	2.11
Mangrove	0.00	1.20
Tidal Flat	-1.00	0.00

Accretion and surface elevation change

In order to account for feedback processes within wetlands that can cause surface elevations to increase or decrease, SLAMM incorporates an accretion parameter. Accretion can be modelled as a function of elevation. While previous studies have examined the sediment type and geochemistry of the Cooks River and its tributaries, there is a paucity of data regarding the spatial distribution of short or long-term sedimentation rates. In addition, especially within intertidal areas, the amount of sedimentation often does not equate to the amount of surface elevation change, defined as the amount by which the surface elevation increases or decreases in height with respect to a local datum or previous surface measurement. Amongst other factors, surface elevation change takes into account soil erosion and compaction, where accretion rates account for positive gains in surface elevation only. In order to incorporate these factors in SLAMM in some manner, it was evaluated that the surface elevation change value was required.

Surface elevation change in wetlands can be measured using a Surface Elevation Table (SET). A network of SETs has been established along the south-east coast of Australia (Rogers *et al.*, 2006); however, the Cooks River is not included in this network. To account for the lack of data for wetlands along Cooks River and its tributaries, SET derived rates of surface elevation change were drawn from a site of similar characteristics, Homebush Bay (Rogers *et al.*, 2005). Rates of surface elevation change for this site were sourced from Bowie (2015b).

Rates of surface elevation change are known to vary spatially throughout a wetland (Temmerman *et al.*, 2003). SLAMM provides a variety of different methods in which the accretion parameter can be assigned. A single accretion rate can be assigned per vegetation type, causing the entire vegetation type to increase by the given amount at each subsite. Alternatively, the accretion rate can be modelled and accretion rates assigned based upon elevation. Whilst uncertainties and compounding of errors may be introduced by utilising the accretion model, it does allow for spatial variability of accretion rates to be incorporated in the final simulations. For this study, modelling of the accretion parameter prior to simulations was considered to best reflect the processes acting within the wetlands of the Cooks River catchment. Rates of surface elevation change were used in place of accretion rates for the reasons detailed above. Based upon the relationship between elevation and rates of surface elevation change observed at Homebush Bay, rates of sedimentation change were modelled to decrease with increasing elevation. The maximum and minimum rates of surface elevation change for mangrove and saltmarsh vegetation communities were set such that the minimum rate for mangrove graded into the maximum rate of saltmarsh surface elevation change. Reedland (areas containing a large proportion of *Phragmites australis*) rate of surface elevation change was deterministically modelled. The rate of surface elevation change for this vegetation type was set as the hypothetical value of 0.105 mm/yr.

Rates of surface elevation change did not vary between subsites; this is an assumption that requires validation (see below for further detail on subsites). Final rates of surface elevation change for different vegetation zones are presented by subsite in Table 4.3.

Rates of surface elevation change (mm/yr)				
Mangrove		Saltmarsh		Reedlands
Maximum	Minimum	Maximum	Minimum	Constant
2.63	0.21	0.21	0.1	0.105

Table 4.3: Minimum and maximum rates of surface elevation change modelled to occur within mangrove and saltmarsh throughout Cooks River sub-sites.

Historic sea-level rise

The historic sea-level rise is utilised in SLAMM to simplistically determine the magnitude of sea-level rise between 1990 and the initial year being modelled for a particular site, and essentially assumes a linear trend in sea-level rise for the given period. Though this method of calculating sea-level rise is understood to introduce numerous uncertainties, the most accurate value for the Cooks River catchment was calculated using the tidal data recorded at Botany Bay and within the Cooks River by the Bureau of Meteorology and MHL (2012), respectively. Given the use of the parameter in SLAMM, tidal data that spanned the time period 1990-2013 were analysed and a mean sea-level rise determined for Botany Bay and Cooks River. The two values were relatively similar when compared against each other. The final value of 2.8 mm/yr was selected from Botany Bay as it was considered to better represent historic sea-level rise, the gauge's position not being affected by river characteristics. This value was also consistent with the global mean sea-level rise for the period 1993-2010 published by the IPCC (2013b).

Tidal range

Tidal range in SLAMM determines the extent of inundation at a given simulated time step and may be used to define the elevation boundaries of each vegetation type. Elevation ranges of vegetation can be set with respect to the tidal range, where the unit of measurement for elevation becomes the half tide unit (half of the tidal range set for a particular site). Given the function of tidal range in SLAMM, this parameter could be considered fundamental to the modelling process. SLAMM, however, will only assign one, discrete tidal range value per subsite. As the tidal plane is a continuous surface along an estuary, the parameterisation of tidal range within SLAMM does not sufficiently capture the tidal effects throughout the estuary. In an attempt to ameliorate the effects of discrete parameterisation of tidal range in SLAMM, a continuous tidal range surface was created for the model extent of this study from which discrete values were obtained on a subsite basis along the river.

The continuous tidal range surface was developed following OEH (2014). Tidal range modelled and utilised in parameterising SLAMM for this study was taken to be the greatest difference in tidal water levels, that is the difference between High High Water Solstice Springs (HHWSS) and Indian Spring Low Water (ISLW). The yearly-average tidal range data were obtained from MHL (2012) and Botany Bay BOM tide gauge. The Manly Hydraulics Laboratory (MHL) collects water level data at three tide gauges along the Cooks River located at Canterbury Road, Illawarra Road Bridge and Tempe Bridge. Data for the Tempe Bridge and Canterbury Road tide gauges reported in MHL (2012) span a 15 year period, 1995-2010, whilst water levels at the Illawarra Road Bridge are only recorded over a 9 year period, 2001-2010. The yearly-averaged tidal range reported for each of the tide gauges was selected for use in modelling of the Cooks River tidal plane. It is noted that the tidal range calculated for the Illawarra Road Bridge tide gauge may contain and introduce error into the modelling due to the short duration of its water level records. However, given that the gauges tidal data record could not be replaced or omitted from the model without further increasing potential errors, the yearly-averaged tidal range for the gauge was evaluated to be sufficient for the purposes of this study.

Tidal range data for Botany Bay was obtained from the BOM tide gauge. The value obtained was compared with the tidal ranges modelled in OEH (2014) and the One-Dimensional Transport with Inflow and Storage (OTIS)(Egbert & Erofeeva, 2010) model for NSW.

Incorporating the four, yearly-averaged tidal range values, a continuous tidal range surface was developed for the Cooks River Estuary following OEH (2014). The output tidal range surface was reclassified into ten classes using the geometrical interval classification method. Mean values of each subsite were calculated and assigned to the respective class. Final subsites were delineated following the method outlined below. The mean tidal range values were again calculated for each subsite, the final tidal range values being presented in Table 4.4.

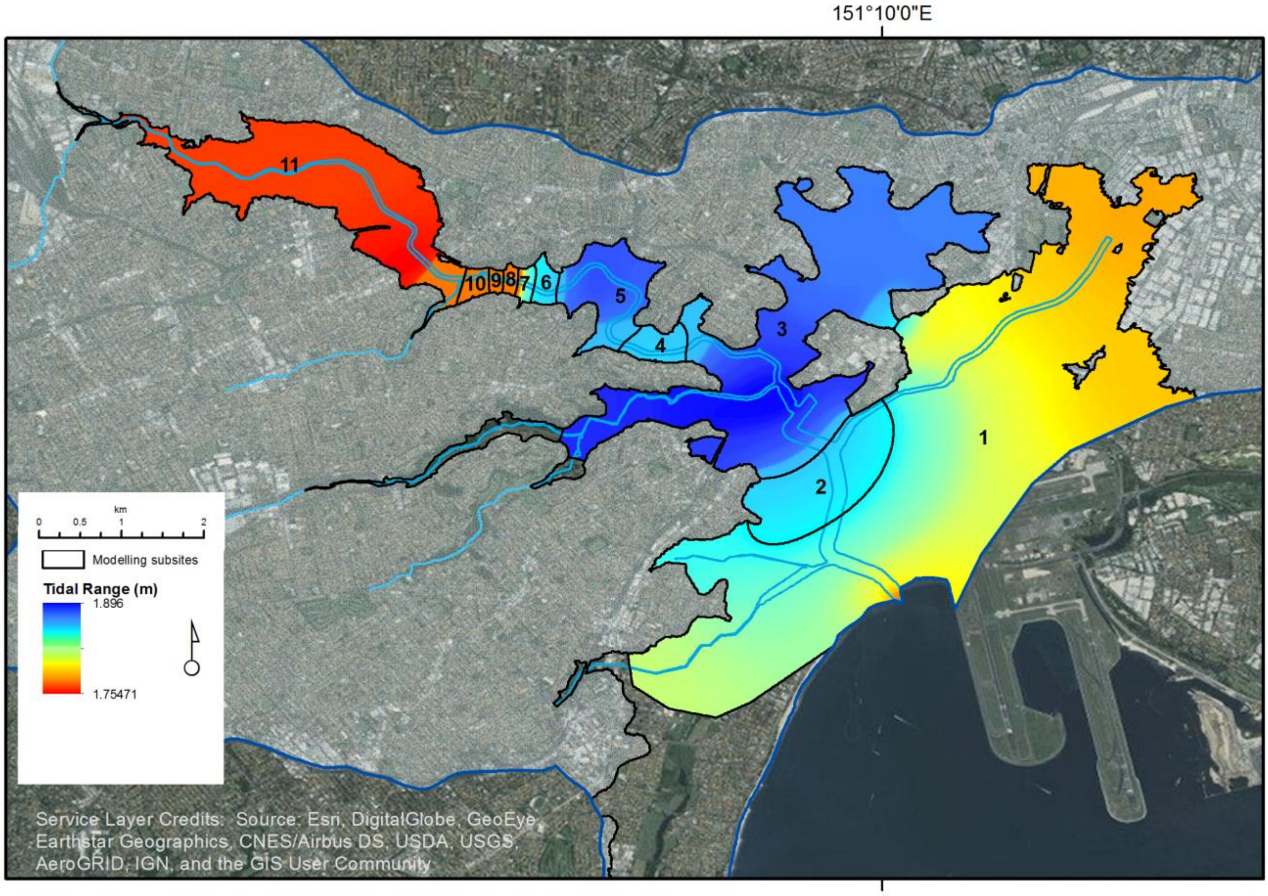

Figure 4.2: Modelled tidal range along the Cooks River, following the methods of OEH (2013b). To allow for tidal changes to be adequately accounted for in SLAMM this tidal range model was used to delineate subsites along the Cooks River, as shown by numbered polygons 1-11.

Table 4.4: Mean tidal range and mean sea level values used to parameterise SLAMM.

Subsite	Tidal Range (m)	Mean Sea Level (m)	Salt Elevation (m)
Global	1.8	0.04	1.2
1	1.844534	0.041746	1.1
2	1.863841	0.046928	1.1
3	1.885837	0.064463	1.15
4	1.873757	0.069208	1.16
5	1.88372	0.063271	1.2
6	1.864989	0.066651	1.2
7	1.846826	0.070949	1.2
8	1.826175	0.075943	1.2
9	1.804435	0.081248	1.2
10	1.782631	0.086591	1.2
11	1.757431	0.092892	1.2

Modelling Subsites

Model subsites were delineated in an attempt to develop a more accurate model of the effect of sea-level rise on the study area. Given parameters, such as tidal range, change throughout the model extent; it was considered best practice to assign parameter values at a subsite level rather than catchment level.

Modelling subsites were delineated based primarily upon tidal range and mean sea level (Table 4.4) of sites along the Cooks River and its tributaries. A mean sea level was modelled in the same manner as the tidal range, following OEH (2014). Dividing the modelled tidal range surface into unique 2cm intervals resulted in the development of 11 subsites within the tidally influenced estuary. Statistics generated on the modelled MSL values for each subsite were then examined to ensure that 95% of the MSL values for an individual subsite did not exceed a 0.5cm variation. Figure 4.2 displays the final delineation of subsites used for modelling the effect of sea level rise on the catchment.

4.2.4 MODEL CALIBRATION

Calibration of the model in SLAMM was conducted using a process of optimisation, whereby parameters were iteratively altered to optimise the goodness of fit between modelled results for the initial year of simulation and the vegetation map of the same year. Calibration of parameters focussed primarily on adjustment of elevation ranges and definition of salt elevation, which is the elevation above which inundation by saline waters occur very rarely. The rates of surface elevation change were held constant. A 10% variance of modelled vegetation at 2013 from the calibration dataset (the initial 2013 vegetation layer previously developed) was considered to be within acceptable limits for further modelling. In certain instances, where, under visual analysis, the SLAMM output appeared to provide refinement to the initial wetland vegetation layers when compared to aerial photography, a variance slightly greater than 10% was accepted. The calibration process was repeated iteratively until the interplay between tidal ranges, elevations and coastal habitat maps at time zero were deemed satisfactory.

4.2.5 MODELLING SCENARIOS

SLAMM was used in this study to estimate the effect of sea-level rise on the Cooks River catchment. High, intermediate and low sea-level rise scenarios were selected based upon the Intergovernmental Panel on Climate Change Fifth Assessment Report (IPCC, 2013a). The IPCC have modelled the potential magnitude of sea-level rise under a variety of possible future pathways, the Representative Concentration Pathways (RCPs). The RCPs describe different future scenarios based upon greenhouse gas emissions and atmospheric concentrations, air pollutant emissions and land use. This study examined the lowest, median and highest increases in SLR based upon RCP 2.6, RCP 4.5 and RCP8.5 respectively.

Values of sea-level rise were adjusted to incorporate the time period considered within SLAMM. The IPCC values of sea-level rise represent future rises with respect to sea level at 1986-2005, whilst SLAMM requires sea-level rise values with respect to 1990. On the basis of the algorithm used to incorporate sea-level rise within SLAMM, the rate of historic sea-level rise set for the study area (See Table 4.4) was used to adjust the RCP values (Table 4.5).

Table 4.5: Sea-level rise scenarios used in SLAMM modelling, the corresponding IPCC Representative Concentration Pathway (RCP) upon which they are based, and the adjusted value applied in SLAMM.

Sea-level rise scenario	RCP	Sea level rise[1] (m)	Sea level rise[2] applied (m)
Low	RCP 2.6 (Minimum)	0.28	0.322
Intermediate	RCP 4.5 (Median)	0.53	0.592
High	RCP 8.5 (Maximum)	0.98	1.022

[1]with respect to sea-level 1986-2005; [2]with respect to sea-level 1990 as required for SLAMM

4.2.6 SPATIAL AND STATISTICAL ANALYSIS

Coastal squeeze

With rising sea levels, wetlands maintain their position within the tidal frame by building elevation *in situ* through processes that increase the soil volume, alter their distribution in a landward direction, or risk being submerged by rising water levels. However, where steep gradients or developed areas flank wetlands, migration of tidal wetlands may be inhibited. This phenomenon has been termed coastal squeeze (Chmura, 2013, Doody, 2004, Pontee, 2013, Torio & Chmura, 2013). Given the extensively developed nature of the Cooks River, little space remains for wetlands to migrate inland, suggesting that coastal squeeze is likely to occur within the catchment. To examine the effect of coastal squeeze on wetland vegetation distribution, further modelling was undertaken and a comparative study of the output was conducted.

Keeping all parameters constant, an additional three scenarios (low, intermediate and high) were simulated in which built-up areas were permitted to convert to a vegetation class over time. Though such situations are completely hypothetical, as it is unlikely any management plan would consider allowing mostly urban areas to be removed in preference of vegetation, the simulations were used as an indication of the effects of coastal squeeze. This was calculated by determining the difference in modelled vegetation extent at 2100 when developed areas were maintained and when vegetation was permitted to ‘encroach’ upon current built-up areas.

Change detection

Change detection between the initial year of modelling, 2013, and 2100 was completed in ENVI 4.8™. The post modelling analysis provided spatial and statistical data on the conversion of vegetation and land use classes by the year 2100 under each sea-level rise scenario and with or without coastal squeeze effects. From such information, patterns of vegetation change and, by extension, the effects of sea-level rise on a variety of vegetation communities were able to be examined.

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4.3 Results and Discussion

4.3.1 MODEL OUTPUTS

When excluding the effects of built areas on the capacity for wetlands to maintain their position within the tidal frame through redistribution to higher elevations, mangrove extent increased under all scenarios and was primarily limited to lower reaches of the Cooks River (Figure 4.4). The best outcomes for mangrove extent were achieved under the highest sea-level rise scenario (Figure 4.5), with mangrove projected to increase between approximately 310-780% from a starting extent of 16.17 (ha) (Table 4.6). Conversely, saltmarsh extent diminished under all sea-level rise scenarios, and was restricted to the upper portions of tributaries; saltmarsh extent was projected to decline between approximately 2-15% from a baseline extent of only 19 ha.

Evidently, the high sea-level rise scenario favoured communities that are typically positioned lower in the tidal frame (i.e. mangrove, tidal flats and estuarine waters), despite the effects of coastal squeeze on wetland adjustment being limited. Land use/wetland vegetation classes that occur higher in the tidal frame (i.e. saltmarsh and developed land) exhibited declines in extent under all scenarios. This outcome implies that rates of sediment supply and deposition within the model are not high enough to compensate for the increased inundation occurring with sea-level rise; consequently, communities positioned higher in the tidal frame at 2013 were projected to convert to lower elevation communities by 2100. As the Cooks River is a highly developed catchment, with many impervious surfaces, it is not surprising that rates of sediment supply are low. However, it should be noted that rates of sediment supply were derived from measurements of mangrove and saltmarsh surface elevation change at Homebush Bay, rather than being derived from empirical, site-specific measurements of mangrove and saltmarsh sedimentation or surface elevation change in the Cooks River. This is a factor that requires further validation through site-specific analyses of sediment accumulation and sediment sources.

The tidal plane of Cooks River within Marrickville and Dulwich Hill are typically broader than other parts of the Cooks River (Figure 4.2). Consequently, the spatial distribution of areas projected to support large extents of mangrove occurred in this area. Greatest establishment of mangrove corresponded to the current distribution of sporting fields and golf clubs along the Cooks River, including Marrickville Golf Club, Kogarah Golf Club and Wills Ground, Earlwood. Importantly, the model also projected conversion of the northern end of the north-south runway at Sydney Airport to mangrove under a high sea-level rise scenario, or a complex of mangrove and *Casuarina* under a low-sea-level rise scenario. Significant areas of built up land northwest of Kogarah Golf Club is projected to convert to mangrove under a high sea-level rise scenario.

Within the Wolli Creek study area, the existing mangrove and saltmarsh area located near Wolli Creek train station was projected to convert to tidal flat under a high sea-level rise scenario. There was also some proliferation of mangrove further along Wolli Creek, though significant expansion was limited by high topography along the northern shoreline. Under the high sea-level rise scenario, translation of saltmarsh to higher elevations was limited by the position of the Wolli Creek train line and the small patch of saltmarsh immediately west of Wolli Creek Train Station was converted to tidal flat. A small patch of saltmarsh located between High Cliff Road and Wolli Creek was projected to diminish in size, but be maintained under all sea-level rise scenarios; this outcome is surprising and requires validation of starting elevations for this saltmarsh patch. This can be achieved by undertaking a DEM validation assessment, which preferably should occur prior to broader assessments of sea level rise impacts in the Sydney Region. Under all sea-level rise scenarios, *Casuarina* was projected to translate to higher elevations that are currently both natural and developed areas. There was little change projected in the configuration of undeveloped land and *Casuarina* in the upper reaches of Wolli Creek, reflecting the influence of tidal attenuation and starting elevation in this area (Figure 4.6).

4.3.2 COASTAL SQUEEZE

Significant declines in the area of mangrove and saltmarsh were projected under all sea-level rise scenarios when built-up areas were not able to convert to other vegetation classes. These effects were most pronounced under the high sea-level rise scenario and preferentially limited the extent of mangrove and *Casuarina* along Cooks River. The largest area of wetland vegetation projected to be impacted by the combined influence of sea-level rise and coastal squeeze occurred along Alexanders Canal, within Sydney Airport (Figure 4.7). Model assumptions presumed that assets in this area would be maintained, despite the low-lying nature of the landscape, thereby having significant influence on wetland extent and ecosystem services. This is clearly evident in the side-by-side comparison of model outputs with and without coastal squeeze effects occurring (Figure 4.8d and 4.8b, respectively).

Built-up areas on the northern shoreline between Canterbury and Marrickville do not abut the shoreline of Cooks River, thereby providing mangrove refuge space under all sea-level rise scenarios. This is unlike the southern shoreline of the Cooks River, where built areas mostly abut the shoreline and limit the establishment of mangrove and saltmarsh. Importantly, the northern shoreline also exhibits a moderate-high tidal range (Figure 4.2), which provides more vertical space to support wetland vegetation. The impact of coastal squeeze on saltmarsh distribution was only marginal and reflects the limited distribution of saltmarsh at the starting of the modelling period (i.e. based on vegetation mapping). Maintaining the extent of saltmarsh in the future should become a management priority to facilitate the maintenance of ecosystems services provided by saltmarsh.

Within the Wolli Creek study area, the translation of wetland communities was restricted in the area between the southern shoreline of Wolli Creek and the train line due to an industrial division on the northern side of Henderson Street acting as a barrier to migration (Figure 4.8).

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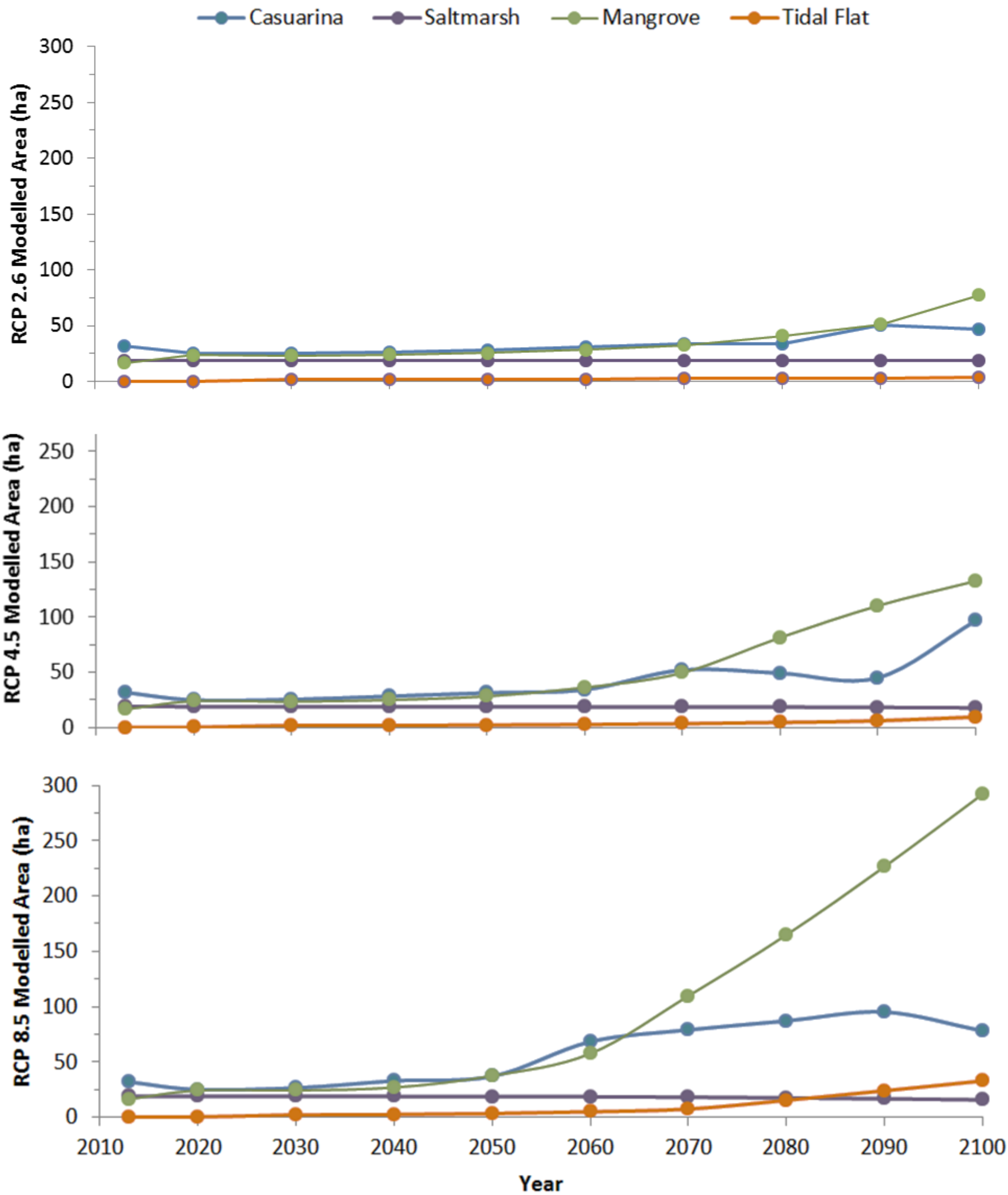

Figure 4.3: Change in extent of vegetation classes over the modelling period under a **a)** a low, **b)** intermediate, and **c)** a high sea level rise scenario.

Table 4.6: Modelled extent (ha) of vegetation classes under low, intermediate, and high sea-level rise scenarios with coastal squeeze effects occurring. Total change in area (ha) and percentage (%) change in area also provided for each sea-level rise scenario.

Scenario	Year	Undeveloped Land	*Casuarina*	Saltmarsh	Mangrove	Tidal Flat	Estuarine Water
Low sea-level rise (RCP 2.6)	2013	568.87	28.79	19.09	16.17	0.27	120.03
	2020	568.56	24.13	19.07	21.13	0.31	120.03
	2030	567.96	24.12	19.06	21.07	0.98	120.03
	2040	567.00	24.26	19.04	21.79	1.11	120.04
	2050	564.87	25.25	18.98	22.85	1.23	120.04
	2060	561.94	25.59	18.93	25.28	1.44	120.05
	2070	556.88	27.30	18.89	28.41	1.71	120.08
	2080	550.03	26.74	18.84	35.59	1.97	120.19
	2090	526.87	41.58	18.78	43.52	2.38	120.46
	2100	508.39	36.37	18.69	66.88	2.75	120.84
	Total Change (ha)	-60.48	7.58	-0.40	50.70	2.48	0.81
	Total Change (%)	-10.63	26.32	-2.11	313.47	922.80	0.68
Intermediate sea-level rise (RCP 4.5)	2013	568.87	28.79	19.09	16.17	0.27	120.03
	2020	568.43	24.13	19.07	21.23	0.33	120.03
	2030	567.58	24.10	19.06	21.36	1.10	120.03
	2040	564.96	25.40	18.98	22.52	1.33	120.04
	2050	561.31	25.89	18.92	25.40	1.66	120.06
	2060	552.58	27.84	18.86	31.72	2.17	120.14
	2070	525.82	43.44	18.76	42.25	2.84	120.50
	2080	502.04	38.89	18.61	69.56	3.95	121.16
	2090	483.24	33.04	18.35	92.95	5.34	122.00
	2100	466.40	30.13	17.97	109.58	8.68	123.24
	Total Change (ha)	-102.47	1.33	-1.12	93.41	8.41	3.20
	Total Change (%)	-18.01	4.63	-5.85	577.50	3128.08	2.67
High sea-level rise (RCP 8.5)	2013	568.87	28.79	19.09	16.17	0.27	120.03
	2020	568.25	24.12	19.07	21.40	0.36	120.03
	2030	566.69	24.30	19.02	21.89	1.29	120.04
	2040	561.09	27.33	18.91	24.00	1.85	120.06
	2050	549.48	29.75	18.83	32.46	2.64	120.21
	2060	505.48	57.44	18.64	47.30	4.20	120.98
	2070	478.59	38.21	18.23	91.43	6.43	122.27
	2080	454.61	33.80	17.58	112.44	14.16	124.26
	2090	430.81	33.01	16.76	129.31	22.17	126.10
	2100	412.65	27.05	15.82	145.16	30.37	127.77
	Total Change (ha)	-156.22	-1.74	-3.27	128.99	30.10	7.74
	Total Change (%)	-27.46	-6.05	-17.11	797.49	11192.93	6.45

Table 4.7: Modelled extent (ha) of vegetation classes under low, intermediate, and high sea-level rise scenarios without coastal squeeze effects occurring. Total change in area (ha) and percentage (%) change in area also provided for each sea-level rise scenario.

Scenario	Year	Developed Land	Undeveloped Land	*Casuarina*	Saltmarsh	Mangrove	Tidal Flat	Estuarine Water
Low sea-level rise (RCP 2.6)	2013	1993.87	568.87	31.92	19.09	16.55	0.29	120.03
	2020	1993.68	568.56	25.06	19.07	23.86	0.36	120.03
	2030	1993.29	567.96	25.22	19.06	23.21	1.82	120.05
	2040	1992.18	567.00	26.24	19.04	24.15	1.74	120.28
	2050	1990.75	564.87	28.31	18.98	25.52	1.88	120.30
	2060	1988.05	561.94	30.99	18.93	28.25	2.11	120.34
	2070	1986.42	556.88	33.28	18.89	32.38	2.39	120.40
	2080	1983.92	550.03	33.90	18.84	40.84	2.67	120.54
	2090	1980.23	526.87	50.12	18.78	51.02	3.10	120.85
	2100	1975.60	508.39	46.41	18.69	77.46	3.48	121.27
	Total Change (ha)	-18.27	-60.48	14.50	-0.40	60.91	3.18	1.24
	Total Change (%)	-0.92	-10.63	45.42	-2.11	367.98	1085.88	1.04
Intermediate sea-level rise (RCP 4.5)	2013	1993.87	568.87	31.92	19.09	16.55	0.29	120.03
	2020	1993.60	568.43	25.10	19.07	24.02	0.38	120.03
	2030	1992.91	567.58	25.45	19.06	23.60	1.96	120.05
	2040	1990.85	564.96	28.45	18.98	25.07	2.01	120.30
	2050	1987.77	561.31	31.49	18.92	28.41	2.38	120.35
	2060	1985.18	552.58	34.35	18.86	36.33	2.93	120.48
	2070	1979.95	525.82	52.24	18.76	49.70	3.64	120.89
	2080	1973.61	502.04	49.34	18.61	81.58	4.78	121.61
	2090	1966.40	483.24	45.20	18.35	110.36	6.23	122.53
	2100	1906.09	466.40	96.73	17.97	132.68	9.65	123.85
	Total Change (ha)	-87.78	-102.47	64.81	-1.12	116.13	9.35	3.82
	Total Change (%)	-4.40	-18.01	203.07	-5.86	701.61	3189.97	3.18
High sea-level rise (RCP 8.5)	2013	1993.87	568.87	31.92	19.09	16.55	0.29	120.03
	2020	1993.51	568.25	25.11	19.07	24.25	0.41	120.03
	2030	1991.75	566.69	26.61	19.02	24.30	2.21	120.05
	2040	1987.69	561.09	32.98	18.91	27.01	2.60	120.35
	2050	1983.76	549.48	37.09	18.83	37.59	3.45	120.56
	2060	1974.83	505.48	68.35	18.64	57.61	5.10	121.42
	2070	1936.62	478.59	79.05	18.23	109.76	7.47	122.82
	2080	1889.56	454.61	87.16	17.58	164.92	15.45	124.95
	2090	1835.20	430.81	95.22	16.76	226.58	24.02	126.93
	2100	1794.39	412.65	78.12	15.82	292.10	32.94	128.87
	Total Change (ha)	-199.48	-156.22	46.20	-3.27	275.55	32.64	8.84
	Total Change (%)	-10.00	-27.46	144.76	-17.11	1664.80	11134.00	7.37

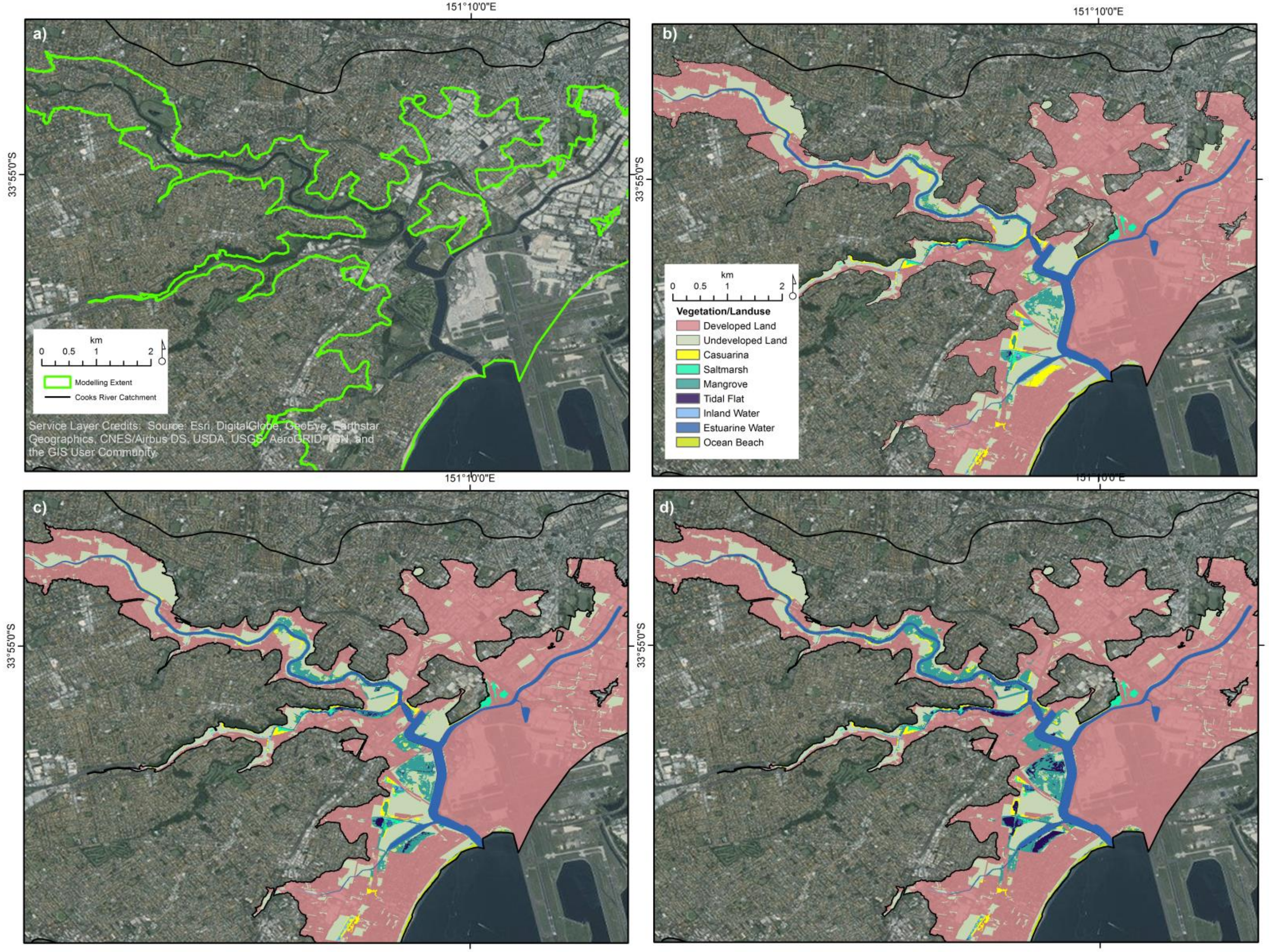

Figure 4.4: **a)** Model extent and modelled vegetation of the Cooks River catchment at 2100 under **b)** low, **c)** intermediate, and **d)** high sea level rise scenarios. In this model output, developed land was considered to remain stable to 2100 and coastal squeeze effects are evident.

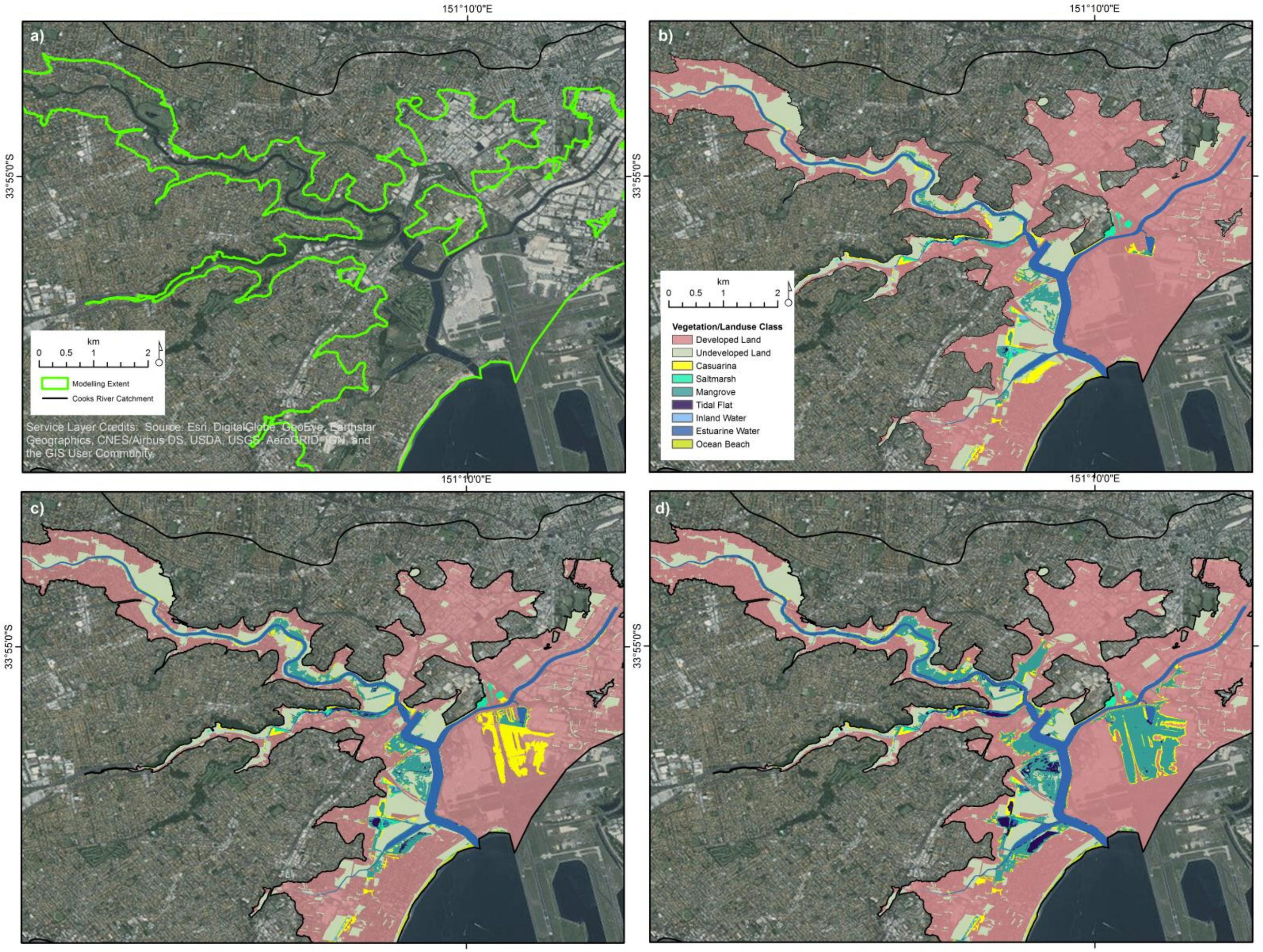

Figure 4.5: **a)** Model extent and **b)** starting vegetation distribution at Cooks River catchment; and comparison of modelled vegetation at 2100 under **c)** a scenario with coastal squeeze occurring due to the occurrence of wetland migration barriers, and d) coastal squeeze limited due to assumptions of built-areas converting to wetlands vegetation.

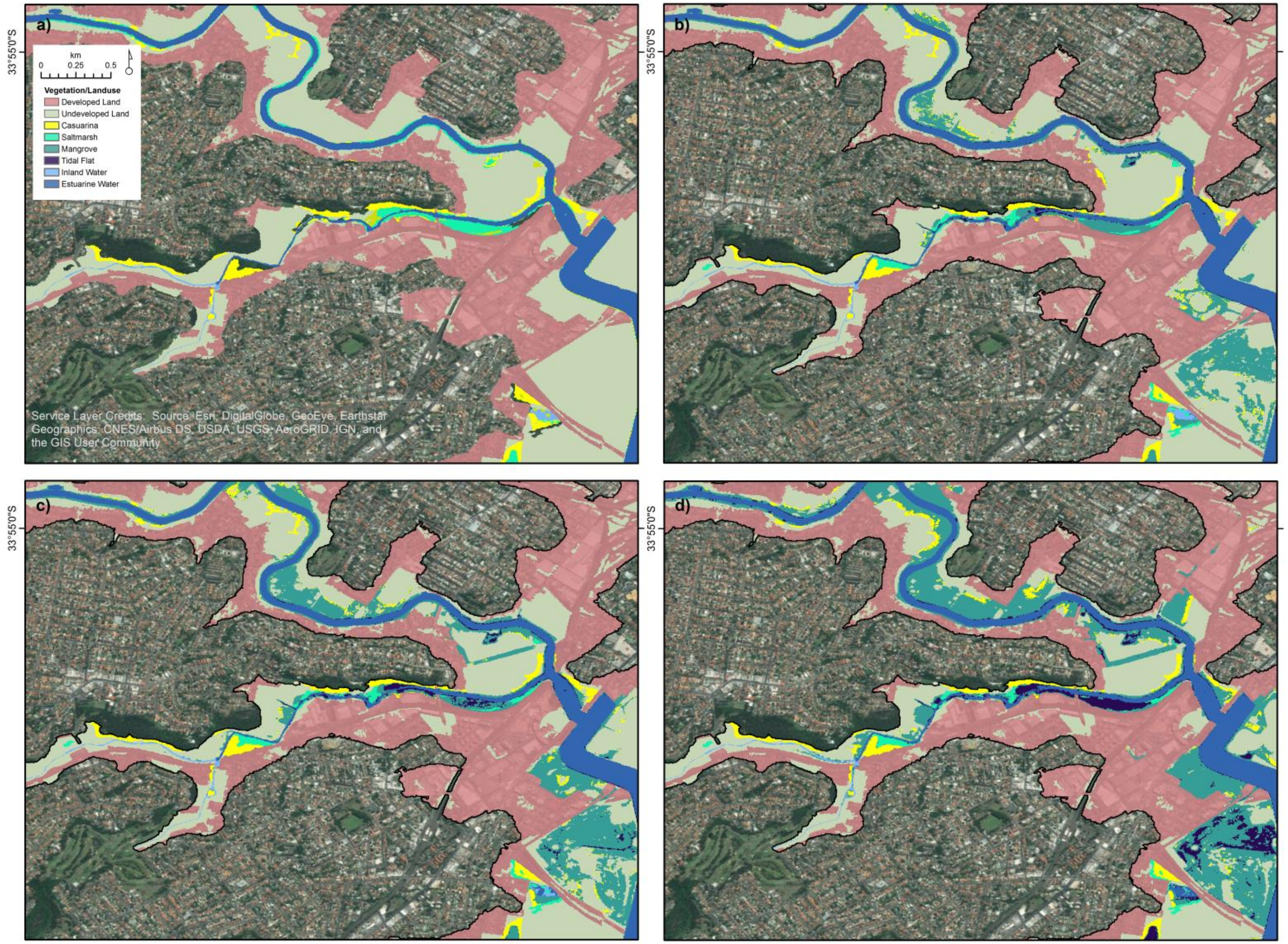

Figure 4.6: **a)** Model extent and modelled vegetation of the Wolli Creek sub-catchment at 2100 under **b)** low, **c)** intermediate, and **d)** high sea level rise scenarios. In this model output, developed land was considered to remain stable to 2100 and coastal squeeze effects are evident.

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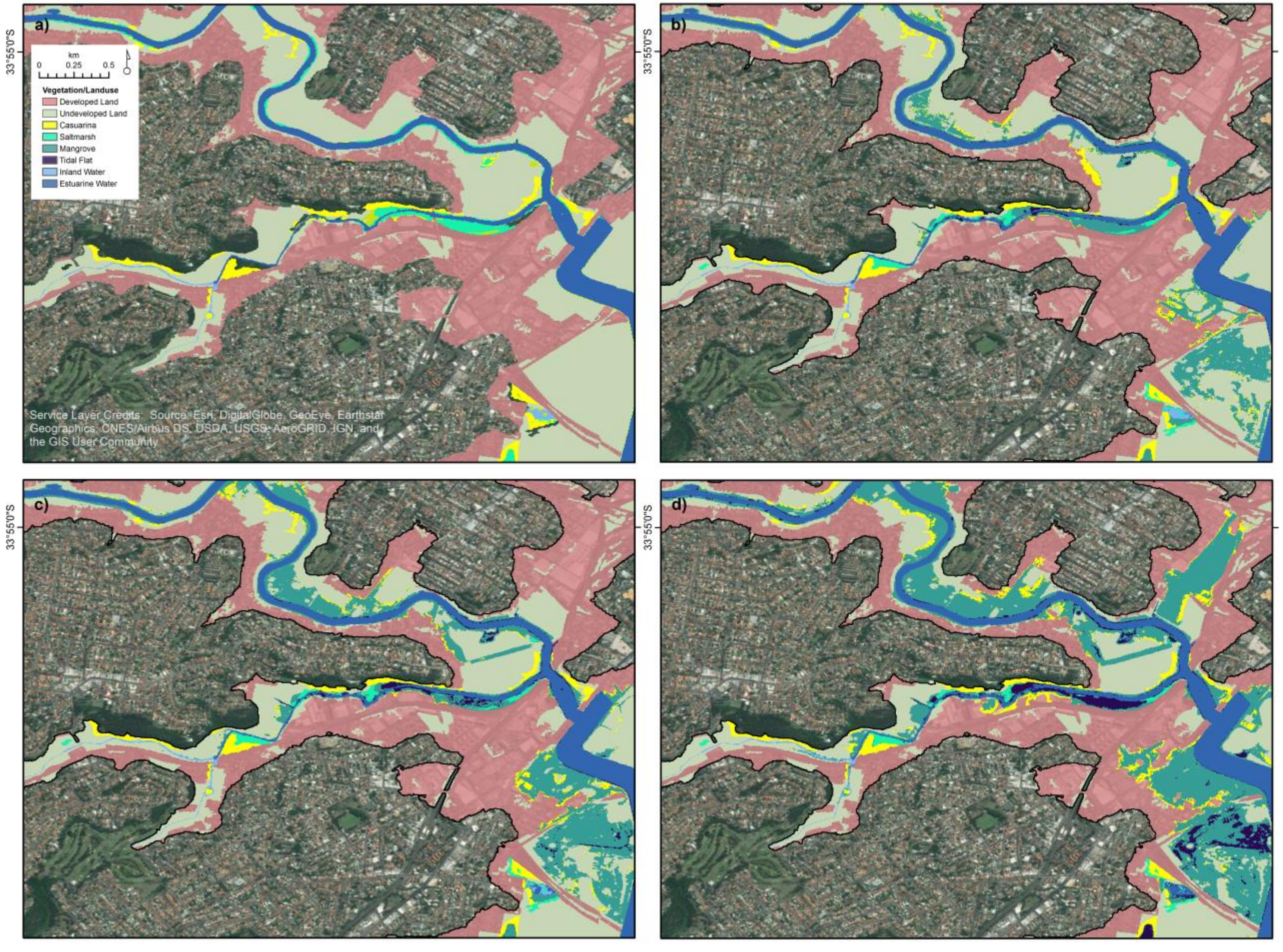

Figure 4.7: **a)** Model extent and modelled vegetation of the Wolli Creek study area at 2100 under **b)** low, **c)** intermediate, and **d)** high sea level rise scenarios. In this model output, developed land was able to convert to coastal wetland and coastal squeeze effects were reduced.

4.3.3 CHANGE DETECTION

Change detection was undertaken to establish which classes primarily converted to wetland vegetation classes within model outputs. When coastal squeeze effects were minimised and developed land was able to convert to wetland classes where possible, approximately 198.54 ha of developed land was projected to convert to *Casuarina*, mangrove or freshwater wetland under a high-sea level rise scenario (Table 4.7, Figure 4.8d). This contrasts with the area of developed land projected to convert under a low sea-level rise scenario, which is in the order of 18.37 ha and was primarily composed of *Casuarina* (Figure 4.8c). The overall pattern was the conversion of high intertidal classes to lower intertidal classes or open water, with this pattern exacerbated under a high sea-level rise scenario.

Under a low sea-level rise scenario, there was projected to be virtually no loss of mangrove extent, and significant gains through the conversion of undeveloped land and *Casuarina* to mangrove. Saltmarsh marginally diminished in size from 19.09 to 18.69 ha, while *Casuarina* exhibited the largest change in area with 70 ha converting to mangrove under a low sea-level rise scenario.

Under a high sea-level rise scenario the model projected conversion of 9.16 ha of mangrove habitat to lower intertidal tidal flat. Significant gains in mangrove extent were modelled to occur through the conversion of developed land (147.61 ha), undeveloped land (133.74 ha) and *Casuarina* (5.90 ha) to mangrove. However, reductions in saltmarsh extent were greater under a high sea-level rise scenario, declining to 15.82 ha, with 2.51 ha converting to tidal flat. Tidal flat increased in area by 32.64ha with significant conversions from mangrove (9.16 ha), *Casuarina* (6.77 ha) and undeveloped land (13.36 ha) to tidal flat under a high sea-level rise scenario.

When coastal squeeze effects were exacerbated by preventing the conversion of developed land to other classes (Table 4.8), the overall outcome for wetland vegetation was a reduction in wetland area (Figure 4.8 a-b). For example, under a low sea-level rise scenario when coastal squeeze effects were exacerbated the model projected a mangrove area of 50.70 ha, contrasting with an area of 77.46 ha when coastal squeeze effects were reduced; under a high sea-level rise scenario, these differences were exacerbated with only 128.99 ha of mangrove projected under a coastal squeeze scenario, compared to 292.10 ha when coastal squeeze effects were minimised. Similarly, saltmarsh was projected to have an area of only15.82 ha. There was little difference in the outcome for saltmarsh between the model projection with or without coastal squeeze effects, however the differences were substantial for *Casuarina* under both a high sea-level rise scenario (27.05 ha with coastal squeeze, 78.12ha without coastal squeeze), and low sea-level rise scenario (7.58 ha with coastal squeeze, 46.41 ha without coastal squeeze).

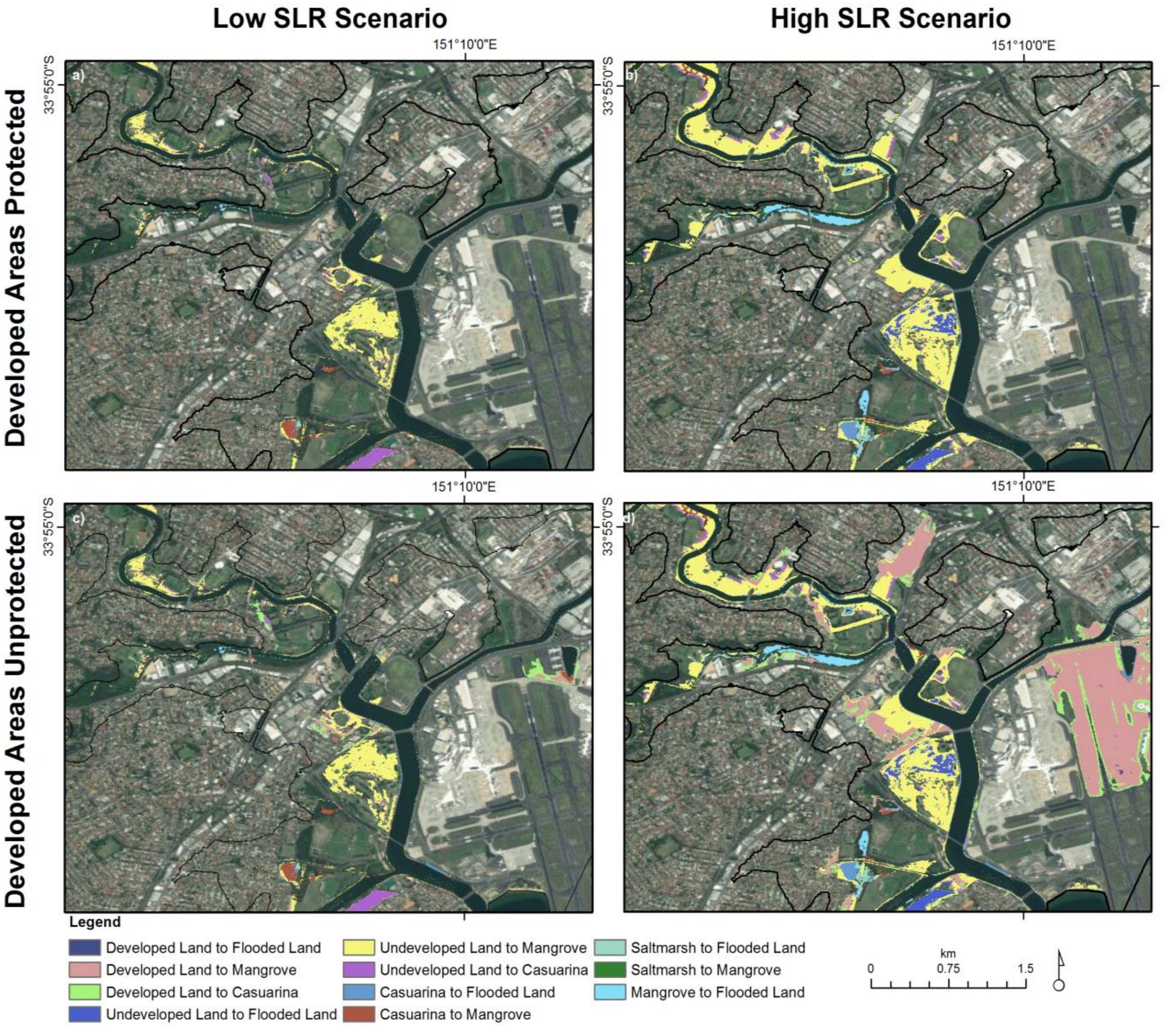

Figure 4.8: Change detection analysis comparing current vegetation distribution with projected vegetation distribution under a **a)** low sea-level rise scenario and developed land is protected from land use changes; **b)** high sea-level rise scenario and developed land is protected from land use changes; **c)** low sea-level rise scenario and developed land is able to convert to other land uses; and **d)** high sea-level rise scenario and developed land is able to convert to other land uses

Table 4.7: Change in area of vegetation classes from 2013 (columns) to 2100 (rows) based on the model assumption that developed land could convert to other classes, and coastal squeeze effects are minimised.

Scenario			2013						
			Developed Land	Undeveloped Land	*Casuarina*	Saltmarsh	Mangrove	Tidal Flat	Estuarine Water
Low sea-level rise (RCP 2.6)	2100	Developed Land	1975.39	0	0	0	0	0	0
		Undeveloped Land	0	509.51	0	0	0	0	0
		Casuarina	9.87	13.35	20.65	0	0	0	0
		Saltmarsh	0	0	0	18.71	0	0	0
		Mangrove	8.50	45.91	8.51	0.17	16.12	0	0
		Tidal Flat	0	0.17	2.08	0.22	0.83	0.20	0
		Estuarine Water	0	0	0.36	0	0	0.01	120.03
Intermediate sea-level rise (RCP 4.5)		Developed Land	1905.82	0	0	0	0	0	0
		Undeveloped Land	0	467.59	0	0	0	0	0
		Casuarina	66.50	8.05	19.28	0	0	0	0
		Saltmarsh	0	0.00	0.00	18.04	0	0	0
		Mangrove	21.43	90.87	7.87	0.42	14.41	0	0
		Tidal Flat	0	2.38	3.78	0.64	2.57	0.16	0
		Estuarine Water	0	0.00	0.68	0.00	0.00	0.05	120.03
(RCP 8.5) High sea-level rise		Developed Land	1793.90	0	0	0	0	0	0
		Undeveloped Land	0	413.48	0	0	0	0	0
		Casuarina	50.25	7.74	17.07	0	0	0	0
		Saltmarsh	0	0	0	15.95	0	0	0
		Mangrove	147.61	133.74	5.90	0.60	7.42	0	0
		Tidal Flat	0.58	13.36	6.77	2.51	9.16	0.01	0.00
		Estuarine Water	0.10	0.03	1.85	0.03	0.40	0.21	120.03

Table 4.8: Change in area (ha) of vegetation classes from 2013 (columns) to 2100 (rows) based on the model assumption that developed land could not convert to other classes, and coastal squeeze effects are exacerbated.

Scenario			2013						
			Developed Land	Undeveloped Land	*Casuarina*	Saltmarsh	Mangrove	Tidal Flat	Estuarine Water
Low sea-level rise (RCP 2.6)	2100	Developed Land	1993.77	0	3.09	0	0.52	0.02	0
		Undeveloped Land	0	509.51	0	0	0	0	0
		Casuarina	0	13.35	20.65	0	0	0	0
		Saltmarsh	0	0	0	18.71	0	0	0
		Mangrove	0	45.91	6.52	0.17	15.67	0	0
		Tidal Flat	0	0.17	1.32	0.22	0.79	0.20	0
		Estuarine Water	0	0	0.03	0	0	0	120.03
Intermediate sea-level rise (RCP 4.5)		Developed Land	1993.77	0.00	3.09	0.00	0.52	0.02	0
		Undeveloped Land	0	467.61	0	0	0	0	0
		Casuarina	0	8.05	19.28	0	0	0	0
		Saltmarsh	0	0	0	18.04	0	0	0
		Mangrove	0	90.84	6.21	0.42	13.98	0	0
		Tidal Flat	0	2.38	2.87	0.64	2.48	0.16	0
		Estuarine Water	0	0	0.15	0	0.00	0.04	120.03
(RCP 8.5) High sea-level rise		Developed Land	1993.77	0.00	3.09	0	0.52	0.02	0
		Undeveloped Land	0	413.48	0.00	0	0	0	0
		Casuarina	0	7.74	17.07	0	0	0	0
		Saltmarsh	0	0.00	0.00	15.95	0	0	0
		Mangrove	0	133.74	5.29	0.60	7.23	0	0
		Tidal Flat	0	13.36	5.31	2.51	8.84	0	0
		Estuarine Water	0	0.03	0.84	0.03	0.39	0.20	120.03

4.3.4 STUDY LIMITATIONS

This study focused upon modelling the effects of sea-level rise on estuarine vegetation, specifically wetland vegetation, of the Cooks River catchment. A number of limitations attributable to data quality, data availability and SLAMM model conceptualisation and implementation should be noted when interpreting or using the results of this study.

- *Quality of the Digital Elevation Model (DEM)*. This study utilised the LPI 5m LiDAR derived DEM for the area that has a reported RMSE of 1m. Elevation data is the basis of the modelling technique used in this study. The accuracy of the elevation model utilised is therefore considered to be paramount, especially in modelling of low-lying wetlands where small errors in the surface representation result in large variations in the hydrological properties, geomorphic adjustments and simulation of wetland persistence or demise. It is noted, therefore, that errors within the DEM may result in significant errors within model output. Future modelling efforts should consider validation of the elevation model prior to implementing the model and, possibly, working upon the technical difficulties that reduce the ability of a 1 metre DEM being used in SLAMM.
- *Paucity of sedimentation and surface elevation change data* within the Cooks River catchment. Though some sedimentation rates are reported for the Cooks River, the rate of change in surface elevation is not recorded. As sedimentation rates do not necessarily correspond to changes in surface elevations within wetlands, it was considered more appropriate to source rates of surface elevation change for mangrove and saltmarsh communities from a wetland environment similar to that at Cooks River, Homebush Bay. It is noted, however, that rates of surface elevation change recorded at Homebush Bay may not directly compare with those within the Cooks River catchment and as such provide only possible indicative values of the parameter.
- *Non-hydrodynamic nature of SLAMM*. The model assumes that sea level will rise equally throughout any waterbody and models future scenarios ‘at equilibrium’. This does not necessarily reflect reality.
- *Deterministic nature of SLAMM*. Deterministic models simulate scenarios based entirely upon the input parameters, without allowing for any randomness or potential change in parameters. As SLAMM is a deterministic model, the results presented within this report represent a number of potential scenarios but do not simulate the range of possible future scenarios. High, intermediate and low sea level scenarios have been chosen to provide an indication of the possible effects of sea level rise on the Cooks River catchment by 2100. An uncertainty analysis should be conducted, however, to account for a range of other possible sea level rise scenarios and potential errors included in the input data (eg. Elevation data, rates of surface elevation change).
- *Discrete treatment of tidal range within SLAMM*. The tidal range changes in a continuous manner throughout the Cooks River and its tributaries. SLAMM, however, accounts for tidal range in a discrete manner, potentially influencing the modelling of vegetation within the tidal frame, such as mangrove and saltmarsh species. This study has adopted an approach that attempts to slightly ameliorate the problem, however, potential error is still present within the final modelling results.
- *Treatment of vegetation succession within SLAMM*. Saltmarshes, as defined for the Australian context within SLAMM, are unable to convert or expand upland in contrast to real-world evidence. Though this may not be a significant problem for the modelling of Cooks River catchment due to the relatively small area initially covered by saltmarsh, interpretation of model results should be conducted with this potential limitation in mind.

4.4 Recommendations

As an outcome of this focus study we can provide multiple recommendations that relate to improving modelling capacity and planning recommendations. These recommendations will be integrated with other aspects of this project in the final project report.

4.4.1 MODELLING

The validity of any model or prediction is related to the accuracy of the input information. Thus, it follows that the accuracy of the model outputs and associated user confidence is only as good as the combined accuracy of the input parameters. To this end, and due to time, resource and data limitations, multiple assumptions were made regarding input model parameters that could be improved with further data collection and validation.

- *DEM validation and correction*: input elevation information is essential for determining the ability of a model to simulate the effect of SLR on coastal wetlands, especially as small errors in elevation for a low-lying, shallow gradient area can propagate significant errors in wetland inundation simulations (Gornitz *et al.*, 2001, Pugh, 2004). This pilot study assumed that the input DEM was accurate and a reliable source of information upon which to project geomorphological and vegetation change could be projected. LiDAR-derived DEMs are typically generated for multiple purposes, and rarely for the purpose of projecting wetland distribution with respect to elevation; accuracy assessments typically exaggerate the accuracy for wetland vegetation (Aguilar & Mills, 2008, Coveney & Fotheringham, 2011, Flood, 2004). Mogensen (2016) demonstrated the improved accuracy when validating the elevations within underlying DEMs and adjusting DEMs to reduce vertical errors. It is highly recommended that further modelling incorporate an assessment of the validity of underlying DEMs and measures to improve their accuracy.
- *Ground-truthing of vegetation distribution patterns*: while vegetation distribution patterns appeared reasonable, some anomalies arose regarding the long-term stability of saltmarsh at some sites throughout the study area. Preliminary analysis indicates that these anomalies may relate to the relationship between vegetation distribution and the accuracy of the DEM at the start of the modelling period (i.e. 2013). The efficacy of these relationships should be confirmed not only through validation of the DEM, but also through extensive ground-truthing of the accuracy of polygons within the input vegetation layer.
- *Spatially explicit accretion and elevation change data*: The second most important component within models of wetland response to sea-level rise is the elevation adjustment occurring within wetlands following sea-level rise. This adjustment is dependent upon many site specific variables that relate to dictate rates of sediment supply, deposition, plant productivity and increases in soil volume (Lovelock *et al.*, 2015, Rogers *et al.*, 2014). This study extrapolated accretion and elevation data from nearby Homebush Bay (Bowie, 2015b); despite its proximity, the validity of using this data at Cooks River should be confirmed through site-specific analyses of accretion and elevation change. This validation exercise could also be extended to confirm the validity of model assumptions regarding the influence of concrete-lined channels on accretion along Cooks River. Following validation of these estimates, there would be capacity to explore the implications of activities that increase or decrease rates of accretion and surface elevation change by enhancing these parameters within model scenarios.
- *Validation of tidal plane modelling*: This study applied the tidal plane models of OEH (2013b), which is undertaken on the basis of relatively little empirical information regarding tidal propagation along Cooks River. The tidal plane model is particularly important for making input parameters spatially explicit and influences model outputs. Analysis of tidal propagation would provide confirmation of the validity of the tidal plane model as a means for creating discrete sub-catchments within the modelling.

- *Validation of model*: SLAMM was used in this study as it is a readily accessible modelling environment that can be applied with limited parameterisation. However, there remain criticisms of this model, many of which are not unfounded (see Mogensen, 2016 for a discussion of these). While every effort to account for these factors has been undertaken in this study, validation of this model by comparison with other models will increase confidence in the model outputs. As a process based model, we recommend the approach of Mogensen (2016), who validated against an empirically derived model, and extending this to apply probabilistic techniques.
- *Uncertainty analysis:* This study examined deterministic results of SLAMM utilising parameters that are inherently variable and include significant uncertainty. An uncertainty analysis will improve the capacity to communicate error and uncertainty of input parameters within SLAMM. Furthermore, by incorporating error into analyses probabilistic modelling of specific vegetation communities can be achieved.

4.4.2 PLANNING AND MANAGEMENT

Despite deficiencies and limitations in modelling, spatial patterning in the distribution of mangrove and saltmarsh was projected, and when intersected with information regarding existing built areas and planning scenarios, this information provides an indication of the fate of mangrove and saltmarsh in the 21st century. There remains scope to alter the fate of mangrove and saltmarsh by altering either processes influencing ecosystem response to sea level through management actions, or altering planning scenarios regarding the distribution of built areas that act as a barrier to wetland migration and enhance coastal squeeze effects.

- *Improving ecosystem health*: autonomous adaptation by ecosystems to climate change and sea-level rise is dependent upon healthy ecosystems. Maintaining or enhancing ecosystem health is essential for enhancing resilience (Lovelock *et al.*, 2009b, Rogers *et al.*, 2016).
- *Enhancing ecosystem elevation*: modelling outputs indicated that the best ecosystem outcomes were achieved when wetlands were positioned higher within the tidal frame. Known as 'elevation capital' (Cahoon & Guntenspergen, 2010, Lovelock *et al.*, 2015), wetlands positioned higher within their optimal tidal position, exhibit greater capacity for survival as they inherently have a longer period of time before they are submerged as sea levels increase. In addition, this is particularly important for saltmarsh which exhibits a very narrow elevation distribution and which can be encroached upon by both mangrove at lower elevations and *casuarina* at higher elevations. Improving elevation capital and maintaining optimal positions within the tidal frame have been enhanced by applying sediment veneers to wetlands, that are typically sourced from dredge spoil (Ford *et al.*, 1999). This technique has yet to be applied in Australia, but is commonly used throughout the USA to improve wetland resilience.
- *Maintaining wetland complexity (mosaics)*: this pilot study demonstrated both the restricted nature of saltmarsh throughout the study region and its reduced capacity to adjust to sea-level rise due to its distribution within a narrow tidal range positioned between the highly adaptive vegetation communities dominated by mangrove and *Casuarina*. Maintenance of the full range of ecosystem services provided by coastal wetlands is dependent upon the spatial mosaicking of wetland communities and habitat complexity. Actions to improve wetland complexity have been achieved by removal of mangrove seedlings where habitat conditions are favourable for both mangrove and saltmarsh (Laegdsgaard *et al.*, 2009). This has typically been undertaken to improve the provision of waterbird habitat that can become restricted when mangrove block waterbird lines of site to waterways; but could also be applied to the removal of *Casuarina* as saltmarsh translates to higher elevations that are occupied by established *Casuarina*. This may be particularly important given the allelopathic capacity of *Casuarina* (El-Baha, 2003).
- *Limiting construction of future barriers to wetland translation* by establishing setbacks, or buffers and restricting development in areas projected to be prone to inundation and that

are likely to support tidal wetlands (Lovelock *et al.*, 2009b). This is particularly important for the northern shoreline of Cooks River which exhibits significant capacity as a wetland vegetation refuge.

- *Removing or managing barriers to wetland translation*. Built areas throughout the study area act as a significant barrier to wetland adaptation and provision of ecosystem services. In this pilot study we explored the implications of maintaining built areas, and this was projected to have an impact on the extent of wetland area by the end of the 21st century under all sea-level rise scenarios. Evidently, there are low-lying areas within the study area, primarily along Alexanders Canal, and within vicinity of Sydney Airport that are particularly vulnerable to inundation and could support significant wetland areas should conversion be facilitated.
- *Exploring complementary land use options*. On the basis of the occurrence of low-lying areas, there is capacity to explore complementary land uses co-occurring within the study area. To this end, the network of golf courses and playing fields aligning the margins of the Cooks River could act as important refuges for mangrove and saltmarsh, and provision of some of this area for wetland vegetation would provide additional ecosystem services.

Chapter Five: Adaptation options and recommendations

5.1 Overview of coastal adaptation options

There are three main approaches to coastal zone management and adaptation for climate change: protect, retreat and accommodate (DCC 2009). These terms are more commonly applied to adaptation of the built environment within the coastal zone, but can also be readily applied to landforms and ecosystems. These terms largely arose from the first assessment report of the Intergovernmental Panel on Climate Change (IPCC) (Gilbert and Vellinga 1990) and can be broadly defined as follows:

- *Protect:* Defend vulnerable areas, especially population centres, economic activities, and natural resources.
- *Accommodate:* Continued occupancy and use of vulnerable areas.
- *Retreat:* Abandon land and structures in vulnerable areas, and resettlement of inhabitants.

Since the first IPCC assessment report, there has been an evolution of coastal adaptation options and terms (Nicholls *et al.* 2007). There is increasing recognition of hybrid or "selective" options, and the use of options that reverse "maladaptation" or build future resilience. Maladaptation has been defined as "action taken ostensibly to reduce vulnerability to climate change that impacts adversely on, or increases the vulnerability of other systems, sectors or social groups" (Barnett and O'Neill 2010). Examples include groynes and other structures that have resulted in the hardening of shorelines that were designed to resolve one set of issues however have led to new and adverse outcomes.

Management plans designed to identify actions to address coastal management issues may offer a range of actions including the "do nothing" option. This term is somewhat contradictory as it refers to a proactive decision not to undertake any adaptation actions. The risks associated pursuing a "doing nothing" option accept that action will be delayed until a future time that includes up to the point when the climate change risk has eventuated. The "do nothing" option more accurately should be regarded as a delayed version of retreat (Nicholls *et al.* 2007) and a more appropriate term may be "no intervention", a term used by Cooper *et al.* (2002). In many cases this option is only provided within management plans as a basis for comparison with other adaptation options (BMT WBM 2012); however no- or limited-intervention options may be valuable and should not be readily dismissed. For example, there are some locations where coastal wetland resilience to expected sea-level rise is sufficiently high (e.g. due to sufficient sediment delivery and/or contributions from belowground biomass production), where there is no need to intervene.

Some management plans also refer to "no-regrets" adaptation options. This refers to a policy or technique for selecting adaptation options, defined as "a policy that would generate net social and/or economic benefits irrespective of whether or not anthropogenic climate change occurs" (Parry *et al.* 2007). The term can also be applied to selecting climate change mitigation options. Tidal reinstatement of degraded coastal landscapes as a climate change adaptation mechanism has the additional benefits of habitat provision and has been funded for these benefits alone.

A range of adaptation options for estuarine shorelines, including tidal wetland ecosystems, are presented in Table 5.1 and discussed below. This list is not exhaustive, but includes adaptation options that might be utilised within the study region as well as innovative options identified within scientific literature.

5.1.1 ESTUARINE SHORELINE ADAPTATION

Depositional shorelines and coastal lowlands on estuaries may be highly developed or occupied by high value coastal ecosystems, including mangrove and saltmarsh. While dikes have historically been used to drain coastal lowlands and minimise inundation risk, this adaptation is commonly regarded as maladaptive. In the context of Australian estuaries, dikes may promote the leaching of acid from acid sulphate soils into estuarine waters (Russell & Helmke, 2002). Where high value ecosystems such as mangrove, saltmarsh and seagrass are present and are to be preserved, hard engineering adaptation options may become limited. Instead, soft engineering and innovative options may become favourable, and may include:

- Pumping of sediment into tidal waters or onto shorelines to promote accretion on estuarine shorelines (Ford *et al.*, 1999)
- Establishing new tidal wetlands or rehabilitating existing wetlands to bind sediments and attenuate wave action. The term living shorelines is used to refer to vegetated shorelines (Currin *et al.*, 2010).
- Conserving existing wetlands that act as a store for flood waters and buffers nearby coastal lowlands from inundation
- Planting semi aquatic vegetation, such as seagrass, to stabilise sediment and reduce erosion (US EPA, 2009)
- Using natural (or more natural) bulkheads and breakwaters, such as oyster beds and seagrass wrack, to dissipate wave action (Adams & Tyson, 2004), and potential deter public access to sensitive foreshore areas.

Integrated coastal zone management is essential when considering innovative adaptation options for estuarine shorelines and coastal lowlands as innovative options can be multi-faceted and reap multiple benefits. For example, establishing new tidal wetlands or rehabilitating existing wetlands has the resilience building benefit of binding sediments to limit erosion and enhances sediment trapping capacity by attenuating wave action; however, tidal wetlands are also efficient sinks of greenhouse gases and may act to mitigate climate change. Global estimates suggest that, on average, saltmarsh contains 162 tonnes of carbon per hectare in the top metre of soil, while for mangrove this value is 255 tonnes per hectare (Duarte *et al.*, 2013). Market-based carbon trading incentives (Section 5.3.3) promote the value of this adaptation/mitigation measure and these incentive schemes may be used to promote conservation of high value ecosystems or encourage land exchange or buy-back.

5.1.2 ESTUARINE WETLAND ADAPTATION

Adaptation options for estuarine shorelines can be readily applied to promote adaptation of estuarine wetlands, including mangrove and saltmarsh. Innovative adaptation actions can be applied not only to maintain mangrove and/or saltmarsh habitat, but to influence the relative distribution of the two (e.g. to promote conservation of saltmarsh from mangrove encroachment). Specific, innovative adaptation options for mangrove and saltmarsh include:

- Removing barriers to landward migration of tidal wetlands (Lovelock *et al.* 2009).
- Limiting construction of future barriers by establishing setbacks, or buffers and restricting development in areas projected to be prone to inundation and that are likely to support tidal wetlands (Lovelock *et al.* 2009).
- Facilitating wetland accommodation by spraying dredge spoil onto wetland surfaces (Ford *et al.* 1999), or introducing clean sediment from terrestrial sources.
- Altering management of sediment laden waters in the upper catchment. Increasing delivery of sediment to catchment and tidal wetlands will promote accretion and allow tidal wetlands to maintain their position within the tidal prism.
- Building resilience by improving ecosystem health and removing drivers of ecosystem decline (Lovelock *et al.* 2009). This may include improving water quality, removing gross pollution, diverting stormwater from tidal wetlands to estuarine waters, or nutrient enrichment of communities.
- Removing mangrove juveniles from saltmarsh to limit the loss of saltmarsh. This is a controversial option that has only been applied in Australia to promote use of saltmarshes by waterbirds, not as an adaptation measure for sea-level rise.

Coastal adaptation can prevent or delay shorelines from reaching thresholds of change by attempting to maintain coastal geomorphology or minimise morphological change. Coastal adaptation may also be directed towards maintaining ecosystem function and quality. It can be focussed on human well-being by attempting to prevent or delay reaching a threshold of change for human settlements or infrastructure. As coastal adaptation can be directed towards landforms, ecosystems or settlements, adaptation options should be considered within the context of a broad range of management outcomes or management goals, as per the approach of the US Environmental Protection Agency (US EPA 2009).

Some adaptation options may meet multiple management goals (e.g. enhancing sediment budgets will maintain landforms and ecosystems and maintain the status quo for settlements and infrastructure) or may be regarded as “no regrets” options. The selection of adaption options that seek to address multiple management goals, while representing an integrated decision making approach, need careful analysis to assess the benefit and impacts against the various objectives (that may not be all complementary) and from the perspectives of the relevant stakeholders. As with any action, care need to be taken to ensure that the achievement of one goal is not at the expense or a maladaptation of another.. This primarily occurs when adaptation options for ecosystems conflict with adaptation options for settlements and infrastructure, though adaptation options for tidal wetlands may conflict with habitat requirements for seagrass. Careful consideration of management goals is required by stakeholders, taking into account the timeframe of the management response, whether any added benefits can be achieved from adaptation options and potential for maladaptation that may occur.

Table 5.1. Potential adaptation options for estuarine shorelines and tidal wetlands that promote adaptation to sea-level rise and climate change

	Adaptation option	Intention	Potential barriers / constraints
Protect options	Seawalls (vertical rock/geotextile structure)	Protect shorelines from erosion by deflecting wave energy Protect shorelines from inundation	Financial cost; decline in ecological value of shoreline; habitat loss
	Revetment (sloping rock/geotextile structure)	Protect shorelines from erosion by absorbing/deflecting wave energy	Financial cost; decline in ecological value of shoreline; habitat loss
	Bulkhead (offshore rock/geotextile structure)	Protect shorelines from erosion by diffusing wave energy	Financial cost; decline in ecological value of shoreline; habitat loss
	Groyne (shore normal structure)	Protect shorelines from erosion by limiting long-shore drift	Financial cost; decline in ecological value of shoreline; habitat loss; wave deflection and potential for erosion elsewhere
	Natural bulkheads – oyster beds	Protect shorelines from erosion by diffusing wave energy	Financial cost; alteration of ecological value of shoreline; habitat change; cutting hazard
	Natural bulkheads – seagrass wrack	Protect shorelines from erosion by absorbing/deflecting wave energy	Financial cost; alteration of ecological value of shoreline; habitat change; odour from decomposition
	Slip gates / ‘smart gates’	Maintain or manipulate ecosystem structure by controlling tidal inundation characteristics	Financial cost; maintenance requirements; alteration of ecological value of shoreline; habitat change
	Culverts	Maintain or manipulate ecosystem structure by controlling tidal inundation characteristics	Financial cost; maintenance requirements (generally low); alteration of ecological value of shoreline; habitat change
	Removing mangrove juveniles form saltmarsh	Inhibit displacement of saltmarsh by mangrove and conserve existing saltmarsh extent	Financial cost (labour intensive, ongoing); loss of mangrove benefits (e.g. carbon sequestration, high capacity to adapt to sea-level rise); trampling impacts

	Adaptation option	Intention	Potential barriers / constraints
Retreat options	Restrict or prohibit development in erosion or inundation risk areas	Creates accommodation space for landforms and ecosystems to retreat to	Change in public amenity; loss of land for development/activity purposes; change in land value
	Increase shoreline setbacks to match erosion and inundation risk	Creates accommodation space for landforms and ecosystems to retreat to	Change in public amenity; loss of land for development/activity purposes; change in land value
	Land buyback	Creates accommodation space for landforms and ecosystems to retreat to	Financial cost; change in public amenity; loss of land for development/activity purposes;
	Land exchange	Creates accommodation space for landforms and ecosystems to retreat to	Financial cost; change in public amenity; loss of land for development/activity purposes;
	Removing barriers to landward migration	Creates accommodation space for landforms and ecosystems to retreat to	Change in public amenity; loss of land for development/activity purposes; change in land value; third party impacts and potential litigation
	Prohibiting construction of future barriers	Creates future accommodation space for landforms and ecosystems to retreat to	Loss of land for development/activity purposes; change in land value
Accommodate Options	Living shoreline replenishment/nourishment	Maintain shoreline profile within the tidal prism	Financial cost; maintenance requirements; alteration of ecological value of shoreline; habitat change
	Living shoreline establishment/rehabilitation	Creates accommodation space for rising water levels Builds resilience to erosion by binding sediments Builds resilience to inundation by enhancing trapping of sediments	Financial cost; maintenance requirements; alteration of ecological value of shoreline; habitat change; change in public amenity; loss of land for development/activity purposes; change in land value;
	Living shoreline conservation	Creates accommodation space for rising water levels Builds resilience to erosion by binding sediments Builds resilience to inundation by enhancing trapping of sediments	Financial cost; maintenance requirements;

	Adaptation option	Intention	Potential barriers / constraints
Other	Eliminate or reduce non-climate stressors	Build ecosystem resilience	Financial cost; viability
	Remove shoreline hardening that may be maladaptive	Prevents erosion of nearby shorelines	Financial cost; alteration of ecological value of shoreline; habitat change; change in public amenity
	Vegetation rehabilitation/ revegetation to build resilience	Builds resilience to erosion by binding sediments Builds resilience to inundation by enhancing trapping of sediments	Financial cost; maintenance requirements; alteration of ecological value of shoreline; habitat change; change in public amenity
	Remove maladaptive dikes	Create accommodation space for rising water. Create space for ecosystem retreat ecosystems Limit release of acidic waters	Financial cost; alteration of ecological value of shoreline; habitat change; change in public amenity
	Market based incentives such as carbon trading	Incentive and facilitation of accommodation and retreat options	Policy and market uncertainty; potential for perverse environmental outcomes
	Limit/monitor adaptation that is maladaptive for ecosystems	Prevent unintentional maladaptation	Maintenance requirements
	Composite systems	Use a range of options. Aim for no regrets and minimise maladaptation	(As per component elements)

5.2 Recommended management plans/strategy for biodiversity adaptation options for mangrove and saltmarsh of the Sydney Region

5.2.1 ADAPTATION NEEDS

Existing knowledge of stressors and opportunities for coastal wetlands within the Sydney region (summarised in Chapter 1), along with findings of the first pass assessment (Chapter 2) and Wolli Creek SLAMM modelling case study (Chapter 3) have been used to identify the adaptation needs for mangrove and saltmarsh of the Sydney region. This assessment has identified two overarching adaptation priorities: management strategies which accommodate vegetation migration under sea-level rise and the importance of strategies which specifically preserve and accommodate saltmarsh:

Importance of management strategies which accommodate vegetation migration

In the first pass assessment, masking with the 90cm inundation modelling and the current distribution of wetland vegetation in Sydney Coastal Councils Group study area demonstrated the potential for increases in estuarine vegetation extent. Importantly, it also identified the potential capacity for mangrove to migrate landward and longitudinally along estuaries when barriers to the occurrence were limited and accommodation space was available. The SLAMM projected mangrove increases between approximately 280-560% under the various sea-level rise scenarios when wetlands were able to maintain their position in the tidal frame (i.e. when built-up areas were able to convert to other vegetation classes). Critically, however, significant declines in the extent of mangrove (and saltmarsh) were projected when built-up areas were not able to convert to other land classes. This effect was most pronounced under the high sea-level rise scenario, further highlighting the vulnerability of coastal wetlands to sea-level rise when there is limited or no opportunity for vegetation migration.

Importance of management strategies which specifically preserve and accommodate saltmarsh

Comparison of historic and aerial photography has demonstrated a consistent increase in mangrove extent and subsequent saltmarsh decline in intertidal flats across the region (Chapter 1: Table 2; Saintilan and Williams 1999; 2000). The decline has prompted the listing of Coastal Saltmarsh as an Endangered Ecological Community in three NSW Bioregions, including the Sydney Basin bioregion. As changes in relative sea-level are likely to have been an important driver of mangrove encroachment (Rogers *et al.* 2006), further consideration of saltmarsh conservation is appropriate in the context of sea-level rise planning. This is highlighted by findings of the SLAMM case study of Wolli Creek, which demonstrated both the restricted nature of saltmarsh throughout the study region and its reduced capacity to adjust to sea-level rise due to its distribution within a narrow tidal range positioned between the highly adaptive vegetation communities dominated by mangrove and *Casuarina*. Importantly, there was little difference in the outcome for saltmarsh between the model projection with or without coastal squeeze effects, meaning that upslope accommodation options alone may be of limited benefit to saltmarsh extent. Maintenance of the full range of ecosystem services provided by coastal wetlands is dependent upon the spatial context of wetland communities in relation to each other, and habitat complexity, including the retention of saltmarsh. Adaptation actions which specifically enhance the extent and resilience of saltmarsh may be required if the biodiversity values and ecosystem services of coastal wetlands are to be retained.

Based on the above adaptation needs, four adaptation options have been identified:

- Eliminate or reduce non-climate stressors to enhance resilience of existing ecosystems;
- Protection through engineering options (including modification of existing structures);
- Elevation maintenance through sediment nourishment;
- Planning for living shoreline establishment and migration.

Each adaptation option is considered in further detail below, with land use and planning constraints and opportunities around their implementation considered in the following section.

5.2.2 ADAPTATION OPTIONS

Eliminate or reduce non-climate stressors

PRINCIPLES AND PRACTICES

Chapter One described a range of climate and non-climate stressors which can impact upon coastal wetlands, with non-climate stressors particularly pertinent in urbanised and industrialised catchments. It has also been demonstrated that coastal wetlands, including mangrove and saltmarsh, have a capacity to adjust and adapt to changes in sea level autonomously (Kirwan *et al.*, 2016). Autonomous adaptation by ecosystems to climate change and sea-level rise, however, is dependent upon healthy ecosystems, with research showing that maintenance or enhancement of ecosystem health is essential for enhancing the resilience of coastal wetlands (Lovelock *et al.*, 2009b, Rogers *et al.*, 2016). For example, Ellison (2012) suggests that while healthy mangroves enable high levels of sediment accretion, degraded mangroves increase the chances of coastal erosion, especially under sea-level rise. By eliminating or reducing non-climate stressors, current vulnerability to sea-level rise may be decreased, and resilience to future sea level change is enhanced.

Important non-climate stressors include chemical and physical disturbances to mangrove and saltmarsh. Petrochemical spills have the potential to cause dieback of vegetation, as heavy oil slicks smother mangrove pneumatophores leading to asphyxiation (Duke 2016). Such dieback may increase susceptibility to erosion and limit surface elevation gain through root production. Management interventions include reducing the exposure by relocating high risk activities (this has largely happened as oil refining functions have left the region), or, if a spill occurs, ensuring that rapid containment response procedures are in place.

Excessive nutrient loading can have adverse impacts, including mortality and subsequent loss of vegetation (especially during arid periods), or may stimulate rates of microbial decomposition and lead to loss of soil structure (Deegan *et al.*, 2012, Lovelock *et al.*, 2009a, Wigand *et al.*, 2014). Engineering works which improve tidal inundation and flushing of mangrove and saltmarsh (see section 4.2.2) may also reduce the impacts of pollutants and improve ecosystem health. Elsewhere, water sensitive urban design (WSUD), such as bioretention basins, raingardens, permeable paving and constructed wetlands, upstream or upslope of tidal wetlands may be effective in limiting the input of nutrients and other pollutants from catchment sources. It should be noted that interventions which limit sediment delivery to mangrove and saltmarsh may represent maladaptation in regards to sea-level rise impact. Although there may be environmental benefits of limiting sediment run-off (for example, to sediment-sensitive ecosystems such as seagrass and coral reefs), the current focus of minimising sediment delivery to estuarine waterways may not be universally appropriate as terrigenous sediment accumulation is the primary means by which mangrove and saltmarsh gain elevation and resilience to sea-level rise. Given the spatial distribution of coral and seagrass in the Sydney region, an approach that enables sediment delivery may be more appropriate at certain locations.

Saltmarsh vegetation is particularly susceptible to the impacts of physical disturbance, due in part to their fragile vegetation form and open vegetation structure. Physical disturbances to saltmarsh in urban areas may include pedestrian or unauthorised vehicle access, dumping of waste (e.g. construction waste) and mowing at the border of saltmarsh and parklands or residential properties (Laegdsgaard *et al.*, 2009). Damage from vehicle impact has been previously assessed at multiple sites along the Georges River (Kelleway, 2005) and Parramatta River (Kelleway *et al.*, 2007), including within National Park estate, Crown land and council managed lands. Loss of vegetation cover associated with physical disturbances will likely reduce the sediment accretion capacity of a wetland, while changes to hydrology have the potential to increase likelihood of mangrove encroachment of saltmarsh. Management interventions to reduce the impact of this stressor include strategic planning to ensure foreshore access by the public, installation and maintenance of effective vehicle gates around sensitive sites, public education and signage and restoration and/or revegetation of impacted sites.

Illegal cutting of mangroves is an important issue in urban estuary management. The NSW Department of Primary Industries is recommending an urban mangrove policy arising from the Hawkesbury Shelf Marine Bioregion Assessment (NSW Marine Estate Management Authority, 2016). The urban mangrove policy would encourage the development of Mangrove Management Plans for local government areas, though these might be better developed at whole-of-estuary scales. Such plans could help balance the conflicting interests associated with mangrove proliferation, by allowing application for a permit to cut under controlled circumstances consistent with the objectives of the plan. We envisage situations where mangrove growth should be encouraged (for shoreline protection, Blue Carbon benefits, wetland elevation gain, and ecosystem values) and other circumstances where mangrove growth may be detrimental to migratory shorebirds, or recreational and aesthetic enjoyment of the estuary. At present, the effect of mangrove cutting on associated ecosystem services is an under-researched topic.

Physical erosion of tidal flats associated with waterway use has received wide publicity in the Parramatta River. This issue has been managed through the imposition of speed restrictions with the aim of reducing wave height, and to a lesser extent the regulation of traffic. Artificial reefs at the seaward edge of the intertidal flat may be an alternative measure for shoreline protection.

POTENTIAL LOCATIONS

The proximity of Towra Point shipping and refinery operations presents a degree of exposure to petrochemical spills and therefor this risk needs to be proactively managed. The site of the former Clyde refinery in the upper reaches of the Parramatta River has been identified as a future urban renewal area under the Draft District Plan prepared by the Greater Sydney Commission (Greater Sydney Commission 2016). This site and the adjacent estuary is likely to havelegacy contamination from the refinery operations that may impact upon future opportunities to expand mangrove and saltmarsh communities. The Cup and Saucer Creek Wetland, constructed as part of Sydney Water's Bank Naturalisation Project provides a good example of efforts to reduce stormwater pollutants entering the Cooks River. Insights from the planning and management of this site should provide guidance to the future locations in relation to the impact of urban pollution (including excessive nutrients) that are likely to be present across most estuaries.

Locations which have been previously identified as having sustained significant damage from physical impacts include those where vehicle impacts have previously been identified (Kelleway 2005; Kelleway *et al.* 2007) including:

- Still Creek, Woronora River (tributary of Georges River)
- Mill Creek (tributary of Georges River)
- Mason Park, Strathfield (Parramatta River)

Locations which may be impacted by boat wake include:

- Parramatta River – especially the shorelines of the main river channel
- Georges River (including areas of Georges River National Park) (J. Kelleway, personal observation)
- Pittwater – There is a high density of recreational boat use in close proximity to mangrove and/or saltmarsh at a number of locations, including Careel Bay, Bayview, McCarrs Creek and Night Bay
- Port Hacking – Areas which may be particularly susceptible include Grays Point, North West Arm and Kareena Park

At each of these sites which have experienced physical disturbance, enhancements to the resilience of saltmarsh may be made by first ensuring the successful exclusion of disturbance, followed by restoration efforts which promote recolonization of vegetation for impacted areas.

Protection through engineering options

PRINCIPLES AND PRACTICES

Engineered structures such as seawalls and floodgates are often an impediment to mangrove and saltmarsh distribution and function, and may limit the opportunities for upslope or upstream migration under sea-level rise. Structures may also have impacts upon estuarine biodiversity and fisheries production as they can act as a barrier to fauna movement, including access to the nursery and feeding values of disconnected wetlands. The socio-economic amenity of these structures in urban areas is primarily to mitigate flood risk for surrounding areas. There may be instances where engineering options can be effective in moderating the hydrology of a site, with implications on wetland structure (e.g. extent of mudflat versus mangrove versus saltmarsh). The suitability of engineering options will likely be dependent upon the management objectives of a site and its engineering history, so will need to be considered on a site by site basis.

There are a range of structures which have been employed in shoreline protection, including seawalls, revetments, bulkheads and groynes (Table 5.1). Modification of existing structures might also be used to improve ecosystem health and resilience, or to introduce tidal exchange to potential wetland migration locations. For example, strategic design of floodgates or culverts could also be used to obtain specific tidal inundation behaviour for areas currently behind roads, impoundments and levees. Structures may be static or dynamic in their hydrological control, with automated 'smart gate' systems able to open and close in response to water level and/or water quality characteristics (Nichols & Baker, 2013). Increasingly, natural materials such as oyster shells and seagrass wrack are being incorporated into engineering structures to enhance their habitat value and minimise adverse environmental impacts. Living shorelines that incorporate mangrove and saltmarsh vegetation have been used widely in the USA as a means of improving shoreline protection (Gedan *et al.*, 2011). Ecosystem-sensitive structures could be designed so as to maximise the multiple benefits they provide (i.e. maximise habitat value whilst also maximising shoreline protection).

Two case studies are presented below to highlight the suitability of engineered options in mangrove and saltmarsh management, and to highlight the importance of strategic and creative planning in these circumstances. The NSW Government's Saltwater Wetlands Rehabilitation Manual (http://www.environment.nsw.gov.au/resources/water/08555saltwetbk.pdf) provides technical information and recommendations for the planning and implementation of engineering options for saltmarsh and mangrove wetlands.

CASE STUDY – HYDROLOGICAL CONTROL AT SYDNEY OLYMPIC PARK

The Wanngal Wetland of Newington Nature Reserve represents one of the largest mudflat, saltmarsh and mangrove complexes in the Sydney region. This complex is located behind a seawall along the boundary with the Parramatta River which was constructed in the late 1800s (NSW National Parks & Wildlife Service & Sydney Olympic Park Authority, 2003). Now tidal inundation is strategically delivered via a series of weirs and gates, whereby water is delivered to the wetland based on detailed hydrological investigations, to favour saltmarsh and mudflat habitats over the growth of mangrove seedlings (Laegdsgaard *et al.*, 2009).

In 2007 a slip gate and boom were installed in Badu Wetlands (Figure 5.1) with the objective of regulating tidal inundation frequency and duration, and improving wetland health. The boom is primarily designed to prevent objects blocking the slip gate, however, is also effective in restricting movement of mangrove propagules (important if planning for saltmarsh over mangrove) and rubbish entering the wetland (Bowie, 2015a). The slip gate and boom were shown to be successful (particularly the slip gate), with improvements in saltmarsh extent and plant health, decreased algal cover, and an increase in benthic fauna and shore birds (Bowie, 2015a).

Another example within Sydney Olympic Park is the construction of gabion walls along Haslam's Creek, which, along with sediment delivery and revegetation techniques has led to the development of saltmarsh dominated intertidal flats.

Figure 5.1: Slip gate (left) and boom (right) at Badu Wetland (Bowie, 2015)

CASE STUDY - OCEANWATCH OYSTER PROJECTS

Oyster reefs were once dominant in Sydney, but by 1989, there were virtually no oysters left in Sydney Harbour (Birch *et al.* 2013). In a project supported by the Sydney Coastal Council Group and Greater Sydney Local Land Services through funding from the Australian Government, OceanWatch Australia has sourced discarded oyster shells which have been incorporated into ‘oyster bags’. These oyster bags will be deployed to five river sites of Sydney Harbour tributaries as part of the initial trial. While this project primarily aims to enhance habitat values for marine organisms, including oysters, and dependent species (e.g. shorebirds) there will likely be benefits in terms of limiting erosion and potentially promoting intertidal sedimentation. If successful, this approach could be trialled for the purpose of enhancing mangrove and saltmarsh sedimentation and improving resilience to sea-level rise.

POTENTIAL LOCATIONS

There are a number of locations where mangrove and saltmarsh currently exist behind engineered control structures. For example, these include Wanngal Wetland, Badu Wetland and Haslams Creek saltmarsh in Sydney Olympic Park (discussed above), Mason Park wetland on Powells Creek (Strathfield) and a created saltmarsh at Federal Park, Glebe. Continued operation of these structures may allow for the maintenance of existing wetland values under rising sea level, though in some instances modification to structures may be required in response to sea-level rise (including in response to any change in management objectives as sea-level rises). It is advised that these decisions be made on a site by site basis and incorporate an adaptive management approach.

Engineering options may also enable hydrological manipulation for off-channel wetlands within the region. Locations such as Landing Light Wetlands (Arncliffe, Cooks River) and Yeramba Lagoon (Georges River National Park), represent examples where relatively large areas of wetland are currently (Landing Lights) or may be (Yeramba Lagoon) connected to estuarine waters and tidal inundation manipulated. Construction of off-channel wetlands can also serve to provide additional ecosystem services such as water quality improvement and habitat provision.

Elevation maintenance through sediment nourishment

PRINCIPLES AND PRACTICES

Wetlands positioned higher within their optimal tidal position hold greater ‘elevation capital’ (Cahoon & Guntenspergen, 2010, Lovelock *et al.*, 2015) and consequently exhibit greater capacity for survival as sea levels increase since they inherently have a longer period of time before they are submerged. This is particularly important for saltmarsh which exhibits a very narrow elevation distribution and which can be encroached upon by both mangrove at lower elevations and *Casuarina* at higher elevations. Improving elevation capital and maintaining optimal positions within the tidal frame have been enhanced by applying sediment veneers to wetlands, that are typically sourced from dredge spoil (Ford *et al.*, 1999). This technique has yet to be applied in Australia, but is commonly used throughout the USA to improve wetland resilience. For example, the introduction of sediment to Long Island Sound, USA, has promoted colonisation by saltmarsh and facilitated a new, more resilient elevation trajectory (Anisfeld *et al.* 2016).

Within mangroves measured using the surface elevation table-marker horizon (SET-MH) method at Homebush Bay, elevation gain in mangroves is tracking above the rate of sea-level rise recorded by tide gauges. Marsh breakup would only occur once the deficit between elevation gain and sea-level rise ("elevation deficit") continued long enough for the wetland to descend below an inundation survival threshold (i.e. "elevation capital" is exhausted). By applying these estimates of mangrove surface elevation change to Cooks River using SLAMM, chapter three indicated that the current mangrove extent is relatively well placed to maintain its position within the tidal frame through sediment accretion processes.

SLAMM indicated that saltmarsh and *Casuarina* are particularly vulnerable to mangrove encroachment in the absence of sediment nourishment. This is because natural sedimentation does not correspond to the degree of sea-level rise that is projected to occur throughout the 21st century and an elevation deficit becomes increasingly apparent. This elevation deficit within saltmarsh and *Casuarina* places these locations at an elevation within the tidal frame that is particularly suitable for mangrove establishment. As the most landward estuarine vegetation, both saltmarsh and *Casuarina* are poorly-placed to adjust to an increasing tidal frame through horizontal migration to higher elevations due to the high degree of urban development within the catchment. Sediment nourishment within saltmarsh and *Casuarina* may provide the most viable and cost effective means of maintaining the extent and position within the tidal frame for these vegetation communities. Installation of SET-MH stations will provide the information needed to assess the volumes of sediment required into the future to ensure the long-term sustainability of these vegetation communities in situ. Currently there are no SET-MH stations located in *Casuarina* in Australia.

SOURCES OF SEDIMENT

Materials for sediment nourishment could be obtained from several sources, including:

- Dredge materials from the adjacent estuary pending the outcome of toxicology tests. Due to the industrial history of Sydney's estuaries (particularly the upper Parramatta River, Cooks River and sections of the Georges River) estuarine sediments may be contaminated. This will impact on the cost effectiveness of the selected remediation option and the potential for maladaptation to occur ,
- Clean fill derived from building excavation. Under the Draft District Plans for Sydney as released by the Greater Sydney Commission (November 2016) Sydney will experience significant growth resulting in substantial construction of medium to high density dwellings across many parts of the Georges and Parramatta River Catchment. There are several areas in close proximity to coastal wetlands in the Sydney region which are either currently undergoing, or planned for high rise construction. This activity alone will generate significant volumes of sediment. There is an opportunity to strategically consider this resource in the context of climate change adaptation including its use for wetland nourishment. Examples include the Wolli Creek area for the Cooks River and Parramatta, Camellia and Silverwater areas for the Parramatta River and Sydney Harbour. Deep excavation of building sites may provide a suitable, clean fill for sediment nourishment, and a mutually beneficial arrangement could be reached with developers who are seeking a cost-effective way to discard of such materials.
- Offshore sands. These sand reserves have previously been identified as a means for beach nourishment in the region and could equally apply to coastal wetlands. A scoping study undertaken for Sydney Coastal Councils Group provides further details regarding this option (AECOM, 2010).

UNCERTAINTY, ADAPTIVE MANAGEMENT AND MONITORING

There are several important uncertainties regarding the response of coastal wetlands to sediment nourishment. These include uncertainty around future rates of sea-level rise (likely to be non-linear and to vary among estuarine locations), the capacity for applied sediments to be reworked and moved within or outside of the target wetland, and the potential for wetland ecogeomorphic feedbacks (such as positive or negative responses of plant growth to sedimentation). While there is some international scientific literature regarding wetland response to sediment nourishment, no such trials have been undertaken in SE Australian coastal wetlands. It is therefore essential that an adaptive management approach be employed based upon rigorous surface elevation and ecosystem response monitoring. The establishment of one or more demonstration sites would be highly valuable in informing coastal wetland managers, practitioners and scientists about the wetland response to sediment nourishment as well as the merits and challenges that might be faced in applying this approach in other settings.

POTENTIAL LOCATIONS

The suitability of sediment nourishment is likely to vary among sites, their expected exposure to sea-level rise and according to local management objectives. Suitability will also be dependent upon the natural sedimentation characteristics of the site. There is also scope to prioritise sites for sediment nourishment to effectively act as a cap to protect against disturbance and redispersal of contaminated sediments. Sediment nourishment may be particularly relevant where there are limited opportunities for wetland migration, either due to topographic or land use constraints. Examples of locations which might be further investigated include:

Kurnell Peninsula, including Towra Point

The first pass assessment identified that large areas of the Kurnell Peninsula supports estuarine vegetation and is indicated to have low to moderate vulnerability to climate change. Large areas of saltmarsh are included within the wetland complexes, and this location represent an important refuge for saltmarsh now and in the future. As a large coastal embayment dominated by marine processes, there is limited opportunities for deposition of terrigenous sediment and sediment nourishment may become a viable strategy for maintaining the current saltmarsh extent.

Wolli Creek (Cooks River)

Due to the steep terrain and infrastructure constraints (e.g. train line) sediment nourishment may represent the most viable, cost effective option for preservation of wetland values.

Muddy Creek (Cooks River)

There is limited space for inland migration within this region of the creek system. Sediment nourishment to increase the natural accretion of sediment in the mangrove ecosystem will allow the ecological communities on both sides of the creek to match sea-level rise with vertical growth.

Upper Parramatta River estuary and Sydney Olympic Park saltmarshes

The existence of rare and threatened saltmarsh species (most notably *Wilsonia backhousei)*, plus the potential for sea-level rise induced erosion to remobilise contaminated substrates may provide further reason to maintain existing saltmarsh in this region. Without land use change there is also limited opportunity for wetland migration in areas such as Duck River, Silverwater.

Saltmarshes of Middle Harbour, Lane Cove River, Port Hacking and Pittwater

Due to the topography of these narrow drowned river valleys there are limited opportunities for up-slope migration of saltmarsh. Adaptation of these sites to sea-level rise may be improved through sediment nourishment.

Planning for living shoreline establishment and migration

PRINCIPLES AND PRACTICES

Built areas throughout the study area can act as a significant barrier to wetland adaptation and provision of ecosystem services. In the Cooks River SLAMM study (Chapter 4) we explored the implications of maintaining built areas, and this was projected to have an impact on the extent of wetland area by the end of the 21st century under all sea-level rise scenarios. Evidently, these low-lying areas within the study area, such as along Alexandria Canal, and within vicinity of Sydney Airport were identified as particularly vulnerable to inundation and could support significant wetland areas should conversion be facilitated. In Sydney Harbour and its tributaries, seawalls armour in excess of 50% of the shoreline (Bulleri and Chapman 2010), while some shorelines of the Georges River are also armoured by seawalls. In some instances proximity to key infrastructure (e.g. Sydney Airport) may dictate the type of shoreline protection which will be undertaken (e.g. toward hard engineering options). In other circumstances, however, there may be opportunities for the creation of living shorelines which promote mangrove and saltmarsh extent and sea-level rise adaptation.

The utilisation of existing open spaces may provide the most feasible opportunities for supporting future wetland establishment and migration. Such an approach will not be without constraints, however, as demand for community use of open space increases with changes in population density in the Sydney metropolitan region. The buyback of private or leasehold land may be a viable approach, especially for sites that exhibit a high vulnerability to sea-level rise and where conversion of existing open spaces is not a realistic option. While buybacks may involve considerable costs, the high value of coastal wetlands – including their provision of vital ecosystem services (estimated value of "$193,843 USD per hectare per year) provide an economic argument in support of this option. These planning issues are dealt with in further detail in section 4.3, while technical aspects of living shorelines are dealt with here.

LIVING SHORELINES

The term living shorelines is used to refer to vegetated shorelines (Currin *et al.*, 2010). It represents a method for establishing new tidal wetlands or rehabilitating existing wetlands which attenuates wave action, minimises erosion and in the case of mangrove and saltmarsh, may offer shoreline resilience to sea-level rise. To date, the living shorelines approach has not been widely utilised in Australia, despite benefits for coastal resilience as well as also other ecosystem benefits, such as nursery habitat to threatened, commercially and recreationally important species, and carbon sequestration and storage (Davis *et al.* 2015, Gittman *et al.* 2016). The creation of mangrove and saltmarsh living shorelines could be achieved through:

- Replacement of hard seawalls (e.g. concrete, rock) with intertidal flats, allowing colonisation by mangroves and/or saltmarsh
- Removal of concrete-line channels (such as stormwater drains) allowing colonisation of banks by mangroves and/or saltmarsh
- Grading of foreshore lands, whether natural or previously 'reclaimed', such as parkland, golf courses so that they offer suitable topography for colonisation or migration of mangroves and/or saltmarsh
- Living shorelines may integrate well with biobanking activities (discussed below)

Provided environmental factors such as geomorphology, hydrology and biogeochemical conditions (e.g. salinity) are appropriate, mangrove and saltmarsh species generally have a strong capacity to vegetate an intertidal area without active planting methods (OEH 2016). An adaptive approach is recommended, however, as strategic planting and/or vegetation maintenance may be required in some circumstances (Carr *et al.* 2008). Newly established communities are not as resilient as mature mangrove and saltmarsh. Therefore, ongoing monitoring and adaptive management may be required during early phases of wetland development.

UTILISING EXISTING OPEN SPACE FOR LIVING SHORELINES

SLAMM modelling of Wolli Creek and the lower Cooks River highlighted the potential for wetland migration under sea-level rise if land conversion was facilitated. Importantly, only 2.2% of the modelled mangrove extent under the high sea-level rise scenario occurs upon land that is currently developed; the remainder being classified as either undeveloped land (36%) or *Casuarina* woodland (62%). This finding highlights the opportunity for existing open spaces to be managed as and/or converted to living shorelines. For the Cooks River large mangrove gains were modelled around Marrickville where the tidal plane is typically broader than other parts of the Cooks River. Consequently, the spatial distribution of areas projected to support large extents of mangrove occurred in this area and corresponded to the current distribution of sporting fields and golf clubs, including Marrickville Golf Club and Kogarah Golf Club. Many of these recreation areas were built on fill and in the long term with sea-level rise may not be able to serve their recreation function. Additionally, conversion to other land uses, such as residential use, may not be feasible as these areas may be established upon fill-sediments that do not provide foundational support.

There are likely to be a number of locations throughout the region where living shoreline establishment could be prioritised to increase the extent of mangrove and saltmarsh. Further analysis of current elevation capital is the minimum requirement to identify suitable locations for living shoreline establishment. Models of wetland ecogeomorphic change would improve projections of elevation trajectories and facilitate identification of optimal locations for living shorelines.

MANAGING LIVING SHORELINES FOR SALTMARSH

As previously discussed, SLAMM modelling showed high potential for mangrove migration in Cooks River and Wolli Creek, but not for saltmarsh. If saltmarsh is to be promoted and managed for then living shorelines may be modified or manipulated to enhance the extent and resilience of saltmarsh over mangrove. This could be done through strategic profiling of shoreline elevation to favour saltmarsh (for example, Haslams Creek), methods which exclude transport of mangrove propagules to an area, or physical removal of non-saltmarsh species. Removal of mangrove seedlings from locations which are suitable for both mangrove and saltmarsh has been undertaken in some places, primarily to maintain waterbird habitat (Laegdsgaard *et al.* 2009). Such an approach could also be applied to the removal of *Casuarina* as saltmarsh migrates to higher elevations that are occupied by established *Casuarina*, though consideration needs to be given to the values of *Casuarina* woodland, which is itself an Endangered Ecological Community under the *Threatened Species Conservation Act 1995*. It should be noted that seedling removal – whether mangrove or *Casuarina* – is not as an adaptation measure for sea-level rise, but instead a tool for maintenance of existing wetland structure and values.

PROPERTY BUY BACK, BROWNFIELD CONVERSION AND PRIVATE-PUBLIC PARTNERSHIPS

The strategic buy back of property, land exchange or a development agreement can be used to facilitate the conversion of shoreline areas to living mangrove and saltmarsh shorelines. While these options may invoke considerable financial costs and land use planning challenges, they can have important socio-environmental benefit, such as expanding the public open space network and linking to blue and green biodiversity grids (e.g. Sydney’s Green Grid as identified in the draft district plans for Sydney (Greater Sydney Commission 2016)). Buy back or development agreements that trade the use of land with limited or restricted development potential for socio-ecological purposes could link to the liveability and sustainability agendas currently pursued by metropolitan, district and local planning for Sydney and complement other cross agency and council initiatives such as the Swim in Parramatta River project. The creation of living shoreline through property buy-back, land exchange or a development agreement may incur lower costs in the longer term relative to hard engineering options that require maintenance under higher sea-level scenarios.

Following buy back, shoreline conversion can be undertaken through active (i.e. re-profiling of shoreline and/or alteration to local hydrology) or passive (i.e. allowing existing shoreline to be inundated by rising sea-level). Given that most buy back properties will be highly modified shorelines, it is likely that some degree of intervention will be required to achieve the most effective biodiversity and sea-level rise adaptation outcomes of these new living shorelines.

Other land use planning incentives can be applied as part of a development agreement that may include floor space bonuses, whereby developers are entitled to planning provisions (such as an increase to number of levels allowed in their development consent) in return for provisions which improve environmental or community benefits related to the project. In this case that may be the conversion of a portion of their landholding to living shoreline and/or the implementation of water sensitive urban design which improves the sea-level rise resilience of adjoining wetlands. Private-public partnership arrangements around the supply of clean fill for local sediment nourishment works may also incentivise developers to engage with living shoreline management (discussed in section 4.2.3).

Current planning priorities for the Sydney region identify the industrial foreshores the Parramatta and Duck Rivers at Camellia, Silverwater, Rydalmere as a region for which 'brownfield conversion' to living shoreline may be particularly suited (refer to section 5.3).

5.3 Opportunities for strategic land use and planning

There are existing legislative controls within New South Wales legislation which regulate activities and development impacts upon mangrove and saltmarsh. Both ecosystems are considered 'marine vegetation' and are therefore regulated by the *NSW Fisheries Management Act 1994*, while saltmarsh is also listed as an Endangered Ecological Community under the *NSW Threatened Species Conservation Act 1995*. In addition, 'coastal saltmarsh' is listed throughout sub-tropical Australia (i.e. across the entire NSW coast) as a Vulnerable Ecological Community under the *Commonwealth Environment Protection and Biodiversity Conservation Act 1999*. While environmental and planning laws and instruments aim to reduce loss of mangrove and saltmarsh, among other objectives, they do not contain provisions for the management of these ecosystems under climate change or sea-level rise. Through the NSW Coastal Planning Guideline: Adapting to Sea Level Rise local Councils are encouraged to give local sea-level rise projections due and proper consideration as part of their strategic and development assessment process. The guideline also states that strategic planning should address and accommodate the effects of sea-level rise on mangrove and saltmarsh, while maintaining amenity and access of public open spaces.

5.3.1 REGIONAL STRATEGIC PLANNING OPPORTUNITIES

There is an important role for strategic planning in accommodating future urban growth in a sustainable manner. In the context of the Sydney Coastal Council Group, this planning, however, needs to occur across multiple scales, from the regional scale, down to the individual wetland site. At the state to regional scale there are several significant planning documents which are either currently in review, forthcoming or have been recently enacted. Elements of these documents which might be harnessed to incorporate coastal wetland adaptation planning within the Sydney region are briefly reviewed here:

NSW COASTAL REFORMS

The Department of Planning and Environment, together with the Office of Environment and Heritage, is developing a new coastal management framework, following the recent creation of the *Coastal Management Act 2016.* The Draft Coastal Management State Environmental Planning Policy (SEPP) and draft maps of coastal management areas are current under public consultation. This SEPP will replace existing coastal SEPPS (including SEPP 14 Coastal Wetlands) and now includes wetlands within Sydney, but will also include development consent procedures for coastal protection works, regulate development on certain land within coastal vulnerability, coastal environment and coastal use areas.

GREATER SYDNEY REGIONAL PLAN

In November 2016, the Greater Sydney Commission released a draft amendment to the Greater Sydney Regional Plan. Entitled 'Towards our Greater Sydney 2056' the draft identified three major centres (with both central Sydney and Parramatta within estuarine zone) and six distinct planning districts (with four of these – North, Central, South and West Central containing the vast majority of Sydney's estuaries).

Sustainability priorities are focussed on maintaining landscapes, protecting waterways, protecting and enhancing biodiversity, building up our resilience against climate change and creating a 'Green Grid' across Greater Sydney. Two central tenets of the 'Green Grid' approach are: 1) to integrate management of the coast and the land; and 2) improve public access to waterway foreshores, wetlands and riparian corridors.

More specifically, the Draft South District plan identifies three priority Green Grid projects, all of which have implications for wetland adaptation

- Cooks River Open Space Corridor – aims to become a regionally significant parkland corridor, with improved water quality and high quality open spaces linking Strathfield and Sydney Olympic Park, through Canterbury, Marrickville and Wolli Creek.
- Wolli Creek Regional Park and Bardwell Valley Parkland – aims to provide open space for recreation, walking and cycling trails, connected patches of ecologically significant vegetation and improved water quality and stormwater management.
- Salt Pan Creek Open Space Corridor – aims to enhance access to the corridor and improve trails and recreational opportunities,

The Draft Plans for the North, Central and West Central districts emphasised the preservation of Sydney Harbour's foreshores and waterways. This has specific relevance to the importance of mangrove and saltmarsh resilience planning:

> *Greater Sydney's major waterways would benefit from clear strategic planning to guide how the waterways are protected, enhanced and enjoyed. Many waterways are managed by a range of stakeholders and we have a clear role in facilitating collaboration between stakeholders. We will explore new forms of governance arrangements for the 'Blue Grid' of waterways in this regard during the review of A Plan for Growing Sydney in 2017.*

The Central District Plan makes a special mention of maximising benefits to the public from the innovative use of golf courses setting the action to *identify opportunities for shared golf courses and open space*. Where golf courses abut waterways, opportunities exist to integrate the natural environment, including saltmarsh and mangroves, into the design of these courses into the future.

The district plans, once finalised, will provide the direction for the revision of local government planning via their Local Environment Plans (LEPs) and also the revision to Sydney's metropolitan plan, A plan for Growing Sydney (as below). At both scales specific sites or areas and associated development controls could be drafted to provide explicit protection for estuarine, saltmarsh and wetland areas. It is also envisaged that catchment based controls, for example requiring water sensitive urban design outcomes for residential and commercial developments, could be implemented to reduce pressures on existing waterways and their environments.

NSW GOVERNMENT'S 'A PLAN FOR GROWING SYDNEY'

It is intended that A Plan for Growing Sydney will guide land use planning decisions for the next 20 years. It contains three directions in relation to sustainability:

- Direction 4.1: Protect our natural environment and biodiversity
- Direction 4.2: Build Sydney's resilience to natural hazards
- Direction 4.3: Manage the impacts of development on the environment.

Specific priorities relating to mangrove and saltmarsh adaptation include:

- Improve the health and resilience of the marine estate (including mangrove and saltmarsh)
- Protect internationally significant wetlands and migratory birds in the Towra Point Nature Reserve Ramsar Wetlands.
- Conserve the natural environment of the Kurnell Peninsula and encourage development that respects the environmental, cultural and economic significance of the area
- Work with councils to protect and improve the health of waterways and aquatic habitats including Parramatta River, Georges River and the South Creek sub-catchment of the Hawkesbury-Nepean Catchment
- Protect internationally significant wetlands and migratory birds at Homebush Bay.
- Work with councils to provide for improved access to the Parramatta River foreshore including the walkway from Ryde to the head of the River.
- Establishing a new 'Priority Growth Area' extending from Greater Parramatta to the Olympic Peninsula to develop more residential, commercial and public space within the western suburbs to accommodate for rising populations in the Sydney region

Together, the directions and priorities identified in the above plans highlight a planning emphasis upon population density increases in areas near Sydney's estuarine waterways, the improvement of waterways and foreshore areas (including importance of public access and amenity), but also the preservation of existing, high value wetlands in the Sydney region and the integration of waterway and land management. These priorities are overwhelmingly positive toward the protection of coastal wetlands, their resilience and potential for adaptation under sea-level rise. However, increases in population densities and the focus on foreshore amenity may offer challenges for implementation of some of the adaptation options discussed in section 4.2. These challenges need not be a deterrent to undertaking wetland accommodation, but they may require innovative and creative planning solutions to achieve mutually beneficial outcomes.

5.3.2 PRIORITY AREAS FOR STRATEGIC PLANNING TOWARD WETLAND ESTABLISHMENT AND ADAPTATION

On the basis of focal areas in this study and consideration of planning priorities in the Sydney region the following areas have been identified as examples for strategic planning that accommodates mangrove and saltmarsh establishment and adaptation.

COOKS RIVER AND WOLLI CREEK OPEN SPACE CORRIDORS

This project identified some scope for wetland expansion within the lower Cooks River and Wolli Creek sub-catchments, dependent upon the capacity for land use change or complementary land uses (e.g. co-existence of mangrove and parkland). The draft 'Towards our Greater Sydney 2056' plan by the Greater Sydney Commission also identified both the Cooks River Open Space Corridor and the Wolli Creek Regional Park (and Bardwell Valley Parkland) as high priorities in Sydney's 'Green Grid.' These priorities aim to create high quality open spaces, access for recreation, connection of ecologically significant vegetation and improvements to water quality and stormwater management. Within this vision there are opportunities for the management and adaptation of existing mangrove and saltmarsh areas, and the opportunity to create living shorelines of mangrove and saltmarsh along the foreshores of both waterways. Given the priorities of open space access and recreation opportunities, it is important that these living shorelines maintain or improve existing foreshore paths and are not considered detrimental to public amenity or the aesthetic values of these paths. Thoughtful design and planning of living shorelines to this end may include designs which promote saltmarsh establishment over mangrove in areas where maintenance of a line of sight and waterway views are prioritised.

REPURPOSING OF INDUSTRIAL FORESHORES THE PARRAMATTA AND DUCK RIVERS (CAMELLIA, SILVERWATER, RYDALMERE)

Recent planning documents highlight the potential for significant change in land use and planning of the upper Parramatta River estuary, including the tributary of Duck River, in alignment with the growing importance of Parramatta as a 'central' Sydney city. Specifically, the NSW Government's A *Plan for Growing Sydney* states a priority to "investigate urban renewal options in Camellia and develop a structure plan to guide future development." More specifically, the Camellia Precinct Land Use and Infrastructure Strategy (2015) is a current proposed plan in which identifies land suitable for rezoning, with The Camelia Masterplan classifying much of the area as a 'proposed business orientated land use transition zone' (NSW Department of Planning and Environment 2015).

The repurposing of Camellia, but also former and current industrial areas of Silverwater and Rydalmere, presents significant opportunities to improve the extent and resilience of mangrove and saltmarsh in the region and incorporate living shorelines which promote their adaptation under sea-level rise. Rezoning of foreshore and low lying areas could be used to increase the horizontal space for wetland migrations and to establish living shorelines. While this may reduce the footprint available for development, this will predominantly be in areas of already high inundation and flooding risk. Further, an effective living shoreline will act as buffer to sea-level rise for surrounding development areas. Development of living shorelines will also increase the extent of open green space, which is currently lacking in the area and could be integrated with plans to increase foreshore access and amenity.

SYDNEY OLYMPIC PARK AND SURROUNDS

Sydney Olympic Park contains the second largest area of mangrove-saltmarsh complex in the Sydney Region (after Towra Point Nature Reserve in Botany Bay). This area has also been identified as an area of high population growth. For Sydney Olympic Park, the Master Plan 2030 is the relevant document that assists in achieving the goals outlined in A Plan for Growing Sydney 2014. The Master Plan came into effect in 2010 and has recently undergone review. The revised plan proposes 1,960,000 m^2 of total additional development area, including 10,700 new dwellings for 23,000 residents, 34,000 professionals and 5,000 students, as well as 100,000 m^2 of commercial space (SOPA and NSW Department of Planning and Environment 2016).

The Master Plan states that 'environmental and recreational values' of the parklands will be preserved and enhanced, and all future developments must incorporate sea-level rise projections. For wetland areas, the Plan adopts the provisions of the Sydney Olympic Park Parklands' Plan of Management 2002. The Sydney Regional Environmental Plan No 24 - Homebush Bay Area guides development of the Homebush Bay Area and thus also facilitates development within the Sydney Olympic Park precinct. According to the SEPP, developments have to consider wetland viability and minimise adverse impacts on wetland habitats. Any urban renewal in close proximity to environmental conservation areas need to be outside a 30 metre environmental buffer zone. Thus, developments in environmental conservation areas will not be approved if they would reduce the ecological value of the area (SEPP 24, 2014).

An increase in the population density of the Sydney Olympic Park region, in concert with land use change and increasing population densities in the corridor between Sydney Olympic Park and Parramatta has implications for wetland adaptation planning in the region. Whilst there are several open space areas within this corridor (e.g. Wilson Park, Blaxland Riverside Park on the southern shore and George Kendall Reserve and Meadowbank Park on the northern shore) which might otherwise be suitable locations for living shoreline establishment and wetland retreat, there will be increasing demand on these spaces for recreation and public amenity. This means that the design of living shoreline and wetland retreat options will need to be complementary to public amenity needs. Options may include living shoreline designs which promote saltmarsh over mangrove (to maintain access and line of site to waterways and aesthetic amenity), and/or the identification of alternative options for recreation in the area (such as alternate playing fields or installation of artificial turf to enhance their carrying capacity).

KURNELL PENINSULA, INCLUDING TOWRA POINT

The Kurnell Peninsula contains the largest areas of mangrove and saltmarsh in the Sydney region, and is therefore a high priority for conservation and management. 'A Plan for Growing Sydney' specifically refers to priorities of improving the health and resilience of the marine estate, protecting the internationally significant wetlands and migratory birds in the Towra Point Nature Reserve Ramsar Wetlands, and conserving the natural environment of the Kurnell Peninsula.

PITTWATER

The steep topography of much of the Pittwater shoreline limits the current distribution of mangrove (17.5 ha) and saltmarsh (2.7 ha). This topography also limits the extent of suitable mangrove and saltmarsh habitat under sea-level rise scenarios. Nevertheless, the first pass assessment (Chapter 3) shows moderate vulnerability including the largest area of mangrove and saltmarsh at Careel Bay, but also Winnererremy Bay (Bayview) and behind Great Mackerel Beach. This assessment also showed potential increases in estuarine wetland area under the 90cm SLR projection (albeit with moderate vulnerability) in Winnererremy Bay and the northwest of the estuary (Great Mackerel Beach, Currawong Beach and The Basin).

Northern Beaches Council currently has a 'Pittwater Waterway Review' underway. This review will consider how to manage competing demands including growth in boat ownership and demand for associated infrastructure, public access to the foreshore and waterway, protection of flora and fauna, recreation, and governance and planning controls, amongst others. This review presents an opportunity to assess future options for mangrove and saltmarsh in the Pittwater estuary.

PORT HACKING

At present, Port Hacking contains a high proportion of saltmarsh (12.8 ha) – relative to extent of mangroves (29.9 ha) – compared to most estuarine areas of Sydney (Table 2.1). Protection of existing saltmarsh areas through elimination or reduction of non-climate stressors is therefore an important first step in Port Hacking. Some of the largest areas of mangrove and saltmarsh within the estuary – including Cabbage Tree Basin, South West Arm, and opposite Grays Point – occur within or bordering Royal National Park. While the public ownership and undeveloped nature of these lands eliminates the need to buy backs or related mechanisms, the conservation status does not prevent degradation or loss of existing areas from either SLR or non-climate stressors. Swallow Rock Reserve (Grays Point) is an important area of tidal wetland outside of the conservation reserve, with moderate vulnerability. Proximity of this site with other wetland areas on the southern shoreline (i.e. within Royal NP) prompts co-ordination of future management with NSW National Parks and Wildlife Service, and potentially the State government's AdaptNSW initiative. 'A Plan for Growing Sydney' identifies the need to protect and enhance the biodiversity and health of waterways, including Port Hacking, as well as identifying Port Hacking as a major tourism asset.

5.3.3 MARKET BASED INCENTIVES

There is growing global interest in the utilisation of market-based schemes to facilitate tidal wetland establishment, restoration and conservation. This is largely based upon the emerging knowledge that 'Blue Carbon' ecosystems (that is, mangrove, saltmarsh and seagrass) may be stores of significant preserved carbon stocks (mostly in their soils and belowground biomass) and may accumulate carbon at a faster rate than most terrestrial ecosystems (McLeod *et al.* 2011).

Although 'Blue Carbon' is not currently captured within the Commonwealth government's national accounting processes, there are indications that this may change in the coming years. The Commonwealth government is currently investigating mechanisms for inclusion both within its National Greenhouse Gas

Inventory (NGGI) and its Emission Reduction Fund (ERF), while its role in the recent establishment of the International Blue Carbon Partnership (IBCP) also points to a promising future.

Outside of government, there are voluntary market frameworks under which credits from Blue Carbon activities (such as mangrove and saltmarsh restoration and preservation) can be traded. In Australia, there are numerous non-government organisations which undertake revegetation works in terrestrial ecosystems to generate carbon credits, though uptake within Blue Carbon ecosystems is still in its infancy. Recent developments in accreditation for Blue Carbon projects through the Verified Carbon Standard (VCS) scheme are likely to increase uptake of carbon credit projects in mangrove and saltmarsh.

While market-based schemes may offer promising incentives for wetland preservation, restoration and creation, their uptake can potentially lead to some unplanned consequences. This is because market-based mechanism often relate to only one of the multiple ecosystem services which an ecosystem might provide. In the example of a Blue Carbon scheme, there may be high uptake of mangrove restoration relative to saltmarsh, due to expectation of higher carbon returns from the former. This gives little consideration of ecosystem benefits of saltmarsh, such as its unique habitat for a range of fauna. Therefore, an integrated approach to coastal wetland management is required to ensure that consideration is given to the benefits and conservation needs of each ecosystem type.

BIOBANKING

Biobanking is a market-based instrument implemented in 2008 under the Threatened Species Conservation Act 1995 and Threatened Species Conservation (Biodiversity Banking) Regulation 2008 designed to conserve biodiversity values (note that this Act and regulation will be replaced in 2017 by the Biodiversity Conservation Act and associated Regulation).

Biobanking was developed to provide protection for sites in perpetuity that are determined to have important conservation outcomes. It can apply to public and private land and may be suitable for existing natural areas along the Parramatta, Georges and Cooks River for future ecological outcomes. The scheme is based on a credit system where land of high biodiversity value can be added to the biobank and the credits can then be sold to a developer which allows the development to directly impact the ecological values upon the land to be developed. Rules for biobanking are set by the NSW Government and have been designed to ensure the scheme is measurable, consistent, secure, transparent and strategic (NSW Department of Environment and Climate Change 2007). These rules may require revision if certain sites are to be protected and or managed for transitional endangered ecological communities as an adaptive approach to rising sea levels associated with climate change. The amount of credit will depend, in part, on the management actions and their benefit to threatened species or populations. A biobanking agreement is registered on the land title and binds current and future owners of the site and ongoing review of compliance is undertaken as part of a compliance assurance strategy overseen by the NSW Government.

Biodiversity credits are based on a principle to improve or maintain biodiversity values. For example, greater credits are given to larger areas of habitat, areas forming biocorridors (as opposed isolated pockets) and sites with a healthy condition. The biodiversity credit system also provides a mechanism to allocate funding for the ongoing maintenance and improvement of a site.

5.3.4 COMMUNITY ENGAGEMENT

One of the key planning challenges identified within this report is addressing the potentially conflicting priorities of utilising existing open space for wetland sea-level rise adaptation and the goals of increasing public amenity and recreational use of Sydney's urban, foreshore areas. While several complimentary strategies have been identified, community engagement with wetland adaptation strategies is essential to their success. Ellison (2012) suggests that community engagement is one of the more critical elements to protecting mangroves as it helps to shape the wider social attitude towards mangroves to one which favours their protection. Community engagement has a role in reducing non-climate stressors, thereby building mangrove and saltmarsh resilience to sea-level rise, but also in the acceptance of adaptation actions which may be confronting (e.g. sediment nourishment) or be perceived to conflict with public amenity (e.g. living shoreline establishment). A relevant example is the Swim in Parramatta Initiative which explicitly identifies community benefits – better public amenity, healthy living, increased property value – associated with what is essentially a water-quality initiative.,

Public awareness and education is an important first step for all adaptation options. In the case of sediment nourishment, public awareness will help to allay concerns regarding the source or potential toxicity of sediments and to demonstrate its public benefit. The public benefits of living shoreline establishment could be integrated within educational resources that will need to be developed as part of the Cooks River and Wolli Creek Open Space Corridors. Similar educational resources could be included as public information is disseminated relating to the rezoning and repurposing of foreshore lands (e.g. Camellia and surrounding Sydney Olympic Park). Many coastal councils with the Sydney region will have existing resources regarding the values and management needs of mangrove and saltmarsh (such as information sheets and signage). Sydney Olympic Park Authority wetland education unit also has educational resources in this regard, which could be utilised for more specific purposes. Development of a Cooks River Wetland Education Centre is one option which could specifically target community engagement and education for future wetland adaptation opportunities in this area.

There is also the potential to directly engage members of the community with wetland adaptation options and their monitoring. Existing community groups might be engaged in the first instance, though there may also be opportunities to resource groups specifically concerned with mangrove and saltmarsh adaptation. For example, in New Zealand, several 'Estuary Care' groups have been established which are made up of both community and organisation members who work in conjunction with management bodies, such as local councils, to ensure the health of estuarine ecosystems (Harty 2009).

5.4 Knowledge Gaps and Future Research

Although strategic and practical adaptation options have been identified, there remain significant knowledge gaps which should be addressed. Recommendations for addressing these knowledge gaps are presented here.

5.4.1 SPATIAL AND TEMPORAL FRAMEWORK FOR FUTURE RESEARCH

Coastal processes operate at a range of temporal and spatial scales and coastal vulnerability assessments need to be scale sensitive. A scaled approach will facilitate prioritisation of areas for high resolution assessment and improve future planning guidance. Planning, engineering and management requires information projected at scales of tens to a few hundred years. This scale requires the integration of information across a range of spatial and temporal scales.

Rogers and Lovelock. (2014) and Sharples *et al.* (2008) advocate a three-tiered approach to coastal vulnerability assessment that is defined on the basis of spatial and temporal scales. First order assessments capture the vulnerability of a region and are ideal for prioritizing areas for high resolution analyses of vulnerability. First order assessments of biophysical vulnerability can be integrated with socio economic information to provide a more holistic first pass assessment of the vulnerability of communities, landscapes and ecosystems to climate change. In this project the first pass assessment constituted a broad scale assessment that integrated long-term geomorphic information across the Sydney Coastal Councils Group region to provide a first order assessment of vulnerability. There remains scope to improve the biophysical aspects of this assessment and integrate the outcomes with socio-economic indicators.

Third order assessments are resource intensive but provide process-based information that can be scaled up to second order resolution. Currently there is limited third order information that is spatially relevant to the Sydney Coastal Councils Group region. In this study, information regarding surface elevation trajectories and vertical accretion have been extrapolated from Homebush Bay to the Cooks River; the validity of this extrapolation requires confirmation. The availability of third pass data at an appropriate spatial scale remains a significant information gap for the Sydney Coastal Councils Group region.

Second order assessments are relevant to the catchment to sub-catchment scale; and consequently are appealing for natural resource management. The SLAMM constitutes a second order assessment that integrates broad-scale information (largely geomorphic in nature and partly integrated in digital elevation models and vegetation distribution), with high resolution information from Homebush Bay regarding surface elevation change and vertical accretion to project future wetlands distribution. There remains scope to improve both the first and second pass assessments applied in this project. This will largely be achieved by the provision of information at the third pass scale.

Planning and management integrates with this three-tiered approach. The scale of second order assessments corresponds to the scale at which regional scale strategic planning occurs and will complement decision making for these purposes. The scale of third order assessments corresponds to the scale at which site-specific activities may occur, such as sediment nourishment, mangrove removal or establishment of living shorelines, and the repurposing of private lands to wetlands and the ecosystem services they provide. In addition, third order information is essential for second order assessment of vulnerability and for improving strategic planning activities at the catchment to sub-catchment scale.

There remains significant scope to improve the resolution and accuracy of assessments at all scales; however key limitations and knowledge gaps remain that are particularly relevant to assessments occurring at the second and third order. Many of these have been detailed in chapter three and include:

- Digital Elevation Model (DEM) validation and correction
- Ground-truthing of vegetation distribution patterns
- Spatially explicit accretion and elevation change data.
- Validation of tidal plane modelling
- Validation of model
- Uncertainty analysis

Ideally, the above steps should be undertaken prior to broader assessments of sea-level rise impacts in the Sydney Region. Once completed, detailed investigation (including SLAMM modelling) of areas identified as priorities for strategic planning which were not undertaken in this report (i.e. The Parramatta and Duck Rivers at Camellia/Silverwater/Rydalmere, the vicinity of Sydney Olympic Park, the Kurnell Peninsula including Towra Point, Pittwater and Port Hacking) should be prioritised.

In a highly urbanised landscape, human interactions with the environment are significant. The approach described above does not prioritise land use types, facilitate complementary land uses or incorporate socio-economic factors and values. Incorporation of vulnerability assessments with socio-economic factors and incorporation of realistic planning outcomes (e.g. classification of potential land-use change based upon social value) using integrated modelling approaches are essential for improving strategic planning and wetland management at the local scale.

5.4.2 FURTHER STUDIES IN SUPPORT OF THE (PROPOSED) URBAN MANGROVE POLICY

The current proposal, as foreshadowed under the urban management policy, to develop Mangrove Management Plans as a context for the granting of cutting permits is potentially controversial, and further research in support of this policy is warranted. In particular, implementation of the policy would benefit from:

- Controlled trials of the effect of pruning on ecosystem services including Blue Carbon sequestration, utilisation by fish and crustaceans, utilisation of mangrove and surrounding habitat by shorebirds
- Historic analysis of mangrove and saltmarsh distribution in urban catchments as context for the development of management plans.

5.4.3 FURTHER STUDIES IN SUPPORT OF SEDIMENT NOURISHMENT STRATEGIES

Given the limited potential for landward encroachment, the relatively modest area of existing intertidal wetland, and the capacity of intertidal wetland for vertical elevation gain, sediment nourishment has emerged in this review as a potentially cost-effective mechanism for the preservation of existing intertidal wetland values in the Sydney urban catchments. However, this is an untested methodology in Australia and the development of the approach would benefit from the following lines of experimentation:

- As proposed above, improved digital elevation modelling is a first step towards determining the amount of "elevation capital" in the region's wetlands, and therefore how much time is available before this capital is exhausted under an accretion deficit. Also, consistent with the above, further SET-MH installation would provide more accurate information on the rate of surface elevation gain, and the nature of accretion deficits as they develop over time.
- Trials of the ecological impacts of sediment nourishment on mangrove and saltmarsh vegetation would be appropriate. The suggested regime of 5cm burial could be tested in controlled settings (including a comparison of estuarine and terrestrial sediment), with measures exploring the impact on vegetation health, and benthic faunal communities. Incorporation of SET-MH stations in these observations would allow the long-term elevation contribution of sediment input to be measured.

5.4.4 ESTABLISH MECHANISMS TO MONITOR AND EVALUATE ADAPTATION OUTCOMES

Most of the adaptation options recommended within this report have not been previously undertaken in SE Australian mangrove or saltmarsh to date. While this may introduce uncertainty regarding the ecosystem response to management options, it also provides an opportunity to gather knowledge and demonstrate options. The monitoring and evaluation of wetland response to sea-level rise at sites within the Sydney region will also compliment investigations undertaken to date within Sydney Olympic Park (Rogers & Saintilan, 2008, Rogers *et al.*, 2005, Rogers *et al.*, 2006) and Towra Point Nature Reserve ((Kelleway *et al.*, 2016; Kelleway *et al.*, in prep.).

Given existing knowledge gaps it is of great importance that adaptive management approaches are undertaken, based upon rigorous surface elevation and ecosystem response monitoring. The establishment of demonstration sites would be highly valuable in informing coastal wetland managers, practitioners and scientists about wetland response to management actions as well as the merits and challenges that might be faced in applying these in other settings.

It should also be noted that demonstration projects will be most instructive if baseline (i.e. pre- adaptation implementation) monitoring and evaluation is maximised. Utilisation of suitable reference locations will also enhance understanding of sea-level rise impacts and wetland response with and without adaptation options.

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