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Benefits, co-benefits and policy implications of restoring coastal wetlands in Auckland: a climate change perspective

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Abstract

The thesis assesses whether and how to integrate the mitigation and adaptation benefits of coastal wetlands into urban land use and land use change decision-making. The case study is that of Auckland, New Zealand, which has a high proportion of coastal wetland relative to the land area.

To address the thesis questions, this research reviews the current knowledge of coastal wetland carbon sequestration and storage (CS&S) capacity, as well as the capacity of wetlands to assist cities to adapt to the effects of climate change by enhancing urban resilience. This data is used to assess firstly, whether these capacities are worth investing in for climate change management purposes, relative to other opportunities. The thesis then examines how the climate change values of coastal wetlands could, or should, inform land use decision-making, including resource consent processing. It is partly informed by the insights gained from the global case studies, as well as the theories and principles that underpin the research idea and questions.

While uncertainty remains in the estimates of CS&S for Auckland’s coastal wetlands due to limited data, this study suggests that a considerable amount of carbon is stored in the sediment of coastal wetlands and can provide potential for mitigation-adaptation outcomes, if properly managed. From a planning perspective, this research demonstrates the need to account for the climate change benefits of coastal wetlands when formulating and implementing land use policies. While the precise benefits may be difficult to quantify, the fact that these benefits exist are clear.

This research is theoretically structured around the key concepts of resilience, ecosystem-based adaptation, climate compatible development and co-benefits which guide the analyses, arguments and recommendations in this research. A framework is proposed in this research for coastal protection and resilience planning in Auckland which draws on the multi-functionality of coastal wetlands and provides for vulnerability mapping and ecological compensation and offsetting as possible planning tools.

Overall, the findings are intended to inform the process of mainstreaming coastal wetlands and their climate change benefits into the land use planning and policy making processes.

Keywords: resilience, climate change, ecosystem-based adaptation, climate compatible development, coastal wetlands, co-benefits, ecological compensation, biodiversity offset.
Dedication

To myself for my hard work, devotion and perseverance;

To my loving mother, father, sisters and brother for they have always been there for me,
and for their endless love and all their continuous support and encouragement;

To my better half, for his insights, constant support and patience that cannot be expressed
in words;

and

To my beloved mother-in-law in her loving memory for her gracious encouragement at the
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1. Introduction

1.1. The idea and focus

When I started this thesis in 2012, there was a growing interest in understanding the role of urban systems for climate change mitigation and adaptation particularly with respect to their potential to act as sinks of carbon (Wright, et al., 2011; Dhakal, 2010; Lal & Augustin, 2011; Otto-Zimmermann, 2011; Bulkeley, 2013; Broto & Bulkeley, 2013). The focus was mostly on understanding the urban carbon dynamics relating to the potential sources and sinks of carbon in urban landscape and how human-induced landscape transformations influence the flux and pools of carbon (IPCC, 2000a; Mustard, et al., 2012; Lal & Augustin, 2011; Brown, et al., 2013). The transformation in urban landscapes refers to human-induced land cover and land use change that involves shifting urban landscapes into different categories of land characterised by some defined parameters. The premise here is that urban land use decisions that lock urban development into a certain pathway could influence carbon sequestration and storage (CS&S) potential and emissions profiles of urban landscape (Mustard, et al., 2012; Houghton, et al., 2012).

My preliminary research (See Chapter 4 for the results) suggested it was possible to categorise different land uses according to such factors as vegetation and soil types and therefore assess changes in carbon sequestration associated with a shift from, for example, urban turfgrass to trees, or by incorporating green space into an urban area. I initially started a review to explore the possibility of designing a decision-making process that could guide land use planners on how to account for the climate change mitigation implications of changes in land use categories. This was because my initial research indicated there was no framework available to assist planners in making such decisions.

For the reasons discussed in Section 1.4, I chose the Auckland region in New Zealand as my case study. I then used the available local and global data on carbon sequestration by different land use categories in urban areas to test the possibility of making reliable estimates of the average rates of carbon sequestration for the land use categories. This preliminary investigation (Chapter 4) identified that using the existing data may provide decision-makers with an insight into how changes between land use categories may result in an increase or decrease in carbon sink capacity of a given urban landscape. However, I concluded that these estimates cannot be used to accurately quantify the scale and magnitude of the changes due to local variations in site-specific parameters.

The preliminary analysis did also confirm that globally wetlands have a relatively high rate of carbon accumulation in their sediments compared to other terrestrial land use categories. These findings were consistent with the emerging recognition in the literature of the climate benefits of coastal wetlands (Mcleod, et al., 2011; Duarte, et al., 2013). The findings of the literature review also provided more evidence in support of the adaptation values of coastal wetlands (Duarte, et al., 2013).

The literature also indicated that despite the significant climate benefits of coastal wetlands, these ecosystems have been least valued in climate change response policies. This has resulted in coastal ecosystems being increasingly under pressure by land use change and urban expansion in coastal areas (Duarte, et al., 2013). As receiving environments, coastal wetlands are vulnerable to the direct (e.g. coastal reclamations) and indirect impacts of land use changes within their catchments (e.g. change in sediment load by such activities as forestry, agriculture or urban development).

Added to these was the argument in the literature that lack of synergy between climate response and urban development (land use management) policies has been undermining the effectiveness of local governments

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1 Climate change is primarily associated with the increased anthropogenic carbon dioxide emissions as a result of a rapid pace of urbanisation (UNEP, 2012)

2 The term ‘land use category’ is used in this research consistent with the IPCC approach (IPCC, 2003). More information is provided in Chapter 3, Section 3.3.1.1.
strategies for addressing climate change (Mickwitz, et al., 2009; Duguma, et al., 2014; IPCC, 2014b). A summary of the preliminary literature review is provided in Section 1.3.

Based on these findings and considering the unique geographic features of the Auckland region with long coastlines, my attention turned to coastal wetlands and their climate change mitigation and adaptation benefits. I therefore focused on assessing Auckland’s coastal wetlands from a mitigation-adaptation perspective, and how this developing area of knowledge could and should influence planning decision-making in Auckland.

The next section sets out the aim and questions of this research. Section 1.4. describes the study area.

### 1.2. Research aim and questions

As discussed, the thesis started with an exploration of the feasibility of developing a framework to inform decisions about the implications of land use change in urban areas on their carbon balance. This initial substantial review established that such a framework was not practical at this stage. However, it identified the significant but under-valued role of coastal wetlands relative to other land uses. Findings of this investigation helped to specify the focus of the research and its aim as illustrated in Figure 1.2-1.

The aim of this research is to assesses whether and how to integrate the mitigation and adaptation benefits of coastal wetlands into urban land use and land use change decision-making in Auckland, New Zealand. To do so, the research seeks to answer the following five research questions for the case study:

- **A.** How do coastal wetlands in the Auckland region currently and potentially contribute to climate change mitigation and adaptation?
- **B.** Whether and how the climate change services of coastal wetlands are taken into account in current coastal resource management and climate change response policies and plans in New Zealand and Auckland, and what are the key challenges and opportunities?
- **C.** What can be learned from the experiences in other jurisdictions in terms of the initiatives and mechanisms that can assist in incorporating climate change services of coastal natural resources into land use, resource management and climate change policies?
- **D.** How do the policies and practices in other jurisdictions compare to the case of Auckland and New Zealand? and
- **E.** Depending on the outcome of A to D, investigate what policy options and planning instrument mixes can be applied to Auckland for mainstreaming climate change benefits of coastal wetlands into land use and resource management decision-making processes

The research expands on the existing knowledge about mitigation and adaptation benefits of coastal wetlands and investigates policy and planning implications of accounting for these benefits through the course of land use decisions and resource management. The research has three facets looking at the planning problem through the lenses of theory (Chapter 2), science (Chapters 4 and 5) and practice (Chapters 6 and 7). However, its primarily focus is on the application of science and theory to examine the practical issues, challenges and opportunities in a specific policy and planning context using the Auckland region as the case study.

The research therefore intends to use the findings and propose a planning framework to guide incorporation of the climate benefits of coastal wetlands into the mainstream urban land use decisions.
Summary of the preliminary literature review

The following provides a summary of the literature review that was carried out as part of the preliminary investigations as discussed in Section 1.1. This includes an overview of the key global planning issues relevant to the focus of this research.

Among the urbanised environments, coastal areas have the highest population density and rate of urban agglomeration (Gaast & Begg, 2012). The low-lying coastal areas occupy only about 2% of the world’s total land area, but they host about 13% of the world urban population (UN-Habitat, 2016). Communities in coastal areas benefit from highly productive and diverse ecological settings that extend across coastal gradient (Kohsaka, et al., 2013).

From an ecological perspective, vegetated coastal ecosystems in coastal areas offer a set of critical ecosystem services particularly in relation to climate change. The growing evidence shows that these ecosystems have significant capacity for carbon sequestration and storage, sediment accretion and coastal protection, along with their multiple other ecological and economic values (Duarte, et al., 2013). However, they are under increased pressure due to development and human-induced land use change. Globally in the past 50 years, about 25 to 50 percent of the areas covered by vegetated coastal habitats have been lost (Duarte, et al., 2013).

Urban environments are also a key element of global response to climate change (Wright, et al., 2011; Bulkeley, 2010). This is primarily because urban settings are the major sources of greenhouse gasses (GHGs) and are among the highly vulnerable systems to the impacts of climate change (Stern, 2007; Satterthwaite, 2008; Bulkeley, 2013; IPCC, 2014a). There are also many potential areas for fostering synergy in urban settings principally because of the diverse and heterogeneous nature of cities (Bulkeley & Betsill, 2013). There is a growing evidence in support of the benefits that can be gained by integrating mitigation and adaptation strategies into sector policies (IPCC, 2007c; IPCC, 2014b).

The common argument in favour of integrating adaptation and mitigation into sector policies is the potential efficiencies in the use of limited resources and capacities and greater economies of scale that can be gained through delivery of multiple benefits by fostering synergies (Locatelli, et al., 2016). This argument has been supported by several other studies including Adelle & Russel (2013), Hallegette, et al. (2011), Mickwitz, et al. (2009), Duguma, et al. (2014), Spencer, et al. (2016), Locatelli, et al. (2015).

The studies suggest that synergies and co-benefits between adaptation and mitigation can be realised both within and across sectors and policy domains and at various horizontal and vertical institutional scales,
with some sectors being more compelling for efficiency gain compared to others. However, a range of institutional, financial, political, technical and cognitive barriers have constrained and challenged the feasibility of achieving synergy between mitigation, adaptation and development (Klein, et al., 2005; Measham, et al., 2011; Biesbroek, et al., 2009; Tompkins & Adger, 2005; Kern & Alber, 2008; Mosera & Ekstrom, 2010; Landauer, et al., 2015; Suckall, et al., 2015; IPCC, 2014b; Locatelli, et al., 2015).

A growing body of literature has recently been focusing on the multiple ecosystem services of coastal wetlands in urban landscapes and increasingly provide scientific evidence in support of the carbon sink and coastal protection capacities of vegetated coastal ecosystems (Duarte, et al., 2013). These studies also suggest that accounting for the ecosystem services of urban landscapes and internalising the benefits into land use and other sector policies has potential adaptation, mitigation, biodiversity and other ecological and economic benefits (Bulkeley, 2013; Elmqvist et al., 2015; Haase, Frantzeskaki and Elmqvist, 2014).

Research in this area has therefore been pursuing an understanding of synergies and trade-offs between land use, natural resource management and climate change and the role of urban ecosystems in achieving win-win solutions (Gómez-Baggethun & Barton, 2013; Viguié & Hallegatte, 2012; Swart & Raes, 2007; Schmidt, et al., 2014; Di Gregorio, et al., 2017).

The current knowledge, however, needs to be transformed into a workable policy and planning language and a set of planning tools in a manner that would facilitate incorporation of the benefits of coastal wetlands into urban land use decision making (Veraart, et al., 2014; Van Enst, et al., 2014; Runhaar, 2016; Runhaar & Driessen, 2012).

An important practical issue is that there is no ‘one-size-fits-all’ solution for mainstreaming and climate policy integration (Eisenack et al., 2014; Rietig, 2012; Adelle and Russel, 2013). This partly relates to the nature and scale of urban planning and resource management issues that generally differ among coastal urban landscapes. The scale and complexity of issues may vary depending on the form, size, structure and dynamics of the urban landscape and its natural ecosystems. Therefore, in practice, each urban setting has its own set of resource management issues in relation to urban development and response to climate change.

One way of addressing this issue is to carry out case studies that can contribute to global knowledge by providing local understanding of the climate benefits of coastal wetlands and policy and planning tools that can assist in mainstreaming of the benefits, if any. The case of Auckland is described in the next section.

1.4. Description of the study area

In this research, the Auckland region in New Zealand is chosen as a case study. The following describes key features of the Auckland region that also shows why it is selected for this research.

The Auckland region is the New Zealand’s largest and most populous urban area. Figure 1.4-1 shows the boundary of the Auckland region including land, coastal and marine areas. The highest concentration of population is in its isthmus (central Auckland) and coastal areas. Auckland is amongst four fastest growing regions in New Zealand with a population forecast of about two million by 2033 (estimated annual growth rate of 1.5% projected for a 30-year period under a medium growth scenario) (Statistics New Zealand, 2017). This increase in population is expected to largely affect transport, housing and other infrastructure. Urban sprawl, housing affordability and reliance on motor vehicles are major issues in the Auckland region (Auckland Council, 2012b). The issues of urban sprawl in the Auckland region is primarily associated with the low-density housing which has been the dominant development pattern in suburban areas driven by expansion of motorways and high rate of car ownership (Auckland Council, 2012b).

Auckland is vulnerable to the impacts of climate change. Climate change is likely to bring about rising sea levels, an increase in floods and droughts, changed wind and rainfall patterns, increased temperatures, reduced frosts, put pressure on ecosystems, and increase the threat of pest species becoming established in
the region (Auckland Council, 2012b). Part of the Auckland’s first spatial plan (The Auckland Plan) (Auckland Council, 2012b) is devoted to strategies towards enabling the city to effectively respond to climate change. Auckland has a relatively ambitious GHGs emissions reduction targets. Consistent with the Government’s national targets, at the regional level, the Auckland Plan sets a target for a 40% decrease in Auckland’s GHGs emissions from 1990 levels by 2040.

The Auckland Council’s projections under business as usual scenario suggest a 39 percent increase in the Auckland’s GHGs emissions by 2031 against 1990 levels (Auckland Council, 2012b). One of the greatest challenges for Auckland in relation to response to climate change is the link between GHGs emissions and development, economic growth, and energy use (Auckland Council, 2012b). The Auckland Plan aims to progress Auckland towards an eco-city or liveable city by transforming the city into a compact, less fossil fuel-dependent, energy efficient and sustainable city (Auckland Council, 2012b).

Coastal ecosystems are significant features and characteristics of the Auckland region (Auckland Council, 2012b). The region has over 1,800 km of coastline\(^3\) and given the geographic and unique location of Auckland, most of the urban parts are sited on a narrow isthmus between two large harbours. Development has tended to be centred on coastal and low-lying areas where communities benefit from the marine environment, but are also at the front line of climate change impacts due to their close proximity to the sea (Auckland Civil Defence, 2011). Most of the reserves, parks, wetlands and rivers in the Auckland region are in danger of encroachment due to urban sprawl where new and existing development diminishes capacity to maintain ecosystem services (Lindsay, et al., 2009).

Management of natural and physical resources in Auckland is regulated under the Resource Management Act 1991 (RMA). The RMA is the New Zealand’s overarching legislative, policy and planning framework for environmental and resource management. The legislative and institutional context in New Zealand and Auckland are discussed in detail in Chapter 6 (Section 6.2).

One of the resource management issues in the Auckland region is the extensive reclamation and drainage of estuaries due to human settlement and development which has resulted in significant reduction in the original areas of wetlands particularly mangrove forest (Lindsay, et al., 2009; Auckland Council, 2011).
On the other hand, increased sedimentation in New Zealand estuaries since the 1940s as a result of deforestation and development activities in catchments, has been the main cause of mangrove expansion in some estuaries across the North Island (Morrisey, et al., 2010; Swales, et al., 2015). Consequently, there has been increasing pressure on local authorities (e.g. the Auckland Council), mainly from coastal communities, to facilitate mangrove removal to enhance the amenity and recreational values of estuaries. Added to this is the evidence of illegal mangrove removal and poisoning in the Auckland region (Auckland Council, 2011). The issues regarding mangrove management in Auckland are further discussed in more details in Chapters 5 and 6.

When this research was initiated in 2012, the Auckland Council was at the initial stages of integrating their resource management and land use plans into a unitary plan known as the Proposed Auckland Unitary Plan (See Chapter 6, Section 6.2 for details). The Plan was ultimately adopted (in part) in November 2016. This research was carried out over this period, during which several other policy debates and resource management reforms have also evolved at national level in New Zealand. The relevant ones include amendments to the Resource Management Act (RMA) coming into effect in 2013, the amendment to the National Policy Statement for Freshwater Management (NPSFM) made in 2014, and a non-statutory guideline for biodiversity offsetting in New Zealand published in 2014.

The evolving policy landscape in New Zealand has been a unique opportunity for this research to carry out a real-time study of a policy process from its inception to its final stage.

1.5. Organisation of chapters

This thesis is organised in nine chapters as described below:

Chapter 2 summarises the findings of a review of the literature to gain an insight into the existing body of knowledge, identify gaps and establish a theoretical context to inform the research design. The chapter outlines the theoretical framework and discusses the key theories and concepts that underpin the research questions.

Chapter 3 describes the research design, outlines the methods used to answer the specific research questions and defines the methodological basis for choosing those methods.

The findings of a systematic review of the literature on carbon sequestration capacity of different land use categories carried out in the early stage of this research (as mentioned in Section 1.1) are provided in Chapter 4. This chapter also includes a brief discussion about why coastal wetlands were identified as the focus of this research.

Chapter 5 provides the findings of an extensive review of the literature addressing the significance of coastal wetlands for climate change mitigation and adaptation. This chapter also includes discussions about the approximate potential of temperate mangrove and saltmarsh ecosystems (similar species to those in Auckland) for carbon storage and sequestration and their role in coastal protection.

The analyses of the New Zealand and Auckland’s relevant policy and planning documents (in Chapter 6) combined with the results of the global case studies (in Chapter 7) provides a basis to identify and discuss the policy options and instruments (in chapter 8) for incorporating climate change services of coastal wetlands into the Auckland Council’s mainstream policy framework.

The final chapter of this thesis (Chapter 9) summarised the key findings with respect to the research questions and outlines the main recommendations, contributions and limitations of the research. This last chapter also provides some directions for future research.
2. Theoretical Framework

2.1. Introduction

Theories explain natural, physical and social phenomena and are a means to make sense of how they work and relate to each other (Eisenhart, 1991; Timmermans & Tavory, 2012). A theory either evolves from other theories or is a product of systematic empirical research, or combination of both, and hence can present various degrees of abstraction and applicability. In any case, theories are valid and well-confirmed propositions unless their truth is falsified by other more rigorous theories, observations or evidence (Popper, 1972).

As with planning generally, environmental and urban planning employ a mix of theoretical and analytical frameworks. Some of these theories are specific to the field of policy and planning (e.g. the theory of bounded rationality (Simon, 1972), communicative planning theory (Healey, 1992) or argumentative theory of planning (Fischer & Forester, 1993). Also, environmental and urban planning are multi and cross-disciplinary domains and hence borrow a wide range of theories from various other pure and applied domains of inquiry, most notably ecology, systems science, geography, social and cognitive science, political science, economics, and decision theory. Many of the recent theories are already multi-disciplinary in nature, meaning that they map landscapes of connected ideas and concepts from various domains. The choice of theories for studies in urban and environmental planning depends on the problem and purpose of the research. It is therefore imperative for research in the domain of planning and policy to clarify how the research links theories to the specific planning problems under investigation.

This chapter starts with a brief outline of the key themes of the research. This is followed by sections discussing the main concepts and approaches underpinning the research idea and questions. The last section of this chapter provides a simplified conceptual framework which help to understand the link between theory and practice within the context of this research. The theoretical framework guides the research design and the choice of analytical frameworks and methods in Chapter 3. It also provides a theoretical basis for the analyses and arguments in Chapters 5 to 8. This chapter in combination with the findings of Chapters 5 and 6 and the insights gained from the global case studies in Chapter 7 eventually feeds into the discussions in Chapter 8.

2.2. Outline of the key themes

Figure 2.2-1 illustrates the type of interactions and consequential trade-offs between climate change, development and natural resource management policies in coastal areas. While coastal natural resources can potentially mitigate climate change and enhance adaptation to coastal hazards, the difference in scale of the issues, and the lack of consensus about the actual benefits of natural resources can create a circular and non-resolvable resource management problem. Uncertainty regarding the long-term sustainability (or existence) of coastal wetlands in response to changing environment, difficulties in deciding on accepting current costs (e.g. altering development options) for deferred benefits (due to the long-term dynamic of natural systems) as well as institutional non-alignment may challenge the decision-making process.
The basic questions are relatively straightforward on their own: what should we do with coastal wetlands which are part of our urban environment? How important are they in our response to climate change? But when it comes to answering these questions, planners or policy makers are often overwhelmed by a range of critical considerations and a cascading set of issues.

Planners and policy makers may know about some values of their wetlands, but with many other competing issues and without sufficient knowledge and strong evidence, they cannot convince other stakeholders to make a choice. They also know these ecosystems are already under critical pressure from urban growth and threatened by climate change, but they cannot stop urban growth, which already centres on the coastlines. They also know climate change is threatening their urban systems, and they need to be able to manage the risks and impacts, but they may not be convinced that wetlands can do the job for them. In addition, their knowledge may not allow them to conclude whether it is worth conserving or reconstituting them or not. But if the community decides to let their wetlands shrink, they will irreversibly lose one of their potentially valuable natural capitals, and would need to accept the known and unknown consequences as a result of that loss. In this situation, there is a limited opportunity for trial and error as development pathways may rule out modifying decisions later on. Putting these and many other extremely interconnected issues together creates a planning dilemma.

To deal with this dilemma, this research seeks to explore the possibility and means of easing the tensions and fostering the synergies between what can at times be three competing objectives (i.e. providing for urban growth and socioeconomic development, conserving natural resources and responding to climate change) that may require trade-offs. It hypothesises that integrating climate change values of coastal wetlands in land use and coastal resource management can help avoid further loss of coastal wetlands (or ideally lead to restoration of wetlands) and enhance resilience to the impacts of climate change.

This leads to using wicked problems and the concept of resilience as the theoretical frameworks for this thesis. These concepts are discussed within the subsequent sections.

### 2.3. The wicked nature of the problem

Addressing a problem primarily needs an understanding of its nature (Bardwell, 1991). The main themes of this research i.e. urban and land use planning, response to climate change and natural resource planning, all exhibit some degree of complexity and uncertainty and the issues related to these subjects can be

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4 The issue of trade-offs in the New Zealand’s legislative and planning contexts are discussed in Section 6.3.1.
classified as ‘wicked’ problems (Rittel & Webber, 1973). The concept of ‘wicked problems’ was first described by Rittel and Webber (1973) in the context of urban planning. Since then, the concept has been widely applied to almost all social and environmental problems including resource and land use management, sustainability and climate change. Rittel and Webber, (1973, pp.160-66) have identified the following ten characteristics for wicked problems:

- There is no definite formulation of a wicked problem.
- Wicked problems have no stopping rules.
- Solutions to wicked problems are not true-or-false, but good or bad.
- There is no immediate and no ultimate test of a solution to a wicked problem.
- Every solution to a wicked problem is a “one-shot operation”; because there is no opportunity to learn by trial-and-error, every attempt counts significantly.
- Wicked problems do not have an enumerable (or an exhaustively describable) set of potential solutions, nor is there a well-described set of permissible operations that may be incorporated into the plan.
- Every wicked problem is essentially unique.
- Every wicked problem can be considered to be a symptom of another problem.
- The causes of a wicked problem can be explained in numerous ways. The choice of explanation determines the nature of the problem’s resolution.
- The planner has no right to be wrong.

A wicked problem is a complex problem that involves enormous interdependencies, uncertainties, circularities, a wide range of stakeholders with conflicting interests and an inherent disagreement about the nature of the problem and the best way to tackle it (Rittel & Webber, 1973; Lazarus, 2008). A wicked problem cuts across spatiotemporal and institutional scales and has no definite solution (Lazarus, 2008). To harness a wicked problem involves dealing with uncertainties, multiple causal factors and high levels of disagreement among stakeholders about the nature of the problem and the best way to tackle it (Lazarus, 2008).

Understanding the properties of wicked problems and possible means to tackle them is part of the problem solving and policy making process in urban and environmental planning. There are no fixed solutions for wicked problems. Tackling wicked problems would rather need a holistic, multi-disciplinary, integrative, collaborative and adaptive approach with proper means to cope with the problems by minimising the need for trade-offs and fostering mutually reinforcing outcomes (Levin, et al., 2012).

Unlike the so-called ‘tame’ problems that are normally approachable and have technical and scientific solutions, there is no one correct ‘solution’ for a wicked problem (Rittel & Webber, 1973). In real-world situations, every wicked problem would be unique to each city due to the city having its own unique identity and hence set of issues. The problem of this type is in essence controversial because many stakeholders often with different and conflicting opinions and interests are involved that challenge the nature of problem and the type of solutions for solving it.

The lack of complete and exhaustive set of solutions for this type of problems implies that any possible solution would be a symptom of another problem (Hartmann, 2012). Moreover, any potential solution would have irreversible consequences and hence would not ethically justify trial-and-error. In fact, wicked problems are never solved, ‘at best they are only re-solved – over and over again’ (Rittel & Webber, 1973, p.160). Successfully addressing wicked policy problems usually involves a range of coordinated and interrelated responses, given their multi-causal nature; it also often involves trade-offs between conflicting goals (Australian Public Service Commission, 2012). The motivation and behaviour of individuals is a key part of the solution as is the involvement of all levels of government and a wide range of non-government organisations (NGOs) (Australian Public Service Commission, 2012).
The Australian Public Service Commission (2012) has identified the following strategies, as possible ways to deal with wicked policy problems particularly those related to climate change:

- Holistic, not partial or linear thinking
- Innovative and flexible approaches
- The ability to work across agency boundaries
- Effectively engaging stakeholders and citizens in understanding the problem and in identifying possible solutions
- Additional core skills
- Tolerating uncertainty and accepting the need for a long-term focus

Climate change and almost all issues that relate to the changing climate and/or are consequences of it are a particular example of ‘wicked problems’ (Incropera, et al., 2015). Climate change has also been characterised as a ‘super wicked problem’ (Levin, et al., 2007; Lazarus, 2008; Levin, et al., 2012). Climate change is said to have the biggest impact on urban systems compared to other less populated areas (UN-HABITAT, 2011). Climate response policies principally seek to reduce GHG levels and adopt measures to enable socio-ecological systems to withstand climatic impacts (Bulkeley, et al., 2009).

The wicked nature of the problems related to climate change partially points to the complex, multi-scale, and evolving nature of the interactions between the three goals of adaptation, mitigation and development (Hamin & Gurran, 2008; De Groot, et al., 2010). It is also associated with the fact that climate adaptation decisions need to be made under deep uncertainty which is associated with the dynamic nature of climate risks such as sea level rise and storm events (Haasnoot, et al., 2013; Kwakkel, et al., 2016). The term ‘uncertainty’ is defined as “any deviation from the unachievable ideal of completely deterministic knowledge of the relevant system” (Walker, et al., 2003). This implies that decision-makers have to deal with uncertain knowledge and information on issues such as future rates of sea level rise and magnitude and frequency of storm events (Buurman & Babovic, 2016). In addition, the connection between climate change adaptation and other environmental, social, economic and political systems adds to the uncertainties and ambiguities that affect climate adaptation decisions (Buurman & Babovic, 2016).

For example, long-term sustainability of coastal wetlands as a means for protecting coastlines against the impacts of climate change is influenced by a range of factors including ecological (e.g. ecosystem dynamics such as growth and decay), anthropogenic (e.g. political and management decisions) and climate change (e.g. sea level rise) (Bouma, et al., 2014). This creates uncertainty on the long-term persistence and sustainability of these ecosystems in response to changing environments (Bouma, et al., 2014). Therefore, managing coastal wetlands for the purpose of climate change adaptation involves uncertainties on persistence of these ecosystems over time (Spalding, et al., 2014a).

The literature has identified a number of approaches which can assist decision making in the face of deep uncertainties such as planning for climate change adaptation. These include Adaptive Policy Making (Walker, et al., 2013), Adaptation Pathways (Reeder & Ranger, 2011; Haasnoot, et al., 2013; Barnett, et al., 2014), Real Options Analysis (Gersonius, et al., 2011) and Transient (time-dependent) Scenarios (Haasnoot, et al., 2015). There are also a wide variety of tools and methodologies that can deal with uncertainty in general. Examples include scenario planning, Monte Carlo Analysis, Multi-layer Decision Analysis and safety margin strategies (Buurman & Babovic, 2016). Choosing the right mix of approaches, tools and methodologies as well as correct framing of uncertainty are necessary to develop robust climate adaptation plans (Buurman & Babovic, 2016). Ranger, et al., (2013) have also discussed the innovations and techniques that were used in the Thames Estuary 2100 (TE2100) project to address and recognise uncertainty in climate projections throughout the planning process. The approach used in the TE2100 project is identified as a “dynamic robustness” approach which aims to develop flexible adaptation strategies that can be modified over time to adapt to new conditions or changes.
The precautionary principle is another approach to respond to uncertainty and supports decision-making in the absence of scientific certainty about consequences of human decisions and action on environment (Walker, et al., 2003). This approach was first globally codified by principle 15 of the Rio declaration (1992) and indicates that lack of full scientific certainty shall not be used as a reason for postponing measures to prevent environmental degradation where there are threats of serious or irreversible damage to the environment (UNFCCC, 2006). This principle can take various forms and has been addressed in different ways ranging from weak (where precaution is limited to serious or irreversible damage and obligations are imposed to ensure the lack of scientific uncertainty is not a reason for delaying actions) to strong (where precaution is broad and not limited to only serious and irreversible harm, and decision-making is informed by science but also influenced by values, judgments and perception of acceptable risks, costs and benefits) (Cooney, 2004). The strong version of the precautionary principle is based on shifting the burden of proof to proponents of harmful activities. This is one of the most important ways to operationalise the precautionary principle and would require the proponents of potentially harmful activities to demonstrate that their proposals are safe to an acceptable level (Cooney, 2004). Article 3.3 of the United Nations Framework Convention on Climate Change (UNFCCC) places an emphasis on the importance of adopting the precautionary principle in planning for climate change mitigation and adaptation (UNFCCC, 2006).

2.4. Resilience, vulnerability and adaptation

The concept of resilience was first used by Holling (1973) as ‘a way to understand nonlinear dynamics as the magnitude of disturbance that a system can experience before it shifts into a different state’ (Folke, et al., 2002). As stated by Swanstrom (Swanstrom, 2008, p.2), ‘resilience is more than a metaphor but less than a theory. At best it is a conceptual framework’ that introduces a dynamic and holistic view to understand processes such as climate change adaptation.

The three broad perspectives of resilience include engineering, ecological and socio-ecological or evolutionary (Gunderson & Holling, 2002; Simmie & Martin, 2010; Davoudi et al., 2013). Under engineering and ecological approaches, resilience refers to the capacity to bounce back or to rebound (Davoudi, 2012).

Engineering resilience is defined as ‘the ability of a system to return to an equilibrium state after a temporary disturbance’ (Holling, 1973, p.17); while ecological resilience is defined as ‘a measure of the ability of these systems to absorb changes [. . .] and still persist’ (Holling, 1973, p.17). Therefore, engineering and ecological resilience, both assume a stable equilibrium to which a resilient system bounces back (engineering) or bounces forth (ecological) (Davoudi, 2012). However, engineering resilience assumes a single equilibrium while ecological resilience suggests existence of multiple equilibria (Holling, 1973). Therefore, these approaches differ in terms of the state of stability or equilibrium (Davoudi, 2012).

The socio-ecological approach to resilience challenges the notion of a stable equilibrium and argues in favour of a dynamic interaction between people and nature as interdependent systems (Folke, et al., 2010). Socio-ecological resilience includes three main elements: (i) the amount of disturbance that a system can absorb without reducing its function, (ii) the degree to which a system is capable of self-organisation in response to change; and (iii) the degree to which a system can develop and enhance its adaptive capacity through learning from change (Carpenter, et al., 2001; Dong, et al., 2016).

By this approach, resilience is the ability of complex social-ecological systems to change, adapt or transform in response to stresses and strains (Carpenter, et al., 2005). Socio-ecological systems ‘are neither humans embedded in an ecological system nor ecosystems embedded in human systems, but rather a different thing altogether. Although the social and ecological components are identifiable, they cannot easily be parsed for either analytic or practical purposes’ (Walker, et al., 2006, p. 13).

Within the context of climate change, resilience is defined as ‘the capacity for a socio-ecological system to absorb stresses and maintain function in the face of external stresses imposed upon it by climate change
and to adapt, reorganize and evolve into more desirable configurations that improve the sustainability of the system, leaving it better prepared for future climate change impacts’ (Salgotra & Gupta, (2015, p.202), adopted from Nelson, et al., (2007)).

This definition is placed under the socio-ecological approach of resilience (Folke, et al., 2002; Walker, et al., 2004; Walker, et al., 2006) which is also referred to as ‘evolutionary resilience’ (Simmie & Martin, 2010; Davoudi, et al., 2013). Socio-ecological (or evolutionary) resilience places an emphasis on the dynamic nature of systems and their ability to transform to a new state with or without external disturbance (Scheffer, 2009, Cited in Davoudi, 2012).

Given the dynamic nature of climate risks and their associated uncertainties (as mentioned in Section 2.3), this perspective of resilience is suggested to be appropriate for studies addressing wicked problems including climate change policies (McEvoy, et al., 2013). The literature of climate resilience suggests that climate adaptation strategies can benefit from adopting the socio-ecological or evolutionary approach of resilience on the basis that such an approach has elements that extend the capacity of socio-ecological systems beyond merely planning for recovery from shocks (e.g. climate hazards) and includes proactive planning for preparedness and adaptive planning for transformation towards a desirable trajectory using the opportunities that emerge from change (Davoudi, et al., 2013).

Based on this discussion, socio-ecological resilience is the most relevant perspective to this research. However, it is important to note that engineering and ecological concepts are also relevant. This is because these concepts represent aspects of resilience that are not mutually exclusive, but interact.

Engineering resilience refers to the capacity of a system (often physical or engineered) to absorb damage without suffering complete failure; while ecological resilience explains how complex natural systems such as ecosystems absorb perturbations and flip into another regime of behaviour by changing variables and processes that control behaviour of the systems (Holling, 1996). Within the context of this research, hard structures such as seawalls are examples of systems with engineering resilience while coastal ecosystems that are capable of withstanding and recovering from the effects of climate change demonstrate a type of ecological resilience. Generally, resilience-based responses to climate change are on a spectrum and interact with each other.

Davoudi, et al., (2013, p.311) posit that resilience in socio-ecological systems can be influenced by:

“(i) their social learning capacity (being prepared) for enhancing their chances of resisting disturbances (being persistent and robust),

(ii) absorbing disturbances without crossing a threshold into an undesirable and possibly irreversible trajectory (being flexible and adaptable) and,

(iii) moving towards a more desirable trajectory (being innovative and transformative)”.

Transformational change involves a change in the nature of the stability landscape, introducing new defining state variables and losing others, as when a household adopts a new direction in making a living or when a region moves from an agrarian to a resource extraction economy (Walker, et al., 2004). It can be a deliberate process, initiated by the people involved, or it can be forced on them by changing environmental or socio-economic conditions (Folke, et al., 2010). Whether transformation is deliberate or forced depends on the level of transformability in the social-ecological system concerned. Transformational change often involves shifts in perception and meaning, social network configurations, patterns of interactions among actors including leadership and political and power relations, and associated organisational and institutional arrangements (Folke, et al., 2010).

The concepts of resilience, vulnerability and adaptation are interrelated and have been widely used together in the literature addressing climate change (Smit & Wandel, 2006; Gallopín, 2006; Leal Filho, et al., 2016). Resilience is argued to be an antonym or the ‘flip side’ of vulnerability (Folke, et al., 2002). In other words, when a system loses resilience it becomes vulnerable to change that previously could be absorbed without
structural change (Folke, et al., 2004). Conversely, reducing vulnerability of socio-ecological systems to the impacts of climate change is considered as increasing resilience to climate change (Davoudi, et al., 2013). In this context both resilience and vulnerability have many aspects in common as they can be considered as a system response to an external shock, risk or stress (Prasad, et al., 2009). Therefore, vulnerability assessment through identifying the risks of climate change to both natural and ecological ecosystems is the key aspect of resilience planning in socio-ecological systems (Malone, 2009).

Vulnerability refers to the ‘state of susceptibility to harm from exposure to stresses associated with environmental and social change and from the absence of capacity to adapt’ (Adger, 2006, p.268). In the context of climate change, IPCC (2007b, p. 883) defines vulnerability as ‘the degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes’. By this definition, vulnerability can be assessed by understanding the nature and magnitude of climate risks, sensitivity and exposure to the risks and the adaptive capacity of a system (IPCC, 2007b).

Within this context, adaptive capacity refers to the potential of a system to adjust to the impacts of climate change, to minimise damages, to use beneficial opportunities and to deal with the consequences of climate change (IPCC, 2007b). Adaptation, therefore, refers to the adjustment of a system to actual or expected stimuli (e.g. climate change) or its effects to minimise the damage or use opportunities (IPCC, 2007b).

Transformational adaptation (to the impacts of climate change) is defined by the IPCC (2014d, p.1758) as “adaptation that changes the fundamental attributes of a system in response to climate and its effects”.

Unlike incremental adaptation which aims “to maintain the essence and integrity of a system or process at a given scale” (IPCC, 2014d, p.1758), transformational adaptation requires changes at a greater scale and may introduce fundamentally different and innovative approaches to preparing for and responding to climate risks (Lonsdale, et al., 2015; Parry, 2017). It, therefore, involves questioning the effectiveness of conventional (or traditional) approaches, practices and governance structures for management of climate risks and may lead to changes in systems of governance, shifts in paradigms and implementation of different practices to improve response to climate risks (Lonsdale, et al., 2015; Parry, 2017).

As discussed above, evolutionary resilience highlights the importance of learning-based adaptive management (Davoudi, et al., 2013) and introduces adaptation as the ability to maintain a response capacity in the face of risks that keep changing and evolving (Hart, 2011). Therefore, adaptation measures that are transformational in nature are key requirements for building and enhancing resilience to the impacts of climate change (Parry, 2017).

Examples of transformational adaptation measures/practices include increasing the scale and intensity of risk management strategies, transforming the location and composition of activities (e.g. coastal retreat), encouraging the use of ecosystem-based approaches to address climate risks, recognising and embracing uncertainty into decision-making processes, prioritising flexible solutions and enhancing involvement and participation of various stakeholders in planning processes (Kates, et al., 2012; Brooks, et al., 2016; Parry, 2017).

Building levees, dykes or seawalls along coastlines and maintaining and adjusting these structures over time are examples of incremental adaptation measures which may result in ‘maladaptation’ (Barnett & O’Neill, 2010) over a long-term (Kates, et al., 2012; Brooks, et al., 2016; Parry, 2017; Brown, et al., 2017). This is because the coastal protection measures give a false sense of security and allow current development patterns to become entrenched (Brown, et al., 2017). They can also adversely affect the ecological processes that regulate natural systems (Pérez, et al., 2010) and result in coastal squeeze.

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5 “Action taken ostensibly to avoid or reduce vulnerability to climate change that impacts adversely on, or increases the vulnerability of other systems, sectors or social groups” (Barnett & O’Neill, 2010: p. 211).

6 Coastal squeeze refers to the loss of coastal wetlands with sea level rise (Bouma, et al., 2014). It occurs in areas where inland migration of coastal wetlands is not possible due to presence of coastal protection structures or intensive coastal development (Bouma, et al., 2014; Lovelock, et al., 2015). This process is further explained in Section 5.3.2.
In practice, a wide range of barriers including institutional values, governance systems, funding arrangements and land use/development policies challenge implementation of transformational adaptation measures (Brown, et al., 2017). These barriers often create path dependency that “manifests as resistance to changing the way things have always been done, even if business as usual seems to be increasingly maladaptive” (Barnett, et al., 2015, p.2). Path dependencies reduce incentives to introduce/implement proactive adaptation strategies and increase the risk of making maladaptive decisions (Brown, et al., 2017).

For instance, implementation of managed retreat, as an example of long-term transformational adaptation mechanism, is often constrained by the property/land use rights of properties in coastal areas and may involve strong community resistance (Hart, 2011). Property rights can also indirectly affect sustainability of coastal ecosystems and contribute to coastal squeeze. This is because they are the main barriers to managed retreat which can enhance sustainability of coastal ecosystems by providing space and facilitating landward migration of coastal ecosystems (U.S. EPA, 2009).

The theory of resilience provides the underpinning framework for a number of other inter-disciplinary approaches, including ecosystem-based adaptation (Section 2.6) which is an approach for building resilience to the effects of climate change using ecosystems and through a collaborative process (Pérez, et al., 2010). The emerging ‘urban sustainability approach’ (Wu, 2014) (Section 2.6) focuses on the spatiotemporal relationships between biodiversity, ecosystem processes (or ecosystem functions), ecosystem services, and human well-being in urban landscapes and regions. The concept of urban sustainability is the recently evolved approach in urban ecology and encompasses other approaches such as urban systems and urban landscape (Wu, 2014). However, it broadens the scope of urban ecology studies to include the dynamic interactions between urbanisation and natural elements of urban landscapes by focusing on ecosystems services and biodiversity (Wu, 2014). The concepts of socio-ecological resilience and urban sustainability are complementary, as the impacts of urbanisation (e.g. land use change) on resilience through altering biodiversity and ecosystem services can be studied under the urban sustainability approach (Wu, 2014).

This research takes the premise of using multi-functional services of coastal wetlands as a means for linking mitigation and adaptation goals with those of development. In other words, assuming coastal wetlands provide climate change services as part of their multiple ecosystem services, they can make a substantial contribution towards adapting to and mitigating climate change (Davoudi, et al., 2013). On this ground, enhancing resilience to climate change through reducing vulnerability to climate impacts and improving carbon sequestration and storage (CS&S) capacity of urban landscapes are hypothesised as two main co-benefits of protection and restoration of coastal wetlands. Changes in patterns of urban landscape (e.g. connectivity and distribution of natural habitats) that are driven by both socio-economic and biophysical processes can affect this particular element of resilience in urban systems (Alberti & Marzluff, 2004).

Given the link between ecosystem services and socio-ecological resilience, the loss of ecosystems means reduced resilience. There are various approaches and tools in conservation ecology (such as environmental compensation or offset (BBOP, 2013)) and urban design (most notably the approach of ‘positive development’ (Birkeland, 2008)) that seek to contribute to enhancing resilience in urban landscapes. These approaches are designed to establish a link between development and conservation outcomes by offsetting the residual adverse effects of development on natural systems (Birkeland & Knight-Lenihan, 2016). The overall purpose is to conserve ecosystems and their ecosystem services while achieving development outcomes. The approach of ‘positive development’ (Birkeland, 2008) expands on these concepts and tools and seeks to provide not only for offsetting but for net positive outcomes with the aim of contributing to the effective reduction of the trend of global ecological deficit.
2.5. Perspectives on urban resilience to climate change

As suggested by Bulkeley (2010, p. 29), 'cities have emerged as key players in the governance of climate change'. There are several ongoing efforts that focus on climate change and cities. ICLEI – Local Governments for Sustainability has played a major role in encouraging mitigation efforts by local governments around the world. The United Nations Human Settlements Programme has set up a Cities and Climate Change Initiative (CCCI)\(^7\), and developed a guide (UN-HABITAT, 2014) for city planners and other allied professionals to better understand, assess, and take action on climate change at the local level. A Knowledge Centre on Cities and Climate Change (K4C) has also developed through a joint collaboration between UN-HABITAT, the World Bank, the United Nations Environment Programme (UNEP) and Cities Alliance (United Nations, 2012). K4C provides online information that can inform local decision-making about climate change.

In late 2016, the United Nations Conference on Housing and Sustainable Urban Development (Habitat III) endorsed a New Urban Agenda (United Nations, 2017). The so called action-oriented New Urban Agenda addresses the multiple dimensions of urban sustainability, takes full account of the 2030 Agenda for Sustainable Development (United Nations, 2015), the Paris Agreement (UNFCCC, 2016) and the Sendai Framework for Disaster Risk Reduction 2015–2030 (UNISDR, 2015), and includes, as a fundamental feature, climate change adaptation and mitigation. The agenda aims to contribute to the achievement of the Sustainable Development Goals in an integrated manner, with a special attention to the goal of making human settlements and cities inclusive, safe, resilient and sustainable by, among other means, reducing risks and vulnerabilities, fostering resilience and ecosystem-based solutions including protecting, conserving, restoring and promoting urban ecosystems and biodiversity.

The Agenda also gives special consideration to environmentally sensitive areas, including urban deltas and coastal areas, highlights the importance of urban ecosystems and their services and establishes a commitment for integration of appropriate measures into sustainable urban planning and promotes a polycentric governance (Ostrom, 2001) model. The New Urban Agenda specifically highlights that effective implementation of the commitments would require a wide-range of means to enable policy frameworks, develop and strengthen capacities and share best practices among governments and institutions at all levels within a polycentric and multi-level governance framework. These new and ongoing initiatives and developments are in essence built upon the mounting research as well as academic and policy discourses over at least the last two decades that primarily focused on tackling the wicked problem of sustainable development and concerned about the challenge of governing urban landscapes in the face of climate change.

A comprehensive review and analysis of the capacities of local governments and urban planning systems to adapt to climate change is provided by Broto and Bulkeley (2013). As a means for mainstreaming climate change, they suggested that integration of climate change across different policy domains is critical to developing effective policy and action. Carmin, et al., (2011) argue that rather than promoting a single model of adaptation, the trends in research findings suggest that it is imperative to identify variations in pathways so that cities can pursue approaches that support their goals and are aligned with their particular administrative, social, cultural, and political contexts.

Carmin, et al., (2011) also argue that adaptation actions are critical to creating resilient, sustainable cities. However, injecting climate considerations into decision-making is a challenge given the range of social, economic and environmental issues competing for attention and support. They further argue that while adaptation actions are consciously or unconsciously underway throughout municipalities, mainstreaming will require alterations in management approaches so that adaptation forms a programmatic course of

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action that is consistent with and can be readily addressed in the context of existing plans, systems, and routines.

The concept of ‘Climate Resilient Cities’ focuses on reducing vulnerability to climate impacts in urban areas by an active adaptive management (Prasad, et al., 2009; Carmin, et al., 2011). This involves engaging with local government in training, capacity building and capital investment programs as principled priorities for building sustainable, resilient communities. In this context, a good understanding of climate change impacts and the hot spots of a city, that represent the most vulnerable sites, plays an important role for effective disaster risk management (Prasad, et al., 2009). Prasad, et al., (2009) argued that cities can increase their resilience by: (i) enhancing knowledge of risks, and utilising tools and resources available to confront threats and build on opportunities, (ii) enhancing their autonomy and their governance system that rely on active collaboration between the different stakeholders, and (iii) developing disaster-resistant infrastructure.

An essential aspect of climate resilient cities is adopting a holistic approach towards coordinating and mainstreaming mitigation and adaptation strategies into urban planning processes to be able to actively deal with climate risks and impacts (Jabareen, 2013). In fact, these strategies must be treated as twin issues (Hamin & Gurran, 2008).

However, little attempt has been made to demonstrate to what extent adaptation strategies support ongoing mitigation policies. In some cases, mitigation and adaptation policies are complementary but in other cases they are inconsistent. For instance, under the mitigation perspective very high-density and concentrated developments are a desirable urban form and land use pattern respectively. While many adaptation activities, although certainly not all, require more land left in open space and/or a less dense built environment. In some cases, mitigation actions are not consistent with each other in the long term. For instance, consolidating urban structure due to climate change mitigation can lead to the loss of green areas (Niemelä, et al., 2010). Such loss can result in an undesirable outcome from the perspective of climate change mitigation, as urban residents have to travel to find recreational services.

The Asian Cities Climate Change Resilience Network (ACCCRN) is an initiative which aims at improving resilience of rapidly urbanizing Asian cities to climate change. According to ACCCRN, a climate resilient city has incorporated flexibility, redundancy, capacity to re-organize and capacity to learn into its urban systems and ways in which people construct and maintain those systems. Therefore, it is able to withstand a variety of climate challenges (Ahadur, et al., 2013). These elements of urban resilience affect urban vulnerability. For instance, if redundancy or flexibility is eliminated, urban vulnerability increases.

ACCCRN also introduces the Shared Learning Dialogue (SLD) process as a framework which has been adapted and implemented locally to reflect national and city contexts and expertise. SLD process is mainly based on capacity building, vulnerability assessment and engaging local experts and vulnerable groups in resilience planning. Development of a common understanding of climate change and urban resilience, share lessons between cities and beyond and preparing action plans for responding to climate risks are the ultimate outcomes of the SLD process (Opitz-Stapleton, et al., 2011).

Recent studies also highlight the role of urban governance in addressing climate change issues (Bulkeley, 2010; Jabareen, 2013). However, there is also a general agreement among academics and practitioners that cities and national governments cannot act separately to effectively tackle climate change (OECD, 2010). The literature therefore suggests that active involvement of different levels of government (i.e. local, regional, national) as well as the private sector and non-governmental stakeholders, within a multi-level governance framework, is required for addressing climate change (OECD, 2010).

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8 ACCCRN, About ACCCRN. [Online] Available at: [https://www.acccrn.net/about-acccrn](https://www.acccrn.net/about-acccrn) [Accessed 2016].
2.6. Perspectives on resilience and urban ecosystem services

Ecosystem services are simply ‘the benefits people obtain from ecosystems’ (Millennium Ecosystem Assessment, 2005, p. v) and can be categorised within the four main categories of ‘provisioning services’ (e.g. food, timber, fuel); ‘regulating services’ (e.g. flood/climate regulation, water purification); ‘cultural services’ (e.g. recreational, aesthetic values) and ‘supporting services’ (e.g. soil formation, nutrient cycling) (Millennium Ecosystem Assessment, 2005). A number of studies (Grimm, et al., 2000; Wu, 2014; Carabine, et al., 2015) exclude supporting services from the classification of ecosystem services on the basis that they are representative of ecosystems functions or processes rather than a specific category of ecosystem services.

From the perspective of urban ecology and resilience, the loss of ecological services provided by an urban ecosystem can lead to loss of ecosystem resilience and options for future generations (Folke, et al., 2004). Loss of ecosystem services as a result of urban growth and development can also reduce resilience of urban systems (Wu & Wu, 2013). Sustainability of urban landscapes is dependent on natural resources that are vulnerable to both human activities and the impacts of climate change. Understanding this vulnerability and its link to ecosystem services and human wellbeing as part of a social learning process can contribute to socio-ecological resilience in urban regions (Tompkins & Adger, 2003). Adaptive and ecosystem-based resource management is therefore contended as an approach that can help enhance resilience of both human and ecological systems (Tompkins & Adger, 2003; Ahern, 2011).

Chapin, et al., (2009) argue that accelerated global changes in climate, environment, and socio-ecological systems demand a transformation in human perceptions of our place in nature and patterns of resource use. Given the challenges of sustaining socio-ecological systems in a rapidly changing world, they advocated a shift to ecosystem stewardship. The central goal of ecosystem stewardship is to sustain the capacity of ecosystems to provide services that benefit society by sustaining or enhancing the integrity and diversity of ecosystems as well as the adaptive capacity and well-being of society (Chapin, et al., 2009).

Ecosystem stewardship requires adaptive governance of coupled socio-ecological systems to provide flexibility to respond to extreme events and unexpected changes. Rather than managing resource stocks and condition, ecosystem stewardship emphasises adaptive management of critical slow variables and feedbacks that determine future trajectories of ecosystem dynamics. Chapin, et al., (2009) suggest a practical approach to implementing ecosystem stewardship through the integration of three broad sustainability strategies: (i) reducing vulnerability; (ii) fostering adaptive capacity and resilience; and (iii) navigating transformations to avoid, or allow escape from, undesirable socio-ecological states. These three strategies are overlapping and complementary. They require more proactive and flexible approaches to defining the future state of the planet than have characterised either exploitative or equilibrial resource management paradigms of the past (Chapin, et al., 2009).

As mentioned earlier, an ‘urban sustainability approach’ (Wu, 2014) is one of the recently evolved approaches of urban ecology that investigates the link between biodiversity, ecosystem functions (or processes), ecosystem services and human well-being in urban landscapes. This approach focuses on the relationships between these four components and urbanisation across both spatial and temporal scale (Wu, 2014). The concepts of urban sustainability and urban resilience are argued to be complementary as they both place an emphasis on the non-linear dynamics and the importance of adaptive responses in urban systems (Walker & Salt, 2012; Wu, 2014).

Human (or social) resilience can be defined as ‘the ability of individuals, communities and governments to deal with shocks and stresses (Carabine, et al., 2015, p. iii). The link between ecosystem services and human (social) resilience is widely discussed in the scholarly literature (Wu & Wu, 2013; Carabine, et al., 2015), but there is limited evidence in practice that shows how ecosystem services link with a specific element of resilience in social systems (e.g. flexibility, diversity, cross-scale interactions, safe failure, or
self-organisation). However, the contribution of ecosystem services to resilience outcomes (e.g. enhancing social capital and well-being) is better understood (Carabine, et al., 2015; Butler & Oluoch-Kosura, 2006).

The role of ecosystem services in reducing exposure to natural hazards and thus contributing to resilience outcomes (e.g. reducing exposure to risk and enhancing adaptive capacity) has been the basis for investment in using ecosystem services for climate change adaptation and disaster risk reduction (Carabine, et al., 2015; Spalding, et al., 2014b). Cycling and storing carbon, as one of the climate regulatory services provided by ecosystems, is also argued to contribute to the ‘basic needs, health and wellbeing’ as one of the outcomes of social resilience (Carabine, et al., 2015).

The Convention on Biological Diversity (CBD) (2009) has developed and is widely promoting the concept of ecosystem-based adaptation (EbA). EbA is an approach based on using ecosystem services for climate change adaptation (Jones, et al., 2012). As reported by the CBD (2009, p. 9) ‘ecosystem-based adaptation which integrates the use of biodiversity and ecosystem services into an overall adaptation strategy, can be cost-effective and generate social, economic and cultural co-benefits and contribute to the conservation of biodiversity’. There is growing evidence that the costs of ecosystem-based approaches are greatly balanced with the benefits they will deliver (Duarte, et al., 2013; Spalding, et al., 2014b; Carabine, et al., 2015; De Groot, et al., 2013).

EbA interventions can be designed to address soil loss, erosion, flooding and natural hazards (Doswald, et al., 2014). The following benefits are identified as contribution of EbA practices to human resilience (Carabine, et al., 2015, p. 16):

- **Social**: improved and secure livelihoods, social cohesion and community, new or preserved recreation areas, better quality land for agriculture/livestock, better water security and protection from loss and damage;
- **Environmental**: biodiversity conservation, carbon sequestration and mitigation benefits, land erosion and degradation prevention, habitat creation and restoration, and mitigation of microclimate variability;
- **Economic**: damage costs prevented, new or improved income, profits, savings compared to alternative adaptation approaches, and income from subsidies’

Multifunctionality (delivering multiple functions or ecosystem services); redundancy and modularisation (delivering similar or backup functions by multiple elements); bio and social diversity; multi-scale networks and connectivity and adaptive planning and design are identified as strategies for building resilience in urban regions (Ahern, 2011). Identifying specific resilience outcomes and their links with ecosystem services can also facilitate measuring the value of ecosystem services for building resilience (Carabine, et al., 2015).

### 2.7. Land use change and environmental outcomes

There is a general consensus in the literature that urbanisation can alter ecosystem functions/processes, and consequently ecosystem services, through land conversion (Alberti & Marzluff, 2004; Alberti, 2008). Land use change is an unavoidable process in urban ecosystems due to the dynamic interactions between socio-economic and biophysical processes operating across multiple scales (Alberti, 2008). In practice, while urban areas rely on ecosystem services of their landscapes, urban growth cannot happen without affecting its natural setting in various forms such as land conversion, use and degradation of natural resources and generating wastes and emissions (Medley, et al., 1995; Alberti & Marzluff, 2004). This way, urban sprawl as a consequence of rapid urbanisation reduces the resilience of a system to transition from wholly natural to wholly anthropogenic land cover (Alberti, 2008).

Given the potential irreversible loss of ecosystem services as a result of land use change and urbanisation, approaches such as the “Precautionary Principle” (as discussed earlier in Section 2.3) and “Environmental Policy Integration (EPI)” can inform and guide decision-making process towards achieving environmental
outcomes while providing for development (Walker, et al., 2003; Cooney, 2004; Runhaar & Driessen, 2012). As defined by Lafferty and Hovden (2003, p. 9), EPI is “the incorporation of environmental objectives into all stages of policymaking in non-environmental policy sectors, with a specific recognition of this goal as a guiding principle for the planning and execution of policy; accompanied by an attempt to aggregate presumed environmental consequences into an overall evaluation of policy, and a commitment to minimise contradictions between environmental and sectoral policies by giving principled priority to the former over the latter.” This concept is further discussed in Section 2.9.

By highlighting the importance of cities contribution to delivering ecosystem services, the concept of ‘Positive Development’ (PD) (Birkeland, 2008) argues that development needs to increase ecological carrying capacity and biodiversity in urban areas and deliver net positive outcomes to nature and communities. The concept posits that this would enable cities to contribute to ecological resilience and sustainability of urban areas (Birkeland & Knight-Lenihan, 2016).

Addressing trade-offs between development and environmental outcomes is a crucial part of the impact assessment process which aims to identify the environmental consequences of development activities/projects (known as Environmental Impact Assessment, EIA) (Morrison-Saunders & Pope, 2013) or policies (known as Strategic Impact Assessment, SEA) (Gibson, 2013). Discussions around trade-offs in the context of New Zealand’s legislative and planning frameworks are provided in Sections 6.3.1 and 6.3.2.

To achieve sustainability, the impact assessment process should promote ‘multiple reinforcing gains’ rather than simply ‘striking a balance’ between environmental and socio-economic outcomes (Gibson, 2006). This way, impact assessment can be considered as a ‘sustainability assessment’ which is simply defined as ‘any process that directs decision-making towards sustainability’ (Morrison-Saunders & Pope, 2013, p. 2).

From this perspective, in order to make a positive contribution to sustainability, development should deliver ‘net gains’ rather than ‘no net losses’ (Morrison-Saunders & Pope, 2013). To achieve this, trade-offs need to be inherent in all decisions made within different stages of an impact assessment process including screening, scoping, approval, monitoring and follow-up. Whereas, conventional impact assessment practices often consider those trade-offs that are only evident at the point of the approval decision (Pope, et al., 2004). Similarly, and under the concept of ‘Positive Development’ (PD), Birkeland and Knight-Lenihan (2016) highlight the importance of accounting for the full life-cycle impacts of development including the adverse effects of resource extraction, transport, storage, construction and disposal.

Establishment of thresholds that aim to maintain and improve ecosystem services can help to manage future land use dynamics by influencing land uses that cannot comply with established standards (Landcare Research, 2011). For example, setting a limit for total nutrient or sediment loads to estuaries (in order to protect and manage coastal wetlands) may influence changes in land use within catchments through reducing the area of impervious surfaces, enhancing public green spaces and limiting agricultural activities. This may also indirectly affect the existing use rights of properties in catchments.

The notions of ‘avoidance’, ‘minimisation’, ‘rehabilitation’ and ‘offset’, as different stages of ‘mitigation hierarchy’ (Mitchell, 1997)\(^9\), are often used to understand the trade-offs through impact assessment process (Morrison-Saunders & Pope, 2013). The mitigation hierarchy (Figure 2.7-1) starts with attempts to avoid the adverse effects of development and continues with efforts to minimise and repair those effects that cannot be avoided. The last option, i.e. offset addresses the residual adverse effects that are remained after all the previous steps.

\(^9\) The mitigation hierarchy is originally rooted in the five categories of mitigation measures (avoid; reduce; repair; compensate; and enhance) identified by Mitchell (1997) (Cited in Tinker, et al., (2005)).
Offset is argued to be a ‘deliberate form of trade-off in which a negative impact with respect to one factor may be compensated for by an enhancement in another, and a loss is traded off for a greater gain elsewhere’ (Gibson, 2006, pp. 6-10). Thus, offsets explicitly introduce the idea of achieving a net gain rather than just minimising losses. The notion of ‘net gain’ as an ideal outcome of offsets can therefore align well with the net sustainability benefits goal of sustainability assessment as mentioned earlier (Morrison-Saunders & Pope, 2013).

Environmental compensation can be defined as ‘the positive environmental measures of comparable worth to maintain the overall stock’ (i.e. natural capital) (Cowell, 1997, p. 293). The terms ‘offsets’, ‘environmental offsets’ or ‘ecological compensation’ have been also used as synonymous with ‘environmental compensation’ in some literature (Persson, 2013; Villarroya, et al., 2014; McKenney, 2005). However, the Business and Biodiversity Offsets Programme (BBOP), considers ‘offset’ as a specific kind of ‘compensation’ that is designed to achieve ‘no net loss’ or ‘preferably a net gain’, while ‘compensation’ is not designed to achieve ‘no net loss’ (BBOP, 2013).

Achieving multiple reinforcing gains (environmental, social and economic benefits or advancement) from development is seen as a ‘win-win-win’ approach that promotes sustainability (Morrison-Saunders & Pope, 2013). The following section provides a brief review of the current concepts and approaches that provides for integration between climate change mitigation and adaptation outcomes with those of development.

2.8. Integrative approaches focusing on triple-wins

Analysis of the existing literature indicates a growing tendency especially at regional and country level towards approaches that emphasise the adoption of more integrated climate change policies and responses towards achieving triple-wins, i.e. generating climate adaptation, mitigation and development benefits (Klein, et al., 2005; Tol, 2005; Spencer, et al., 2016). This emphasis is based on the idea that with limited resources available to invest in response to climate change and the need for cost-effective measures, policy mixes that seek triple-wins should promote efficient use of existing capacities and resources to minimise potential trade-offs and inconsistencies and enhance complementarities and synergies between mitigation and adaptation policies with those of development (Klein, et al., 2007; Spencer, et al., 2016; Thornton & Comberti, 2017).
The need for integration of climate policies into urban and spatial planning is associated with the inherent nature of planning issues that involves uncertainty, complexity and normativity (Hartmann, 2012). The primary practical challenge with unifying adaptation and mitigation is that they operate at different spatial and temporal scales and are perceived as competing over means of implementation or limited financial resources (Hamin & Gurran, 2008). Moreover, the conventional perception of inverse relationship between economic development and nature conservation and lack of an agreed operational framework that links theory to practice have made it difficult to reconcile the three goals without or with minimum need for trade-offs (Connop, et al., 2016; De Groot, et al., 2010; Dahlberg & Burlando, 2009). As discussed in Section 2.3, planning for climate change adaptation requires dealing with and managing uncertainty associated with the dynamic nature of climate risks. This adds to the complexity of integrating different objectives.

The concept of triple-wins originated from the concept of ‘climate-smart agriculture’, defined by the Food and Agriculture Organisation of the United Nations (FAO, 2010, p. ii) as “agriculture that sustainably increases productivity, resilience (adaptation), reduces/removes greenhouse gases (mitigation), and enhances achievement of national food security and development goals”. This concept is articulated by various integrative resilience-centred approaches including ‘Climate Compatible Development’ and ‘Climate-Smart Landscapes’, which are discussed below. These approaches articulate triple-wins as single policies that deliver multiple benefits for climate adaptation, mitigation and development (including ecosystem conservation) (Tompkins, et al., 2013).

The concept of ‘Climate Compatible Development’ (CCD) (Mitchell & Maxwell, 2010) is an effort to integrate both adaptation and mitigation strategies with those of development. This concept brings together and builds on the concepts of ‘climate resilient development’ and ‘low carbon development’ (Figure 2.8-1). CCD is defined as “development that minimises the harm caused by climate impacts, while maximising the many human development opportunities presented by a low emission, more resilient, future” (Mitchell & Maxwell, 2010, p. 1). CCD is an emerging field of research and policy that was driven, among others, by a need to assist international funding organisations in financing integrated programmes and projects in developing countries that aim to achieve triple-wins (Ellis, et al., 2013).

The concept has also been used in developed countries for local financing of CCD projects implemented by private sectors e.g. in United States and Germany (Whitley & Mohanty, 2013; Whitley & Mohanty, 2012) or for broader resilience and risk management programmes, e.g. in the Netherlands (Veraart, et al., 2014). The concept has also been applied to coastal management. For example, a study by Huxham, et al., (2015) in Kenya concluded that CCD has a significant potential for generating large economic and environmental gains, in contrast to the status quo.

Figure 2.8-1 illustrates the conceptual relationship between development, mitigation and adaptation. The middle section in the figure illustrates the overlap between the three strategies or policies where they can mutually reinforce each other to produce ‘triple wins’. At the core of this approach is the concept of co-benefits and synergies that is shown in the lower area of the figure where adaptation and mitigation strategies overlap.

Another advancement is the transition towards investments in ‘Low-Carbon Climate-Resilient Development’ (LCCRD) (Corfee-Morlot, et al., 2012), which is essentially the same as CCD with a stronger focus on infrastructure. This approach is a further effort to strengthen the links between mitigation, adaptation and development towards achieving climate compatible development. LCCRD aims to encourage and reinforce investments in greening built-infrastructure (e.g. in the energy, transport, water and building sectors) and improving green infrastructure (e.g. through urban greening to enable achieving both low-carbon and climate-resilient development).
Integrated Coastal Zone Management (ICZM) (or Integrated Coastal Management, ICM) is considered as one of the mechanisms which can support implementation of CCD in coastal areas (CDKN, 2013). The concept of ICZM was first established in 1992 during the Earth Summit of Rio de Janeiro (or the United Nations Conference on Environment and Development, UNCED) and seeks to achieve sustainability through balancing the interests of socio-economic development and conservation of natural resources in coastal areas (Karim & Hoque, 2009). As defined by Post and Lundin (1996, p.1), ICZM is “a process of governance and consists of the legal and institutional framework necessary to ensure that development and management plans for coastal zones are integrated with environmental (including social) goals and are made with the participation of those affected”. The principles of ICZM, as highlighted in the UNCED’s Agenda 21, include economically efficient resource usage; sustainable resource usage and precautionary action (Turner & Bower, 1999).

ICZM provides for a dynamic and adaptive management process that places an emphasis on enhancing knowledge of the behaviour of coastal processes and of human behaviour and value of coastal ecosystems (Turner & Bower, 1999; Post & Lundin, 1996; Karim & Hoque, 2009). It is, therefore, recognised as the most appropriate process to deal with the impacts of climate change in coastal areas (IPCC, 2007b). To overcome the sectoral and intergovernmental fragmentation in management of coastal areas, ICZM encourages horizontal and vertical institutional integration and provides for the best use of available science and techniques\(^\text{10}\) as well as adoption a precautionary approach to deal with the complexities and uncertainties exist in the coastal zones (Cicin-Sain, et al., 1995; Turner & Bower, 1999).

ICZM also provides for an ecosystem-based approach to management of coastal resources which involves considering the whole ecosystem, including humans and the environment and their continuous interaction, instead of managing one issue or resource in isolation (UNEP, 2010). Ecosystem and landscape approaches are types of holistic approaches that seek to reduce trade-offs and enhance synergies between multiple objectives including social, economic and environmental (Minang, et al., 2015). They can therefore contribute to sustainable development and socio-ecological resilience (Minang, et al., 2015). As discussed before, holistic approaches can help address wicked problems. It is argued that integrated approaches such as ‘ecosystem’ and ‘landscape’ approaches can help to manage wicked climate-related problems and promote climate-resilient pathways (Minang, et al., 2015). Considerations of ICZM regarding the use of an integrated and precautionary approach in management of coastal resources have informed the recommendations proposed in this research (Chapter 8).

\(^{10}\) Examples included risk assessment, economic valuation, vulnerability assessments, resource accounting, benefit-cost analysis and outcome-based monitoring (Cicin-Sain, et al., 1995).
The ecosystem approach is defined as ‘a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way’\textsuperscript{11}. Landscape approaches have been referred to as ‘a set of concepts, tools, methods and approaches deployed in landscapes in a bid to achieve multiple economic, social, environmental objectives (multifunctionality) through processes that recognize, reconcile and synergize interests, attitudes and actions of multiple actors’ (Minang, et al., 2015, p. 8). Embedding an ‘ecosystem’ or ‘landscape’ approach in land use planning and decision-making, will therefore require a robust understanding of the multiple ecosystem services and how they may be affected by external drivers and pressures (Haines-Young & Potschin, 2010; Minang, et al., 2015).

The recent emerging ‘Climate-Smart Landscapes’ concept (Scherr, et al., 2012; Harvey, et al., 2014) builds upon the multifunctionality of natural landscapes and provides for a ‘landscape approach’ to manage interconnected components of the landscape in order to deliver multiple benefits such as biodiversity support, mitigation of natural disasters, carbon sequestration and storage and supporting sustainable commercial activities. Climate-smart landscapes are defined as the ‘landscape actions and processes that seek to integrate change mitigation and adaptation alongside multiple social, economic and environmental objectives’ (Minang, et al., 2015, p. 10). Based on this approach, landscapes are viewed as ‘learning laboratories’ where the impacts of climate change and socio-economic development can be felt and the response tested (Minang, et al., 2015). It is argued that the climate-smartness of a landscape depends on its (human, social, natural and physical) assets that are generally difficult to understand and manage.

As mentioned, ecosystems across urban landscapes provide multiple services. Many scholars, planners and urban practitioners have drawn attention to the multiple services that coastal ecosystems provide in urban landscapes (Duarte, et al., 2013; Costanza, et al., 2008; Crooks, et al., 2011). Aside from their diverse number of ecological, cultural and aesthetic services, coastal wetlands (including mangrove, saltmarsh and seagrass habitats) provide a range of climate change services that can make a contribution towards adapting to and mitigating climate change while providing for development and conservation outcomes (Spalding, et al., 2014b; Duarte, et al., 2013). They can potentially enhance the capacity of urban systems to act as carbon sinks (pools) while simultaneously offering ways to create urban environments that are more resilient and adaptive to the impact of climate change (e.g. the role of coastal wetlands in protecting against storm surges and sea level rise that are anticipated with climate change) (Spalding, et al., 2014b).

This multi-functionality of coastal wetlands has increased attention towards the sustainable use and management of these ecosystems to contribute to climate change mitigation and adaptation while achieving biodiversity conservation outcomes (Duarte, et al., 2013; Crooks, et al., 2011). Within the context of climate compatible development and climate-smart landscapes, coastal wetlands have the potential to assist in achieving triple-wins. This can be achieved by internalising the multiple benefits of coastal wetlands into mainstream conservation, development or climate change policies. While the long-term sustainability of coastal wetlands remains uncertain due to climate change (Section 2.3) this approach helps justify the creation of coastal buffers using multiple measures.

The approach of co-benefits has been used in several global environmental policies (Spencer, et al., 2016). For example, mechanisms such as ‘biodiversity in climate change funding’ uses biodiversity co-benefits to contribute to climate change mitigation and adaptation (OECD, 2014a). Most of the IPCC mechanisms including ‘Reducing Emissions from Deforestation and Degradation’ (REDD), ‘Carbon Finance’ and the recently developed approach of ‘Blue Carbon’ use climate change mitigation (or carbon sequestration and storage) co-benefit of biodiversity policies to contribute to climate change mitigation.

Ecosystem-based approaches for climate change adaptation (known as ecosystem-based adaptation discussed in Section 2.6) also provide for multiple co-benefits compared to engineering options (Jones, et

al., 2012; Spencer, et al., 2016). The following principles are identified by Spencer, et al. (2016) as a conceptual guide that can be used in developing climate change mitigation-adaptation strategies that seek to deliver multiple co-benefits and maximise human health, environmental, social and economic outcomes.

- Provide incentives at multiple scales (e.g. local, regional) and timeframes (e.g. immediate, long-term)
- Promote long-term integrated impact assessment to monitor the success and refine methods
- Foster multi-sectorial, multi-level networks to share the lessons learned

Spencer, et al., (2016) also posit that lack of comprehensive and integrated data on the co-benefits of mitigation-adaptation strategies should not prevent action. Alternatively, utilising current scientific knowledge and undertaking ‘no regret’ co-benefits projects can contribute to safeguarding social and ecological outcomes; while ongoing assessment provides an opportunity to improve knowledge and refine methods.

2.9. Tools for policy integration

The terms ‘integration’ and ‘mainstreaming’ are frequently used in various places in this chapter and throughout the entire thesis. They refer to one of the most fundamental axioms of environmental policy and governance namely Environmental Policy Integration (EPI) (discussed earlier in Section 2.7) that has been a central feature of the environmental and development discourse over the last five decades (Lafferty & Hovden, 2003; Jordan & Lenschow, 2010; Runhaar & Driessen, 2012). In the context of climate change, this axiom is articulated as Climate Policy Integration (CPI) (Adelle & Russel, 2013; Mickwitz, et al., 2009).

EPI is a subset of the general concept of policy integration (PI) originally discussed by Underdal (1980) in the context of integrated marine policies. The literature conceptualises CPI as a distinct form of EPI. There are various interpretations of the EPI and CPI. The literature refers to EPI as a policy process or process of governing, a policy output or outcome, as a set of concrete tools, a broad policy agenda or paradigm for policy change, and a first-order principle for operationalisation of sustainable development (Persson, 2013; Jordan & Lenschow, 2010; Adelle & Russel, 2013).

By Underdal’s definition (Underdal, 1980, p. 162), “a policy is integrated to the extent that it recognizes its consequences as decision premises, aggregates them into an overall evaluation, and penetrates all policy levels and all government agencies involved in its execution”. This is the basis of EPI definition provided by Lafferty and Hovden (2003) (See Section 2.7). The majority of studies adopt the same connotation for CPI by substituting ‘climate’ with ‘environment’ (in EPI definition), although this approach has been scrutinised by a few authors (for example see Adelle & Russel, 2013).

EPI is referred to as both a policy goal requiring policy change and a process necessary for achieving the change (Lafferty & Hovden, 2003). One of the premises of EPI, and as such CPI, is that they require reconstituting environmental protection as integral part of development process and therefore demand for a shift in policy priorities towards making protection of the life-supporting capacity of the nature a principled priority rather than subsidiary to sector policies (Lafferty & Hovden, 2003; Adelle & Russel, 2013; Eckerberg & Nilsson, 2013). According to Lafferty and Hovden (2003, p. 9) “the whole point of EPI is, at the very least, to avoid situations where environmental degradation becomes subsidiary; and, in the context of sustainable development, to ensure that the long-term carrying capacity of nature becomes a principal or overarching societal objective.”

Jordan and Lenschow (2010; 2009) have categorised the EPI instruments into three broad types of communicative, organisational and procedural. By their definition, communicative instruments include long term visions and strategies that set out directions for detailed reforms but leave the operation of policy integration to individual organisations and sectors. In contrast organisational instrument are those that alter the organisational context in which policy decisions are made e.g. through amalgamation or strengthening
of existing institutions and actors or creation of new actors and networks. Likewise, the procedural instruments seek to alter decision making processes e.g. extended right for environmental authorities, green budgeting and strategic environmental assessment (SEA). They argue that while communicative instruments are generally used for rhetorical purposes, the two other instruments especially the procedural ones tend to be more interventional and hence have in most cases been adopted with greater reluctance.

Jordan and Lenschow’s analyses of instruments within and across various jurisdictions and from political, institutional and cognitive perspectives concluded that communicative instruments have generally been more popular that the organisational and procedural ones. Their analysis also reveals the crucial role (and in most cases lack) of political support for the success of EPI and an evident absence of a comprehensive and coordinated design for institutionalisation of EPI. They maintain that for an EPI to be successful, it would be crucial to maintain the consistency and continuity of instruments and measures through the entire policy cycle.

A domain specific treatment of EPI in urban planning is provided by Runhaar, et al., (2009). They classified planning tools into two main groups of substance-oriented (i.e. tools that produce knowledge on the state of the environment or effects of future plans to support decision making e.g. EIA) and process-oriented (i.e. interactive planning tools that aim at facilitating dialogues and joint actions through the planning and policy process), and a third group of hybrid tools that have elements of the two other tools. In analysing the potential contribution of the hybrid tools to EPI, they distinguished three forms of EPI: (1) coordination (that seek to avoid conflicting policies or compensate for adverse impacts of sector policies on environment e.g. environmental zoning and mitigation of environmental impacts); (2) harmonization (that seek to align environmental objectives with sectoral objectives to promote synergy); and (3) prioritization (making environmental objectives a principled priority e.g. preserving particular areas for nature or taking precautionary principle to prevent likely environmental degradation). Runhaar, et al., (2014) also classified tools as legal (or regulatory) tools, economic tools and communicative (or informational) tools.

The precautionary principle (as discussed in Section 2.3) is viewed as a fundamental principle underlying environmental policy (Cooney, 2004). ‘Environmental mainstreaming’ generally has the same meaning as EPI. Dalal-Clayton and Bass (2009, p. 11) define ‘environmental mainstreaming’ as “the informed inclusion of relevant environmental concerns into the decisions of institutions that drive national, local and sectoral development policy, rules, plans, investment and action.” They have developed a comprehensive list of instruments and tools for environmental mainstreaming positing that in order for an effective mainstreaming, a mix of approaches and tools should be developed and applied to different stages of policy and planning cycle and over time, as shown in Figure 2.9-1. The tools and instruments used or suggested for EPI and CPI are diverse and wide-ranging (Runhaar, 2016). The literature on EPI and CPI suggest that integrative perspectives will need an appropriate mix of consistent instruments and tools across sectors or subsystems and all stages of the policy cycle (Adelle & Russel, 2013; Eckerberg & Nilsson, 2013; Jordan & Lenschow, 2010; Mickwitz, et al., 2009; Runhaar & Driessen, 2012; Runhaar, 2016; Howlett & Rayner., 2007; Candel & Biesbrock, 2016).
Howlett and Rayner (2007) have discussed design principles for the choice of policy instrument mixes with a view to identify appropriate mixes of policy instruments and avoid inconsistent outcomes due to the lack of coherence among policy instruments within complex decision-making and operational contexts. In a recent study, Candel and Biesbroek (2016) have proposed a multi-dimensional framework that distinguishes four dimensions of integration: (1) policy frames, (2) sub-system involvement, (3) policy goals, and (4) policy instruments; and discussed various manifestations of each dimension across a spectrum of policy integration from low to high degree of integration.

They argue that policy integration should be understood as a complex process with cross-scale dynamics and elements that are not necessarily in accord with each other. Policy integration from this perspective seeks to address cross-cutting social problems that also involve high degree of uncertainty, deadlock interaction patterns and controversy among stakeholders (or wicked problems as discussed earlier). This complexity of the policy integration process implies that in some instances even lower degrees of integration may be most feasible and appropriate for the governance of wicked problems (Candel & Biesbroek, 2016).

A comprehensive review and analysis of tools that address Climate Compatible Development (CCD as discussed earlier in this chapter) or its related aspects was provided by Climate and Development Knowledge Network (CDKN, 2011). The study identified a diverse range of tools that only a few simultaneously covered mitigation, adaptation and development, while the majority of tools focused primarily on the earlier stages of the policy cycle. The tools were mapped against various stages of the policy cycle and a number of thematic areas that represented the overlaps between mitigation, adaptation and development as illustrated in the concept of CCD (Figure 2.8-1). This study concludes that the choice and application of tools is context dependent, and suggests that an appropriate mix of tools for integrated
planning for CCD would be the one that consistently span across all stages of the policy and planning cycle.

2.10. Linking theory to practice: A simplified conceptual framework

There is a large body of research on the relationship between theory and practice in planning. The majority of the articles written on this subject have identified and discussed a ‘disconnect’ between theory and practice and attempted to fill in the gap (Harrison, 2014; Pierre, 2016; Allmendinger, 2002). Many authors have questioned and scrutinised the utility of theory for addressing real planning problems (Binder, 2011). Hence there is a wide spectrum of arguments about the usefulness of theory for planning, with some overly concerned about realism and the primacy of empirically based planning theory, while others privilege theory over practice and are hence labelled anti-realism (Binder, 2011; Harrison, 2014).

The various perspectives discussed in the preceding sections provide useful insights into the current understanding of the linkage between urban sustainability, resilience, adaptation and vulnerability to climate change, ecosystem services of urban natural resources and land use planning within the broader context of urban climate change governance (Bulkeley, 2010). As discussed, these approaches are principally designed to inform and guide policy processes and decisions through various stages of the policy and planning cycle towards achieving win-win and triple-wins policy outcomes. Putting these overarching frameworks into practice would entail some form of policy integration or mainstreaming and therefore requires appropriate policy instruments and planning tools as discussed in Section 2.9.

Drawing on the literature, especially CDKN (2011), Tompkins et al. (2013), Runhaar (2016) and Candel and Biesbroek (2016), and the theoretical discussion in previous sections, a conceptual framework, as depicted in Figure 2.10-1, was developed.

This framework illustrates a series of hypothetical scenarios where various possible win-win and triple-win outcomes can be achieved under different priority and trade-offs between a set of competing goals. In the context of the discussions in this chapter, the prioritisation involves trade-offs between the four competing goals of mitigation, adaptation, development and conservation. The square graphs represent the pairwise dichotomies between development (i.e. growth and social-economic wellbeing) and conservation (preservation of natural resources); and between mitigation (emission reduction and enhancing carbon) and adaptation (building resilience and adaptive capacity).

The three circles build on the rubric of CCD and represent overlaps between mitigation, adaptation and development. Larger overlap represents stronger integration between the three strategies or goals and the size of circles represent the position of the associated strategy or goal in the priority squares. The scenarios are ordered in a way that the narratives at the lower part of the leftmost and rightmost hypothetical scenarios respectively describe the lowest and highest ends of the scales for each key policy dimension discussed in this section.

The conceptual framework as described above is principally built on the perspectives that link urban sustainability to climate resilience and ecosystem services, which are reflected in the descriptions of the high and low ends of the spectrums for policy frames. This conceptualisation provides a guidance to identify the key parameters and variables that may influence the process of policy integration or mainstreaming within a given planning context. These will inform and guide the methodology and analytical framework in the next chapter.
A: Adaptation – M: Mitigation – C: Conservation – D: Development

Figure 2.10-1. A simplified conceptual framework illustrating possible states of policy integration

Note: Corporate goals are the goals that are adopted within sectors with a strategic view to addressing a broader concern about a cross-cutting problem (Rainey, et al., 2015).
3. Research design and methodology

3.1. Introduction

This chapter describes the research design, outlines the methods used to answer the specific research questions and defines the methodological basis for choosing those methods. As mentioned in Section 1.2, this research is addressing the following five specific questions:

A. How do coastal wetlands in the Auckland region currently and potentially contribute to climate change mitigation and adaptation?
B. Whether and how the climate change services of coastal wetlands are taken into account in current coastal resource management and climate change response policies and plans in New Zealand and Auckland, and what are the key challenges and opportunities?
C. What can be learned from the experiences in other jurisdictions in terms of the initiatives and mechanisms that can assist in incorporating climate change services of coastal natural resources into land use, resource management and climate change policies?
D. How do the policies and practices in other jurisdictions compare to the case of Auckland and New Zealand? and
E. Depending on the outcome of A to D, investigate what policy options and planning instrument mixes can be applied to Auckland for mainstreaming climate change benefits of coastal wetlands into land use and resource management decision-making processes.

From a methodological perspective, while many of the social, behavioural and political aspects of urban planning are subjects of interest for qualitative research, understanding the characteristics, form, function and dynamics of urban systems and their physical and natural resources often requires rigorous quantitative techniques and hence tend to fall into the domain of science rather than policy. Notwithstanding the difference, the two facets of urban planning are intertwined and inextricable. Theories that advocate evidence-based policy and planning practices provide frameworks that describe the interactions between policy and science (Head, 2010).

These theories posit that policies that shape and transform urban landscapes need to be plausibly informed by research evidence in order to effectively lead to sustainable outcomes (Krizek, et al., 2009). However, there are many challenges involved in embedding science and evidence in policy and planning processes including the divergence in the interpretation of the term ‘evidence’ within this context (Krizek, et al., 2009).

This research is a crossroad between the two domains of science and policy within a context of urban and environmental planning, and therefore addresses the challenge of bridging the theory and practice from both theoretical and methodological perspectives. Given this nature of the research, a combination of qualitative and quantitative research methods including literature review, content analysis of policy documents, case studies, statistical and spatial (GIS) analyses and questionnaire survey was used in order to answer the research questions.

These methods were used to gain an in-depth understanding of (i) climate change mitigation and adaptation values of coastal wetlands, (ii) the current policies and approaches to management of these ecosystems in the context of climate change and, (iii) the existing opportunities, gaps and policy options to incorporate the climate change values of coastal wetlands into the planning system.

3.2. Research design

Urban planning is an interdisciplinary science and a technical and political process (Taylor, 1998; Krizek, et al., 2009). The choice and application of methods and evaluation frameworks are context dependent. This means that to meet specific objectives of a research, any given combination of issues and problems
in an urban context is to be analysed and evaluated using a customised set of methods and techniques (Bracken, 2014). The methodology of this research follows this tradition.

Figure 3.2-1 shows the research design and process which is based on the input/output relations between various components of the research. Arrows in the graph indicate the key inputs and outputs between boxes. The research process starts with a problem formulation, framing and scoping phase followed by detailed observation, review, analysis and discussion, and concludes with a last phase of interpretation and synthesis. Chapters 4 and 5 outline findings of the analytical studies carried out to address the scientific aspects of this research (Questions A & B). Likewise, chapters 6 and 7 represent outcomes of the work carried out to analyse and discuss the policy aspects of this research (Questions C & D) and include discussion of issues, comparison of cases and the available opportunities. Therefore, appropriate qualitative and quantitative methods are applied to chapters corresponding to the nature of issues and questions that are addressed in each chapter. Outputs of all analytical works and theoretical discussions are fed into Chapter 8 that outlines the key planning considerations and their implications.

Chapter 9, as the last chapter, combines and summarises findings of the research including the reviews, analyses, interpretations and discussions.

The research design does not follow a distinct sequential process, and rather involves a series of recursive modifications, where the output of a review and analysis is used as an input to a preceding phase in order to refine or confirm the assumptions or interpretations.
Figure 3.2-1. Research design and process (Source: Author)
3.3. Methods used to address the research questions

3.3.1. Systematic literature review and data collection

3.3.1.1. Review of literature on carbon sequestration by land use categories

As mentioned in Chapter 1, this research initially started with a review of the literature addressing the interaction between land use change and carbon sequestration capacity of different land use categories. The primary objective was to examine whether it is possible to provide planners with a way of assessing the implications of land use change on carbon sequestration. The first essential step to achieve this objective was to analyse and understand the variations in carbon sequestration between different land categories.

A systematic literature review process carried out in August 2013 to find out the relevant literature and collect data to estimate the carbon sequestration potential by different land use categories. The land use categories selected for this research were urban trees\textsuperscript{12}, natural forests\textsuperscript{13}, plantation, wetlands, agricultural lands and urban turfgrass/lawns. These categories are selected mainly because their carbon sequestration potential is well recognised and addressed in the literature. They are also consistent with the broad land use categories\textsuperscript{14} identified in the 2003 IPCC report (IPCC, 2003). These categories are a mixture of land cover (e.g. natural forest, wetlands) and land use (e.g. plantation, urban trees, agricultural lands, urban turfgrass/lawns) classes. However, for convenience and to be consistent with the IPCC approach, this research refers to these categories as ‘land use’ categories (IPCC, 2003, p. 2.5). Among the selected land use categories, urban trees and urban turfgrasses/lawns are mainly associated with urban settings, while agricultural lands, natural forests, wetlands and plantation are mostly found in rural areas.

The literature review process included a systematic search of local (New Zealand and Auckland) and global literature and is explained in Appendix 1. The findings of this review to explore the initial idea of the research are provided in Chapter 4.

3.3.1.2. Review of literature on carbon sequestration and storage (CS&S) by coastal wetlands

A systematic literature review carried out between December 2014 and December 2015 to collect data to provide an estimate of the potential of Auckland’s coastal wetlands for CS&S. Findings (provided in Chapter 5) were then used as a basis for policy and planning discussions in Chapters 6 to 8 written between January 2016 and February 2017. The literature addressing the climate change benefits of coastal ecosystems have mainly focused on saline coastal wetlands including mangrove and saltmarsh habitats (Rogers, et al., 2014; Chmura, et al., 2003). However, seagrass beds have been also grouped with mangroves and saltmarshes as vegetated coastal habitats that can contribute to climate change mitigation and adaptation (Duarte, et al., 2013; Mcleod, et al., 2011).

\textsuperscript{12} There is no clear and commonly agreed definition for ‘urban trees’ or ‘urban forests’ within the literature. The combination of words implies that the terms refer to individual stands or communities of trees within urban areas or cities. The urban tree category in this research includes studies that have reported carbon sequestration rates for ‘urban trees’ or ‘urban forests’, without specifying whether they were natural or planted.

\textsuperscript{13} The natural forest category includes studies that have provided information on carbon sequestration by ‘forests’, and have either specified that the forest under study was a natural forest, or just made no mention of the terms that could suggest otherwise; such as ‘plantation’, ‘secondary forest’ or ‘planted forest’.

\textsuperscript{14} As identified in the IPCC report, the six broad land use categories include forest land, cropland, grassland, wetlands, settlements and other land (e.g. bare soil, rock, ice, and all unmanaged land areas that do not fall into any of the other five categories).
No categories of New Zealand’s coastal wetlands are identified within the literature. Coastal wetland types in the Auckland region include mangrove forests, saltmarsh habitats and salt meadows\(^{15}\) (Auckland Council, 2013). To be consistent with the global literature, this research considers mangrove forests, saltmarsh wetlands and seagrass beds as the categories of saline coastal wetlands. However, given the lack of local and national data on the aerial extent of seagrass ecosystems, they were excluded from the CS&S estimates in this research. The information regarding mangrove and saltmarsh species in the Auckland region is provided in Appendix 2. The systematic literature review involved collection of data on the rate of CS&S for these species from both local and global literature and is explained in Appendix 2.

### 3.3.2. Using ArcGIS 10.1 to estimate aerial extent

To estimate the total CS&S capacity of Auckland’s coastal wetlands, it was necessary to have information about the aerial extent of these ecosystems in the Auckland region. At the time of writing this part of the research, there was no reported information about the area of mangrove, saltmarsh and seagrass ecosystems in the Auckland region. The Auckland Low Carbon Action Plan 2014 (Auckland Council, 2014) provides an estimate of the area of saltwater wetlands in Auckland (22,500 ha); however, no definition of saltwater wetlands is provided in the Plan.

The available national data included GIS shapefiles of the New Zealand Land Cover Database (LCDBv.4, 2014) (Landcare Research, 2014). Similar to the situation in Auckland, the national database in New Zealand also lacked information for seagrass habitats. The LCDBv.4 2014 included a specific GIS shapefile for mangrove ecosystems across New Zealand, but did not include such information for saltmarsh habitats. It rather included a specific land use class for ‘herbaceous saline vegetation’ that represents salt-tolerant plants found in estuarine or coastal wetlands (Thompson, et al., 2003). This class includes saltmarsh species and therefore its GIS layer was used to estimate the area of saltmarsh ecosystems in Auckland. The analyses were carried out between October 2014 and January 2015. By the end of that period the information provided by the LCDBv.4 was the only information that could be used to estimate spatial extent of mangrove and saltmarsh habitats in the Auckland region.

### 3.3.3. Statistical analysis using SPSS 16

This section describes the statistical procedure (One-way ANOVA and LSD and Bonferroni Post Hoc tests) used in the research to analyse carbon sequestration data for different land use categories described in Section 3.3.1.1. The results are provided in Chapter 4, Section 4.3.

One-way ANOVA is a recognised and commonly-used method to identify whether there are any statistically significant differences between the means of more than two independent groups (Penny & Henson, 2006). One-way ANOVA can be used if there is only one independent (qualitative) variable (e.g. land use class) and one dependent (quantitative) variable (e.g. carbon sequestration)\(^{16}\) (Heron, 2009). It was used in this research to test whether or not the mean carbon sequestration values among the six land use categories were significantly different. Before comparison of means through ANOVA, homogeneity of variances was tested using Levene's test (Levene, 1960). Levene's test is recognised to be more appropriate for testing homogeneity of variances compared to F-test (Rasch & Guiard, 2004). It is also available through SPSS and thus was used to assess the equality of variances across the selected groups.

---

15 Salt meadows grow in areas beyond saltmarshes where the land is drier (Wassilieff, 2006) and include flat mat-forming or turf-forming plants such as glasswort (Sarcocornia quinqueflora), sea primrose (Samolus repens), remuremu (Selliera radicans) and bachelors button (Cotula coronopifolia) (Auckland Council, 2013).

Data reported for biomass carbon sequestration (40 observations) and data on carbon sequestration in soil/sediment (50 observations) by different land use categories were independently analysed in SPSS 16. Results of the Levene’s test for biomass data confirmed homogeneity of variances among carbon sequestration values (p-value>0.05). One-way ANOVA indicated that there were significant differences in average values of biomass carbon sequestration among categories (p-value<0.05) (See Table 4.3-5). Given the homogeneity of variances, the Fisher’s least significant differences (LSD) and Bonferroni post-hoc tests were used to identify what categories show significant difference.

The LSD test was chosen because it is the most commonly used test for ANOVA (e.g. Bowers, et al., (2004); Hilton & Armstrong, (2006)). This method uses t-test for pair-wise comparisons between group means which has proved to be effective where limited number of categories exist (as it was the case in this research) (Carmer & Swanson, 1973; Brown, 2005). The t-test was used to identify the differences in mean values of biomass carbon sequestration between categories. The LSD post-hoc test is often accompanied by the Bonferroni test (e.g. Bowers, et al., (2004); Haynes, et al., (2004); Magogwe & Oliver, (2007)) which is a more conservative test and similar to the LSD uses t-test for pair-wise comparisons, but also controls the overall error rate (Hilton & Armstrong, 2006). Thus, a combination of these two post-hoc tests provides reliable results for the purpose of the statistical analysis in this research.

3.3.4. Content analysis of policy documents

Content analysis is a widely-used method in qualitative research (Hsieh & Shannon, 2005). It is used as a flexible method to interpret meaning from the content of text data and analyse information based on the objective of the research (Cavanagh, 1997). The primary objective of content analysis is ‘to provide knowledge and understanding of the phenomenon under study’ (Downe-Wamboldt, 1992, p. 314). It is therefore a useful method that can be used in this research to analyse information and the content of policy and planning documents.

The policy documents that were analysed included national and regional policies and strategies (both statutory and non-statutory) that (i) regulate the management and protection of coastal resources including wetlands, and (ii) address planning themes involving protection and management of natural resources in general and specifically the marine environment, response to climate change, and management of natural hazards.

At the central government level, the key selected documents included the Resource Management Act (1991) and its relevant amendment (the Resource Management (Energy and Climate Change) Amendment Act 2004); the Climate Change Response Act (2002); the New Zealand Coastal Policy Statement (1994 & 2010); the Building Act (2004); the Local Government Act (2002); the Civil Defence and Emergency Management Act, (2002); the National CDEM Strategy (2008); the National Policy Statement for Freshwater Management (2014); and cabinet papers, guidance and technical materials released by the Ministry for the Environment, Department of Conservation, Landcare Research; New Zealand Climate Change Research Institute (NZCCRI), and National Institute of Water and Atmospheric Research (NIWA).

The key planning documents at the local government level, i.e. the Auckland region, selected for the review in this research included the Auckland Plan (2012); the Proposed Auckland Unitary Plan (2013); the Auckland Unitary Plan (operative in part) (2016); the Hauraki Gulf Marine Park Act (2000); the Auckland CDEM Group Plans (2011-2016 & 2016-2021) and the Auckland’s Energy Resilience and Low Carbon Action Plan (Low Carbon Auckland). A wide range of other material and resources mainly those published by the Auckland Council, Landcare research and NIWA were also reviewed and analysed based on the objectives of the research.
Amongst the three approaches to content analysis (i.e. conventional\textsuperscript{17}, directed\textsuperscript{18} or summative) (Hsieh & Shannon, 2005) (Table 3.3-1), this research selected to use the summative content analysis which involves analysing documents using a number of keywords (certain words or themes) and interpreting the underlying context (Hsieh & Shannon, 2005). This approach to content analysis starts with quantifying (counting of) certain words or ‘manifest content’ (known as manifest content analysis) and extends to include latent content analysis which allows interpretation of the context (Zhang & Wildemuth, 2016). The approach used in this research is mainly focused on understanding the underlying context rather than focusing on counting the frequency of specific words. The steps for content analysis used in this research are explained in Appendix 3.

<table>
<thead>
<tr>
<th>Type of Content Analysis</th>
<th>Study Starts With</th>
<th>Timing of Defining Codes or Keywords</th>
<th>Source of Codes or Keywords</th>
</tr>
</thead>
<tbody>
<tr>
<td>Conventional content analysis</td>
<td>Observation</td>
<td>Codes are defined during data analysis</td>
<td>Codes are derived from data</td>
</tr>
<tr>
<td>Directed content analysis</td>
<td>Theory</td>
<td>Codes are defined before and during data analysis</td>
<td>Codes are derived from theory or relevant research findings</td>
</tr>
<tr>
<td>Summative content analysis</td>
<td>Keywords</td>
<td>Keywords are identified before and during data analysis</td>
<td>Keywords are derived from interest of researchers or review of literature</td>
</tr>
</tbody>
</table>

3.3.5. Questionnaire survey

Policy processes typically involve stakeholders with a diversity of interests, affiliations and levels of knowledge and understanding of the policy issues. Understanding the perspectives and positions of the people involved in a policy process can provide valuable input into the analyses of policies. With this view and to complement the qualitative documentary research and literature review in this research, a questionnaire-based survey was conducted between September 2014 and August 2015 to gain an insight into how the key stakeholders and participants in the policy processes of interest for this research perceive and evaluate the ecosystem services of Auckland’s coastal wetlands and how they differ in their perspectives to the issues and possible solutions.

The questionnaire method was used as a tool to identify and analyse the perceptions and knowledge of the issues among different groups of participants (Dahlberg & McCaig, 2010). The survey was therefore not intended to be used for statistical inference from sample to the general population. The survey process started with the design of the questionnaire, selection of participants and formal ethics approval\textsuperscript{19}, and was followed by distribution of the questionnaires among participants through email communications, collecting the completed questionnaires and analyses of the information.

\textsuperscript{17} Conventional content analysis is for grounded theory development. In this approach, coding categories are derived directly from the text data. It is often used when there is limited literature or theory on a phenomenon under study (Hsieh & Shannon, 2005).

\textsuperscript{18} Directed content analysis starts with an existing theory or relevant research findings about a phenomenon that is incomplete and would benefit from further description. In this approach, coding categories are identified based on the existing theory or research findings, but new codes will emerge during data analysis and the initial coding categories will be refined and modified (Hsieh & Shannon, 2005).

\textsuperscript{19} The Ethics approval is required for conducting the questionnaire survey in this research.
A semi-structured questionnaire approach was used to design the questionnaire. This method is a relatively cost-effective way to collect more detailed information from participants (Mathers, et al., 2007). The questionnaire contained nine questions including rating-scale questions, multiple-choice questions and open-ended questions (comment boxes). A standard format was used to maintain the logical sequence and relations between questions.

For the specific purpose of this questionnaire survey, experience in policy making, planning and research in the field of wetland management especially coastal wetlands in Auckland was a necessary condition for shortlisting potential participants for the questionnaire survey. The participants were also preferred to be familiar with the Proposed Auckland Unitary Plan provisions relating to coastal wetlands and climate change. Therefore, a ‘purposive’ sampling method (Palys, 2008) was employed to identify the survey participants. This method is a type of non-probability sampling methods that is generally used where the purpose is to choose participants that have expertise in the research area of interest and therefore would best be able to provide the needed information. A specific type of this method is ‘criterion sampling’ which involves selecting participants based on certain criteria.

Detailed information about the structure and content of the questionnaire and the process to identify the potential participants is provided in Appendix 4. A copy of the Ethics approval (Ref. 013770) from the University of Auckland’s Human Participants Ethics Committee (UAHPEC) is provided in Appendix 5. Summary of the responses to the questionnaire is provided in Chapter 6 (Section 6.5.5). More details of the responses are provided in Appendix 6.

3.3.6. Case studies

A case study method was selected as it helps “to derive an up close or otherwise in-depth understanding of a single or small number of cases set in their real-world context” (Yin, 2011, p. 4). The main case study and the focus of this research is the Auckland region in New Zealand while also considering a small number of other cases (jurisdictions in a number of selected countries), under a comparative case study approach (Mees & Driessen, 2011). This provides an opportunity to understand the knowledge and experience of the cases regarding the research interests (Yin, 2011). A comparative case study method is applied as it is a commonly used approach to increase the validity of research within the context of urban policy and planning (e.g. Sykes, (2008); Mees & Driessen, (2011); Goodrick, (2014)). It is also used to identify similarities or differences between a number of cases that seek a common focus or goal (Goodrick, 2014). This approach is appropriate where there is a need to compare two or more cases in order to provide more generalizable knowledge to answer the questions of “how” (Goodrick, 2014).

The comparative case study approach in this research was used to investigate (i) ‘how’ climate change services of coastal wetlands were considered in climate-related, urban development strategies of other jurisdictions and (ii) ‘how’ sea level rise and coastal hazards were addressed within land use and climate change policies in coastal areas. The first step involved the identification of cases through a systematic review of peer-reviewed and grey literature including an extensive review of the global initiatives/programs/networks working in the field of climate change and local government responses to climate change. This process is explained in the next section.

Second, the contexts in which the selected cases operate was studied through conducting a review of the national/local governance systems they are subject to as well as general climate change policy responses from central government that can provide guidance on climate change responses at the local/municipal level. Third, an in-depth review was conducted of the climate change adaptation and mitigation as well as coastal hazards management strategies to find out how they consider and recognise the climate change benefits of coastal wetlands and deal with land use planning and development in coastal areas. The findings were then summarized (Chapter 7) and used to identify similarities and differences between the cases and the case of Auckland.
The case studies selected for this research included the cities of New York and New Orleans in the US, the city of Jakarta in Indonesia, the state of Tamil Nadu in India and the city of Rotterdam in the Netherlands. These cases were identified through a systematic review of available resources which is explained in Appendix 7.
4. Land use change and its implications on carbon sequestration

4.1. Introduction

This chapter provides the findings of a systematic review of literature and statistical analyses that were carried out in the initial stage of this research to understand whether it is possible to provide planners with a way of assessing the implications of land use change on carbon sequestration. The methodology is explained in Chapter 3 (Section 3.3.1.1).

4.2. Land use change and carbon balance

The literature consistently identifies land use change as one of the main determinants of the balance of carbon within an ecosystem (Lal, 2004b). The debate is over how land use change could lead to a net gain or loss of carbon in an ecosystem and how carbon sequestration varies with changes in land use (Post & Kwon, 2000; Ingram & Fernandes, 2001; Guo & Gifford, 2002; Lal, 2004b). Carbon sequestration is referred to as ‘transferring atmospheric carbon dioxide \(\text{CO}_2\) into long-lived pools and storing it securely so it is not immediately re-emitted’ (Lal, 2004b, p. 1623).

As a complementary measure to the emission reduction from the source, climate response policies include measures to use biological sequestration or bio-sequestration capacities of urban environments (Lal & Augustin, 2011; Piana, 2013). Bio-sequestration includes direct removal of \(\text{CO}_2\) from the atmosphere by biological processes and through activities that maintain and enhance the capacity of biological resources to capture and store atmospheric carbon (IPCC, 2005). In practice, bio-sequestration has been largely focused on measures such as afforestation, avoided deforestation, reforestation and practices that enhance soil carbon and biological sinks (IPCC, 2005).

The process involves plants that capture atmospheric \(\text{CO}_2\) through photosynthesis and store carbon in their biomass (above- and below-ground) and soil (Jansson, et al., 2010) (Figure 4.2-1). The carbon stored in biomass is argued to be short-lived (decades to centuries) as it returns to the atmosphere when the plants decay or are harvested (Jansson, et al., 2010). Long-term (millennia) carbon sequestration occurs when the carbon captured by the above-ground biomass moves to the belowground biomass (root system) and enters the soil carbon pool (Jansson, et al., 2010) which includes soil organic carbon (SOC) and soil inorganic carbon (SIC) (Lal, 2004a). The term ‘soil carbon sequestration’ implies ‘removal of atmospheric \(\text{CO}_2\) by plants and storage of fixed carbon as soil organic matter’ (Lal, 2004a, p. 9). Carbon sequestration rate represents how much carbon can be sequestered by plants or soil over a certain period (Graham, et al., 1992; Niu & Duiker, 2006) and is often reported in the standard unit of \(\text{t} \text{C ha}^{-1} \text{yr}^{-1}\) (\(=\text{Mg} \text{C ha}^{-1} \text{yr}^{-1}\)) (Pregitzer & Euskirchen, 2004).

Combinations of (i) natural processes (e.g. respiration, growth, death and decomposition of organic matter) and (ii) human influences (e.g. management and maintenance activities, land use change and deforestation) determine the net carbon sequestration by a particular land use over a long-term period (Mitsch, et al., 2010; Nowak, et al., 2013).
Figure 4.2-1. A simplified illustration of the carbon cycle through soil, plant and atmosphere (Jana, et al., 2010)

A large number of studies have discussed the relationships between land use change and carbon sequestration (mainly soil carbon sequestration) (e.g. Ingram & Fernandes, (2001); Lal, (2004a) & (2004b)) or investigated changes in carbon sequestration and soil carbon pools (mainly SOC) of a particular land use in relation to land use change and/or management practices (e.g. Post & Kwon, (2000); Follett, (2001); Groenendijk, et al., (2002); Desjardins, et al., (2005); Silver, et al., (2004); Söderström, et al., (2014); Sharma, et al., (2014)). For example, results of a meta-analysis by Guo and Gifford, (2002) indicated that generally SOC declines after land use changes from pasture to plantation (-10%), native forest to plantation (-13%), native forest to crop (-42%) and pasture to crop (-59%). Findings of their research also revealed that SOC may increase after land use changes from crop to pasture (+19%), crop to plantation (+18%) and crop to secondary forest (+53%). According to the results of a study conducted by Post and Kwon, (2000), there was an increasing trend of SOC sequestration rates from temperate regions to subtropical regions when agricultural land is converted to secondary forest.

In New Zealand, Groenendijk, et al., (2002) pointed out that afforestation of pasture with Radiata pine (Pinus radiata) decreased SOC by 15% to a depth of 12–18 cm mainly due to increasing SOC mineralization. In the Cerrado region of Central Brazil, Neufeldt, et al., (2002) also observed that reforestation of pasture with pine led to a clear reduction of SOC compared to eucalyptus plantation. Desjardins, et al., (2005) discussed that converting cropland into perennial forage may result in a substantial increase in carbon sequestration. The impacts of management practices on carbon sequestration have been mainly studied for agricultural lands and grasslands (Follett, 2001; Desjardins, et al., 2005; Salinger, et al., 2005; Zirkle, 2010). These studies indicated increased rates of carbon sequestration as a result of conservation tillage (i.e., no-till, ridge-till, and mulch-tillage), frequent fertilisation and irrigation and maintaining higher levels of residue cover on conventionally tilled croplands.

Overall, studies concluded that the amount of carbon released to or removed from the atmosphere as a result of land use change depends on multiple factors including land use history, plant species, land management strategies and soil organic carbon content (Post & Kwon, 2000; Guo & Gifford, 2002; Lal, 2004a; Lal, 2004b). Despite the sizable number of studies addressing variations in carbon sequestration and SOC associated with changes in land use, comparative studies of the average rates of carbon sequestration between different categories of land use are lacking.
4.3. Carbon sequestration by land use categories

4.3.1. Data sources

As explained in Chapter 3 (Section 3.3.1.1), the land use categories selected in this review were urban trees, natural forests, plantation, wetlands, agricultural lands and urban turfgrass/lawns.

The average values of carbon sequestration from 121 observations, reported in peer-reviewed and grey literature, were included in the dataset (Appendix 1, Table A1-3). The search strategy is explained in Chapter 3 (Section 3.3.1.1). Of the total number of 121 observations selected in this research, 100 observations (including 40 observations on carbon sequestration rates in above- and below-ground biomass, 50 observations on the rate of carbon accumulation in soils/sediments and 10 observations on the rate of carbon sequestration in both biomass and soil) were reported for gross carbon sequestration; whereas 21 observations were reported for net carbon sequestration (Table 4.3-1). Studies on soil carbon sequestration have mostly measured and reported SOC. As mentioned in Chapter 3 (Section 3.3.3.1), One-way analysis of variance (ANOVA) with LSD and Bonferroni Post Hoc tests, using SPSS version 16, were performed to detect differences in average rates of carbon sequestration between different categories.

<table>
<thead>
<tr>
<th>Category</th>
<th>Total</th>
<th>Gross carbon sequestration</th>
<th>Net carbon sequestration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Carbon sequestration in biomass</td>
<td>Carbon sequestration in soil</td>
</tr>
<tr>
<td>Urban tree</td>
<td>15</td>
<td>14</td>
<td>No data</td>
</tr>
<tr>
<td>Natural forest</td>
<td>22</td>
<td>8</td>
<td>No data</td>
</tr>
<tr>
<td>Plantation</td>
<td>34</td>
<td>11</td>
<td>11</td>
</tr>
<tr>
<td>Wetland</td>
<td>27</td>
<td>7</td>
<td>17</td>
</tr>
<tr>
<td>Agricultural land</td>
<td>16</td>
<td>No data</td>
<td>15</td>
</tr>
<tr>
<td>Urban turfgrass</td>
<td>7</td>
<td>No data</td>
<td>7</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>121</td>
<td>40</td>
<td>50</td>
</tr>
</tbody>
</table>

Carbon sequestration (gross) rate in biomass is often defined as the difference between the amounts of carbon stored in biomass in successive years (i.e. year y and year y+1) (Liu & Li, 2012; Nowak, et al., 2013). Annual biomass accumulation has been also estimated by dividing the amount of carbon stored in biomass over a period of time by the number of years (Lasco, et al., 2004). Some studies have only considered the carbon stored in above-ground biomass (i.e. trunk or stem, branch, leaf) while some other studies have taken into account the carbon stored in both above- and below-ground (root) biomass. To estimate carbon sequestration in biomass, studies have used biomass equations to calculate dry biomass of trees. The biomass can be converted to carbon using a biomass transformation coefficient of 0.5 (Dai, et al., 2013). The amount of carbon sequestered annually is then estimated based on the annual growth rates of tree DBH (diameter at breast height). Where biomass equations were only available for calculating above-ground biomass, a root-to-shoot ratio of 0.25 has been commonly used to estimate the whole tree biomass (Liu & Li, 2012).

A number of studies, mainly those focused on natural forests, have estimated carbon sequestration as a function of net primary productivity (NPP) and considered carbon sequestration as the annual ratio of carbon fixation in NPP (Zhao, et al., 2010; Kaul, et al., 2010). NPP is defined as ‘the balance between the light energy fixed through photosynthesis (gross primary productivity) and the portion lost through respiration and mortality, thus resulting in net carbon sequestration from the atmosphere by vegetation’ (Zhao, et al., 2010, p. 808). Based on this definition, the carbon sequestration estimated through NPP is considered as net carbon sequestration in this study.
In some studies, \((e.g.\) Chen, et al., (1999); Gielen, et al., (2013))\), carbon sequestration is estimated based on a measure of net ecosystem productivity (NEP). NEP is generally defined as the net balance between CO\(_2\) uptake and release (Black, et al., 2007; Gielen, et al., 2013). The following equations show the difference between NPP and NEP (Vetter, et al., 2005):

\[
\begin{align*}
\text{NPP} &= \text{GPP} - R_a \\
\text{NEP} &= \text{NPP} - R_h \\
\end{align*}
\]

GPP: gross primary production  
NPP: net primary production  
NEP: net ecosystem production  
\(R_a\): Autotrophic respiration by plant  
\(R_h\): Heterotrophic respiration by bacteria, fungi, and animals

NEP is mostly calculated by measuring the rates of photosynthesis and respiration (e.g. Chen, et al., 1999), while other sources of emissions (non-CO\(_2\) carbon fluxes) such as DOC (Dissolved Organic Compounds) leaching and VOC (Volatile Organic Compounds) emissions (e.g. Black, et al., 2007; Gielen, et al., 2013) or harvesting/thinning (e.g. Vetter, et al., 2005) are also considered in the calculation of NEP. Net carbon sequestration is also estimated by deducting the amount of carbon sequestrated due to tree growth from the amount of carbon lost due to tree mortality and decay (e.g. Dai, et al., 2013). Some studies (e.g. Herrero & Bravo, 2012) also included emissions due to harvesting to estimate the amount of lost carbon.

Alternatively, net ecosystem exchange (NEE) has been used in a number of other studies (e.g. Zha, et al., 2004; Ueyama, et al., 2011) to calculate net carbon sequestration. NEE is defined as the balance between carbon uptake through GPP and carbon release through both autotrophic (\(R_a\)) and heterotrophic (\(R_h\)) respiration (i.e. ecosystem respiration, Re). The following equations define the relationship between GPP, NEE and Re:

\[
\begin{align*}
\text{Gross primary productivity (GPP)} &= \text{NEE} - \text{Re} \\
\text{Re} &= \text{Ecosystem respiration} = R_a + R_h (\text{Coleman, 2010}) \\
\end{align*}
\]

These studies have used Eddy Covariance technique to measure CO\(_2\) flux.

Carbon sequestration in soil is estimated as a function of soil carbon content (mainly SOC) and changes in soil carbon over a period of time is used to estimate the amount of carbon sequestered per year (Qian & Follett, 2002; Hedley, et al., 2009; Fonseca, et al., 2012; Sierra, et al., 2013). Percentage of organic carbon (%OC) and the bulk density (BD) of the soil are used to estimate carbon sequestration over a period of time (Townsend-Small & Czimczik, 2010; Selhorst & Lal, 2013).

Where wetlands were the focus of studies (e.g. Bernal & Mitsch, 2012), sediment carbon sequestration (accumulation) is estimated by measuring the sediment organic carbon content and sediment accretion over a given period of time. To estimate sediment accretion over time, studies mainly used radiometric dating (radiocarbon/radioisotopic analyses) of soil cores with \(^{137}\text{Cs}\) and/or \(^{210}\text{Pb}\) to understand the timing and long-term rate of sediment accumulation (Choi & Wang, 2004; Mitsch, et al., 2010; Brevik & Homburg, 2004; Callaway, et al., 2012).

As shown in Table 4.3-1, there are variations in the number of observations reported for biomass and soil carbon sequestration by different categories of land use. The gross rate of carbon sequestration in urban tree and natural forest categories have been only reported for biomass; while the gross carbon sequestration rates in wetlands and agricultural lands have been mainly reported for soil/sediment carbon sequestration. The number of observations reported for biomass and soil carbon sequestration in plantation category is identical; whereas all reported observations of carbon sequestration rates for urban turfgrasses represent soil carbon sequestration.
The natural forest category has the highest number of observations reported for net carbon sequestration. Only one observation of net carbon sequestration has been reported for each category of urban tree and wetland. Data reported for agricultural lands and urban turfgrasses does not include net carbon sequestration. There is also only one observation reported for combined carbon sequestration in biomass and soil in the agricultural land category.

Given that the number of observations among land use categories varies for each type of carbon sequestration (i.e. biomass, soil, biomass + soil, net) and that there is only one observation for some categories as discussed above, the average carbon sequestration for categories are estimated only for biomass and soil carbon sequestration.

4.3.2. Variations in carbon sequestration values within categories

Average values of above- and below-ground biomass carbon sequestration in urban trees are from case studies conducted in the United States, China and Korea. The values range from 0.3 t C ha\(^{-1}\) yr\(^{-1}\) in intensively maintained trees in campus area of the Auburn University (AU) in Auburn city, Alabama USA, to 4.09 t C ha\(^{-1}\) yr\(^{-1}\) in Lincoln city, USA (Appendix 1, Table A1-3). The difference between minimum and maximum values can be attributed to differences in the density and composition of tree species. Trees in the maintained landscape of AU had a lower percent of canopy cover and exhibited smaller mean total height, crown width, and basal area compared to the trees in the protected area (arboretum) of AU. Therefore, the protected and naturalized trees in the arboretum (that were in better condition than those in the campus) sequestered over six times more carbon on a per hectare basis than trees in the main campus (Martin, et al., 2012). However, the total area studied in the research by Martin, et al., (2012) was small relative to other case studies conducted in large cities. As mentioned before, the amount of carbon sequestered in biomass of urban trees is mainly estimated using an allometric relationship between DBH and dry mass, so results can also vary depending on whether a species-specific or more generic allometric formula is used.

The rates of biomass (above- and below-ground) carbon sequestration vary between 3.7-9.50 t C ha\(^{-1}\) yr\(^{-1}\) in natural forest and between 1.27- 8.13 t C ha\(^{-1}\) yr\(^{-1}\) in plantation category (Table 4.3-2). Since both values are reported for carbon sequestration in biomass, the variation can be attributed to the differences in growth rate, forest structure and climate. The highest rates of carbon sequestration in natural forest and plantation categories were reported for broad-leaved poplar forest in China and moso bamboo plantation in Taiwan, respectively. This represents the high growth rate in broad-leaved trees (Jo, 2002; Zhao, et al., 2010; Yu, et al., 2011) and the fast growth rate and high biomass productivity in moso bamboo plantation (Yen & Lee, 2011). Soil carbon sequestration in plantation category varies from 0.13 t C ha\(^{-1}\) yr\(^{-1}\) to 3.2 t C ha\(^{-1}\) yr\(^{-1}\) (Table 4.3-2). This variation in soil carbon sequestration can be associated with the differences in soil organic carbon, climate, land use history and organic carbon input from above-ground biomass (Lemma, et al., 2006; Ueyama, et al., 2011; Sierra, et al., 2013).

In the wetland category, above- and below-ground biomass carbon sequestration varies between 0.45 t C ha\(^{-1}\) yr\(^{-1}\) and 4.71 t C ha\(^{-1}\) yr\(^{-1}\). The lowest rate is reported for a peatland in Finland; and the highest rate is reported for a mangrove forest in India. The rates of sediment carbon sequestration in this category varies between 0.3 t C ha\(^{-1}\) yr\(^{-1}\) for a coastal lagoon in Los Angeles, USA and 3.17 t C ha\(^{-1}\) yr\(^{-1}\) for a depressional freshwater wetland in Ohio, USA. Variations in sediment accretion rates and organic carbon in sediments can explain the differences in sediment carbon sequestration across wetlands (Callaway, et al., 2012; Bernal & Mitsch, 2012).

The rate of carbon sequestration in soil varies between 0.16-6.1 t C ha\(^{-1}\) yr\(^{-1}\) and 0.69-3.55 t C ha\(^{-1}\) yr\(^{-1}\) in the agricultural land and urban turfgrass/lawn categories, respectively. These variations can be associated with differences in soil carbon content, management practices (e.g. use of fertiliser, tillage system) and land use history (Hedley, et al., 2009; Toma, et al., 2013).
Table 4.3-2. Variations in carbon sequestration data

<table>
<thead>
<tr>
<th>Category</th>
<th>Carbon sequestration in biomass (t C ha(^{-1}) yr(^{-1}))</th>
<th>Carbon sequestration in soil (t C ha(^{-1}) yr(^{-1}))</th>
<th>Range*</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Min</td>
<td>Max</td>
<td>Range*</td>
</tr>
<tr>
<td>Urban tree</td>
<td>0.3</td>
<td>4.09</td>
<td>3.79</td>
</tr>
<tr>
<td>Natural forest</td>
<td>3.7</td>
<td>9.5</td>
<td>5.8</td>
</tr>
<tr>
<td>Plantation</td>
<td>1.27</td>
<td>8.13</td>
<td>6.86</td>
</tr>
<tr>
<td>Wetland</td>
<td>0.45</td>
<td>4.71</td>
<td>4.26</td>
</tr>
<tr>
<td>Agricultural land</td>
<td>0.16</td>
<td>6.1</td>
<td>5.94</td>
</tr>
<tr>
<td>Urban turfgrass/lawn</td>
<td>0.69</td>
<td>3.55</td>
<td>2.86</td>
</tr>
</tbody>
</table>

* Range = Max - Min

Table 4.3-3 is a summary of the literature addressing the factors affecting carbon sequestration (both gross and net) in biomass and soil of urban tree, natural forest, plantation, wetland, agricultural land and turfgrass/lawn. As shown in the table, a wide range of different factors influence carbon sequestration, which can be classified in the broad categories of (I) Bio-physical, (II) Management and maintenance, (III) Climate and (IV) Measurement method.

Table 4.3-3. Factors influencing soil and biomass carbon sequestration (gross and net) by different landscape types as reported in the literature

<table>
<thead>
<tr>
<th>Broad category</th>
<th>(I)</th>
<th>(II)</th>
<th>(III)</th>
<th>(IV)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bio-physical</td>
<td>Management &amp; maintenance</td>
<td>Climate</td>
<td>Measurement method</td>
</tr>
<tr>
<td>Factors/landscape type</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban tree</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
</tr>
<tr>
<td>Natural forest</td>
<td>2</td>
<td>2</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td>Plantation</td>
<td>6</td>
<td>7</td>
<td>8</td>
<td>9</td>
</tr>
<tr>
<td>Wetland</td>
<td>8</td>
<td>9</td>
<td>10</td>
<td>11</td>
</tr>
<tr>
<td>Agricultural land &amp; turfgrass</td>
<td>7</td>
<td>15</td>
<td>8</td>
<td>12</td>
</tr>
</tbody>
</table>

1 Nowak & Crane, 2002; Jo, 2002; Yang, et al., 2005; Zhao, et al., 2010; Yu, et al., 2011; Liu & Li, 2012
2 Zhao, et al., 2010; Ueyama, et al., 2011; Herren & Bravo, 2012
3 Lasco & Pollhin, 2003; Yu, et al., 2011
4 Zha, et al., 2004; Vetter, et al., 2005
5 Yu, et al, 2011
6 Kaul, et al., 2010
7 Silver, et al., 2000
8 Turunen, et al., 2002; Mitsch, et al., 2010
9 Turunen, et al., 2002; Bernal & Mitsch, 2008
10 Mitsch, et al., 2010; Turunen, et al., 2002; Brevik & Homburg, 2004
11 Chunra, et al., 2003; Belyea & Malmer, 2004; Bernal & Mitsch, 2008; Mitsch, et al., 2010
12 Minkkinen, et al., 2001
13 Turunen, et al., 2002; Chmura, et al., 2003; Bernal & Mitsch, 2008; Mitsch, et al., 2010
14 Turunen, et al., 2002
15 Follett, 2001; Lal, 2004b; Desjardins, et al., 2005; Salinger, et al., 2005; Zirkle, et al., 2011
4.3.3. Variations in carbon sequestration values between categories

Average values of carbon sequestration in above- and below-ground biomass and soil of the selected categories are provided in Table 4.3-4. As the data indicates, natural forests and plantation have the highest rates of biomass carbon sequestration, while soil carbon sequestration is highest in urban turfgrasses and wetlands, respectively. Based on the results of a one-way ANOVA, there is no statistically significant difference in soil carbon sequestration between the categories (F (3, 46) = 2, p-value>0.05); while biomass carbon sequestration significantly differs across categories (F (3, 36) = 13.71, p-value<0.001).

Table 4.3-4. Average rate of carbon sequestration in biomass and soil of the selected land use categories

<table>
<thead>
<tr>
<th>Category</th>
<th>No. of observation (Total)</th>
<th>Carbon sequestration in biomass</th>
<th>Carbon sequestration in soil</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Average (t C ha(^{-1}) yr(^{-1}))</td>
<td>No. of observation</td>
</tr>
<tr>
<td>Urban tree</td>
<td>15</td>
<td>1.95</td>
<td>14</td>
</tr>
<tr>
<td>Natural forest</td>
<td>22</td>
<td>6.29</td>
<td>8</td>
</tr>
<tr>
<td>Plantation</td>
<td>34</td>
<td>4.95</td>
<td>11</td>
</tr>
<tr>
<td>Wetland</td>
<td>27</td>
<td>1.88</td>
<td>7</td>
</tr>
<tr>
<td>Agricultural land</td>
<td>16</td>
<td>-</td>
<td>No data</td>
</tr>
<tr>
<td>Urban turfgrass/lawn</td>
<td>7</td>
<td>-</td>
<td>No data</td>
</tr>
</tbody>
</table>

*Total number of observations:121

Pair-wise comparisons (post hoc tests) of the estimated average carbon sequestration values for land use categories show that there are significant differences in average values of biomass carbon sequestration between urban trees, natural forests and plantation, and between wetlands, natural forests and plantation (p-values ≤ 0.001) (Table 4.3-5). There was no statistically significant difference in biomass carbon sequestration among urban trees and wetlands as well as between natural forests and plantation (p-value>0.05).

Table 4.3-5. Pairwise comparisons (post hoc tests) of average biomass carbon sequestration rates between categories

<table>
<thead>
<tr>
<th>Comparison</th>
<th>Difference in Means</th>
<th>SE</th>
<th>p-value</th>
<th>LSD</th>
<th>Bonferroni</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban tree - Natural forest</td>
<td>-4.34</td>
<td>0.81</td>
<td>&lt;.001</td>
<td>&lt;.001</td>
<td></td>
</tr>
<tr>
<td>Urban tree – Plantation</td>
<td>-3</td>
<td>0.73</td>
<td>&lt;.001</td>
<td>&lt;.001</td>
<td></td>
</tr>
<tr>
<td>Urban tree – Wetland</td>
<td>0.75</td>
<td>0.84</td>
<td>.930</td>
<td>1.000</td>
<td></td>
</tr>
<tr>
<td>Natural forest - Urban tree</td>
<td>4.34</td>
<td>0.81</td>
<td>&lt;.001</td>
<td>&lt;.001</td>
<td></td>
</tr>
<tr>
<td>Natural forest – Plantation</td>
<td>1.34</td>
<td>0.85</td>
<td>.122</td>
<td>.732</td>
<td></td>
</tr>
<tr>
<td>Natural forest – Wetland</td>
<td>4.42</td>
<td>0.94</td>
<td>&lt;.001</td>
<td>&lt;.001</td>
<td></td>
</tr>
<tr>
<td>Plantation - Urban tree</td>
<td>3</td>
<td>0.73</td>
<td>&lt;.001</td>
<td>&lt;.001</td>
<td></td>
</tr>
<tr>
<td>Plantation - Natural forest</td>
<td>-1.34</td>
<td>0.85</td>
<td>.122</td>
<td>.732</td>
<td></td>
</tr>
<tr>
<td>Plantation – Wetland</td>
<td>3.07</td>
<td>0.88</td>
<td>.001</td>
<td>.008</td>
<td></td>
</tr>
<tr>
<td>Wetland – Natural forest</td>
<td>-4.42</td>
<td>0.94</td>
<td>&lt;.001</td>
<td>&lt;.001</td>
<td></td>
</tr>
<tr>
<td>Wetland – Plantation</td>
<td>-3.07</td>
<td>0.88</td>
<td>.001</td>
<td>.008</td>
<td></td>
</tr>
<tr>
<td>Wetland – Urban tree</td>
<td>-0.75</td>
<td>0.84</td>
<td>.930</td>
<td>1.000</td>
<td></td>
</tr>
</tbody>
</table>

Significance level (α = 0.05)

Note: Levene’s test of equality of variance:
(a) biomass carbon sequestration data: statistic=1.77, df1=3, df2=36, p=0.169
(b) Soil/sediment carbon sequestration data: statistic=0.32, df1=3, df2=46, p=0.811

Results also show lower variability (i.e. lower range) in biomass sequestration rates for urban trees, and wetlands compared to plantation and natural forests. The difference between the lowest and the highest rates of soil carbon sequestration is highest in the agricultural land category; while urban turfgrass and wetlands have the lowest range of data. The lower variability in urban tree and urban turfgrass categories might be an artefact of there being relatively fewer case studies. Observations in these categories also come from relatively few countries, which might have an influence on variability.
4.4. Conclusion

This chapter examines the variations in carbon sequestration rates across the broad categories of land use to understand how changes in land use may affect carbon sequestration. In doing so, data on carbon sequestration by the broad categories of land use (including urban trees, urban turfgrasses, natural forests, plantation, wetlands and agricultural lands) from 121 observations (Appendix 1, Table A1-3) were compiled and used to estimate the average value of carbon sequestration in biomass and soil/sediment of the selected categories. The literature review identified 40 observations in urban tree, natural forest, plantation and wetland categories reported for carbon sequestration rates in biomass (both above- and below-ground), while data on carbon sequestration in soil/sediment included 50 observations in plantation, wetlands, agricultural land and urban turfgrass categories (Table 4.3-1). In total, 31 observations were reported for net carbon sequestration and combined sequestration (i.e. soil and biomass). Given that there was only one observation reported for net carbon sequestration in urban tree and wetland categories and also only one observation reported for combined sequestration in the agricultural land category, the average values of net and combined (soil plus biomass) carbon sequestration cannot be compared between categories and thus are not estimated.

Pair-wise comparisons of average carbon sequestration rates between categories indicate statistically significant differences in biomass carbon sequestration between all categories except between urban trees and wetlands as well as between natural forests and plantation. However, there is no significant difference in soil/sediment carbon sequestration across categories. The review of the literature also found that carbon sequestration in the selected land use categories is influenced by a wide range of parameters that cause variations in carbon sequestration rates in the biomass and soil/sediment across different observations. Findings show that the rate of carbon sequestration in biomass particularly in trees (urban trees, natural forests and plantation) is mainly associated with growth rate and biomass production which varies between different species and decreases as tree age increases (Nowak & Crane, 2002). The rate of carbon sequestration in soil/sediment is greatly influenced by the rate of the organic carbon input to the soil and therefore the amount of soil organic carbon stored within a given period is the main indicator of the rate of carbon accumulation. In wetlands, the rate of sediment accretion (or peat accumulation) is also a main parameter used to estimate carbon sequestration.

While the data used in this research represents the best available, these estimates are still limited by the quantity and variability of available data. Moreover, as mentioned before, carbon sequestration in both soil and biomass is greatly influenced by parameters which show spatial variations. The average rates of carbon sequestration estimated in this study provide an indication of the gross carbon sequestration and may allow decision-makers to capture the trend towards or away from higher levels of gross carbon sequestration linked to land use change. Applying such approximate estimates to quantify the scale and magnitude of change at local scale involves high level of uncertainty that undermine the ability to make reliable estimates of the scale of the impact.

In the absence of empirical data, local estimates can benefit from observations reported for a specific land use with similar conditions in terms of the parameters affecting carbon sequestration. But, the lack of local data may constrain accurate measurement of the changes in carbon sequestration for each category and for the entire landscape.

Given the limited data on net carbon sequestration, it is difficult to conclude whether shifts between land use types would have a definite influence on net carbon sequestration. In addition, the definitions and methods used to estimate net carbon sequestration are not consistent among studies. These inconsistencies impose a limitation on using the net carbon sequestration data even where the data is reported for one type of land use.
4.5. Narrowing the research focus

According to the above estimates (Table 4.3-4), the highest rates of biomass carbon sequestration are in natural forests and plantation, while urban turfgrasses and wetlands have the greatest rates of carbon sequestration in soil/sediment. As the literature (Townsend-Small & Czimczik, 2010; Selhorst & Lal, 2013) concluded, the high rate of carbon accumulation in turfgrass soil is due to the frequent irrigation and fertilisation which increase plants productivity and SOC content. These can also result in $\text{N}_2\text{O}$ emissions from turfgrasses which can offset carbon sequestration by these ecosystems. Maintenance of turfgrasses (e.g. mowing, irrigation and fertilisation) also generates $\text{CO}_2$ emission due to the fossil fuels consumption. It is also the case in planted forests and urban trees where frequent pruning and harvesting can release significant amounts of $\text{CO}_2$ to the atmosphere (Jo, 2002; Nowak, et al., 2002; Zhao, et al., 2010).

Taking maintenance activities into account, wetlands have a great advantage over managed land use types in terms of long-term net carbon sequestration and storage particularly in their sediment. However, some types of wetlands, including freshwater wetlands and peatlands, also generate significant sources of $\text{CH}_4$ (methane) which may offset $\text{CO}_2$ uptake by these ecosystems in a long run (Frolking, et al., 2006). In contrast, coastal marine wetlands including mangrove forests, saltmarshes and seagrass beds have very low rate of $\text{CH}_4$ emission mainly because methanogenic bacteria are inhibited by salinity (IPCC, 2000a; Poffenbarger, et al., 2011).

The global literature also reported a higher rate of carbon sequestration in sediments of coastal wetlands compared to terrestrial ecosystems (IPCC, et al., 2003; Murray, et al., 2011; Mcleod, et al., 2011). It is argued that carbon locked up within the sediments of coastal ecosystems can stay for a very long period (e.g. millennia) if it remains undisturbed (Howard, et al., 2017; Pendleton, et al., 2012). There is also growing interest in blue carbon (i.e. the carbon sequestered and stored by coastal and marine ecosystems particularly by mangroves, saltmarshes and seagrasses) within the global literature over the last few years (UNEP & CIFOR, 2014).

In addition to carbon sequestration, coastal wetlands have also another important contribution to climate change. They provide natural buffers and protect coastal properties against storm surges, sea level rise and erosion as long as they have the ability to can keep pace with the changing environment (Koch, et al., 2009; Gedan, et al., 2011; Duarte, et al., 2013; Spalding, et al., 2014a). This decreases vulnerability of coastal communities to the effects of climate change and enhances adaptation and resilience to climate change. As stated in Chapter 1, this is particularly the case given Auckland’s unusually long coastline relative to the urbanized area. Therefore, the focus of this research was narrowed down to analysing the climate change services of Auckland’s coastal wetlands. This also includes considering the policy options to improve land use decision-making with respect to the climate change values of coastal wetlands. Findings of this research regarding these issues are outlined within the following chapters.
5. Climate change services of coastal wetlands

5.1. Introduction

The implications of land use change on the carbon sequestration capacity of different land use categories were discussed in the previous chapter. The discussions highlighted the significance of coastal wetland ecosystems (i.e. mangrove forests, saltmarshes and seagrass beds) in providing climate change services through sequestering and storing carbon and protecting coastline against coastal hazards including the impacts of climate change. The analyses and discussions in Chapter 4 resulted in the selection of coastal wetlands as the focus of this research.

This chapter aims to answer the following question:

- How do coastal wetlands in the Auckland region currently and potentially contribute to climate change mitigation and adaptation?

In addressing this question, this chapter first summarises the global evidence that supports the significance of coastal wetlands and their climate change services. The review of global literature also aims to identify the factors that influence the capacity of coastal wetlands for carbon sequestration and storage (CS&S) and coastal protection. This is followed by a discussion on the key features of coastal wetlands in the Auckland region, and their potential contribution to climate change mitigation and adaptation.

5.2. Global review

Coastal wetlands are habitats in estuarine hydrosystems which include ‘all areas of subtidal and intertidal zones in estuaries, and also wet ground in supratidal zones where surface water and groundwater receive saline contributions from wave splash, or airborne salt in sea spray’ (Johnson & Gerbeaux, 2004, p. 20). Coastal wetlands are also identified as ‘tidally-influenced wetlands specifically including mangroves, saltmarshes, seagrasses and tidal freshwater systems’ (Herr, et al., 2012, p. 10). This definition includes both saline and freshwater coastal wetlands. As mentioned in Chapter 2 (Section 3.3.1.2), the literature addressing climate change benefits of coastal ecosystems have mainly focused on saline coastal wetlands including mangrove and saltmarsh habitats (Rogers, et al., 2014; Chmura, et al., 2003). However, seagrass beds have been also grouped with mangroves and saltmarshes as vegetated coastal habitats that can contribute to climate change mitigation and adaptation (Duarte, et al., 2013; Mcleod, et al., 2011).

Coastal wetlands are valuable ecosystems that provide a wide range of significant functions (De Groot, et al., 2012). They support marine and terrestrial biodiversity, provide wildlife habitats, purify and replenish water and provide recreational and economic opportunities (Barbier, et al., 2011). Among the diverse range of benefits provided by coastal wetlands, their role in carbon sequestration and coastal protection have been often undervalued until recently when their potential for storing carbon (termed as ‘blue carbon’20) and protection against storm events received increased global attention (Duarte, et al., 2013; Lovelock & McAllister, 2013). Despite the significance of coastal wetlands for climate change mitigation and adaptation, these ecosystems are also vulnerable to the effects of climate change such as sea level rise (Spalding, et al., 2014a). Understanding their vulnerability is therefore essential in order to protect and enhance their climate change values in a long run (Spalding, et al., 2014a). The following sections discuss the potential of coastal wetlands for providing climate change services based on a review of the global literature.

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20 As defined by the Blue Carbon Initiative (http://thebluecarboninitiative.org/ [Accessed 2016]), blue carbon is the carbon stored in coastal and marine ecosystems including mangroves, tidal marshes and seagrasses.
5.2.1. Contribution of coastal wetlands to climate change mitigation

As mentioned in the previous chapter, coastal wetlands including mangrove forests, saltmarshes and seagrass beds capture carbon dioxide (CO$_2$) and store carbon both in the plants (biomass) and in the sediment underneath them (Howard, et al., 2017; Mcleod, et al., 2011). Over the last few years, there has been a growing interest within the scientific communities and international organisations/initiatives in the role of coastal and marine ecosystems as sinks of carbon or blue carbon ecosystems (UNEP & CIFOR, 2014).

The average amount of carbon stored per hectare in the organic-rich soils of coastal wetlands is between two and eight times greater than that in terrestrial forests (Murray, et al., 2011) (Figure 5.2-1). Default values of carbon stock by tropical and temperate forests are reported to be less than 400 t C ha$^{-1}$ (IPCC, 2003), whereas mangrove ecosystems, on average, can approximately store 1,000 t C ha$^{-1}$ (Donato, et al., 2011; Adame, et al., 2013). As reported by Mcleod, et al., (2011), the annual carbon burial (sequestration) rate of mangroves, saltmarshes and seagrass beds exceed that of terrestrial ecosystems (Figure 5.2-2).

Approximately 95% of the carbon in seagrass ecosystems is stored in their sediments; while mangrove and saltmarsh ecosystems can store carbon in their biomass as well as in their sediments (Murray, et al., 2011). As also mentioned in Chapter 4 (Section 4.5), the carbon within the sediments of coastal vegetation is ‘locked up’ due to the low-oxygen conditions and a number of other factors that hinder decomposition at depth (Pendleton, et al., 2012). This characteristic of coastal wetlands particularly mangroves and saltmarshes gives them an advantage over terrestrial ecosystems in terms of their ability for a long-term carbon burial (Lawrence, et al., 2012). However, when coastal wetlands are degraded or converted to other land uses, the sediment carbon is exposed to oxygen and is metabolised into CO$_2$ through bacterial (microbial) activities (Lawrence, et al., 2012), releasing the stored carbon directly to the atmosphere.

This process is less understood for seagrass ecosystems, but there is an argument that the sediment carbon can be released into the water column through a process of mineralisation or oxidation when exposed to aerobic water (Lawrence, et al., 2012). A study by Lovelock, et al., (2011) suggested that conversion or destruction of coastal wetlands can change these ecosystems from being net sinks into net sources of carbon over a long term, on the basis that disturbance stops carbon sequestration and results in releasing the stored carbon into the atmosphere or the water column.

![Figure 5.2-1. Average soil and biomass carbon storage in coastal wetlands and tropical forests (Murray, et al., 2011)](image)

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Quantitative estimates of emissions associated with disturbance or conversion of coastal habitats are scarce within the literature (UNEP & CIFOR, 2014). Using the best available data on the aerial extent of coastal wetlands, rates of land use conversion and values of near-surface carbon stocks in coastal wetlands, Pendleton, et al., (2012) provided the first global estimates of the emissions due to degradation of coastal wetland habitats (UNEP & CIFOR, 2014). They argued that the magnitude of post-conversion CO₂ release depends on the type of ecosystem being disturbed and the level (intensity) of disturbance. The level of disturbance indicates the amount of carbon loss due to biomass or sediment removal or alteration (Pendleton, et al., 2012). The approach used by Pendleton, et al., (2012) is also used in this research to estimate the approximate emissions due to removal or destruction of coastal wetlands in the Auckland region and is further explained in section 5.6.

Despite the significant role of coastal wetlands in the carbon cycle, most of the studies on carbon sequestration have focused on terrestrial ecosystems (such as forests) and coastal wetlands have been less studied (UNEP & CIFOR, 2014). The REDD²² mechanism, under the UNFCCC and its Kyoto Protocol, has for long been focusing on emissions and removals of greenhouse gases (GHGs) from terrestrial (land) areas; and promoting protection of terrestrial forests and more recently peatlands for carbon management (UNEP & CIFOR, 2014). However, the recent trends show a growing consensus among the scientific community around the idea that management of coastal ecosystems through avoided emission, conservation, restoration and sustainable use can play a key role in management of the global carbon cycle and thus can strongly contribute to climate change mitigation (Howard, et al., 2017; Mcleod, et al., 2011). As part of the blue carbon initiatives, various projects for restoration of degraded coastal wetlands are proceeding in some countries including Senegal, India, Abu Dhabi, Ecuador, Indonesia, Madagascar and Mozambique (UNEP & CIFOR, 2014).

The 2006 IPCC guidelines (IPCC, 2006) had generally excluded coastal wetlands, mainly due to the lack of sufficient scientific information to support Tier 1 (basic) methodologies for how to include coastal wetlands in national inventories (Ausseil, et al., 2015). However, the 2013 IPCC Wetlands Supplement to the 2006 IPCC guidelines (IPCC, 2014c), broadens the scope of wetland reporting to cover all wetlands in any land use class (Ausseil, et al., 2015) and covers inland organic soils and wetlands on mineral soils, coastal wetlands including mangrove forests, tidal marshes and seagrass meadows and constructed wetlands for wastewater treatment. Although still non-mandatory, it can be considered as the first international attempt to address the emission implications due to degradation of coastal wetlands.

22 Reducing Emissions from Deforestation and Forest Degradation
The Social Cost of Carbon (SCC), rather than the market value of carbon, is commonly used to estimate the monetary value of the carbon stored in coastal wetlands (Barnett, et al., 2016) or the emissions resulting from degradation of these ecosystems (Pendleton, et al., 2012). The SCC is the estimated price of the economic or social costs or damages caused by each additional tonne of CO$_2$ emission; and has been commonly used to assess the benefits of climate change mitigation policies (Nordhaus, 2014). The market price (value) of carbon is determined “by the avoided cost of regulatory controls on carbon and not avoided damages” (Pendleton, et al., 2012, p. e43542) and thus does not necessarily equal to the SCC. Determining SCC is a complex and challenging process that involves the use of sophisticated Integrated Assessment Models (IAMs) and depends on the assumptions around the time period used for the analysis (Pizer, et al., 2014). The average SCC values of $37 and $83 per tonne of CO$_2$ are reported for the US and the United Kingdom, respectively (Foote, et al., 2015; Barnett, et al., 2016).

In 2015, the first global methodology (Methodology for Tidal Wetland and Seagrass Restoration, v1.0) (Verified Carbon Standard, 2015a) to quantify the blue carbon credits from tidal wetlands and seagrass restoration activities was released by the Verified Carbon Standard (VCS). It facilitates and encourages development of blue carbon projects that aim to provide carbon credits through restoration and improved management of coastal wetlands (UNEP & CIFOR, 2014). A methodology for quantifying the GHG benefits of tidal wetland restoration in the Mississippi Delta (Restoration of Degraded Deltaic Wetlands of the Mississippi Delta, v.2.0) had been also approved by the American Carbon Registry in 2012 (American Carbon Registry, 2012).

### 5.2.2. Factors affecting carbon sequestration and storage in coastal wetlands

#### 5.2.2.1. Mangrove ecosystems

##### 5.2.2.1.1. Carbon storage

Studies that investigated variations in carbon density across mangrove ecosystems at the global scale are very scarce. By analysing soil carbon storage data for mangrove sites worldwide, Chmura, et al., (2003) concluded that carbon density in mangrove ecosystems is significantly influenced by mean annual temperature ($p < 0.005$) mainly because temperature affects decomposition rate. It is also argued that temperature can affect productivity and growth rate in coastal vegetation and therefore can influence carbon balance and storage in these ecosystems (Ellison, 2000). According to Mcleod, et al., (2011), temperature affects carbon storage in coastal vegetation by influencing the underlying metabolic processes of carbon gain through photosynthesis and carbon loss through plant and microbial respiration.

Chmura, et al., (2003) also concluded that differences in local factors such as distance from open waters are the main causes of great variations in carbon density among mangrove forests within a given region. As an example, they found that mangroves adjacent to coastal waters at Shark River, Florida, had lower amount of soil carbon density compared to mangroves located farther from open waters. They argued that this could be due to differences in tidal flooding, sediment deposition rate and suspended sediment supply which are drivers for variations in carbon density in tidal saline wetlands (Chmura, et al., 2003). Similarly, a study of the mangrove ecosystems at Pollen Island, Auckland (Yang, et al., 2013), showed that interior mangroves (those located farther from the tidal creek) had greater amount of sediment carbon than soils in mudflat and fringe mangroves adjacent to coastal waters. This study argues that the difference in sediment carbon content can be attributed to the greater contribution of below-ground biomass to the soil carbon content and to the lower rate of soil mineralisation in interior mangroves compared to mudflat and fringe mangrove.

According to Yang, et al., (2013), under the stressful environmental conditions at interior site (Low PH level and seasonally fluctuating total dissolved salt (TDS) concentration), mangroves allocated more carbon into below-ground biomass through an adaptation process than mangroves at fringe and mudflat sites. Therefore, interior mangroves were shorter in height and had higher stem density compared to fringe mangrove.
mangroves. In addition, minimum tidal flow and turbulence at the interior site resulted in greater amount of fine sediments deposition and consequently higher carbon stock in interior sediments.

Saintilin, et al., (2013) found no apparent impact of latitude on carbon storage among mangrove ecosystems across South Eastern Australia. However, they reported the lowest density of soil carbon for mangroves within the colder climate at the southernmost sites. Their results also indicated significant impact of vegetation type (p < 0.0001) on soil carbon storage, while the impact of geomorphic settings (fluvial, marine) on variations in soil carbon density was not significant (p = 0.0699). It is also argued that due to differences in root morphology and below-ground biomass, soil organic carbon varies among different mangrove species (Donato, et al., 2011; Lovelock, et al., 2014).

Overall, results of the above studies suggest that soil carbon storage in mangrove ecosystems is mainly affected by temperature, mangrove species and distance from open waters (i.e. seaward distance).

5.2.2.1.2. Carbon sequestration

The only available comprehensive study of variations in carbon sequestration across mangrove ecosystems worldwide is done by Chmura, et al., (2003) who found no significant influence of temperature on carbon sequestration across mangrove sites. However, they concluded that temperature can affect carbon sequestration in coastal wetland soils due to its effect on decomposition rate. The impact of latitude on carbon sequestration in mangrove sediments has not been investigated neither in global nor in regional studies. Carbon sequestration rate in coastal vegetation is positively correlated with sediment accretion rates (Chmura, et al., 2003; Howe, et al., 2008). According to the results of studies at a local or regional scale, changes in tidal range (Saintilan, et al., 2013), mangrove species (Lovelock, et al., 2014), seaward distance (Chmura, et al., 2003) and rainfall (Howe, et al., 2008) can affect sediment accretion rate and thereby affect sequestration rates in mangrove habitats.

It is also argued that the balance between rainfall and evaporation affects sediment salinity in coastal ecosystems (Ouyang & Lee, 2014) and salinity is negatively correlated with carbon sequestration due to its impact on decomposition rates (Loomis & Craft, 2010). Lovelock, et al., (2014) have concluded that plant species composition can strongly affect carbon sequestration in mangrove ecosystems, attributed to differences in organic sediment input (Lovelock, et al., 2014).

5.2.2.2. Saltmarsh ecosystems

5.2.2.2.1. Carbon storage

As reported by Chmura, et al., (2003), mean annual temperature affected carbon density in both communities of Spartina patens and Spartina alterniflora marshes globally. They found that soil carbon density in both vegetation types decreased with increasing temperature and concluded that it could be due to greater decomposition rate in warmer climates. However, the impact of temperature on soil carbon density was only significant (p <0.005) for Spartina patens marshes. Mcleod, et al., (2011) also concluded that temperature effects soil carbon dynamic in coastal wetlands by influencing the rate of photosynthesis and respiration. Some other studies (Ouyang & Lee, 2014; Loomis & Craft, 2010) suggest that the balance between rainfall and evaporation can influence soil carbon density in coastal marshes. This is because sediment salinity is a function of rainfall and evaporation and negatively correlated with soil bulk density and soil organic carbon.

Saintilin, et al., (2013) found no apparent impact of latitude on carbon storage among saltmarsh ecosystems across South Eastern Australia. However, the lowest amount of carbon density was found for saltmarsh vegetation across the southernmost sites (Westernport Bay, Southern Victoria) that had colder climate compared to other sampling sites (Hawkesbury River and Hunter River, New South Wales). Their

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23 The vertical difference between the high tide and the low tide is identified as tidal range.
results also showed that carbon storage in saltmarsh habitats was significantly influenced by saltmarsh species. They concluded that differences in standing biomass and root structure among different saltmarsh species were associated with variations in soil carbon density among different saltmarsh species.

Similarly, a study by Lovelock, et al., (2014) showed that saltmarsh species with highly developed rhizomes have greater amount of carbon density compared to those with shallow roots. For example, saltmarsh dominated with *Juncus kraussii* had greater amount of soil organic carbon than saltmarsh dominated with *Sarcocornia quinqueflora* (Saintilan, et al., 2013; Lovelock, et al., 2014). The difference is likely due to the perennial rhizomes in *Juncus* marsh that greatly contribute to soil carbon compared to shallow roots of *Sarcocornia* marsh (Saintilan, et al., 2013; Lovelock, et al., 2014). It is also reported that differences in position (elevation) of marsh species in the tidal zone account for differences in soil carbon density in saltmarsh habitats (Chmura, et al., 2003; Saintilan, et al., 2013; Lovelock, et al., 2014).

### 5.2.2.2. Carbon sequestration

Similar to mangrove ecosystems, carbon sequestration in sediments of saltmarsh ecosystems is argued to be affected by temperature, due to its impact on primary productivity (Kirwan, et al., 2009), soil carbon dynamics (Ouyang & Lee, 2014) and decomposition rate (Chmura, et al., 2003). Chmura et al. (2003) found no significant impact of temperature on carbon sequestration within global saltmarsh studies. They concluded that temperature may influence carbon sequestration in saltmarsh ecosystems by affecting decomposition rate, but local or regional factors that influence sediment accretion rates are the dominant drivers for carbon sequestration in coastal wetlands.

Using a generalized linear model in a study on saltmarsh vegetation, Saintilan, et al., (2013) found that vegetation types was the major driver for vertical sediment accumulation (p < 0.0001), followed by tidal range and marsh zone (p < 0.0005). They also showed that saltmarsh vegetation of upper intertidal zone had lower rates of soil carbon sequestration compared to the lower tidal zone vegetation (Chmura, et al., 2003; Saintilan, et al., 2013). This difference is associated with the lower frequency, depth and duration of inundation in upper intertidal zones than lower areas which affect vertical soil accretion (Chmura, et al., 2003; Saintilan, et al., 2013).

Howe, et al., (2008) concluded that sediment accretion rates among saltmarsh vegetation at Hunter Estuary, South Eastern Australia, were greatly influenced by rainfall and mean estuary water level. Sediment accretion rate maintains the position of coastal vegetation in the tidal frame (Howe, et al., 2008) and positively correlates with carbon sequestration (Howe, et al., 2008; Ouyang & Lee, 2014). Sediment salinity that can affect carbon sequestration rates, due to its impact on decomposition rate, is also a function of rainfall and evaporation (Loomis & Craft, 2010).

As an update for the study by Chmura, et al., (2003), Ouyang and Lee, (2014) compiled a global database for saltmarsh ecosystems and analysed how latitudinal and biogeography parameters influence carbon sequestration in saltmarsh sediments. They concluded that sediment carbon sequestration in saltmarsh habitats is influenced by a diverse range of biogeochemical and biotic factors including tidal range, latitude, halophyte genera and habitat elevation. They used multiple linear regressions and found that variation in carbon sequestration among the global data was mainly influenced by tidal range (51.7%) and latitude (29.6%). Results of a regression tree analysis applied in their study indicated that latitude sits on top of the hierarchy and mean tidal range (MTR) occupies the primary branches of the tree. They found that saltmarsh ecosystems across mid-latitudes (between 48.5 and 58.5° N) had the highest average rate of soil carbon sequestration, while sequestration rates decreased towards the poles and the equator.

Ouyang and Lee, (2014) discussed that the differences in the length of growing season and sediment salinity, due to changes in latitude, could affect primary productivity, decay rate and soil properties and consequently influence sequestration rate in saltmarsh sediment. In addition, variations in temperature due to latitudinal changes could affect carbon dynamics through changes in biomass productivity and decomposition rate. They argued that difference in tidal range could lead to variations in carbon
sequestration as tidal range indicates frequency and duration of tidal flooding and thus affects belowground biomass productivity, sediment aeration and organic matter decomposition. Their study showed that carbon sequestration in saltmarsh sediments varied with respect to elevation of marsh vegetation across tidal zone. They discussed that the difference could be associated with variations in sediment accretion rate, root productivity and sediment input that are influenced by frequency and duration of inundation.

As reported by Ouyang and Lee, (2014), quality and quantity of soil organic matter that affect decomposition rate is different within different species of saltmarsh vegetation. Therefore, they concluded that plant species can affect soil carbon sequestration in saltmarsh ecosystems. Finding of their study indicated the highest rate of soil carbon sequestration within Spartina-dominated saltmarshes, whereas Distichlis-dominated marshes showed the lowest rate of soil carbon sequestration.


5.2.3. Conclusion

Global studies have mainly accounted for the factors that affect carbon sequestration and storage (CS&S) in sediments of coastal wetlands. Findings of the global review suggest that soil carbon storage in coastal wetlands is a function of soil bulk density and organic carbon content of the soil and soil carbon sequestration is driven by sediment accretion rate, soil bulk density and soil carbon density. Therefore, factors affecting these parameters can also affect the rate of CS&S in the soil. These factors include temperature, rainfall, evaporation, plant species, tidal range and position of vegetation in tidal frame. There is a general agreement between most climate models that the capacity of both terrestrial and marine ecosystems to capture and store atmospheric carbon will be influenced by climate change in the future. However, the magnitude of these impacts is highly uncertain (Friedlingstein, et al., 2006). Table 5.2-1 provides summaries of the factors influencing CS&S among mangrove and saltmarsh ecosystems as reported in global studies.

Table 5.2-1. Factors affecting variations in soil carbon sequestration and storage in coastal wetlands (Chmura, et al., 2003; Howe, et al., 2008; Loomis & Craft, 2010; Mcleod, et al., 2011; Saintilan & Rogers, 2013; Saintilan, et al., 2013; Lovelock, et al., 2014; Ouyang & Lee, 2014)

<table>
<thead>
<tr>
<th>Factor</th>
<th>Mechanism of affecting carbon sequestration and storage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>Temperature affects productivity and decomposition rate and consequently affects sediment CS&amp;S.</td>
</tr>
<tr>
<td>Rainfall &amp; evaporation</td>
<td>The balance between rainfall and evaporation identifies sediment salinity and salinity affects soil bulk density and decomposition rate.</td>
</tr>
<tr>
<td>Plant species</td>
<td>Differences in biomass productivity and root structure among different species of mangrove and saltmarsh vegetation affect soil carbon content and sediment accretion rate and therefore affect soil carbon stock and sequestration.</td>
</tr>
<tr>
<td>Tidal range</td>
<td>Tidal range determines frequency and duration of tidal inundation (flooding) and therefore influences CS&amp;S through affecting belowground biomass productivity, decomposition rate, sediment aeration and deposition.</td>
</tr>
<tr>
<td>Elevation from sea level**</td>
<td>Position of mangrove and saltmarsh vegetation across tidal range indicates frequency and duration of tidal inundation and influences CS&amp;S through affecting root productivity, suspended sediment supply and sediment accretion rate.</td>
</tr>
<tr>
<td>Seaward distance***</td>
<td></td>
</tr>
</tbody>
</table>

**Mainly associated with saltmarsh vegetation   ***Mainly associated with mangrove habitats
5.3. Adaptation services of coastal wetlands

5.3.1. Protection from coastal hazards and erosion

The valuable function of coastal wetlands in protecting shorelines has been highlighted in recent decades particularly after the most damaging events such as the 2004 Indian Ocean tsunami; hurricanes Katrina and Rita (2005) and Hurricane Sandy (2012) (Barbier, et al., 2013). The mechanisms by which coastal wetlands provide protection against storms include reducing the area of open water needed for wind to form waves (fetch), increasing drag on water flow and absorbing wave energy (Koch, et al., 2009; Spalding, et al., 2013; Marsooli & Wu, 2014). In fact, multiple interactions of wave motion with trunks and roots of coastal plants and the consequent bottom friction are the most important mechanisms that contribute to the reduction in waves’ height and energy (Koch, et al., 2009; Gedan, et al., 2011; Mitra, 2013).

Studies showed that damage from Hurricane Katrina in coastal Louisiana was lower in earthen levees fronted by extensive canopies of forested wetlands than turf covered earthen levees along the man-made shipping channel (Mississippi River Gulf Outlet, MRGO) (Day, et al., 2007; Barbier & Enchelmeyer, 2014). Similarly, a number of studies that conducted a post-tsunami (the 2004 Indian Ocean Tsunami) survey suggested that mangrove forests provided a significant defence against tsunami waves in the affected areas of India and Sri Lanka (Danielsen, et al., 2005; Dahdouh-Guebas, et al., 2005; Kathiresan & Rajendran, 2005; Vermaat & Thampanya, 2006). Consequently, there has been increasing global attention to the concept of ‘bioshield’ (i.e. use of vegetation for improving protection against natural disasters) since 2004 (Feagin, et al., 2010). A number of other studies including those by Hamzah et al. (1999), Hiraishi and Harada (2003), Latief and Hadi (2007) and Tanaka et al. (2007) have also provided evidence in support of the effectiveness of mangrove forests for protection against tsunami waves.

However, due to the sudden and destructive nature of tsunami events, the evidence that coastal wetlands can protect against tsunami waves is more tenuous (Gedan, et al., 2011). A few studies have argued that the reduced damage from the 2004 tsunami in some places could not be linked with the presence of coastal wetlands. For example, further statistical analysis of the post-tsunami data by Kerr, et al., (2006) found no relationship between presence and absence of mangroves with tsunami-caused human death. Based on these results, Kerr, et al., (2006) argued against discourses that overstate efficiency of mangroves for tsunami protection.

By analysing data from 57 sites throughout the Indian Ocean, Chatenoux and Peduzzi, (2007) also argued that mangrove forests affected by the 2004 tsunami were mostly located in sheltered areas such as bays, lagoons and estuaries. It is, therefore, not clear whether the minor damage to properties in these areas was due to the protective capacity of the mangrove forests or because the survey areas were sheltered from direct exposure to open sea. They concluded that the extent of tsunami-induced inundation was associated with the distance from the earthquake epicenter and nearshore bathymetry and coastal vegetation was unlikely to reduce inundation from tsunamis. Gedan, et al., (2011) have also concluded that large tsunami waves and storm surges can overwhelm the attenuation effect of coastal wetlands vegetation.

Spalding, et al., (2014b) studied coastal protection capacity of natural ecosystems and argued that both natural and engineering coastal defences may be less effective in protecting coastlines against very high waves (10 – 15m height) in extreme tsunami events. However they posited that the growing consensus among scholars clearly supports the protective role of mangrove ecosystems in attenuating wave energy.

Anecdotal information collected after Hurricane Andrew in 1992 in Louisiana suggested that marsh wetlands reduced the wave height by 4.7 cm/km (Costanza, et al., 2008). Zhang, et al., (2012) indicated that without mangrove forest, inundation from Wilma would extend more than 70% further inland in the Gulf Coast of South Florida. Analysing peak water level at two mangrove sites in south western Florida
indicated that the storm wave heights from hurricanes Charley (2004) and Wilma (2005) decreased by an average of 4.2 cm/km and 9.4 cm/km when passed through riverine mangroves and mangrove-interior marsh, respectively (Krauss, et al., 2009). Figure 5.3-1 illustrates storm surge reduction by the mangrove-interior marsh site during Hurricane Charley, as reported by Krauss, et al., (2009).

In Queensland, Australia, damage from Cyclone Larry (2006) was significantly less in areas with intact mangrove cover (Bell & Lovelock, 2013). According to Mitra, (2013), the damage resulting from the 1970 typhoon and associated tidal waves in Bangladesh would not have been so extensive, if vast areas of mangrove swamps would not have been lost and replaced with paddy fields. Likewise, villages in the Kutch areas of Gujarat, India, where mangrove forests suffered from continuous encroachment, were heavily damaged due to the destructive cyclone in 1983 (Mitra, 2013). It is also reported that during the 1999 super cyclone in India, properties in a village that was protected by artificial embankment experienced greater damage (US$ 154 per household) compared to properties in a village that was protected by mangrove forest (US$ 33 per household) (Badola & Hussain, 2005).

The protective role of coastal mangroves against tropical storms (‘Wilma’ and ‘Gamma’, 2005) in Turneffe Atoll, Belize, has been highlighted in a study by Granek and Ruttenberg, (2007). They found that the retention rate of their installed research equipment was greater in intact mangrove areas than adjacent cleared mangrove areas, after their study site was hit by the storms. They concluded that removal of mangroves decreases coastal protection not only against hurricanes, but also during less energetic but more frequent events, such as tropical storms. Additional evidence that implies coastal wetlands protect shorelines comes from a number of studies that argue damage from major storm events has been lower in areas with coastal wetlands including mangroves (Badola & Hussain, 2005; Kerr & Baird, 2007; Costanza, et al., 2008; Das & Vincent, 2009).
Mangroves have significantly reduced the economic impacts and fatalities associated with the 1999 super cyclone that struck Orissa, India (Badola & Hussain, 2005; Das & Vincent, 2009). It was estimated that without mangroves cover, there would have been 1.72 additional deaths per village within 10 km of the coast (Das & Vincent, 2009). Observations of the areas affected by this cyclone revealed that no damage occurred in areas with dense and luxuriant mangrove forest. By contrast, maximum damage was reported for Mahanadi delta in India, where large-scale deforestation and reclamation of mangroves have been occurred (Mitra, 2013).

The results of storm modelling by Wamsley, et al., (2010) showed that the wetlands of coastal Louisiana can attenuate wave heights up to 16.6 cm per kilometre of wetland. However, this potential depends on the characteristics of storm waves and coastal wetlands. A meta-analysis of wave attenuation data from 15 sites in USA, England, the Netherlands and China indicated that wave height reduction was greater in vegetated sites than non-vegetated mudflats (Gedan, et al., 2011).

Unlike mangrove and saltmarsh habitats, the role of seagrass beds in reducing wave height and energy is less studied (Bouma, et al., 2014). It is argued that due to the high flexibility of seagrass beds, which can easily bend under current, their wave attenuation function is smaller than mangrove or saltmarsh ecosystems unless they are very dense (high biomass) (Bouma, et al., 2014). It is also suggested that seagrass beds are less effective in areas with macro-tidal range (i.e. more than 4 metres) and strong tidal currents (Bouma, et al., 2014).
A number of studies argued that the storm protection function of coastal wetlands is reduced as the magnitude of storm surge increases (Day, et al., 2007; Kerr & Baird, 2007; Gedan, et al., 2011). Similarly, Boutwell and Westra (2014) analysed data from 13 hurricanes and tropical storms over a twelve-year period (1997-2008) and found that coastal wetlands in Louisiana most effectively mitigated damage from hurricanes in Category One (i.e. wind speed: 74-95 mile per hour; surge height: 4-5 feet, Kantha, 2006). Their results showed that, compared to Category One hurricanes, the coastal wetlands in Louisiana have lower capacity to mitigate damage from tropical storms and the Category Two hurricanes, and were far less protective against Category Three hurricanes. Massel, et al., (1999) also pointed out that coastal vegetation is most effective in reducing damage from typical storm-generated waves.

Apart from wave attenuation, mangroves and saltmarshes have also a critical role in reducing the risk of coastal erosion by trapping and stabilising sediment while dissipating wave energy (Mitra, 2013). This function of coastal vegetation in trapping sediment is attributed to the complex root system as well as the existence of a vast expanse of aerial roots (pneumatophores) that increase residence time of water and allows assimilation of suspended sediment (Mitra, 2013). By accumulating and holding sediments, coastal wetlands can modify a shoreline and maintain the long-term integrity of coastal settings (Gedan, et al., 2011). They are, therefore, referred to as ‘ecosystem engineers’ as they can modify and shape their habitat by trapping sediment (Kristensen, et al., 2008; Bouma, et al., 2010).

A field survey in Southern Thailand where approximately 50% of mangrove forests have been lost since 1961, found that during a period from 1967 to 1998 coastlines have experienced an annual erosion rate of 0.01 to 0.32 Km\(^2\)/year (Thampanya, et al., 2006). According to this survey, net accretion was only found in coastlines with intact mangrove cover. Studies reported that mangroves fringing muddy open waters can annually accumulate sediment up to 1000 t/km\(^2\) (Ellison, 2009; Woodroffe & Davies, 2009). As reported by Wolanski (2007) saltmarsh wetlands in the UK have the capacity of vertical accretion of sediments of 4.3 mm/yr\(^{-1}\) in average and of 82 mm/yr\(^{-1}\) at the highest rate.

Studies have employed different methods to quantify wave attenuation by coastal vegetation. These methods mainly include numerical wave modelling combined with laboratory experiments to examine wave evolution in the presence of vegetation (Marsooli & Wu, 2014; Stark, et al., 2016; Smith, et al., 2016). However, to identify the amount of avoided erosion and inundation due to presences of vegetation, it is essential to quantify how different plants with different size and characteristics can alter water level and erosion under different storm conditions (Guannel, et al., 2015). A number of numerical models have been developed for specific habitats under specific conditions. As reported by Guannel, et al., (2015), examples of these models have been used in the studies by Augustin, et al., (2009), Li and Zhang, (2010), and Maza, et al., (2013).

The InVEST Nearshore Waves and Erosion model\(^{27}\) is a newly developed model by the Natural Capital Project (Stanford University, USA) that provides quantitative estimates of the storm protection function of coastal ecosystems in terms of avoided erosion and flood mitigation. The model is composed of two primary sub-models: (i) the ‘profile generator’ and the ‘nearshore waves and erosion’. The model combines information of wave characteristics, nearshore bathymetry, backshore characteristics, location and physical characteristics of natural habitats and quantifies the avoided inundation and erosion by coastal vegetation under different management strategies.

\(^{24}\) Category Two hurricanes are characterised by: wind speed: 69-110 mile per hour; surge height: 6-8 feet (Kantha, 2006).

\(^{25}\) Category Three hurricanes are characterised by: wind speed: 111-130 mile per hour; surge height: 9-12 feet (Kantha, 2006).

\(^{26}\) Typical storm-generated are characterised by: wind speed: 39-73 mile per hour; surge height: 0-3 feet (Kantha, 2006).

The latest version (2015) of this model also incorporates the differences between various types of coastal wetlands with respect to their wave attenuation and erosion control functions. It, therefore, requires information about height, diameter and density of mangroves, marshes and seagrass beds. To differentiate between various mangrove species in terms of their coastal protection function, the model allows users to include information on the height, diameter and density of the mangroves’ root system, trunk and canopy\textsuperscript{28}. As an open source extension for ArcGIS, the model is new and no practical application to a case study has yet been reported in the literature; however, there is ongoing work to test the model.

Overall, it is argued that findings of the existing studies provide reasonable evidence that supports the importance of coastal wetlands in protecting shorelines from erosion, storm surge and small tsunami waves that consequently can reduce economic impacts on coastal properties (Alongi, 2008; Gedan, et al., 2011; Spalding, et al., 2014a&b). However, similar to hard-defence structures, wetlands as a stand-alone solution may not provide protection against extreme coastal hazards such as large-scale erosion, river meandering, severe storm surges and large tsunami waves (Spalding et al. 2014b).

Studies have also suggested combining green (nature-based) and grey (engineered) measures for increasing protection against coastal hazards (Gedan, et al., 2011; van Wesenbeeck, et al., 2016; Spalding, et al., 2014a). Such a combined solution needs to be implemented alongside other risk reduction measures including avoidance of new development in areas subject to coastal hazards, improved building design and standards, increased public awareness, efficient early warning system and evacuation procedures (Spalding, et al., 2014a) (Figure 5.3-2).

\textsuperscript{28} The idea for including differences between mangrove species in the model was partly informed by an online forum, to which the author had active contribution.
5.3.2. Adaptation to sea level rise and long-term sustainability of coastal wetlands

Climate change is expected to affect coastal wetlands due to rising sea level, more intense and frequent storm events and increasing temperature (Erwin, 2009; Lundquist, et al., 2014a). These changes may result in expansion or degradation of coastal wetlands over a long-term period (Lundquist, et al., 2014a). Due to their inherent ability to accumulate sediment and enhance surface elevation, mangrove and saltmarsh ecosystems are naturally able to cope with changes in sea level (Figure 5.3-3) (Lovelock, et al., 2015; Spalding, et al., 2014a). This gives mangrove and saltmarsh habitats an advantage over engineering structures, as they are highly dynamic systems and have in-built ability to respond to the changes in sea level (Spalding, et al., 2014a). Whereas, engineering structures often need costly updates.

Where enough space is available, mangrove and saltmarsh habitats also migrate landwards or seawards in response to sea level rise (Spalding, et al., 2014a). Therefore, long term sustainability of mangrove and saltmarsh ecosystems is affected in areas where there is not enough space for coastal wetlands to migrate inland (known as coastal squeeze) or where sediment accretion cannot keep pace with sea level rise (Ravit & Weis, 2014; Bouma, et al., 2014). In fact, the fate of coastal wetlands with respect to sea level rise is dependent on whether the balance in sediment accretion rates (considering erosion processes and relative

Figure 5.3-3. Elevation change due to sediment accretion by coastal wetlands (Spalding, et al., 2014a) (A) Contemporary natural shoreline; (B) Sediment accretion by coastal wetlands and elevation change in response to sea level rise; (C) Hard protection structure which can alter sedimentation, prevent wetland migration and cause subsidence and wetland loss; (D) Hybrid interventions which allow inland migration and sediment accretion. (HAT: Highest astronomical tide; MSL: Mean sea level)
loss of elevation due to sub-surface subsidence) in mangrove habitats is higher than sea level rise or whether there is enough space for them to migrate inland. This indicates that coastal wetlands will be eventually submerged as a result of sea level rise, if they cannot keep pace with changes in the sea level and if their landward migration is not possible. Therefore, as suggested by Webb, et al., (2013), assessing the vulnerability of coastal wetlands is the key step in developing a coastal adaptation plan. Monitoring is also essential to assess persistence and responses of coastal ecosystems over time (Webb, et al., 2013; Sasmito, et al., 2015).

Mechanisms for adaptation to sea level rise are classified into three main categories of ‘protect’, ‘retreat’ and ‘accommodate’ (IPCC, 2001; IPCC, 2000b). Strategies within the ‘protect’ category aim to reduce the risk of coastal hazards by decreasing their probability of occurrence mainly through construction of hard/engineering structures; while ‘accommodate’ strategies focus on improving the ability of society to cope with the effects of coastal hazards (e.g. elevating and retrofitting buildings) (IPCC, 2001). Retreat strategies mainly reduce the risk of hazards by limiting their potential effects (re-location of coastal properties) (IPCC, 2001). Schematic illustration of these adaptation strategies is provided in Figure 5.3-4.

Managed retreat is increasingly undertaken (e.g. in the Netherlands and the United Kingdom, Shepard, et al., 2011) to re-connect coastal lands to the tidal systems through, for example, opening seawalls and filling drainage channels. However, it requires purchasing coastal properties which in most cases has been challenging and costly (Spalding, et al., 2014b).

5.3.3. Restoration of coastal wetlands for coastal protection and its economic benefits

There are increasing examples of restoration of coastal wetlands (particularly mangroves and saltmarshes) for coastal protection and stabilisation including the restoration of tidal marshes in Scheldt Estuary, the Netherlands (Eertman, et al., 2002), mangroves rehabilitation in Australia along the coastlines of New South Wales (Stewart & Fairfull, 2008), Western Port Bay, Victoria (Kirkman & Boon, 2012) and Brisbane.

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29 Section 6.6 of the 2001 IPCC report and Section 15.3.3 of the 2000 IPCC report.
(Saenger, 1996); restoration of Gulf Coast wetlands as natural storm surge barriers (Barbier & Enchelmeyer, 2014); protection and restoration of mangroves in coasts of Guyana, South America (Anthony & Gratiot, 2012) and mangrove restoration activities in Bangladesh and Philippines (Spalding, et al., 2014b).

Unlike hard infrastructure with high establishment and maintenance cost, conservation and restoration of coastal wetlands provides a cost-effective option (Duarte, et al., 2013). For example, construction of a 500-m rock wall in St. Kilda, north of Adelaide in South Australia, cost around AUD 1 million (or AUD 2,000 per metre) (Harty, 2009). Whereas, based on the data reported for mangroves restoration projects, Lewis (2001) estimated that mangrove restoration approximately cost between US$ 225 – 216,000 ha$^{-1}$ (or US$ 0.02 – 22 per m$^2$). In Vietnam, plantation of 12,000 ha of mangroves between 1994 and 2002 did approximately cost US$ 1.1 million. However, it saved US$ 7.3 million per year because it reduced the cost of levee and dyke maintenance while protected inland areas against Wukong typhoon in 2000 (Convention on Biological Diversity, 2009). The planted mangrove forests have also provided livelihood benefits to coastal communities.

Marshland stabilisation, marshland creation and freshwater diversion are strategies for restoring New Orleans’s coastal wetlands as part of the coastal defence system and cost about US$ 2 per m$^2$, US$ 4.5 per m$^2$ and US$ 14.3 million respectively. The cost for heightening a dyke by 1m, as a hard defence measure in New Orleans, is estimated at between US$ 7-8 million per km (Carabine, et al., 2015).

As mentioned earlier, coastal wetlands can naturally adapt to the effects of climate change such as sea level rise, while hard defence structures may require costly upgrades (Duarte, et al., 2013). It is also argued that artificial structures may affect natural sediment exchange and therefore exacerbate coastal erosion (Spalding, et al., 2014b). In addition, coastal wetlands which are restored for coastal protection, can also provide other co-benefits such as biodiversity support, carbon storage, fisheries, tourism and recreation (Spalding, et al., 2014a; Duarte, et al., 2013; Alongi, 2012). It is, therefore, suggested that the value of these co-benefits needs to be considered in cost-benefit assessment that involves comparing nature based (e.g. coastal wetland restoration) with hard structures (Costanza, et al., 2008; Costanza, et al., 2014).

Barbier (2016) has also suggested considering and assessing the additional benefits of restoration of estuarine and coastal ecosystems (ECEs) in terms of increasing Net Domestic Product (NDP) and economic wealth. He applied a wealth accounting framework to a saltmarsh restoration project in Louisiana, and concluded that creation of approximately 100km$^2$ marsh wetlands over 2012–2031 (as suggested in 2012 Louisiana Coastal Master Plan) can result in net wealth of over US$ 37 per person in real term (i.e. 2005-dollar value). He has concluded that ECEs which provide coastal protection functions are important ecological capitals which need to be restored and maintained.

However, it is suggested that achieving multiple co-benefits from mangrove protection or restoration may be challenging in practice (Atkinson, et al., 2016; Faulkner, 2004; Lee, et al., 2006). For example, coastal wetlands with high values for coastal protection are often located at close proximity to urban or semi-urban areas and are likely to have degraded biodiversity and fishery services as are exposed to human-induced impacts such as pollution, hydrological alteration, etc. (Faulkner, 2004; Lee, et al., 2006).

Atkinson, et al., (2016) mapped priority mangrove sites for four ecosystem services (coastal protection, fisheries, biodiversity and carbon storage) in Fiji and concluded that managing a mangrove site for a particular ecosystem service, could also provide other co-benefits, but at different levels (Table 5.3-1). For example, 47.7% of priority areas for coastal protection services were also priority areas for carbon storage services. As the table indicates, the co-benefits for biodiversity were much lower compared to other services. Atkinson, et al., (2016) highlighted the importance of mapping of ecosystem services for understanding the possible trade-offs and synergies in ecosystem services and the efficient allocation of the limited resources for mangrove conservation.
Table 5.3-1. Percent by area of co-benefits when mangroves are managed for a specific ecosystem service
(Atkinson, et al., 2016)

<table>
<thead>
<tr>
<th>The objective ES being managed</th>
<th>Coastal protection</th>
<th>Fisheries</th>
<th>Biodiversity</th>
<th>Carbon storage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coastal protection</td>
<td>-</td>
<td>42.6</td>
<td>16.1</td>
<td>47.7</td>
</tr>
<tr>
<td>Fisheries</td>
<td>41.8</td>
<td>-</td>
<td>0.7</td>
<td>39.3</td>
</tr>
<tr>
<td>Biodiversity</td>
<td>16.3</td>
<td>0.7</td>
<td>-</td>
<td>14.5</td>
</tr>
<tr>
<td>Carbon storage</td>
<td>46</td>
<td>41.2</td>
<td>13.8</td>
<td>-</td>
</tr>
</tbody>
</table>

They used a benefit-transfer approach for economic valuation of the selected ecosystem services using available data and literature from Fiji and neighbouring islands. However, due to the limited local data, they have made a number of assumptions that resulted in some level of uncertainty in their estimates. For example, they only considered the width of mangrove forests in calculating the wave attenuation function of mangrove ecosystems and did not take into account the other contributing parameters such as species diversity and density. To estimate carbon storage, they assumed a uniform and constant carbon stock of 900 t CO$_2$e ha$^{-1}$ and speculated that about 60% of this stock can be potentially released where mangrove removal results in a 45% annual rate of decay. Given the lack of local data, species-specific carbon storage and sediment characteristics were not considered in calculating the carbon storage value of mangrove ecosystems.

Studies on the valuation of protective services of coastal wetlands are relatively new and often follow two approaches: (a) estimate the cost as an equivalent measure of protection and (b) estimate the degree to which wetland presence is associated with lower damage (Boutwell & Westra, 2014). Most of the studies have adopted the latter approach which is called Damage Cost Avoided (DCA) method (Costanza, et al., 2008; Barbier, et al., 2013; Barbier & Enchelmeyer, 2014; Boutwell & Westra, 2014).

Through an economic analysis of the damage to coastal properties and infrastructure caused by 34 major hurricanes striking the US coast between 1980 and 2004, Costanza, et al., (2008) quantified and mapped (Figure 5.3-5) the storm protection values of the US coastal wetlands and concluded that these ecosystems had significantly reduced damage to coastal communities. They suggested that information on historical damage, storm characteristics (intensity and probability), wetlands location and characteristics, built infrastructure location and value and population distribution is required to monetise the protective value of coastal wetlands.

According to the estimates by Costanza, et al., (2008), the annual value of the US coastal wetlands for storm protection is between US$ 250 to US$ 51,000 ha$^{-1}$ yr$^{-1}$, with an average of US$ 8,240 ha$^{-1}$ yr$^{-1}$ (median=US$ 3,230 ha^{-1} yr^{-1};$ SD = US$ 12,418 ha^{-1} yr^{-1}$). The relatively high variation in storm protection value of coastal wetlands in different locations is argued to be due to differences in type, extent, and condition of the wetlands, storm strength and the value of properties and infrastructure in the affected areas (Costanza, et al., 2008). Similarly, in a literature review by Schmidt, et al., (2014), a value of US$ 196 – 1,140 ha$^{-1}$ yr$^{-1}$ was estimated as the storm protection value of estuarine marshes in Georgia.

The recent studies by De Groot, et al., (2012) and Costanza, et al., (2014) provided a value of US$ 194,000 ha$^{-1}$ yr$^{-1}$ as the value of ecosystem services provided by coastal wetlands including tidal marsh, mangroves and saltmarsh wetlands. This estimate excludes carbon sequestration but includes the value of 13 ecosystems services (including disturbance moderation) which are classified within the broad categories of ‘provisioning’, ‘regulating’, ‘habitat’ and ‘cultural’ services. A study by the Economics of Ecosystems and Biodiversity (TEEB) global initiative (Russi, et al., 2013), identified a value of US$ 215,000 ha$^{-1}$ yr$^{-1}$ for 11 ecosystem services (including moderation of extreme event and climate regulation) provided by mangrove and tidal marshes.
Figure 5.3-5. Map of the total value of the US coastal wetlands for storm protection estimated for 1 km x 1 km pixels (Costanza, et al., 2008)

The results of an economic valuation of coastal wetlands with respect to their function of reducing expected property damage from hurricane storm surges in south-eastern Louisiana, including Greater New Orleans, showed that an increase by 0.1 in wetland continuity (i.e. the wetland-water ratio per m along the transect) and an increase by 0.001 in vegetation roughness (through changes in wetland vegetation) reduced property damage by US$ 99 -133 and US$ 24 - 43, respectively (Barbier, 2013). Boutwell and Westra, (2014) estimated an annual value of US$ 374.72 ha\(^{-1}\) as the coastal protection value of estuarine wetlands in Louisiana.

A value of US$ 300,000 km\(^{-2}\) has been also estimated as the value of Malaysian mangroves only for storm protection and flood control (Convention on Biological Diversity, 2009). This estimate has been based on the cost of replacing the mangroves with artificial structures such as seawalls that provide the same protection (Ramsar Secretariat, 2001, Cited in Gilman, et al., 2008). Results of a study by Barbier (2007) showed that over 1979-1996 and 1996-2004, a 1 km\(^2\) loss of mangroves in Southern Thailand increased the storm damage by about US$ 585,000 and US$ 187,898 km\(^{-2}\), respectively.

5.3.4. Factors affecting wave attenuation at coastal areas

The wave attenuation function of coastal wetlands varies across time and space (Barbier, et al., 2008; Koch, et al., 2009; Spalding, et al., 2013). The results of a storm surge modelling in Southern Louisiana indicated that wave attenuation is influenced by bottom friction caused by wetland vegetation, the surrounding coastal landscape and the characteristics of the storm surge itself (strength and duration) (Wamsley, et al., 2010). The literature also suggests that the rate of wave attenuation by mangroves depends on the vegetation structure such as trunk diameter, root structure, stem density and the characteristics of the incident waves (Mitra, 2013; Marsooli & Wu, 2014). Tanaka, et al., (2007) reported that Rhizophora species particularly Rhizophora apiculata and Rhizophora mucronata provided effective coastal protection.
against tsunami damage in Sri Lanka and Thailand owing to their complex aerial root structure (Known as prop roots, McIvor, et al., (2012a)). They also suggested that a combination of different mangrove species can provide more effective protection as the level of resistance varies between different species. Similar to the prop roots of *Rhizophora* spp., the pneumatophores of *Avicennia* spp. create higher wave attenuation at shallow depths. As water level increases, the rate of attenuation is influenced by the size, density and height of trunks and branches (McIvor, et al., 2012a).

There is evidence that dense mangrove forest reduces wave energy more effectively than low density forest due to the greater drag force to moving water (Mazda, et al., 1997; Massel, et al., 1999). A study by Bao (2011) found that in areas with tall and dense mangrove forests, a smaller width of mangroves provides acceptable coastal protection. He concluded that a minimum width of 40-80 m is required for *Rhizophora mucronata* to be able to provide coastal protection from a 3-m wave; whereas the minimum width for *Avicennia marina* varied from 80 - 120 to >240 m between two different sites due to differences in tree height, density and canopy closure.

It is suggested that quantifying the relationship between the features of mangrove forests and the level of attenuation provided under different wave scenarios can inform restoration and management of mangrove forests towards maximising their potential for coastal protection. For example, in areas with dense mangrove forests and small waves, a narrow width of mangroves may provide effective protection; while a wider band of mangroves is required in an area with frequent storms and less dense mangrove forests (McIvor, et al., 2012a).

Lacambra, et al., (2008) reported that the optimum width of mangrove forests ranges from 100 to 1,500 metres depending on the type of mangrove species and local characteristics. They also provided examples of 500 to 1,000 metres of mangroves established as a buffer zone across the coastline of Mekong delta in Vietnam and mangrove greenbelts of 20 to 50 metres in Philippines, without specifying the mangrove species. In Malaysia, under the 1950s regulations, a mangrove buffer zone of 200 m is required in front of a coastal defence structure protecting agricultural lands (Othman, 1994, Cited in Lacambra, et al., 2008). Evidence indicates that well-established, extensive mangrove cover (~1,000 m wide) at Pichavaram mangrove forest in Tamil Nadu, India strongly protected coastal villages against the Indian Ocean Tsunami. The mangrove forest in this area included a 10 m wide *Rhizophora apiculata* and *Rhizophora mucronata* trees followed by approximately 1000 m wide *Bruguiera*, *Excoecaria* and *Avicennia marina* (Government of Tamil Nadu, 2015).

Analysis of the hydrological data from a tsunami that struck Papua New Guinea in 1998, showed that a 100-m wide forest belt at a density of 3,000 trees/ha (or 30 trees/100m²) can reduce the maximum tsunami flow pressure by about 90 percent (Hiraishi & Harada, 2003). Furthermore, based on the results of studies that examined the relationship between waves height and coastal vegetation, Hiraishi (2008) concluded that a total of 400 trees per 10 m² of coastline is needed to decrease the tsunami power to the level less than 15% of original.

Bouma, et al., (2014) argued that the size (width) of intertidal ecosystems needed to provide a specific wave attenuation function increases with increase in tidal range (vertical difference between low and high tides) (Figure 5.3-6). As the figure shows, a 1000 m wide coastal habitat can provide 90% attenuation when the tidal range is 5 m, but its ability to attenuate a wave decreases by 50% when the tidal range reaches 8 m. Fragmentation also affects wave attenuation by coastal wetlands (Renaud, et al., 2013; Spalding, et al., 2014b). It is argued that fragmented wetlands can channel and increase the wave energy, and thus less fragmented ecosystems provide greater attenuation (Renaud, et al., 2013). As mentioned in the previous section, wetland continuity and roughness could also affect the attenuation effect of coastal wetlands.
Based on review of a number of studies, Spalding, et al., (2014c) identified properties of mangrove forests that determine the wave attenuation potential of mangrove forests (Figure 5.3-7). As the figure indicates, the average width of mangrove forest required for coastal protection ranges between hundreds of metres to thousands of metres according to the type of hazard. They have concluded that similar to hard protection structures, mangroves are unlikely to provide effective protection against tsunami waves.

A study by Mazda, et al., (2006) identified that wave attenuation ($r$) can be considered as a proportion of wave height reduction per metre of land traversed (e.g. $r = 0.01$ means a 1% reduction in wave height per metre which indicates 100% reduction across a 100-metre stretch). Their study showed that attenuation was two times smaller for large storm waves (average height of 164 cm) with large traverse distances (average distance of 5,863 m) than small wind waves (average height of 22 cm) with small traverse distances (average distance of 63 m).

Findings of the study by Mazda, et al., (2006) is consistent with the results of a number of previous studies (Knutson, et al., 1982; Wayne, 1976; Morgan, et al., 2009) which also showed more than 60% attenuation of small waves (<0.3 m) happen in under 20 m traverse distances. Importantly, great reduction in wave height and energy across relatively small traverse distances and a nonlinear relationship between wave attenuation and wetland size indicate that even narrow and small wetlands (e.g. narrow fringing marsh wetlands composed of Spartina alterniflora) can provide protection but mainly against small waves (Morgan, et al., 2009; Gedan, et al., 2011). A study in the UK (Möller, et al., 1999, Cited in Spalding, et al., 2014b) also concluded that saltmarsh habitats could reduce height and energy of short-period wind waves (water depth between 0.52 m and 1.39 m) by an average of 61% and 82%.
It is also reported that the presence of barrier islands, wind speed, wave shoaling (as a result of decrease in water depth) and storm duration can affect surge height and magnitude at wetland margins (Gönnert, et al., 2001; Resio & Westerink, 2008). For instance, a steady wind during Hurricane Rita overwhelmed drag from coastal vegetation and caused an increase of surge heights across nearly 40 km of saltmarsh in Louisiana (Resio & Westerink, 2008). Also, wind-induced short-period waves are subjected to greater attenuation compared to long-period swell waves (Mitra, 2013). Another conclusion was that wave attenuation decreases with an increase in water depth over coastal wetland habitats (i.e. canopies become submerged). However, submerged coastal wetland vegetation can still attenuate waves (Spalding, et al., 2013).

Field measurements of flood attenuation within a large tidal marsh wetland across the south-western part of the Netherlands (the Western Scheldt estuary) indicated that tidal marsh wetlands have the ability to attenuate storm waves (Stark, et al., 2016). However, this function varies between different locations and is associated with the presence and geometry of nearby channels, marsh width and elevation as well as the wave height (Stark, et al., 2016). The maximum attenuation was found across narrow channels with a large width of tidal marsh platform; while narrow marshes across wider channels showed a smaller rate of attenuation (Stark, et al., 2016).

A study by Shepard, et al., (2011) found that there is a positive correlation between vegetation density, marsh size and biomass production with wave dissipation and coastline stabilisation. Through a meta-analysis of 75 peer-reviewed studies, Shepard, et al., (2011) identified the most important factors that affect wave attenuation in saltmarsh ecosystems (Figure 5.3-8).

Overall, studies conclude that the wave attenuation function of saltmarsh ecosystems is associated with marsh width and elevation (Stark, et al., 2016), stem stiffness (Bouma, et al., 2014), standing biomass (Koch, et al., 2009; Bouma, et al., 2014), inundation height (Möller, et al., 2011), wave characteristics

<table>
<thead>
<tr>
<th>Hazard</th>
<th>Waves</th>
<th>Storm surges</th>
<th>Tsunami</th>
<th>Erosion</th>
<th>Sea level rise</th>
</tr>
</thead>
<tbody>
<tr>
<td>Width</td>
<td>Hundreds of meters needed to significantly reduce waves (wave height is reduced by 13-66% per 100m of mangroves)</td>
<td>Hundreds of meters needed to significantly reduce wind and waves on top of surge</td>
<td>Hundreds of meters needed to reduce tsunami flood depth by 5-30%</td>
<td>Sufficient mangrove forest width needs to be present to maintain sediment balance. This can help to prevent erosion and may encourage active soil build-up.</td>
<td></td>
</tr>
<tr>
<td>Structure</td>
<td>The more obstacles the better: dense aerial root systems and branches help attenuate waves</td>
<td>Open channels and lagoons allow free passage, while dense aerial root systems and canopies obstruct flow</td>
<td>Mangroves do not provide a secure defence (nor do any engineered defences)</td>
<td>Complex aerial root systems help slow water flows, allowing sediment to settle and causing sediment to accrete rather than erode.</td>
<td></td>
</tr>
<tr>
<td>Tree Size</td>
<td>Young &amp; small mangroves can already be effective</td>
<td>Smaller trees and shrubs may be overtopped by tsunamis and the very largest storm surges</td>
<td>Young trees already enable soils to build up. The more biomass input into the soil the better.</td>
<td>Allow room for landward retreat of the mangrove</td>
<td></td>
</tr>
<tr>
<td>Link to other ecosystems</td>
<td>Sand dunes, barrier islands, saltmarshes, seagrasses and coral reefs can all play an additional role in reducing waves</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Underpinning factors</td>
<td>Healthy mangroves are a prerequisite for all aspects of coastal protection. Healthy mangroves require sufficient sediment and fresh water supply and connections with other ecosystems. Conversely, pollution, subsidence (due to deep groundwater/oil extraction or oxidation upon conversion) and unsustainable use jeopardizes mangroves.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 5.3-7. Mangrove forest properties and their role in coastal protection (Spalding, et al., 2014c)
(Shepard, et al., 2011) and presence and geometry of nearby channels (Stark, et al., 2016). Sediment accumulated in saltmarsh platforms can also contribute to wave attenuation if strongly stabilised (Bouma, et al., 2014). Seasonal variations in wave attenuation resulting from changes in aboveground biomass is mainly reported for a number of saltmarsh species (e.g. Salicornia spp.) in areas with temperate climates where aboveground biomass partly or completely disappears during winter. The absence of biomass decreases the wave attenuation function (Bouma, et al., 2014).

Compared to mangrove forests and saltmarsh habitats, relatively few studies have investigated the factors that can potentially influence wave attenuation in seagrass ecosystems. The main studies, as reported by Bouma, et al., (2014), include Manca, et al., (2012), Maza, et al., (2013), Paul, et al., (2012), and Infantes, et al., (2011). These studies suggest that wave attenuation by seagrass beds is strongly correlated with water depth and seasonal variations of shoot density. A minimum density of 1000 shoots m$^{-2}$ is required for the seagrass *Ruppia maritima* to be able to provide a wave attenuation function (Chen, et al., 2007, Cited in Koch, et al., 2009). Water quality is also argued to affect wave attenuation potential of seagrass beds (Bouma, et al., 2014). The happens in nutrient-rich environments, where seagrasses are more fragile and easily break when exposed to waves (Bouma, et al., 2014). The indirect impacts of poor water quality on wave attenuation through affecting ecosystems’ stability and vegetation development has been less studied and is a potential area for future research (Bouma, et al., 2014).

Overall, a complex set of interactions between storms and wetlands influence wave attenuation capacity of coastal wetlands and multiple parameters such as structure, size and density of wetlands as well as the surrounding local bathymetry, topography and storm characteristics determine this capacity (Wamsley, et al., 2010; Gedan, et al., 2011; Spalding, et al., 2014b). These differences make wave attenuation too complicated to be described with a simple rule (Gedan, et al., 2011). A full list of factors affecting the coastal protection function of coastal wetlands, based on the findings of a number of recent peer-reviewed articles in the literature (Lacambra, et al., 2008; Shepard, et al., 2011; Gedan, et al., 2011; McIvor, et al., 2012b; McIvor, et al., 2012a) is provided by Spalding et al (2014b) (Table 5.3-2). Given the spatial and temporal variations in risk reduction potential of coastal wetlands, storm attenuation benefits of coastal wetlands need to be estimated and quantified at the site level (Spalding, et al., 2013; Bouma, et al., 2014).
As suggested by Costanza, et al., (2008) information on historical damage, storm characteristics (intensity and probability), wetlands location and characteristics, built infrastructure location and value, population distribution, etc. is required to estimate the storm protection values of coastal wetlands.

The subsequent sections look at mangrove and saltmarsh ecosystems in the Auckland region with respect to their climate change benefits.

Table 5.3-2. Factors affecting the coastal protection function of coastal wetlands (Spalding, et al., 2014b)

<table>
<thead>
<tr>
<th>Coastal wetlands</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abiotic variables</td>
</tr>
<tr>
<td>Wave characteristics</td>
</tr>
<tr>
<td>Adjacent land use</td>
</tr>
<tr>
<td>Soil characteristics</td>
</tr>
<tr>
<td>Presence and frequency of disturbances</td>
</tr>
<tr>
<td>Topography</td>
</tr>
<tr>
<td>Slope</td>
</tr>
<tr>
<td>Bathymetry</td>
</tr>
<tr>
<td>Water depth over plants</td>
</tr>
<tr>
<td>Drainage systems</td>
</tr>
<tr>
<td>Prevailing tides</td>
</tr>
<tr>
<td>Distance to shore</td>
</tr>
<tr>
<td>Distance from sediment source</td>
</tr>
<tr>
<td>Distance to other ecosystems</td>
</tr>
<tr>
<td>Exposure</td>
</tr>
<tr>
<td>Ecosystem variables</td>
</tr>
<tr>
<td>Habitat width</td>
</tr>
<tr>
<td>Plant density</td>
</tr>
<tr>
<td>Vegetation structure, mangroves: canopy height, aerial root, physiognomy, age class distribution, sub-canopy elements</td>
</tr>
<tr>
<td>Vegetation structure, salt marshes: plant height, vegetation stiffness</td>
</tr>
<tr>
<td>Resistance and resilience (capacity to survive or recover from impacts)</td>
</tr>
<tr>
<td>Fragmentation</td>
</tr>
<tr>
<td>Dominant species</td>
</tr>
</tbody>
</table>

5.4. Distribution of mangroves and saltmarshes in the Auckland region

Mangroves are naturally growing in warm tropical and subtropical coastlines, but a few mangrove species, including the New Zealand’s only mangrove species (*Avicennia marina subsp. Australasica*), can also thrive in colder climates (Morrisey, et al., 2007). In New Zealand, mono-species mangrove forests are only found across the northern part of the North Island of about latitude 38°S, from Kawhia Harbour on the west coast and Ohiwa Harbour on the east coast (Harty, 2009). It is reported that *Avicennia marina subsp. Australasica* is native to New Zealand and has existed for about 19 million years.

In Auckland, mangroves grow in the shallow areas of most of the region’s estuaries including behind beach deposits at the mouths of streams and along streams (Auckland Council, 2013). Based on the information from the LCDBv.4 (as explained in Chapter 3, Section 3.3.2), the area of mangrove forests in the Auckland region in 2012 is estimated at 8,928 ha. Adding this figure to the existing estimates of the total extent of mangrove forest in the Northland, Bay of Plenty and Waikato regions (Table 5.4-1) gives a total area of 21,637 ha, which is within the range of the total area of mangroves in New Zealand (estimated to be between 19,349 and 22,500 ha, Silliman, et al., 2009; Morrisey, et al., 2010; Patterson & Cole, 2013). This

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[^30]: [Accessed May 2016]
means that the figure of 8,928 ha is a reasonably valid estimate of the extent of Auckland’s mangroves in 2012.

Table 5.4-1. Available data on mangrove extent in New Zealand’s regions

<table>
<thead>
<tr>
<th>Region</th>
<th>Estuary</th>
<th>Area (ha)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Auckland</td>
<td>Total region</td>
<td>8,928</td>
<td>This research (area in 2012)</td>
</tr>
<tr>
<td></td>
<td>Kaipara Harbour</td>
<td>6,167</td>
<td>Morrisey, et al., 2010. P.107</td>
</tr>
<tr>
<td>Northland</td>
<td>Total region</td>
<td>6,300</td>
<td>Northland Regional Council(^{31})</td>
</tr>
<tr>
<td>Bay of Plenty</td>
<td>Firth of Thames</td>
<td>1,100</td>
<td>Morrisey, et al., 2010., P.107</td>
</tr>
<tr>
<td></td>
<td>Rangaunu Harbour</td>
<td>2,415.5</td>
<td>Personal communication (email) with the Waikato Regional Council on 12 January 2016</td>
</tr>
<tr>
<td></td>
<td>Tauranga Harbour</td>
<td>623</td>
<td>Personal communication (email) with the Waikato Regional Council on 12 January 2016</td>
</tr>
<tr>
<td>Waikato</td>
<td>-</td>
<td>2,270.50</td>
<td>The latest estimate of the total area of NZ’s mangrove habitats (19,349) is provided by Patterson and Cole (2013). An estimate of 22,500 is reported by Silliman et al. (2009), as the total area of mangrove habitats in New Zealand. Morrisey, et al., (2010) has also reported that the area of mangrove forest in Firth of Thames, Rangaunu, Tauranga and Kaipara Harbours accounts for 47% of the New Zealand total. Therefore, the total area of mangrove forest in New Zealand is estimated at 21,900 ha.</td>
</tr>
</tbody>
</table>

Total    | 21,637

Saltmarsh wetlands in Auckland are usually found at the head of estuaries and in low energy environments where wave action is reduced and sediment is deposited (Auckland Council, 2013). The main saltmarsh species in the Auckland region are mentioned in Chapter 3, Section 3.3.1.2. The latest GIS layers provided by the LCDBv.4 (2014) gives an estimated total area of 2,056 ha in 2012 for herbaceous saline vegetation (considered as the saltmarsh category) in the Auckland region. The spatial extent of mangrove and saltmarsh ecosystems in the Auckland region in 2012, prepared in this research, is shown in Figure 5.4-1.

Figure 5.4-1. Spatial extent of mangrove and saltmarsh ecosystems in the Auckland region in 2012 (This study)
5.5. Modification of mangrove and saltmarsh habitats over time

5.5.1. Historic loss

Information about the original extent of coastal wetlands in the Auckland region is lacking, but as reported by Lindsay, et al., (2009) only 3% of the original area (86,956 ha) of coastal forest across the Auckland region currently remains. Although, it is not clearly stated what ecosystems are considered as coastal forests.

Mangrove habitats in New Zealand have been historically reclaimed for agricultural activities. A 1973 MSc thesis (Winn, 1973) reported a growing trend of reclamation of the Waitemata harbour in Auckland from approximately 400 acres (162 hectares) in 1915 to 2,400 acres (970 hectares) in 1970. There is also evidence of extensive reclamation of mangroves/coastal wetlands in the Kaipara Harbour prior to 1970s (Auckland Council, 2011). This corresponds to the global trend of degradation of coastal wetlands. As an evidence, in New York City, only 15% (between 2,270 ha and 4,050 ha) of the original extent of coastal wetlands (between 15,000 ha and 27,000 ha) is left based on the data for 2012, which indicates a loss rate of 85% (New York City Department of Parks and Recreation, 2012).

It is reported that extensive reclamation and drainage during the early 20th century was presumably the main cause of the net loss of the mangrove forests across the Auckland region. Foreshore reclamation for construction of ports and harbours, as well as building causeways in estuaries are argued to be the main causes of saltmarsh degradation in New Zealand (Hume, 2003). In response to concerns over the extent of reclamations in New Zealand, the Harbours Amendment Act 1977 included a provision that prohibited the infilling of estuaries and tidal ecosystems for agricultural purposes (Auckland Council, 2011).

5.5.2. Expansion of mangroves

Geological studies of northern New Zealand have indicated that sediment transport into estuaries and harbours increased dramatically following human, and particularly European, settlement and was mainly associated with removal of terrestrial vegetative cover and catchment-based land use activities (Morrisey, et al., 2007&2010). Deposition of sediments in estuaries and harbours resulted in a reduction in tidal prism (the volume of water flowing in and out of an estuary or harbour) and provided a desirable habitat for mangrove expansion (Bell, et al., 2000).

In addition, changes to water flow associated with reclamation of estuaries for roads and causeways are argued to have slowed down the ability of estuaries to flush sediments back to coastal environment and therefore have led to increased sediment deposition in estuaries (Auckland Council, 2011). This has adversely affected the natural process of beach nourishment and accelerated the rate of mangrove expansion (Auckland Council, 2011). An example is construction of the Herald Island causeway in 1958 which has led to ongoing sedimentation within the confined bay (Auckland Regional Council, 2004). Causeway construction in Pahurehure Inlet (Manukau Harbour) has also resulted in mangrove expansion.

Evidence indicates that seaward expansion of mangroves (averaging 4% per year) since 1940s has occurred in northern New Zealand mainly as a response to increased sedimentation due to catchment deforestation (Morrisey, et al., 2010; Swales, et al., 2015). However, there is no strong evidence of extensive change in mangrove area in most places where there are concerns about mangrove expansion (Auckland Council, 2011). There is also little evidence in support of the argument that expansion of mangroves has resulted in loss of saltmarsh ecosystems in Auckland. It is because mangroves have generally expanded seaward, while saltmarsh habitats mainly grow landwards of mangroves and in higher elevation. It is, however, argued that expansion of mangroves may adversely affect the roosting and feeding areas of wading birds.

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while providing habitats for other bird species that depend on mangroves for feeding, roosting and breeding.\textsuperscript{35}

Data on mangrove expansion for a number of estuaries in northern New Zealand is reported by Morrisey, et al., (2010). Table 5.5-1 provides a summary of the data on mangrove expansion in a number of estuaries across the Auckland region. As the data shows, the area of mangrove forests has increased in most estuaries between 1939 and 2007. However, there has been a decrease in the area of mangroves in a number of estuaries such as central Waitemata Harbour and Shoal Bay between 1959 and 2007. As indicated in the table, mangrove forests across the selected estuaries have expanded by approximately 212 hectares between 1939 and 2007. However, as stated by Morrisey, et al., (2007), given the lack of quantitative estimates of the rate of mangrove loss in New Zealand, it is not possible to determine a net loss or gain in the extent of mangrove forests up to the present.

Table 5.5-1. Historical changes in the area of mangrove habitats across a number of estuaries in the Auckland region (Morrisey, et al., 2010)

<table>
<thead>
<tr>
<th>Location</th>
<th>Time period</th>
<th>Habitat area (ha)</th>
<th>Habitat area change (ha)</th>
<th>Original report</th>
</tr>
</thead>
<tbody>
<tr>
<td>Whangapoua, Great Barrier Island, Auckland</td>
<td>1960–1999</td>
<td>88–162.8</td>
<td>74.8</td>
<td>Morrisey et al. 1999</td>
</tr>
<tr>
<td>Lucas Creek, Auckland</td>
<td>1950–1996</td>
<td>47.5–50.5</td>
<td>3</td>
<td>Morrisey et al. 1999</td>
</tr>
<tr>
<td><strong>Total change (1939-2007)</strong></td>
<td></td>
<td></td>
<td><strong>+ 212.3</strong></td>
<td></td>
</tr>
</tbody>
</table>

5.5.3. Mangrove removal and saltmarsh degradation

The concerns over expansion of mangroves have led to a commonly held perception, particularly among coastal communities, that mangroves are a pest or weed and have not naturally been part of New Zealand estuaries (Green, et al., 2003; Harty, 2009). The perception has resulted in an increased number of consent applications from coastal communities to remove mangroves and to prevent further expansion (Morrisey, et al., 2007) with the purpose of reinstating navigational, recreational and amenity values of estuaries (Lundquist, et al., 2014b). Aside from consented mangrove removal, there is also evidence of illegal removals (cutting and poisoning of mangrove trees) in a number of Auckland’s estuaries. An example is Whangateau Harbour where coastal communities have raised serious concerns about mangrove expansion (Auckland Council, 2011) and illegally removed and poisoned mangrove trees on the thinking that the removal would improve the amenity and recreational values of the coast (Figure 5.5-1).

However, findings from a monitoring of a number of mangrove removal sites (Lundquist, et al., 2014b) indicated that due to hydrodynamic differences (e.g., wind-wave exposure, tidal currents) between different estuaries, removal of mangroves may not necessarily result in the sites to return to their previous sandflat condition. The observations indicated that recovery to a sandy status has not occurred in most removal sites (Lundquist, et al., 2014b).

\textsuperscript{35} http://www.marinenz.org.nz/documents/new_zelands_mangroves_summary.pdf [Accessed May 2016]
According to the information from Lundquist, et al., (2014b) and the Auckland Council (unpublished data), approximately 131 ha of mangroves were legally removed from different sites across the Auckland region between 2006 and 2015. However, given the lack of data on the area of removal for a number of sites and the unknown area of illegal removal, this figure may not represent the actual removal of mangroves in the region. Given the annual mangrove loss within the last ten years, a value of 0.15% was estimated in this study as the rate of mangrove loss per year (around 13 ha per year). This number is much lower than the global rate of mangrove loss (0.7–3 % per year, Pendleton, et al., 2012). This estimate only indicates a gross loss, as information about the extent of mangrove expansion in the Auckland region between 2006 and 2015 was not available.

No data is specifically reported for the loss or conversion rate of saltmarshes in New Zealand and Auckland. A recent study by Stevens and Robertson (2014) on coastal habitats in Havelock Estuary, Marlborough region, New Zealand, found that the area of saltmarshes has decreased by 15% between 2001 and 2014 mainly as a result of cordgrass (Spartina) removal. Assuming similar trend in Auckland, the rate of saltmarsh loss in the Auckland region is roughly estimated at 1% per year.

![Figure 5.5-1. a: Poisoned mangrove tree, Whangateau Harbour, Auckland; b: Illegal cutting of mangroves, Whangateau Harbour, Auckland (Whangateau HarbourCare Group, 2015b)](image)

5.6. Carbon storage potential of Auckland’s coastal wetlands

5.6.1. Estimating sediment CS&S

Using data from both local and similar global studies (as explained in Chapter 3, Section 3.3.1.2), this research estimated that temperate *Avicennia marina* can potentially store between 104 - 144 t C ha\(^{-1}\) in their sediment (to 100 cm depth) with an average value of 124 t C ha\(^{-1}\) (equal to 454 t CO\(_2\)e ha\(^{-1}\)). This mangrove species can also sequester approximately 5 tonnes of CO\(_2\) per hectare per year (Table 5.6-1). Saltmarsh ecosystems in temperate climates (Cfa/Cfb) dominated by *Spartina alterniflora*, *Spartina anglica*, *Sarcocornia quinqueflora* and *Juncus kraussii* store approximately 307 t C ha\(^{-1}\) (1125 t CO\(_2\)e ha\(^{-1}\)) and sequester 8 tonnes of CO\(_2\) per hectare per year.

As noted in Chapter 3 (Section 3.3.1.2), these estimates were made in 2015 when only limited data (Yang, et al., 2013 & Tran, 2014) on carbon storage by mangrove habitats in New Zealand and Auckland were available.

The most recent empirical data reported by Bulmer, et al., (2016), shows that mangrove habitats at three estuaries in Auckland (Whangateau, Bayswater, Mangere) stored carbon in their sediment (to 100 cm depth).
depth) at an average value of 89.3 t C ha\(^{-1}\). The selected mangrove forests also stored approximately 34 t C ha\(^{-1}\) in their above- and below-ground biomass (Bulmer, et al., 2016).

The sediment carbon estimate by Bulmer, et al., (2016) is slightly less but similar to the average value estimated in this research. It is also less than the average value of carbon storage by mangroves at Pollen Island, Auckland (103 t C ha\(^{-1}\)), as reported by Yang, et al., (2013). Variation in sediment carbon storage across mangrove sites in Auckland is argued to be associated with the differences in forest characteristics in terms of height, density, below- and above-ground biomass and distance from the seaward edge across the sites (Bulmer, et al., 2016).

According to the empirical information provided by Bulmer, et al., (2016) and Yang, et al., (2013), the average value of carbon storage in the sediment of Auckland’s mangrove forests is approximately 97 t C ha\(^{-1}\). Considering these estimates, mangrove forests in the Auckland region with an area of 8,928 ha have potentially stored 860,000 tonnes of carbon in their sediments, which is equal to three million tonnes of carbon dioxide. Together with saltmarsh ecosystems, their carbon storage potential is around 1.5 million tonnes equal to 5.5 million tonnes of CO\(_2\) (Table 5.6-2). These ecosystems can also accumulate carbon in their sediment at the rate of 17,000 tonnes of carbon per year which is equivalent to 62,000 tonnes of carbon dioxide per year. Auckland’s CO\(_2\) emission per person is estimated at about 7 t CO\(_2\)e in 2014 (Auckland Council, 2014). The CO\(_2\) equivalent of the annual carbon accumulation in mangrove and saltmarsh sediments can therefore offset emissions from 8,700 people.

Considering a range between NZ$ 19 to NZ$ 40 per tonne of CO\(_2\), as the estimates of the cost of GHG emissions (New Zealand Transport Agency, 2016), the value of the carbon stored only in the sediment (to 100 cm depth) of mangrove and saltmarsh ecosystems is estimated between NZ$ 104 and NZ$ 220 million. It is important to note that these values (i.e. a range between 19 to 40 NZ$ per tonne of CO\(_2\)), do not

### Table 5.6-1. Sediment CS&S for mangrove (Avicennia marina) and saltmarsh habitats (Spartina alterniflora, Spartina anglica, Sarcocornia quinqueflora, Juncus kraussii) across temperate climate (Cfa/Cfb), (This research)

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Sediment carbon stock (t C ha(^{-1}))</th>
<th>Sediment carbon stock (t CO(_2)e ha(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean ± SE</td>
<td>Range</td>
</tr>
<tr>
<td><strong>Mangroves</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Avicennia marina</em></td>
<td>124 ± 20</td>
<td>104 - 144</td>
</tr>
<tr>
<td><strong>Saltmarshes (Total):</strong></td>
<td>307 ± 31</td>
<td>276 - 338</td>
</tr>
<tr>
<td><em>Spartina alterniflora</em></td>
<td>259 ± 32</td>
<td>227 - 291</td>
</tr>
<tr>
<td><em>Spartina anglica</em></td>
<td>327 ± 51</td>
<td>276 - 378</td>
</tr>
<tr>
<td><em>Sarcocornia quinqueflora</em></td>
<td>442 ± 124</td>
<td>318 - 566</td>
</tr>
<tr>
<td><em>Juncus kraussii</em></td>
<td>327 ± 16</td>
<td>311 -343</td>
</tr>
<tr>
<td><strong>Carbon sequestration in sediment (t C ha(^{-1})yr(^{-1}))</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Mangroves</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Avicennia marina</em></td>
<td>1.4 ± 0.3</td>
<td>1 – 1.7</td>
</tr>
<tr>
<td><strong>Saltmarshes (Total):</strong></td>
<td>2.3 ± 0.4</td>
<td>1.9 – 2.7</td>
</tr>
<tr>
<td><em>Spartina alterniflora</em></td>
<td>2.1 ± 0.6</td>
<td>1.5 – 2.7</td>
</tr>
<tr>
<td><em>Spartina anglica</em></td>
<td>4.2 ± 1.2</td>
<td>3 – 5.4</td>
</tr>
<tr>
<td><em>Sarcocornia quinqueflora</em></td>
<td>1.2 ± 0.3</td>
<td>0.9 – 1.5</td>
</tr>
<tr>
<td><em>Juncus kraussii</em></td>
<td>2.1</td>
<td></td>
</tr>
</tbody>
</table>

Note: the figures are rounded.
meaningfully represent the social cost of carbon (SCC) (See Section 5.2.1), rather they are dependent on the supply of emission trading units internationally (Chapman, 2015).

If a SCC of US$ 220 per tonne CO₂ (Moore & Diaz, 2015) is considered, the carbon storage value of Auckland’s mangrove forests\(^{36}\) (\(\approx 131 \text{ t C ha}^{-1}\) equal to \(\approx 480 \text{ t CO}_2\text{e ha}^{-1}\)) is worth about US$ 100,000 per hectare.

<table>
<thead>
<tr>
<th>Coastal wetland</th>
<th>Sediment carbon storage (t C ha(^{-1})) (Ave.)</th>
<th>Area in Auckland in 2012 (ha)</th>
<th>Sediment carbon storage Tonne carbon Tonne CO₂e</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves</td>
<td>97</td>
<td>8,928</td>
<td>866,016 3,175,681</td>
</tr>
<tr>
<td>Saltmarshes</td>
<td>307</td>
<td>2,056</td>
<td>631,192 2,314,581</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td>10,984</td>
<td>1,497,208 5,490,262</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Carbon sequestration in sediment t C ha(^{-1}) yr(^{-1}) (Ave.)</th>
<th>Area in Auckland in 2012 (ha)</th>
<th>Carbon sequestration in sediment t C yr(^{-1}) (Ave.) t CO₂e yr(^{-1}) (Ave.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves</td>
<td>1.4 5</td>
<td>8,928</td>
</tr>
<tr>
<td>Saltmarshes</td>
<td>2.3 8.4</td>
<td>2,056</td>
</tr>
<tr>
<td>Total</td>
<td>10,984</td>
<td>16,960</td>
</tr>
</tbody>
</table>

Note: the figures are rounded.

### 5.6.2. Net carbon sequestration

Methane emission from coastal saline wetlands is very low compared to freshwater wetlands (IPCC, 2000a), mainly because methanogenic bacteria are inhibited by salinity (Poffenbarger, et al., 2011). However, studies that quantified methane emission from temperate coastal wetlands are still scarce. Livesley and Andrusiak (2012) concluded that temperate mangrove and saltmarsh ecosystems in Victoria, Australia, have emitted little methane and nitrous oxide (N\(_2\)O), but stored large amounts of carbon in their sediment. Studies suggest that mangrove and saltmarsh ecosystems can annually emit about 1.85 t CO₂e ha\(^{-1}\) (Murray, et al., 2011) and 0.4 t CO₂e ha\(^{-1}\) (Poffenbarger, et al., 2011) of methane, respectively. Considering methane emissions from mangrove and saltmarsh ecosystems, their potential for the net carbon sequestration (sequestration less methane emissions) is estimated in this research at 44,000 tonnes per year (12,000 tonnes of carbon per year, 67% of their gross sediment sequestration per year) (Table 5.6-3).

Considering a range between NZ$ 19 to NZ$ 40 per tonne of CO₂, (as explained above), the value of the net carbon annually sequestered by these ecosystems ranges between NZ$ 830,000 and NZ$ 1.7 million. This estimate significantly increases (US$ 9.6 million/yr\(^{-1}\) \(\approx\) NZ$ 13.6 million/yr\(^{-1}\)), if the SCC value of US$ 220 (equal to around 310 NZ$ per tonne), as estimated by Moore and Diaz (2015), is used.

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\(^{36}\)This value includes both biomass and sediment carbon storage as reported by Yang, et al., (2013) and Bulmer, et al., (2016) for mangrove forests in the Auckland region.
Table 5.6-3. Methane emission and net CO\(_2\) removal for mangrove and saltmarsh habitats

<table>
<thead>
<tr>
<th></th>
<th>Sediment carbon sequestration (t CO(_2)/ha yr(^{-1})) (Mean)</th>
<th>Methane emission (t CO(_2)/ha yr(^{-1})) (Mean)</th>
<th>Area (ha)</th>
<th>Net CO(_2) removal (t CO(_2)/ha yr(^{-1})) (Mean)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangrove</td>
<td>5</td>
<td>1.85</td>
<td>8,928</td>
<td>28,123</td>
</tr>
<tr>
<td>Saltmarsh</td>
<td>8</td>
<td>0.4</td>
<td>2,056</td>
<td>15,626</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>13</strong></td>
<td><strong>2.25</strong></td>
<td><strong>10,984</strong></td>
<td><strong>43,749</strong></td>
</tr>
</tbody>
</table>

5.6.3. CO\(_2\) emissions due to modification of coastal wetlands

Studies that examined CO\(_2\) emissions due to coastal wetland degradation are scarce both globally (Macklin, et al., 2014) and in New Zealand. By measuring sediment CO\(_2\) efflux across cleared and intact mangrove areas in northern New Zealand, Bulmer, et al., (2015) concluded that regardless of an immediate carbon loss due to mangrove clearance, the majority of sediment carbon will be released over a number of years to decades once mangroves are removed.

This thesis used the approach suggested by Pendleton, et al., (2012) to estimate the approximate CO\(_2\) emission as a result of coastal wetlands degradation in Auckland. This approach focuses on ‘near-surface’ carbon which refers to the carbon within vegetation biomass and the top metre of sediment. This is partly because these carbon pools are more susceptible to land use change and also because carbon losses from deep sediments (>1 m) are not well understood and information in this field is still scarce (Pendleton, et al., 2012).

Information about the aerial extent of mangrove and saltmarsh habitats in the Auckland region, their approximate degradation rate (% of area lost per year) and their sediment and biomass carbon stock was used to estimate the amount of potential CO\(_2\) emission due to degradation of these ecosystems. Emissions were estimated as 25% and 100% of the total amount of the stored carbon. The high end of 100% would apply to high impact disturbance activities which convert coastal wetlands to a qualitatively different land use that prevents recovery of carbon. The low end of 25% would apply if the new land use allows redistributing of near-surface carbon (Pendleton, et al., 2012).

The annual degradation rates of 0.15% and 1% (as explained in Section 5.3.3) were used for mangrove and saltmarsh habitats, respectively. Considering the total aerial extent of these ecosystems in the Auckland region, this implies degradation of 13 hectares mangrove forests and 20 hectares saltmarsh wetlands per year which can potentially release between 8,000 and 33,000 tonnes of CO\(_2\) to the atmosphere, with an average estimate of 20,000 (t CO\(_2\)) (Table 5.6-4).

This is greater than the CO\(_2\) emission from mineral soil change in the first year of conversion of 2,000 hectares of vegetated wetland to perennial cropland (17,462 t CO\(_2\)/yr\(^{-1}\)) as estimated using the following equation adopted from the New Zealand Greenhouse Gas Inventory 1990–2013 (Ministry for the Environment, 2016):

\[
\text{Emissions from mineral soil changes} = \frac{[(\text{Mineral soil carbon at steady state in the new land use} - \text{Mineral soil carbon at steady state in the previous land use)}/ 20 \text{ years (transition period)}] \times \text{Activity data (Area)}}
\]
Table 5.6-4. Estimates of carbon emission from degradation and disturbance of Auckland’s mangrove and saltmarsh habitats

<table>
<thead>
<tr>
<th></th>
<th>Area (ha)</th>
<th>Degradation rate (% yr⁻¹)</th>
<th>Carbon stock (t CO₂e ha⁻¹)</th>
<th>Near-surface carbon (top metre sediment + biomass) susceptible (t CO₂e ha⁻¹) 25% - 100% disturbance range (mean)</th>
<th>CO₂e emission (t CO₂e yr⁻¹) range (mean)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Living biomass</td>
<td>Sediment</td>
<td>481</td>
</tr>
<tr>
<td>Mangrove</td>
<td>8,928</td>
<td>0.15</td>
<td>(Bulmer, et al., 2016)</td>
<td>356</td>
<td>120 – 481</td>
</tr>
<tr>
<td>Saltmarsh</td>
<td>2,056</td>
<td>1</td>
<td>(Lawrence, et al., 2012)</td>
<td>52</td>
<td>294 – 1,177</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td></td>
<td>(This study)</td>
<td>1,177</td>
<td>414 – 1,658</td>
</tr>
</tbody>
</table>

Note: the figures are rounded.

Considering a range between NZ$ 19 and NZ$ 40, (as explained earlier), emission of 20,000 tonnes of CO₂ will approximately cost NZ$ 390,000 to $820,000 per year.

This approach to estimate CO₂ emission due to conversion of coastal wetlands is a conservative approach. First, because it only considers disturbance of the first top metre of sediment; while it is identified that deep sediments (>1 m depth) contain high level of carbon and may affected by land use change activities in coastal areas (Pendleton, et al., 2012). This means that even the high-end scenario (i.e. 100% carbon loss) provides an estimate that could in reality be less than the actual carbon loss (Pendleton, et al., 2012). Second, the low-end scenario (i.e. 25% carbon loss) assumes that land use change activities retain 75% of carbon which is extremely conservative. Third, these estimates only indicate the loss of carbon which is stored in sediments and biomass of mangrove and saltmarsh habitats and is lost when these ecosystems are disturbed (biomass and soils are disturbed) (Pendleton, et al., 2012). There is also loss of additional carbon as a result of biomass and sediment carbon sequestration that is lost when coastal vegetation is removed.

In addition, removal of mangrove biomass will expose the sediment carbon to oxygen and thus the stored carbon will be oxidised and gradually emitted in the form of CO₂. The timing of CO₂ release from sediment post-clearance of mangrove biomass is not clear yet, but as mentioned earlier mangrove removal will modify (decrease) the sediment carbon over a long-term run (Bulmer, et al., 2015). Estimates of CO₂ emission also depend on the method for mangrove removal. Possible use of fossil fuel powered machineries might also be considered as an additional source of emission, which is not included in the estimates of emission in this research. There is no clear indication of emission implications of mangrove removal activities, which may over time, offset the total carbon sequestered and stored in coastal wetland ecosystems.

It can be argued that CO₂ emission as a result of mangrove removal in Auckland has been most likely offset by mangrove expansion. However, the net benefits remain uncertain unless data on the net changes in mangrove areas becomes available. Based on the information from the LCDB.v.4, this research estimated the 1996 and 2012 aerial extent of mangroves at 8,917 ha and 8,928 ha, respectively. These figures suggest that mangrove forests in Auckland have been a net sink of carbon during 1996 and 2012.

5.6.4. Limitation of the approach and data sources

Using global data for local estimates of CS&S of coastal wetlands is uncertain due to high variations associated with differences in factors that affect CS&S in coastal wetlands. To reduce uncertainty, this study only used data reported for coastal wetlands with similar conditions (climate and halophyte species)
to coastal wetlands in Auckland. However, due to limited data availability within the literature, some uncertainties still remain for these estimations. Equally, assuming uniform carbon densities to a depth of one metre may result in over or under estimation of sediment carbon storage for saltmarsh studies that reported carbon density for less than one-metre depth of sediment. The current estimates only represent the approximate potential of mangrove and saltmarsh habitats for CS&S in their sediment and therefore do not reflect the overall blue carbon capacity of Auckland’s coastal wetlands. Incorporating data on the rate of biomass CS&S as well as data for CS&S in seagrass ecosystems will increase the current estimates.

5.6.5. Carbon storage of coastal wetlands and terrestrial ecosystems

Terrestrial ecosystems in New Zealand can potentially store between 279 and 693 tonnes of carbon (CO$_2$ equivalent) in the top metre of a typical hectare of their soils (Table 5.6-5). On a per hectare basis, it is lower than the estimates of carbon storage within the sediment of coastal mangrove and saltmarsh habitats (1,481 t CO$_2$e ha$^{-1}$). While there is limited information on the rate of carbon sequestration within the soils of terrestrial ecosystems, the available data shows the annual rate of carbon sequestration in sediment is greater in mangroves, saltmarshes and seagrasses (5-8 t CO$_2$e ha$^{-1}$ yr$^{-1}$, Table 5.6-2) than terrestrial forests (0.15-0.19 t CO$_2$e ha$^{-1}$ yr$^{-1}$, Ouyang & Lee, 2014). It is reported that carbon accumulation in biomass and soil of terrestrial forest ecosystems either stops or occurs at low rate once it reaches a saturation point (Das & Mukherjee, 2015).

Carbon sequestration in coastal ecosystems is a function of sediment accretion rates. Therefore, the amount of sediment carbon within intertidal coastal ecosystems increases over time through increasing sediment surface elevation as a result of sediment and organic matter accretion. In addition, as noted before, carbon can stay trapped in the sediments of coastal vegetation over millennia, if undisturbed (Pendleton, et al., 2012). This characteristic of coastal wetlands particularly mangrove forests and saltmarsh wetlands gives them an advantage over terrestrial ecosystems in terms of their ability for a long-term carbon burial (Lawrence, et al., 2012).

Table 5.6-5. Average carbon storage in sediments and soils of temperate coastal and terrestrial ecosystems

<table>
<thead>
<tr>
<th>Land use category</th>
<th>Average carbon stock in top metre of soil/sediment (t CO$_2$e ha$^{-1}$)</th>
<th>Location</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperate Mangrove &amp; saltmarsh habitats</td>
<td>1481</td>
<td></td>
<td>Yang, et al., (2013); Bulmer, et al., (2016) and this study</td>
</tr>
<tr>
<td>Mixed Indigenous forest*</td>
<td>643.49</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Exotic forest*</td>
<td>488.09</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Unimproved pasture*</td>
<td>488.12</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Improved pasture*</td>
<td>532.68</td>
<td>New Zealand</td>
<td>Scott, et al., 2002</td>
</tr>
<tr>
<td>Broadleaf Indigenous Forest*</td>
<td>450.48</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cropland*</td>
<td>692.97</td>
<td>New Zealand</td>
<td>Coomes, et al., 2002</td>
</tr>
<tr>
<td>Scrub*</td>
<td>540.66</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Indigenous forest</td>
<td>279.06</td>
<td>New Zealand</td>
<td></td>
</tr>
<tr>
<td>Shrubland</td>
<td>367.43</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: Data for these land use categories is provided for moist-temperate climates with High Clay Activity (HCA) soil type which matches Auckland’s climate and dominant soil, based on the information provided by Scott et al (2012).
5.7. Coastal protection function of Auckland’s coastal wetlands

The low-lying coastal areas of the Auckland region are potentially exposed to erosion and inundation due to sea-level rise (SLR), storm tides and tsunami. Results of a risk assessment and analysis of the nationally significant hazards and risks in Auckland indicated a moderate risk for storm surge and tsunami (regional/local) and a very high risk for cyclone and coastal erosion (Auckland Civil Defence, 2011). Sea level rise increases inundation caused by king tides, storm surges and waves (Parliamentary Commissioner for the Environment, 2015). In New Zealand, the sea level is projected to increase by about 30cm between 2015 and 2065 and thus it is predicted that coastal communities living in low-lying coastal lands will likely experience more frequent and severe flooding (Parliamentary Commissioner for the Environment, 2015).

The frequency and magnitude of storm events are also expected to increase as a result of sea level rise. For example, projections indicate that a 100-year storm event (i.e. a storm event which occurs once every hundred years) will occur every four years in Auckland if sea level rises by 30cm (Figure 5.7-1) (Parliamentary Commissioner for the Environment, 2015). With sea level rise, the area of inundation as the combined effects of tide, storm surge and waves will also increase and reach further inland (Rouse, et al., 2017).

It is argued that sea level rise will continue to increase beyond 2100 even if GHG emissions are reduced; thus, a key change for New Zealand is to adapt to the increasing risk of climate change to coastal areas (NZCCC, 2014; Rouse, et al., 2017). To guide and assist local governments in planning for climate change adaptation, the Ministry for the Environment (MfE) has been mainly involved in developing guidance and information documents which can be used to identify, assess and manage the climate risks in New Zealand’s coastal areas. This is further discussed in Chapter 6, Section 6.4.1.

Figure 5.7-1. Changes in the frequency of exceedance of 100-year storms due to changes in sea level rise at four ports in New Zealand (Parliamentary Commissioner for the Environment, 2015)

There is evidence which indicate the most flood prone sites are areas that were historically occupied by wetlands. An example is an area in Bay of Plenty where the land inundation following a flood event (2004) occurred in an area which had been previously covered by coastal wetlands (Figure 5.7-2) (Clarkson, et al., 2013). The Auckland Council has prepared coastal inundation maps for the Auckland region that
identify areas that will be inundated by flood events with different annual return rates (ARI) of 5, 20, 50, and 100 years (or with the probability of 20%, 5%, 2% and 1%). In addition, the coastal areas at risk of inundation and flooding with annual return intervals of 50 and 100 years identify inundation as a result of sea level rise by 1 and 2 metres. These maps are available to the public through the Auckland Council GIS viewer. Tsunami evacuation zones are also prepared for all regions across the country (including the Auckland region) by the Ministry of Civil Defence and Emergency Management and are available on the websites of the local civil defence groups (Figure 5.7-3).

Figure 5.7-2. The extent of land inundation following 2004 flooding in Bay of Plenty and the historic wetland area (Clarkson, et al., 2013, from Gerbeaux)

Figure 5.7-3. Examples of coastal inundation (left: dark blue shows) and tsunami evacuation (right) maps in Devonport, Auckland

Dark blue: Area that will be inundated as a result of 1 in 100-year storm event plus 1m SLR; Light blue: Area that will be inundated as a result of 1 in 100-year storm event plus 2m SLR; Red zone: The highest risk zone; Orange zone: Area that is likely to be evacuated during official warnings and (iii) Yellow zone: Area which needs to be evacuated in the large tsunami events. Source: Auckland Council GIS Viewer.

Despite vulnerability of the coastal areas to sea level rise, coastal erosion and tsunami, development still occurs in Auckland’s coastal areas (Figure 5.7-4). Coastal communities at the Omaha beach and Whangateau Harbour are at risk of storm inundation as the King tide event of 200mm on 22 Feb 2015, flooded the footpath of the waterfront properties (Figure 5.7-5). As Figure 5.7-6a, shows mangroves are the only buffer between the land and the sea at Whangateau Harbour and their removal may enhance exposure to coastal hazards. It is reported that there is considerable shore erosion in areas where mangroves have been removed and consequently, local owners have attempted to prevent further erosion by dumping rock and concrete along the coast (Figure 5.7-6b).

Figure 5.7-4. Development in Orewa, Auckland, within close proximity of the open water (Whangateau HarbourCare Group, 2014)

Figure 5.7-5. Land inundated following a King tide event (200mm) at Point Wells, Whangateau Harbour (25 Feb 2015) (Whangateau HarbourCare Group, 2015a)
As mentioned before, there is growing evidence regarding the importance of coastal vegetation for protection of the coast from the effects of storm surges, sea level rise and erosion. However, much of the built-up areas most vulnerable to storm activity (i.e. too exposed and high energy environments) are the areas where coastal wetlands would not naturally occur or have a small/no chance to establish. Although, as mentioned by Bouma, et al., (2014), it is feasible to establish wetlands in exposed, harsh environments by using proper engineering measures. It is also indicated that natural or well established coastal wetlands (mainly mangroves) adjusted to low energy environments with mild average hydrodynamic stress may protect coastal areas against rare extreme events, particularly if they have occupied wide intertidal areas (Bouma, et al., 2014).

There are polarised views about the ecological significance of New Zealand mangrove and its coastal protection benefits (Morrisey, et al., 2007; De Luca, 2015). The divergence in view is primarily around incomparability of New Zealand’s mangroves with their tropical counterparts in terms of their climate adaptation benefits (Morrisey, et al., 2007). A few studies have discussed the role of coastal wetlands (mainly mangroves) in protection from storm surges in New Zealand (Morrisey, et al., 2007; Swales, et al., 2007; Morrisey, et al., 2010). These studies, however, have not specifically focused on the potential of mangrove forests for coastal protection. Instead, this function of mangroves has been discussed in general together with other ecological and biological aspects of mangroves such as primary productivity and supporting aquatic and terrestrial fauna and flora. Morrisey, et al., (2010) have also briefly outlined the differences in wave attenuation between temperate and tropical mangrove forests.

As part of a comprehensive study on mangrove habitat expansion in the southern Firth of Thames (52 km southeast of Auckland), Swales et al. (2007) concluded that the wide and the firmly established mangrove forest in the Firth of Thames (approximately 800-m wide, 1-4 metres height) can provide a substantial natural defence against tsunamis, wind waves and swell generated by storms (Swales, et al., 2007). The coastal protection role of mangroves in this study is determined by examining the extent to which the mangrove forest can protect a stopbank from wave-induced coastal erosion and inundation due to sea-level rise and storm events. The study also found that mangroves in the Firth of Thames have minimal effect on attenuating locally-generated short-period (e.g., 5 to 15 minute) tsunami waves due to their shallow root system. However, they can still provide a buffer to the dynamic components of tsunami owing to the wide band of the forest (Swales, et al., 2007).

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A value of NZ$ 4,900 per hectare is estimated as the value of disturbance regulation by mangrove ecosystems in New Zealand (Patterson & Cole, 2013). Given the lack of specific New Zealand data, this value is estimated based on the information from global literature particularly the study by Costanza, et al., (1997). Excluding other services has been due to the lack of reliable local and inapplicable global data.

As discussed in Section 5.3.4, the wave attenuation function of mangrove forests is dependent on a wide range of variables of which plant species is one. This suggests that simply considering dominant mangrove species, mangroves in New Zealand are likely to provide less protection against storm surges than tropical mangrove forests dominated by more robust mangrove species with complex root structures such as Rhizophora spp. However, this comparison cannot provide a robust basis for the argument that New Zealand’s mangrove forests are generally less effective at mitigating storm surges compared to tropical mangrove forests.

Based on the literature, the wave attenuation function of coastal vegetation is strongly site-specific and physical parameters such as tidal range and the width of foreshore area (which also determine long-term persistence of coastal wetlands), together with a range of other factors, play a critical role in determining the storm protection capacity of coastal wetlands (Mclvor, et al., 2012a; Bouma, et al., 2014; Spalding, et al., 2014a). This suggests that mangrove forests cannot be compared because of the different levels of protection that they may provide under different storm conditions due to differences in site-specific biophysical variables.

There is, however, no doubt that coastal vegetation particularly mangrove forests and saltmarsh wetlands provide coastal defence functions. Studies provided examples of the wave attenuation function of Avicennia marina (e.g. Kathiresan & Rajendran, 2005; Badola and Hussain, 2005; Alongi, 2008; Quartel, et al., 2007; Bao, 2011).

As mentioned before (See section 5.3.2), mangrove and saltmarsh vegetation can enhance surface elevation through biophysical processes which include sediment deposition as a result of wave attenuation. This function of coastal wetlands enables them to naturally adapt to the long-term changes in sea level and therefore protect the coastline against sea level rise. However, long-term persistence and resilience of coastal wetlands primarily depends on whether the increase in surface elevation is at a rate equal to or exceeding the rate of sea level rise. It also strongly depends on the availability of foreshore space that enables intertidal habitats to go through natural processes such as inland migration, decay and recovery (Bouma, et al., 2014; Spalding, et al., 2014a).

As reported by Swales, et al., (2007), colonisation of mangroves in the southern Firth of Thames has accelerated sediment accretion, with average sediment accretion rate (SAR) ranging from 5 to 100mm yr\(^{-1}\) on the forest fringe. They concluded that average SAR over the last 50 years in the southern Firth of Thames has substantially outpaced the historic rate of sea-level rise (1.4 mm yr\(^{-1}\) since 1899). They also argued that the infrequent occurrence of tidal inundation was the driver behind the reduced SAR (7-12 mm yr\(^{-1}\)) in old-growth forest. A SAR of up to 14 mm yr\(^{-1}\) was reported for intact mangrove forests at Tauranga Harbour (North Island, New Zealand); while elevation loss rate ranging from 9 to 38 mm yr\(^{-1}\) was observed within areas where mangroves were cleared (Stokes, et al., 2010, Cited in Krauss, et al., 2014).

A recent study by Swales, et al., (2015) showed that the major mangrove seedling recruitment event at the Firth of Thames occurred more than a decade after transition to the high SAR phase. Based on these findings, they concluded that mangroves occupy tidal flats once they reach an ecologically suitable elevation. They argued that New Zealand mangroves are opportunistic species in that their development on tidal flats is driven by physical processes. This suggests that patterns and rates of mud-deposition are

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39 Examples of disturbance regulation service include ‘storm protection, flood control, drought recovery, and other aspects of habitat response to environmental variability mainly controlled by vegetation structure’ (Patterson & Cole, 2013, p.499).
mainly controlled by physical processes\textsuperscript{40}, irrespective of the presence of mangroves. Swales, et al., (2015) argued that this behaviour of mangroves can be associated with the characteristics of the estuary which is wave-dominated and exposed to high rate of mineral sedimentation. They also posited that mangroves (\textit{Avicennia marina}) prevent sediment resuspension but do not measurably contribute to enhancing sediment accretion over the annual to decadal timescales.

According to the findings of a study on the fate of mangrove forests in Indo-Pacific region (Lovelock, et al., 2015), subject to sediment availability, mangrove forests in New Zealand are unlikely to be submerged by the projected 0.48m and 0.63m increase in sea level by 2100, even if their landward migration is not possible. However, parts of the mangrove forests mostly located at the eastern estuaries will most likely be submerged between 2070 and 2090 under a more extreme scenario of sea level rise (1.4m by 2100) (Figure 5.7-7). These findings highlight the importance of the maintenance of sediment supply in estuaries and planning for landward migration of mangrove forests in areas where sediment supply is restricted. However, lack of sufficient inland space for coastal wetlands migration mainly due to intensive development\textsuperscript{41} as well as the lack of political support and interest for landward retreat of human settlements remain the key challenges (Lovelock et al. 2015). In areas where space for landward migration is vital to ensure long-term survival of coastal ecosystems, any decision in favour of future development or keeping the line of existing development implies loss of coastal ecosystems in the long run.

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{figure5.7-7.png}
\caption{Figure 5.7-7. Predicted decade of mangrove forest submergence (due to sea level rise, SLR) in New Zealand, indicated by the colour scale. The darkest blue areas indicate no submergence predicted by 2100 (Lovelock et al. 2015)}
\end{figure}

The most recent report by the Arctic Monitoring and Assessment Programme (AMAP, 2017) concludes that when all sources of sea level rise are considered (not just those from the Arctic), the rise in global sea level by 2100 would be at least 52 cm for a GHG reduction scenario\textsuperscript{42} (compared with 2006) and 74 cm

\textsuperscript{40} Tidal asymmetry, hydrodynamic resuspension and onshore tidal advection of mud; shoreward wave-energy dissipation due to shoaling and fluid mud concentrations; scour and settling lags and flocculation in shallow water depths at high slack tide.

\textsuperscript{41} This will result in wetland loss with sea level rise and is known as “coastal squeeze” (Bouma, et al., 2014) as mentioned in Section 5.3.2.

\textsuperscript{42} RCP4.5: In this scenario, reductions in emissions lead to stabilization of greenhouse gas concentrations (at approximately 650 ppm CO$_2$-equivalent) in the atmosphere by 2100 and a stabilized end-of-century global average temperature rise of 1.7–3.1$^\circ$C above pre-industrial levels (AMAP, 2017).
for a business-as-usual scenario. These estimates are slightly higher than the IPCC estimates which were used in the study by Lovelock, et al., (2015). It is therefore important that the long-term sustainability of coastal wetlands in Auckland is assessed against higher rates of sea level rise under different climate change emission reduction scenarios as suggested by Lawrence, et al., (2013).

Overall, based on the findings of the global literature review, a more precise knowledge of the coastal protection function of Auckland’s mangrove and saltmarsh ecosystems requires site-specific measurements and modelling that incorporates a wide range of variables involved in wave attenuation. However, it is likely that mangrove and saltmarsh ecosystems in the Auckland region can provide effective protection against storm events in areas where a number of minimum conditions as discussed in Section 5.3.4 and characterised in Figure 5.3-7 are evident.

Given the findings of the global literature, it is unlikely that coastal mangrove and saltmarsh habitats provide protection against tsunamis unless they occupy a wide area on the intertidal platform (with width more than 1km). Examples are extensive mangrove ecosystems at Tauhoa, Auckland (Figure 5.7-8). Moreover, Auckland’s mangrove and saltmarsh ecosystems play a very important role in protecting the coastline against erosion and sea level rise. Management and monitoring of surface elevation in estuaries is also of critical importance as the balance between sea level rise and surface elevation determines the long-term sustainability of mangrove and saltmarsh habitats.

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**5.8. Summary and conclusion**

Using the available local and global data, this study estimates that Auckland’s coastal mangrove and saltmarsh habitats can approximately store 1.5 million tonnes of carbon in the first metre of their sediment and can annually accumulate carbon in their sediment at the rate of 44,000 tonnes CO$_2$e (corrected for methane emissions). The monetary value of the carbon stored in the sediment of mangrove and saltmarsh ecosystems was estimated at approximately NZS 220 million. The value of carbon sequestration is also about 1.7 million per year. These estimates do not include the value of other ecosystem services provided by mangrove and saltmarsh ecosystems. On a per hectare basis, Auckland’s coastal mangrove and saltmarsh habitats appear to be able to sequester more carbon in their sediments than terrestrial forests.

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43 RCP8.5: RCP-8.5 is a high-emission business-as-usual scenario, leading to a global non-stabilised (1370 ppm CO$_2$-equivalent) temperature rise of 3.8–6°C by 2100 (AMAP, 2017).
This suggests that even with the smaller above ground biomass and aerial extent of coastal wetlands in Auckland, these ecosystems have the potential to contribute substantially to long-term carbon storage.

The estimates provided in this study are based on the best information and data available in the peer-reviewed local and global publications. However, these estimates are still limited by the quantity and variability of available data, particularly data on CS&S by saltmarsh species including *Sarcocornia quinqueflora* and *Juncus kraussii*. In addition, due to the reasons stated in Section 2.6.4, the calculations in this research underestimate the potential of Auckland’s coastal wetlands for carbon sequestration and storage.

Given a wide range of parameters affecting the wave attenuation function of coastal wetlands, estimating the true value of the protective function of Auckland’s coastal wetlands requires site-specific modelling and measurement. Lack of such reliable data makes it difficult to compare the true protective values of coastal wetlands with other protection options such as groynes or seawalls. However, as studies concluded (See Section 5.3.3 for more information), coastal wetlands are generally cost-effective options compared to engineered structures. Moreover, restoration of coastal wetlands for storm protection can also offer a number of other co-benefits (such as CS&S) which need to be considered in comparison between natural and engineered options in terms of their long-term costs and benefits.

It should be also noted that areas where wetlands are easiest to protect and enhance (mostly low energy, less exposed coasts), or where this may be more acceptable by local communities, are not necessarily those areas likely to benefit the most from their protection abilities. However, these coastal wetlands, if not disturbed, still provide effective protection against erosion, sea level rise and rare extreme events.

Considering the findings from the extensive review of literature in this chapter, this research suggests that while the climate change benefits of Auckland’s mangrove forests and saltmarsh wetlands are positive, the scale and the true value of these benefits is uncertain. The discussions in this chapter demonstrated the principles and highlighted the paucity of data. While quantifying the true CS&S values of Auckland’s mangrove and saltmarsh habitats is difficult, it is possible to attach an approximate value when making decisions about land use activities affecting coastal wetlands, by characterising the types of coastal wetland around a coast, and applying a social cost of carbon (SCC).

Equally, there is sufficient evidence of positive benefits to promote land use activities that protect and enhance mangroves. However, factors determining long-term resilience and persistence of mangrove forests need to be taken into account when making decisions to incorporate mangroves in coastal protection.
6. Coastal resource management and response to climate change in New Zealand and Auckland

6.1. Introduction

This chapter addresses the following question:

- Whether and how the climate change services of coastal wetlands are taken into account in current coastal resource management and climate change response policies and plans in New Zealand and Auckland, and what are the key challenges and opportunities?

In order to answer this question, this chapter provides findings of a comprehensive review and analysis of the relevant statutory and non-statutory planning and policy documents in New Zealand and Auckland. As mentioned in Chapter 3 (Section 3.4.1), the focus is on national and regional policies and strategies that regulate the management and protection of coastal resources including wetlands. The review also includes documents that address protection and management of natural resources particularly marine environment, responses to climate change and the management of natural hazards.

The review identifies and discusses a number of key issues and opportunities regarding incorporation of the values of coastal wetlands into the mainstream policy and planning processes.

Findings of this chapter feed into the next chapter which presents findings of an extensive review of the climate change response policies and initiatives in a number of overseas jurisdictions. It informs a discussion on disparities or similarities between New Zealand (the case of Auckland) and global cases in terms of management of coastal wetlands for climate change mitigation and adaptation and approaches to improve resilience to the impacts of climate change.

The methods applied to the analyses in this chapter include document analysis (content analysis of policy documents) and questionnaire survey explained in Chapter 3 (Sections 3.4.1 and 3.4.2).

6.2. Governance and legislative framework

New Zealand is a unitary system with a central government and 78 local authorities that consist of 11 regional councils (the top tier of local government) and 67 territorial authorities (the second tier of local government). Territorial authorities in New Zealand include 12 City Councils, 54 District Councils, and the Auckland Council (Department of Internal Affairs, 2016). There are six territorial authorities in New Zealand which have the powers of regional councils and are referred to as unitary authorities (Department of Internal Affairs, 2016). They include the Auckland Council, the Nelson City Council, the Gisborne, Tasman, and Marlborough district councils and the Chatham Islands Council (Department of Internal Affairs, 2016).

Central government in New Zealand has a broad political, legislative and financial power, and has a key role in provision of social services and delivering on national outcomes (Department of the Prime Minister and Cabinet, 2008). At the local and regional level, local governments exercise their powers conferred to them under various statutes to serve their communities within their jurisdictions. While the local governments in New Zealand collect about 60 percent of their revenue from local taxes (i.e. rates), they only contribute to about 10 percent of the total national public expenditure and less than 10 percent of the total public revenue (Local Government New Zealand, 2015).

Local governments in New Zealand are generally reliant on central government’s policy directions and guidance and to some extent its financial support to be able to deliver on their responsibilities, especially with respect to provision of infrastructure, which on average accounts for about half of the total operating expenditures of local councils (Local Government New Zealand, 2015). Like many other unitary systems, the New Zealand’s local authorities are also constrained with their limited control over their financial
policies which are largely influenced by the central government’s both fiscal and monetary policies (Local Government New Zealand, 2015; NZ Institute of Economic Research, 2012). This means that local governments need to continuously adjust their policies and plans to be able to keep pace with and respond to the dynamics of the market driven by the central government policies.

Since the release of an independent inquiry into regulatory performance in local government by New Zealand’s Productivity Commission in 2013, there have been debates about the lack of proper alignment between central government and local government through the drafting and enactment of new regulations that create mandates for local governments (NZ Productivity Commission, 2013).

Despite this broad reliance on a central system of governance, local government in New Zealand is the primary means of managing environmental issues. That is, regional councils are responsible for management of biophysical resources including freshwater, soil, air and the coastal marine area across their region (along with other tasks such as transport). While regulating land use activities including residential development and infrastructure as well as other service delivery (e.g. drainage, water, sewage) are the responsibility of the territorial authorities (i.e. city and district councils) (Harker, et al., 2016).

The Resource Management Act (RMA) 1991, is the primary legislation that seeks to promote the sustainable management of New Zealand’s natural and physical resources (s5). RMA defines the governance structure for environmental and resource management. Under the RMA (s30&31), local governments are required to develop resource management plans that identify local resource management issues, and set out objectives, policies and methods (including rules) to address those issues within the context of the overall purpose of the RMA, which is the sustainable management of natural and physical resources.

The key role of central government is to provide statutory directions to local authorities with respect to environmental matters of national importance as defined in the RMA (s6&7). In doing so, central government under the RMA (s43&45) may choose to prepare National Policy Statements (NPS) and National Environmental Standards (NES). These are the key instruments that are intended to enable the central government to coordinate and steer individual activities of the local governments in exercising their power to manage their natural and physical resources (Ministry for the Environment, 2012).

Under the RMA (s44&55) local authorities are required to give effect to NESs and NPSs and align their local outcomes, objectives and policies with the national outcomes when preparing and reviewing their statutory plans. In fact, the RMA establishes a four-tier administrative regime including national policy statements, regional policy statements, regional plans and district plans. This ensures that a consistent approach balances the local and national aspirations and needs and enables an integrated management of resources at national level (Ministry for the Environment, 2012). The RMA requires regional councils to prepare and implement mandatory Regional Policy Statements (RPSs) (s60(1)), mandatory Regional Coastal Plans (RCPs) (s64) and optional Regional Plans (s65) (Figure 6.2-1).

The RMA (s57) requires at least one Coastal Policy Statement (CPS) to exist at all times and must set out policies in relation to the sustainable management of coastal environment of New Zealand. The purpose of the New Zealand Coastal Policy Statement (NZCPS) as indicated in the RMA (s56) is to state policies in order to achieve the purpose of the RMA in relation to the coastal environment of New Zealand. Amongst the five national policy statements in effect, the NZCPS is the only one which is mandatory under the RMA (Department of Conservation, 2010a). The current NZCPS 2010 requires a strategic and integrated approach for planning and management of natural and physical resources in the coastal environment (Ministry for the Environment, 2012). Development of a national coastal policy statement and mandates for its review is argued to be a good attempt towards ICZM (See Section 2.8) which places an emphasis on evaluation and adoption of changes (Peart, 2007). In addition, the mandatory requirement

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44 These include the National Policy Statement for Freshwater Management, the National Policy Statement for Renewable Electricity Generation, the National Policy Statement on Electricity Transmission, the New Zealand Coastal Policy Statement and the National Policy Statement on Urban Development Capacity.
that the hierarchy of coastal policy plans are consistent with and must give effects to the provisions of the NZCPS also provides opportunities for integrated, rather than sectoral management of the coastal environment and is consistent with the integration principles of ICZM (Heemskerk, et al., 2001; Bremer & Glavovic, 2013). The NZCPS also provides for sustainable management of the coastal environment and adoption of a precautionary approach to use and management of coastal resources. This gives regard to the principles of ICZM as mentioned in Section 2.8.

Given to the establishment of a strong administrative structure supported by a court structure, the RMA is argued to be a good example of a national resource management legislation which supports the approach of ICZM (Heemskerk, et al., 2001; Bremer & Glavovic, 2013); however, it has never been formally described as an exercise in integrated coastal management (Bremer & Glavovic, 2013).

Figure 6.2-1. Environmental governance under the RMA (Ministry for the Environment, 2008b, p.61)

Local authorities in New Zealand operate under the Local Government Act (LGA). The LGA is a statutory framework, enacted in 2002, that sets out the purpose, structure, functions, powers and accountabilities of local governments in New Zealand (Kessaram, 2013). Under the LGA (s12&13), local authorities have full rights, powers and privileges to carry on or undertake any activity or business towards the purpose of
a local government, as long as those activities comply with the provisions set out in the LGA and other enactments, and the general law. The LGA (s14(1)(h)) enables local authorities to contribute to community interests, taking a ‘sustainable development’ approach. This partly overlaps with the purpose of the RMA (s5), which promotes ‘sustainable management’ of resources towards providing for social, economic and cultural wellbeing of the current generation while meeting the foreseeable needs of the future generation. This recognition of ‘sustainability’ as the steering purpose of local government is also consistent with ICZM which provides for sustainable management of coastal resources.

Under the LGA, all local and regional councils are required to develop a long-term plan (LTP) that sets out the local governments community outcomes and outlines a strategic direction to achieve those outcomes. While overarching in its scope, the LTP in itself does not override any of the other legislative mandates. To achieve environmental outcomes, local governments need to align their objectives in LTP with the objectives set out in their statutory and non-statutory policy statements and plans under the RMA.

On 1 November 2010, the Auckland region formed a unitary authority controlled by the Auckland Council, replacing the previous eight local government entities (One regional council and seven district councils). The current Auckland Council consists of the governing body (Mayor and 20 Councillors) and 21 local boards, which represent the interests of local communities (Shirley, et al., 2016). There are also six council controlled organisations (CCO) and a new transport agency to support the Auckland Council (Shirley, et al., 2016).

The Auckland Plan which is a requirement under the Local Government (Auckland Council) Amendment Act 2010 (LGACAA 2010, s79) was prepared and released by the newly formed Auckland Council in September 2011 and formally adopted by the Council in March 2012 (Auckland Council, 2012a). The Auckland Plan is a 30-year spatial plan which sets out a high level strategic direction for the Auckland region particularly in relation to transport, land use, housing and economic development (Auckland Council, 2012a).

The Auckland Plan has been subject to scrutiny especially with regard to its role after its approval and a lack of legislative support under the RMA regime (Beattie, 2011). The Auckland Plan was initially considered as a non-statutory plan as the RMA had no requirements for plan makers to give effect to the plan (Beattie, 2011). It has been argued that this failure to link the Auckland plan to the RMA (1991) through statute, could affect the plan’s efficiency to influence the strategic direction of Auckland growth management and infrastructure provision (Beattie, 2011).

The Auckland Unitary Plan (AUP) is the main regulatory instrument to implement the strategic directions of the Auckland Plan. The AUP has involved an extensive process of public consultation after its first draft (i.e. the Proposed Auckland Unitary Plan or PAUP) was released in March 2013 (Auckland Council, 2016a). The AUP Operative in part, was notified by the council on 15 November 2016 and identified the parts to be treated as operative under the RMA (s86F) and the parts that would be subject to appeal (Auckland Council, 2016a).

The AUP replaces the existing Regional Policy Statement (RPS) and 13 district and regional plans including Auckland Regional Coastal Plan. The RPS is at the top of the AUP hierarchical policy framework and provides an overview of the resource management issues of the region and policies and methods (regulatory and non-regulatory) to achieve the purpose of the RMA in integrated management of natural and physical resources of the whole region. The regional and district plan provisions including

45 Carruthers, B., A weighting game: What statutory weight will, s.l.: Russell McVeagh.
46 The AUP provisions on mangrove management in SEA-M (Chapter D, D9) and in coastal marine area outside identified SEA (Chapter F, F2) were still under appeal at the time of updating and finalising this section (November 2016 to January 2017).
47 The exception is the Hauraki Gulf Islands section of the Auckland Council District Plan which is not replaced with the AUP. However, the Hauraki Gulf Islands are regulated by the provisions under the AUP regional policy statement, the regional coastal plan and the regional plan.
objectives, policies and rules give effect to the RPS (Auckland Council, 2016b). The relationship of the AUP to the RMA and other policy statements and plans is demonstrated in Figure 6.2-2.

The following sections present findings of an extensive review of the New Zealand and Auckland’s policy documents (including statutes, non-statutory plans and guidelines) that are relevant to management of the coastal environment, including wetlands, and relate to climate change, coastal hazards and integrated catchment management as well as the roles and responsibilities of the related authorities involved in making and implementing those statutes and policies.

Figure 6.2-2. Relationship of the Unitary Plan to other policy statements and plans (PAUP, Chapter A, 2.2.3) (Auckland Council, 2016b)

6.3. Management of coastal resources including coastal wetlands

An outstanding feature of the current legislative and planning system in New Zealand is its strong recognition of the importance of protecting the coastal environment at a strategic level particularly by the RMA (1991), the NZCPS (2010) and the Fisheries Act (1996). Legislative mandates for management of the coastal environment are primarily set out in the RMA. The extent of the coastal management jurisdictions is illustrated in Figure 6.3-1 (Department of Conservation, 2010b).

Under section 6 of the RMA, preservation of the natural character of the coastal environment and wetlands is a matter of national significance. Although the RMA is not specific about what characteristics make a coastal environment a matter of national significance or what constitutes an outstanding natural landscape, it sets out directions for the NZCPS and other national policies and strategies that provides for further clarifications on the matters of national significance. This strategic level recognition of the significance of the coastal environment is based on an approach that principally focuses on protection of indigenous biodiversity, natural character, public access to the coast, and both recreational and cultural values of the coastal environment, while sustaining fisheries habitat.
6.3.1. Definitions

Although the term ‘coastal environment’ appears under several RMA provisions, there is no definition for ‘coastal environment’ in the Act. The RMA (s2) defines ‘Coastal Marine Area’ (CMA) that describes morphological components and the geographic extent of CMA. By this definition, a coastal marine area is composed of foreshore, seabed, water and the air space above the water; and its extent covers an area below the line of mean high water springs on the landward side to the outer limits of the territorial sea on the seaward side (Figure 6.3-1). The extent and characteristics of the ‘coastal environment’ are identified in the NZCPS 2010. The seaward extent of the coastal environment is relatively straightforward; however, delineating landward extent of the coastal environment is challenging due to the variations between places and with time (Department of Conservation, 2010c).

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48 Coastal marine area (CMA) means the foreshore, seabed, and coastal water, and the air space above the water—
   a. of which the seaward boundary is the outer limits of the territorial sea;
   b. of which the landward boundary is the line of mean high water springs, except that where that line crosses a river, the landward boundary at that point shall be whichever is the lesser of—(i) 1 kilometre upstream from the mouth of the river; or(ii) the point upstream that is calculated by multiplying the width of the river mouth by 5 (RMA, s2).

49 Recognise that ‘coastal environment’ includes:
   a. the coastal marine area;
   b. islands within the coastal marine area;
   c. areas where coastal processes, influences or qualities are significant, including coastal lakes, lagoons, tidal estuaries, saltmarshes, coastal wetlands, and the margins of these;
   d. areas at risk from coastal hazards;
   e. coastal vegetation and the habitat of indigenous coastal species including migratory birds;
   f. elements and features that contribute to the natural character, landscape, visual qualities or amenity values;
   g. items of cultural and historic heritage in the coastal marine area or on the coast;
   h. inter-related coastal marine and terrestrial systems, including the intertidal zone; and
   i. physical resources and built facilities, including infrastructure, that have modified the coastal environment (NZCPS, 2010, p.11).

50 The seaward extent of the coastal environment includes the coastal marine area (CMA) (mean high water springs to 12 nautical miles), except where this cross a river and includes islands within the CMA.
This has resulted in the local authorities being left to decide the landward extent of their coastal resources based on their technical information and subjective judgment provided that the judgments comply with the principles of integrated and sustainable resource management required under the RMA (Department of Conservation, 2010c). Clarification of the issues surrounding the extent and characteristics of the coastal environment (especially the landward extent) has been carried out on a case by case basis and in some instances assisted by case law (Department of Conservation, 2010c).

To guide local authorities in defining their coastal environment, the Department of Conservation has developed a series of guidelines for implementation of the NZCPS (Department of Conservation, 2010a), which includes guidance on defining the extent and characteristics of the coastal environments (Department of Conservation, 2010c). The Department highlights the necessity of strategic planning and integration of freshwater and catchment management with their downstream coastal environment.

The RMA does not distinguish between coastal and terrestrial wetlands; rather it refers (s2) to the term ‘wetland’ in general. The NZCPS 2010 includes two references to the term ‘coastal wetland’, but does not include a definition for ‘coastal wetland’. Likewise, no definition of ‘coastal wetland’ is provided within the Auckland Plan and the AUP. As mentioned in Chapter 5, an informative fact sheet titled ‘coastal wetlands and estuaries’ prepared by the Auckland Council (Auckland Council, 2013) indicates that coastal wetlands in the Auckland region include mangrove swamps, saltmarshes and salt meadows.

Lack of specific and clear definition of coastal wetlands within the policy documents and legislation at both central and local levels is a potential source of ambiguity in using the term ‘coastal wetland’ in legislation and plans. For example, the term ‘coastal wetland’ is used separately from ‘saltmarsh’ in the NZCPS (Policy 1(2)(c); Policy 11(b)(iii)) and the AUP (e.g. Chapter E, E15, p. 3; Chapter I, I541, p. 12); whereas, as stated in Chapter 5, saltmarsh habitats are categorised as part of coastal wetlands within the scientific literature (Lee, et al., 2006; Herr, et al., 2012; Herr, et al., 2015; UNEP & CIFOR, 2014).

Moreover, unlike global classification of coastal (marine) wetlands (i.e. mangroves, saltmarshes and seagrass beds) (Herr, et al., 2015) seagrass beds are not a separate category in Auckland’s coastal wetlands classification (Auckland Council, 2013), and glasswort (Sacrocornia quinqueflora), a dominant salt meadow species in Auckland, is included in the category of saltmarsh wetlands within the global literature (e.g. Ouyang & Lee (2014)). This complicates comparisons with the international data.

Esplanade reserves (i.e. a 20-metre strip along the edge of major rivers, lakes and the coastlines) in New Zealand, historically derived from the concept of ‘Queens Chain’ to retain public access to coastlines and riverbanks, are good mechanisms to protect riparian and coastal margins for several purposes including mitigation of natural hazards (RMA, s229(a)(v)). Esplanade reserves are owned by the territorial authorities and apply to subdivisions under 4 acres (Ranson, 2015).

The RMA defines sustainable management in Section 5 (s5(2)) where the purpose of the Act to promote the sustainable management of natural and physical resources is provided. There have been debates around the nature of the sustainable management approach mandated by the RMA. One school of thought argues that the matters set out in paragraphs (a) to (c) in s 5(2) (See the footnote for definition of sustainable management in the RMA) provide a basic level of environmental protection and set out environmental bottom lines that must be met in all cases (Dawson, 2013). This interpretation is consistent with the

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51 The term ‘queen’s Chain’ is used ‘to describe various types of land status which provide public access and/or protect conservation values beside many, but not all, water bodies in New Zealand’ (Ministry of Justice, 2010).

52 In this Act, sustainable management means managing the use, development, and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic, and cultural wellbeing and for their health and safety while—

(a) Sustaining the potential of natural and physical resources (excluding minerals) to meet the reasonably foreseeable needs of future generations; and
(b) Safeguarding the life-supporting capacity of air, water, soil, and ecosystems; and
(c) Avoiding, remedying, or mitigating any adverse effects of activities on the environment.
legislative intent of the RMA when the Act was introduced to Parliament by Simon Upton (Former Minister for the Environment). He stated that the purpose of the RMA is not concerned with planning and controlling economic activity, nor about trade-offs; rather the Act identifies a set of biophysical bottom lines and provides for sustainable management through sustaining, safeguarding, avoiding, remediating, and mitigating the adverse effects of the use of natural resources (Dalal-Clayton & Sadler, 2014).

The other school of thought is based on an ‘overall broad judgement’ approach to interpret sustainable management as defined in Section 5 of the RMA (Dawson, 2013). This approach requires that the ecological values (stated in paragraphs (a) to (c) in s 5(2)) are judged alongside the other matters embodied in the main text of the RMA (e.g. matters of national importance set out in Sections 6 and 7) (Ministry for the Environment, 2012a). This approach has been dominantly taken by the Courts in decision-making under the RMA. This means that in effect, as interpreted by the Courts, regulatory authorities have a mandate to adopt an overall broad judgment approach and trade off values, including ecological ones, rather than meeting biophysical bottom lines (Ministry for the Environment, 2012a; Dawson, 2013).

The overall broad judgment approach was critised in the King Salmon decision by the Supreme Court in 2014 (Atkins & Dawson, 2014). The Supreme Court decision highlighted the importance of certainty or at least clarity of policy direction in higher order policy documents, particularly the NZCPS (policies 13&15) and suggested that applying an overall broad judgment and consideration of Section 5 of the RMA is not appropriate where the planning documents clearly set environmental bottom lines that must be given effect to (Atkins, et al., 2014).

6.3.2. Manage development impacts: Avoid, remedy, mitigate, offset

Given the specific nature of the King Salmon case (Daya-Winterbottom, 2014), the Supreme Court’s findings are not relevant to resource consents and under Section 104 of the RMA, using an overall judgment, which allows for comparison of conflicting considerations, is still appropriate for making decisions on resource consent applications (Atkins, et al., 2014). However, the Supreme Court’s findings affect the formulation of policy and plan documents by councils and in turn may influence resource consents through changing the status of development activities (Atkins, et al., 2014).

As mentioned earlier, the RMA (1991) provides for ‘avoidance, remediation or mitigation’ of the adverse effects of development on natural and physical resources as the requirement for sustainable management (s5(2)(c)). The NZCPS 2010 also adopts the same approach and requires ‘avoidance, remediation or mitigation’ of adverse effects of activities on indigenous ecosystems and habitats in the coastal environment which are particularly vulnerable to modification, including ‘estuaries, lagoons, coastal wetlands, dunelands, intertidal zones, rocky reef systems, eelgrass and saltmarsh’ (Policy 11(b)(iii)). The focus is, however, on habitats with important biodiversity, recreational, commercial, traditional or cultural values (Policy 11(b)). The climate change values of coastal wetlands are not referred to in this policy.

The place of ‘biodiversity offset’ in New Zealand’s current planning system is described in the non-statutory guidance on ‘Good Practice Biodiversity Offsetting in New Zealand’ that was released in 2014 (New Zealand Government, 2014). Biodiversity offsetting is a particular form of ‘offset’ or ‘ecological compensation’ (as discussed in Chapter 2) and is regarded as an early attempt to internalise the environmental costs of development projects that adversely affect the environment (von Hase & ten Kate, 2017). The definition of biodiversity offset in the guideline is derived from the Business and Biodiversity Offsets Programme (BBOP) (BBOP, 2012d) which is currently recognised internationally as a reference framework and standard for biodiversity offsetting (Holloway, 2014).

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53 New Zealand King Salmon’s proposals to establish salmon farms in the Marlborough Sounds (Atkins, et al., 2014).
54 Biodiversity offsetting is defined as ‘measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground’ (New Zealand Government, 2014, p. 3).
This definition constitutes the core principles of biodiversity offsets, i.e. ‘additionality’, ‘comparability’ and ‘longevity’ (Pilgrim & Bennun, 2014; OECD, 2014b). That is offsets are required to create ‘additional’ or ‘comparable’ biodiversity gains to compensate for the biodiversity losses caused by development over project lifetime (Bull, et al., 2013; Pilgrim & Bennun, 2014). The guidance provides for ‘no net loss’ and preferably a ‘net gain’ of biodiversity through offsetting significant (or more than minor) residual adverse effects after appropriate prevention, mitigation and on-site rehabilitation measures have been taken. This approach challenges the notion of avoid-remedy-mitigate (RMA, s5(2)(c)) which does not necessarily provide for a net ecological benefit as an outcome of land use/development activities (Knight-Lenihan, 2013; Knight-Lenihan, 2014).

The RMA (1991), the Conservation Act (1987) and the Crown Minerals Act (1991) provide legislative context for ‘biodiversity offsetting’, but none of them mandate the approach (Brown & Penelope, 2016). Policies under the RMA generally show a preference for ‘like-for-like’ offset activities, but they also provide for ‘trading up’ (out-of-kind) in situations where an area with lower conservation values can be traded with an area with more significant values (Brown & Penelope, 2016).

Mitigation, offset and compensation are now distinguished from one another according to the resolutions made by the Environment Court (Brown & Penelope, 2016). Mitigation is referred to ‘any action that directly addresses environmental damage within the impact footprints’, while offset includes ‘explicitly calculated positive actions undertaken outside the direct impact footprint’ (Brown & Penelope, 2016, p.36). Compensation is referred to positive actions as a result of development that are not classified under the offset and mitigation categories (Brown & Penelope, 2016).

Over the last few years, there has been a growing interest within the scientific community for enhancing knowledge about the issues, challenges and benefits of biodiversity offsets in the New Zealand context. Examples include studies by Stephens (2012); Knight-Lenihan (2013 and (2014); Overton, et al., (2013); Overton and Stephens, (2015); Brown, et al., (2015); Birkeland and Knight-Lenihan, (2016) and Brown and Penelope, (2016).

The above authors have generally argued that the current New Zealand regulatory framework that underpins biodiversity offsetting is weak and does not sufficiently advocate the potential benefits; and hence, it will likely result in a net decline of biodiversity rather a net gain. The literature has, therefore, called for a coherent national-level policy framework such as a national policy statement that could provide more robust and clear guidance to local government and would restrict the extent of discretion within the local decision-making process.

As argued by Brown and Penelope, (2016, p. 39), offset requirements in New Zealand are often ‘scattered, disconnected and ultimately non-strategic’. These authors have therefore called for incorporating offsets into a wider-level (e.g. landscape scale) conservation planning to improve the offset outcomes compared to the current ad-hoc approach. They have also recommended that regional authorities develop regional biodiversity plans and identify conservation priorities and targets that can be met via biodiversity offsetting. This approach is consistent with the ‘conservation benefits matching’ approach of biodiversity offsetting as will be discussed later in this section.

Issues related to ‘property rights’ (or existing use rights) and their erosion is argued to be a key barrier to biodiversity conservation on private lands (Kneebone, et al., 2000a; Kneebone, et al., 2000b; Greenhalgh, et al., 2010). The reports prepared by the Ministerial Advisory Committee in 200055, emphasised the importance of the management and conservation of indigenous biodiversity on private lands without compromising the legitimate rights of land owners to gain economic benefits from their lands (Norton, 2001). The majority of submissions on those reports sought clarity on the nature of property rights with

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respect to biodiversity, as well as on the relationship between existing use rights and compensation (Hill, 2001).

Despite the lack of a legal mandate and national policy statement about biodiversity offsetting (Brown, et al., 2013; Maseyk, et al., 2016), local governments are increasingly incorporating provisions that require developers or property-owners seeking to perform construction/development activities to offset residual adverse effects on biodiversity that cannot be avoided, remedied and mitigated (Holloway, 2014). However, it is argued that ecological compensation (including biodiversity offsetting) for development activities is often discretionary, inconsistently applied, and poorly monitored (Brown, et al., 2013). Moreover, the current approach to ecological compensation in New Zealand is argued to lean toward trading-off (i.e. balancing positive and adverse effects of the proposed activity on the environment) which theoretically implies ‘no net loss’ but in practice compensation without effective enhancement may result in a ‘net decline’ of ecological values (Birkeland & Knight-Lenihan, 2016).

As stated in the biodiversity offsetting guideline (New Zealand Government, 2014, p. 20), social and cultural values associated with biodiversity (particularly valued habitats or species), as well as ecosystem functions (e.g. nutrient cycling rates) need to be considered when designing offset plans. For example, if a wetland within an impact site has a flood reduction function, the restored wetland in a proposed offset site also needs to have the same function. There is, however, no explicit reference in the guideline to the protective role of coastal wetlands against sea level rise and storm events and other values associated with biodiversity, such as carbon sequestration.

The current accounting system (model) for biodiversity offsets in New Zealand applies to ‘like-for-like’ offsets and is focused on biodiversity values of ecosystems. The model considers three hierarchical levels to categorise biodiversity including (i) biodiversity type (e.g. forest, wetland); (ii) biodiversity components (e.g. canopy, diversity) and, (iii) biodiversity attributes (e.g. canopy cover, height of canopy, species diversity) and uses a ‘disaggregated area x condition’ currency to demonstrate no net loss for biodiversity (Maseyk, et al., 2015). In doing so, the model calculates a Net Present Biodiversity Value (NPBV) for each biodiversity attribute by comparing losses and gains between the impact and the offset sites (Maseyk, et al., 2015). The concept of NPBV was first introduced by Overton, et al., (2013, p. 102) and is defined as ‘a basic measure against which the no-net-loss criterion can be measured’.

Area x condition is the most popular and commonly used type of currencies to assess ecological losses and gains in offset projects (BBOP, 2012e). However, the use of ‘area x condition’ currencies involves some limitations such as difficulty in benchmarking and lack of ecological context (Department of Conservation, 2014; Birkeland & Knight-Lenihan, 2016). Disaggregated currencies ‘capture more information by measuring multiple attributes and their condition’ (Department of Conservation, 2014, p. 4) and maintain the identity of individual biodiversity components (BBOP, 2012e).

The use of disaggregated currencies is therefore argued to better provide for avoiding trade-offs between biodiversity components (Department of Conservation, 2014). There is also another type of currencies generally known as ‘context-dependent’ currencies (BBOP, 2012e), but are also referred to as ‘ecological integrity’ (Department of Conservation, 2014) or ‘conservation benefit matching’ currencies (Birkeland & Knight-Lenihan, 2016). These currencies provide for consideration of high level (e.g. national, regional) conservation goals and priorities when assessing biodiversity losses and gains (BBOP, 2012e; Department of Conservation, 2014).

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Context-dependent currencies can act as an overarching framework for offsets under ‘area x condition’ where actions are done in the context of identified prioritised areas for enhancement (Birkeland & Knight-Lenihan, 2016). They have the potential to contribute to a net gain of biodiversity within an appropriate legislative and regulatory setting (BBOP, 2012c; Birkeland & Knight-Lenihan, 2016). However, using these currencies requires good quality data (which is often not available for all aspects and components of biodiversity) and monitoring to identify whether an offset will contribute to achieving the high-level goals (Department of Conservation, 2014; Birkeland & Knight-Lenihan, 2016). One Plan 2014, prepared by Manawatu-Wanganui (Horizons) Regional Council (Horizons Regional Council, 2014), represents an example of practicing the ‘conservation benefit matching’ approach in New Zealand. The plan sets out terrestrial and aquatic ecological improvement targets which will be taken into account in managing and planning land use activities in relevant catchments.

The AUP refers to ‘biodiversity offset’, ‘no net loss’ and preferably a ‘net gain’ in ecological values (Appendix 8; Chapter D, D9.3.1(d); Chapter E, E3.8.1(1)(d)) and generally provides for offsetting any residual adverse effects of a development or activity on the environment that cannot be avoided, remedied or mitigated (Chapter E, E15.3(3)). It also provides for offsetting any remaining ‘more than minor’ adverse effects on lakes, rivers, streams or wetlands through restoration and enhancement that are ‘like-for-like’, achieve no net loss and are implemented in the closest proximity to the subject site or within the same catchment (Chapter E, E3.3(4)). However, it is not clear whether the term ‘wetlands’ refers to freshwater (terrestrial and coastal) or coastal saline wetlands.

Similar to the biodiversity offset guideline, the AUP does not provide for offsetting adverse effects of land use activities on the carbon sequestration and coastal protection values of coastal wetlands. There is also no reference to ‘offset’ where policies and rules addressing mangrove removal in coastal marine zones are provided (Chapter F, F2). In effect, it is not clear whether and how mangrove removal activities are required to avoid, remedy, mitigate or offset the adverse effects of removing mangroves in terms of managing climate change. It contradicts the general requirement for offsetting the adverse effects on wetlands as mentioned above.

### 6.3.3. Policy evaluation: section 32 of the RMA

To achieve the overall purpose of the RMA (i.e. sustainable management), the effects of individual land use/development activities on natural and physical resources (including the coastal marine environment) must be identified through a project-based Environmental Assessment (EA), termed Assessment of Environmental Effects (AEF)\(^{58}\). There is no mandatory requirement for undertaking Strategic Environmental Assessment (SEA) in New Zealand (van Roon, et al., 2016). Section 32 of the RMA is the closest statutory tool which provides for a strategic analysis of policies (Policy-based Environmental Assessment) that is typical of SEA; however, it does not mandate SEA (van Roon, et al., 2016). Section 32 provides for identification and assessment of the effects of proposals (e.g. plans, plan changes and policy statements) on the community, the economy, and the environment through using a number of evaluation methods including cost-benefit assessment (s32, (2)(b)).

The use of cost-benefit assessment is also proposed for policies that place importance on the need to demonstrate a net benefit (Ministry for the Environment, 2014b). Under this approach, all costs and benefits of a proposal should be quantified (if practicable) and assessed. There is, however, no reference to the ‘co-benefits’ associated with different policies particularly their climate change co-benefits (e.g. climate change mitigation potential of coastal protection strategies). Likewise, the emission implications

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\(^{58}\) Ministry for the environment, *A guide to preparing a basic assessment of environmental effects*, Wellington: Ministry for the environment.
of policies (e.g. mangrove removal policies) that can be approximately quantified (e.g. through using Social Cost of Carbon\textsuperscript{59}) are not recognised in assessing the costs and benefits of proposals.

Although, quantification (monetising) of costs and benefits is encouraged, the use of qualitative information that is collected through different methods such as ‘case studies and literature reviews’ is also considered when benefits are difficult to monetise or quantify. For example, it is recognised that quantifying the impacts on a number of values such as biodiversity, cultural, recreational and amenity may be difficult due to the methodological limitations or for ethical reasons (Ministry for the Environment, 2014b). For quantitative estimates, the assumptions, limitations and the scope of quantitative information need to be clearly stated (Ministry for the Environment, 2014b).

6.3.4. Policy response to mangrove expansion

Given the expansion of mangroves in some estuaries across the Auckland region and increased pressure mainly from coastal communities to remove mangroves (as discussed in chapter 5), the earlier notified version of the AUP (i.e. PAUP, September 2013) provided for a ‘permitted activity’ rule to facilitate removal of mangroves (including seedling) to a state they existed in 1996. However, removal was not allowed in areas (i) where mangroves provide important ecological values, (ii) of active coastal erosion where mangroves provide a buffer against coastal processes causing erosion; and (iii) where the sediments contain high levels of contaminants at risk of being re-suspended.

The only use of the term ‘mangrove’ in the Auckland Plan (Chapter 7, Clause 459) is where the Plan posits that increased sediment inputs to marine areas caused by land-based activities can degrade coastal habitats e.g. by mangrove expansion. In this statement, mangrove expansion is seen as a threat to coastal habitats; without an explicit indication of the coastal habitat types affected by mangrove expansion. This specific reference of the Auckland Plan to mangroves is lacking a context and a clear approach to management of coastal ecosystems in general and cumulative mangrove loss in particular.

The issue of mangrove removal was one of the most challenging and contentious issues through the submission and hearing processes of the PAUP (PAUP Independent Hearings Panel, 2016) with a wide range of conflicting views put forward by both proponents and opponents. A number of submitters, particularly those from coastal communities, sought a different date (rather than the proposed date of 1996) as the base date for permitted mangrove removal activities in a number of Auckland’s estuaries such as the Whangateau Harbour (Omaha), the Kaipara Harbour and the Manukau Harbour (PAUP Independent Hearings Panel, 2016). Their reasoning was that the proposed date ‘was arbitrary and unnecessarily restrictive on mangrove removal’ (PAUP Independent Hearings Panel, 2016, p.8). The benchmark date of 1996 was also scrutinised for not being based on a robust technical analysis (PAUP Independent Hearings Panel, 2016).

Another group of submitters (such as the Mangrove Protection Society, supported by the Royal Forest and Bird Protection Society, the Environmental Defence Society, and the Waitākere Local Board) opposed the permitted activity status for mangrove removal and demanded a resource consent process for any removal activity except for seedling removal (PAUP Independent Hearings Panel, 2016). Their rational was mainly an emphasis on the importance and significance of mangroves for their ecological values particularly supporting threatened bird species (such as banded rail).

Considering the outcomes of the submissions and hearings, the Auckland Unitary Plan Independent Hearings Panel (the Panel) provided for a less permissive mangrove removal rule and recommended that removal of mangroves to be treated as a ‘discretionary activity\textsuperscript{60}’ that requires case-by-case assessments

\textsuperscript{59} Social cost of carbon (SCC) is the estimated price of the economic or social costs or damages caused by each additional tonne of CO\textsubscript{2} emission, and has been commonly used to assess the benefits of climate change mitigation policies (Nordhaus, 2014).

\textsuperscript{60} Resource consent is required for a discretionary activity. Council may approve or decline a proposal for a discretionary activity. In assessing the proposed activity, council can consider all relevant objectives and policies
based on a thorough consideration of the values of mangroves rather than against a particular date (PAUP Independent Hearings Panel, 2016). The Panel’s recommendation was based on the consideration of the complexity of mangrove management (including removal) which requires accounting for several competing values such as ecological, biological, natural character, amenity and cultural values as well as public access and use of the coastal environment (PAUP Independent Hearings Panel, 2016).

The latest notified AUP (15 November 2016) which included the Council’s decisions based on the Panel’s recommendation, has therefore identified mangrove removal as a ‘discretionary activity’ that requires a resource consent (AUP, Chapter E, E3, p.8) (Auckland Council, 2016b). Consistent with the Panel’s recommendation, removal of mangrove seedlings is considered as a ‘permitted activity’ (AUP, Chapter E, E3, p.8). Small-scale mangrove removal (up to 200m$^2$) is also permitted to enable the use and maintenance of lawful structures and public infrastructure (AUP, Chapter F, F2, Table F2.19.4). The AUP also outlines that removal of mangroves to improve amenity and landscape values of coastal areas must be assessed and weighed against the important ecological and biological values (services) of mangrove ecosystems (AUP, Chapter F, F2, p.14).

Given that expansion of mangroves has been influenced by increased sediment deposition (mainly due to catchment land use activities such as deforestation and construction as discussed in Chapter 5), the AUP (15 November 2016) included more specific provisions addressing the link between sediment control and mangrove management compared to its earlier notified version (i.e. PAUP). For instance, the AUP provisions on mangrove management in coastal marine area include a specific objective for reducing the amount of sediment entering the coastal area to control mangrove colonisation (Chapter F, F2.7.2 (4)).

The AUP also requires proposals for mangrove removal to assess sediment input and identify catchment initiatives for reducing the amount of sediment and nutrient entering coastal areas (Chapter F, F2.23.2(12)(b)). This might have satisfied the concerns raised by a number of submitters (including submissions from the Royal Forest and Bird Protection Society; the Environmental Defence Society; Friends of Oakley Creek Te Auaunga and a number independent public submissions) that required the PAUP to provide a stronger link between mangrove removal activities and sediment-generated activities in the catchment (Auckland Council, 2016c).

Although the condition of 1996 as the base year is lifted from the provisions, the AUP is still based on the idea to control and reverse expansion of mangroves primarily via its removal in combination with upstream land use and stormwater management. Optimistically the decisions in the AUP may help reduce the speed of mangrove removal in Auckland, due mainly to the provision that requires full account of all competing values of mangroves through resource consent processes. However, the complexity of such assessment especially in the absence or paucity of data may lead to prolonged case law processes. Despite the above potential positive trajectory, there is a lack of clarity on how this approach to the management of mangroves fits into the framework of the biodiversity offsetting as set out in the AUP.

### 6.3.5. Values of the coastal environment

Saltmarsh and seagrass habitats in Auckland are protected under the AUP, mainly for their significance in providing habitats for birds, fish and supporting indigenous biodiversity (e.g. Chapter F, F2, F2.21.4.2; Chapter I, I541.8.2.1(17); Chapter E, E3.6.1.9(7); Chapter F, F2.21.5.6 (1)(f)).

The Auckland Plan recognises the importance of the coastal and marine environment for providing recreational (e.g. clauses 349, 362, 448), ecological (e.g. clause 458), economical (e.g. clauses 362, 374, 457), cultural (e.g. clause 312) and amenity values (e.g. clause 556). Of these values, the recreational values of the coastal environment in Auckland receive significant importance on the ground that they contribute to the economy and liveability of the region (Clause 362). While the Auckland plan is supposed to be

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within the Unitary Plan, all potential environmental effects, and any matters outlined in s.104 of the RMA without limitation in decision-making (Auckland Council, 2016b).
strategic (and spatial) in approach, it remains opaque about the values and the potential or actual benefits of the coastal wetlands and their specific elements in Auckland (i.e. mangroves, saltmarshes and seagrass beds).

As indicated in the AUP (Chapter B, B7, p.10), areas with high ecological values are identified as ‘Significant Ecological Areas’ (SEAs) using a number of significance criteria as set out in Schedules 3 and 4 of the Plan. The criteria used to identify Significant Ecological Areas – Marine (SEA-M) include (a) recognised international or national significance; (b) threat status and rarity; (c) uniqueness or distinctiveness; (d) diversity; (e) stepping stones, buffers and migration pathways and (f) representativeness. There is explicit reference to mangrove, saltmarsh, seagrass and terrestrial coastal vegetation as intact ecosystems that form ecological sequences across intertidal gradients (AUP, Schedule 4, p. 3).

The current criteria are mainly focused on the significance of marine ecosystems for protecting indigenous biodiversity. Other services of these ecosystems particularly their climate change services are not taken into account. A number of mangrove and saltmarsh habitats at different estuaries across the region are identified as SEA-M, particularly for their importance as habitats for wading birds. Seagrass meadows at one estuary in the region (Kaipara Harbour) are also included within the SEA-M, mainly for their significance for supporting a wide variety of aquatic fauna and flora (AUP, Schedule 4, p. 8).

6.3.6. Approaches to integrated management

New Zealand’s current legislative and policy framework generally promotes and in some instances, mandates local governments to adopt and apply an integrated approach to resource management. The legislation that places strong emphasis on linking catchment management with marine management is the Hauraki Gulf Marine Park Act 2000 (The HGMP Act 2000) which specifically provides for ‘integrated management’ of the Hauraki Gulf’s environment. The HGMP61 is located in the upper North Island of New Zealand and has given a special importance because of its significant social, economic and cultural values, as well as high vulnerability to human activities (Hauraki Gulf Forum, 2014). The HGMP includes more than half of the coastlines and coastal wetlands of the Auckland region.

The HGMP Act seeks to achieve the purpose of ‘integrated management’ through (i) integrating the management efforts of different agencies involved in the management of the HGMP, (ii) providing a set of common management objectives for the Gulf and (iii) linking the management of catchment activities with the management of the Gulf’s coastal marine environment (Hauraki Gulf Forum, 2014).

Policy 4 of the NZCPS 2010 also explicitly sets out requirements for integrated management of the coastal environment. Similar to the approach taken by the HGMP Act, the NZCPS also provides for a coordinated and collaborative management of activities within the coastal environment with a focus on sediment generating activities that affect the quality of water in the coastal environment and marine ecosystems.

Unlike the ‘effect-based’ approach of the RMA, the HGMP Act mainly focuses on ‘protection’ and ‘where appropriate’ enhancement of the ‘life-supporting capacity’ of the environment of the Hauraki Gulf, its islands, and catchment’ (HGMP Act, s8(a)). The definition of the ‘life-supporting capacity’62, as provided by the Hauraki Gulf Forum (Hauraki Gulf Forum, 2009), does not explicitly refer to the climate change values of the Gulf’s marine environment; however, the HGMP Act itself provides a legislative basis to

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61 The HGMP includes the foreshore and seabed of the Hauraki Gulf, Waitemata Harbour, Firth of Thames, and east coast of the Coromandel Peninsula (Gaskin & Rayner, 2013).

62 The life-supporting capacity of the environment of the Gulf and its islands includes ‘the capacity: a. to provide for: (i) the historic, traditional, cultural, and spiritual relationship of the tangata whenua of the Gulf with the Gulf and its islands; and (ii) the social, economic, recreational, and cultural wellbeing of people and communities; b. to use the resources of the Gulf by the people and communities of the Gulf and New Zealand for economic activities and recreation; c. to maintain the soil, air, water, and ecosystems of the Gulf’ (Hauraki Gulf Forum, 2009, p.92).
‘protect’ and ‘enhance’ ecological values of the Gulf’s marine environment including coastal wetland ecosystems.

The AUP gives effect to these policy statements by including objectives, policies and provisions that focus on controlling discharges to the coastal marine area and maintaining coastal water quality including benthic sediments. The Auckland Council as one of the regulatory authorities involved in the management of HGMP, is required to give effect to the provisions of the HGMP Act primarily through regional policy statements. The AUP provisions of particular relevance to the Gulf, mainly provide for a comprehensive management of sediment run-off from earthworks and land use activities, a more controlled management of livestock access to the coastal marine area, improving the quality of stromwater and run-off and limiting urban growth within the proposed Rural Urban Boundary (Hauraki Gulf Forum, 2009).

Management of mangrove expansion through removal of expanded mangroves, is also identified as one of the AUP provisions related to the HGMP (Hauraki Gulf Forum, 2014). It seems that the current policies with respect to mangroves are expected to result in a gradual decline and ultimately a net reduction in the extent of coastal wetlands relative to their current state.

The National Policy Statement for Freshwater Management 2014 (NPSFM) provides direction to local authorities on integrated management of freshwater and particularly provides for improvement and management of two important values including ecosystem health and human health for recreation. It requires regional councils to identify ‘Freshwater Management Units’ (FMUs) as spatial scales for management purposes (Ministry for the Environment, 2014a). This spatial zoning provides opportunity for better management of catchment-based activities that affect coastal marine ecosystems.

To manage cumulative effects, the NPSFM includes provisions that require the freshwater quantity and quality limits to be set by making the best use of available information including scientific and socio-economic knowledge (Ministry for the Environment, 2014a). Although estuaries and coastal ecosystems are excluded from the provisions of the NPSFM, they must be ‘given regard to’ when setting limits for freshwater management (NPSFM, Policy A1). The impacts of climate change must also be considered in determining the limits. The NPSFM is planned to be fully implemented by every regional council by 31 December 2025 (Ministry for the Environment, 2014a).

The Auckland Council, has already identified ‘Consolidated Receiving Environments’ (CREs) across the Auckland region (Figure 6.3-2). CREs are considered as the primary basis for identification of FMUs (Auckland Council, 2015c). Boundaries of the CREs are identified ‘based on amalgamating surface water catchments for the three harbours in Auckland’ (Auckland Council, 2015c, p.9) and each CRE constitutes either an estuary or part of the coast (Sea Change, 2016, Appendix 4). To give effect to the provisions of the NPSFM (2014), the AUP sets out requirements for managing adverse effects of development activities on freshwater and coastal water (Chapter E, E1. P.1).

The AUP provides for maintaining and improving the quality of coastal water mainly through mitigating the generation and discharge of contaminates (Chapter E, E1.3(8)(b)) and sediment (Chapter F, F2.3.3(9&10)). The Auckland Council has also developed a new programme known as the Healthy Waterways (Wai Ora) to ensure implementation of the NPSFM and the AUP’s provisions on freshwater management. The Auckland Council’s Sustainable Catchments Programme is another initiative that seeks to provide a framework for an integrated catchment planning through iwi and community participation (Environment Guide, 2015). This programme is currently implemented in seven priority

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63 The NPSFM 2014 is the amendment to the NPSFM 2011.
64 ‘Freshwater management unit’ is ‘the water body, multiple water bodies or any part of a water body determined by the regional council as the appropriate spatial scale for setting freshwater objectives and limits and for freshwater accounting and management purposes’ (Ministry for the Environment, 2014, p.7).
catchments across Auckland and aims to identify and promote a range of activities to protect long-term health of the receiving environments.

The Auckland Plan (Chapter 7, Strategic Direction 7) provides for development of a marine spatial plan for the Hauraki Gulf, Kaipara Harbour, Manukau Harbour and West Coast. The marine spatial plan for the Hauraki Gulf, known as Sea Change (Tai Timu Tai Pari) which has been developed over four years since 2012, was released in December 2016. The plan uses the best available scientific information and Mātauranga Māori knowledge to identify a roadmap for a collaborative management of the HGMP. The plan itself is a non-statutory document, but its findings will be used to inform changes to the statutory policies particularly in terms of managing catchment activities that affect the HGMP.

Figure 6.3-2. The Auckland Council’s Consolidated Receiving Environments (Sea Change, 2016, Appendix 4, p.328)

The Sea Change plan refers to the potential of aquaculture activities, e.g. shellfish farming to absorb carbon and contribute to climate change mitigation (Sea Change, 2016). There is no mention of the ‘blue carbon’ and coastal protection potential of coastal wetlands within the HGMP. But as part of the activities to address sediment management in the coastal marine areas, the plan provides for retaining mangroves as ‘effective natural means of trapping sediment’ (Sea Change, 2016, p.141).

The plan recognises the adverse effects of climate change (e.g. more frequent and intense storm events and sea level rise) on coastal ecosystems and marine biodiversity and identifies a specific theme to restore ‘healthy functioning ecosystems throughout the Hauraki Gulf Marine Park including those in freshwater, estuarine, inshore and deepwater areas’ (Sea Change, 2016, p.116). Establishment of a long-term research program to provide more information about the dynamics of the HGMP’s ecosystems and a monitoring
system to measure the ecosystem health of the HGMP are examples of activities identified to achieve the objectives of the proposed biodiversity theme. Restoration of seagrass beds as nursery habitats for fish species is also considered in the plan.

A limit of 2mm per year above the baseline rate\textsuperscript{66} is identified as the acceptable rate of sedimentation in estuaries and coastal areas across the HGMP by 2050 (Sea Change, 2016). The plan refers to this limit as ‘the ecological adverse-effects threshold’ (Sea Change, 2016, Appendix 4, p.320). Appendix 4 of the Sea Change plan provides more information on the sediment control objectives, but it does not clearly specify whether the limit for sedimentation rate has also taken into account the long-term sustainability of mangroves, particularly with respect to sea level rise. The plan is also not specific about how the effects of climate change (as proposed in the NPSFM), particularly sea level rise, are considered in determining the sedimentation limit.

As mentioned in Chapter 5, sediment availability is an important parameter that affects the potential of mangrove ecosystems to keep pace with sea level rise (Lovelock, et al., 2015). It is therefore necessary to make sure that the identified sedimentation limit will not adversely affect Auckland’s mangrove ecosystems over a long run. According to the plan, the sedimentation limit will be monitored across a number of monitoring sites identified across the HGMP. The baseline rates will be established in 2019 and the monitoring will be carried out every five years starting from 2025 (i.e. 2025, 2030, 2035, 2040, 2045, 2050).

The plan is supplemented with an online spatial planning tool, SeaSketch\textsuperscript{67}, that provides an opportunity for community involvement in the planning and management of the HGMP. A wide range of information about the natural ecosystems of the HGMP is available to the public through the SeaSketch website. For example, the layer of ‘Marine Habitats’ under the category of ‘Marine Environment’, demonstrates the boundaries of different habitat types including mangrove and saltmarsh ecosystems.

Overall, establishment of the FMUs in the Auckland region and adopting a catchment-wide approach to address the cumulative effects of activities on freshwater and coastal environment are significant steps towards the integrated management of natural resources as promoted by different statutes. However, as noted above, it is necessary to make sure that the identified limit for sedimentation rate has taken into account future rises in sea level and the sustainability of mangrove ecosystems.

6.3.7. Current restoration activities

There are currently community-based trusts that encourage public involvement in protecting and/or enhancing dunes and wetlands. The ‘Dune Restoration Trust of New Zealand’ is specifically established to protect and restore sand dunes given the important function of dunes as natural barriers between land and sea. The ‘National Wetland Trust’ is also working to protect and restore wetlands in New Zealand and recognises the values of wetlands in reducing flood risks and storing carbon. However, these efforts appear to be mainly focused on terrestrial freshwater wetlands including peat bogs which are also recognised for their carbon storage capacity.

Wetlands restoration activities have been initiated in New Zealand to reduce vulnerability (increase resilience) of wetlands to multiple threats including the effects of climate change. The first pilot site in New Zealand is the Whangamarino Wetlands in the Waikato Region which is recognised as the second largest bog and swamp complex in the North Island. Restoration of these wetlands aims to provide

\textsuperscript{66} Baseline rate is defined as the rate when the catchment was fully forested. It varies across different estuaries (Sea Change, 2016, Appendix 4).

\textsuperscript{67} http://www.seasketch.org/#projecthomepage/52322dd05d3e2c665a00d119
significant benefits in terms of disaster prevention and flood control, as well as a wide range of co-benefits including recreational opportunities, carbon sequestration and water supply.\footnote{UNFCCC, Whangamarino Wetlands. [Online] Available at: \url{https://unfccc.int/files/adaptation/application/pdf/38eba.pdf} [Accessed 2016].}

\section*{6.4. Responses to climate change and coastal hazards}


Under the UNFCCC, New Zealand is committed to annually report its total human-induced emissions and removals of greenhouse gases.\footnote{Under the 2006 IPCC guidelines (IPCC, 2006), the main categories include forest land, cropland, grassland, wetlands, settlements and other land (Ministry for the Environment, 2016).} New Zealand’s latest National Greenhouse Gas Inventory Report (1990-2014), released in May 2016, provides net emissions estimates for the wetland category, as one of six land use categories of the Land use, land use change and forestry (LULUCF) sector. The wetland category consists of two sub-categories including open water and vegetated wetland. Estuarine (i.e. tidal areas including mangroves) is included within both sub-categories.\footnote{http://www.mfe.govt.nz/climate-change/international-forums-and-agreements/united-nations-framework-convention-climate [Accessed June 2016].} As the report indicates, the wetland category (in general) has been a net remover in 1990 (-20.7 Kt CO\textsubscript{2}e) and has changed to a net emitter (2.7 Kt CO\textsubscript{2}e) in 2014.\footnote{http://www.mfe.govt.nz/climate-change/international-forums-and-agreements/united-nations-framework-convention-climate [Accessed June 2016].} This is associated with changes in land use patterns during this period.\footnote{http://www.mfe.govt.nz/climate-change/international-forums-and-agreements/united-nations-framework-convention-climate [Accessed June 2016].} There has been a decrease in the area of lands converted to wetlands between 1990 and 2014. There were 14,386 ha of lands in a state of conversion to wetlands in 1990; while this figure has been dropped to 7,055 ha as at 2014. It is reported that ‘these lands have been converted to Wetlands during the previous 28 years but have not yet reached steady state and entered the Wetlands remaining wetlands category’ (Ministry for the Environment, 2016, p. 278).
Despite the inclusion of mangrove ecosystems within the area reported under the wetland category, carbon sequestration from mangroves is not included in the net emission and removal estimates for this category. It is because the 2006 IPCC Guidelines (IPCC, 2006) provide no guidance for including carbon storage or emissions from mangrove ecosystems in the estimates of net emissions and removals for the wetland category.

New Zealand has incorporated its international climate change obligations into domestic legislation largely through the Resource Management (Energy and Climate Change) Amendment Act 2004 and the Climate Change Response Act 2002 (CCRA 2002) (Baillie, 2012). The RMA sets out the roles and responsibilities of the central and local governments in relation to response to climate change. The main focus of the RMA provisions regarding climate change mitigation is on the use and development of renewable energy (RMA, s7(j)). The CCRA 2002 was amended in 2008 (Climate Change Response, Emissions Trading Amendment Act 2008) to establish the New Zealand Emission Trading Scheme (NZ ETS) (Baillie, 2012).

The NZ ETS supports and encourages farmers and land owners to obtain carbon credits by undertaking activities (e.g. tree planting, sustainable farming) that could increase carbon storage and reduce emission of GHGs from land. Carbon credits that are traded in New Zealand Units (NZUs) can be sold to those who want to offset their emissions or can provide funding for restoration projects (Ministry for Primary Industries, 2016). The New Zealand Emission Unit Register (NZEUR) developed in 2007 manages the accounting, reporting and reconciliation of emissions and unit holdings and transactions as part of the NZ ETS. Based on the new changes to the NZ ETS, the one-for-two subsidy which allows non-forestry businesses to pay one emissions unit for every two tonnes of carbon dioxide equivalent emissions will be phased out over three years starting from 1 January 2017 (New Zealand Parliament, 2016).

The ‘Permanent Forest Sink Initiative’ (PFSI) and the ‘Afforestation Grant Scheme’ (AGS) are also other national market-based mechanisms that support and encourage carbon management through forestry and plantation activities (Ministry of Agriculture and Forestry, 2011). However, coastal wetlands are not addressed in the NZ ETS, PFSI and AGS.

Sections 70A and 104E of the RMA restrict the ability of local authorities to consider the effects of greenhouse gases (GHGs) emissions on climate change (except for the use and development of renewable energy) when considering applications for a discharge or coastal permit. However, local authorities are required to have particular regard to the effects of climate change when planning and managing their coastal resources (RMA, s7(i)). Local authorities can delegate part or all of their responsibilities for hazard management to appropriate agencies who have the resources and capacity for hazard management, provided that councils supply an appropriate level of guidance and support to those agencies (Ministry for the Environment, 2008a).

The result of the above legislative arrangements is that management of climate change mitigation is a central government responsibility through the NZ ETS, while climate change adaptation is done primarily by local governments. However, local governments can still take on initiatives that can contribute to climate change emissions reductions through co-benefits associated with, for example, adopting a ‘compact city’ urban form.

No single entity is responsible for the management of natural hazards in New Zealand. The responsibility is rather distributed over several entities including the Ministry of Civil Defence and Emergency Management (MCDEM), regional councils, territorial authorities, emergency management officers, Civil Defence and Emergency Management Groups, Engineering Lifelines Groups and Canterbury Earthquake Recovery Authority who are responsible for and contribute to natural hazard management (Saunders, et al., 2013). The RMA (1991), the Building Act (2004), the Civil Defence Emergency Management (CDEM)

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Act (2002), the LGA (2002) and the NZCPS (1994&2010) are the main statutes that have primary influence on managing natural hazards and effects of climate change in New Zealand (Hart, 2011; Saunders, et al., 2013).

However, the current provisions of these statutes do not require adoption of long-term dynamic adaptation strategies such as ‘transformational change’ to future coastal hazards (Hart, 2011). The only exceptions are the NZCPS 1994 and 2010 that have included explicit references to adaptation strategies such as ‘abandonment or relocation of existing structures’ (NZCPS 1994, Policy 3.4.6) and ‘managed retreat’ in the context of climate change (NZCPS 2010) (Hart, 2011). Potential barriers to managed retreat are further discussed in the following sections.

Under the RMA (s88(2)(b)), each application for a resource consent (including any application to subdivide, use or develop on, by or near land subject to natural hazard risk) must be accompanied by an assessments of environmental effects (AEEs) that needs to include any risk to the neighbourhood, the wider community, or the environment through natural hazards (Schedule 4, Clause 7(1)(f)).

Under the RMA (s.30 & s.106) and the Building Act (s71-74), consent authorities are required to avoid or mitigate natural hazards and can refuse to grant a building consent for development on land subject to natural hazards. An example of the local level policies to address natural hazards is the Environment Canterbury’s Regional Coastal Environment Plan which identifies setback distances for developments in coastal areas (Ministry for the Environment, 2008b). There is also case law (e.g. Gallagher v Tasman District case law77) that has refused appeals where the risks associated with coastal hazards and sea level rise were strong and obvious.

Under section 71 of the Building Act, consent authorities may refuse to grant a building consent if the land is subject or is likely to be subject to one or more natural hazards, or the construction work is likely to accelerate, worsen, or result in a natural hazard on that land or any other property. However, this restriction does not apply if adequate provision has been made to protect the land or building work, or to restore any damage to the land or other property as a result of the building work (s71(2)).

The Building Act (2002) provisions regarding building on land subject to natural hazards do not differentiate between lands subject to one, and land subject to more than one, natural hazards, despite, the level and extent of vulnerability differing between land subject to one hazard and land subject to more than one hazard. According to the 2014 amendment to the LGA, a separate infrastructure strategy that includes explicit consideration of resilience of infrastructure in the event of natural disasters is required for a period of at least 30 consecutive financial years (Willis, 2014).

The CDEM Act (2002) provides for adopting an integrated approach of 4Rs (Reduction, Readiness, Response, Recover) for management of civil defence emergencies including natural hazards (Ministry of Civil Defence and Emergency Management, 2016a). In practice, the emphasis is however on the last three components (Hart, 2011). The CDEM Act (2002) provides no guidance or direction on the nature or type of risk reduction strategies or actions except for referring to cost-effective risk reduction measures. The generic definition of ‘hazard’ in the CDEM Act (2002) can include climate-induced coastal hazards; but there is no explicit reference to ‘climate change’ in the body of the CDEM Act (2002).

The National CDEM Strategy (2008)78 (Ministry of Civil Defence and Emergency Management, 2008) and the National CDEM Plan (2015) (Ministry of Civil Defence and Emergency Management, 2016b) (as requirements under the CDEM Act 2002) provide for reducing the risk of natural hazards to communities and highlight the need for using a combination of protective measures as one of the principles underlying risk reduction. However, they do not include any reference to the importance of natural buffers and coastal

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77 NZEnvC 245 (New Zealand Environment Court, 2014)
78 The National CDEM Strategy, developed in 2008, is currently under review to align with a commitment New Zealand made to the international Sendai Framework for Disaster Risk Reduction (Auckland Civil Defence, 2016).
wetlands in reducing the risk of coastal hazards and do not provide for protection and enhancement of these ecosystems as a risk reduction solution in coastal areas.

The first comprehensive review of the physical impacts of climate change on New Zealand’s coastline and estuaries was conducted by Hicks (1990) prior to the first IPCC assessment report. He provided the first general assessment of the vulnerability of New Zealand’s coasts to sea level rise in the range of 20-60cm (Moore & Smith, 1996). New Zealand has a history of coastal vulnerability mapping. The first mapping approach was developed by Gibb, et al., (1992) that used eight parameters to identify the susceptibility of 110 locations around New Zealand coast to coastal hazards. The national-scale maps demonstrating sensitivity or vulnerability of New Zealand coastline to coastal inundation and erosion were developed in 2012 by National Institute of Water & Atmospheric (NIWA) Research Ltd (Goodhue, et al., 2012).

For preparing these maps, Coastal Sensitivity Indices (CSI) were identified using a combination of four geomorphic (exposure, hinterland, sediment type and landform type) and three oceanographic (high tide range, change in storm surge and wave height) variables. These variables were selected by using information from international studies on coastal sensitivity. Given the large scale of mapping, the CSIs for inundation and coastal erosion were calculated using the best available New Zealand-wide information. However, as stated by Goodhue, et al., (2012), subject to availability of information, the CSIs could be improved by incorporating data on rates of coastal erosion/accretion and socio-economic variables. It is suggested that councils can combine the national CSI information for their region with additional local information (e.g. land use pattern, hazards reducers/exacerbators) to provide more detailed and robust coastal vulnerability maps.

The NZCPS 2010 is the only regulation that provides for protection and restoration of natural defences to reduce the risk of coastal hazards (taking account of climate change). Policy 26 of the NZCPS includes beaches, estuaries, wetlands, intertidal areas, coastal vegetation, dunes and barrier islands as natural defences; but it does not clearly specify whether coastal wetlands (i.e. mangrove, saltmarsh and seagrass ecosystems) are types of natural defences. The previous version of the NZCPS (i.e. the NZCPS 1994, Policy 3.4.3) had included mangroves within the list of ecosystems that act as natural defences and had specifically provided for the recognition, maintenance and where appropriate enhancement of their coastal protection ability. But, due to community concerns over the undesired expansion of mangroves in some estuaries, the current version of the NZCPS (i.e. the NZCPS 2010) has removed any references to mangrove ecosystems (Orchard, 2011).

6.4.1. Guidance to local governments

The important role of local government in managing climate risks and planning for climate change adaptation is well recognised within the New Zealand’s legislative framework (Britton, et al., 2011; Lawrence, et al., 2015; Rouse, et al., 2017). As mentioned in Section 5.7, the Ministry for the Environment (MfE) has developed a series of guidelines to assist local governments assess the potential impacts of climate change on their functions and operations, adopt appropriate measures, develop and implement their climate adaptation policies and incorporate them into their decision-making frameworks (Hart, 2011). The non-statutory guidelines that have been provided by the Ministry for the Environment include:

- Climate change effects and impacts assessment: A guidance manual for local government in New Zealand, 2008 (Ministry for the Environment, 2008a) (Climate change projections for New Zealand updated in June 2016)

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79 Elevation; storm wave run-up; gradient; tsunami; lithology; landform; horizontal trend; short-term fluctuation
80 Hinterland: Land or region behind the beach and dune or ridge systems. Can be rising ground, sea cliffs, lagoon or wetlands (https://www.niwa.co.nz/coasts/nzcoast/tools-and-visualisations/coastal-terms-and-definitions) [Accessed Jan 2017].
- Coastal hazards and climate change: A guidance manual for local government in New Zealand (Ministry for the Environment, 2008b)
- Preparing for coastal change: A guide for local government in New Zealand (Ministry for the Environment, 2009)

The Ministry for the Environment is in the process of updating the current guidelines and is developing further guidance and national direction on implementing the resource management legislation and planning for climate change. These guidance manuals specify and explain the effects of climate change in New Zealand and discuss different adaptation strategies that local governments can undertake to manage the risk of climate change and associated coastal hazards. A framework for risk assessment based on the New Zealand Standard for Risk Management (AS/NZS4360) is also provided to support local governments’ decisions on managing the risks of climate change and coastal hazards (Ministry for the Environment, 2008b). The risk-based approach set out in the MfE guidance on coastal hazards and climate change (Ministry for the Environment, 2008b) also enables planners to link the climate change effects provision in Part 2 of the RMA with coastal development policies.

As part of the risk assessment process, local governments are encouraged to take a number of key considerations into account when identifying the risks from coastal hazards (Table 6.4-1). As the Table shows, the height and width of natural frontal barriers (e.g. dunes) need to be considered in identification of coastal hazard risks in coastal areas. Maintenance and enhancement of natural coastal defences and buffers through initiatives such as ‘Coastcare’ is also encouraged in the guidance as one of the requirements for effective regional and district rules (Ministry for the Environment, 2008b).

Table 6.4-1. Key criteria that local governments are encouraged to consider in identifying the risks from climate change and coastal hazards (Ministry for the Environment, 2008b)

<table>
<thead>
<tr>
<th>Key consideration</th>
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<td>Hazard sources and pathways</td>
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• Existence of particular environmental issues (e.g. significant mangroves, wetlands, seabird feeding or nesting areas, dune ecosystems)
• Level of access
• Cultural or heritage values of the site

To be effective in managing coastal hazards, the regional and district plans are also encouraged to include rules and mechanisms that (i) are based on the level of risk and vulnerability of the receiving environment, (ii) adopt a risk avoidance approach for new development and a risk reduction approach for existing development in coastal hazard zones, (iii) identify areas where structural coastal protection measures (or ‘hold the line’ options) can be appropriate long-term solutions. The guidance also suggests identifying coastal hazard zones based on the likelihood of hazard occurrences (e.g. from ‘exceptionally unlikely’ category with < 1% probability of occurrence to ‘virtually certain’ category with > 99% probability of occurrence) (Ministry for the Environment, 2008b).

Adopting risk avoidance strategies (i.e. avoiding new development in coastal hazard areas) is generally suggested in the guideline as a preferred measure for land use planning; however, a number of risk mitigation and risk transfer strategies are also discussed in the guidance. Possible measures for risk reduction and risk transfer in the guideline includes (i) Increased public information and awareness of coastal hazard risk through non-statutory (e.g. public talks and meetings, effective use of media) and statutory mechanisms (e.g. incorporating hazard and risk information in regional and district plans, Land Information Memoranda (LIM)83, placing notices of coastal hazard risk on property titles); (ii) Planned or managed retreat and its possible implementation methods (e.g. subsidies for relocation, property title covenants, transferable development rights); (iii) Emergency management and (iv) Efficient insurance system which plays a proactive role in hazard risk management. Consent notice or covenant attached to the title of individual properties in coastal hazard zones is suggested as a possible measure to increase public awareness and reduce financial liability on local authorities. However, it is also stated that they may not necessarily be effective as they are unlikely to affect the land value or change the buyers’ perception.

Regular/periodic update of hazard and risk information particularly those provided in LIM reports, also helps to increase public awareness. However, local councils need to be conscious of the potential liabilities associated with the provision of hazard information and ensure that the hazard information are accurate to the extent that would not make them unreasonably liable for any damage incurred as a result of their usage (Ministry for the Environment, 2008b). For example, significant public concerns were raised following the Kapiti Coast District Council incorporated coastal erosion risks to LIM reports for properties within ‘erosion hazard zones’ (1,800 coastal properties along the Kapiti coast) and placed restrictions on development and subdivision within these areas (Parliamentary Commissioner for the Environment, 2015). Consequently, the council was challenged in the High Court and finally decided to remove the ‘erosion hazard zones’ from LIM reports. The council was criticized for being unmindful of the potential impacts of its decision on ‘the value and marketability of coastal properties’ (Parliamentary Commissioner for the Environment, 2015, p. 65). The Court did not agree that no information should be placed on the LIM; rather, that the form of the information should be amended to reflect the draft and untested nature of the information placed by the council on the LIM (Weir v Kapiti Coast District Council, 2013).

As the potential barriers to managed retreat, the guideline briefly refers to public resistance, financial issues, existing use rights in coastal hazard areas and lack of clear transition mechanisms. Issues around existing use rights, as the key barrier to the effective implementation of managed retreat in particular, and to reducing the risk of coastal hazards and climate change in general are discussed in the next section.

83 Under the Local Government and Official Information and Meetings Act (LGOIMA) (1987), information about the potential natural hazards must be also included in the land information memorandum (LIM) which summaries all information that a council holds on a particular piece of land or building (s44A (2)). The natural hazards include, but are not limited to, potential erosion, avulsion, falling debris, subsidence, slippage, alluvion, or inundation (s 44A (2)).
Although the guideline refers to natural coastal buffers, it does not specify the coastal protection role of coastal wetlands. The only natural coastal feature that is considered is ‘dune systems’. The term ‘mangrove’ is only used once in the coastal hazards and climate change guidance (Ministry for the Environment, 2008b), where it is referred to as an example of particular environmental issues that need to be considered in identifying the risk to the receiving environment. The guidance also identifies sensitivity of a number of coastal landform types (including saltmarsh) to the different effects of climate change (including sea level rise, storm surge, precipitation, wave height and wave direction) (Ministry for the Environment, 2008b, p.33) but information about sensitivity of mangrove and seagrass ecosystems is missing.

The guidance strongly provides for adopting ‘no regrets’, ‘low regrets’ and ‘win-win’ adaptation strategies to mitigate the risks from climate change and coastal hazards. As indicated in the guideline (Ministry for the Environment, 2008b), ‘no regrets’ and ‘win-win’ strategies refer to policies, decisions and measures that help to manage and control the climate and coastal hazard risks while providing net social, economic and environmental benefits. They also refer to adaptation strategies that help manage several coastal hazard or climate related risks at once with at least no net adverse effect. Although not explicit, this approach provides an opportunity to consider ‘ecosystem-based adaptation’ which, as mentioned in Chapter 2, integrates ecosystem services into the overall climate change adaptation strategies to provide multiple co-benefits. The existing guidelines do not, however, provide directions on how adaptation measures can help deliver mitigation outcomes as their co-benefits.

Despite the existence of a relatively strong guidance, local authorities have no obligation to consider and apply the directions in the guidelines, incorporate them into the decision-making system, monitor effectiveness of the actions and report on the outcomes. This may undermine the effectiveness of the existing guidelines.

In addition to the MfE’s guidance manuals, there is a growing number of studies that have discussed different dimensions of coastal adaptation in New Zealand. Recent examples include a study by Blackett, et al., (2010) on the key factors affecting delivery of positive environmental outcomes with respect to management of coastal erosion in six case studies in New Zealand; a study by Britton, et al., (2011) on steps and requirements for effective coastal adaptation based on engaging communities and institutional decision-makers; a study by Lawrence, et al., (2013) that highlighted the importance of developing flexible adaptation strategies and introduced a simplified and cost-effective methodology to explore changes in flood frequencies under alternative climate change emission reduction scenarios rather than single ‘best estimate’ scenarios; a study by Lawrence, et al., (2015) on barriers/enablers that influence the ability of local government to deliver flexible and adaptive responses to climate risks that keep changing and evolving; a study by Reisinger, et al., (2015) on the role of managed retreat as an adaptation mechanism which can promote resilience in highly developed coastal areas; a study by Manning, et al., (2015) on the importance of adopting a dynamic approach in planning for and managing climate risks; and the most recent study by Rouse, et al., (2017) which puts together the findings from the previous research and provides recommendations for developing adaptation strategies which are flexible and can better respond to the future risks of climate change in New Zealand’s coastal areas.

These studies have highlighted the importance of developing adaptation pathways towards more flexible, adaptive and transformational coastal protection and adaptation strategies which enhance resilience to climate risks that keep changing and evolving. They also provide for recognising and considering the dynamic nature of climate risks and their associated uncertainty in planning for climate change adaptation in New Zealand’s coastal areas. This is consistent with the conclusions of global literature (as discussed in Section 2.4) that shifting to transformational adaptation measures is required to manage the future impacts of climate change. The New Zealand studies have also identified managed retreat as a long-term adaptation strategy which can enhance the flexibility of local response options (e.g. Reisinger, et al., 2015; Lawrence, et al., 2015; Manning, et al., 2015).
The findings of both global and local studies are considered in proposing the potential policy options and recommendations for the Auckland region in Chapter 8 of this research.

### 6.4.2. Managed retreat and issues around existing use rights

Managed retreat is defined ‘as any strategic decision to withdraw, relocate or abandon private or public assets that are at risk of being impacted by coastal hazards’ (Ministry for the Environment, 2008b, p.70). It is proposed as a fundamental risk reduction measure that needs to be considered for New Zealand’s coastal areas within the next few decades (Ministry for the Environment, 2008b). Significant increase in structural protection measures is identified as an alternative to managed retreat and can be appropriate in certain circumstances. However, structural measures, especially if implemented at a large scale, are often capital-intensive, can adversely affect natural coastal processes, amenity and public access values and are generally considered as being unsustainable in a long-term period (Ministry for the Environment, 2008b).

As mentioned earlier, the ‘existing use rights’ is identified as the key barrier to effective implementation of managed retreat in New Zealand. The limitations posed by the existing use rights can contribute to coastal squeeze and challenge the nature of resilience and adaptive management in the context of climate change (See Sections 2.4 and 2.7 for more details). Under Section 10 of the RMA, it is permissible for existing uses of land to contravene new district plan rules or proposed rules, within limits. However, regional councils can override the existing use rights in regional plan policies and rules, which must then be given effect by district councils (Ministry for the Environment, 2008b).

Currently, the use rights that are granted to a land use activity (e.g. new building) in coastal marine area are effectively permanent and cannot be extinguished. However, councils can control the use rights for alteration or extension to the existing buildings. It means that controlling the use rights in coastal hazard areas is not currently feasible if the existing buildings are not altered (Ministry for the Environment, 2008b). In fact, the current fee simple titles that are applied to the most coastal residential properties in New Zealand, provide the owners with exclusive permanent use of that title known as a property right. This system does not reflect the potential impermanence of the land in coastal hazard areas and therefore does not discourage investment in shorefront properties in areas subject to coastal hazards (Turbott, 2006).

Retreat (both voluntary and managed) is argued to be a challenging process in most of Auckland’s coastal areas given the highly urbanised coastal environment, high value of land and community resistance (Environment Waikato, 2006; Hart, 2011). A detailed study by Environment Waikato in 2006 (Environment Waikato, 2006), has discussed the potentials and challenges of managed retreat and its possible implementation options in Auckland, Waikato and Bay of Plenty Regions. Hart (2011) has also addressed the barriers and opportunities of managed retreat for the Auckland region. Both studies concluded that the absolute preference in living in a shorefront location compounded by the high and increasing value of coastal properties outweigh the risk of hazards and thus voluntary retreat is unlikely to occur in Auckland. Therefore, regional or district rules are required to enforce managed retreat over a long-term period. The studies also outlined a range of other reasons that make people refrain from leaving the coastal areas. These include:

- Lack of tangible evidence of risks or potential losses
- A perception that any likely damage will either be minor or can be prevented without significant cost; or can be recovered from insurance
- The low probability of actual loss compared to the high capital gain over a short period of time

Elevating buildings, as a micro-retreat option (Ministry for the Environment, 2008b) is currently considered as the only standard that must be complied with by new development activities in Auckland’s coastal areas at risk of inundation and 1m sea level rise. It also applies to the extensions to existing buildings.
6.4.3. Insurance

In New Zealand, the Earthquake Commission (EQC) natural disaster insurance known as ‘EQCover’ is a national insurance scheme that insures residential homes, land and contents against damage caused by natural disasters including earthquakes, natural landslips, volcanic eruptions, hydrothermal activities, tsunamis, fire and floods/storms (Earthquake Commission, 2016). However, it does not cover the cost of coastal erosion (Turbott, 2006). EQC may not cover the damage if properties have a hazard notice under section 36 of the Building Act 1991. Private insurers may also take a similar approach (Turbott, 2006). Premiums for EQCover are calculated at the same rate regardless of the location of the property and its potential risk level.

As at November 2016, the maximum annual premium (for one home and its contents) was $180 (+GST) which provides a maximum cover of $100,000 (+GST) for the home and $20,000 (+GST) for contents (EQC New Zealand Government, 2015). Property owners can also get top-up from private insurers to cover damage to their properties above the EQC limit (Earthquake Commission, 2016). Private insurers generally consult LIMs and district plans for information about the flood risk when making decisions on applications for insurance of properties against floods (Greater Wellington Regional Council, 2014). Insurers also consider information from national and international databases including history of flood events/flood claims in the area, level of hazard, frequency and duration of flooding and measures that are taken to reduce the risk of flood (Greater Wellington Regional Council, 2014).

6.5. Auckland’s policy responses to climate change and coastal hazards

Auckland is a member of the C40 Cities Climate Leadership Group (C40) since December 2015 (Auckland Council, 2015a). The C40 is a global network, currently consisting of 83 leading cities around the world that are working together to reduce GHG emissions and mitigate climate risks (C40 Cities, 2015). In addition, Auckland is participating in the Compact of Mayors initiative, which is the largest global initiative created collectively by city mayors from around the world (Auckland Council, 2014). The Auckland Council is also a signatory of a local government declaration on action on climate change jointly declared by local government leaders from across New Zealand (Auckland Council, 2014).

In Auckland, the provisions for climate change and coastal hazards are mainly included in the Auckland Plan (2012), the AUP (2016), the Auckland CDEM Group Plans (2011-2016 & 2016-2021) and the non-statutory Energy Resilience and Low Carbon Action Plan (Low Carbon Auckland, 2014). The relevant provisions are further discussed within the following sections.

6.5.1. Auckland Plan

The proposed directives and actions for adaptation to climate change in the Auckland Plan focus on a flexible risk-based approach which identifies opportunities and risks associated with climate change and intends to increase resilience to the impacts of climate change (Auckland Plan, Chapter 8). The Plan refers to minimum setbacks, raising floor levels, innovative building design, and managed retreats (if necessary) as examples of the tools for climate resilience and provides for progressive incorporation of these tools into the Unitary Plan over time (Chapter 8, Clause 526).

Strategic Direction 7 of the Auckland Plan sets out a priority for building resilience to natural hazards. To measure and monitor progress towards achieving this target, the Auckland Plan (Chapter 15, P.365) identifies a measure that is the ‘percentage of residents who understand their risk from natural hazards and are undertaking measures to mitigate or reduce their risk’. These types of qualitative variable are costly to quantify and generally prone to biased interpretations. This can lead to potential scrutiny of the

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84 The proposed reform package on EQCover provides for ‘no content insurance’ (EQC New Zealand Government, 2015).
Auckland Council’s liabilities if a major natural hazard results in irreversible damage to properties and lives in the areas prone to hazards.

This highlights the necessity of public awareness initiatives/programmes to improve the resident’s knowledge and information about the potential risks of climate change and natural hazards. Without sufficient knowledge to make sensible decisions, relying on residents’ understanding of their own exposure to risk might be of limited use.

The Auckland Plan does not include consistent provisions associated with future housing development in coastal areas vulnerable to natural hazards. Clause 467\(^{85}\) of the plan sets out a strong direction to avoid future residential development in areas at risk of natural hazards, whereas Directive 7.15\(^{86}\) allows future development in at risk areas subject to the acceptance and management of the risks. The Development Strategy of the Auckland Plan also enables more opportunities for people to live ‘near the coast’ (Auckland Plan, Section D, Clause 103).

The carbon sequestration functions of forestry and planting programmes are highlighted in the Auckland Plan (Clause 501) which is a reflection of the significance of these ecosystems under the NZ ETS. The Auckland Plan also provides for protection and enhancement of Auckland’s green infrastructure\(^{87}\) (GI) networks as a measure for climate change mitigation (Directive 8.2). However, it only refers to green roofs and urban allotments as examples of GI in the Auckland region (Clause 526; Box 8.2). Protection, promotion and enhancement of the quality of ‘indigenous ecosystems’ is also indicated in the Auckland Plan as measures that can contribute to both adaptation to and mitigation of climate change (Clause 526). However, the Plan does not specify the type of indigenous ecosystems that can provide mitigation and adaptation services and provides no information on how they could contribute to climate change adaptation.

6.5.2. The Auckland Unitary Plan

Responding to climate change was identified as one of the issues of regional significance in the PAUP (Chapter B, 1). The AUP identifies ‘environmental risk’ as one of the significant resource management issues for the Auckland region (Chapter B, B1.4(9)). The broad category of environmental risk includes issues such as natural hazards and climate change together with a number of other risks\(^{88}\).

Despite Auckland’s engagement in the global initiatives to mitigate climate change, the AUP does not include specific provisions on climate change mitigation where it refers to objectives and policies addressing natural hazards and climate change (Chapter B, B10.2). Whereas, the Proposed AUP (PAUP) provided for addressing both climate change mitigation and adaptation (Chapter B, 9). For example, increasing carbon sinks as well as energy efficiency and the greater use of renewable energies was included in the PAUP as a policy to mitigate climate change (PAUP, Chapter B, 9). But the terms ‘carbon sink’ and ‘carbon sequestration’ are missing in the RPS of the AUP.

Under the provisions of the AUP (Chapter B, B10.2), protection and restoration of sand dune and vegetation on sand dune are specifically encouraged for reducing the impacts of coastal hazards and erosion. Whereas, the role of coastal wetland vegetation, particularly mangroves in protection against storm surges and sea level rise are not recognised in the current policies. As part of the nature-based measure to address coastal hazards, the AUP only provides for beach nourishment and dune stabilisation and identifies them as ‘permitted activities’ as long as they comply with the associated standards (Chapter...

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85 ‘... future housing development must be located away from natural hazards’ (Auckland Plan, Clause 467)
86 ‘Avoid placing communities and critical infrastructure and lifeline utilities in locations at risk from natural hazards, unless the risks are manageable and acceptable’ (Auckland Plan, Directive 7.15).
87 The Auckland Plan (Clause 500) refers to green infrastructure (GI) as ‘a strategically planned and delivered network of high-quality green spaces and other environmental features that deliver ecological services and quality of life benefits required by the communities they serve’
88 Hazardous substances, contaminated land and genetically modified organisms.
Activities in hazard areas are also assessed in terms of the extent to which they include non-structural solutions to protect infrastructure from the hazard and resulting adverse effects, among other things (AUP, Chapter E, E36.8.2(16)(d)). However, as an example of ‘non-structural solutions’, the AUP only refers to ‘planting or the retention or enhancement of natural landform buffers’ (AUP, Chapter E, E36.9(2)(g)). There is no clear reference to enhancement or restoration of coastal wetlands.

Under the AUP, mangroves cannot be removed in areas where they mitigate coastal erosion. There is currently no map which shows the areas susceptible to coastal erosion in the Auckland region; however, these areas may in part cover the areas at risk of sea level rise and inundation. Apart from mangroves, there is no explicit statement in the Auckland Plan accounting for the climate change values of saltmarsh and seagrass ecosystems. The AUP provides for protection of saltmarsh and seagrass ecosystems, however, only in terms of their role in supporting biodiversity.

Under the current provisions of the AUP (2016), new housing development in areas subject to coastal storm inundation of 1% AEP\(^89\) plus 1m sea level rise is permitted if the finished floor level of habitable rooms is above the flood elevation and sea level rise (Chapter E, E36, Table E36.4.1). The new rules are presumably set to facilitate development in coastal areas at risk of storm inundation and sea level rise as the requirement regarding the structural integrity of buildings during storm events (that had been previously included in the PAUP) is not included in the new version of AUP.

Information about the infrastructure failure in the Auckland region (e.g. electricity outage, damage to water supply facilities, road disruption etc.) that can be caused by various drivers including natural hazards is publicly available through the Auckland Council website.\(^90\) The AUP provisions on natural hazards and flooding (Chapter E36) provides for assessing the risk of natural hazards (including the likely effects of climate change) to people, property, infrastructure and the environment when making decisions for subdivision, use and development (Chapter E, E36, E36.2.). The AUP allows placing new infrastructure in areas subject to natural hazards, but provides for an adaptive management response taking account of a longer-term rise in sea level (E36.3(8)).

The RPS provisions on the coastal environment, as indicated in the AUP, (Chapter B, B8) provide a strong mandate for appropriate setback of development from the coastal marine environment to (i) protect public open space values and access (Chapter B, B8.4.2(c)), (ii) to protect the natural character and amenity values of the coastal environment (Chapter B, B8.3.2(7)) and (iii) to not compromise the ability of future generations to have access to and along the coast Chapter B, B8.4.2(d)). It is also stated that the future impacts of coastal erosion and sea level rise need to be considered when identifying appropriate setback from the coastal edge (Chapter B, B8.6. p.14). The AUP rules related to the ‘general coastal marine zone’ (Chapter F, F2) therefore require a setback of development from mean high water springs in order to avoid being affected by erosion, coastal hazards and sea level rise (Chapter F, F2, F2.23.2(4)(xi)).

Compared to the PAUP, the AUP provides more specific zoning categories that better reflect the vulnerability of coastal and inland areas to coastal hazards and flooding. The new zoning applied to the AUP rules for natural hazards management, is however more hazard/risk specific compared to the previous PAUP provisions. The current provisions classify activities within different hazard zones including:

1. Areas at risk of erosion,
2. Areas at risk of coastal storm inundation 1% annual exceedance probability (AEP)
3. Areas at risk of coastal storm inundation 1% annual exceedance probability (AEP) plus 1 metre sea level rise,
4. Areas at risk of 1% annual exceedance probability (AEP) floodplain,

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\(^89\) Annual Exceedance Probability
(v) Overland flow paths, and;
(vi) Areas at risk of land instability.

The coastal inundation maps prepared by the Auckland Council (Explained in Chapter 5, Section 5.7) are good examples of risk communication actions by the council as a local authority. As discussed in chapter 5 (Section 5.7), despite the risk of storm inundation, erosion, land subsidence and sea level rise in coastal areas, residential dwelling and commercial facilities have been and are still being allowed to be constructed very close to the sea (e.g. Orewa beach, Omaha beach/Whangateau Harbour).

As reported by the Auckland Council (Auckland Council, 2015b), a 640-metre-long seawall that will cost approximately $5 million91 ($7,800 per metre) is planned to be constructed along the northern end of the Orewa beach to protect the area against storm damage and the future effects of sea level rise. It is also stated that low-level planting will be used to enhance the seawall and help with sand deposition. As mentioned in Chapter 5 (Section 5.7), Omaha beach (Whangateau Harbour) is also at risk of storm and inundation and in some parts of the estuary mangroves provide a buffer against erosion and inundation. However, coastal communities generally support mangrove removal (Auckland Council, 2016c) and there is also evidence of illegal mangrove removal.

Policy 3 of the NZCPS explicitly provides for adopting a precautionary approach in management of coastal resources vulnerable to the effects of climate change, while the AUP makes no rule to identify and protect coastal wetland habitats that are adversely affected by the cumulative effects of land use change and climate change.

6.5.3. Auckland CDEM Group Plans (2011-2016 & 2016-2021)

The first Auckland CDEM Group Plan (2011-2016) had no references to the role of nature-based initiatives to mitigate the risk of coastal hazards. However, the new statutory CDEM Group Plan (2016-2021) (Auckland Civil Defence, 2016) titled ‘Working together to build a resilient Auckland’ includes ‘resilience’ as the Auckland-specific strategic goal in addition to the 4Rs (Reduction, Readiness, Response, Recover) proposed in the CDEM Act 2002.

The resilience-based approach in the CDEM group plan is consistent with the Strategic Direction 7 of the Auckland Plan that sets out a priority for building resilience to natural hazards (Auckland Civil Defence, 2016). The risk assessment framework proposed for the Auckland region is based on identifying the likelihood and consequences of an event and is similar to the risk assessment methodology provided in the non-statutory coastal hazards and climate change guidance manuals (Ministry for the Environment, 2008 a&b) as discussed before. The semi-quantitative risk-assessment methodology will consider the exposure and vulnerability of each of the elements at risk including the ‘four well-beings of sustainability’ (i.e. social, environmental, cultural and economic). However, the plan does not clearly specify criteria that are used to identify vulnerability of coastal areas to coastal hazards and the effects of climate change such as sea level rise.

As indicated in the CDEM Plan (2016-2021), Auckland Council Natural Hazards Risk Management Action Plan (NHRMAP) that is currently being developed will provide best practice risk reduction and resilience outcomes and include more details about how the potential risks from natural hazards can be greatly reduced (Auckland Civil Defence, 2016).

The Auckland’s CDEM Plan (2016-2021) refers to restoration of wetlands in line with restoration of catchments, soil, vegetation and other natural systems as an integral part of hazard management. But, it does not specify how this approach to restoration is linked to the current land use planning policies and what type of wetlands are considered for restoration as part of the hazard management strategy. The approach, however, is a step forward to integrate hazard management strategies with wetland restoration

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activities, though the role of coastal wetlands is not explicitly recognised. The plan does not refer to natural hazard management strategies that can provide multiple benefits (‘no regret’ and ‘win-win’ strategies) as proposed in the non-statutory coastal hazards and climate change guidance. Nonetheless, it provides for incorporating sustainable, energy resilient, low carbon and adaptive land planning and management initiatives into CDEM activities, as one of the actions proposed to improve Auckland’s resilience to natural hazards.

6.5.4. Auckland’s Energy Resilience and Low Carbon Action Plan

The Auckland’s Energy Resilience and Low Carbon Action Plan (Low Carbon Plan) (Auckland Council, 2014) is a non-statutory plan developed to assist in achieving the Auckland Plan’s strategic directions on reducing GHGs emissions and improving the city’s energy resilience. It is also aimed to work with, and contribute to the outcomes of the AUP (Auckland Council, 2014). The Low Carbon Plan mainly focuses on reducing Auckland’s GHG emissions through developing an energy-efficient transport system, promoting greener buildings and greater use of renewable energies.

The carbon sequestration potential of coastal and marine ecosystems is recognised in the Low Carbon Plan; however, the actions to enhance this capacity are planned to be completed by 2040 (Figure 6.5-1). The Low Carbon Plan refers to the limited understanding of the role of coastal and marine ecosystems for carbon sequestration and provides for exploring marine sequestration potential. To fill this gap, the Plan identifies two actions (Table 6.5-1) to examine coastal and marine carbon sinks and the implications of marine preservation on GHG reductions and emissions (Auckland Council, 2014).

![Figure 6.5-1. Pathway to 2040 for exploring marine sequestration potential (Auckland Council, 2014)](image)

<table>
<thead>
<tr>
<th>Exploring marine sequestration potential</th>
<th>Delivery lead and contributors</th>
<th>Completion date</th>
<th>Status</th>
</tr>
</thead>
<tbody>
<tr>
<td>Action 21: Evaluate existing research on the potential of coastal and marine ecosystems to sequester carbon. New research programmes are supported to fill any research gaps. (E)</td>
<td>Auckland Council, Research institutions</td>
<td>2020</td>
<td>Started</td>
</tr>
<tr>
<td>Action 22: Ensure the Marine Spatial Plan provides for enhanced coastal and marine sequestration. (E)</td>
<td>Auckland Council, Research Institutions</td>
<td>2030</td>
<td>Started</td>
</tr>
</tbody>
</table>

Table 6.5-1. Actions towards exploring marine sequestration potential (Auckland Council, 2014)

Note: E means ‘enabling’ actions

Despite this positive recognition of the carbon sink potential of coastal and marine ecosystems, the fact that the Low Carbon Plan is a non-statutory document under the Auckland Plan poses a potential challenge for its implementation through the AUP. This is because the Auckland Plan is prepared under the LGA; while the AUP is a mandatory document under the RMA that precludes involvement of local governments...
in climate change mitigation. This means that implementation of the findings of the Low Carbon Plan (except for the use of renewable energies) is effectively unlikely to happen through the AUP.

### 6.5.5. Responses from the questionnaire survey

As mentioned in Chapter 3 (Section 3.4.2), a questionnaire survey was conducted to gain an understanding of the knowledge and perception of the participants about ecosystem services of coastal wetlands and their management issues. This section summarizes the results and presents the responses that were selected by majority of participants. Details are provided in Appendix 6.

With respect to the first question about ecosystem services of mangrove ecosystems in Auckland, respondents gave ‘very high’\(^2\) importance to the role of mangroves in storm protection and erosion control, as well as nutrient cycling and water purification (Table 6.5.2). Mangroves were also perceived to have ‘high’ values for carbon sequestration and storage (CS&S) and biodiversity protection. However, majority of the respondents gave ‘moderate’ score to the level of importance given in the PAUP (2013) to storm protection and erosion control values of mangroves. As responses indicated, the role of mangroves in CS&S and nutrient/water cycling was not considered in the PAUP.

Table 6.5.2. A summary of responses to Question 1 of the questionnaire regarding the degree of importance of different ecosystem services provided by coastal wetland categories (responses provided by the highest number of respondents are only presented in the table)

<table>
<thead>
<tr>
<th>Habitat</th>
<th>(i) Degree of importance (As you would assign to each service)</th>
<th>(ii) Degree of importance (Given in the PAUP)</th>
</tr>
</thead>
<tbody>
<tr>
<td>(A) Provisioning Services:</td>
<td>Mangroves: Low (3 of 8)</td>
<td>Not considered in the PAUP (3 of 8)</td>
</tr>
<tr>
<td>Production of food/medicine for human</td>
<td>Saltmarshes: Low (4 of 8)</td>
<td>Not considered in the PAUP (5 of 8)</td>
</tr>
<tr>
<td></td>
<td>Seagrasses: Low (4 of 8)</td>
<td>Not considered in the PAUP (5 of 8)</td>
</tr>
<tr>
<td>(B) Regulating services:</td>
<td>Mangroves: High (5 of 8)</td>
<td>Not considered in the PAUP (4 of 8)</td>
</tr>
<tr>
<td>Carbon Storage and Sequestration</td>
<td>Saltmarshes: High (3 of 8)</td>
<td>Not considered in the PAUP (4 of 8)</td>
</tr>
<tr>
<td></td>
<td>Seagrasses: Low (4 of 8)</td>
<td>Not considered in the PAUP (4 of 8)</td>
</tr>
<tr>
<td>(C) Regulating services:</td>
<td>Mangroves: Very high (4 of 8)</td>
<td>Moderate (3 of 8)</td>
</tr>
<tr>
<td>Storm Protection</td>
<td>Saltmarshes: High (3 of 8)</td>
<td>Not considered in the PAUP (2 of 7)</td>
</tr>
<tr>
<td></td>
<td>Seagrasses: Low (3 of 8)</td>
<td>Not considered in the PAUP (2 of 7)</td>
</tr>
<tr>
<td>(D) Regulating services:</td>
<td>Mangroves: Very high (4 of 8)</td>
<td>Moderate (3 of 8)</td>
</tr>
<tr>
<td>Erosion Control</td>
<td>Saltmarshes: Very high (4 of 8)</td>
<td>Not considered in the PAUP (2 of 7)</td>
</tr>
<tr>
<td></td>
<td>Seagrasses: Low: Very low (2 of 8 each)</td>
<td>Not considered in the PAUP (3 of 8)</td>
</tr>
<tr>
<td>(E) Supporting Services:</td>
<td>Mangroves: Very high (4 of 8)</td>
<td>Not considered in the PAUP (4 of 8)</td>
</tr>
<tr>
<td>(biogeochemical cycles, e.g.</td>
<td>Saltmarshes: High (3 of 8)</td>
<td>Not considered in the PAUP (4 of 8)</td>
</tr>
<tr>
<td>Nutrient cycling, water cycling</td>
<td>Seagrasses: High: Moderate (3 of 8 each)</td>
<td>Not considered in the PAUP (4 of 8)</td>
</tr>
<tr>
<td>(F) Supporting Services:</td>
<td>Mangroves: High (5 of 8)</td>
<td>Low (3 of 8)</td>
</tr>
<tr>
<td>(e.g. Support threatened plant</td>
<td>Saltmarshes: High (6 of 8)</td>
<td>Very high and low (2 of 8 each)</td>
</tr>
<tr>
<td>and animal species, supporting</td>
<td>Seagrasses: High (6 of 8)</td>
<td>Low (3 of 8)</td>
</tr>
<tr>
<td>biodiversity)</td>
<td></td>
<td>Not considered in the PAUP (2 of 8)</td>
</tr>
<tr>
<td>(G) Cultural Services (e.g.</td>
<td>Mangroves: High/Low (3 of 7 each)</td>
<td>Very low (4 of 8)</td>
</tr>
<tr>
<td>Social, religious, recreational</td>
<td>Saltmarshes: High (4 of 7)</td>
<td>Not considered in the PAUP (3 of 8)</td>
</tr>
<tr>
<td></td>
<td>Seagrasses: High (4 of 7)</td>
<td>Not considered in the PAUP (3 of 8)</td>
</tr>
<tr>
<td>(H) Other (Please specify):</td>
<td>Mangroves: Moderate (1 of 1)</td>
<td>Not considered in the PAUP (1 of 1)</td>
</tr>
<tr>
<td>juvenile fish nursery</td>
<td>Saltmarshes: Very low (1 of 1)</td>
<td>Not considered in the PAUP (1 of 1)</td>
</tr>
<tr>
<td></td>
<td>Seagrasses: High (1 of 1)</td>
<td>Low (1 of 1)</td>
</tr>
</tbody>
</table>

For saltmarsh ecosystems, respondents gave ‘high’ importance to CS&S, storm protection, nutrient and water cycling, biodiversity protection and cultural values. However, respondents believed that except for biodiversity protection, none of these services has been considered in the PAUP. Respondents gave ‘high’ importance to biodiversity protection, juvenile fish nursery and cultural services provided by seagarsss ecosystems. However, they indicated that cultural values of these ecosystems are not considered in the PAUP.

\(^2\) The qualitative rating scale includes ‘very high’, ‘high’, ‘moderate’, ‘low’ and ‘very low’
In relation to question 2, the majority of respondents identified the group of ‘relevant academics and researchers’ as entities with ‘high’ level of knowledge about mitigation and adaptation services of coastal wetlands; while, the three groups of ‘public’, ‘developers’ and ‘councilors’ were identified by participants as entities with ‘low’ level of knowledge about the climate change values of coastal wetlands (Table 6.5-3).

Table 6.5-3. A summary of responses to Question 2 of the questionnaire regarding the level of knowledge of mitigation and adaptation services of coastal wetlands by various relevant groups (responses provided by the highest number of respondents are only presented in the table)

<table>
<thead>
<tr>
<th>Group</th>
<th>Mitigation services</th>
<th>Adaptation services</th>
</tr>
</thead>
<tbody>
<tr>
<td>(A) Councillors</td>
<td>Low (6 of 8)</td>
<td>Low (4 of 8)</td>
</tr>
<tr>
<td>(B) Planners (Council officers)</td>
<td>Low (4 of 8)</td>
<td>Medium/Low (3 of 8 each)</td>
</tr>
<tr>
<td>(C) Council Controlled Organisations</td>
<td>Medium/Low (3 of 8 each)</td>
<td>Low (4 of 8)</td>
</tr>
<tr>
<td>(D) Developers</td>
<td>Low (6 of 8)</td>
<td>Low (6 of 8)</td>
</tr>
<tr>
<td>(E) Relevant academics and researchers</td>
<td>High (7 of 8)</td>
<td>High (7 of 8)</td>
</tr>
<tr>
<td>(F) Relevant consultants</td>
<td>High/Low (3 of 8 each)</td>
<td>High/Medium (3 of 8 each)</td>
</tr>
<tr>
<td>(G) Public</td>
<td>Low (8 of 8)</td>
<td>Low (7 of 8)</td>
</tr>
<tr>
<td>(H) Harbour care or Wai care groups¹</td>
<td>Medium (6 of 8)</td>
<td>Medium (4 of 8)</td>
</tr>
<tr>
<td>(I) Other Non-Governmental Organisations</td>
<td>Medium (5 of 8)</td>
<td>Medium (5 of 8)</td>
</tr>
<tr>
<td>(J) Other (Please specify): iwi</td>
<td>High (1 of 1)</td>
<td>High (1 of 1)</td>
</tr>
</tbody>
</table>

With respect to the potential of different habitat types for carbon storage in biomass, ‘mangroves’ were ranked third after ‘natural forest’ (First) and ‘planted forest’ (Second). ‘Mangroves’ were also ranked third for their biomass carbon sequestration after ‘planted forest’ and ‘vegetated non-coastal wetlands’ (First) and ‘natural forest’ (Second). ‘Natural forest’ and ‘vegetated non-coastal wetlands’ were ranked first for their potential for ‘soil carbon storage’ and ‘soil carbon sequestration’ respectively. Respondents also believed that there is no or insufficient information about (i) biomass carbon storage by mangroves, saltmarshes and seagrasses and (ii) biomass carbon sequestration by saltmarshes and seagrasses (See Appendix 6, answer to Question 3).

The majority of respondents (5 out of 7) ‘strongly’ agreed that compared to other ecosystems, in terms of their size and climate change services, coastal wetlands in the Auckland region are important elements of the Auckland urban landscape. 4 out of 7 respondents also agreed that an effective climate change mitigation and adaptation policy in the Auckland region would include controlling land use in connection with protection and/or restoration of coastal wetlands. They reasoned that coastal wetlands effectively contribute to climate change mitigation and adaptation and future changes in land use will threaten these ecosystems and makes them more vulnerable to the impacts of land-use change and coastal hazards. However, 3 out of 8 respondents gave higher importance to the role of coastal wetlands in climate change mitigation and adaptation (Appendix 6, answers to Questions 4 & 5).

The majority of respondents (4 out of 7) believed that the PAUP has given higher importance to land use change policies that allow further land development in coastal areas. However, 3 out of 7 respondents believed that the PAUP has given a higher importance to protection of coastal ecosystems as significant habitats for indigenous flora and fauna, and as important sites for Mana Whenua (Appendix 6, answer to Question 6).

Eutrophication and increased sedimentation rates due to land use change in catchments as well as water pollution and reclamation were identified as the most important threats to the coastal wetlands in the Auckland region (Appendix 6, answers to Question 7).

With respect to parameters that need to be considered in development of coastal areas, majority of respondents (6 out of 7) gave ‘high’ importance to vulnerability of low-lying coastal areas to the effects of climate change (risk assessment approach) and 4 out of 8 believed that this parameter is given ‘low’ importance in the PAUP. Ability to retreat and servicing issues were identified by one respondent as the
parameters with ‘high’ and ‘medium’ importance, respectively. However, ability to retreat is not
considered in the PAUP and servicing is given a low importance (Appendix 6, answers to Question 8).

The following issues are identified by the respondents as the most fundamental problems influencing
intention and capacity of the local government to contribute to the climate change mitigation (Appendix 6,
answers to Question 9):

- Central government management of greenhouse gas emissions initiatives
- Legislative change
- Lack of independence to take action at local, regional and national level
- General lack of public interest and action in forcing their leaders to change things
- Agency capture by the development community

Participants in the survey identified the following ways in which the planning process in Auckland can
take into account the climate change services of coastal wetlands (Appendix 6, answers to Question 4.1):

- ‘Classify as significant natural areas or some other form of protection rules’
- ‘Anyway’
- ‘Many areas in wider Auckland will be flooded with sea level rise, so the region needs to
understand wetland services to take advantage of these’
- ‘By protecting them’
- ‘The protection of coastal wetlands is acknowledged in planning documents but seldom given
effect by the rules. Often a lack of integration across different rules e.g. zoning and ecological
protection overlays result in poor protection of these areas. A lack of a clear biodiversity
mitigation and offset guideline also inhibits best practice for compensating loss of these
ecosystems’
- ‘These areas need to be mapped, potential at-risk and expansion zones identified, and rules should
protect these areas (with consent assessment criteria related to these functions)’
- ‘By delineating a wide ‘no go’ zone around the entire coastline where long-term, private property
rights are restricted, the land is kept in public ownership and the regeneration of natural
ecosystems and processes is actively supported and encouraged. Obviously, this is a long-term
project! One of my great concerns is that the very rich (who tend to own property along the coast)
will begin to force the rest of us to pay for very expensive and ultimately (King Canute like) doomed
to failure ‘solutions’ to protect their private assets… After all a cliff is just an eroding hill. Why
build your house at the top of one and expect the hill to stop eroding?’

They generally believed that a response needs to be taken at national, regional and local scales as all levels
have a role to play with clear direction at national and regional levels and action at local level (Appendix
6, answers to Question 4.2).

6.6. Key policy and planning challenges and opportunities

This section discusses several key issues and opportunities that are identified through the review of the
current policies and plans in New Zealand and Auckland in the previous sections of this chapter. The
review primarily focused on the current state of resource management of coastal wetlands in general and
mangroves in particular in relation to their climate change services.

6.6.1. The issue of mangrove expansion

The nature of the resource management issues associated with mangroves is different from the two other
intertidal coastal wetlands (i.e. saltmarshes and seagrasses) in the Auckland region. While the three
ecosystems are generally valued, at both national and regional level, for their biodiversity and habitat functions, the main planning issue with the Auckland’s mangroves is their expansion and colonisation. This issue is visibly different from the common resource management problems where the main concern with respect to natural resources is typically their shrinkage rather than expansion. Consequently, unlike the global mangrove restoration initiatives which aim to enhance mangrove areas, ‘mangrove restoration’ activities in New Zealand aim to remove mangroves to ‘restore’ the prior sandier, unvegetated states of estuaries.  

The legislative framework in New Zealand generally supports sustainable management, use and protection of the coastal natural resources. However, the issue of mangrove expansion has largely been within the jurisdictions of the local governments. Several academic and research institutes, advocacy groups, local communities and businesses have been actively involved in the debates around the problem of mangrove expansion. Negative impression about the expansion of mangroves and its effects on coastal communities and properties has resulted in a strong public resistance against their protection and enhancement.

The resolutions in the most recent version of the AUP have been the result of relatively long debate among proponents and opponents of mangrove removal in Auckland. The AUP recognises ecological values of mangroves, but still allows removal of the established mangrove stands although in a more restrictive manner compared to its previous versions. The restrictive position of the AUP may potentially reduce the speed and possibly the extent of mangrove removal, but the lack of knowledge about ecological values of mangroves and coastal wetlands in general and the paucity of evidence about the positive impacts of mangrove removal generates concerns about the long-term sustainability of these ecosystems.

Of particular importance for this research are the potential climate change mitigation and adaptation benefits of the coastal wetlands including mangroves that are kept out of the current policy agendas. This is discussed further below.

### 6.6.2. Policy positions on climate change services of coastal wetlands

Chapter 5 demonstrated that coastal wetlands including mangroves can considerably contribute to climate change mitigation and adaptation by protecting coastlines against sea level rise and storm events and storing carbon mainly in their sediments.

The current legislative framework is almost ignorant of the contribution of coastal ecosystems (including coastal wetlands) to climate change mitigation, and underestimates the adaptation benefits of these ecosystems. Auckland has a voluntary (non-mandatory) Energy Resilience and Low Carbon Action Plan that has a long-term goal of mainstreaming marine ecosystems as sinks of carbon. But this will be realised only once sufficient knowledge about the carbon sequestration capacity of the marine ecosystems is developed. By that time, there will be a gradual reduction in the mangrove stocks, which means there will be some CO\(_2\) emissions due to mangrove removal.

In addition, it is unlikely that the Low Carbon Action Plan could influence the policies and rules in the AUP, as the RMA precludes the local governments from involving in climate change mitigation. In fact, a key issue especially with the current legislative and governance framework for climate change is the divergence in the distribution of responsibilities for mitigation and adaptation between local and central government in New Zealand.

Coastal wetlands are not currently part of the ETS or other carbon market mechanisms in New Zealand, which is effectively a reflection of the lack of global mandate and requirement to incorporate emissions and removals from coastal ecosystems in climate mitigation policies. The 2013 IPCC supplement on wetlands (IPCC, 2014c) provides an opportunity for incorporating blue carbon in national and local climate change policies.

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change policies. However, lack of sufficient empirical data is an obstacle for reporting on emissions due to wetland removal and the amount of carbon sequestration by wetlands in national GHG inventories.

As mentioned in Chapter 5, global interest in ‘blue carbon’ has been growing significantly over the last decade and a number of global initiatives are specifically developed to promote restoration and protection of blue carbon ecosystems including mangroves, saltmarshes and seagrasses. Auckland’s membership in C40 is a potential opportunity for knowledge exchange, particularly regarding the climate change mitigation values of coastal wetlands.

The estimates in this research (Chapter 5) indicated that Auckland mangroves and saltmarsh sequester about 44,000 tonnes carbon (CO$_2$) per year (net sequestration$^{94}$). Based on a SCC value estimated at US$220 per tonne (Moore & Diaz, 2015) this is worth about US$ 9.6 million per year. This research also found that depending on severity (magnitude) of clearance, removing one hectare of mangrove and one hectare of saltmarsh could result in a carbon loss between 400 and 1,600 tonnes of CO$_2$, averaging 1000. Using the same SCC value (Moore & Diaz, 2015), this is equivalent to about US$220,000.

The importance of ecosystems as natural defences against coastal hazards and the need to ‘protect’ and ‘enhance’ natural defences are specifically addressed in the mandatory national coastal policy statement (NZCPS). However, unlike the NZCPS 1994 that had explicitly identified mangroves as natural defences (Policy 3.4.3), mangroves are excluded from the list of natural defence ecosystems in the NZCPS 2010 (Policy 26).

The current approach to coastal protection in the AUP is focusing on some form of nature based solutions including plantation on sand dunes and beach nourishment, but does not provide for restoration of coastal wetlands. This approach is institutionalised in the form of community groups who support and encourage planting on sand dunes largely because of their more tangible amenity benefits. The AUP also recognises that mangroves provide erosion control benefits and supports their protection where they control coastal erosion. However, there is no reference to the protective role of coastal wetlands in mitigating inland inundation and adapting to sea level rise.

The Strategic Environmental Assessment (SEA) approach, common in Europe and Canada, is allowed under the RMA, although there is no formal framework for undertaking SEA. Section 32 of the RMA provides for identification and evaluation of the environmental, social, cultural and economic effects of policies through a range of evaluation methods such as cost-benefit assessment. There is, however, no reference to the ‘co-benefits’ and emission implications of different policies particularly within the context of climate change.

6.6.3. Implication of the current provisions on mangrove ecosystems

The current policies are at odds with the notion of no-net-loss and are expected to result in depletion of mangrove resources over time. There is no limit for the acceptable level of mangrove removal, which is going to be determined and decided on a case-by-case basis, meaning there is no clear indication for where the no-net-loss should be set for mangroves and what ecological values of mangroves are accounted for in an offset scheme.

The initially set date of 1996 in the PAUP as a baseline was debated through the development of the plan and eventually taken out of its most recent version. Arguably, using a date as a baseline could only be appropriate if there is certainty about the extent and patterns of mangrove expansion and a sound knowledge about the ecology of mangroves and their interactions with their surrounding landscape. Such an acceptably complete knowledge is generally unattainable in practice and there is always uncertainty in science about effects of changes to ecosystems and landscapes. Considering the dynamic nature of the changes in coastal ecosystems including mangroves, setting a date also implies that ecological characters

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94 Gross carbon sequestration less methane emission
of the coastal wetlands (including ecosystem functions and services at a certain point in time and their natural variations over time) are known and can be used to determine a limit for the level of acceptable change. This was not the case in Auckland.

In the absence of a sound scientific basis and rationale, the current policies in the AUP require case by case assessment of applications for mangrove removal in accordance with the conditions set out for the removal activities within coastal marine areas. The discretionary status of the mangrove removal activities in the AUP, is an indication of the AUP’s restrictive position on mangrove removal and a potentially positive opportunity for careful management of the remaining mangrove stocks.

However, there is a lack of clarity in the existing policies about the approach for identification and protection of coastal wetlands that will be adversely affected by the effects of climate change. As discussed in Chapter 5, while wetlands adapt, they are also limited by space. As sea level rises, wetlands will migrate landward and this will be affected in areas where coastal wetlands are limited by coastal properties and infrastructure. As discussed in Section 5.7, changes in sea level rise, frequency and magnitude of storm events and sediment availability affect long-term existence of coastal wetlands in Auckland and therefore need to be considered in planning for the use of these ecosystems for the purpose of climate change mitigation and adaptation. According to the findings of a study by Lovelock, et al., (2015), mangrove forests in New Zealand (including Auckland) can keep pace with increase in the sea level by 48cm and 63cm by 2100; however, due to the dynamic nature of climate risks, long-term sustainability of coastal wetlands needs to be identified under the higher projections of sea level rise and more frequent storm events.

The current limit for sedimentation rate in estuaries across the HGMP (2mm above the baseline rate by 2050) and establishment of the FMUs are efforts to control and manage the cumulative effects of land use activities on freshwater and marine ecosystems. It is, however, necessary to ensure that it does not adversely affect the long-term sustainability of mangrove and saltmarsh ecosystems.

The provisions of the biodiversity offsetting guidance and the AUP regarding offsetting the residual adverse effects of development activities on biodiversity, provide an opportunity which can drive conservation and restoration of coastal wetlands. However, as mentioned, there is still not a clear link between mangrove removal and offset projects. Expansion of mangroves and the current public view could potentially challenge any initiative in relation to enhancement and restoration of mangroves under the existing offsetting approach.

### 6.6.4. Nature-based approaches in the current provisions

Under the current AUP rules, development in coastal areas at risk of storm inundation (as a result of 1% AEP storm) and 1m sea level rise is permitted if the building elevation is higher than the flood elevation and 1m sea level rise (AUP, Chapter E, E36, p.12 & Table E36.4.1). The developers seeking to build in these areas are not legally required to use flood-resistant materials or contribute to enhancement or restoration of coastal wetlands as an ecosystem-based adaptation measure. The only mandatory requirement is for developers and property owners to elevate their building floors (AUP, Chapter E, E36, Standard E36.6.1.1).

There is a general reference to ‘planting’ and ‘retention or enhancement of natural landform buffers’ as possible non-structural measures, but there is no explicit reference to coastal wetlands. As noted earlier, measures such as ‘beach nourishment’ and ‘dune restoration’ are also specifically included within the list of activities which aim to provide defence against coastal hazards.

The guidance manuals prepared by the Ministry of the Environment provide useful information about different non-structural adaptation strategies that local governments can adopt to decrease vulnerability to the impacts of climate change and coastal hazards (Section 6.4.1). There is, however, no legal mandate for local governments to incorporate these provisions into their land use and climate adaptation strategies. The
current guidelines also promote adaptation strategies that can provide multiple benefits and refer to them as ‘win-win’ or ‘no regret’ strategies, although there is no explicit reference to the term ‘co-benefits’.

The recent Auckland’s CDEM Group Plan (2016-2021) adopts a risk-based approach, similar to that proposed in the MfE guidance manual, i.e. identifying the likelihood and consequences of natural hazards to assess the overall risk. However, it does not refer to the criteria which are proposed in the MfE guidance as the key considerations in assessing the vulnerability of coastal communities and ecosystems to the potential coastal hazards. The CDEM Plan includes one reference to restoration of catchments, wetlands, soil and other natural systems as an integral part of hazard management, but it is not clear whether and how the coastal wetlands are also considered in hazard management strategies. The plan does not include references to hazard management strategies that can provide multiple benefits particularly climate change co-benefits.

6.7. Summary

Overall, coastal wetlands are vulnerable ecosystems and in the past decade in Auckland have come under increased public scrutiny and policy liberalisation to allow area reduction by, for example, mangrove seedling removal. Underestimating the value of coastal wetlands ecosystems, in the Auckland’s planning system could be a reflection of the Auckland Council’s response to the perceived undesirable expansion of mangroves in some estuaries of Auckland and the consequent public pressure especially by advocates of mangrove removal in parts of the Auckland coastal areas. It can be also partly attributed to the lack of guidance by central government to make local governments take stronger policies to consider the values of coastal wetlands (particularly mangroves) in climate change response policies.

As mentioned, limited understanding and lack of local empirical information particularly regarding the mitigation benefits of coastal wetlands, together with the lack of global mandate to incorporate coastal ecosystems in climate mitigation policies is another contributing factor. In fact, the failure to incorporate coastal wetlands reflects the historical situation which is now beginning to change. For example, under the 2013 IPCC guidelines, GHG emissions and removals from coastal wetlands can be also incorporated in the national GHG inventories.

The fact that local governments are not allowed to consider the effects of activities in terms of GHG emission raises the question of whether and how carbon emission impacts of mangrove removal activities could or should be accounted for in the land use consent processes. The concept of ‘biodiversity offset’ and its ideal purpose to achieve ‘a net gain’ provides an opportunity for restoring coastal wetlands (affected by development and land use activities) and can contribute to climate change mitigation and adaptation. That is offsetting, by contributing to biodiversity values, can enhance coastal wetland systems and hence can increase adaptation to the impacts of climate change on coastal areas. This in turn creates a co-benefit of carbon storage and sequestration. Issues regarding property rights (including the existing use rights) highlight the importance of regional planning provisions (which can affect district plan’s rules) in planning for coastal wetlands protection and management.

The next section provides an overview of the current policy and planning documents in a number of selected coastal cities outside New Zealand to understand how coastal wetlands and their climate change services are managed in other jurisdictions and how land use strategies address adaptation to the effects of climate change and coastal hazards. It also includes a discussion about differences and similarities between Auckland and the global cases in terms of the approaches and strategies used to manage coastal wetlands and issues related to climate change issues.
7. Global experience, comparison with Auckland

7.1. Introduction

This chapter addresses the following questions:

- What can be learned from the experiences in other jurisdictions in terms of the initiatives and mechanisms that can assist in incorporating climate change services of coastal natural resources into land use, resource management and climate change policies?
- How do the policies and practices in other jurisdictions compare to the case of Auckland and New Zealand?

In order to answer these questions, this chapter first provides findings of a comprehensive review of the climate change response policies and initiatives in a number of jurisdictions (Sections 7.2 to 7.5). Key findings are then summarised in Section 7.6. As explained in Chapter 3 (Section 3.4.3), the case studies selected in this research are the coastal cities of New York and New Orleans in the United States, Rotterdam in the Netherlands, Jakarta in Indonesia and the coastal state of Tamil Nadu in India (Figure 7.1-1).

The purpose of this review is to identify initiatives that involve protection and enhancement of coastal natural features as a means of enhancing adaptive capacity of the coasts to the effects of climate change and increasing CS&S. The review also identifies approaches that have the potential to reduce vulnerability and enhance resilience to the effects of climate change in coastal areas by limiting development and promoting nature-based solutions in coastal areas. Section 7.7 of this chapter seeks to identify disparities or similarities between New Zealand (the case of Auckland) and global cases in terms of management of coastal wetlands for climate change mitigation and adaptation.

Findings of this chapter feed into the next chapter and inform a discussion on the possible policy options for mainstreaming protection and rehabilitation of coastal wetlands into the Auckland’s climate change and land use planning processes. The methods applied to the analyses in this chapter include case studies and systematic literature review that are explained in Chapter 3.
7.1.1. Outline of the review

To facilitate analysis of similarities and differences between the selected cases and the case of Auckland, and to identify transferable interventions or initiatives that can be further explored for their applicability to Auckland, each case study follows a consistent outline as below:

- Characteristics of the urban landscape and vulnerability to climate change
- Governance structure and climate response plans
- Adaptation to sea level rise and coastal hazards
- Initiatives for integrating climate change services of coastal natural features into climate change strategies or plans
- Key initiatives

7.2. New York City & New Orleans, The United States

7.2.1. Governance structure and federal climate response plans

The United States has a complex government structure. In the United States, numerous entities and agencies provide various types of public services at federal, state and local levels of governance. Local governments including municipalities are the third tier of governance and each level of governance has generally its own legislative, fiscal and administrative autonomy. Most of the local governments in the United States have the power and autonomy to set and regulate their own constitutions. However, cities and municipalities have limited power to levy taxes especially property taxes and largely depend on state regulations and federal grants to finance their plans.

The highest form of municipal legislation is a Local Law. Local legislative bodies (such as city councils) are granted by the State Constitution broad powers to enact their local laws equivalent to the laws enacted by state legislature, which might be subject to certain conditions. Cities as local units of government have also the power and autonomy to revise and change their city-wide rules and regulations (Charters) within their jurisdictions as long as there is no inconsistency with the state and federal constitutions and other general laws. In the case of a city, town or village, the granted powers in the United States include the power to adopt zoning regulations and conduct comprehensive planning. However, federal standards apply to some types of land use control such as floodplain regulations, which are land use controls governing the amount, type and location of development within defined flood-prone areas.

Municipalities must adopt their local floodplain regulations to be eligible for the National Flood Insurance Programme (NFIP) which is administered by the Federal Emergency Management Agency (FEMA). Since March 2016, only the states with approved climate change adaptation and hazard mitigation plan are eligible for the FEMA’s disaster preparedness funds. The new rules, however, do not affect federal funds for recovery after a natural disaster. Property owners with federally insured mortgages on buildings located in the high-risk areas (known as Special Flood Hazard Areas or SFHAs) must purchase flood insurance ‘up to the amount of the outstanding balance on the mortgage, or the National Flood Insurance...”

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97 Under the NFIP, the land in the floodplain within a community subject to a 1 percent or greater chance of flooding in any given year.
Under the NFIP, the SFHAs include areas that are subject to inundation as a result of a 100-year flood or a base flood (1% annual chance flood) (Becker, 2015). Areas between the limits of the base flood and a 500-year flood (0.2% annual chance flood) are identified as ‘moderate flood hazard zones’, while areas of ‘minimal flood hazard’ refer to areas outside the SFHAs and higher than the elevation of a 500-year flood98.

The annual insurance premium is defined based on the flood elevation and the elevation of buildings. Higher insurance premiums apply to buildings below the base flood elevation (BFE), with the fee decreasing as the buildings elevation increases (Figure 7.2-1). The lowest premiums apply to buildings at higher elevation above the base flood level (FEMA, 2013). This method encourages people to increase elevation of their buildings in order to minimise the insurance premiums. The government (FEMA) supports this by charging lower insurance fees if homeowners invest in strengthening their buildings against floods and take actions to mitigate flood risk, like elevating mechanical equipment. Similar incentives exist for communities that adopt and enforce a floodplain management ordinance to reduce future flood risks to new construction in the SFHAs. Property owners can also obtain excess flood insurance from private insurers to cover the losses beyond the coverage of the NFIP (Brown, 2016).

The SFHAs along with other risk zones are identified on a special Flood Insurance Rate Map (FIRM) developed specifically for this purpose (FEMA, 2011). Information provided by FIRMs establishes a basis for identification of an annual flood insurance premium under the NFIP.

The United States is a signatory to both the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol. It has ratified the UNFCCC in 199299 but it has not yet ratified the Kyoto Protocol. The US states and municipal governments have taken extensive local actions including adopting laws and policies on climate change, renewable energy, and energy efficiency in order to reduce their emission of GHGs100. In the absence of federal leadership and national guidance on climate change, the local efforts at state, local, and regional levels are argued as examples of ‘bottom-up governance’ or ‘multi-level governance’ (Wheeler, 2008).

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The Environmental Protection Agency (EPA) has developed a number of regulations regarding GHGs reduction (focusing on transport and energy sectors and municipal landfills) under the Clean Air Act (Burger, et al., 2016). The EPA has also provided grants to many states to support their GHGs reduction targets and mitigation plans. The local climate change action plans, developed before 2008, were mainly focused on climate change mitigation rather than adaptation (Wheeler, 2008). In June 2013, President Obama’s Climate Action Plan was announced and outlined a wide range of actions to reduce carbon pollution, increase the use of renewable energies and clean (low carbon) energy sources and enhance resilience to the effects of climate change\textsuperscript{101}. In response to the Executive Order 13653 regarding preparing a Climate Action Plan, signed by President Obama in November 2013, the EPA prepared and finalised a Climate Change Adaptation Plan in June 2014\textsuperscript{102}. The plan identifies 10 priority actions that the EPA needs to take to integrate climate change adaptation in its policies and plans to ensure their effectiveness under future climatic conditions.

- ‘Fulfill Strategic Measures in FY 2011–2015 EPA Strategic Plan
- Protect Agency facilities and operations
- Factor legal considerations into adaptation efforts
- Strengthen adaptive capacity of EPA staff and partners through training
- Develop decision-support tools that enable EPA staff and partners to integrate climate adaptation planning into their work
- Identify cross-EPA science needs related to climate adaptation
- Partner with tribes to increase adaptive capacity
- Focus on most vulnerable people and places
- Measure and evaluate performance
- Develop and Implement Program and Regional Office Implementation Plans’ (US Environmental Protection Agency, 2010, p. 42)

### 7.2.2. New York City

#### 7.2.2.1. Characteristics of the urban landscape and vulnerability to climate change

New York City, as the most populous city in the United States\textsuperscript{103}, is a coastal city which is largely flat and low-lying and is located in south eastern New York state (Gornitz, et al., 2002). New York is vulnerable to sea level rise and extreme weather conditions (Gornitz, et al., 2002). The city includes approximately 837 km (520 miles) of coastline (The City of New York, 2015) and every resident lives within 8 km of the waterfront (La Rocco, 2004). Hurricane Sandy in October 2012 caused extensive damage particularly to the properties and infrastructure within the city’s low-lying ocean-facing neighbourhoods (The City of New York, 2013a). The city’s coastline has changed significantly since human settlement in the 17\textsuperscript{th} century and most of the natural features including tidal wetlands which once protected the city against coastal hazards have been lost over time (The City of New York, 2013a).

Wetlands are dominant natural features across much of New York City’s natural waterfront (NYC Dept. of Parks and Recreation, 2012). However, around 85% of the coastal wetlands across New York-New Jerseys Harbour Estuary have been lost over the last century (NYC Dept. of Parks and Recreation, 2012). Until recently, the city’s remaining wetlands and natural features that could provide important buffers against coastal storms and sea level rise had often been considered as underutilized property that could be reclaimed and developed (NYC Dept. of Parks and Recreation, 2012). As defined in the New York City

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\textsuperscript{103} U.S. Census Bureau, Federal, State, & Local Governments Main. [Online] Available at: http://www.census.gov/govs/ [Accessed 2016]
Wetland Strategy (2012), tidal wetlands include estuarine intertidal flats (mudflats, sand bars, and beaches), estuarine emergent wetlands (vegetated flats), or low and high salt marshes that are flooded on a regular or irregular basis (NYC Dept. of Parks and Recreation, 2012).

In response to the drastic reduction in the area of tidal wetlands in New York, the Tidal Wetland Act was passed by the New York State in 1973 and sought to regulate development activities in tidal wetlands and their adjacent areas (i.e. areas up to 300 feet inland from the wetland boundary within New York State, areas up to 150 feet inland within New York City)\(^\text{104}\). Under the Tidal Wetland Act, ‘Tidal Wetlands Permit Programme’ which is administered by the New York State Department of Environmental Conservation requires a permit for any activities that may alter wetlands or their adjacent areas.

However, it is stated that despite the requirements of state laws in terms of a transition area for inland migration of wetlands, development has occurred much closer to the wetland boundary (NYC Dept. of Parks and Recreation, 2012). The New York City’s Wetland Strategy has been developed under the Local Law 31 and aimed to avoid a net loss of the New York City’s wetlands (including tidal wetlands) while improving and maximising their ecological services (NYC Dept. of Parks and Recreation, 2012).

**7.2.2.2. Governance structure and climate response plans**

New York City has its own charter and set of local laws (New York State Department of State, 2011). The city is a consolidated assembly of several counties, towns, cities and villages, which has been formed through a long-term reorganisation process. The city’s government has the form of a strong mayor-council, with the mayor serving as the chief executive and administrative head of the city, and the council as the policy making body. The mayor usually has the power to prepare the budget, appoint and remove certain officials, and to exercise broad veto powers over council actions (New York State Department of State, 2011).

In New York, some matters of state concern are within the jurisdiction of state legislature and in principle a local government may not adopt a local law relating to those matters, unless the state legislature has specifically granted such power to the local government by law (New York State Department of State, 2011). These include matters which by their nature require intervention of the state and intergovernmental cooperation at state level, such as transportation and highways, education, health, banking and taxation. New York City needs to work with the state and other municipalities to deliver these types of regional services. New York City also needs to work with regional planning councils. These are intergovernmental boards that are locally formed by adjoining counties and primarily provide a regional approach to issues that extend beyond the boundaries of local government.

In 2009 and under the Executive Order 24, the New York State Climate Action Council (NYSCAC) was created and was charged with preparing the New York State Climate Action Plan. The New York State Climate Action Plan Interim Report was released by the CAC in 2010 and identified the state’s main strategies for reducing GHGs emission and improving adaptation to the impacts of climate change and natural disasters. However, the CAC has not finalised the New York State Climate Action Plan and the recommendations included in the Interim Report have not been officially endorsed by the state\(^\text{105}\).

New York City’s first commitment to reduce greenhouse gas emission dates back to 2007 when the first PlaNYC (i.e. Greener, Greater New York), the city’s long term, sustainability plan, was released by New York City’s Mayor Michael Bloomberg (The City of New York, 2013a). By local law, PlaNYC needs to be updated every four years and progress needs to be reported annually (Local Governments for Sustainability USA, 2010). The first two versions of PlaNYC (PlaNYC 2007 and PlaNYC 2011) were titled ‘A Greener, Greater New York’ and included initiatives to improve the quality of life for residents,


support green infrastructure, and reduce the emission of GHGs along with a number of other sustainability objectives. PlaNYC 2007 includes a goal to create New York City Climate Change Adaptation Task Force. In order to support the goals identified in PlaNYC and to inform the New York City Climate Change Adaptation Task Force on matters relevant to climate change, the New York City Panel on Climate Change (NPCC), consisting of leading climate scientists, was established by Michael Bloomberg in 2008.

New York City has adopted a flexible adaptation approach to respond to the impacts of climate change (Solecki & Rosenzweig, 2014). This approach seeks to promote strategies that address both existing and future climate risks. Gradual retreat from the most at-risk areas and converting these areas to open and green spaces (such as parkland) that could provide flood protection had been considered as part of the climate change adaptation strategies in the city’s climate change programme published in 2008 (NYC Dept. of Environmental Protection, 2008). However, due to the high-density development in coastal areas, retreat from the coast or abandonment are not considered in the proposed coastal protection strategies in PlaNYC 2013 and OneNYC 2015 (The City of New York, 2013a). Instead, the strategies are mainly focused on protection and aim to create more resilient shorelines that can withstand future hazards (The City of New York, 2013a).

Following Hurricane Sandy, the third PlaNYC (PlaNYC 2013: A Stronger, More Resilient New York) was prepared by the second New York City Panel on Climate Change (NPCC2) and specifically included strategies for improving the city’s resilience to the impacts of climate change particularly sea-level rise and more frequent storm surges. OneNYC 2015 (The Plan for a Strong and Just City), is the most recent plan which provides the city’s vision for the next century and sets out goals and targets as well as action plans/initiatives to address the city’s challenges including climate change. Strengthening the city’s coastal defence against flooding and sea level rise is identified as one of the goals towards a resilient city in OneNYC 2015.

7.2.2.3. Adaptation to sea level rise and coastal hazards

Coastal protection plan

After Hurricane Sandy in October 2012 and its extensive damage, PlaNYC 2013 outlined a comprehensive coastal protection plan for increasing resilience to coastal hazards and sea level rise (The City of New York, 2013a). The coastal protection plan was developed based on the findings of a multi-faceted analysis which identified the vulnerability of coastal areas and examined the feasibility of different protection measures for vulnerable areas (The City of New York, 2013a). This analysis was also used to develop the coastal risk map for New York City (Figure 7.2-2). This map was then used to prioritise implementation of the proposed set of measures in areas which were exposed to greater risk of coastal hazards.

The vulnerability analysis of coastal areas in the New York City was based on the following factors:

- Percentage of particularly vulnerable populations (e.g. elderly or those with disabilities)
- Coastal flood elevation
- Underlying geomorphology (e.g. oceanfront beaches), geology (e.g. glacial outwash plains) soil data (e.g. soft; mix of soft and dense) and elevation of the coastline
- The distance over which the wind blows in one direction (defined as fetch) to identify susceptibility to wind-driven wave action
- Exposure to high velocity breaking waves
- Performance of existing coastal features (e.g. revetments, bulkheads or piers) in reducing damage and flooding during Sandy
- Expected costs for disaster response and recovery, considering the value of properties at risk

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The New York City also examined effectiveness of a single engineering solution (i.e. construction of massive harbour-wide storm surge barriers) to protect the city against future storms and coastal hazards. The results demonstrated that the risks and complications that such a single solution may pose to the city were much greater than its benefits (The City of New York, 2013a). Consequently, an integrated system of discrete coastal projects was suggested as an appropriate approach to coastal protection. The projects were individually designed so that they match the specific characteristics of a given coastal area and were managed with an integrated approach that primarily sought to achieve more effective coastal protection across the entire coastal landscape (The City of New York, 2013a).

Figure 7.2-2. Coastal risk map developed for New York City as part of the comprehensive coastal protection plan (The City of New York, 2013a, PlaNYC 2013, p.57)

The coastal protection plan identified 37 initiatives and projects as the Phase One initiatives which are currently being implemented and are spread over a 10-year period. Sophisticated storm surge modelling using digital hydrodynamic models was used to evaluate the effectiveness and performance of the suggested protection measures in reducing wave heights and energy of Sandy-like storms and other projected 100-year and 500-year storms. Through a cost-benefit analysis, the construction and long-term maintenance costs and associated benefits were analysed for each measure and the results were used to facilitate comparison between different measures (The City of New York, 2013a).

**Zoning and insurance policies**

Following Hurricane Sandy, the New York City Council adopted the Flood Resilience Zoning Text Amendment in 2013 to facilitate reconstruction of damaged buildings in compliance with the new requirements of the Building Code and higher flood elevations issued by the FEMA (Section 7.2.1) (The City of New York, 2013b). The building code requirements are based on the FIRMs that were updated for the city post Sandy. Currently, Preliminary FIRMs for the coastal areas of the city are available to the
public through FEMA’s Geoplatform\(^{107}\) (Figure 7.2-3). They include up-to-date information about the SFHAs and base flood elevation (BFE).

The flood insurance premium was increased following Hurricane Sandy \(\text{(The City of New York, 2015)}\). Construction within the flood zones, must comply with the flood resistant construction standards and building code requirements \(\text{(FEMA, 2013)}\). These standards provide for using flood-resistant materials in buildings, elevating buildings above anticipated flood levels and designing buildings in a way that withstand the wave’s pressure \(\text{(FEMA, 2013)}\). Applying these standards will decrease flood insurance premiums and make buildings less vulnerable.

![Figure 7.2-3. Screenshot of the FEMA’s Geoplatform that provides flood hazard data for coastal areas of New York, The figures in parenthesis indicate BFE. The website also provides information on the estimated ground elevation\(^{108}\) – Zones starting with A or V are sub-zones of the SFHAs - Zone X (unshaded) represent areas with low flood risk\(^{109}\).](image)

**Resilient neighbourhoods initiative**

The main purpose of the ‘Resilient Neighbourhoods’ initiative in New York City is to identify changes in land use and zoning regulations that are required to improve the adaptive capacity and resilience of the neighbourhoods against coastal hazards (e.g. storm surges) and other climate events\(^ {110}\). This initiative was launched in 2013 and is funded by a federal grant\(^ {111}\). It is identified in OneNYC 2015 as a programme to evaluate land use and zoning regulations through a public participation process in order to improve the city’s adaptive capacity \(\text{(The City of New York, 2015)}\). Ten neighbourhoods that were severely damaged by Hurricane Sandy and had their unique challenges and opportunities have been selected for the first phase of the study on Resilient Neighbourhoods. Through an extensive public outreach process and working with the affected communities in each neighbourhood, the issues are identified and specific objectives and recommendations are developed and finalised\(^ {112}\).

**Voluntary Buyout and Acquisition Programme**


\(^{111}\) The U.S. Department of Housing and Urban Development (HUD) Community Development Block Grant for Disaster Recovery.

At the state level and as part of the post-Sandy recovery plans, the state of New York created a voluntary Buyout and Acquisition Programme as an incentive-based mechanism to encourage moving away from the high-risk coastal areas and turning those areas into their natural state. The full pre-storm fair market value of homes (up to the FHA Federal Housing Administration loan limit) will be paid to homeowners eligible for a buyout. Families that relocate within their home county or borough are also eligible for an incentive of up to 5%. The voluntary Buyout and Acquisition Programme is currently operating in a number of neighbourhoods (selected by cooperation of the individual homeowners) on Long Island and Staten Island and aims to turn the parcels of land into natural coastal buffers such as wetlands, open space, or stormwater management systems. PlaNYC 2013 has also identified an initiative to identify eligible communities for the New York Smart Home Buyout Program through collaboration with the State.

**Transferable Development Rights (TDR)**

To create financial incentives to shift development away from flood zones, it is suggested that properties in coastal areas that are not at maximum floor-area ratio (FAR), could sell FAR or development rights to developers in commercial districts outside the flood zones where more growth is desired. The main requirement of TDR is availability of ‘unutilised FAR’ in coastal regions which can be used to increase development in incipient commercial areas outside the flood zone. ‘Coastal Protection Bonus’ can be also used to finance coastal protection projects by generating and then selling bonus FAR to developers in inland commercial districts. This mechanism is based on the ‘District Improvement Bonus’ that has been used in Manhattan for infrastructure improvement.

7.2.2.4. Initiatives for integrating climate change services of coastal natural features into climate change strategies/plans

**Incorporating coastal wetlands into the coastal protection strategies**

President Obama’s Climate Action Plan (2013) recognised the role of forest in climate change mitigation and provided for protection and restoration of forests, grasslands and wetlands in the face of a changing climate. The plan also refers to the significant investment for enhancing barrier islands and marsh ecosystems that protect coastal communities against storm surges.

The experience of Hurricane Sandy revealed the important role of coastal natural features (e.g. nourished beaches, sand dunes and restored saltmarsh wetlands) in mitigating inland flooding, sand migration and reducing damage to properties. Post-Sandy’s observations demonstrated that areas behind sand dunes (with vegetation) and restored tidal wetlands (with elevated edges) suffered significantly less damage compared to other areas which were not protected by natural features. Findings of a storm surge modelling, which was used to evaluate coastal protection strategies in PlaNYC 2013, also indicated that if properly located, natural features (living shorelines) including beaches, tidal wetlands, dunes, oyster reefs, living breakwaters and coastal/maritime forests can provide a wave attenuating function. The modelling also indicated that the wave attenuation function of tidal wetlands can be further strengthened.

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113 NYS Governor’s Office of Storm Recovery, Buyout & Acquisition Programs. [Online] Available at: [https://stormrecovery.ny.gov/housing/buyout-acquisition-programs](https://stormrecovery.ny.gov/housing/buyout-acquisition-programs) [Accessed 2016].

114 [https://stormrecovery.ny.gov/housing/buyout-acquisition-programs](https://stormrecovery.ny.gov/housing/buyout-acquisition-programs) [Accessed 2016]

115 [https://stormrecovery.ny.gov/housing/buyout-acquisition-programs](https://stormrecovery.ny.gov/housing/buyout-acquisition-programs) [Accessed 2016]

116 The ratio of a building's total floor area (gross floor area) to the size of the piece of land upon which it is built.

117 Living shorelines are referred to as ‘coastal edges that incorporate combination of reefs, living breakwaters, maritime or coastal forests, and tidal wetlands to reduce wave action and erosion’ (The City of New York, 2013a, p.53).
by either changing the depth at which these coastal habitats are placed or by slightly increasing hardened elements such as rock (The City of New York, 2013a).

The results of a preliminary cost-benefit analysis, which was used to identify the best coastal protection measure for Howard Beach, indicated that a hybrid measure including both natural and hard engineering structures appears to be the most cost effective solution. This study that was conducted by the Nature Conservancy in 2013 (The Nature Conservancy, 2013) demonstrated that natural coastal features including tidal wetlands (saltmarshes), dunes, seagrass beds and coastal forests can and should be considered in conjunction with engineering structures in order to provide sustainable protection against storm surge and sea level rise. The study found the greatest benefit/cost ratio (B/C ratio) for a hybrid measure including the elements of operable flood gates, restored marsh and ribbed mussel as well as berm and rock groins (The Nature Conservancy, 2013).

Given the findings of the relevant studies, modelling and the experiences from Sandy, PlaNYC 2013 and OneNYC 2015 have incorporated nature-based initiatives into the city’s coastal protection measures. A number of initiatives specifically provide for restoration and enhancement of tidal wetlands (saltmarsh ecosystems) in some coastal areas such as Howard Beach and Jamaica Bay that are highly exposed to coastal inundation (Figure 7.2-4). PlaNYC 2013 does not provide for restoration or protection of coastal wetlands for their CS&S potential. OneNYC 2015 has included a quantitative indicator as ‘acres of coastal ecosystems restored’ to measure progress towards achieving the coastal protection goal.

Sediment from maintenance dredging activities has been used to restore saltmarsh wetlands in Jamaica Bay (Messaros, et al., 2012). Surface elevation (sediment erosion/accretion) is also one of the parameters that has been regularly monitored to ensure long-term ecological benefits of the restored tidal wetlands (Messaros, et al., 2012). Changes in tidal range, sea level, sediment accretion and subsidence are identified as main parameters that affect the hydrology (i.e. frequency and duration of saltmarsh flooding) of saltmarsh ecosystems within Jamaica Bay and thus were considered in restoration activities (Messaros, et al., 2012).

Enhancing the area of coastal wetlands in Staten Island is also part of activities that aim to expand the existing Bluebell (stormwater management system) to reduce the impacts of storm-related flooding while providing ecological benefits (NYC Dept. of Environmental Protection, 2008; Solecki, et al., 2014). Goal 2 (Protecting America’s Waters) of the EPA’s 2014 Climate Change Adaptation Plan provides for restoration and protection of watersheds, wetlands, oceans, and aquatic ecosystems. However, the focus of restoration and protection of these ecosystems is on reducing their vulnerability to the potential impacts of climate change such as sea level rise, changes in rainfall patterns and rising temperature. The plan does not provide for protection and restoration of coastal wetlands for their coastal protection or carbon sequestration potential. In contrast, the New York State Climate Action Plan Interim Report (2010) has included references to the contribution of freshwater and tidal wetlands to sequester carbon and mitigate the effects of climate change (NYSCAC, 2010). The report has specifically provided for protection and restoration of tidal wetlands for their climate change mitigation and adaptation services.

The relevant policies from the New York State Climate Action Plan Interim Report (NYSCAC, 2010) include:

- ‘Protect and restore freshwater and tidal wetlands through acquisition of fee or easement and regulation to prevent releases of GHGs which will allow existing freshwater and tidal wetlands to continue to sequester carbon and mitigate the effects of more intense storm events caused by climate change’ (NYSCAC, Chapter 9, p.9-22).
- ‘Protect and enhance the stability and function of stream, river, and aquatic coastal systems to accommodate changing climate conditions’ (NYSCAC, Chapter 11, p.11-31).
- ‘Infrastructure investments should be designed and constructed to protect and preserve natural resources and ecosystems that provide essential climate-adaptation services or benefits in addition to meeting transportation needs’ (NYSCAC, Chapter 11, p.11-74).
Mitigation banking or bio-banking (e.g. wetland mitigation banking) is one of the three broad approaches for ‘offset’ or ‘compensatory mitigation’ (OECD, 2016). The other two approaches include ‘one-off’ or ‘permittee-responsible mitigation’ and ‘in-lieu fee mitigation’ (OECD, 2016, p. 26). Under the one-off or permittee responsible mitigation, the developer is required to carry out the offset project at or adjacent the impact site or at another location usually within the same watershed. The developer takes on the financial and legal liability of the offset project under the one-off approach (OECD, 2014c).

Unlike the one-off approach, in-lieu fee and mitigation banking are forms of ‘third-party’ offset or compensatory mitigation because financial and legal liability is transferred from the developer to a third-party or ‘offset provider’ (OECD, 2016). A third party can be a government agency, corporation, non-profit organization, or a private entity that undertakes the offset project under a formal agreement with a regulatory agency.

Under a wetland mitigation banking approach, ‘offset credits’ that are generated through large-scale wetland restoration projects can be sold to developers to compensate for the adverse effects of their development activities on wetlands (OECD, 2014c). Since the offset has already been created prior to the adverse effects of development, the mitigation banking approach significantly minimises the risk that offset objectives are not met (OECD, 2016).

The United States was one of the early adopters of this approach to balance economic initiatives with wetlands restoration and to prevent no net loss in wetlands area. The U.S. Wetland Mitigation banking has its roots in the in the US Clean Water Act (1972) (Chapter 404(b)(1)). The effectiveness of wetland mitigation banking in the United States was not fully appreciated in the 1980s, mainly due to the irreplaceable nature of wetland ecosystems and lack of immediate evidence regarding the success of mitigation banking after the wetland is established (Race & Christie, 1982; Race, 1985). Development of

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119 U.S. Army Corps of Engineers, Compensatory Mitigation Rule, s.l.: U.S. Army Corps of Engineers (USACE).
comprehensive and interagency national guidance on mitigation banking (e.g. National Wetlands Mitigation Action Plan, 2002; Final Compensatory Mitigation Rule, 2008) provided further instructions and supported achievement of the goal of no net loss of wetlands.\footnote{https://www.epa.gov/cwa-404/mitigation-banking-factsheet [Accessed 2016]}

Given the importance of coastal wetlands as defensive buffers against sea level rise and storm surges and challenges in financing coastal wetland restoration projects, PlaNYC 2011 (The City of New York, 2011) identified initiatives to restore tidal wetlands through mechanisms such as ‘in-lieu fee mitigation’ or ‘mitigation banking’. Initiative 33 of the comprehensive coastal protection plan (PlaNYC, 2013) also provides for a study to examine the feasibility of ‘wetland mitigation banking’ in New York City. The proposed approach for wetland mitigation banking in New York City provides an opportunity for restoration of tidal wetlands in areas with high risk of flooding where wetlands can contribute to coastal protection (The Nature Conservancy, 2013). It encourages off-site mitigation projects which compared to the conventional on-site projects are more feasible in New York City, given the limited or lack of space in most coastal areas and high value of coastal lands (The City of New York, 2011).

Under this process, wetlands are established, enhanced or restored in critical areas (including coastal areas exposed to high risk of flooding and sea level rise) and restoration costs will be covered by selling ‘mitigation credits’ to developers who seek to build on or near wetlands in low flood-risk areas (The Nature Conservancy, 2013; Freshkills Park Alliance, 2016). It also facilitates compliance with permit requirements for projects as the mitigation banking sites are already selected and approved (New York City Economic Development Corporation, 2013).

The New York City’s first wetland mitigation banking initiative, the MARSHES\footnote{Mitigation and Restoration Strategies for Habitat and Ecological Sustainability (MARSHES)}\footnote{Service area refers to a geographic area in which permitted impacts can be compensated for at a given bank.}, was developed to support restoration and protection of the valuable wetlands through utilising federal recovery funds (C40 Cities, 2016). The first pilot mitigation banking project in New York City, which is currently in progress (Freshkills Park Alliance, 2016) is aimed to restore a degraded tidal marsh wetland (surrounding Saw Mill Creek on Staten Island) for reinstating its coastal protection and ecological values (Freshkills Park Alliance, 2016). The project will restore 68.94 acres (approximately 28 ha) of the wetland and will cost approximately US$14 million. It is expected to create about 18 credits that will be used to compensate unavoidable impacts of development activities on wetlands at other sites (Freshkills Park Alliance, 2016). New York City has leveraged the public (e.g. federal disaster recovery fund) and private economic investment to support the Saw Mill Creek Pilot project (New York City Economic Development Corporation, 2016; C40 Cities, 2016).

Restoration of the Saw Mill Creek tidal marsh wetland is expected to achieve three main goals of (i) increasing coastal resilience and protection against storm events, flooding and sea level rise, (ii) restoring significant ecological values of the wetland, and (iii) mainstreaming the compensatory mitigation approach to wetlands in New York City (New York City Office of Management and Budget, 2015).

The process to identify the Saw Mill Creek as the pilot New York City wetland mitigation bank project included an exhaustive consultation and evaluation of a number of potential locations (New York City Office of Management and Budget, 2015). The potential sites were evaluated and screened out based on four criteria including (i) the potential of a site to serve the selected service area\footnote{Service area refers to a geographic area in which permitted impacts can be compensated for at a given bank.}, (ii) site ownership and control, (iii) the site’s current ecological suitability and future ecosystem services that would result from restoration, and (iv) technical considerations (New York City Office of Management and Budget, 2015). Except for the first criterion, each of the other three criteria included a number of sub-criteria which were used to screen out the potential sites.

The service area is often identified through a negotiated process and may include watershed/catchment boundaries or the ecological unit boundaries of surrounding hydrologic basins (New York City Office of...
Management and Budget, 2015). The service area can be divided into the ‘primary’ and ‘secondary’ service areas. The mitigation credits required by activities/projects in primary service area are preferably provided by the selected mitigation bank; while decisions on the use of credits by activities/projects in secondary service area would be made by a regulatory authority on a case-by-case basis (New York City Office of Management and Budget, 2015).

7.2.2.5. Key initiatives

The key initiatives in New York City include:

- A comprehensive coastal protection plan was developed based on vulnerability of coastal areas to floods and sea level rise and the climate change projections were integrated into land use planning and development in coastal areas.
- Nature-based initiatives were incorporated into the coastal protection measures and specific initiatives were developed to restore and enhance tidal wetlands in a number of coastal areas.
- Wetlands Mitigation Banking programme was adopted and employed as a financial mechanism to support wetland restoration activities.
- Quantitative indicators were developed including (i) the linear feet of coastal defences completed, (ii) the acres of coastal ecosystems restored and (iii) the number of residents benefiting from coastal defences and restored ecosystems to measure the progress towards achieving the coastal protection goal.
- The city developed ‘Resilient Neighbourhoods’ programme to evaluate and update existing local and citywide land use and zoning regulations and to establish a framework for adaptive land use planning.

7.2.3. New Orleans

7.2.3.1. Characteristics of the urban landscape and vulnerability to climate change

New Orleans is a coastal city in the state of Louisiana which is located on the Gulf of Mexico. The city expands over 438 km$^2$ of lands where wetlands include one third of the city’s land areas (i.e. 146 km$^2$) (City of New Orleans, 2015). The city is built upon the deltaic soils (deposited by the Mississippi River over 7000 years) and due to its unique geography on the Gulf of Mexico, is highly exposed to the impacts of climate change such as sea level rise and more frequent and intense storm events (City of New Orleans, 2015).

The Mississippi Deltaic Plain (MDP), (including wetlands and low upland ridges), which is located in southeast New Orleans, has been formed as a series of overlapping delta lobes over millennia before human impact. Human activities over the last two decades have affected the hydrology and the sediment balance in the Mississippi basin resulting in extensive degradation of Louisiana’s wetlands that once buffered Louisiana’s coastline against storm surge and flood events (Day, et al., 2007). Reclamation and drainage of wetlands in the low-lying areas of the city spread development into the locations that were previously uninhabitable (City of New Orleans, 2015).

The Hurricanes Katrina and Rita in 2005 devastated the Gulf Coast and caused extensive damage to Louisiana’s coastline and the city of New Orleans. Analysis of the impacts of Hurricanes Katrina and Rita (2005) showed that deterioration of the Mississippi Deltaic Plain (MDP), including wetlands and low upland ridges, increased the vulnerability of coastal communities to the adverse effects of extreme storm events (Day, et al., 2007). Louisiana and New Orleans were among the areas that were severely affected by the Hurricanes Katrina and Rita in 2005. Coastal areas such as all Mississippi beachfront towns experienced the worst property damage mainly because of the catastrophic failure of the levee systems particularly the shipping channel known as the Mississippi River Gulf Outlet (MRGO) (Dixon, et al., 2006). A number of studies and observations argued that areas with extensive canopies of forested wetlands
experienced less damage compared to areas with no or damaged coastal vegetation (Day, et al., 2007; Barbier & Enchelmeyer, 2014).

7.2.3.2. Governance structure and climate response plans

The administrative divisions in the State of Louisiana are called parishes which is similar to counties in other states. In the State of Louisiana, the City of New Orleans is consolidated with the parish of Orleans and constituted a city-parish jurisdiction that has the powers and responsibilities of both types of local government. As noted before, in the United States, the responsibilities and functions of local governments within states vary based on the states' constitutions (Col, 2007).

Under the Louisiana’s state constitution, local governments (i.e. both municipalities and parishes) are granted with a great autonomy to regulate local issues and the state is prohibited from intervention in local matters unless specified (Col, 2007). Similar to New York City, the city of New Orleans has also a strong mayor-council form of government with the mayor serving as the chief executive and administrative head of the city, while the council serves as the city's primary legislative and policy making body. Along with overseeing the city's day-to-day operation, the mayor has the power to sign legislation into law, propose budget and appoint mayoral directors. The city council then adopts the city budget proposed by the mayor, approves the directors appointed by the mayor and levies taxes. The council has also the power to make or amend city laws, policies and ordinances125.

Unlike the New York State, the State of Louisiana does not have a Climate Action Council (CAC). The state still lacks a climate change mitigation plan; however, the Louisiana's Comprehensive Master Plan for a Sustainable Coast (2007) (Coastal Protection and Restoration Authority, 2007) addresses land loss from sea level rise, subsidence and other factors over the next 50 years and indicates a combination of coastal restoration and protection initiatives to achieve the state’s approach towards a sustainable coast. The Master Plans 2007 and 2012 were prepared by the Louisiana Coastal Protection and Restoration Authority (CPRA) as a requirement under Act 8 of the First Extraordinary Session of 2005126.

In January 2016, the State of Louisiana won a $92,629,249 grant from NDRC127 to support implementation of Louisiana Strategic Adaptations for Future Environments Programme (Louisiana Safe or LA SAFE) (National Disaster Resiliency Competition, 2016). LA SAFE seeks to complement the Louisiana’s Coastal Master Plan (2012) and outlines a development strategy for the state’s hurricane-impacted areas (Louisiana Office of Community Development Disaster Recovery Unit, 2015).

The city of New Orleans has developed a number of planning documents which address climate change mitigation and adaptation. Main examples include development of UNOP (Unified New Orleans Plan to guide the deployment of disaster recovery funding) in 2007 (City of New Orleans, 2007), GreeNOLA (The city’s sustainable rebuilding and recovery plan) in 2008 (The Louisiana Disaster Recovery Foundation, 2008); the city’s Master Plan (the City Charter-mandated Plan for the 21st Century: New Orleans 2030) in 2010 (City of New Orleans, 2010); and Resilient New Orleans in 2015 (City of New Orleans, 2015).

The GHGs mitigation strategies included in GreeNOLA mainly provide for encouraging green buildings, enhancing energy efficiency in residential and transport sectors and greater use of alternative (renewable) energies and clean fuels (The Louisiana Disaster Recovery Foundation, 2008). A number of programmes including the Green Building Ordinance, Green Jobs Training, a Green Council, the Solar America City and wetlands assimilation project are among the initiatives that contribute to reducing GHGs emission.

127 National Disaster Resiliency Competition (NDRC)
The wetland assimilation project is a response to the Louisiana’s natural wetland loss partly caused by insufficient sedimentation, land subsidence and relative sea level rise. The project is based on an idea to allow release of treated and sanitary nutrient-rich wastewater into suitable wetlands to activate soil formation and vertical accretion by stimulating root growth as a result of the increased organic matter deposition. This model is considered as a green approach to waste water treatment with low maintenance and operating costs and also intended to increase wetland’s carbon sequestration rate by stimulating plant growth and productivity (Louisiana Department of Environmental Quality, 2016).

Resilient New Orleans 2015 (City of New Orleans, 2015) as the latest climate-related, sustainability plan in New Orleans utilises local expertise and global best practices to improve the city’s resilience and adaptive capacity to the future challenges facing the city (e.g. climate change, poverty, crime). The plan was developed through a joint collaboration with 100 Resilient Cities Network, pioneered by the Rockefeller Foundation and outlines a number of visions, goals and actions to guide the city towards a resilient future by 2050. By adopting a ‘Resilience Dividend’ approach, the city aims ‘to improve its overall strength and endure multiple shocks and stresses rather than investing to reduce impacts from a single hazard or improve a single metric’ (City of New Orleans, 2015, p.25). The Resilient New Orleans 2015 includes a specific action to develop and implement a Climate Action Plan for New Orleans which will help to support the city’s GHGs reduction targets.

7.2.3.3. Adaptation to sea level rise and coastal hazards

Coastal Plan

The Louisiana's Comprehensive Master Plan 2007 outlined coastal protection strategies that relied on ‘multiple lines of defence’ concept. This approach involves integrating both natural features and structural measures to provide effective protection against flood and hurricane events (Coastal Protection and Restoration Authority, 2007). The Louisiana’s Coastal Master Plan 2012, built on the strategies proposed in the 2007 Master Plan and identified science-based initiatives to reduce the risks of coastal hazards on the coastline of Louisiana over a 50-year period (Coastal Protection and Restoration Authority, 2012). The objectives identified in the plan seek to increase flood protection for coastal communities, improve natural processes to rebuild and restore coastal ecosystems and enhance recreational, cultural and economic functions of coastal habitats (Coastal Protection and Restoration Authority, 2012).

Broad categories of projects and a list of project types under each category is provided in Table 7.2-1. Spatial distribution of these projects is demonstrated in Figure 7.2-5. To select the final projects for inclusion in the Master Plan, a number of primary factors (decision drivers) and decision criteria were considered and the practical implication of different project options were analysed through using science based tools such as Predictive Models and Planning Tools (Coastal Protection and Restoration Authority, 2012). For example, the implications of individual projects on ecosystem services of coastal habitats (e.g. saltwater fisheries, habitat provision, carbon sequestration and nitrogen uptake) were evaluated and the results were used to identify which projects better meet the priorities and preferences set out in the plan. The 2012 Master Plan finally provides a number of projects (109 projects) to be implemented on the Southwest Coast, Central Coast and Southeast Coast during two phases: 2012-2031 and 2032-2061.

As Table 7.2-1 indicates, the Louisiana’s Coastal Master Plan (2012) includes non-structural solutions to enhance and strengthen coastal protection. Adopting land use policies to avoid unwise development in high-risk coastal zones, informing coastal communities about the risk of living in flood risk zones, enacting local regulations and improving building codes are included as the programmatic non-structural measures suggested in the Master Plan.
Table 7.2-1. Project types identified in the 2012 Louisiana’s Coastal Master Plan (Coastal Protection and Restoration Authority, 2012)

<table>
<thead>
<tr>
<th>General category of project</th>
<th>Project type</th>
</tr>
</thead>
<tbody>
<tr>
<td>Restoration</td>
<td>Barrier Island/Headland Restoration, Hydrologic Restoration, Marsh Creation, Marsh Creation, Oyster Barrier Reefs.</td>
</tr>
<tr>
<td></td>
<td>Ridge Restoration, Sediment Diversion, Channel Realignment, Bank Stabilization, Shoreline Protection</td>
</tr>
<tr>
<td>Protection:</td>
<td>Earthen Levee</td>
</tr>
<tr>
<td>Structural</td>
<td>Concrete Wall</td>
</tr>
<tr>
<td></td>
<td>Floodgate</td>
</tr>
<tr>
<td></td>
<td>Pumps</td>
</tr>
<tr>
<td>Protection:</td>
<td>Elevation</td>
</tr>
<tr>
<td>Non-Structural</td>
<td>Flood proofing</td>
</tr>
<tr>
<td></td>
<td>Voluntary Acquisition</td>
</tr>
<tr>
<td>Protection:</td>
<td>Land use planning</td>
</tr>
<tr>
<td>Non-structural Programmatic</td>
<td>Building codes</td>
</tr>
<tr>
<td>Measures</td>
<td>Implementation of flood damage prevention ordinances</td>
</tr>
<tr>
<td></td>
<td>Education</td>
</tr>
</tbody>
</table>

Figure 7.2-5. Examples of coastal protection and restoration projects in central and southeast coasts of Louisiana (Coastal Protection and Restoration Authority, 2012)
Zoning

The land use and zoning regulations in the Louisiana’s Coastal Master Plan 2012 (Coastal Protection and Restoration Authority, 2012) aim to ensure that construction of new flood protection structures will not encourage further development in high-risk coastal areas (induced risk of coastal protection). They also need to ensure that coastal wetlands in high-risk areas remain intact. As Table 7.2-1 indicates, the Louisiana’s Coastal Master Plan 2012 provides different non-structural protection strategies depending on the elevation or depth of flood in coastal areas:

- **Elevation:**
  ‘This option involves raising residential structures so that their lowest floor is higher than projected flood depths. This measure was considered for areas with a projected flood depth of between 3 and 18 feet’ (Coastal Protection and Restoration Authority, 2012, p.72)

- **Flood proofing:**
  ‘This option retrofits structures so they can be resistant to flood damage. Residential and commercial flood proofing was considered for areas with projected flood depths of 3 feet or less’ (Coastal Protection and Restoration Authority, 2012, p.72)

- **Voluntary Acquisition:**
  ‘This option applies to areas where projected flood depths make elevation or flood proofing infeasible and where residential structures would need to be elevated higher than 18 feet’ (Coastal Protection and Restoration Authority, 2012, p.72)

However, in the case of voluntary acquisition, the plan provides for public participation and engagement in developing and refining the options that suit the communities’ requirements. The 15-page LA SAFE strategy 2015 (Louisiana Office of Community Development Disaster Recovery Unit, 2015) focuses on the following three typological development zones in Louisiana and outlines the main principles for each zone. However, the actions and projects that will be implemented in each zone are not indicated in the strategy. The zones include (Louisiana Office of Community Development Disaster Recovery Unit, 2015):

- **Resettlement zones:**
  Areas projected to experience flood inundation over 14 feet above base flood levels in a 100-year storm event over the next 50 years.

- **Retrofit zones:**
  Areas projected to experience between 3 feet and 14 feet of flood inundation in a 100-year storm event over the next 50 years.

- **Reshaping zones:**
  Areas projected to experience less than 3 feet of flood inundation in a 100-year storm event over the next 50 years.

**Resilience retrofit programme**

As a non-structural measure for coastal protection, the Resilient New Orleans 2015 has set out a goal to ‘incentivise property owners to invest in risk reduction’ and has identified an action to establish a ‘Resilience Retrofit’ programme. This programme uses a similar financing structure as is currently being used in energy efficiency programmes (e.g. PACE programme) and offers low-interest loans to property owners to retrofit their buildings. The loans are repaid through property tax bills over a period of 20 years and reduction in insurance premiums serves as an incentive to encourage property owners to improve the flood resiliency of their homes (City of New Orleans, 2015). This programme is planned to be implemented in the Gentilly neighbourhood which is selected as the city’s first Resilience District.
The city is working to expand an energy-efficiency programme through partnership with Deutsche Bank in order to provide private finances for homeowners to invest in stormwater management and retrofitting practices (LaRose, 2016). Under this programme, property owners with less than 80 percent of the area's median income are eligible to receive up to $5,000 to retrofit their properties. The appropriate retrofitting and mitigation options for each property need to be identified through consultation with a programme consultant or expert (LaRose, 2016). Funded by a federal grant\(^\text{128}\), the projects in the Gentilly Resilience District aim to retain and control flood water (that previously had been kept behind levees and floodwalls), reduce land subsidence and encourage neighbourhood revitalization. Greater use of green infrastructure (swales and rain retention gardens) and implementing resiliency retrofit programmes are identified as examples of these projects (LaRose, 2016).

### 7.2.3.4. Initiatives for integrating climate change services of coastal natural features into climate change strategies/plans

**Incorporating coastal wetlands into the coastal protection strategies**

Prior to the Hurricane Katrina, the wetland restoration (under the Coastal Wetlands Planning, Protection and Restoration Act, CWPPRA, 1990) and the initiatives for coastal protection against hurricanes and other hazards were discrete areas of activity, with the coastal protection measures principally based on hard structures (Coastal Protection and Restoration Authority, 2007). The approach for protection and management of wetlands prior to Hurricane Katrina was simply to avoid further loss of Louisiana’s wetlands, rather than coastal protection. The rapid loss of coastal wetlands in Southern Louisiana which once acted as a buffer to protect the coastline from the destructive impacts of storm surge and hurricanes, increased attention towards wetlands restoration particularly after the disastrous flooding of Hurricane Katrina (Day, et al., 2007; Barbier & Enchelmeyer, 2014). A number of scientists argued that since the coastal wetlands and barrier islands in the Mississippi Deltaic Plain (MDP) are the first line of defence against storm surges, the effects of future hurricane surges will worsen in South Louisiana particularly in New Orleans if the current trend of coastal wetland loss continues (Tibbetts, 2006).

Therefore, restoration of coastal wetlands as well as barrier shoreline restoration and ridge habitat restoration were included within the coastal protection strategies provided in the Louisiana’s Coastal Master Plans 2007 and 2012 (Coastal Protection and Restoration Authority, 2007&2012) and the recent Resilient New Orleans 2015 (City of New Orleans, 2015). The coastal protection strategies built on the ‘multiple lines of defence’ concept which requires using natural features to complement structural measures such as floodgates, seawalls and levees. The Louisiana’s Coastal Master Plan (2012) has included 28 marsh creation and restoration projects as part of the non-structural coastal protection initiatives along Louisiana’s coastline. These projects are planned to be implemented in two phases (i.e. 2012-2032 and 2032-2061). Out of the $50 billion budgeted for the Master Plan, the highest budget (i.e. $20 billion) is allocated to marsh creation projects (Coastal Protection and Restoration Authority, 2012).

The coastal wetland restoration projects in South Louisiana are mainly funded under the CWPPRA (1990). The federal Coastal Impact Assistance Programme (CIAP) also provides financial support to coastal wetland restoration activities in Louisiana (Coastal Protection and Restoration Authority, 2012). Different techniques including ‘sediment trapping’ and ‘beneficial use of dredged material’ are used in the wetland restoration projects under the CWPPRA\(^\text{129}\). These techniques are based on using sediments from rivers and maintenance dredging activities to restore or re-build coastal wetlands. An example of a wetland restoration project using the beneficial use of dredged material technique is ‘CWPPRA Project BA-39 Mississippi

\(^{128}\) $141 million grant through the HUD National Disaster Resilience Competition (NDRC) in January 2016

River Sediment Delivery – Bayou Dupont’ that used sediment from the Mississippi River to create new marsh wetlands that had turned to open water (Figure 7.2-6).\(^{130}\)

![Creating new marsh wetlands in ‘CWPPRA Project Bayou Dupont’ using the restoration technique known as the beneficial use of dredged material. The red boat near the levee is pumping sediment to the fragile wetlands in top left of the image.](image)

As indicated before, the Resilient New Orleans 2015 provides for developing a climate action plan to achieve the city’s emission targets. It is mentioned that the action plan will also augment the city’s adaptation strategies already underway as many of them such as wetland restoration, green infrastructure enhancement, and transportation improvements can provide mitigation benefits (City of New Orleans, 2015).

**Evaluating the impacts of coastal protection strategies on carbon sequestration**

The protection and restoration projects proposed in the Coastal Master Plan (2012) were evaluated in terms of their potential impacts on a set of proposed criteria (e.g. ecosystem services) over a period of 50 years (from 2010 to 2060). The impacts of the proposed protection and restoration projects (e.g. marsh creation/restoration) on carbon sequestration (as one of the ecosystem services provided by coastal habitats) were estimated through the Wetland Morphology model (Coastal Protection and Restoration Authority, 2012). The impacts of protection and restoration projects on maintaining/building land (i.e. different projects could change land and water areas and their fractions in a pixel) and vertical accretion/surface elevation (i.e. different projects could change mineral and organic matter accumulation) were identified and used to estimate changes in the soil organic carbon (SOC) storage within a certain depth (e.g., 1 m) and SOC sequestration under different projects (Eq. 1&2).

**SOC storage over 1-m depth of soil:**

\[
SOC \text{ (cell)} = (BD \ast OM\%/2.2) \ast 100cm\ast 25ha\ast \%Land
\]

\(SOC\): soil organic carbon total amount (metric tonnes per 500m grid cell)

\(BD\): soil bulk density (g/cm\(^3\))


OM%: organic matter content
2.2: conversion factor from SOM to SOC for coastal Louisiana wetlands (SOM% = 2.2 * SOC%)
100cm: top 1 metre soil depth
25ha: area of each 500m x 500m pixel
%Land: land percentage in the 500m grid cell from the landscape change sub-model

**SOC sequestration potential from elevation change:**
\[
\Delta SOC \text{ (cell)} = (BD \times \frac{OM\%}{2.2}) \times H
\]  
(Eq. 2)

\(\Delta SOC\): SOC sequestration rate due to elevation change (tC/ha/yr)

H: vertical accretion rate (cm/yr)

Projects will also change distribution of vegetation types (e.g., freshwater, brackish, saline marshes, swamp forest) due to changes in salinity and inundation regimes. Soil bulk density (BD) and organic matter (OM) are considered as a function of vegetation types and thus subject to change under different projects. Updated information on BD and OM were therefore used to estimate SOC storage and sequestration. The information was obtained from soil data provided by the Coastwide Reference Monitoring System (CRMS)\(^{132}\).

CRMS is an ecosystem monitoring programme that was developed to address the mandates of the CWPPRA (1990) with respect to monitoring the effectiveness of all CWPPRA restoration projects (Steyer, et al., 2003). The programme provides empirical information for 390 reference sites that can be used to evaluate the cumulative effects of individual restoration projects on the ecosystem (Coastal Protection and Restoration Authority, 2007). The information is available online through the CRMS website and its spatial viewer maps (GIS viewer maps; Figure 7.2-7). As examples of data provided for Louisiana’s wetlands by the CRMS include organic material, bulk density and surface elevation/accretion (SVI: Submergence Vulnerability Index). SVI represents the vulnerability of a site to submergence based on the site’s elevation and relative sea level rise.

Figure 7.2-7. A screenshot of the information provided by the CRMS developed for Louisiana’s coastal wetlands\(^{133}\)

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\(^{132}\) https://lacoast.gov/crms2/home.aspx

\(^{133}\) https://lacoast.gov/crms2/home.aspx
Introducing blue carbon to the carbon market to support coastal wetlands restoration

In September 2012, the American Carbon Registry (ACR) certified the very first modular Wetland Carbon Offset Methodology (Restoration of Degraded Deltaic Wetlands of the Mississippi Delta v2.0), for generating and monetizing blue carbon credits from wetland restoration activities in the Mississippi Delta (Louisiana, Texas, Mississippi)\textsuperscript{134}. This methodology was developed by a New Orleans-based environmental consulting company, Tierra Resources that is actively working in the field of wetlands restoration. In addition to the carbon sequestered by mangrove forests, seagrass beds, and salt marshes, blue carbon in Louisiana also includes carbon in the sediment and biomass of tidally influenced cypress-tupelo forests and freshwater marshes (Mack, et al., 2014).

The methodology was aimed to create a financial mechanism to support coastal wetland restoration activities in the Mississippi Delta\textsuperscript{135}. This methodology was used to quantify the increase in the amount of carbon sequestered and stored in biomass (aboveground, belowground), and soil (i.e. organic carbon) over and above the baseline scenario. Changes in carbon dioxide, methane or nitrous oxide (attributable to the project activity), is also quantified and deducted from net emission reductions (Tierra Resources, 2012). It also quantifies GHGs emission as a result of wetlands degradation and erosion of the wetland soil horizon (specifically the top 50 cm) (Tierra Resources, 2012).

Under this methodology, applicants (e.g. landowners) can document and quantify the GHGs benefit of their wetland restoration projects and sell the certified offset credits in the voluntary carbon market (Tierra Resources, 2012). The methodology allows applicants to use empirical methods as well as data from peer-reviewed scientific literature to estimate carbon offset credits, provided that such data complies with the valid sampling and statistical procedures outlined in the methodology (Tierra Resources, 2012). In case of using secondary data, applicants are required to take into account the associated uncertainties and estimate the net anthropogenic GHGs removal in a conservative manner (Tierra Resources, 2012).

The first Louisiana wetland restoration project that used the wetland restoration methodology is currently underway in a privately-owned coastal cypress swamp west of New Orleans in St. Charles Parish\textsuperscript{136}. The project is known as the ‘Luling Oxidation Pond Wetlands Assimilation System’ and uses treated municipal wastewater to restore a 950-acre wetland which is at risk of subsidence (sinking) and saltwater intrusion. The project is the first project in Louisiana that demonstrates the true costs and benefits associated with commercial wetland carbon practices\textsuperscript{137}. As indicated by Tierra Resources, the net carbon sequestration as a result of this project, is expected to be between 1,000 – 7,000 metric tonnes of CO\textsubscript{2}e per year\textsuperscript{138}. Carbon credits from the project will be issued once the results are audited and validated by an independent verifier.

The findings of a two-year assessment indicated that restoration of coastal wetlands in Louisiana can potentially produce over 1.8 million offsets (carbon credits)\textsuperscript{139} per year (Mack, et al., 2014). This value is estimated based on the difference between the carbon sequestration rate of an approved baseline (business as usual) and the carbon sequestration rate resulting from the restoration activity (Mack, et al., 2014).


\textsuperscript{139} A carbon offset or a carbon credit is ‘a metric ton reduction in emissions of carbon dioxide or greenhouse gases made in order to compensate for, or to offset, an emission made elsewhere’ (Mack, et al., 2014. P.12).
Depending on the dollar value of the carbon credits, restoration of Louisiana’s coastal wetlands can provide a total of $540 million (at $4.40 per offset) to almost $1.6 billion (at $10.80 per offset) carbon finance revenue over a period of 50 years (Mack, et al., 2014). This revenue can be used to support wetland restoration efforts in Louisiana. The estimates provided by Mack et al. (2014) are based on data from 47 peer-reviewed literature on soil and biomass carbon sequestration and GHGs emission from various wetland types in the Mississippi River deltaic plain, as well as other areas of the world (Mack, et al., 2014).

Restoration of forested wetlands by using treated municipal effluent (known as wetland assimilation systems) showed the highest net offset yield per acre; while river diversions and mangrove plantings indicated the potential to generate the largest amount of carbon credits in Louisiana as they can be applied over extended areas (Mack, et al., 2014).

The first pilot mangrove plantation project in Louisiana successfully implemented aerial planting of black mangrove (*Avicennia germinans*) seeds in a privately-owned saltmarsh wetland to restore the wetland for protection against erosion, hurricane surge and increasing wetland’s carbon storage (blue carbon)\(^{140}\). The high cost of wetland restoration (between $20,000-$150,000 per acre) which far exceeds the potential carbon revenue in some cases, is identified as the primary barrier to wetland carbon commercialisation in this pilot project (Mack, et al., 2014). Therefore, it is suggested that large-scale carbon finance projects need to be leveraged with traditional restoration funding programmes to encourage private investment in wetland restoration projects (Mack, et al., 2014). However, given that 80% of wetlands in Louisiana are privately owned, this may cause challenges in terms of using state and federal budget in projects which provide private benefits.

Under the carbon finance programme and in order to encourage landowners to participate in carbon financing projects, outreach programmes were implemented during 2012 and 2013 to provide information about emerging opportunities to restore wetlands via carbon offset.

**Planting mangroves for coastal protection and carbon offsetting potential**

In August 2015, the successful results of a three-year mangrove plantation project in southeast Louisiana was publicised (The Climate Trust, 2015). The pilot project successfully implemented aerial planting of black mangrove (*Avicennia germinans*) seeds in a privately-owned saltmarsh ecosystem across Terrebonne and Lafourche parishes\(^{141}\). The objective was to restore the wetland for protection against erosion, hurricane surge and increasing wetland’s carbon storage (blue carbon). Air seeding of mangroves (by crop duster airplane) was tested for the first time in this project and was introduced as a cost-efficient technique (approximately 3% of the cost of conventional restoration techniques) for large-scale wetland restoration activities (The Climate Trust, 2015). Mangrove were selected for wetland restoration due to their ability to thrive in saltwater environments while providing a wide range of ecosystem services such as carbon sequestration, erosion control, storm surge protection and habitat protection\(^{142}\). The technique used in this project has the potential to be expanded to other areas of coastal Louisiana and around the world (The Climate Trust, 2015). A similar seeding project, supported by Entergy Corporation was planned to be conducted on a 60-acre site in 2016\(^{143}\).


7.2.3.5. Key initiatives

The key initiatives in New Orleans include:

- Comprehensive Coastal Master Plans were developed. The design and implementation of coastal protection measures were based on the concept of ‘multiple lines of defence’ which involves combining natural features and structural measures.
- Wetland restoration was integrated with flood protection initiatives to provide effective coastal protection.
- Land use planning was changed to encourage smart growth in coastal areas (e.g. maintaining a buffer zone near levees, discouraging unwise development in low-lying and flood-prone areas and protecting and restoring wetlands particularly in high-risk zones).
- Coastal Louisiana was classified into three main categories including Reshape, Retrofit, and Resettlement zones, based on their vulnerability to flood inundation.
- The Wetland Carbon Offset Methodology was developed and the methodology was applied to a pilot restoration project.

7.3. Tamil Nadu State, India

7.3.1. Characteristics of the urban landscape and its vulnerability to climate change

Tamil Nadu is one of the India’s coastal states which is situated in the southern part of the country and contains approximately 1076 km of coastline. The state is bounded by the Indian Ocean on the south and the Bay of Bengal on the east (Government of Tamil Nadu, 2015). It is classified as a highly vulnerable coastal state and has been struck by 44 cyclones between 1891 and 2011, of which 30 were severe cyclonic storms (Government of Tamil Nadu, 2015). The state of Tamil Nadu was one of the areas affected by the Indian Ocean Tsunami in 2004 and suffered extensive damage particularly in its coastal districts (Shaw, 2015). Post-tsunami observation revealed that villages (particularly those around the Peechavaram mangrove area) that were protected by extensive cover of mangroves (approximately 1000 m wide) were protected against the tsunami waves (Government of Tamil Nadu, 2015).

Coastal ecosystems (including mangroves) across the state are under increasing pressure from shrimp farming and industrial development (Government of Tamil Nadu, 2015). Studies have also indicated that climate change and human activities will change the extent and composition of mangrove ecosystems in India. The state of Tamil Nadu includes approximately 39 km$^2$ (3,900 ha) of mangrove forests (mainly distributed along the east coast of the state) that are composed of species such as *Rhizophora*, *Bruguiera*, *Excoecaria* and *Avicennia* (Government of Tamil Nadu, 2015).

7.3.2. Governance structure and climate response plans

The Republic of India is a country located in South Asia and is composed of 29 states and 7 union territories\textsuperscript{144}. India is a federal republic with a parliamentary form of government. The hierarchy of administrative divisions in India include state/union territory, district, tehsils, and village. The Constitution of India is the country’s supreme legal documents and governs the parliamentary system (Korabik, et al., 2017). The President of India is the head of state; while the Prime Minister leads the government of India\textsuperscript{145}. Each Indian state is governed by a Governor who is appointed by the President and has all executive powers

\textsuperscript{144} Government of India, Profile. [Online] Available at: https://india.gov.in/india-glance/profile [Accessed December 2016]

\textsuperscript{145} https://india.gov.in/india-glance/profile [Accessed December 2016]
of the state according to the Constitution of India\textsuperscript{146}. Similar to the system of government in states, each union territory is administered by an Administrator who is also appointed by the President\textsuperscript{147}.

The governing authority of the Indian State of Tamil Nadu is the government of Tamil Nadu and the legislature of the state has been unicameral since 1986. The Governor acts as the constitution head of the state while the Chief Minister heads the council of ministers\textsuperscript{148}. India ratified the UNFCCC and the Kyoto Protocol in 1993 and 2002, respectively\textsuperscript{149}. The National Action Plan on Climate Change (NAPCC) that was released in 2008 is the main climate change policy in the country and identifies eight national missions to address climate change mitigation and adaptation (Prime Minister’s Council on Climate Change, 2008). A total of 32 states and union territories have currently prepared the State Action Plan on Climate Change (SAPCC) to incorporate climate change consideration into their planning and decision making processes (Ministry of Environment Forest and Climate Change, 2015). Tamil Nadu State Action Plan on Climate Change (TNSAPCC) was developed through a four-year planning process (from July 2010 to July 2014) and was released in 2015. The TNSAPCC provides a climate response framework to enhance the state’s resilience and adaptive capacity to the impacts of climate change and natural hazards (Government of Tamil Nadu, 2015).

In addition to the NAPCC, the Government of India has developed a wide range of initiatives to decrease GHGs emission and improve adaptation to the impacts of climate change. The mitigation initiatives are mainly focused on improving energy efficiency and promoting the use of renewable energies across the country; while adaptation initiatives mainly provide for improving water quality, water use efficiency and agriculture practices (Ministry of Environment Forest and Climate Change, 2015).

The ‘Smart Cities Mission’ is one of the mitigation strategies that was launched in June 2015 and is planned to be implemented within a five-year period (2015-2020). This programme is built on three main principles including (i) area improvement (Retrofitting) (ii) city renewal (Re-development) and (iii) city extension (Greenfield development) and aims to provide strategic action plans for area developments in 100 cities across the country. The cities are identified based on a number of criteria within two stages; however, vulnerability to coastal hazards is not included in the criteria used to select cities and coastal cities are not given a priority in this programme. Retrofitting in this programme involves identification of core infrastructure gaps in already built up areas and improving sewerage, power supply and conservation of buildings and heritage\textsuperscript{150}.

Property owners in India can insure their properties against different flood events (caused by rain or storm) through their general house insurance policy (PolicyBazaar, 2016). However, flood insurance is not very common or popular in India. The recent flood events have resulted in increased insurance premium in some Indian states (e.g. Kashmir) (Parvaiz, 2015) and considering the options to identify insurance premium based on the flood risk potential of different regions (Business Standard, 2015).

7.3.3. Adaptation to sea level rise and coastal hazards

\textit{Zoning regulation}

Under the Environmental Protection Act (1986) of India, vulnerable coastal areas (i.e. coastal land up to 500m from the high tide line and a stage of 100m along banks of creeks, estuaries, backwater and rivers subject to tidal fluctuations) are identified as ‘Coastal Regulation Zones’ (CRZs) where establishment and expansion of industries, operations and processes are restricted (Ministry of Environment and Forests,
CRZs are classified into four zones and different level of restriction applies to construction and development activities in each zone. The zones include (Ministry of Environment and Forests, 1991):

CRZ-I: the area between low and high tide line that includes ecologically sensitive ecosystems (including mangroves) of the coast. No development activities are permitted in this zone; however, exploration and extraction of oil and natural gas is permitted in areas that are not ecologically sensitive and important.

CRZ-II: areas that have already been developed up to or close to the shoreline. Buildings are allowed only on the landward side of the existing road or on the landward side of existing authorised structures.

CRZ-III: Areas that are relatively undisturbed and those which fall outside the categories I or II. The area up to 200 metres from the high tide line is identified as ‘No Development Zone.

CRZ-IV: Coastal stretches in the Andaman and Nicobar, Lakshadweep and small islands, except those designated as CRZ-I, CRZ-II or CRZ-III. New construction of buildings is not permitted within 200 metres of the high tide line.

In response to the damage caused by the 2004 tsunami, it was suggested that a vulnerability line is identified for coastal areas and development activities will be regulated on the seaward side of that line (Ministry of Environment and Forests, 2008). Factors including elevation, geology, geomorphology, sea level trends, horizontal shoreline displacement, tidal ranges and wave heights were suggested to be considered in identifying the vulnerability line. Consequently, CRZ Notification 2011 (which replaced CRZ Notification of 1991) provided for mapping the hazard line along the coastline of the country. The mapping process was initiated in 2012 and is planned to be completed by 2017 (The Times of India, 2013). It is one of the components of the Integrated Coastal Zone Management (ICZM) Project in India that is funded by the World Bank and is ongoing in a number of coastal states. Mapping of the CRZ-I (ecologically sensitive areas) and high tide line for the coastline of India is also in progress (The Times of India, 2016).

**Integrated Coastal Zone Management (ICZM) Project**

An Integrated Coastal Zone Management (ICZM) Project, through cooperation with the World Bank, has been initiated in India since 2010 and is planned to be completed by 2017. This project aims to demonstrate the ICZM approach in the pilot states of Gujarat, Orissa and West Bengal in order to build a capacity at the national level for implementation of comprehensive coastal management approach in the country (The World Bank, 2010). As reported by the Tamil Nadu Department of Environment, the state under the World Bank Assistance, has initiated the process of developing an ICZM Plan since 2007 to support decision-making and planning for development in coastal areas and to provide strategies for mitigating coastal hazards. The ICZM Plan has not yet been released and is part of the broad Coastal Disaster Risk Reduction Project (CDRRP) in Tamil Nadu\(^1\).

Under the ICZM project in Tamil Nadu, land use, capability and vulnerability maps were prepared for the entire length of coastline (from the coast to 2.5 km inland at the scale of 1:5000) and are being used to identify the setback lines in coastal areas\(^2\). Seven parameters including elevation, geology, geomorphology, sea level trends, horizontal shore line displacement (erosion /accretion), tidal ranges and wave heights were considered to evaluate the vulnerability of coastal areas\(^3\). The significant

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\(^1\) Tamil Nadu Department of Environment, *Coastal Disaster Risk Reduction Project (CDRRP)*. [Online] Available at: [http://www.environment.tn.nic.in/etrp.html](http://www.environment.tn.nic.in/etrp.html) [Accessed 2016].


environmental issues of the Tamil Nadu state, including mangroves degradation, have been also addressed in the ICZM Plan through an inter-sector impact assessment.

### 7.3.4. Initiatives for integrating climate change services of coastal natural features into climate change strategies/plans

**Protection and enhancement of mangroves for increasing carbon sinks and coastal protection**

The national mission for a green India mainly provides for increasing the extent and density of forests throughout the country to protect a wide range of ecological services particularly biodiversity and carbon sequestration (Prime Minister’s Council on Climate Change, 2008). This mission has been further expanded and addressed in a separate document prepared by the Ministry of Environment and Forests (2010). As indicated in this document, the scope of greening in this mission is not only limited to plantation and trees, but also includes restoration of coastal ecosystems and habitats. For example, restoration of mangroves (up to 0.1 million hectare) is identified as one of the action plans to achieve the targets of the green India mission (Ministry of Environment and Forests, 2010).

During 2006-2007, the Tamil Nadu Forest Department planted mangroves over an area of 1,650 ha along the coastline to enhance coastal protection (Government of Tamil Nadu, 2015). The National Cyclone Risk Mitigation Project (which is implemented by the Government of India with the assistance of World Bank) supports shelterbelt and mangrove plantations in Tamil Nadu. Consistent with the green India national mission, the TNSAPCC 2015 has identified specific strategies for mangrove plantation and restoration (developing bioshield) for improving coastal protection and enhancing carbon sequestration. The plan also provides for a research project to identify carbon sequestration potential of all forest types across the state (Government of Tamil Nadu, 2015). A bio-shield project by SEEDS India was also implemented along the coast of Lighthouse Panchayat in Tamil Nadu in 2008 to strengthen resilience of the local communities to natural hazards through establishing a multi-layered and multi-species bio-shield154.

A blue carbon project is currently being implemented at Sundarbans mangrove forest (the largest mangrove forest on Earth between West Bengal in India and Southern Bangladesh) and aims to plant about 4,600 ha of mangroves that are expected to sequester approximately 1,000,000 tCO₂e (net sequestration) over a 20-year period (≈ 51,000 tCO₂e per year) (Verified Carbon Standard, 2015b). These values are ex-ante estimations and are based on a conservative annual net carbon sequestration rate of 11.1 tCO₂e/ha which includes above- and below-ground biomass, dead wood and soil organic carbon (SOC) (Verified Carbon Standard, 2015b). The project documents available on the VCS website155 do not clearly state how the value of 11.1 tCO₂e/ha is estimated, however, there are references to both global and local scientific literature within the project’s verification report (Verified Carbon Standard, 2015c). The global studies by Chmura, et al., (2003) and Murray, et al., (2011) are included within the list of studies used in this report. These studies were also used in this research as sources for information on CS&S by coastal wetlands.

Net GHG emission reductions or removals in this project are calculated under the Clean Development Mechanism (CDM) methodology (CDM AR-AM0014, Version 3)156 and are validated against the VCS Standard Version 3.5 (Verified Carbon Standard, 2015a). Following the project validation in 2015, the verified carbon credits are issued to Livelihoods (the project’s funding company) by the UNFCCC (Wylie, et al., 2016).

The Sundarbans restoration project is classified under the ‘Agriculture, Forestry, and Other Land Use’ (AFOLU) projects, the category of Afforestation, Reforestation, and Restoration (ARR) for mangroves

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156 Clean Development Mechanism, A/R Large-scale Methodology Afforestation and reforestation of degraded mangrove habitats, AR-AM0014 Version 03.0.
(Verified Carbon Standard, 2015b) and is primarily aimed to generate carbon offset credits (or emission reduction) as well as to enhance protection against coastal hazards (Wylie, et al., 2016). The project has also supported local livelihood by providing job opportunities for coastal communities (Wylie, et al., 2016).

**Mangroves For the Future (MFF)**

The 2004 tsunami led to a reaffirmation of the important role of mangroves and shelterbelt plantation in reducing the tsunami damage to coastal areas (Government of Tamil Nadu, 2015) and was the main impetus for developing the ‘Mangroves For the Future’ (MFF) regional initiative in 2006. MFF is coordinated by IUCN (International Union for Conservation of Nature) and UNDP (United Nations Development Programme) and seeks to enhance and build resilience to coastal hazards through supporting mangrove plantation and restoration activities. MFF activities are focused in countries including India, Indonesia, Maldives, Seychelles, Sri Lanka, Thailand, Bangladesh, Cambodia, Myanmar, Pakistan and Vietnam.

The MFF activities in each country is coordinated and managed through a National Coordinating Body (NCB). MFF in India seeks to improve management of coastal and marine ecosystems (specifically mangroves and coral reefs) by providing a scientific knowledge base that can support national policy making processes. It also aims to promote sustainable livelihood opportunities for coastal communities. Considering a number of factors including human pressure, pollution, degradation, cyclone, climate change and sea level rise, seascape setting and other unique features, five states in India have been selected as the priority states for the MFF projects. These states include West Bengal, Gujarat, Orissa, Andhra and Tamil Nadu. NCB in India is guided through a National Strategy and Action Plan (NSAP).

The NSAP includes two main categories (i) Conservation Strategy and Action Plan for mangroves, and (ii) Restoration Strategy and Action Plan for potential and/or degraded mangrove areas. It has identified four priority areas including ‘(i) Environmentally sustainable livelihoods to reduce pressure on coastal ecosystems, (ii) Plantation of mangroves for creating green belts, (iii) Civil society awareness, participation and coastal decision making and sustainable financing and (iv) Improving knowledge gaps.’

### 7.3.5. Key initiatives

The key initiatives in Tamil Nadu include:

- Development of the Tamil Nadu State Action Plan on Climate Change (TMSAPCC) that provides for mangrove plantation and restoration to improve coastal protection and carbon sequestration.
- National regulation for development in vulnerable coastal areas (i.e. Coastal Regulation Zone) that prohibits development in coastal areas containing ecologically sensitive coastal ecosystems (including mangroves) as well as in areas up to 200 metres from the high tide line (i.e. No Development Zone).
- Participation in the regional Mangroves for the Future (MFF) initiative to restore mangrove forest in a number of pilot coastal cities primarily for improving resilience to coastal hazards and the effects of climate change; but also for supporting local livelihood and enhancing biodiversity.
- Considering mangrove restoration as one of the actions to achieve the Green India National Mission that seeks to increase carbon sink potential of the country.

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157 Mangroves for the Future, *Who we are.* [Online] Available at: [https://www.mangrovesforthefuture.org/who-we-are/about/who-we-are/](https://www.mangrovesforthefuture.org/who-we-are/about/who-we-are/) [Accessed 2016].


7.4. Jakarta, Indonesia

7.4.1. Characteristics of the urban landscape and its vulnerability to climate change

The Special Capital Region of Jakarta is located on the northwest coast of Java Island and is the largest and most populous metropolitan district of Indonesia (Gunawan & Pusaka, 2016). Flooding caused by sea water or high tide is the most common climate-related hazard that occurs frequently in North Jakarta (Firman, et al., 2011). North Jakarta experienced a different flood in 2007 and sea water which was rushed through the streets caused inundation (up to 1.5 metres deep) for several days. Land subsidence was identified as the major underlying cause of the 2007 flood which caused extensive damage to coastal lands and claimed the lives of 76 people (Government of Indonesia, 2014).

Around 40% of Jakarta’s land, particularly in North Jakarta is below sea level (Firman, et al., 2011). Estimates indicate that if the current rate of land subsidence (mainly associated with the large scale extraction of groundwater) continues, approximately 80% of North Jakarta will be five metres below sea level by 2030 (C40 Cities, 2016). Land subsidence, rapid population growth, increased climate-related hazards and other challenges such as pollution, have made Jakarta one of the most vulnerable coastal cities to climate change in Southeast Asia (Yusuf & Francisco, 2009).

In Jakarta, mangrove ecosystems along with coral reefs and seagrass beds are found along Jakarta Bay in North Jakarta (Arifin, 2004). Muara Angke, a national wildlife sanctuary and Pantai Indah Kapuk are the last remaining true mangrove ecosystems of Jakarta Bay and consist of mangrove species such as Avicennia alba, Avicennia marina and Rhizophora mucronata (Nur, et al., 2001; Kusmana, 2014). Mangrove forests around Pantai Indah Kapuk area are dominated by Avicennia marina (Yulianda, et al., 2014) and cover a total area of 320 hectares of which 180 hectares is protected (Government of Indonesia, 2014). Mangrove forests in Jakarta are under pressure from coastal development and aquaculture activities and have been significantly diminished over years (Nur, et al., 2001). For example, the conservation area of mangroves in Muara Angke has decreased from 1,335 ha in 1960 to 173 ha in 2001 (Nur, et al., 2001). Pollution, poor water quality, illegal cutting and subsidence are among the other threats to mangrove ecosystems in Jakarta (Government of Indonesia, 2014).

7.4.2. Governance structure and climate response plans

Indonesia, officially the Republic of Indonesia, has a republican form of government including an elected legislature and president 161. The president of Indonesia is both head of state and head of government 162. The hierarchy of administrative divisions in Indonesia include province, regency, city, district and administrative villages 163. Under article 33 of the country’s constitution, ‘land and water and natural resources therein shall be controlled by the State and be utilized for the greatest welfare of the people’ (Idrus, 2009, p.138). Each of 34 provinces in Indonesia has its own legislature, governor and autonomy in determining its own administrations (Djalante & Thomalla, 2012).

Some provinces are vested with greater legislative power and act more independently from the central government than other provinces. This applies to five of 34 provinces in Indonesia including Aceh, Jakarta, Yogyakarta, Papua, and West Papua. These are referred to as provinces with special status. Jakarta has a governor and comprises five administrative cities/municipalities including Central Jakarta, North Jakarta, West Jakarta, South Jakarta and East Jakarta (Firman, et al., 2011).

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The establishment of the National Committee on Climate Change (KNPI) in 1990 was the first attempt to address climate change in Indonesia (Djalante & Thomalla, 2012). In 1992, Indonesia signed and ratified the UNFCCC (Djalante & Thomalla, 2012); however, until 2004 it did not ratify the Kyoto Protocol (Green, et al., 2010). The National Action Plan for Climate Change (RAN-PI) was launched in 2007 and outlined strategies to decrease the rate of environmental degradation and to create development systems that are resilient to climate impacts (Djalante & Thomalla, 2012). In 2008, the National Council on Climate Change (DNPI) was established and was charged with formulation and implementation of climate change-related policies and programmes (Green, et al., 2010).

The National Action Plan on Greenhouse Gas Emission Reduction (RAN-GRK) that was officially established by a presidential decree in 2011 identifies sectoral allocation for meeting the GHGs reduction target of 26% (below the country’s business-as-usual level) by 2020 (Di Gregorio, et al., 2015). The implementation guidelines of the RAN-GRK at national and sub-national (provincial) levels have been also developed by the Ministry of National Development Planning (BAPPENAS) (Di Gregorio, et al., 2015). Improvement, rehabilitation, operation and maintenance of marsh reclamation network (including peatlands) and timber forest plantation are among the actions plans included within the forestry and peatland sector. The National Action Plan for Climate Change Adaptation (RAN-API) was developed in 2014 with the overall aim of mainstreaming adaptation initiatives into the national development planning. Under RAN-API provincial governments are required to develop their local adaptation strategies/plans in accordance with the specific needs and vulnerability of their regions (Ministry of National Development Planning, 2014).

The Spatial Capital Region of Jakarta does not currently have a policy or an action plan which is specifically tailored to climate change (Firman, et al., 2011). However, the first National Capital Integrated Coastal Development (NCICD) Master Plan that was released in 2014, aims to increase the resilience of coastal communities of northern Jakarta to flooding from the sea (Government of Indonesia, 2014). NCICD is referred to as the Jakarta’s climate change adaptation strategy in the 2014 report by Connecting Delta Cities (CDC) global network (Connecting Delta Cities, 2014). The Master Plan was developed through a long-term cooperation between the governments of Indonesia and the Netherlands and included a wide range of structural and non-structural measures to protect the waterfront of North Jakarta (Connecting Delta Cities, 2014).

Damage caused by rainfall and coastal flooding is generally covered through general property insurance policies in Indonesia (Sipahutar, 2013). The new flood insurance policy in Indonesia (adopted in 2013) do not apply the old property classification system (industrial, commercial or residential) to identify insurance premium. Instead, it classifies properties in terms of the risk of exposure to floods (low164, medium165 and high166) and the insurance premium is determined according to a property’s potential risk. For example, the insurance premium for properties in high-risk zones such as North Jakarta is 0.52% of the property’s price (Sipahutar, 2013).

7.4.3. Adaptation to sea level rise and coastal hazards

Coastal Master Plan

The coastal protection strategies outlined in the Jakarta’s Master Plan are classified within two main categories including (i) short-term no regret solutions and (b) principle long-term solutions. Reinforcing existing flood protection structures (e.g. dykes), reducing the rate of subsidence and improving the city drainage system are identified as the short-term urgent actions for reducing the risk of floods in North Jakarta (Government of Indonesia, 2014). Out of three alternatives (abandon, on-shore protection, or off-
shore protection) as long-term solutions, the Master Plan considers off-shore protection as the most robust and practical solution for protecting North Jakarta against future floods and sea level rise.

The selected option requires extensive land reclamation and closure of some parts of the Jakarta Bay. The concept of the Great Garuda (the national symbol of Indonesia; Figure 7.4-1) is chosen in designing the outer sea dyke to maximise flood protection and to create a symbolic icon for the city (Government of Indonesia, 2014). The outer seawall will be developed far off-shore in deep water and requires extensive (approximately 1,250 ha) land creation (reclamation). The environmental impacts of closing off the bay on mangrove ecosystems and marine life are addressed in the plan; however, the plan does not include the adverse environmental effects of the large-scale land reclamation. There are currently many concerns regarding the environmental and social effects of constructing the giant seawall and its inefficiency in slowing subsidence (Jakarta Post, 2015 and 2016). However, the works to create the seawall are currently progressing as the government has considered the project as an important contributor to protect Jakarta against sinking below sea level (Beo Da Costa, 2016).

Jakarta Climate Adaptation Tools (JCAT) is a research project that was initiated in 2011 and was planned to develop tools ‘to assess, compare, and optimise options for climate adaptation of Jakarta as a delta city’ (Connecting Delta Cities, 2014). The project is funded and implemented by the Delta Alliance and Knowledge for Climate (Connecting Delta Cities, 2014). One of the project’s outputs is a coastal economic exposure assessment tool that provides estimates of the economic values of assets that are potentially at risk of coastal floods with different return periods (or exceedance probabilities) (Ward, et al., 2014).

7.4.4. Initiatives for integrating climate change services of coastal natural features into climate change strategies/plans

The national mitigation (RAN-GRK) and adaptation (RAN-API) plans in Indonesia do not currently consider carbon storage and coastal protection functions of coastal wetlands ecosystems; however, the RAN-API provides for improving the quantity and quality of coral reefs and forests in the priority river basin areas to preserve biodiversity (Ministry of National Development Planning, 2014).

**Mangrove plantation for coastal protection**

Closing off Jakarta Bay with an offshore outer seawall is one of the actions suggested in the Master Plan to protect the city from future floods and inundation (Government of Indonesia, 2014). This closure that will be implemented within two phases will turn the bay into a fresh water lake and will accelerate degradation of the remaining mangrove habitats (Government of Indonesia, 2014). To mitigate loss of mangrove ecosystems in North Jakarta and to utilise the coastal protection potential of mangrove forests, the Master Plan has identified a number of locations (outside the dyke) for mangrove plantation and development (Government of Indonesia, 2014). Construction of a waterfront park along the outer edges of the east and west wings of the Great Garuda provides a greenbelt of mangroves along the coastline that act as a natural buffer and offers coastal protection as well as recreational values (Figure 7.4-1). A mangrove park is also designed for the western tip of the Great Garuda western wing for providing a storm protection buffer which can also create recreational and educational opportunities. Mangrove restoration and creation are included in the second phase of flood protection activities (Government of Indonesia, 2014).
A green network including a large city level park, waterfront boulevards, mangroves and wetlands and the city street network is considered as the core spatial design element for the Great Garuda (Government of Indonesia, 2014). The plan has also identified a number of areas (mostly densely populated coastal neighbourhoods) where residents will be re-located to safer places and vegetation will be replaced instead (Figure 7.4-2).

Tree-planting campaigns organised by NGOs and environmental foundations have been conducted in some parts of North Jakarta to re-plant mangroves in order to protect the properties and infrastructure against floods. The example is a three-year mangrove plantation project by AEON Environmental Foundation, initiated in 2011, that has planted 39,000 saplings of mangroves in North Jakarta over two years (2011 and 2012) (AEON Environmental Foundation, 2012). In March 2015, the governments of Indonesia and the Netherlands launched a large coastal safety project that will be implemented at the North Coast of Java over a five-year period (Wetlands International, 2015). The purpose of this project is to reduce the risk of erosion and storms in the coastline of Demak district by building a stable coastline through integration of mangrove restoration, a small-scale hard coastal protection structure and land use practices (Wetlands International, 2015).
Figure 7.4-2. Coastal protection measures proposed for a densely populated coastal neighbourhood (Pluit) in North Jakarta (Government of Indonesia, 2014)

**Blue Carbon Project**

The Research Centre for Coastal and Marine Resources, Ministry of Marine Affairs and Fisheries, is currently undertaking a Blue Carbon Project in eight pilot sites in Indonesia (Poernomo & Pranowo, 2015). This project is aimed to estimate the potential of coastal ecosystems (mangrove and seagrass) in storing and sequestering carbon in the pilot sites and to support development of a national strategy on blue carbon (Poernomo & Pranowo, 2015).

**Mangroves for the Future (MFF)**

Similar to India, Indonesia has also developed a National Strategy and Action Plan (NSAP) to outline the MFF activities throughout the country. The latest NSAP for Indonesia (2012-2015) has identified a special focus on strengthening the small scale, local ecosystems management and restoration projects to improve public awareness of the potential of coastal ecosystems (particularly mangroves) to cope with the effects of climate change and economic opportunities they can provide. District/local level projects also seek to support and promote sustainable livelihood opportunities and increase resilience of coastal communities. Considering the role of coastal wetlands ecosystems (mangroves, saltmarshes and seagrasses) in capturing carbon, the MFF Indonesia also aims to participate and contribute to the national GHGs reduction through restoration of mangrove ecosystems167.

### 7.4.5. Key initiatives

The key initiatives in Jakarta include:

- Development of a Coastal Master Plan and designing and implementing coastal protection measures based on the concept of ‘living with water’.
- Including a green network as the core criterion for the spatial development of the city and providing for mangrove plantation and restoration as part of the coastal protection measures which can provide multiple benefits.
- Development of a Blue Carbon Project at the national level to examine the CS&S potential of mangrove and seagrass ecosystems in a number of pilot sites across the country.

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• Changes in land use particularly in densely populated coastal neighbourhoods (i.e. Constructing dyke, relocating communities to safer places and creating a re-vegetated buffer zone).

7.5. Rotterdam, The Netherlands

7.5.1. Characteristics of the urban landscape and its vulnerability to climate change

Rotterdam is a delta city in the province of South Holland in the Netherlands (Rotterdam Climate Initiative, 2013). Most parts of the city have been historically developed on a low-lying peat which once formed the delta of the rivers Rhine and Meuse (Rotterdam Climate Initiative, 2013). Like many other cities in the Netherlands, Rotterdam has a long tradition of ‘living with water’ and has historically developed innovative solutions to protect the city against the threats of the water from the rivers and especially from the sea. Rotterdam is vulnerable to the effects of climate change particularly extreme rainfall and rising sea level which will cause more frequent flooding of the city (Rotterdam Climate Initiative, 2013).

As a delta city, urban areas in Rotterdam have been historically developed in a way that minimises the risk of flooding. Low-lying areas of the city which are predominantly below sea level (up to 6.67 metres below NAP168 in some areas) are protected by dykes and are mainly included within the inner-dyke urban districts (Rotterdam Climate Initiative, 2013). Outer-dyke areas that encompass most parts of the city (including the main port) have mainly higher elevation than the inner-dyke areas and are not protected by dykes. These areas are protected by a storm surge barrier (Maeslant storm surge barrier); however, due to their direct exposure to the river and the sea, they are highly vulnerable to rising sea level. Risk of flooding has been always taken into account when determining the elevation of new constructions in Rotterdam, especially within the outer-dyke urban districts. For example, the elevation of new developments in the quaysides of the outer-dyke areas ranges between 3 and 5.5 metres above NAP.

Given the differences in the vulnerability of the inner- and outer-dyke areas to flooding, different adaptation measures are designed to protect these areas. A combination of prevention and adaptation measures which also include ‘building with nature’ solutions are proposed to make a multi-level flood protection system in outer-dyke Rotterdam. Examples include adaptive (flood-proof) construction and design of buildings and outdoor areas, green banks and tidal forest improvement and floating buildings. Measures to protect the inner-dyke areas are mainly focused on prevention and aim to reinforce the existing flood defence system and restore the ‘sponge function’ of the city (Rotterdam Climate Initiative, 2013).

7.5.2. Governance structure and climate response plans

Since 1848, the political system in the Netherlands has been a parliamentary representative democracy within a framework of constitutional monarchy169. The constitution of the Netherlands defines the role and position of the monarch as the head of state and regulates the general principles of the Dutch parliamentary system. The parliament consists of two chambers and the government is the executive. The Netherlands has a relatively prolonged and complicated law making procedure with the second chamber of the parliament playing an important role in the law-making process. Likewise, the process for forming the government involves several stages of post-election bargaining, which has been criticised for being too complicated, time consuming and making the incumbent government non-responsive during the negotiations (Netherlands Institute for Multiparty Democracy, 2008).

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168 National Amsterdam Level equal to mean sea level
The Netherlands has a unitary system with three tiers of government. The central government is made up of council of Ministers, which is the deliberative council of the Dutch cabinet. The prime minister as the head of the Netherlands government also leads the largest party of the coalition that support regulated free market, economic liberalism and the ideology of welfare state. The primary duty of the prime minister is to formulate policies and ensure policies are coordinated among various ministers. The government is also responsible for appointing mayors, provincial governors and the judiciary members (Netherlands Institute for Multiparty Democracy, 2008).

The Netherlands’s 12 provinces sit between the central government and municipalities. Each province has its own provincial (regional) council and executive, responsible for a range of duties including town and country planning, environmental management and public transport (Netherlands Institute for Multiparty Democracy, 2008). A municipality is the third and last tier of the government in the Netherlands. The duties of municipalities are exclusively confined to the matters that directly affect their residents. Municipalities provide public services such as social welfare and recreational amenities, and are responsible for building and maintaining local roads and streets in cities. A local plan developed by a municipality regulates land use within its jurisdiction and must be consistent with the related provincial structure plan and need the agreement of the provincial authority to be implemented (Netherlands Institute for Multiparty Democracy, 2008).

Although the state power has been devolved to provinces and municipalities over the recent years, the decisions by municipalities are, to a greater extent, overseen and controlled by provinces. A municipality’s annual budget and any decision with financial repercussions must pass the approval of the supervising province. Both provinces and municipalities are highly dependent on the central government for funding their activities. Public participation is a key element of the policy and planning process in Netherlands (Netherlands Institute for Multiparty Democracy, 2008).

The Netherlands is a party to the UNFCCC and the Kyoto Protocol, ratified them in 1993 and 2002, respectively. As a member of the European Union (EU), the Netherlands’s emission reduction targets should comply with the binding commitments from the EU (i.e. 20% reduction of greenhouse gas emissions in 2020 compared to 1990). In 2009, the country developed a set of climate policies regarding GHGs reduction targets, but it lacks a coherent set of climate rules (laws) to ensure compliance with the EU binding commitments (Peeters, 2013). Following the concerns from the public wanting active contribution of the government to climate change, the judicial body (Hague District Court) has ordered the central government to do something about climate change (Griggs, 2015). The Environmental Management Act (1993) of the Netherlands is the main legislation that addresses GHGs reduction.

Unlike climate change mitigation, the country has adopted a much stronger legislative approach regarding climate change adaptation. The Delta Act that was enacted in 2012 seeks to protect the country against flooding and freshwater scarcity and provides for the establishment of a Delta Commissioner together with a Delta-Committee, a Delta-Fund and a Delta-Programme (Peeters, 2013). The Netherlands’ Climate Agenda (2013) outlines the country’s overall approach and policy framework with respect to climate change mitigation and adaptation. The national initiatives and programmes regarding climate change mitigation are mainly focused on promoting the use of renewable/clean energy sources and include SER Energy Agreement, the Incentive Regulation for Sustainable Energy (SDE+) and Green Deals (Ministry of Infrastructure and the Environment, 2013).

The role of coastal wetlands (mainly saltmarsh wetlands in the Netherlands) for climate change mitigation (i.e. carbon sequestration) is not addressed in the national climate agenda. The agenda does not also explicitly refer to the coastal protection function of coastal saltmarsh ecosystems and does not provide for

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restoration of coastal wetlands for improving protection. It refers to the Delta Programme as the main national initiative that aims to improve adaptation to climate change and coastal flooding.

Climate change initiatives in Rotterdam have been mainly started since 2007 when the city joined the Clinton Climate Initiative\(^\text{171}\) (CCI) and established its own Rotterdam Climate Initiative (RCI). The RCI outlined a goal to reduce Rotterdam’s GHGs emission by 50% (as compared to 1990) by 2025 and make the city 100% climate proof by 2025.

Rotterdam is also one of the participating cities in the C40 Cities Climate Leadership Group and the Connecting Delta Cities (CDC) network and plays a leading role in developing smart adaptation measures for delta cities \((\text{Rotterdam Climate Initiative, 2009})\). For example, Rotterdam has been involved in the process of developing the climate change adaptation and mitigation strategies for cities such as New York, New Orleans, Jakarta, Shanghai, Singapore, Hong Kong, London and Ho Chi Minh City \((\text{Rotterdam Climate Initiative, 2009})\).

With respect to climate change mitigation, the Mitigation Action Programme (a 13-page document) was published in 2010 and identified opportunities to reduce the city’s GHGs emission by 50% by 2025. The opportunities included in the action programme are classified within the three main categories including (i) Energy conservation, (ii) Energy exchange and (iii) Greenification of energy \((\text{Rotterdam Climate Initiative, 2010})\). Making the city greener by increasing the inner-city green spaces (e.g. parks, green roofs) is suggested as one of the opportunities which can cool down the urban environment and result in saving energy \((\text{Rotterdam Climate Initiative, 2010})\).

### 7.5.3. Adaptation to sea level rise and coastal hazards

The national government in the Netherlands holds the key responsibility of water management and climate change adaptation \((\text{Keskitalo, et al., 2014})\). Provincial governments are involved in designing flood and water management strategies and have the main responsibility to implement the strategies \((\text{Keskitalo, et al., 2014})\). There are 24 water districts in the Netherlands that are governed by water boards\(^\text{172}\). The water boards are responsible for maintaining dykes and managing water levels in canals and water quality, whereas managing main rivers, waterways, large lakes and seas is the responsibility of the national minister \((\text{Keskitalo, et al., 2014})\).

The Calamities and Compensation Act 1998 (WTS) provides legal framework regarding disaster loss compensation in the Netherlands. Under this Act, the government aims to compensate the loss (as a result of disasters such as floods) using public resources such as tax income and state loans. However, the damage caused by storm surges is not covered under WTS due to significant financial costs associated with floods and the difficulty of estimating the costs beforehand \((\text{Botzen & Van Den Bergh, 2008})\). The general house insurance excludes coverage for flood related damage; however, damage due to extreme rainfall is currently covered by 40% of the insurance policies \((\text{Keskitalo, et al., 2014})\). The state recognises the role of individuals in preventing and minimising potential flood damage; however, the government is responsible for protecting communities against floods and coastal hazards and acts under WTS to compensate people for the damage suffered \((\text{Keskitalo, et al., 2014})\).

There has been no private flood insurance policy in the Netherlands until 2012 when the Lloyd’s Coverholder Neerlandse scheme started to provide limited coverage (up to 75,000 Euros) for flood damage\(^\text{173}\). Under this scheme, property owners can purchase insurance coverage for water damage caused

\(^{171}\) CCI is an international climate program established by the former President of the United States Bill Clinton and aims to unite the world’s megacities to take actions to tackle climate change and to reduce GHGs emission (Rotterdam Climate Initiative, 2009; Geerlings & Van Duin, 2011).


by flood, earthquake, or terrorism attacks (O'Callaghan, 2016) and premium discounts will be applied to policyholders who take measures to retrofit (flood-proof) their home.\(^{174}\) The Neerlandse online underwriting tool\(^ {175}\) allows the general public to find out the level of risk (high or low) to their properties and the measure they can take to reduce high risks (O'Callaghan, 2016).

The Dutch insurance association has been exploring the possibility of designing a flood insurance scheme so that the state has only a minimal role or even no role in compensating for the damage caused by floods (Keskitalo, et al., 2014). However, attempts to develop a broader flood insurance coverage have failed so far. It is argued to be associated with the ‘extreme low-probability/high-impact nature of flood risks in the Netherlands that results in relatively high premiums for limited commercial flood insurance coverage’\(^ {176}\).

The government’s proposal for a compulsory flood insurance scheme has not been successful to gain public support due to it being seen as an unfair way of taxation (O'Callaghan, 2016). It is also argued that compulsory insurance may adversely affect risk awareness and mitigation, given that people generally do not have incentive to understand risks or to take mitigation measures (O'Callaghan, 2016).

**Delta Programme and Rotterdam Climate Proof Programme**

The Netherlands has adopted a strong and coherent approach in improving the country’s adaptive capacity to coastal inundation and the effects of climate change. Since 2010 and under the direction of the Delta Programme Commissioner, central government and municipalities as well as provinces and district water boards have been jointly involved in developing and identifying appropriate Delta Decisions and preferential strategies with the aim of making the country water-robust by 2050 (Ministry of Infrastructure and the Environment and Ministry of Economic Affairs, 2015). Six Delta Programmes are published so far and each includes regional sub-programmes for six regions across the country. The first and the second Delta Programme reports that were published in 2011 and 2012 respectively were aimed at exploring possible adaptation strategies; while the 2013 and 2014 reports identified promising strategies. The 2015 and the 2016 (the latest) Delta Programmes have formulated the preferred strategies (van Loon-Steensma, 2015).

The Rotterdam Climate Proof (RCP) Programme was launched in 2008 and focused its activities on three main categories including (i) Knowledge development, (ii) Implementation of climate change adaptation measures and (iii) Presenting Rotterdam internationally as an innovative delta city (Rotterdam Climate Initiative, 2009; Geerlings & Van Duin, 2011). Under the RCP programme, the knowledge developed within the context of the national research programme ‘Knowledge for Climate’ and the national ‘Delta Programme’ was used to prepare the Rotterdam Climate Change Adaptation Strategy in 2013 (Rotterdam Climate Initiative, 2013).

The Rotterdam Climate Change Adaptation Strategy (2013) sets out a vision to make Rotterdam a ‘climate proof city’ by 2025; while other functions of the city, as being a safe, attractive and economically strong place to live, are also maintained and improved. The strategy defines adaptation as the ‘solutions being found in all aspects of the urban environment that make it possible to alleviate the system and make it more resilient’ (Rotterdam Climate Initiative, 2013, p.7). Strengthening the city’s existing flood defence systems and encouraging green-blue adaptation (e.g. Green Team initiative)\(^ {177}\) which requires greater use of ecosystem services are considered as the core design principles that guide adaptation measures for the city of Rotterdam.

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\(^{175}\) http://neerlandse.nl/verduidelijking/over-neerlandse/


\(^{177}\) The local ‘Green Team’ initiatives with the motto of ‘Paving out, Plants in/Tile out, Green in’ encourage people to use more plants in their gardens.
Incorporating coastal wetlands into the coastal protection strategies

The Wadden Sea, located in the central part of the Netherlands’ Wadden region, is one of the world’s largest intertidal areas and includes the main saltmarsh habitat in the country (van Loon-Steensma, 2015). Due to the protective function of its barrier islands and wide intertidal flats, it plays a key role in protecting the Dutch mainland against flooding. The first (2011) and the second (2012) Wadden Region Delta Programmes (as one of the six regional sub-programmes of the Delta Programme) recognised the importance of saltmarsh ecosystems in attenuating wave energy and included an activity to investigate this potential (van Loon-Steensma, 2015). Since then, a number of research projects examined the effectiveness of saltmarshes along the Wadden Sea coast in reducing the wave height and energy. Those studies have mainly focused on modelling; however, due to the lack of sufficient funding, their results have not been verified through field measurement (van Loon-Steensma, 2015).

With respect to the findings of research projects and extensive literature review, saltmarsh habitats were incorporated into the preferred strategies for the Dutch Wadden region in 2014 and were included as green elements in a major dyke research project conducted by the northern water boards. It is consistent with the ‘Building with Nature’ approach originally developed by the Dutch that promotes the use of ecosystems and natural processes within development of hard engineering infrastructure (van Loon-Steensma, 2015).

However, some concerns have been raised by a number of Wadden Sea experts arguing that the integration of saltmarsh ecosystems into flood defence strategies could not be based on the modelling that had not been verified by field measurement (van Loon-Steensma, 2015). In response to these concerns, several other research programmes are underway to evaluate and calibrate the existing models under the Wadden Programme. For instance, the process of comparing the projected wave height reduction (from modelling analysis) with observed heights on dykes after storms has already initiated.

Another major concern questions the possibility of trade-offs between protection and restoration of saltmarsh ecosystems for biodiversity conservation and doing the same for the purpose of coastal protection. This concern is based on the argument by the Wadden experts that the extent and features of saltmarsh ecosystems that are required for coastal protection are in part different from those required for biodiversity protection (van Loon-Steensma, 2015). For instance, while the biodiversity conservation outcomes can be best achieved by maintaining a relatively low-lying and dynamic saltmarsh, effective protection during storm events requires a stable saltmarsh with relatively higher surface (van Loon-Steensma, 2015).

To protect the coastal protection function of saltmarsh ecosystems, the latest Wadden Region Delta Programme (2016) has included activities such as beach replenishment (nourishment) and increasing the natural sediment accretion through adapted saltmarsh management (Ministry of Infrastructure and the Environment and Ministry of Economic Affairs, 2015). Under the Climate Buffer Coalition, which is based on the concept of ‘Natural Climate Buffer’, seven Dutch nature conservation organizations are working to develop climate buffers (i.e. nature areas specially designed to reduce the consequences of climate change) and nature-based protection measures at different locations across the country to make the Netherlands more resilient and more beautiful (www.klimaatbuffers.nl).

Green-blue adaptation

The current flood protection system of Rotterdam is mainly comprised of extensive networks of dykes, storm surge barriers and canals. While the strategy provides for maintaining and strengthening of the existing robust system; it also incorporates green-blue adaptation measures to improve protection from flooding and to enhance the capacity of the city to act as a ‘sponge’ and store water during flood events. These measures are designed based on the area-specific requirements and differ between inner-dyke and
outer-dyke urban districts. Examples include green roofs and façades, blue roofs, bioswales, porous paving stones, water squares (plaza) and tidal parks for the inner-dyke areas and building with nature approaches such as improving tidal forests and green banks for the outer-dyke urban areas (Rotterdam Climate Initiative, 2013).

Incorporation of green-blue adaptation measures into the flood protection measures is argued to support the goal for a waterproof city which is robust and consists of grey, blue and green patches with a mix of paving and vegetation. It will also help to make the city a more attractive place to live in. The appropriate use of ecosystem services is identified as a win-win (no regret) measure which will not only contribute to making the city less vulnerable and more resilient, but will also provide a wide range of other benefits (Rotterdam Climate Initiative, 2013).

Results of a study carried out by the Knowledge for Climate programme which was used to determine the green adaptation measures in the Climate Change Adaptation Strategy, identified ‘riparian wetlands restoration/creation’ as one of the green adaptation measures for Rotterdam (Knowledge for Climate Programme Office, 2012). The study concluded that restoration or creation of riparian wetlands (i.e. wetlands adjacent to rivers and subject to sporadic or seasonal flooding) can directly contribute to flood protection as they can attenuate wave energy, reduce current velocity and increase water storage. There is an initiative in Rotterdam that encourages conversion of old, vacant harbours into estuarine habitats (Knowledge for Climate Programme Office, 2012). Creation of a tidal park as a green measure is proposed in the climate change adaptation strategy to reinforce a dyke in Stadshavens (i.e. city harbour) (Rotterdam Climate Initiative, 2013).

### 7.5.5. Key initiatives

The key initiatives in Rotterdam include:

- Development of a climate change adaptation strategy that provides area-specific adaptation measures, based on the vulnerability of different parts of the city to the effects of climate change.
- Development of comprehensive national adaptation strategies under the Delta Program through cooperation of central government, municipalities and district water boards.
- Incorporation of ‘green-blue’ adaptation measures (e.g. creation/restoration of riparian wetlands) into the measures proposed to make the city climate proof.
- Water-driven urban development that considers risk of flooding in spatial development of the city.
7.6. Summary of the findings

7.6.1. Incorporating adaptation benefits of coastal wetlands into coastal protection

The cases have adopted various but mainly similar approaches to improve resilience to sea level rise and coastal hazards and to protect and restore coastal wetlands for enhancing coastal protection and carbon sequestration. The hurricanes Katrina (2005) and Sandy (2012) were the drivers for preparation of the comprehensive coastal protection plans in New York and Louisiana. They were also the key drivers for incorporation of the coastal wetlands into the coastal protection strategies.

Preservation and restoration of wetlands was part of the policies and plans in New York and Louisiana even before hurricanes Katrina and Sandy. Enactment of the ‘Tidal Wetland Act’ by New York State in 1973, in response to extensive destruction of tidal wetlands, provided a regulatory basis for protecting tidal wetlands and managing land use activities in coastal areas. Louisiana had also been preserving and restoring parts of its wetlands under the federal law (CWPPRA) since 1990. However, the approach to protect and restore tidal wetlands prior to hurricanes Katrina and Sandy was simply to avoid further loss of tidal wetlands rather than coastal protection.

It was only during Hurricane Katrina that the failure of the hurricane protection systems (mainly levees) diverted the attention to restoration of the lost wetlands which were once extended along the Louisiana’s shoreline. Consequently, the Act 8 of 2005 that was enacted after Hurricane Katrina integrated the wetland restoration projects into the Louisiana’s mainstream flood protection activities. Louisiana’s Coastal Master Plans (2007 and 2012) and the recent Resilient New Orleans (2015) have therefore included wetland restoration initiatives as part of their proposed coastal protection strategies.

Hurricane Sandy also proved the protective role of the restored saltmarsh ecosystems (with elevated edges) and vegetated sand dunes and reinforced the need for further activities to restore the lost wetlands. Restoration of tidal wetlands was included as a nature-based coastal protection measure in PlaNYC 2013 that was developed after Sandy. The recent OneNYC 2015 has also included an indicator of the ‘acres of coastal ecosystems restored’ to evaluate the progress of coastal protection measures towards achieving coastal protection goals.

The Indian Ocean Tsunami (2004), revealed the importance of greenbelts and mangrove forests in protecting coastal communities and infrastructure from destructive tsunami waves. The Mangroves for the Future (MFF) was launched in 2006 to strengthen coastal protection through restoration of mangrove forests primarily in countries that were affected by the 2004 tsunami. Development of this global initiative, reflects international consensus on building coastal resilience and highlights the significance of mangrove ecosystems in reducing damage from coastal hazards. Under the MFF programme, the governments of India and Indonesia developed their detailed national strategy and action plans (NSAPs) and identified priority areas and targeted projects for protection and restoration of mangrove forests. The MFF activities in India and Indonesia are aimed to contribute to enhancing coastal resilience through restoration and sustainable management of mangrove ecosystems.

Jakarta’s Integrated Coastal Protection Plan (2014) largely relies on engineering structures (i.e. a large-scale off-shore dyke) for protection of coastal areas and providing a basis for economic development. However, the plan provides for mangrove restoration to compensate for the loss of mangroves in Jakarta Bay caused by the closure of the bay (to construct the giant seawall) as well as providing educational, recreational and coastal protection values.

Since 2010, the Netherlands has been investigating the role of saltmarsh ecosystems for protection of the coastline within the Wadden Region. The research was started under the Wadden Region Delta Programme.

178 Coastal Wetlands Planning, Protection and Restoration Act, CWPPRA, 1990
in 2011 and included extensive literature reviews and modelling. Based on the results of the investigations, integration of saltmarshes into flood defence measures was defined as one of the preferred strategies for the Dutch Wadden Region and restoration and preservation of saltmarsh ecosystems was hence incorporated into the Wadden Region’s conservation plans in 2014.

However, despite the findings of the research projects, the approach in Wadden has been debated by a number of Wadden Sea experts. The main criticism is that the modelling has not been tested in real world situations and therefore the results do not necessarily represent the actual role of the saltmarshes for protection of coastlines. Another criticism is that there is a difference between preservation and restoration of saltmarsh ecosystems for the purpose of coastal protection and doing the same for the purpose of biodiversity conservation, and that this will require a trade-off between the two objectives.

Although, Rotterdam’s climate adaptation strategy (2013) does not specifically refer to the coastal protection capacity of tidal wetlands, it refers to the concepts of ‘building with nature’ and ‘green adaptation’ and provides for nature-based initiatives (such as creation of tidal parks and green banks) that improve adaptation to the effects and climate change as well as the ecological value of the city.

### 7.6.2. Non-structural coastal protection measures

#### 7.6.2.1. Land use and zoning regulations

The experiences of hurricanes Sandy and Katrina revealed that strategies that relied solely on a single measure of coastal protection (mostly engineering structures) have not only been ineffective but also have caused and aggravated the damage during the extreme events. Therefore, the coastal protection plans in New York City and New Orleans included a combination of measures that can be classified as nature-based (e.g. wetland restoration), structural (e.g. hard engineering structures) and non-structural (e.g. land use and zoning practices, insurance). Different strategies have been tailored to address specific requirements of different coastal areas and have been principally developed based on a vulnerability assessment.

The Louisiana’s Master Plan (2012) has strict land use regulations in areas where the coastal protection measures may encourage growth and development in those areas. This effect is referred to as ‘Induced Development’, which in this case might be due to protection of coastal lands against flood and other coastal hazards. The regulations in the Master Plan include requiring buffer zones near levees, restrictions on development in wetland areas, and the voluntary acquisition of lands. Under these regulations, coastal wetlands in high-risk areas need to remain intact and undeveloped. The Master Plan provides different non-structural protection strategies (i.e. elevation, flood proofing and voluntary acquisition) depending on the elevation or depth of flood in coastal areas. For example, elevating buildings is an option in areas where flood depth ranges between 3 and 18 feet; while flood proofing is considered as an appropriate measure in areas with a flood depth of 3 feet or less. Alternatively, voluntary acquisition is considered in areas where flood proofing or elevating buildings are not practical (given the high flood depths) and where buildings would need to be elevated higher than 18 feet (5.5m).

The ‘LA SAFE Programme’ in Louisiana is another initiative that complements the initiatives set out in Louisiana’s Master Plan. The programme has identified three distinct zones (i.e. Reshape, Retrofit, and Resettlement) across coastal Louisiana and developed land use strategies specific to each zone. The ‘Resilient Retrofit Programme’ in New Orleans is an example of a non-structural measure at the city scale that works by providing low-interest loans to people to insulate their houses.

A typical example of the non-structural measures in New York City is the ‘Resilient Neighbourhood Programme’ which is funded from the federal government budget. The main purpose of this initiative is to identify changes in land use and zoning regulations that are required to improve the adaptive capacity and resilience of the neighbourhoods against coastal hazards (e.g. storm surges) and other climate events. The voluntary ‘Buyout and Acquisition Programme’ in New York State is one of the initiatives at state level
that is developed after Hurricane Sandy to encourage property owners in high-risk coastal areas to retreat from these areas. The initiative provides the opportunity for property owners to voluntarily sell their properties to the state government who then converts the properties into wetlands or green spaces.

Non-structural strategies proposed in Tamil Nadu’s Action Plan on Climate Change (2014) include integration of building design alternatives into land use regulations to ensure that buildings and structures will withstand impacts of climate change, extreme rainfall and coastal hazards. The strategies also include restrictions on development in flood prone areas, protecting and resettling encroachments, preservation of the coastal vegetation, making regulations to require development of greenery and green belts.

The national zoning system in India, Under the Coastal Regulation Zone (CRZ) Notification, also provides for protection of natural ecosystems as buffers against coastal hazards and promotes restoration of mangroves as ‘bio-shield’ across coastal areas. After the 2004 Indian Ocean Tsunami, the CRZ Notification review committee proposed a hazard line or vulnerability line to be defined for coastal areas to regulate activities on the seaward side of that line. Consequently, CRZ Notification 2011 provided for mapping the hazard line along the coastline of the country. The mapping process was initiated in 2012 and is planned to be completed by 2017.

7.6.2.2. Insurance policies

Insurance is an incentive measure that spreads the risks associated with assets or activities across a wider community of beneficiaries. Insurance regulation is viewed as part of the non-structural measure for protection of coastal areas. The availability of insurance for coastal hazard areas and the extent to which insurance policies cover damage may facilitate or limit development in at-risk areas.

The insurance policies and regulations vary among the cases. The flood insurance system in New York and New Orleans operates under the National Flood Insurance Programme (NFIP) which is administered by the FEMA. Flood insurance is mandatory for properties in the high-risk zones (SFHAs) that have secured federal or federally-related loans or lines of credits. Location of properties and the difference between building elevation and the BFE provides a basis to determine the annual flood insurance premium. This encourages property owners to adopt and enforce management regulations to reduce flood damage. Property owners also have the option to insure their properties with private insurers to cover any extra losses not covered by the NFIP.

The NFIP annual premium was substantially increased after hurricanes Katrina and Sandy. The SFHAs in New York City and New Orleans as well as the BFE information are available to public through FEMA’s online Geoplatform179. Unlike the United States, there is no national flood insurance scheme in the Netherlands, India and Indonesia which requires flood insurance for properties in flood hazard areas.

In the Netherlands, under the Calamities and Compensation Act 1998 (WTS), the central government is required to provide compensations for losses caused by disasters including floods except for any damage caused by storm surges. This exception is on the ground that storm surge related costs maybe significant and difficult to estimate in advance. The relatively very limited size and extent of flood insurance schemes offered by private insurers in the Netherlands can be mainly associated with the liability that is vested with the government for compensation of flood damage. That is, the government is the key entity responsible for providing flood protection. The proposal from the Dutch Association of Insurers regarding compulsory flood insurance has not been successful due to a public view that considers the proposal as an unfair form of taxation.

In India and Indonesia, damage from floods (including coastal and surface floods) is generally covered through general property insurance policies. Although, flood insurance policies are not very popular in India and Indonesia, the recent floods and their associated damage to properties, have made property

owners to protect their properties against future natural disasters. The current insurance policies in India do not consider the properties’ potential flood risk in determining insurance premium. However, since 2015 and after the severe floods in India, the possibility of charging different premiums based on the level of the flood risk has been considered. The new insurance scheme in Indonesia (effective from March 2013) has also classified properties with respect to flood risks and has applied the highest flood insurance premium to properties located in high-risk zones.

### 7.6.3. Mitigation benefits of coastal wetlands

Except for Tamil Nadu’s Action Plan on Climate Change (2014) that explicitly refers to the carbon sequestration potential of mangrove ecosystems, the climate change mitigation policies in other cases have not specifically recognised the carbon sequestration function of coastal wetlands. Despite the recognition of the carbon sink capacity of mangroves in Tamil Nadu’s Action Plan, no estimate of the carbon storage capacity of mangrove forests is provided. The plan provides for a number of research programmes during the 12th & 13th five year plans (FYP) to identify the carbon sequestration potential of all forest types in Tamil Nadu. The MFF activities in Indonesia also aim to contribute to the country’s emission reduction targets by enhancing carbon sequestration through mangrove restoration.

Given the extensive area of mangrove forests in Indonesia and India and their degradation status, global initiatives such as ‘The Blue Carbon Initiative’\(^{180}\) and ‘The Livelihoods Carbon Fund’\(^{181}\) are supporting mangrove restoration programmes in these countries.\(^{182}\) The blue carbon projects in India and Indonesia are intended to provide multiple benefits including carbon sequestration, coastal protection, biodiversity conservation and improving local livelihoods.

Resilient New Orleans (2015) has included a reference to the mitigation benefits of coastal adaptation strategies such as wetland restoration, but it does not specifically refer to tidal wetlands. Implications of coastal protection and restoration activities proposed in Louisiana Coastal Master Plan (2007) on carbon sequestration have been evaluated within a period of 50 years (2010-2060). The information (soil bulk density and organic matter) used for this evaluation were obtained from the database developed by the Coastwide Reference Monitoring System (CRMS) as part of the initiatives under the CWPPRA (1990). As mentioned in Chapter 5, lack of sufficient empirical data for CS&S of coastal wetlands in Auckland undermines providing an accurate estimate of the blue carbon potential of these ecosystems. Whereas, the database in Louisiana provides site-specific carbon information that can be used to identify the blue carbon potential of coastal wetlands as well as their vulnerability to changes in sea level rise.

Development of the wetland carbon offset methodology by a New Orleans-based company and its approval by the American Carbon Registry in 2012 can be seen as a step forward towards incorporating blue carbon into the carbon market. Through this process, the net carbon offset credits generated by wetland restoration projects are used to offset carbon emissions. The methodology has been first used in a wetland restoration

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\(^{180}\) This program attempts to mitigate climate change through sustainable management and restoration of coastal wetlands ecosystems (including mangroves, tidal marshes and seagrasses). The initiative is coordinated by Conservation International (CI), the International Union for Conservation of Nature (IUCN), and the Intergovernmental Oceanographic Commission of the United Nations Educational, Scientific, and Cultural Organization (IOC-UNESCO) [Accessed 2016].

\(^{181}\) This fund is a type of carbon investment fund which aims to improve rural economy through large-scale agroforestry activities, restoration of degraded natural ecosystems (e.g. mangrove forests) and rural energy projects in developing countries in Africa, Asia and Latin America. It currently has 10 investors (Danone, Schneider Electric, Crédit Agricole S.A., Michelin, Hermès, SAP, CDC Climat, La Poste, Firmenich, Voyageurs du Monde) which receive carbon credits for their financial contribution. Since its establishment in 2011, the fund has supported plantation of 130 million trees which have resulted in carbon sequestration of around 10 million tons of CO\(_2\) [Accessed June 2016].

project (Luling Cypress Wetlands Pilot Project) at west New Orleans and is expected to provide annual net carbon sequestration between 1000-7000 t CO$_2$e for a 950-acre project.

Since mangrove planting was identified as one of the restoration techniques with greatest carbon offset potential, the first pilot project to plant mangroves in areas up to 40,000 acres across coastal Terrebone and Lafourche parishes has been started and is currently underway in Louisiana. The quantitative estimates of the carbon sequestration potential of coastal wetlands restoration in Louisiana was provided by Mack, et al., (2014) as a result of a two-year study. The study compiled and used empirical data (on biomass and soil carbon sequestration, methane and nitrous oxide emissions) from 47 peer-reviewed literature sources reported for coastal wetlands in the Mississippi River deltaic plain, as well as other areas of the world.

In addition to providing carbon credits, restoration of degraded coastal wetlands in Louisiana is intended to provide protection from erosion and hurricane surge by creating natural buffers.

Exceeding the restoration costs over the generated carbon revenue is one of the main challenges of blue carbon projects, particularly large-scale projects in Louisiana. It can be partly associated with the lack of connection between the co-benefits of restoration with the carbon market. That is, the carbon market particularly the regulatory (compliance) carbon market is designed to identify the most efficient (i.e. cost effective) way of mitigating emissions. However, the existing carbon market does not appreciate other potential benefits of carbon offsetting programmes, such as wetland restoration. As reported by Mack, et al., (2014) the multiple co-benefits of offset projects are points of interest for buyers in a voluntary carbon market and may increase the price of carbon credits. As mentioned in Chapter 5 (Section 5.2.1), the market value of carbon is different from the Social Cost of Carbon (SCC) which estimates the economic or social costs or damages caused by each additional tonne of CO$_2$ emission (Nordhaus, 2014).

A summary of the key points from all five cases and the case of Auckland is provided in Table 7.6-1.
**Table 7.6-1. Summary of the selected cases and the case of Auckland**

<table>
<thead>
<tr>
<th>The case studied</th>
<th>New York, USA</th>
<th>New Orleans, USA</th>
<th>Jakarta, Indonesia</th>
<th>Tamil Nadu, India</th>
<th>Rotterdam, The Netherlands</th>
<th>Auckland, New Zealand</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dominant coastal</td>
<td>Saltmarsh</td>
<td>Saltmarsh</td>
<td>Mangrove</td>
<td>Mangrove</td>
<td>Saltmarsh</td>
<td>Mangrove</td>
</tr>
</tbody>
</table>
| Climate response plans (addressing both Mitigation & adaptation) | National | - President Obama’s Climate Action Plan (2013) *  
- National Action Plan on GHG Emission Reduction (2011) *  
- PlanNYC (2007-2013) *  
- GreentLNO (2008) **  
- New Orleans 2030 (City’s Master Plan, 2010) *  
- Rotterdam Climate Change Adaptation Strategy (2013) ** | National | - Auckland Plan (2011) *  
- Auckland Unitary Plan (2012-2016) *  
- Auckland CDEM Group Plan (2016-2021) *  
- Low Carbon Auckland (2014) ** |
| Response to sea level rise and coastal hazards | National | - National Flood Insurance Programme (NFIP)  
- Flood Insurance Rate Maps (FIRMs) | National | - Integrated Coastal Zone Management (ICZM) Project | National | - Delta Programme | National | - MIE Adaptation guidelines |
| Local | - Voluntary Buyout and Acquisition Programme  
- Coastal protection plan (2013)  
- Resilient neighbourhoods initiative  
- Transferable Development Rights | - Coastal protection plan (2007)  
- Zoning regulation (LA SAFE strategy)  
- Resilience retrofit programme | Local | - Coastal protection plan | National | - ICZM plan (not released yet) | National | - Rotterdam Climate Proof Programme | National | - Land use and zoning rules |
| Initiatives for integrating climate change services of coastal natural features into climate change strategies/plans | National | - Incorporating coastal wetlands into the coastal protection strategies  
- Wetland Mitigation Banking  
- Monetary valuation of the ecosystem services of tidal saltwater wetlands | National | - Mangrove plantation and development for coastal protection  
- Blue Carbon Project (Research Project)  
- Mangroves For the Future (MFF) | National | - Protection and enhancement of mangroves for increasing carbon sinks and coastal protection  
- Mangroves For the Future (MFF) | National | - Incorporating coastal wetlands into the coastal protection strategies  
- Green-blue adaptation | National | - Protection and restoration of sand dunes for coastal protection  
- Research Project for examining the potential of marine ecosystems for carbon sequestration |

*Statutory documents (Mandated by law)

**Non-statutory documents (e.g. advisory)
7.7. Discussion: Similarities & differences

The cases reviewed in sections 7.2 to 7.5 have some similarities and dissimilarities with each other and with the case of Auckland in terms of their geographic, legislative and governance characteristics noting that the two cases of New York and New Orleans have similar models of the States’ governance. These differences and similarities are discussed in this section in order to gain an insight into the opportunities for mainstreaming protection and enhancement of coastal wetlands into the Auckland’s climate change and land use planning processes. It also informs a discussion on the mechanisms that can be applied to improve the region’s adaptive capacity and resilience to sea level rise and coastal hazards.

7.7.1. Governance structure and climate response plans

In almost all cases and in the case of Auckland, the central government is the dominant power. The forms, functions and powers of local governments vary among the jurisdictions. The various forms of local governments have generally some form of dependency on central government either financially or through the decision-making process or a combination of both. While the states and the local governments, including municipalities, in New Orleans and New York have a relatively high degree of legislative and executive independence, they are also reliant on federal government aid to be able to fund their projects.

The share of the total state general revenues from the federal aid varies among states. For instance, in the 2012 fiscal year, the state governments received on average about 32.8% of their total general revenue from the federal government, ranging from 45.8% in Mississippi to 20% in Alaska. These figures in Louisiana and New York were 44.3% and 37.7%, respectively (Walczak, 2016).

From a political perspective, except for the Netherlands which has a political system similar to New Zealand (Unitary Constitutional Monarchy), other cases have different forms of governance and political structure with different levels of centralization and democratic distribution of power. New Zealand and the Netherlands also differ from the other four cases in their constitutional form of governance, in the sense that both have a form of decentralised unitary state.

The United States and India with their state governments are examples of workable federal systems in large nations with very diverse populations. The United States has a complex government structure, which in many aspects is fundamentally different from the way the government is structured and works in New Zealand.

Unlike New Zealand where a national statute (Local Government Act 2002) defines the structure, powers and functions of all local governments, most of the local governments in the United States have the power and autonomy to enact their own local laws subject to certain conditions. Local Law in the United States system of local governments is a form of municipal legislation granted by the State Constitution. Depending on the law, it can even take precedence over a federal law; however, it varies city to city and state to state in the United States.

The government in the cities of New Orleans and New York has the form of a strong mayor-council, with the mayor serving as the chief executive and administrative head of the city, and the council as the policy making body. The mayor usually has the power to prepare the budget, appoint and remove certain officials, and to exercise broad veto powers over council actions. These features of the local government in New Orleans and New York City have some similarities with the structure of local council and the mayoral system in New Zealand. However, mayors of the territorial authorities (and chairs of regional councils) in New Zealand have limited powers of a largely executive nature.

While mayors in New Zealand have the ability to make major decisions, councils can vote down the proposals made by mayors. This can potentially undermine the capacity of the mayors in the New Zealand system to intervene in formulation and delivery of innovative legally binding policies especially in response to unexpected events. This suggests that in a similar situation, a mayor in New York City or New...
Orleans can make more agile decisions and quickly respond in fast pacing situations such as in response to a natural disaster. A typical example of this is the role that the mayor of New York (Michael Bloomberg) played in developing the package of statutory policies and instruments that currently exist in New York for adaptation, resilience and the use of natural features for coastal protection.

New York City also needs to work with regional planning councils. These regional planning councils are similar to the regional councils in New Zealand in terms of their functions but they do not have the power to regulate or tax, rather they are formed and funded by local, state and federal government and are vehicles for local governments to share their resources on issues of regional concerns.

Similar to New Zealand, Indonesia’s system of governance is unitary and power is concentrated in the central government. Five of 34 provinces in Indonesia including Aceh, Jakarta, Yogyakarta, Papua, and West Papua are granted with greater legislative power and autonomy than the other provinces. This is different from New Zealand where all regions are in a similar position under the LGA.

In the United States, there was no federal level plan or strategy for climate adaptation until 2014 when the first Climate Action Plan was developed by the EPA under an executive order by President Barack Obama (Executive Order 13653). However, states and municipalities have been active and developed their own mitigation and adaptation policies and initiatives. In the absence of the federal level guidance, active involvement of states and municipalities in developing climate change response plans represents an example of multi-level governance.

Similar to the cases in United States, there is not only no restriction for local governments in New Zealand to develop their own strategies and actions for coastal protection, but also they have mandates to do so under the RMA and NZCPS. This is in contrast with the climate mitigation that is mainly kept out of the jurisdiction of local government in New Zealand.

The agile responses to climate hazards in New York and Louisiana (post Sandy and Katrina) reflect on the role of the local leadership and the capacity of local governments to use their authority at hand to intervene and initiate a response with minimal interruption. The case of Louisiana also represents a collaborative model of response to climate change with the response being a result of a collaborative effort between municipalities, counties and the state of Louisiana. Although the two cases indicate that both collaborative and assertive models of intervention can independently deliver immediate responses, they also suggest that a responsive and adaptive model of climate governance can benefit from using the power of the local authorities supported by a horizontal model of collaborative governance.
7.7.2. Approaches to coastal protection

7.7.2.1. Coastal protection plans

Among the five cases, the New York’s post-Sandy ‘Comprehensive Coastal Protection Plan’ (PlaNYC 2013) and the Louisiana’s post-Katrina ‘Comprehensive Master Plan for a Sustainable Coast’ (2007 and 2012) have adopted a comprehensive approach to coastal protection at the city and regional scales, respectively.

In Auckland, provisions related to coastal protection and hazard management are included in various statutory policy documents including the Auckland Plan, the Auckland Unitary Plan and the Auckland’s CDEM Group Plan. Unlike New York and Louisiana, there is no standalone plan or a dedicated chapter in the existing plans for coastal protection. An integrated national-level coastal adaptation plan, similar to the Delta Programme in the Netherlands which identifies different flood protection strategies across the country, has not been developed for New Zealand.

This is largely attributable to the differences in governance and legislative framework between the two countries and is not necessarily a negative point for New Zealand. As mentioned in this chapter, coastal protection against floods in the Netherlands is a responsibility of central government. Whereas responsibility for coastal protection in general and adaptation to climate change in particular is vested with the local governments in New Zealand. This is because of the difference in geography as well as history and nature of flood events in the Netherlands. While, more than half of the Netherlands is below sea level and the country has been fighting inundation for centuries, New Zealand’s coasts have a relatively elevated topography and thus less vulnerable to inundation.

Under the New York City’s coastal protection plan, coastal areas are ranked based on their overall level of vulnerability and implementation of protection strategies is prioritised for the most vulnerable coastal areas. The protection strategies in New York and Louisiana are spatially explicit and are evaluated based on a series of factors such as costs, benefits, long-term effectiveness and implications on other protection measures and ecosystem services.

In Auckland, areas at risk of hazards such as tsunami, coastal inundation and earthquakes are separately mapped, but there is no map classifying coastal areas based on their overall level of vulnerability and demonstrating high risk areas similar to the coastal risk map prepared for New York City (Figure 7.2-2). Consequently, the Auckland region lacks site-specific coastal protection/adaptation strategies based on variations in vulnerability of different coastal areas. However, as mentioned in Chapter 6 (Section 6.4), national coastal sensitivity indices (CSIs) for coastal inundation and erosion were developed for New Zealand coastlines by National Institute of Water & Atmospheric (NIWA) Research Ltd (Goodhue, et al., 2012) and can be used for identifying coastal sensitivity at regional scale. Similar to the approach adopted in New York City to develop the city’s coastal risk map (Figure 7.2-2), CSIs in New Zealand are also based on a combination of geomorphic and oceanographic variables.

A number of socio-economic and physical parameters such as vulnerable communities, coastal assets and their protection cost were also used in developing the New York City’s coastal risk map. However, such variables are not included in the national CSI mapping for New Zealand, mainly because of the scale of the maps and lack of sufficient New Zealand-wide information (Goodhue, et al., 2012). The NIWA report recommends regional councils to use more detailed local data and tailor CSIs for their regions.

For example, councils are encouraged to incorporate data about land use/land cover types, hazard reducers (e.g. artificial beach nourishment, or beach control structures such as groynes and seawalls) and hazard exacerbators (e.g. the presence of stream mouths, vehicle and pedestrian access, or storm water outfalls) to provide more robust regional CSIs, focusing on areas with high sensitivity based on the national maps (Goodhue, et al., 2012).
The climate change and coastal hazard guidance manual (Ministry for the Environment, 2008b) has also identified a number of key criteria that local governments need to consider in assessing vulnerability.

Consistent with the non-structural coastal protection solutions in other jurisdictions, the available guidance manuals in New Zealand provide information about the possible risk reduction and risk transfer strategies including but not limited to the measures such as managed retreat, coastal setback zones, maintaining and enhancing natural buffers and integrating a wide range of coastal hazard risk rather than treating separate hazards independently. However, there is no legal mandate for local authorities to consider and apply the provisions provided in these guidance manuals.

The recent Auckland’s CDEM Group Plan (2016-2021) identifies the issues, activities and actions for each of the five themes of ‘reduction’, ‘readiness’, ‘response’, ‘recovery’ and ‘resilience’. It does not identify specific coastal protection and adaptation strategies, rather it refers to the development and implementation of the Natural Hazard Risk Management Action Plan (NHRMAP) which delivers best practice risk reduction and resilience outcomes. The plan also refers to a risk assessment approach to identify the risk of different natural hazards in Auckland. However, it does not clearly identify how the vulnerability of coastal areas should be assessed and how different risk management measures are identified and evaluated in terms of their long-term effectiveness and implications on other strategies and coastal processes.

Non-structural measures in Auckland can be generally classified as land use and building controls, public education, emergency preparedness, early warnings and insurance. As mentioned earlier, these measures are included in different policy documents and are not integrated in one plan.

Incorporation of tidal wetlands into the coastal protection strategies in New York and Louisiana is an example of ‘ecosystem-based adaptation’ that, as explained in Chapter 2, promotes the use of ecosystem services for coastal protection. In Auckland, nature-based protection measures are limited to dune restoration and beach nourishment and the role of coastal wetlands for protection against coastal inundation and sea level rise is not recognised. The Auckland’s CDEM Group Plan recognises restoration of wetlands and other types of restoration as an integral part of hazard management strategies; but it does not specify the type of wetlands and an implementation framework for wetland restoration.

The Auckland’s CDEM Group Plan emphasises the role of community engagement in developing and implementing successful disaster risk reduction strategies. This approach is similar to the ‘Resilient Neighbourhoods’ initiative in New York City that aims to identify neighbourhood-specific adaptation strategies through community participation and involvement.

7.7.2.2. Zoning regulation and incentive mechanisms

All cases except for Jakarta have developed zoning regulation and policies to reduce the risk of coastal hazards to coastal communities and infrastructure. As identified in the Louisiana's Strategic Adaptations for Future Environments (LA SAFE) coastal areas are classified into three zones of ‘re-settlement’, ‘retrofit’ and ‘reshaping’. This zoning system is consistent with the broad categories of adaptation strategies, i.e. ‘protect’, ‘accommodate’ and ‘retreat’, as proposed by the IPCC (IPCC, 2001; IPCC, 2000b). This zoning scheme does not exist in Auckland and its application, particularly retreat, may create significant public resistance and challenges as discussed in Chapter 6 (Section 6.4.2).

PlaNYC 2013 has considered the feasibility of the possible adaptation options based on the extension of the city into the coastal areas and has excluded the two options of ‘retreat’ and ‘abandon’ from its coastal adaptation strategies. This has resulted in the need for New York to focus on the ‘protection’ measures and support these with a series of incentive-based strategies, such as property insurance, to put a limitation on the further expansion of development into the coastal areas. However, due to the interests from coastal communities in Oakwood Beach (Staten Island), ‘managed retreat’ is currently taking place in this coastal neighbourhood of New York City.
The current zoning system in Auckland has some similarities with the flood hazard zoning system that is developed by the Federal Emergency Management Agency (FEMA) for the US coastal cities. Both systems identify areas at risk of inundation and demonstrate them through online maps that are accessible for public. However, unlike the FEMA’s flood hazard maps, the level of risk (e.g. high, medium, low) is not identified for Auckland’s coastal areas. The Auckland Council’s online flood maps also do not include information about the base flood elevation (BFE) for properties within coastal hazard areas the same way that is provided through the FEMA’s online maps. It is important that frequency and extent of coastal inundation is identified based on different scenarios of sea level rise; otherwise planning will be based on static risks and of little adaptation (anticipatory) value (Lawrence, et al., 2013).

The regulatory zoning system in India includes a specific category for ecologically sensitive ecosystems (including mangroves) and does not allow development in this zone. It is different with the zoning system in Auckland where no specific zone is identified for coastal wetlands and development is not restricted in areas where mangroves grow and exist. A similar concept in New Zealand is an esplanade reserve (a 20-metre Queen’s Chain strip) alongside waterways, (i.e. rivers, lakes, sea boundaries) which legally applies to subdivisions under 4 hectares. However, esplanade reserves are inland of mean high water springs (MHWS); whereas most coastal wetlands (all mangroves) are seaward of MHWS.

In New York City, under the state’s law, an area up to 150 feet inland from a tidal wetland boundary is identified as a transition area and any activity within this area requires a permit from the New York State Department of Environmental Conservation\textsuperscript{183}. In Auckland, there is not currently a buffer for coastal wetlands, but similar to New York’s regulation, removal of mangroves requires a resource consent from the Auckland council as the regulatory authority. The exception is removal of mangrove seedlings and small-scale mangrove removal (up to 200m$^2$) that are identified as permitted activities and do not require a resource consent.

\textbf{7.7.2.2.1. Insurance}

Compared to the national insurance scheme in New Zealand (EQC insurance), the NFIP in the United States plays a better role to incentivise people to take measures to retrofit their buildings. However, it does not necessarily limit development or investment in coastal areas.

In New Zealand, location and elevation of buildings do not influence the insurance premium under the EQC insurance scheme. It was also the case in India and Indonesia until recent floods and their extensive damage highlighted the need for linking risk zones with insurance premium. The EQC insurance applies the same rule to properties inside and outside coastal hazard areas and do not incentivise property owners to undertake risk mitigation measures. Whereas, under the NFIP, owners of properties with the floor elevation below the BFE are subject to the highest insurance premium unless they undertake measures to elevate their buildings.

In the Netherlands, flood damage is partly covered by the central government, but the damage from storm surges is not covered. Alternatively, people are informed about the risks they are exposed to and the measures they can take if the risk is high. This information is provided through the Neerlandse online underwriting tool. The strategy is providing people with transparent information, so that they can understand true levels of risk.

New Zealand has already started activities to inform people about the risk of natural hazards. Currently, information about natural hazards are included in the LIM reports that are prepared and available for all individual properties. Tsunami evacuation maps are also prepared and available for all New Zealand coastlines. The Auckland Council’s flooding maps are also good measures to enhance public awareness about coastal hazards and sea level rise. However, as noted above, they can be further improved to provide

more hazard specific information including level of risk, flood elevation and possible risk reduction measures.

### 7.7.2.3. Gaining adaptation benefits through restoration of coastal wetlands

Cases have adopted different approaches to incorporate coastal ecosystems into their coastal protection strategies with a common purpose of strengthening coastal protection using ecosystem services of coastal natural features. These approaches are represented using various concepts such as ‘Building with Nature’ and ‘Natural Climate Buffer’ in the Netherlands, ‘Multiple Line of Defence’ in Louisiana and ‘Living Shorelines’ in New York.

In most cases, improving coastal protection has been the main driver for restoration of coastal wetlands. New Orleans’ most recent development plan (Resilient New Orleans 2015) refers to and promotes the mitigation benefits of adaptation measures including wetland restoration. However, it does not explicitly refer to coastal wetlands and their carbon sequestration potential. This incorporation of mitigation co-benefits of adaptation measures seems to have been the result of the blue carbon research projects in Louisiana and New Orleans over the last several years. The competitive position of Louisiana with respect to availability of data and information can partly be attributed to how the federal law (i.e. CWPPRA 1990) has driven extensive studies to produce empirical information specifically to inform wetland restoration projects.

Unlike the cases reviewed, Auckland’s provisions on climate change and coastal hazards management neither provide for restoration of coastal wetlands nor require their protection to mitigate the risk of coastal inundation and sea level rise. As noted in Chapter 5 (Section 5.2.4.3) and Chapter 6 (Sections 6.3.4 and 6.5.1), lack of public appreciation of coastal wetlands particularly mangroves and their values in Auckland is one of the most notable barriers against protection and enhancement of these ecosystems for the purpose of coastal protection.

In contrast, the public in Louisiana and New Orleans for example strongly support restoration and protection of wetlands both coastal marshes and terrestrial forested wetlands. That support is mainly due to the concerns over continuing loss of Louisiana’s coastal wetland forests and the recognition of their coastal protection values after the Hurricane Katrina, whereas in Auckland, expansion of mangroves and lack of sufficient awareness about the values of coastal wetlands have been the source of misconception. The only recognised coastal natural features for coastal protection in Auckland are sand dunes that are protected and restored under the current provisions.

In fact, expansion of mangroves distinguishes New Zealand and Auckland from the other cases reviewed. While, the use of sediment is proposed as a technique to restore tidal wetlands in Louisiana and New York, sedimentation rate in the Auckland’s estuaries need to be reduced to prevent mangrove expansion. Surface elevation (the balance between sediment accretion, subsidence and erosion) is one of the parameters that is regularly monitored for the restored saltmarsh wetlands in Jamaica Bay (New York City). In Auckland, establishment of a monitoring system for measuring sedimentation rate (that should not exceed 2mm per year above the baseline rate), under the Hauraki Gulf Marine Spatial Plan (Sea Change), is an effort to address the cumulative effects of land use activities on coastal environment. However, parameters such as tidal range and surface elevation that affect the long-term sustainability of coastal wetlands particularly mangrove and saltmarsh ecosystems are not considered in the proposed monitoring programme.

Act No. 8 enacted in November 2005 after Hurricane Katrina, specifically provided for integration of coastal wetland restoration and hurricane protection activities in Louisiana to create a sustainable coast. The New Zealand Coastal Policy Statement 2010, as a statutory framework to manage activities in coastal areas, recognises the important role of natural coastal features in protecting coastlines against coastal hazards and the effects of climate change. But, as mentioned in Chapter 6 (Section 6.4), in response to public concerns over mangrove expansion, the NZCPS 2010 excludes mangroves from the list of natural coastal defence systems.
Without a fundamental shift in current thinking, there is little hope for a significant plan change over a foreseeable future unless a devastating climate hazard triggers a thorough review of the current views towards the coastal wetlands and in particular Auckland’s mangroves.

7.7.3. Mechanisms to support restoration of coastal wetlands

7.7.3.1. Introducing blue carbon to the carbon market

Among the cases, Louisiana stands out as a state that has provided innovative financing strategies for wetland restoration. It is also known as a place where there are opportunities for investment in innovative economic and environmental improvements. None of the jurisdictions in the cases studied has provided quantitative estimates of the CS&S capacity of coastal wetlands in their planning documents. However, based on empirical information from both local and global literature, Mack et al. (2014) reported the carbon offset credits as a result of coastal wetlands restoration in Louisiana.

A similar study has not been yet carried out for New Zealand or Auckland. The research summary in Chapter 5 (Section 5.6) provides the first estimate of the blue carbon potential of mangrove and saltmarsh ecosystems in Auckland. Similar to the approach taken by Mack, et al., (2014), this research also used the best available information from both local and global peer-reviewed literature to provide an approximate estimate of the CS&S of Auckland’s mangrove and saltmarsh habitats. However, given the limited local data, the estimates include some degree of uncertainty which needs to be accounted for.

As mentioned in Chapter 6, the carbon sink capacity of coastal wetlands is not currently considered in assessing land use policies that affect coastal wetlands. In contrast, the final coastal protection and restoration strategies proposed in the Louisiana’s Coastal Master Plan 2012 were selected based on their impacts on a range of ecosystem services including carbon sequestration. Similar to the non-statutory climate change response plan in India, the non-statutory Low Carbon Auckland Plan provides for a research programme to explore the carbon abatement potential of coastal and marine ecosystems. However, it sets out a long-term target that raises the question of what would happen to the lost mangrove stock until knowledge about the carbon sequestration capacity is developed.

Moreover, in all the jurisdictions studied, the legislative framework does not restrict the ability of local governments to actively contribute to climate change mitigation. Whereas, in New Zealand, contribution of the local governments to climate change mitigation is principally limited to policies that promotes the use of renewable energies. Effectively this means that the local governments in New Zealand are constrained to incorporate the emission implications of land use activities in resource consenting process, particularly in relation to coastal areas.

7.7.3.2. Wetland mitigation banking

The wetland mitigation banking in New York City is proposed as a mechanism to compensate for the adverse effects of development activities on wetlands. New York City has customised this approach to allow restoration of wetlands in pre-selected areas including areas at high risk of inundation due to storm surges and sea level rise (The Nature Conservancy, 2013; Freshkills Park Alliance, 2016). This is because of the shortage and high values of land in most of the city’s coastal areas that constrain implementation of onsite mitigation banking projects (The City of New York, 2013a).

Wetland mitigation banking is not currently implemented in New Zealand and there is also no legal requirement for it to be undertaken. However, the New Zealand government’s Guidance on Good Practice Biodiversity Offsetting in New Zealand (New Zealand Government, 2014), although non-statutory, provides an opportunity for wetland mitigation banking in New Zealand. Unlike the United states, the biodiversity offset in New Zealand is not legally binding under national legislation. As noted in Chapter 6 (Section 6.3.2), the RMA requires that adverse effects be avoided, remedied or mitigated, but the Act does not include a requirement for offsetting the adverse effects or achieving a no net loss outcome.
The biodiversity offsetting guidance does not explicitly refer to the mitigation banking as an offsetting instrument, but it provides for achieving biodiversity gains prior to development impacts. It is proposed as a good practice to address ‘the time lag between when biodiversity is lost and when biodiversity gains are fully delivered’ (New Zealand Government, 2014, p. 30). It implicitly implies ‘mitigation banking’ where offset occurs prior to development impacts and the generated offset credits will be sold to developers.

The biodiversity offset projects in New Zealand are currently implemented under the ‘one-off’ or ‘permittee-responsible mitigation’ approach which requires a developer to carry out the offset project at or within the closest proximity to the impact site (i.e. on-site offset) or within the same catchment or ecological district (i.e. off-site offset) (New Zealand Government, 2014). This approach provides flexibility for identifying offset sites where on-site offsetting is limited. The concept of freshwater management unit is also an opportunity to set geographical boundaries for the offset sites.

Biodiversity offsetting presents a conceptually attractive approach to conservation. However, there are some limitations arising from the design and implementation of offset schemes (Bull, et al., 2013) which needs to be considered and addressed for achieving a successful offset. Otherwise offset could potentially result in adverse rather than positive outcomes (Brown & Penelope, 2016). These limitations and the key proposed approaches to address them are briefly outlined in Table 7.7-1.

<table>
<thead>
<tr>
<th>Limitation</th>
<th>Proposed approaches to address the limitation</th>
</tr>
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<tbody>
<tr>
<td>Difficulty in achieving equivalence between ecological losses and gains</td>
<td>• Using practical biodiversity surrogates and proxies to assess the baseline condition</td>
</tr>
<tr>
<td></td>
<td>• Using reliable assumptions to predict and measure conservation benefits relative to reference scenarios</td>
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<tr>
<td></td>
<td>• Avoid ‘out-of-kind’ offsets unless trading up from losses with little or no conservation value</td>
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<td></td>
<td>• Compare biodiversity losses in the impact site and biodiversity gains in the offset site in three dimensions: biodiversity type, location/space and time</td>
</tr>
<tr>
<td>Limits to what can be offset on a ‘like-for-like’ (or in-kind) basis</td>
<td>• Assessing the risk of ‘non-offsetable’ residual biodiversity losses through a risk assessment</td>
</tr>
<tr>
<td></td>
<td>• Identifying measures to minimise the risk considering the level of the risk and uncertainty in delivering offset outcomes</td>
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<tr>
<td></td>
<td>• Establishing thresholds for what can and cannot be offset</td>
</tr>
<tr>
<td>Uncertainty in achieving ‘no net loss’ and ecological gains</td>
<td>• Assessment of the biodiversity condition in the offset area prior to the offset project occurring or in ‘without offset’ scenario</td>
</tr>
<tr>
<td></td>
<td>• Providing an explicit definition for no net loss and identifying no net loss against a dynamic baseline that considers trends in biodiversity condition</td>
</tr>
<tr>
<td></td>
<td>• Designing offset actions in a way that deliver ‘significant’ biodiversity gains not only no net loss targets.</td>
</tr>
<tr>
<td></td>
<td>• Assessing and measuring net gain at the scale of a landscape beyond the project site</td>
</tr>
<tr>
<td></td>
<td>• Achieving biodiversity gains prior to the development project occurring, where possible</td>
</tr>
</tbody>
</table>

As the table indicates, uncertainty in achieving ecological gain versus certainty in occurring ecological loss due to development activities, is one of the common criticisms of biodiversity offsets (Bull, et al., 2013). Achieving biodiversity gains prior to development impacts is proposed as a good practice to address this issue (Bull, et al., 2013; New Zealand Government, 2014). Bio-banking (e.g. mitigation banking) can significantly minimise the risk that biodiversity objectives are not met, as offset has established prior to development impacts (OECD, 2016). However, bio-banking is not an appropriate offsetting mechanism where there is a low demand for offsets (OECD, 2016). Establishment of pre-selected offset sites also offers a number of other advantages compared to the permittee-responsible mitigation:
• It may involve adopting a landscape-wide approach in identifying the size and location of the offset sites in a way that contributes to achieving national or regional outcomes (BBOP, 2012e; OECD, 2016).
• It may involve extensive planning and the use of scientific expertise which may not be always available to many permittee-responsible offsetting proposals.\(^{184}\)
• It can facilitate the resource consenting process for development activities as the offset site has been already approved.\(^{185}\)

No net loss and preferably a net gain goals of biodiversity offsets require the adverse effects to be either balanced (for no net loss) or outweighed (for net gain) by biodiversity gains through offset activities (Aiama, et al., 2015). The term ‘net’ in ‘no net loss’ and ‘net gain’ implies some imperfections in achieving a balance between the inevitable biodiversity loss at the development site and the biodiversity gain off-site, which requires careful consideration of time lag between the loss and gain of biodiversity, and the extent and type of the lost and gained biodiversity (Aiama, et al., 2015). Such an imperfection is mainly associated with the inherent limitations of available information on ecosystems and their dynamics. It is therefore argued that designing offset activities to deliver no net loss together with the lack of appropriate offsetting frameworks often results in a net decline/reduction in biodiversity (Birkeland & Knight-Lenihan, 2016). Thus, it is recommended that offset projects take a precautionary approach and overcompensate for residual impacts to ensure the no net loss targets are achieved (Aiama, et al., 2015; Birkeland & Knight-Lenihan, 2016).

Despite these inherent difficulties, biodiversity offsetting addresses the ‘residual impacts’ that were previously left unmeasured and uncompensated. Therefore, it bridges a gap that were previously completely ignored, even if it is doing that imperfectly (von Hase & ten Kate, 2017). Thus, it is generally acknowledged that biodiversity offsetting provides the potential for significant improvement in biodiversity and ecological conditions, if it is correctly framed and implemented (von Hase & ten Kate, 2017). The current and growing scientific information and knowledge can provide a basis for improving policy frameworks and decision making relating to offsets.

7.8. Conclusion

The experiences of post hazard adaptation and resilience planning in other jurisdictions particularly the cases of Louisiana and New York suggest that effective coastal resilience policies necessitate a risk based approach based on a multi-facet analysis that applies a set of variables to determine vulnerability of coastal areas to coastal hazards including the effects of climate change. The guidance manuals prepared for local governments in New Zealand also promote a similar approach.

A missing element in Auckland is a strategic coastal protection and resilience plan. Such a plan will need to identify the most vulnerable (high risk) areas and provide spatially explicit adaptation/protection strategies based on the degree of vulnerability and other site-specific characteristics of Auckland’s coastal areas. The proposed resilience approach for coastal areas is principally based on the concept of multifunctionality of the natural coastal features, with particular attention being paid to the adaptation benefits of coastal wetlands and their mitigation co-benefits. This approach also ensures that the vulnerability of coastal wetlands to the future effects of climate change such as sea level rise is identified and well managed.

As discussed in previous chapters and based on the review of policies from other jurisdictions, it is arguable that the evidence of positive benefits is sufficient to promote land use activities that protect and enhance coastal wetlands particularly mangrove and saltmarsh ecosystems. Offsetting as a conservation tool, can


also contribute to climate change mitigation and adaptation, if its framework is expanded to provide for a wider range of ecosystem services such as carbon sequestration and coastal protection. The wetland mitigation banking approach used in New York City is an example that provides for achieving multiple benefits (including climate change adaptation) from restoration of tidal wetlands. Pre-selection of offset sites with a holistic approach (e.g. a catchment- or region-wide approach) and based on regional (or national) conservation and coastal protection priorities can also ensure that offset projects will contribute to achieving the strategic goals and outcomes.

Considering the findings of the extensive literature review, global case studies and the information provided by the available guidance manuals, possible planning strategies or options for Auckland under an integrated coastal protection and resilience planning approach are discussed in the following chapter.
8. Options for Auckland

8.1. Introduction

This chapter draws on findings of the previous chapters including the theories and principles discussed in Chapter 2, and seeks to address the last question of this research:

- Depending on the outcome of the previous research questions (i.e. questions A to D), investigate what policy options and instrument mixes can be applied to Auckland for mainstreaming climate change benefits of coastal wetlands into land use and resource management decision-making processes.

The chapter starts with a discussion on the need for an integrated coastal protection and resilience planning approach and discusses the key requirements and potential challenges.

8.2. The need for an integrated coastal protection and resilience plan

The theories and practices that are discussed in previous chapters suggest that tackling the conflicts between urban development, natural resource management, and response to climate change in coastal areas requires an integrated, adaptive and ecologically positive approach to land use and resource management. Findings of the previous chapters also highlighted the need for an ecosystem-based approach that principally builds upon the theories of resilience, accounts for the multiple benefits of coastal natural resources (including wetlands) and focuses on vulnerabilities of coastal environments to natural hazards (including the foreseeable impacts of climate change and the impacts of upstream land use activities on the receiving coastal environments).

Depending on the context, bringing these elements together may also involve some strategic changes to legislative and operational frameworks that govern implementation of mitigation and adaptation policies. These changes may be needed primarily to reduce institutional sources of divergence between these two key means of response to climate change. In the New Zealand context, the RMA’s stance on the role of central and local governments regarding adaptation and mitigation is an example of such divergence at national level which is discussed in detail in Chapter 6.

The global case studies showed that in most jurisdictions such an integrated approach had not been adopted until a catastrophic event like a hurricane or flood triggered a rethinking about the benefits of ecosystems that would have otherwise been given lower priorities. Other indirect and non-use benefits of coastal ecosystems such as carbon sequestration are either understated or in principle are too intangible to be incorporated into the mainstream policy agendas. The global examples of post-disaster reconstitution of policies and the consequent innovative mechanisms that have been developed to enhance resilience of the coastal cities to natural disasters provide valuable insights into the policy and planning processes in other coastal cities with similar vulnerabilities, including Auckland.

A key observation from the case studies was that in most instances, the policies have been shifting towards nature-based solutions with a stronger focus on multiple benefits of coastal natural ecosystems. Whilst urban development policies predominantly show a tendency to trade-off between social, environmental and economic goals based on cost-benefit analysis, there has also been an emerging interest in approaches and mechanisms that consciously seek win-win outcomes. As discussed in previous chapters, the win-win solutions largely focus on synergies between two or more policies or strategies and multiple benefits that can be derived from a single policy or measure.

The market based and regulatory mechanisms that were discussed in Chapter 7 basically provide for win-win outcomes and have the potential of generating multiple benefits. The biodiversity offsetting, for example, is a positive approach that seeks to ensure net ecological gain or at its minimum no net loss. The flood insurance policies that encourage retreat from highly vulnerable coastal areas are primarily incentive-
based hazard management measures to reduce vulnerabilities, which can also release coastal lands for ecological restoration or mitigation banking.

These mechanisms and other tools such as vulnerability mapping, knowledge sharing and public engagement provide a framework that can be tailored to integrate policies and achieve win-win outcomes in Auckland taking into account the opportunities and challenges that are identified and discussed in preceding chapters.

8.2.1. Possible pathway for Auckland

The proposed pathway that is further discussed in the following sections is a framework that is basically built upon the concepts of co-benefits, resilience, Climate Compatible Development (CCD) and Integrated Coastal Zone Management (ICZM). The policy options and recommendations proposed in this chapter are also informed by the experience of the global cases as well as the findings of New Zealand’s studies on coastal adaptation (See Section 6.4.1).

The framework suggests an integrated coastal protection and resilience planning for Auckland. This planning approach focuses on the resilience of Auckland’s coastal areas as a core objective of land use and urban planning and applies mechanisms that promote the use of coastal wetlands to enhance resilience of the coastal environment. Therefore, focusing on climate change adaptation and resilience as the primary policy objectives and using mechanisms such as offset that incorporates climate change values of the coastal ecosystems could assist in achieving net ecological and mitigation co-benefits.

The integrated approach means a strategic view to management of coastal hazards and coastal resources that recognises vulnerabilities of coastal areas and adequately accounts for the multiple benefits of coastal ecosystems. This approach would help to shift the focus from a re-active coastal hazard management (managing coastal hazards including the effects of climate change as a private property protection issue) which is currently the case in Auckland (Hart, 2011) to a pro-active coastal hazard management and adaptation (i.e. managing coastal hazards including the effects of climate change as a coastal system and sustainable-management issue).

With this approach, coastal resources would need to be planned for and managed within a ‘coastal process unit’ which could extend the coastal boundaries to include catchments and processes that influence the receiving coastal environments. The National Policy Statement for Freshwater Management 2014 (NPSFM) which requires establishment of the Freshwater Management Units (FMUs) (Chapter 6, Section 6.3.6), provides a useful framework for achieving the integrated management of freshwater and marine resources, mainly through managing the cumulative effects.

The principles and requirements for an integrated coastal protection plan are further discussed within the following sections.

8.2.2. Principles

To ensure that the proposed integrated coastal protection approach is consistent with the provisions set out in the NZCPS and other strategic policy documents in New Zealand, the approach will need to be based on a number of key principles as outlined below. These principles are built upon the findings from the preceding chapters and are informed by the main concepts and theories as discussed in Chapter 2. The key principles include:

- Effective adaptation strategies need to be site-specific and take into account the strategic relation of the sites, and the long-term benefits and costs of their implementation.
Consideration of uncertainty\(^{186}\) and the dynamic nature of climate risks is fundamental in developing coastal protection strategies to make sure they can deliver long-term adaptation outcomes (this is consistent with the findings of local studies as provided in Section 6.4.1)

Where an adaptation strategy identifies protection as an appropriate long-term strategy for a site, coastal protection options need to:

- Consider a combination of measures including nature-based (natural), non-structural and structural solutions primarily to minimize risks
- Recognise the multiple benefits of coastal ecosystems and incorporate nature-based measures into the policy agenda (as principled priority)

Particular attention needs to be paid to ecological connectivity between fragmented coastal natural elements when developing policies and rules for land use control.

Land use and development activities which affect coastal wetlands need to be weighed against the costs associated with loss of ecological values including climate change values of coastal wetlands.

An integrated governance model (consistent with the concepts of ICZM and CCD as discussed in Section 2.8) involves a full range of coastal stakeholders in the decision-making process including the communities, corporations and territorial authorities.

Open communication and information sharing is a key enabler of urban resilience and adaptive planning in response to unanticipated external stressors. Data and knowledge representation and mapping tools that are designed and tailored specifically to exchange local information, can:

- Help coastal communities understand their vulnerability to coastal hazards, the value of nature-based solutions and how they can help to reduce climate risks
- Improve common understanding of the planning and policy issues, possible options and the benefits and costs of actions or inactions
- Enhance transparency about outcomes, policy interventions and their effectiveness
- Help reduce potential resistance through policy processes; and
- Inform adaptation, restoration and conservation decisions in coastal areas

The following sections discuss the key requirements and challenges of the proposed strategic coastal protection approach.

8.3. **Area-specific strategies for coastal protection**

Protection strategies for coastal areas in the Auckland region need to be tailored based on the specific site characteristics and the nature and severity of the hazards and risks. This highlights the importance of coastal vulnerability maps that are discussed in Section 8.4. Technical, environmental, social and economic restrictions should also be considered in identifying the most appropriate measures for different coastal areas across the region.

The strategic approach to identify protection/adaptation strategies needs to take into account the advantages (benefits) and disadvantages (costs/damage) of different options over a long-term period. It may result in developing long-term management approaches for some areas that are different from present day practices. For example, in the short-term, protection of assets in a number of coastal areas may be tenable in terms of their cost and benefit given the high value of assets and the reasonably low cost of protection. However, over a longer term and given the future climate impacts, the protection measures may not be effective and people may have no choice but to leave their homes. In the latter case, there will be a need for guidance for decision makers to help them decide on an appropriate path towards a potentially different situation in the future.

\(^{186}\) Available approaches and methodologies to manage and deal with uncertainty are provided in Section 2.3.
8.3.1. Compatibility with catchment activities and other protection measures

An integrated approach to formulation and development of coastal protection strategies needs to ensure that activities (e.g. landward diversions, storm water management structures, sediment control measures) within the entire catchment are properly integrated into those strategies. For example, catchment sediment control activities may impede the flow and exchange of water and sediments to downstream coastal wetlands, and result in the reduction in the area or functioning of the impacted wetlands. Similarly, offshore activities granted resource consents may also influence intertidal wetland extent and health through changes in onshore or along-shore currents or resuspension and deposition of marine sediments disturbed by dredging, dumping, artificial reef construction etc. Coastal protection strategies need to identify and influence these types of activities to ensure that the catchment-based and offshore activities will not adversely impact the opportunities for coastal protection. The NZCPS uses a similar approach where it requires that any adverse effect of a hard protection structure on the coastal environment must be minimised (NZCPS, Policy 27(3)). However, this approach proposes assessments throughout adjacent catchment(s), which is more thorough but also more challenging.

Coastal protection strategies also need to be compatible and complement each other. For instance, as indicated in the Louisiana Comprehensive Coastal Master Plan (2007) (Coastal Protection and Restoration Authority, 2007), levees as hard protection measures, can impede the flow of water and sediments into swamps and marshes, and this can result in the loss of the wetlands and eventually lead to inundation of coastal lands. Therefore, evaluation of hard structures (e.g. levees, seawalls) should consider their adverse impacts on coastal processes and sediment supply and the consequent effects on land or wetlands.

8.4. Zoning based on vulnerability

As mentioned in Chapter 7 (Section 7.8), effective coastal resilience policies necessitate a risk based approach based on a multi-faceted analysis that applies a set of variables to determine vulnerability of coastal areas to coastal hazards including the effects of climate change. Applying a risk-based approach to Auckland means that coastal protection strategies in Auckland will need to be based on a comprehensive assessment of the vulnerabilities of Auckland’s coastal environment to natural hazards.

In an ideal situation where the common barriers including time, resources, knowledge and political resistance are of less concern, an assessment method should include every possible factor that can help to gain a comprehensive view of the vulnerabilities in the coastal areas. As discussed in Chapters 6 (Section 6.4) and 7 (Section 7.7.2.1), the national coastal sensitivity indices (CSIs) for inundation and coastal erosion can be used to develop regional CSIs for the Auckland region. As recommended by Goodhue, et al., (2012) and Ministry for the Environment (2008b) and drawing on the coastal risk mapping in New York City (Chapter 7, Section 7.2.2.3), the following socio-economic and physical parameters can be also used to develop more accurate regional CSIs:

- Density and concentration of population particularly vulnerable populations (e.g. elderly or those with disabilities that are potentially more vulnerable to storm events)
- Values of assets at risk of coastal hazards (e.g. GDP per capita and GDP per unit land)
- The expected cost for protection of coastal assets and infrastructure
- The expected cost for disaster response and recovery
- The presence of hazards reducers and exacerbators (See Section 7.7.2.1 for more information)
- The rate of subsidence, erosion and accretion (mm/yr)

Combining information from the two separate CSI maps (coastal inundation and erosion) can result in a coastal risk map for the Auckland region similar to the map developed for New York City (Figure 7.2-2).

187 Apart from sediment supply by freshwater resources (e.g. streams, rivers), resuspension of fine sediments during storm events and king tides and their accumulation during calm weather will serve as an additional source of sediment for coastal ecosystems.
Such a map can be used to prioritise investments on coastal protection. This type of mapping can also be valuable in identifying site-specific coastal adaptation strategies tailored to different coastal areas.

The Auckland Council’s current coastal inundation maps can be used to rank coastal areas until such time as a complete coastal vulnerability assessment provides for more robust zoning of the coastal areas. However, as suggested by Lawrence, et al., (2013), to reflect the uncertainties, different climate change emission reduction scenarios can be used to estimate the changes in frequency of coastal inundation and rates of sea level rise. Otherwise, reliance on a single emission reduction scenario and static rate of sea level rise creates path dependency (as discussed in Section 2.4) and results in inflexible coastal adaptation strategies which would likely fail to address changing risks over time.

One approach to identify and rank inundation zones could be estimating the area (or extent) and frequency of coastal inundation under different emission reduction scenarios combined with changes in sea level rise. The area which will be inundated under all scenarios can be classified as an area with “very high” risk of coastal inundation. Accordingly, areas that will be less frequently inundated can be classified as areas with “high” and “moderate” risks of coastal inundation.

In addition, there needs to be a mechanism to allow the review of the boundary of coastal hazard zones to be regularly updated according to sea level rise projections.

8.4.1. Protect and retreat zones in Auckland

As a guidance, which can be further improved, areas with ‘very high’ and ‘high’ risk of coastal inundation with high flood frequency where it is not feasible to raise elevation of the buildings (high flood elevation) can be identified as ‘retreat zone’. Retreat zones can also include areas where:

(i) Coastal protection (using hard engineered structures) cannot provide effective protection particularly over a long-term period, or

(ii) Coastal protection (using hard engineered structures) is not economically feasible (particularly over a long run) and wouldn’t outweigh the costs (it is particularly the case in areas with at-risk assets of low value and density), or

(iii) No feasible technical solution would be available.

Areas outside the retreat zone where frequent inundation is not expected and increasing buildings floor height is feasible, can be identified as ‘protect zone’. The protect zone can also include areas where coastal protection is cost-effective and feasible (e.g. in areas with at-risk assets of high values).

This zoning approach is mainly based on the IPCC classification of adaptation responses (i.e. protect, accommodate and retreat) (IPCC, 2001; IPCC, 2000b) (See Section 5.3.2) and a similar approach adopted by the Louisiana's Strategic Adaptations for Future Environments (LA SAFE) (Chapter 7, Section 7.2.3.3).

The possibility of ‘No Action’ (abandon) can also be examined where there is no technically feasible solution or if the expected benefits from coastal protection would not justify the costs.

Once the retreat and protect zones are established, land use and zoning regulations can be applied with the levels of restrictions determined based on the magnitude of the inundation/flood risk. Strategies within the protect zone can include those that aim to protect people and infrastructure through a combination of structural, non-structural and nature-based protection measures. Examples include construction of dykes, levees and groynes as structural protection measures; warning system, emergency plans, elevating/retrofitting buildings and insurance policies as non-structural measures; and wetland creation or restoration as nature-based adaptation approaches (Nicholls, 2011). Except for wetland creation or restoration, these strategies are currently undertaken in the Auckland region as part of the coastal protection programmes.

Adaptation/protection strategies within the retreat zone need to be developed in a way that minimises potential risks by pulling the properties back from the coast or by landward relocation of buildings and
infrastructure. These strategies mainly focus on non-structural and natural measures rather than hard protection (Nicholls, 2011; Ministry for the Environment, 2008b).

Possible non-structural strategies based on (i) the findings of an extensive review of scientific literature in Chapter 5 and discussions in Chapters 6 and 7, (ii) the findings of the case studies review particularly the cases of New York and Louisiana, (iii) the information from the MfE guidance manuals (Ministry for the Environment, 2008a&2008b), and (ii) the information from studies addressing Auckland’s vulnerability to coastal inundation and sea level rise (e.g. studies by Turbott, 2006 and Hart, 2011), include:

- Modifying private property rights and setting regional rules that support establishment of finite terms for the use of land within coastal hazard risk areas and changing the permanent utility of titles to reflect the impermanence of the land (e.g. changes in house titles from fee simple to leasehold land with a defined term of tenure). This strategy aims to address the issues around the existing use rights discussed in Chapter 6 (Section 6.4.2).

- Avoiding establishment of new structural coastal protection measures and limiting maintenance or upgrading of existing structural protection measures in the retreat zones

- Developing a specific flood insurance scheme at the national scale that considers the elevation of properties and their overall exposure to coastal hazards in estimating insurance premiums. For example, similar to the National Flood Insurance Program (NFIP) in the United States, a house with ground elevation 1m below the Base Flood Elevation (BFE) will pay a higher premium than a house with ground elevation equal to BFE and a house with ground elevation 1m higher than BFE. This option is proposed based on the discussion on flood insurance programmes in different case studies including New Zealand (Chapter 7, Section 7.7.2.2.1).

- Examining the possibility of ‘No Insurance’ for new buildings within the retreat zone and ‘high insurance premium’ for new buildings in the protect zone as well as existing properties within both retreat and protect zones if the building elevation is below BFE: The insurance policy options, however, will not necessarily prevent development or investment in coastal areas if it is not accompanied by land use and zoning regulation that restrict new development. It is mainly because some property owners especially in high value areas may still be willing and able to invest in coastal properties even in the absence of insurance cover. These options may also have disproportionate effects on low- and middle-income homeowners with mortgages, while buyers with no need to mortgage (normally high-income buyer) are not impacted.

**8.4.2. Potential challenges**

The legal and practical feasibility of the above land use policy options as well as their social and economic implications need further investigation which is beyond the scope of this research. A potential consequence of risk based zoning of coastal areas would be on values of the properties especially in areas classified as high or very high risk. While a reduction in property values as a result of the zoning regulations can discourage further development in the high or very high risk areas and also reduce liability of the Auckland Council to likely damage to properties as a result of the risks defined for the zones, it will also incur some costs to compensate the existing property owners in these areas. It will also result in strong public resistance against leaving their houses (e.g. strong public resistance to managed retreat from property owners at Mission Bay/Kohimarama, Hart, 2011) and over the effects of the zoning policies on properties’ value.

As mentioned in Chapter 6 (Section 6.4.1), inclusion of coastal erosion risks to LIM reports for the properties located within ‘erosion hazard zones’ in Kapiti Coast caused significant public concerns and legal challenges and eventually resulted in removing this information from LIM reports and withdrawal of development restriction from the Kapiti Coast Proposed District Plan 2012.
Another potential issue is that considering some areas for protection while assigning some others as retreat may be perceived as inequitable, especially if the areas set aside for protection have high social or economic status compared to retreat areas. These types of issues can be mitigated if people are well informed of the long-term possibility of sea level rise and coastal hazards. As discussed in Chapter 6 (Section 6.4.2), issues around property rights are one of the main barriers to the implementation of retreat programmes. Compensation of private property owners under a managed retreat strategy entails a complex set of political, legal and economic and financial issues that is beyond the scope of this research. However, some of the potential implications of this method are discussed below.

Compensation potentially involves purchase of at risk properties by local authorities (in this case the Auckland Council). An obvious issue is that the potentially high prices of the properties in coastal areas makes the purchase of those properties unaffordable for councils. Some councils use the Public Works Act (1981) as the guide to voluntary purchase at the coast and in flood plains e.g. Hutt City reach of the Hutt river flood risk management plan (Greater Wellington Regional Council, 2016).

While local authorities generally avoid disturbing local property markets, any effort to compensate property owners has the potential to alter the market. For example, purchasing of properties by a local authority may create a perception in the community that properties within coastal hazard zones will eventually be purchased by councils. This may consequentially encourage investment in coastal hazard areas with the expectation of future compensation possibly with a higher price due to the relatively higher rate of property price increase in coastal areas. This will require a thorough consideration by the local authorities of the effects of the compensation measures on public perception within a broader context of the problems that may arise as a result of that.

In contrast, a relatively feasible option could be based on incentive-based mechanisms (e.g. insurance) and participatory land use planning approaches without immediate purchase of the at-risk properties. Examples from the case studies include voluntary ‘Buyout and Acquisition Programme’ in New York State; the ‘Resilient Neighbourhood Programme’ and ‘Transferable Development Rights’ in New York City and the ‘Resilient Retrofit Programme’ in New Orleans.

The zoning regulations of the types listed earlier can incrementally alter the public perception of the lands and properties within hazard risk zones over a long-term period, and therefore provides for a natural adjustment of the values of the at-risk properties by market forces rather than an induced intervention. This also provides local authorities with an opportunity to adjust the regulations and makes the changes manageable for councils. This option should take into account the long-term nature of the process and uncertainties relating to the occurrence of hazards.

Even if the councils evaluate the compensation as an affordable option, there will still be an issue with its funding and consequentially the wider community pushback. Councils will then need to clearly identify the sources and mechanisms of funding the compensations.

### 8.5. Ecosystem-based adaptation

#### 8.5.1. Principle

As mentioned in Chapter 2, ecosystem-based adaptation (EbA) aims to integrate biodiversity and ecosystem services into an overall adaptation strategy to help communities adapt to the adverse effects of climate change (Convention on Biological Diversity, 2009; OECD, 2014a). It is an example of strategies under the broad category of ‘biodiversity in climate change funding’ that leverages biodiversity co-benefits to contribute to climate change mitigation and adaptation (OECD, 2014a). As noted earlier in this chapter, a strategic coastal protection plan for the Auckland region (as proposed) needs to adopt a robust ecosystem approach to increase coastal resiliency and reduce vulnerability to climate change impacts. According to the findings of the extensive literature review (Chapter 5) and the case studies (Chapter 7), the rationale
for incorporating coastal wetlands as natural features into the coastal protection strategies can be summarised as the followings:

a. Coastal wetlands have the potential to attenuate wave energy and mitigate the extent of inundation as a result of storm events.

b. Coastal wetlands have a built-in capacity to ‘evolve’ and ‘adapt’ to natural changes including storms and sea level rise over time. They can do this through rebuilding or migrating landwards, and maintaining their form through sediment accretion (provided sources of sediment are not disrupted).

c. Providing climate benefits by coastal wetlands depends on sustainability and persistence of these ecosystems in response to changing environments. As discussed in the previous chapters (Sections 2.3; 5.3.2; 5.7; 6.6.3), sustainability of coastal wetlands is influenced by a range of ecological, anthropogenic and climate change factors. This creates uncertainty on the long-term persistence of this ecosystems.

d. Subject to sediment availability and undisturbed hydrological conditions, mangrove ecosystems in New Zealand including Auckland’s mangrove forests can keep pace with sea level rise and will mainly remain unsubmerged until 2100 (Loveland, et al., 2015). This highlights the importance of mangrove management to make the best use of their coastal protection services.

e. Integrating coastal wetlands with structural protection measures (e.g. seawalls, levees) can reduce the cost of structures’ maintenance (e.g. the results of modelling in the Netherlands, Wadden Sea; the experience in Louisiana). It is because they reduce the day-to-day exposure of structures and therefore increase their life and reduce maintenance commitments.

f. Coastal wetlands provide climate change mitigation co-benefits through long-term sequestration and storage of carbon mainly in their sediments.

g. The sediment carbon storage in Auckland’s coastal wetlands is greater than terrestrial forests, highlighting the role of coastal wetland ecosystems as permanent carbon sinks.

h. Compared to freshwater wetlands, coastal marine wetlands have less methane emission as salinity hinders the Methanogenesis process.

i. The ‘blue carbon’ potential of coastal wetlands can provide incentive to encourage restoration of these ecosystems.

Given the uncertainty over the long-term persistence of coastal wetlands, understanding the timeframe within which coastal wetlands in Auckland can keep pace with sea level rise and increased frequency and magnitude of storm events is therefore necessary for making decisions on the use of these ecosystems for the purpose of climate change mitigation and adaptation. A zoning mechanism which reflects the different sea level rise scenarios and allows for a flexible response to climate change (as discussed in Section 8.4), can also help with understanding and responding to the timeframe.

### 8.6. Requirements

#### 8.6.1.1. Incorporating the climate change values into cost-benefit assessment

As discussed in Chapter 6 (Section 6.3.3), under section 32 of the RMA, all costs and benefits of a proposal (e.g. a new policy or regulation) should be quantified (if practicable) and assessed (Ministry for the Environment, 2014b). The concept of ‘co-benefits’ particularly in the context of climate change needs to be also taken into account when evaluating different policies (e.g. coastal protection strategies) under section 32 of the RMA.

In fact, the actual expenses of coastal protection strategies such as wetland restoration need to be assessed against the income/revenue that can be earned, for instance in terms of contributing to reducing cumulative GHG concentrations in the atmosphere, as well as the avoided costs (damage) from reducing the impact of climate change on the coasts. The social cost of carbon associated with degradation/removal of coastal
wetlands (e.g. mangrove forests) (See Sections 5.6.1, 5.6.2 & 5.6.3 for more information) needs to be also included in assessing the costs associated with land use and development projects which affect coastal wetlands. However, it should be noted that SCC is only a broad estimate, and actual amounts will vary from country to country due to the variation in policy and economic contexts. But it can be used to give an idea of the scale of the value.

These information, subject to the clear identification of assumptions, limitations and the scope of quantitative information as proposed in the guide to section 32 of the RMA 1991 (Ministry for the Environment, 2014b) can provide a basis for considering the co-benefits of ecosystem-based coastal protection policies such as coastal wetlands enhancement. They can also inform offset programmes (as will be discussed in the next section) about the costs and benefits associated with the climate change co-benefits of biodiversity.

As discussed in Chapter 5 (Section 5.3.4), to identify the amount of avoided erosion and inundation due to presences of vegetation, it is essential to quantify how differently sized plants with differing characteristics can alter water level and shoreline erosion under different storm conditions (Guannel, et al., 2015). Hence, the ability to quantify the protective benefits of wetlands is difficult, as is the ability to conduct cost-benefit against other protection options such as groynes or seawalls. However, it is possible to compare wetland restoration (or plantation) cost with the cost of construction, long-term operations and maintenance of engineered coastal defence structures, as discussed in Chapter 5 (Section 5.3.3).

Any cost-benefit analysis should also include the other co-benefits associated with restoration of coastal wetlands, while noting coastal wetlands with high protection value are unlikely to have high biodiversity and fisheries production values (See Section 5.3.3 for details). In addition, as mentioned in Chapter 7 (Section 7.5.4), using coastal wetlands for coastal protection may require emphasis on sediment accretion and silt stabilisation possibly at the expense of biodiversity and recreational values. This is, however, mainly the case for using saltmarsh ecosystems for coastal protection which requires saltmarsh with elevated edge (the experience form New York City and Wadden Region in the Netherlands). These trade-offs can be addressed by identifying and prioritising biodiversity conservation and coastal protection objectives as further discussed within the following sections.

8.6.1.2. Compensating for the residual impacts of development on coastal wetlands

The underpinning approach to hamper the spread of mangroves is principally at odds with the notion of conservation and preferably rehabilitation under the existing offsetting approach. Unless the Auckland Council decides to set a limit for no net loss of mangroves and requires a targeted offset scheme for mangrove removal across the region, any resource consent that will be granted to an application for mangrove removal would in practice mean that the mangroves allowed for removal lack significant ecological values and hence would not be eligible for offsetting.

In the context of this research, offsets would involve protecting and enhancing coastal wetlands (not limited to mangroves) through developer contributions, where applicable. Large-scale developments such as buildings and sub-divisions could be targeted initially and contribution would be explicitly tied to offsetting the risks of climate change. This could work by developers contributing to enhance/rehabilitate coastal wetlands identified as having particular climate change risk reduction potential, and earning credits in the process. Co-benefits of wetlands restoration programmes, particularly carbon credits can be used to partially cover restoration costs (similar to the blue carbon projects in Indonesia and Louisiana). The requirements of this approach are discussed within the following sections.

As noted above, under the current policies, removal of mangroves is allowed (although discretionary) in areas where mangroves do not provide significant biodiversity values and do not contribute to erosion control. In such cases, provided that mangroves do not contribute to protection against storm surges and sea level rise and/or the area is not at high or very high risk of inundation (as explained in Section 8.4), the possibility of ‘out-of-kind’ or ‘like for better’ offsets can be explored and the loss of CS&S as a result of
mangrove removal can be traded with restoration or enhancement of other coastal habitats such as saltmarsh or seagrass ecosystems that are at risk of degradation. The offset size can be identified in a way that results in a net gain in carbon sequestration. However, this strongly depends on reliable and accurate data about CS&S potential of these three coastal wetland ecosystems that is currently unavailable for Auckland. As outlined earlier, the estimates provided in this research represent approximate values of CS&S by Auckland’s mangroves and saltmarsh ecosystems but are limited by insufficient Auckland-specific data. Therefore, it is necessary to have site-specific measurements of the stored carbon and the cost can be covered by the applicants seeking mangrove removal.

While, ‘like-for-like’ offsets are preferred for achieving the offset objectives (BBOP, 2012b), there are situations where ‘out-of-kind’ or ‘like-for-better’ offset is recommended. This may occur whereby better outcomes could be achieved by restoring or protecting a more highly valued ecosystem which might be identified as a priority for conservation purposes (BBOP, 2012a). In other words, ‘out-of-kind’ means offsetting ‘low priority impact through high priority gains’ (BBOP, 2012e, p. 10) or trading up. Therefore, ‘like for better’ offset can be seen as a strategy in the case of mangrove removal in Auckland where the mangroves allowed for removal are assumed to have no or low ecological values. Instead, the carbon loss as a result of mangrove removal can be offset by restoring saltmarsh and seagrass habitats with high ecological values.

However, identifying ecological equivalence in ‘out-of-kind’ offsets are more challenging than ‘in-kind’ offsets (Bull, et al., 2013). Context-dependent currencies (See Section 6.3.2 for more detail) are also essential where ‘out-of-kind’ offsets are being considered (BBOP, 2012e). As discussed in Chapter 2, offsets provide a tool for addressing the residual adverse effects of development on the environment, but their associated risks and challenges (as discussed in Chapter 7, Section 7.7.3.2) need to be clearly accounted for and managed in the design, during and post implementation of offset projects.

**8.6.1.2.1. Incorporating climate change values into resource consent process and offsetting**

This research identified the main variables that affect the climate change mitigation and adaptation values of coastal wetlands. For example, species type, carbon density, sediment accretion rate and tidal range (subject to similarity in climatic condition) are the key bio-physical factors that influence CS&S in coastal wetlands (Refer to Section 5.2.2, Chapter 5). The coastal protection potential of coastal wetlands is also influenced by a wide range of variables including species type, habitat size (length and width), species density, sediment accretion rate and tidal range as well as storm characteristics, sea level rise, topography, slope and seabed bathymetry (Refer to Section 5.3.5, Chapter 5). These variables can be considered within the biodiversity accounting model as the attributes for carbon storage and the coastal protection functions of coastal wetlands. This helps to compare between the impact and the offset sites in terms of parameters affecting carbon storage and coastal protection. However, as noted earlier, the information provided in this research identify the basis and needs to be further improved.

Incorporation of the climate change mitigation and adaptation values into offset programmes implies that the Auckland Unitary Plan (AUP) policies explicitly account for the values of mangroves with respect to protection against storm events and sea level rise. It also means that proper measures (e.g. avoiding mangrove removal, managed retreat, development setback lines) would be needed to protect mangroves in areas with very high and high exposure to storms and sea level rise or in areas with very high and high coastal vulnerability index (as explained in Section 8.4.1). This research suggests that the discretionary mangrove removal activities need to be evaluated based on ‘their implications in terms of affecting the stored carbon and increasing vulnerability to sea level rise, flooding and storm events’.

The proposed policy language can be as follows:
‘In reviewing resource consent applications, the Council will consider the climate change benefits (including carbon sequestration and storage and coastal protection against sea level rise, storm surge, and extreme climatic events) of coastal wetlands and/or natural coastal buffers’.

The AUP refers to ‘loss of ecosystem services’ as part of the indigenous biodiversity values in significant ecological areas that are required to be avoided, remedied, mitigated or offset (Chapter D, D9.3(2)(h)). It can either include ‘loss of the carbon stored in coastal wetland ecosystems’ or modify the current provision to ‘loss of ecosystem services including carbon sequestration and storage’. It provides an opportunity to include carbon loss as part of the ecological losses caused by development.

As noted above, the coastal protection function of mangroves varies across estuaries due to variations in multiple factors. Moreover, long-term persistence and sustainability of mangrove ecosystems is greatly dependent on availability of space for inland migration due to sea level rise and sediment for keeping pace with sea level rise (Chapter 5, Section 5.3.2). Therefore, it is important to identify areas where mangroves can provide long-term protection and avoid removing mangroves where these areas overlap with the most vulnerable coastal areas.

In the long-term, it is also ideal to plan for retreat from these areas and turn them into the natural state (e.g. coastal wetlands). But, strong public interest to remove mangroves and resistance to their protection, remains the key challenge. A preliminary decision tree to help identifying areas where mangrove can provide long-term protection is further discussed in Section 8.6.1.3.

8.6.1.2.2. Pre-selected offset areas, a strategy to deal with uncertainty in achieving ecological gains

As outlined in chapter 7, mitigation banking offers an opportunity to minimise the risk of uncertainty in achieving ecological gains, as offset has occurred prior to development impacts. It can also contribute to achieving the higher-level goals, consistent with the ‘conservation benefit matching’ approach (as explained in Chapter 6, Section 6.3.2), if priority offset sites and objectives are identified under a landscape or regional scale approach. There is, therefore, a need to identify and prioritise catchment or regional ecological targets and manage land use activities and offset projects in a way that contributes to achieving these targets. It helps to ensure that conservation and coastal protection efforts are targeted to where they will make the most difference.

In New Zealand, regional councils are currently establishing freshwater management units (FMUs)—essentially large-scale catchments – in order to identify land use activities needing to be controlled to achieve receiving environment goals. The FMUs can be also used as a spatial scale for identifying ecological improvement targets and offset sites which can provide credits for land use activities within the FMU boundaries.

Ideally, a national offsetting framework, by taking a strategic approach, can identify the conservation targets for each ecosystem across different regions and prioritise conservation and offset activities to improve the ecological values of the most threatened ecosystems. It can also be combined with a strategic coastal protection plan to identify areas where conservation and enhancement of coastal wetlands will improve adaptation to the effects of climate change.

As mentioned earlier, sites with high priority for biodiversity conservation do not necessarily coincide with areas where provision of particular ecosystem services (e.g. carbon sequestration, or coastal protection) is of high priority. Thus, it is important to identify whether offsets are targeted to improve biodiversity, ecosystem function, ecosystem services or a specified combination of these.
8.6.1.2.3. Modifying the existing use rights to account for biodiversity conservation on private lands

As mentioned in Chapter 6 (Section 6.4.2), issues around property rights and their erosion seem to be the most challenging barrier to preserving biodiversity on private lands. In the case of private coastal wetlands in Auckland (although their extent is not yet clear), it would be necessary to clarify the legal position of property owners regarding requiring compensation for the activities that may adversely affect coastal wetlands and their ecological values.

8.6.1.3. Identifying areas where mangrove can provide long-term protection

This research proposes a decision tree (Figure 8.6-1) to help identify mangrove sites which can be good candidates for long-term coastal protection.

This framework is built upon the following main principles:

- Whether the hydrological condition allows consistent, unimpeded and minimum sediment delivery that ensures long-term sustainability/survival of the mangrove ecosystem and enables appropriate levels of sediment accumulation
- Whether the characteristics of the site and mangrove habitat allow the acceptable level of protection, or if not, whether enhancement of the mangrove area can improve the protection level
- Whether landward migration of mangroves is allowed (both physically and legally) or if not, whether there is a possibility to create space for landward migration

Subject to availability of information, mangrove recovery can be also considered in identifying areas where mangroves are good candidate for coastal protection.

The proposed framework only provides a basis for consideration and will need further investigation. An investigation in this area will require collaborative work involving various experts and practitioners of multiple disciplines. Moreover, any decision regarding management and protection of coastal wetlands needs to include and consider other valuable services particularly their contribution to long-term CS&S.

A requirement for application of this framework is the estimation of the site-specific storm protection potential of mangrove forests. Chapter 5 (Section 5.3.4) explains the parameters that affect the coastal protection potential of coastal wetlands particularly mangroves. Since it is a new area of research, there is currently limited information available about the methods that can be used to identify the capacity of coastal wetlands for reducing damage from storm surge and erosion. The only developed model that is available online is the newly developed InVEST Nearshore Waves and Erosion Model\(^\text{188}\) that quantifies the protective services provided by natural habitats of nearshore environments in terms of avoided erosion and flood mitigation. The data requirements are explained in Chapter 5 (Section 5.3.1) and the full instruction is available online through the Natural Capital Project website\(^\text{189}\). Feasibility of using this model for a specific mangrove or saltmarsh area in Auckland can be examined through future studies. The proposed decision tree can be used in conjunction with scenarios of future uncertain climate change (e.g. under different sea level rise scenarios and associated storm and flood scenarios) to avoid planning based on static climate risks. This means that regulatory authorities need to clearly identify which scenario (or set of scenarios) is being used to base a decision on. The decision process would also need to be revisited regularly as revised scenarios are released.


8.6.1.4. Monitoring surface elevation and tidal range in coastal areas

As mentioned in Chapter 5 (Section 5.3.2), the balance between sediment accretion and sea level rise determines the ability of mangrove and saltmarsh ecosystems to keep pace with sea level rise. Any changes to upstream sediment supply (i.e. minimising sediment supply to estuaries) will affect the balance between sediment accretion and sea level rise and may result in ‘elevation deficit’ which threatens long-term sustainability of mangrove ecosystems. In such conditions if landward migration of mangroves is not possible, mangroves will be gradually submerged due to sea level rise (Lovelock, et al., 2015). In fact, the sedimentation inputs, the rate of sea level rise and the magnitude (threshold level) of storms are the key factors that determine the efficacy of protection capacity and longevity of coastal wetlands.

It is therefore necessary to monitor the surface elevation (i.e. balance between sediment accretion rate, erosion and subsidence) in Auckland’s mangrove and saltmarsh ecosystems over time to ensure that they can keep pace with sea level rise over a long run. The Auckland Council regularly monitors the quality of marine water across the Auckland region (Auckland Council, 2015d). It would be beneficial to add parameters such as sediment organic carbon, bulk density and surface elevation/accretion (SVI: Submergence Vulnerability Index) similar to the information provided by the online Coastwide Reference Monitoring System (CRMS) developed for Louisiana’s coastal wetlands (See Figure 7.2-7 in Chapter 7). The monitoring system can also help to identify the cumulative effects of activities on coastal wetlands.
8.6.1.5. Improve public awareness

In the case of New Zealand and Auckland, the problem of mangrove expansion in combination with the lack of local information about the mitigation benefits and debates around the adaptation values of mangroves has been among the key challenges for land use and urban planning in Auckland. The literature suggests that these types of knowledge issues can lead to misconception and consequently misrepresentation of the values of the coastal ecosystems through land use planning and decision making (Uittenbroek, et al., 2013; Tribbia & Moser, 2008; Moser & Ekstrom, 2010). As explained in Chapter 6 (Section 6.4), coastal communities in some estuaries at risk of inundation and sea level rise (e.g. Omaha beach/Whangateau Harbour) strongly demand removal of mangroves mainly to reinstate amenity values of the coast. This can be attributed to the lack of understanding about coastal dynamics that determine mangrove colonisation. Both removal and planting mangroves may not be permanent as mangrove settlement patterns are dictated by catchment runoff and hydraulics.

Understanding of the characteristics and impacts of sea level rise is also poor which compounds the other misunderstandings about mangroves and their carbon co-benefits.

One thing that locals are often not aware of is that changes in the erosion and accretion patterns of sandy shorelines may take decades and would be beyond the life experience of many. Conveying the science about coastal processes is therefore critical for locals to understand risks and probabilities even without accounting for more extreme events under climate change. Such knowledge sharing would involve enhancing public awareness about the diverse values of coastal wetlands particularly mangroves, their role as natural buffers and uncertainties in achieving sandy estuaries as a result of mangrove removal.

8.7. Summary

This chapter proposed a framework for developing an integrated coastal protection and resilience plan for the Auckland region that is based on the understanding of vulnerabilities of both coastal communities and resources (wetlands) to climate hazards. The framework is aimed at mainstreaming climate benefits of coastal wetlands into Auckland’s land use and resource management decisions. The proposed framework provides the principles and identifies a set of policy mixes and instruments as a guidance for Auckland to ensure that its coastal natural resources will be properly protected and enhanced for their adaptation and mitigation benefits. This framework can be further expanded and improved.

This approach provides for the improvement of Auckland’s current planning system such that urban development, whether coastal or inland within a given catchment, needs to account for the implications on the climate mitigation and adaptation benefits of coastal wetlands. This includes the coastal protection and carbon sequestration and storage benefits under different scenarios.

If pursued seriously, the proposed approach would potentially constitute a fundamentally different way of assessing ecological impacts and their mitigation and compensation. This in turn requires a shift in the assessment of land use practice, and therefore resource consent processing criteria, to include the need to create a net benefit measured by (in this instance) climate change response capacity. Understanding the dynamic nature of climate risks including sea level rise and storm events and their impacts on sustainability of coastal wetlands together with developing flexible measures to adapt to the long-term risks of climate remain the key considerations and challenges.
9. Conclusion

This chapter provides a synopsis of findings in relation to the research questions, and discusses the findings in relation to the broad theoretical framework in Chapter 2. The chapter follows by an outline of the key contributions and limitations of this research and identifies topics for future research.

9.1. Addressing the research questions

As discussed in Chapter 1, the preliminary review in this research used the carbon sequestration data for six selected land use categories to understand how changes in land use may affect carbon sequestration. The purpose was to examine whether it is possible to provide planners with a way of assessing the emission implications of land use change activities. The review found that carbon sequestration in both biomass and soil is greatly influenced by parameters that vary locally and hence using global data to quantify the magnitude and scale of change at local scale involves high level of uncertainty. The results of that review together with the findings of a preliminary review of global literature about climate change mitigation and adaptation values of coastal wetlands assisted in narrowing the focus of this research to be on coastal wetlands and their climate change values in the Auckland region.

As discussed in Chapter 1, this research then addressed the following specific questions:

- How do coastal wetlands in the Auckland region currently and potentially contribute to climate change mitigation and adaptation?
- Whether and how the climate change services of coastal wetlands are taken into account in current coastal resource management and climate change response policies and plans in New Zealand and Auckland, and what are the key challenges and opportunities?
- What can be learned from the experiences in other jurisdictions in terms of the initiatives and mechanisms that can assist in incorporating climate change services of coastal natural resources into land use, resource management and climate change policies?
- How do the policies and practices in other jurisdictions compare to the case of Auckland and New Zealand?
- Depending on the outcome of A to D, investigate what policy options and planning instrument mixes can be applied to Auckland for mainstreaming climate change benefits of coastal wetlands into land use and resource management decision-making processes.

How do coastal wetlands in the Auckland region currently and potentially contribute to climate change mitigation and adaptation?

The climate change services of Auckland’s coastal wetlands are discussed in Chapter 5. Given the lack of national and local estimates of the aerial extent of seagrass ecosystems, the coastal wetlands studied in this research only included mangrove forests and saltmarsh ecosystems (See Chapter 3). The review of the global literature in Chapter 5 showed that carbon sink and coastal protection capacity of coastal wetlands are influenced by a wide range of bio-physical and climatic parameters and thus are difficult to be exactly quantified and estimated without site-specific information.

In the Auckland region, the exact contribution of its coastal wetlands to climate change mitigation and adaptation is currently unclear due to the lack of empirical data. However, based on the global literature review, the fact that these ecosystems can store carbon and protect against sea level rise and storm surges is undeniable. The discussions in chapters 5 and 7 strongly suggest that coastal wetlands are valuable ecosystems that provide a wide range of ecological services including climate change benefits. The survey results in Chapter 6 also showed that majority of the participants surveyed gave high to very high importance to carbon storage and sequestration as well as storm surge protection values of mangrove forest and saltmarsh ecosystems.
The best available local and global data used in this research suggest that Auckland’s coastal mangrove and saltmarsh habitats with an approximate area of 10,984 ha can potentially store about 1.5 million tonnes of carbon in the first meter of their sediment which is equal to 5.5 million tonnes of CO$_2$ (Chapter 5). Considering a range between NZ$ 19 to NZ$ 40 per tonne of CO$_2$, as the current cost of GHG emissions (New Zealand Transport Agency, 2016), the value of the carbon stored only in the sediment of mangrove and saltmarsh ecosystems is estimated between NZ$ 104 and NZ$ 220 million. Based on a social cost of carbon (SCC) estimated at US$220 per tonne (Moore and Diaz, 2015) this is worth about US$ 1,200 million. The results also provided an approximate value of 44,000 tonnes of CO$_2$ per year as the net rate of carbon sequestration by temperate mangrove and saltmarsh habitats. Based on the SCC, this estimate suggests about US$9.6 million per year. These estimates can provide an approximate value that can be used as a basis for decision-making until more accurate local data is available. Note that the lower value of carbon reflects current market demand, while SCC more fully accounts for the actual costs of an extra tonne of CO$_2$ entering the atmosphere as a result of an action, in this case coastal wetland losses (See Section 5.2.1, Chapter 5).

Unlike carbon sequestration and storage capacity which is quantifiable for Auckland based on local and global data (although with some degree of uncertainty), quantifying the coastal protection values of Auckland’s mangrove and saltmarsh ecosystems was not practical for two reasons. First, as discussed in Chapter 5, the coastal protection capacity of coastal wetlands is greatly influenced by a range of parameters that vary depending on local conditions and therefore empirical local data is required for reliable and accurate estimates. Second, studies that have quantified coastal protection values of coastal wetland are globally limited.

While the few global and local studies that monetised the storm protection values of coastal wetlands provide some insights into the significance of coastal wetlands for protection of shorelines against storms and climate hazards, their application to New Zealand is limited due mainly to the range of variations in local conditions and the lack of clear distinction between wetland types in the studies. These studies, however, provide some useful insights suggesting that coastal wetlands in Auckland can potentially protect coastlines if the rate of sea level rise and frequency of coastal storms do not advance beyond the ability of these ecosystems to respond and if there is enough space for landward migration of these ecosystems. Managing and maintaining sediment delivery to estuaries is therefore highly important, as sediment availability is a key factor determining the ability of coastal wetlands to keep pace with sea level rise.

In general, the current knowledge suggests that the capacity of mangrove forest and saltmarsh ecosystems to protect coastlines vary by their properties and the type of coastal hazard (section 5.3.4) and that the health, integrity and connectivity between ecosystems are the prerequisites for effective coastal protection by these ecosystems. Moreover, compared to hard structures that require costly upgrade and maintenance, coastal wetlands can naturally adapt to the changes in SLR and recover after climatic disasters and could therefore be more cost-effective options in a long term.

In summary, these findings highlight that while the climate benefits of coastal wetlands are undeniable, the scale of benefits are uncertain in Auckland. If not properly dealt with, this uncertainty can significantly undermine the whole purpose of internalising the climate values of coastal wetlands into the Auckland’s mainstream urban planning and land-use decision-making process.

In addressing the issue of uncertainty, this research recognises that the lack of sufficient evidence and uncertainty in knowledge is a substantial element of policy and decision-making and cannot be easily avoided in practice. However, uncertainty in decision making and planning can be addressed through a variety of ways.

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The spectrum would arguably include a range of approaches from the ones that completely ignore the values of ecosystems at the very early stages of policy process, to those that partially recognise the values at strategic level (e.g. vision and strategic priorities) but not through the implementation phase, and ultimately to the approaches that fully incorporate the values into all stages of the policy and planning cycle with various strategies to deal with uncertainties. This thesis argues that the latter approach is necessary for addressing the issue of uncertainty in the context of environmental policy integration or mainstreaming.

Here, the theoretical perspectives discussed in Chapter 2 provide a basis to frame the issue of uncertainty in the context of environmental mainstreaming. In the first instance, mainstreaming requires an understanding and sufficient recognition of the significance of the environmental concerns (in this case ecological values of coastal wetlands). Such knowledge and understanding is necessary in all stages of the policy cycle and particularly crucial during the first stages i.e. setting and prioritising goals and for decision making through implementation of the policies e.g. through resource consent process and quantification of net ecological benefits for offsetting ecological losses.

Environmental mainstreaming is fundamentally based on a precautionary approach (or prudent foresight) that in principle aims to guide decision-making where the decisions might have irreversible consequences especially in a long-run because of the uncertainty and lack of consensus regarding the very nature of the problem (the harm) and its possible solutions or the effectiveness of the outcomes. In practice, this approach influences the decision-making by shifting the ‘burden of proof’ to the proponents of potentially harmful activities (e.g. mangrove removal). This approach is consistently applicable to all stages of policy and planning cycle.

In the case of the decision about mangrove removal, from a precautionary perspective, while incomplete, the current evidence about the climate values of coastal wetlands would be sufficient to include conservation or restoration of these ecosystems in the policy agenda. Removal of wetlands as shared resources would mean the loss of a potential carbon sink and cost-effective means of protecting coastlines against climate hazards.

Shifting the burden of proof from the community (the Auckland Council) to applicants of mangrove removal requires applicants to provide evidence in support of their proposals and accept the associated liabilities. This would apply to both the public and privately-owned wetlands (mangrove forests) on the basis that the benefits of ecosystems are not confined to defined boundaries. This rationale would hence justify the mechanisms such as ecological compensation to ensure no net loss (and net gain in the long-term) of the potential benefits of the coastal wetlands for carbon sequestration and coastal protection.

Likewise, the complex, in-determinant and wicked nature of the environmental problems signifies the importance of learning-based adaptive strategies within the framework of the resilience theory as discussed in Chapter 2. The resilience perspective suggests an incremental and transformative shift in social norms and constructs through adaptive governance of natural and physical resources in the face of climate surprise. In this regard, the resilience framework and precautionary approach can provide guidance on how to address the issues of uncertainty through the planning process.

In the case of the Auckland’s problem with mangrove management, this research argued that there is sufficient evidence to dispel the existing scepticism about the values of coastal wetlands (including mangroves) for climate change mitigation and adaptation. The uncertainty is hence not about existence of the values, rather it is about the scale of benefits. The latter implies identifying, in quantitative terms, the net carbon sequestration and storage and coastal buffer capacity of mangroves. However, quantification of values within a long-term scale is likely to be problematic due to the uncertainty about the response of mangroves to the rate of sea level rise and increased frequency of storms. Using assessment tools especially designed for addressing uncertainty (as mentioned in Section 2.3) can help inform decisions in the face of uncertainty.
While this level of detailed understanding of the values may be required for decisions about how much, where and when mangroves can be removed, and whether and how that should be compensated by mechanisms such as ecological offsetting and carbon credit schemes, it is not necessary for strategic planning and decisions about whether those values should or should not be explicitly included in statutory policies such as regional coastal policy statements or zoning regulations.

This highlights the importance of understanding where the uncertainties are placed in the policy cycle and planning process, whether the uncertainty is about benefits or valuation of benefits and how the planning system frames and defines outcomes. Equally important is how these issues are communicated broadly with public and key actors; given that often lack of consensus about the nature and sources of uncertainty, where it lies, whether and how it affects outcomes of interest and how it can be resolved is itself a source of conflict. As discussed, in Auckland, the uncertainty is more about valuation of benefits, which can manifest itself through implementation of the plans and policies.

Within this framework, in order for Auckland to deal with this issue, the region will need to adopt appropriate mechanisms to implement the precautionary principle as discussed earlier, develop methodologies to guide the ecological valuation of its mangrove ecosystems, reassess its policies by learning from results of the evaluations in practice and communicate consequences of the incremental policy change and adaptive resource management on resilience and adaptive capacity of the region.

Whether and how the climate change services of coastal wetlands are taken into account in current coastal resource management and climate change response policies and plans in New Zealand and Auckland, and what are the key challenges and opportunities?

This question was addressed in Chapter 6 through a comprehensive review of both national and local policy documents relating to the management of coastal resources, responding to climate change and management of coastal hazards in New Zealand and Auckland. The findings suggest that although the importance of preserving the coastal and marine environment is recognised within the current legislative framework and reflected in the planning practice in New Zealand and Auckland, the existing framework remains silent about mitigation values and understates the adaptation values of coastal wetlands. The following summarises the major downsides of the current legislative and planning framework with respect to accounting for the climate values and benefits of coastal wetlands and highlights some opportunities for improvement.

As discussed in Chapter 6, coastal wetlands are primarily managed under the Resource Management Act (RMA), the New Zealand Coastal Policy Statement (NZCPS), Regional Policy Statements (RPS) and Regional Coastal Plans. The NZCPS and other national guidelines provide relatively clear directions for local governments to apply an integrated approach when defining, planning and managing their regional coastal environment. The integrated approach primarily involves taking account of cumulative effects of land use activities within catchments of the coastal receiving environments through trans-jurisdictional, strategic and collaborative planning.

Despite the strategic emphasis on integrated management of coastal natural resources, the ambiguities about the geographic (particularly landward) extent of coastal environments and the lack of a clearly defined and mandated connection between land use and resource management within coastal areas and their upstream catchments along with a combination of institutional, political, financial and technical hurdles have constrained realisation of the integrated approach. The New Zealand’s National Policy Statement for Freshwater Management, enacted in 2014, (NPS-FM) has been promising in filling this gap primarily by mandating catchment-wide planning and management of freshwater and coastal water resources.

However, there is a lack of a clear definition and classification for coastal wetlands and their values, a paucity of knowledge and a general public misconception about ecological (especially the climate) values of coastal wetlands. In addition, there is a division of responsibilities between local and national
governments for mitigation and adaptation. These remain to be the key challenges for planning and management of coastal wetlands in relation to their climate benefits. Lack of a definite operational framework and guidance for local government on the choice and application of appropriate mixes of policy instruments and measures may undermine the effectiveness of the NPS-FM.

Despite the potential utility of incorporating planning instruments such as biodiversity offsetting (or generally environmental compensation) into the mainstream statutory planning processes for integrated management of land use activities, the non-mandatory status of existing biodiversity offsetting in New Zealand undermines its effectiveness (See Section 6.3.2). Likewise, the currently fragmented legislative and policy landscape that governs the New Zealand’s response to climate change is not in accord with the idea of using market-based methods such as carbon credit schemes for coastal resource management. This can be partly attributed to the difficulties in placing coastal restoration efforts in the context of carbon credit within the global or national carbon credit schemes. Added to these issues is the lack of an inclusive framework linking land use decisions with changes in coastal natural resources, coastal hazards and response to climate change.

Notwithstanding the drawbacks discussed above, the current legislative, policy and planning landscape provides some opportunities for reconstituting the values and benefits of coastal wetlands including their adaptation and mitigation values into the mainstream planning and management of coastal and freshwater resources in New Zealand. As discussed, among the most prominent opportunities are the provisions in the NZCPS for protection and enhancement of natural buffers including coastal wetlands, the non-mandatory biodiversity offset guidelines and the provisions in the NPS-FM for establishment of freshwater management units.

In the context of Auckland, the current policy and planning practice in relation to management of coastal wetlands has largely been a response to the unwelcomed problem of mangrove expansion associated with sedimentation within catchments. The recent Auckland Unitary Plan (AUP) treats saltmarshes as habitats threatened by expanding mangroves and hardly talks about seagrass ecosystems. Evidently, the public pressure and the policy discourse on how the mangrove expansion should be managed have been the primary drivers behind the biased policy position towards mangroves, which has also been influenced by the lack of knowledge about seagrass ecosystems and the relatively large extent of mangrove forest compared to saltmarsh habitats (as discussed, mangroves constitute about 80% of the Auckland’s total extent of mangrove and saltmarsh ecosystems).

The review in Chapter 6 suggests that the issue of mangrove management has not been thoroughly discussed in the context of climate change and specifically from the viewpoint of their mitigation and adaptation benefits. The recognition of carbon sequestration capacity of marine ecosystems in the non-statutory Auckland’s low carbon strategy (Auckland Council, 2014) is arguably a communicative strategy that is neither aimed for nor likely to influence the AUP’s policies. As a non-mandatory strategy, the document was primarily aimed at supporting the Auckland Plan strategic directions on reducing GHGs emissions and improving energy resilience. While the strategy aims, at a very high level, to determine and enhance carbon sequestration capacity of the Auckland’s marine environment, the long-term horizon of the strategy (2020-2040) and its non-mandatory nature undermine its effectiveness in practice. The long-term target to improve knowledge about the carbon sequestration capacity of marine ecosystems in the Auckland’s low carbon strategy raises the question of what would happen to the lost mangrove stock until knowledge about the carbon sequestration capacity is developed.

Although the AUP takes a more conservative approach to mangrove management, removal of mangroves as a discretionary activity is still allowed (subject to case-by-case assessment) in areas where they do not provide significant ecological values or do not contribute to protection against active coastal erosion. The current rules for resource consent do not qualify the removed mangroves for biodiversity offsets on the basis that mangroves that are consented for removal are fundamentally the ones that are identified as lacking biodiversity values. Likewise, the rules do not take into account the potential ecosystem services
of the same removed mangroves such as carbon sequestration and storage and coastal protection (against sea level rise and storm surges). These types of services are not currently taken into account in assessing the mangrove removal applications.

The problem of coastal wetland (mangrove) management in Auckland is clearly a resource management issue that has been isolated from the Auckland’s climate response policies. As a result of this fragmentation, climate values have literally no impact on decisions for preserving or restoring coastal wetlands in Auckland. As a unitary authority, the Auckland Council has more opportunity for policy integration compared to other regions of New Zealand where the distribution of responsibilities between regional and territorial authorities often influences the decisions on cross jurisdictional issues that require some form of policy integration.

Despite this opportunity, there is still an apparent disconnect between the Auckland’s land use policies, management of coastal wetlands and climate response. Within this context, incorporating climate values of the Auckland’s coastal wetlands into its resource management and climate response policies would imply preservation or restoration of these resources, which seems to be at odds with the current policy paradigm that principally seeks to control mangrove expansion rather than preserving or restoring them. Considering these issues in the context of the theoretical framework discussed in Chapter 2 highlights the need for a holistic and procedural framework with an appropriate mix of mechanisms and tools to enable policy integration across the entire policy cycle.

In terms of opportunities, the recent global attempts such as developing methodologies for quantifying the blue carbon potential of coastal wetlands and the recent IPCC approach to incorporate the emissions and removals from coastal wetlands are the opportunities for incorporation of the climate change benefits of coastal wetlands in the national mitigation strategies.

However, as discussed, limitations such as insufficient local data particularly regarding carbon sink capacity of coastal wetlands, negative public impression about mangroves mainly because of their expansion and the divergence in the distribution of responsibilities for mitigation and adaptation between local and central government, will likely challenge incorporation of climate change services of coastal wetlands into the land use planning for Auckland’s coastal areas. Moreover, issues around the existing use rights in coastal areas may challenge long-term planning for coastal protection (See Section 2.4).

What can be learned from the experiences in other jurisdictions in terms of the initiatives and mechanisms that can assist in incorporating climate change services of coastal natural resources into land use, resource management and climate change policies?

This question was addressed as part of the global case studies in Chapter 7. The following outlines the key observations.

All cases studied are big coastal cities with various types of coastal wetlands, a long history of dealing with extreme climate hazards and the depletion of coastal wetlands as a major resource management problem. While hardening shorelines with engineered coastal defence has been a conventional approach to shielding coastal urban areas from natural hazards in these cases, this paradigm has dramatically shifted towards an emphasis on resilience and ecosystem-based adaptation. The initiatives adopted in the cases studied involve a mix of hard, soft and nature-based solutions and strongly focus on multiple benefits of coastal natural ecosystems as a means to foster synergies between policies.

Except for Rotterdam with a long history of flood and coastal inundation, the policies and initiatives for using nature-based solutions for coastal protection in other jurisdictions are driven by tsunamis and hurricanes. The initiatives in all cases have been aimed at and resulted in protection and recovery of wetlands, while the restoration and conservation of wetlands have been previously discrete areas of policy and action. While in all cases, coastal wetlands are used for coastal protection, the initiatives used to mainstream coastal wetlands into coastal protection strategies and plans vary among the cases studied.
The review of cases suggested that the choice of initiatives and policy mixes can be highly context dependent and largely depends on how jurisdictions, individually or collaboratively, deal with trade-offs and co-benefits, set priorities and mix their policies and instruments within a broader strategic framework. Rotterdam, for example had to deal with a trade-off between biodiversity and coastal protection for saltmarshes given wetlands with elevated edge are likely to have less biodiversity values. Louisiana pumps sediment to restore wetlands principally for the purpose of coastal protection, noting that the restored wetlands would provide multiple other co-benefits.

The relationship between institutional and governance structure of the cases reviewed and their climate change initiatives is not straightforward. However, the review suggested that where local governments and specially mayors have more constitutional autonomy, there is generally more potential for agile planning response. This is crucial for building resilience and post-hazard recovery and restructuring.

While in developed countries, adaptation values of coastal wetlands are incorporated into local planning and policy systems, restoration of mangroves for coastal protection in developing countries takes the form of internationally financed projects/programmes and hence not fully incorporated into plans and policies. The Netherlands has also adopted a building with nature approach and incorporated its natural systems (tidal parks) into its coastal protection policies.

Also given that almost all initiatives and plans are very recent and that restoration of wetlands may take years or decades to fully establish, at this stage, it is not possible to comment on actual outcomes and effectiveness of the plans and initiatives. However, the initiatives provide some useful insights into what opportunities exist and can be tailored and used in other coastal urban areas towards enhancing coastal resilience while achieving multiple other benefits.

While reducing vulnerability and enhancing adaptation to coastal hazards including climate impacts is the main focus and dominant discourse in most cases, the comprehensive and integrated planning approach adopted by a number of jurisdictions has provided them with opportunities to account for other co-benefits of ecosystem-based adaptation initiatives. In Louisiana, for example, the inclusion of coastal wetlands in local carbon market has been used as an incentive to encourage energy sector to invest in wetland restoration aimed at coastal protection. New York City uses a mitigation banking mechanism to restore wetlands in low lying areas where they can function as coastal buffers and provide other ecological benefits.

The review of cases suggests that a mix of structural, non-structural and nature-based strategies and tools can be utilised for coastal protection while providing for other co-benefits. Regulatory and voluntary policy instruments such as vulnerability assessment and zoning, insurance policies and voluntary buyout and acquisition programmes are examples of non-structural solutions that can limit further development in vulnerable coastal areas or facilitate retreat from these areas.

As mentioned, compared to hard protection structures (e.g. levees or seawalls) enhancement of coastal wetlands is generally a cost-efficient, long-term solution for protection against sea level rise and storm surges. This is because the cost of wetland restoration is normally less than building cost of engineering structures and coastal wetlands will not require extensive maintenance or upgrade. Moreover, unlike levees and seawalls that will likely affect estuaries and coastal ecosystems by changing sediment supply or saltwater intrusion, restoration of coastal wetlands will rarely disrupt other habitats. Coastal wetlands which are located seaward of engineering structures can also reduce the maintenance costs of hard measures as they provide a buffer and prevent frequent exposure of structures to waves.

However, coastal wetlands may not be effective in all instances particularly at the time of extreme climatic events. They are also often limited by existing coastal development which affect their long-term sustainability. The review therefore suggests using a combination of soft and hard engineering measures to provide effective protection against coastal hazards. As mentioned, tsunamis, hurricanes and floods have been the main drivers for incorporating natural systems into coastal protection strategies. In most cases,
effectiveness of natural systems has been identified through post-coastal hazard observations of the damages. This suggests the importance of precautionary approach and proactive planning for coastal protection.

**How do the policies and practices in other jurisdictions compare to the case of Auckland and New Zealand?**

The differences and similarities between Auckland and the cases studied are discussed in Chapter 7 and summarised below:

The nature of resource management problems in relation to coastal wetlands differs between the cases studied and the case of Auckland. While based on limited data, unlike the cases studied, mangroves appear to be expanding in Auckland. This major difference could influence priorities, policy mixes and outcomes in relation to management of coastal wetlands within the framework of response to climate change. For example, while Auckland plans to reduce sediment input into its estuaries to control and prevent mangrove expansion, Louisiana pumps sediment to its coastlines to restore or create saltmarsh wetlands for protection against coastal hazards.

Another difference is that the composition of coastal wetlands in Auckland is different from the cases studied. However, this difference does not undermine the overall conclusions of this review. Amongst other key differences are the lack of an integrated and comprehensive coastal protection plan in Auckland, the divergence between coastal wetland management and coastal protection against climate hazards (as was the case in Louisiana and New York prior hurricanes Katrina and Sandy) and non-mandatory nature of mechanisms such as ecological compensation and offsetting.

Moreover, the Coastwide Reference Monitoring System (CRMS) designed for coastal wetlands in Louisiana provides site-specific information such as surface elevation/accretion (SVI: Submergence Vulnerability Index) which is used to identify the vulnerability of coastal wetlands to the changes in sea level based on their ability to keep pace with SLR (Chapter 7). Such an indicator to monitor the balance between surface sediment accretion and erosion/subsidence in coastal areas still does not exist in Auckland. The current policies are not also clear about how the vulnerability of coastal ecosystems to the future changes in sea level will be determined and how the vulnerable coastal ecosystems will be protected. Auckland is also unclear about long-term sustainability of its mangrove and saltmarsh ecosystems especially with respect to their sediment requirements within the framework of the spatial planning for Hauraki Gulf Marine Park (HGMP).

Similar to Auckland, the carbon sink potential of coastal wetlands is only recognised within the non-statutory plans of some cases. For the cases studied, this does not seem to have been a prerequisite for protection or restoration of coastal wetlands, as the statutory policies in these cases already provide for enhancement and restoration of mangrove and saltmarsh ecosystems for coastal protection. This is opposite to Auckland, where the overall intention of statutory policies is towards removal of mangroves.

Compared to New Zealand, the greater autonomy and power of local governments in the United States (e.g. the cases of New York and New Orleans) also seems to have influenced the role of local governments to intervene in formulation and delivery of innovative but legally binding policies for local climate change mitigation and adaptation.

**Depending on the outcome of the previous questions (i.e. questions A to D), investigate what policy options and planning instrument mixes can be applied to Auckland for mainstreaming climate change benefits of coastal wetlands into land use and resource management decision-making processes?**

In Chapter 8, this research proposed an integrated approach for Auckland in an attempt to guide the Auckland’s land use and resource management processes towards achieving win-win and preferably triple-wins outcomes. This approach was developed on the basis that Auckland, as New Zealand’s important
economic, commercial and social hub, needs to be resilient to the foreseeable climate hazards and ensuring long-term sustainability of the region’s coastal wetlands can provide that capacity to the entire region. Alongside this capacity, coastal wetlands are significant sinks of carbon and can also offer multiple other ecological and economic benefits for the region.

The approach is informed by the theories of urban sustainability, resilience and ecosystem-based adaptation, guided by the rubric of climate compatible development and the global case studies, and suggests a mix of tools and mechanisms with a set of parameters and variable as a guiding framework for Auckland to develop an integrated coastal protection and resilience plan. The framework includes a combination of policy tools and instruments such as vulnerability assessment, zoning regulations, incentive-based mechanisms, cost-benefit assessment, strategic environmental assessment, ecological offsetting or bio-banking and monitoring. These are briefly outlined below:

**Vulnerability assessment**: is proposed to identify vulnerability of Auckland’s coastal areas to multiple risks such as coastal inundation, sea level rise and erosion and to develop site-specific coastal protection strategies;

**Participatory land use planning approaches**: Some examples from the case studies include voluntary ‘Buyout and Acquisition Programmes’ in New York State; the ‘Resilient Neighbourhood Programme’ and ‘Transferable Development Rights’ in New York City and the ‘Resilient Retrofit Programme’ in New Orleans;

**Strategic planning**: this involves developing site-specific coastal protection measures (including a mix of nature-based, structural and non-structural options) with a regional perspective and based on the level of vulnerability, with a focus on the most vulnerable coastal areas. This requires holistic and spatial planning to ensure the tools and strategies assigned to each site are consistent and complementary and in combination provide for regional outcomes. The concept of ‘co-benefits’ particularly in the context of climate change also needs to be also taken into account when evaluating different policies (e.g. coastal protection strategies) under section 32 of the RMA;

**Ecological Assessment**: that involves incorporating climate change values of coastal wetlands into zoning regulation and consent planning process to require development activities with potential impacts on coastal wetlands to be assessed based on their impacts on the climate change values of coastal wetlands. This includes weightings against the costs (using social cost estimates) associated with loss of ecological values including climate change values of coastal wetlands;

**Ecological compensation and offsetting**: If properly designed with a regional and holistic view and linked with other tools and instruments, this mechanism can provide significant ecological benefits and enhance ecological sustainability of both coastal wetlands and other terrestrial ecosystems. This involves designing a mandatory offset or ecological compensation scheme including identifying and zoning offset sites based on a holistic regional approach, developing an accounting system, methodologies and procedures to support ecological valuation, decision making and monitoring;

**Regulation for private property rights**: by setting regional rules that support establishment of finite terms for the use of land within coastal hazard risk areas and changing the permanent utility of titles to reflect the impermanence of the land (e.g. changes in house titles from fee simple to lease-hold land with a defined term of tenure);

**Zoning**: once vulnerabilities are identified, Auckland’s coastal areas can then be grouped into ‘retreat’ and ‘protect’ zones (as explained in Chapter 8, Section 8.4.1);

**Incentive-based mechanisms**: including a specific flood insurance scheme at the national scale that considers the elevation of properties and their overall exposure to coastal hazards in estimating insurance premiums, and examining feasibility of high flood insurance premium or no insurance for new properties
Monitoring: of e.g. surface elevation and tidal range in coastal areas and plan for landward migration of mangroves in areas where surface elevation cannot keep up with sea level rise. Monitoring of values and social impact triggers (i.e. anticipatory behaviour triggers rather than just biophysical) can also help in developing proper adaptation measures. This is because monitoring only environmental changes could lead to taking action too late.

The proposed strategic framework is based on the assumption that protection, restoration or enhancement of coastal wetlands for coastal protection is a win-win approach as it can offer multiple benefits rather than one. However, areas where coastal wetlands can provide high protection values are not necessarily the areas where they provide high biodiversity values. In addition, creation of saltmarsh habitats for coastal protection requires sediment deposition and accretion which may not be appropriate for achieving biodiversity conservation objectives. Therefore, as suggested in Chapter 8, it is important that the priority areas for restoration of coastal wetlands for different purposes (e.g. coastal protection, biodiversity conservation) are identified with an ecosystem-based approach and based on the regional or local priorities and ecological improvement targets.

9.2. Summary of findings

The fact that coastal wetlands have significant climate change mitigation and adaptation benefits is undeniable. Compared to terrestrial ecosystems, coastal wetlands have greater potential to accumulate carbon in their sediments and if undisturbed can retain carbon for a long period of time. Coastal marine wetlands are also greatly advantaged over terrestrial freshwater wetlands for their lower methane emission due to their salinity. The coastal protection capacity of coastal wetlands is also evident. It is therefore worthwhile to account for these benefits in land use planning.

Land use decisions that rely on climate benefits of coastal wetlands need to be conscious of the long-term sustainability and resilience of these resources against impacts of climate change. Long-term sustainability and resilience of coastal wetlands primarily depends on the balance between sediment accretion and sea level rise and availability of landward space for inland migration of coastal wetlands.

Both adaptation and mitigation capacities of coastal wetlands are measurable but require accounting for a wide range of parameters that vary locally. This variability creates uncertainties that could affect applicability of global data for local estimates. Uncertainties in data can be managed primarily by applying precautionary approach to all stages of the policy cycle. Also, the growing global knowledge (e.g. blue carbon methodologies) and incorporation of coastal wetlands into carbon market are opportunities that can drive local data generation.

In the context of mangrove removal, the precautionary principle implies that applicants to take the burden of proof and provide sufficient evidence for their proposals. The Auckland Council’s current policy position and rules is built upon this approach but to operationalise the principle would require a mechanism to enable inclusion of climate benefits of coastal wetlands in the consent process and to provide guidance for applicants to assess the amount of carbon loss due to mangrove removal and the means for compensation of the lost carbon.

Decisions on whether and how mangroves should be managed (i.e. removed or not, and if removed how the losses should be compensated, if significant) would need to be made within a broader strategic and regional framework that sets out regional priorities and outcomes, and establishes the relationships between policy instruments and tools (e.g. zoning rules and consent process, offset schemes, vulnerability assessment, insurance schemes etc.).

A strategic approach needs to link priorities to outcomes. The conceptual framework in Chapter 2 provides a basis to formulate regional priorities and outcomes in a way that helps achieving the triple-wins. Based
on this approach, catchment or regional coastal protection and biodiversity conservation targets can be identified and offset sites can be selected in areas that help achieving these targets.

The resilience approach and polycentric governance are prerequisites for achieving triple-wins outcomes. In the New Zealand context, where there is a division of responsibilities for mitigation and adaptation between central and local governments, focusing on ecosystem-based adaptation and resilience-oriented initiatives with potential mitigation co-benefits can enhance capacity of local governments to foster win-win outcomes.

Open communication and information sharing is a key enabler of resilience and adaptive planning in response to foreseeable climate impacts. Public awareness of the benefits of coastal wetlands can reduce resistance and gain public acceptance of the policy initiatives to maintain coastal wetlands.

While climate change threatens coastal wetlands, in the context of Auckland, it is a potential opportunity for conservation of coastal wetlands (particularly mangroves) if the climate change benefits are properly accounted for in the land use decisions through resource consenting process. This suggests that in the case of Auckland, internalising climate benefits into land use decisions can lead to conservation and enhancement of mangroves. However, in practice, it will be the priorities and policy goals that will likely drive conservation and enhancement, and hence contribute to climate compatible development.

9.3. Contribution and limitations

This research has focused on a relatively new area of inquiry and is the first of its kind both in New Zealand and globally. That is, the research attempted to address the problem of integrating climate change mitigation-adaptation values of coastal wetlands into land use decisions and planning for coastal resource management. To do so, the research looked at the problem through the lenses of science, policy and practice and tried to bring together the current theories, knowledge and practice (global experience) and applied that to explore the policy aspects of the research. The key contributions of the research are outlined below:

As discussed, incorporation of climate benefits of coastal wetlands into mainstream policy and planning process requires knowledge of the climate values of the wetlands. When this research started in 2012, the information about carbon sequestration and storage (CS&S) capacity of coastal wetlands was scarce in Auckland. This research combined the global data into a new database of the CS&S for temperate mangrove and saltmarsh species similar to those in Auckland. This data can contribute to enhancing our understanding of the approximate potential of Auckland’s mangrove and saltmarsh ecosystems and can also be applied to other regions with similar condition to Auckland. This data represents the best available information in the absence of local empirical data for individual wetlands. The research discussed and suggested means to address the uncertainties and limitations associated with the use of the data.

The research also provided the first comprehensive assessment of the parameters that affect the wave attenuation potential of mangrove and saltmarsh ecosystems. Findings of this assessment can inform the debates around the incomparability of the New Zealand’s mangrove species with its tropical counterparts.

In addition, the research identified the key parameters that can be used in New Zealand and Auckland (and globally) to determine the long-term sustainability of mangrove and saltmarsh ecosystems in relation to their resilience to the impacts of climate change. The decision tree developed in Chapter 8 builds on those parameters and knowledge and can be customised and used to identify areas where mangroves can provide long term protection against coastal hazards.

Moreover, the research identified the variables that affect the capacity of coastal wetlands for CS&S and coastal protection. As part of the process to identify ecological equivalence between impact and offset sites, these variables can be included in the offset accounting model as attributes for carbon storage and coastal protection and can also be used to facilitate comparison between the sites.
As part of the scoping process for this research, it was necessary to identify how conversions between various land use categories would affect carbon sequestration capacity of an urban landscape. This was identified in Chapter 4 and the results partly helped to narrow the focus of this thesis.

The comprehensive review of the current legislative and planning system in New Zealand and Auckland in Chapter 6 is the first review of its kind in New Zealand that inspected the system from this particular perspective. It identified the major challenges and opportunities in relation to the policy questions and focus of this research.

This research provided a policy and planning framework with a mix of recommended planning tools (outlined in Chapter 8) that can inform the decisions through various stages of planning for coastal wetlands and land use activities in Auckland. The proposed framework is built upon the theories discussed in Chapter 2 especially the concepts of co-benefits and resilience and informed by the findings of the comparative case studies in Chapter 7. The framework seeks to ensure that the coastal natural resources in the Auckland region will be properly protected and enhanced for their adaptation and mitigation benefits. The comparative analysis of the cases studied provides an overview of the global experiences that can also be used by other local governments in New Zealand.

The framework is intended to identify opportunities that exist to fill in the current policy, planning and implementation gaps as discussed in Chapter 6 (Section 6.5). This might be of particular interest for the central government in New Zealand through its on-going review of the policy and implementation guidelines for local authorities. It also provides options for local authorities (in this case the Auckland Council) to take into consideration when developing or reviewing their local policies and plans for their coastal environments.

Drawing on the theories and perspectives about the key concepts discussed in Chapter 2, the research has developed a conceptual framework that simplifies the relations between various dimensions of policy integration and the key concepts especially the concepts of win-win and triple-wins outcomes and trade-offs from the perspective of the Climate Compatible Development. This can be used as a framework to represent the complexity of the policy integration in a simple conceptualisation. It can be used to demonstrate the existing state and various possible future states of a policy system, illustrate how policy outcomes are influenced by the way conflicting goals are prioritised and hence guide the policy and planning process. In the context of New Zealand, this framework may be of particular interest for both central and local government, but mostly for central government to identify their policy priorities and provide guidance for local governments.

Notwithstanding the significance of the results, this research had a number of limitations as follows:

As discussed in Chapter 2, this research was formulated to address a wicked planning problem and designed to cover a broad range of complex and interrelated issues. Analysis and synthesis of this complexity has been a big challenge for this research that could undermine the coherence and consistency of the entire thesis. As a guiding framework, this research employed a set of key concepts and the rubric of Climate Compatible Development as discussed in Chapter 2 and applied a systematic approach to establish the connections between various stages of the research and to eventually bind the findings to come to cohesive conclusions.

Data was another major limitation for this research. It was not the intention in this research to generate empirical data. The available data was rather used for estimations. While the data used in this research to estimate the carbon sequestration and storage (CS&S) capacity of Auckland’s mangrove and saltmarsh ecosystems represent the best available, the variations in global data and the limited Auckland-specific empirical data created some issues in relation to reliability and applicability of this information for making local estimates. To reduce the uncertainty associated with using global data, this study only used data reported for coastal wetlands with similar conditions (climate and halophyte species) to coastal wetlands.
in Auckland, so the estimated values can be used as an approximate CS&S potential of mangrove and saltmarsh ecosystems in the Auckland region.

In addition, due to the lack of information about the aerial extent of seagrass ecosystems in the Auckland region, these ecosystems were excluded from the carbon estimates of this research. Thus, the estimated values do not represent the overall capacity of Auckland’s coastal wetlands for CS&S. A number of other methodological limitations are also discussed in Chapter 5.

In regard to the questionnaire survey, the number of people that could be selected as the participants for the survey (or the population of interest) was small. This primarily happened because the purpose of the survey required a selective sampling method as discussed in Chapter 3 and out of a limited number of the potential participants only 53% accepted to answer the questions. However, the survey was only used for the purpose of triangulation, and hence a large sample size was not required for this research. Limitations related to the questionnaire survey have not affected the outcomes of this research as the final conclusion and recommendations have not relied on the survey responses. However, given the acceptable response rate, comments and responses from the survey are used in this research to primarily support the discussions in Chapter 6.

9.4. Directions for further research

The subject of this research i.e. climate policy integration and many of the concepts, approaches and ideas discussed in this thesis are relatively recent and evolving. So, it was by no means surprising to come across some unknowns or emerging questions in such a progressive area of research. Through the various stages of the research, I came up with a number of new questions beyond the scope of this research. Some of these key questions are listed below as research topics that I believe can guide future research. They can be treated individually, but in my view, there will be more benefit if they are considered within a broader multi-disciplinary research programme. This is because of the nature of the problems in this area of research that requires knowledge from various disciplines to cross at some point, and putting them within a research programme can help to achieve this.

In this research, I approached the gap between theory and practice in this area of inquiry and proposed and used a conceptual framework with an attempt to fill in the gap. However, there remain opportunities to contribute to bridging the gap between science, theory and practice in this area. The conceptual framework I suggested can be a good start to expand on. Following are the key research topics:

- The trade-offs and co-benefits between ecological (biodiversity) conservation, carbon sequestration, coastal protection and development (whether and how quantifying these trade-offs and synergies can support land use decisions?);
- Addressing uncertainty in climate risks and its impacts on long-term sustainability of coastal wetlands in Auckland
- A methodology and proxy for (monetary) valuation of ecosystem services of coastal wetlands, in the paucity of local empirical data
- Valuation of the ecosystem services of Auckland’s coastal wetlands
- Evaluating storm protection potential of coastal wetlands in Auckland’s estuaries
- Blue carbon potential of Auckland’s coastal wetlands
- Understanding what and how ecosystem services are perceived and valued by the public, and various other actors and what influences the perceptions and values
- A methodology for assessing out of kind offset ecological equivalence (e.g. catchment plantation vs. coastal wetlands, and carbon storage vs. sequestration)
- Lifecycle assessment for carbon abatement and offsetting
- Carrying capacity and offsetting: how much wetland can be sustainability removed? Estimating acceptable level or limit of mangrove removal to achieve no net loss of mangrove ecosystems
• Comparative study of the relationship between governance and the choice of policy mixes and instruments for mainstreaming coastal wetlands into policies
• How does uncertainty unfold through various stages of the policy cycle, and how it can be identified and addressed?
• The link between priorities and outcomes: how priorities determine win-win or triple-wins outcomes and how they can be measured for monitoring policy performance?
• A framework to assess and monitor effectiveness of the blue carbon, ecosystem-based adaptation and related initiatives
Appendix 1: Review of literature on carbon sequestration by land use categories (Chapter 4)

The literature review process started with a review of peer-reviewed and grey literature in Auckland and New Zealand. The review included a keyword-based search of the ‘Springer Link’ and ‘Web of Knowledge’ using different combinations of search statements as identified in Table A1-1. Excluding duplicate entries, the search identified a total of 72 studies, which were then reviewed to extract the information about values of carbon sequestration. The individual review of documents started with review of abstracts followed by in-depth review of content to avoid unnecessary examination of the irrelevant materials. For the book chapters or big articles with no recorded abstract, the whole text was searched using the term ‘sequestration’. A combination of keywords (Auckland/New Zealand + carbon sequestration) were entered in the Google search engine to search the grey literature.

The review of peer-reviewed and grey literature found a few studies and estimates on the rate of carbon sequestration by land use categories in New Zealand. These included studies by Hollinger, et al., (1993); Huh, et al., (2008); and Carswell, et al., (2009). Unpublished data on the rate of carbon sequestration for a 69-year old Kauri plantation in the Taranaki Region was also reported through the website of the New Zealand’s Ministry for Primary Industries191. Carswell, et al., (2009) has also stated that limited information on the rate of carbon sequestration is available for broad classes of New Zealand vegetation.

Table A1-1. Keywords used to search ‘Web of Knowledge’ and ‘Springer Link’ for studies on carbon sequestration in New Zealand (1990 – August 2013)

<table>
<thead>
<tr>
<th>Subject</th>
<th>No.</th>
<th>Combination of search terms</th>
<th>Springer Link</th>
<th>Web of Knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>Studies on carbon sequestration in Auckland and New Zealand</td>
<td>1.</td>
<td>With the exact phrase: Auckland carbon sequestration</td>
<td>TOPIC: (Auckland) AND TITLE: (“carbon sequestration”)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>With the exact phrase: New Zealand carbon sequestration</td>
<td>TOPIC: (New Zealand) AND TITLE: (“carbon sequestration”)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>With the exact phrase: carbon sequestration; With at least one of the words: Auckland</td>
<td>TOPIC: (Auckland carbon sequestration) AND TITLE: (“carbon sequestration”)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>4.</td>
<td>With the exact phrase: New Zealand; With at least one of the words: sequestration</td>
<td>TOPIC: (New Zealand carbon sequestration) AND TITLE: (“carbon sequestration”)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>5.</td>
<td>With the exact phrase: carbon sequestration; Where the title contains: Auckland</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>6.</td>
<td>With the exact phrase: carbon sequestration; Where the title contains: New Zealand</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The global review constituted a systematic search of citations in the “Web of Knowledge” covering a period from 1990 to September 2013. The search looked for relevant studies on carbon sequestration rates within the six land use categories, using 29 different keyword combinations as listed in Table A1-2 and shown in Figure A1-1 The search returned 483 studies which were then reduced to 471 after duplicate papers were removed.

Of the identified studies, only those that reported quantitative values for carbon sequestration in their abstracts were included in the database of this research. There were two considerations for this approach to filtering: (i) to allow a more inclusive and relevant document retrieval, the combinations of search statements were consciously chosen to be least specific about values, and (ii) the filtering approach 191 The website has been changed since 2013 and the former information is not available any longer. However, the carbon sequestration rate for 69-year old Kauri plantation in Taranaki Region is also reported by Kimberley, et al., (2014).
assumed that where there were no carbon sequestration values in abstracts, the articles were more likely not focused on quantitative or comparative analysis of carbon sequestration values. This process filtered the studies down to 61 (Figure A1-1).

Table A1-2. Keywords used to search ‘Web of Knowledge’ for global literature on carbon sequestration (1990 – September 2013)

<table>
<thead>
<tr>
<th>Subject</th>
<th>No.</th>
<th>Combination of search terms (In Title)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Studies on carbon sequestration in urban trees</td>
<td>1.</td>
<td>Carbon + Sequestration + “Urban trees”</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>Carbon + “Urban forest”</td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>Carbon + Green space + Urban</td>
</tr>
<tr>
<td></td>
<td>4.</td>
<td>Carbon + Sequestration + “Urban forests”</td>
</tr>
<tr>
<td></td>
<td>5.</td>
<td>“Carbon sequestration” + Urban + Forests</td>
</tr>
<tr>
<td></td>
<td>6.</td>
<td>Carbon + Sequestration + Urban + Trees</td>
</tr>
<tr>
<td>Studies on carbon sequestration in natural forests</td>
<td>1.</td>
<td>“Carbon sequestration” + “Forest”</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>Carbon + Sequestration + Tree</td>
</tr>
<tr>
<td>Studies on carbon sequestration in plantation</td>
<td>1.</td>
<td>“Carbon sequestration” + Reforestation</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>Sequestration + Plantation</td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>“Carbon accumulation” + Plantation</td>
</tr>
<tr>
<td></td>
<td>4.</td>
<td>“Secondary forest” + “Carbon”</td>
</tr>
<tr>
<td></td>
<td>5.</td>
<td>“Carbon sequestration” + “Land use change”</td>
</tr>
<tr>
<td>Studies on carbon sequestration in wetlands</td>
<td>1.</td>
<td>“Carbon Sequestration” + Wetland</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>“Carbon balance” + Wetland</td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>Carbon + sequestration + wetland</td>
</tr>
<tr>
<td>Studies on carbon sequestration in agricultural lands</td>
<td>1.</td>
<td>“Carbon sequestration” + Cropland</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>“Carbon sequestration” + “Agricultural land”</td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>“Carbon sequestration” + “Farmland”</td>
</tr>
<tr>
<td></td>
<td>4.</td>
<td>“Carbon sequestration” + “Cropland”</td>
</tr>
<tr>
<td></td>
<td>5.</td>
<td>“Cropland” + Sequester + Carbon</td>
</tr>
<tr>
<td></td>
<td>6.</td>
<td>“Cropland” + Sequestration</td>
</tr>
<tr>
<td></td>
<td>7.</td>
<td>Carbon + Sequestration + Paddy soil</td>
</tr>
<tr>
<td></td>
<td>8.</td>
<td>“Carbon sequestration” + Rangeland</td>
</tr>
<tr>
<td></td>
<td>9.</td>
<td>“Carbon sequestration” + Pasture</td>
</tr>
<tr>
<td></td>
<td>10.</td>
<td>Sequestration + Pastoral</td>
</tr>
<tr>
<td>Studies on carbon sequestration in turfgrass/lawn</td>
<td>1.</td>
<td>“Carbon sequestration” + Grassland</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>“Carbon sequestration” + Lawn</td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>“Carbon sequestration” + Turfgrass</td>
</tr>
</tbody>
</table>
Full texts of the selected publications were then reviewed and in total 121 reported observations were selected and included in the dataset of this research. The dataset is provided in Table A1-3.

A number of studies applied measurement units that represent carbon sequestration in g C m$^{-2}$ yr$^{-1}$ or Kg C m$^{-2}$ yr$^{-1}$. These units were converted to standard units, i.e. Mg C ha$^{-1}$ yr$^{-1}$, by multiplying the quantities by $10^2$ and 10 respectively for each unit of measurement.
Table A1-3. Carbon sequestration rates by the selected land use categories

<table>
<thead>
<tr>
<th>No</th>
<th>Location</th>
<th>Climate</th>
<th>Vegetation type</th>
<th>Carbon Seq. t C ha(^{-1}) yr(^{-1})</th>
<th>Standard Error (SE)</th>
<th>Parameter measured to calculate carbon sequestration</th>
<th>Net Seq.</th>
<th>Method</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Biomass</td>
<td>Soil</td>
<td>Sediment accretion</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Carbon</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>USA, Jersey City</td>
<td>A</td>
<td>Urban trees</td>
<td>1.83</td>
<td>0.34</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td>Nowak et al., 2013</td>
</tr>
<tr>
<td>2</td>
<td>USA, Sacramento city</td>
<td>C</td>
<td>Urban trees</td>
<td>3.77</td>
<td>0.64</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>USA, Atlanta City</td>
<td>C</td>
<td>Urban trees</td>
<td>2.29</td>
<td>0.17</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>USA, Syracuse city</td>
<td>C</td>
<td>Urban trees</td>
<td>2.85</td>
<td>0.3</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>USA, Baltimore City</td>
<td>C</td>
<td>Urban trees</td>
<td>2.82</td>
<td>0.36</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>USA, Lincoln City</td>
<td>D</td>
<td>Urban trees</td>
<td>4.09</td>
<td>0.63</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>Korea, Chuncheon City</td>
<td>D</td>
<td>Urban trees</td>
<td>0.56</td>
<td>0.08</td>
<td>√ (AG/BG)</td>
<td></td>
<td></td>
<td>Jo, 2002</td>
</tr>
<tr>
<td>8</td>
<td>Korea, Kangleung City</td>
<td>D</td>
<td>Urban trees</td>
<td>0.71</td>
<td>0.1</td>
<td>√ (AG/BG)</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>9</td>
<td>Korea, Kangnam City</td>
<td>D</td>
<td>Urban trees</td>
<td>0.53</td>
<td>0.06</td>
<td>√ (AG/BG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>Korea, Junglang City</td>
<td>D</td>
<td>Urban trees</td>
<td>0.8</td>
<td>0.12</td>
<td>√ (AG/BG)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>11</td>
<td>China, Shenyang City</td>
<td>D</td>
<td>Urban forest</td>
<td>2.84</td>
<td>0.58</td>
<td>√ (AG/BG)</td>
<td></td>
<td></td>
<td>Liu &amp; Li, 2012</td>
</tr>
<tr>
<td>12</td>
<td>China, Beijing</td>
<td>D</td>
<td>Urban forest</td>
<td>2.22</td>
<td>0.25</td>
<td>√ (AG/BG)</td>
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<td></td>
<td>Yang et al., 2005; Cited from Liu &amp; Li, 2012</td>
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<tr>
<td>13</td>
<td>China, Hangzhou</td>
<td>C</td>
<td>Urban forests</td>
<td>1.66</td>
<td>N/I</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td>Zhao et al., 2010</td>
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<tr>
<td>14</td>
<td>USA, Auburn City</td>
<td>D</td>
<td>Protected trees in arboretum</td>
<td>1.7</td>
<td>N/I</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td>Biomass &amp; growth equations</td>
</tr>
<tr>
<td>15</td>
<td>USA, Auburn City</td>
<td>D</td>
<td>Intensively maintained trees</td>
<td>0.3</td>
<td>N/I</td>
<td>√ (AG)</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Natural forest

<table>
<thead>
<tr>
<th>No</th>
<th>Location</th>
<th>Climate</th>
<th>Vegetation type</th>
<th>Carbon Seq. t C ha(^{-1}) yr(^{-1})</th>
<th>Standard Error (SE)</th>
<th>Parameter measured to calculate carbon sequestration</th>
<th>Net Seq.</th>
<th>Method</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
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<td></td>
<td></td>
<td>Biomass</td>
<td>Soil</td>
<td>Sediment accretion</td>
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<td></td>
<td></td>
<td></td>
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</tr>
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<td>C</td>
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<td>Canada, Saskatchewan</td>
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<td>Old aspen forest (1994)</td>
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Climate classes according to the main groups of Köppen climate classification are: A: Tropical/megathermal climates, B: Dry (arid and semi-arid) climates, C: Temperate/mesothermal climates, D: Continental/microthermal climates, E: Polar and alpine climates (McKnight and Hess, 2000)
<table>
<thead>
<tr>
<th></th>
<th>Location</th>
<th>Region</th>
<th>Forest Type</th>
<th>Growth / Mortality / Harvest</th>
<th>NEE / NEP</th>
<th>Model Simulation</th>
<th>Biomass &amp; Growth Equations</th>
<th>CO2 Flux Measurement</th>
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<td>(Eddy covariance)</td>
<td>SOC (SOC)</td>
<td>CO2 flux measurement</td>
<td>Gielen et al., 2013</td>
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<td>NEP/harvest</td>
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<td>Vetter et al., 2005</td>
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<td>D</td>
<td>Mixed forest</td>
<td>(AG/BG)</td>
<td>√ (AG/BG)</td>
<td>Growth / mortality / harvest</td>
<td>Dai et al., 2013</td>
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<td>√ (AG/BG)</td>
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<td>Yu et al., 2011</td>
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<td>Growth / mortality / harvest</td>
<td>Sang et al., 2002, Cited from Yu et al., 2011</td>
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<td>Hollinger et al., 1993, Cited from Tate et al., 1997</td>
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<td>Tree Species</td>
<td>Biomass &amp; Growth Equations</td>
<td>Soil Sampling</td>
<td>Other Notes</td>
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<td>Joshi et al., 2013</td>
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* Type of biomass is not specified.
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<tr>
<th>No.</th>
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<th>N/I</th>
<th>Methodology</th>
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<td>29.</td>
<td>China, Yuanmou</td>
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<td>Ireland</td>
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<td>Biomass equations &amp; respiration models Black et al., 2007</td>
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<td><strong>Soil sampling</strong></td>
<td>Tang &amp; Li, 2013</td>
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**Wetland**

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### Agricultural land

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### Urban turfgrass/lawn

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<td>N/I</td>
<td></td>
<td>Soil sampling</td>
</tr>
<tr>
<td>6.</td>
<td>USA, Denver, Fort Collins</td>
<td>Turfgrass (Golf course)</td>
<td>0.9</td>
<td>N/I</td>
<td></td>
<td>Soil sampling</td>
</tr>
<tr>
<td></td>
<td>USA, Irvine</td>
<td>C</td>
<td>Turfgrass (Park)</td>
<td>1.4</td>
<td>N/I</td>
<td>SOC</td>
</tr>
<tr>
<td>---</td>
<td>------------</td>
<td>---</td>
<td>------------------</td>
<td>-----</td>
<td>-----</td>
<td>-----</td>
</tr>
</tbody>
</table>
Appendix 2: Review of literature on carbon sequestration and storage (CS&S) by coastal wetlands (Chapter 5)

- Mangrove and saltmarsh species in Auckland, New Zealand

Unlike tropical mangrove forests, temperate mangrove forests in New Zealand are composed of mono-specific stands of *Avicennia marina subsp. Australasica* which is locally called ‘Manawa’. It is one of several taxa within the genus *Avicennia* that occurs in both Northern and Southern Hemispheres (Morrisey, et al., 2007). However, the subspecies within which Manawa is included is confined to New Zealand, South-Eastern Australia and Lord Howe Island (New South Wales, Australia) (Morrisey, et al., 2007). *Avicennia marina subsp. Australasica* is also called *Avicennia resinifera* or *Avicennia marina var. resinifera* (New Zealand Plant Conservation Network, 2014).

Dominant species of saltmarsh wetlands in Auckland include sea rush (*Juncus krausii var. australiensis*), Oioi (*Apodasmia similis*), glasswort (*Sacrocornia quinqueflora*), and saltmarsh ribbon wood (*Plagianthus divaricatus*). Among other saltmarsh species found in Auckland are cordgrass (*Spartina spp.*) and arrow grass (*Triglochin palustris*). Cordgrass is a non-native plant that was introduced to New Zealand in 1950s to protect estuarine tidal flats through sediment stabilization (Dodgshun, et al., 2007). Cordgrass species in Auckland include *Spartina anglica*, *Spartina alterniflora* and *Spartina townsendii* that are listed as pest species.

- Review for local data

There is a considerable amount of research that studied mangrove forests, saltmarsh wetlands and seagrass ecosystems in New Zealand and Auckland (e.g. Ward & Lambie, 1999; Turner, et al., 1999; van Houte-Howes, 2004; Ellis, et al., 2004; Turner & Schwarz, 2006; Alfaro, 2006; Halliday, et al., 2006; Lovelock, et al., 2007; Morrisey, et al., 2007; Townsend, et al., 2008; Lovelock, 2008; Townsend, et al., 2010; Morrisey, et al., 2010; Bulmer, et al., 2012; Murray, 2013). However, these studies have mainly focused on biology and management of mangrove, saltmarsh and seagrass ecosystems and do not provide data for carbon sequestration and storage (CS&S).

The most comprehensive studies about the New Zealand mangrove were conducted by Morrisey, et al., in (2007) and (2010). These studies provided a wide range of information about mangrove ecology, historic loss and recent expansion of mangroves, the role of mangroves in estuaries and marine environment and mangrove management initiatives in New Zealand. The 2007 report also provided some estimates of primary production by the New Zealand mangrove, as well as a number of saltmarsh and seagrass ecosystems in Australia, USA and Mexico. However, given the difference between primary production and carbon sequestration (Pace & Lovett, 2012), this information was not used to estimate CS&S potential of mangrove forests in New Zealand.

To identify empirical studies on carbon sequestration and storage (CS&S) by mangrove and saltmarsh species in Auckland, a review of the peer-reviewed literature using different keyword combinations in the ‘Web of Science’ and ‘Google Scholar’ (Table A2-1) was conducted during the research period (Dec 2014 - Dec 2015).

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In total, 272 studies (excluding duplicates) were found as the result of this search and were reviewed for the estimates of carbon sequestration rates within the selected species of mangrove and saltmarsh habitats. The review found no studies in New Zealand reporting the rate of ‘carbon sequestration’ for mangrove and saltmarsh species. Only two studies by Yang, et al., (2013) and Tran (2014) reported quantitative estimates of ‘carbon storage’ in the sediment and biomass (above- and below-ground) of *Avicennia marina* in New Zealand. There was, however, no reported data of carbon storage by saltmarsh species in New Zealand.

The review also found the most recent and comprehensive global database of carbon accumulation rates (CAR)\(^{198}\) in the sediment of saltmarsh ecosystems provided by Ouyang and Lee, (2014). They compiled data from 50 studies between 1983 and 2013 and provided a dataset consisting 143 observations across the Southern and Northern hemispheres. Their database includes no study in New Zealand, but includes 36 observations of sediment carbon density and CAR for a number of saltmarsh species similar to those found in New Zealand (i.e. *Juncus Kraussii, Sarcocornia quinqueflora, Spartina alterniflora* and *Spartina anglica*).

A comprehensive global dataset known as ‘the Blue Carbon Data Set’\(^{199}\) includes CS&S data for coastal wetlands from studies published between 1988 and 2011. However, no estimate of CS&S by *Avicennia marina subsp. Australasica* is included in the dataset. All estimates reported for CS&S in saltmarsh species in the blue carbon dataset are also included in the dataset provided by Ouyang and Lee, (2014).

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\(^{198}\) Carbon accumulation rates (CAR) in sediment, is also referred to as carbon sequestration (Ouyang & Lee, 2014).

Table A2-1. Keywords used to search ‘Web of Science’ and ‘Google Scholar’ for studies on CS&S by mangrove and saltmarsh ecosystems in New Zealand (1990 to 2015, Search period: December 2014 to December 2015)

<table>
<thead>
<tr>
<th>Subject</th>
<th>Database Search statement</th>
</tr>
</thead>
</table>
| Studies on CS&S of Avicennia marina subsp australasica | Web of Science: TOPIC:(Avicennia marina) AND TOPIC:(New Zealand) AND TITLE: (carbon)  
Google Scholar: with all of the words: (carbon) (New Zealand); with the exact phrase: (Avicennia marina subsp australasica) with all of the words: (New Zealand); with the exact phrase: (Avicennia marina subsp australasica) with all of the words: (carbon sequestration) (New Zealand); with the exact phrase: (Avicennia marina var resinifera) |
| Studies on CS&S of saltmarsh species found in Auckland | Web of Science: TOPIC:(Juncus krausii var. australiensis) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Juncus krausii var. australiensis) AND TOPIC: (New Zealand)  
TOPIC:(Apodasmia similis) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Plagianthus divaricatus) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Triglochin palustris) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Spartina anglica) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Spartina alterniflora) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Spartina townsendii) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
TOPIC:(Sarcocornia quinqueflora) AND TOPIC: (New Zealand) AND TITLE: (carbon)  
with all of the words: (carbon) (New Zealand); with the exact phrase: (Juncus krausii var. australiensis)  
with all of the words: (carbon) (New Zealand); with the exact phrase: (Juncus krausii)  
with all of the words: (New Zealand); with the exact phrase: (Plagianthus divaricatus)  
with all of the words: (New Zealand); with the exact phrase: (Triglochin palustris)  
with all of the words: (New Zealand); with the exact phrase: (Spartina anglica); with the exact phrase: (New Zealand)  
with all of the words: (“carbon sequestration”) (Spartina anglica); with the exact phrase: (New Zealand)  
with all of the words: (“carbon stock”) (Spartina anglica); with the exact phrase: (New Zealand)  
with all of the words: (“carbon storage”) (Spartina anglica); with the exact phrase: (New Zealand)  
with all of the words: (“carbon sequestration”) (Spartina townsendii); with the exact phrase: (New Zealand)  
with all of the words: (“carbon stock”) (Spartina townsendii); with the exact phrase: (New Zealand)  
with all of the words: (“carbon storage”) (Spartina townsendii); with the exact phrase: (New Zealand)  
with all of the words: (“carbon sequestration”) (Spartina quinqueflora); with the exact phrase: (New Zealand)  
with all of the words: (“carbon stock”) (Spartina quinqueflora); with the exact phrase: (New Zealand)  
with all of the words: (“carbon storage”) (Spartina quinqueflora); with the exact phrase: (New Zealand) |
Review for similar data from global literature

In the absence of sufficient local data, an alternative approach applied to this research was to use data from the global literature. To use the global data for local estimates, it is necessary to identify and use the data on CS&S from coastal wetlands with similar conditions to coastal wetlands in Auckland. In doing so, this research first reviewed the literature outlining the factors that affect CS&S in coastal wetlands. The factors were then used to identify and select the global cases similar to Auckland. Once the global case studies were identified, an average value of data from these studies was used to estimate the approximate potential of CS&S by mangrove and saltmarsh habitats in Auckland. Since CS&S are influenced by those factors, similarity in the factors between the case studies and the Auckland region were considered to be indicative of the similarity in CS&S.

The findings of the literature review regarding the factors affecting CS&S in mangrove and saltmarsh ecosystems are briefly outlined below. Detailed information is provided in Chapter 5, Section 5.2.2.

In total, studies have mainly accounted for the parameters that affect CS&S in sediments of coastal wetlands. The studies suggest that sediment CS&S in coastal wetlands are a function of soil bulk density and carbon content of the soil; however, carbon sequestration in sediment is also driven by sediment accretion rate (SAR) (Chmura, et al., 2003; Ouyang & Lee, 2014).

Limited information is available on sediment accretion rate (SAR) across mangrove and saltmarsh habitats in New Zealand and Auckland. A few recent studies suggest that SAR is highly variable across and within coastal ecosystems. In the Firth of Thames (52 km southeast of Auckland), SARs between 7–12 mm yr\(^{-1}\) and 33-100 mm yr\(^{-1}\) have been reported (Swales, et al., 2015). The SAR for Auckland estuaries is reported to be between 3 and 5 mm yr\(^{-1}\) (Swales & Gibbs, 2014). Sediment accretion rate of 14 mm yr\(^{-1}\) is reported for an intact mangrove site in New Zealand (Stokes, et al., 2010, Cited in Krauss, et al., 2014). Young and Harvey (1996) also reported SARs between 5.5 and 6.4 mm yr\(^{-1}\) in the Hauraki Plains (75 km south-east of Auckland). Variations in surface elevation and sediment accretion among estuaries is argued to be influenced by multiple drivers such as sediment input, tidal regime, vegetation type and root density (Stokes, et al., 2009).

As concluded by the global literature (Chmura, et al., 2003; Loomis & Craft, 2010; McLeod, et al., 2011; Saintilan, et al., 2013; Lovelock, et al., 2014; Ouyang & Lee, 2014), climactic factors (temperature, rainfall and evaporation), plant species, tidal range and position of vegetation in the tidal frame can affect sediment CS&S in coastal wetlands by influencing soil bulk density, carbon content and SAR. Amongst these parameters, information about climate, plant species and tidal range is often reported within the studies. However, due to the lack of information about tidal range for Auckland’s mangrove and saltmarsh habitats, only climate and plant species were considered to filter the global studies on CS&S in mangrove and saltmarsh ecosystems.

According to the Köppen climate classification system, Auckland’s climate is classified in the Cfb climate category (Garr & Fitzharris, 1991). This category is identified as a ‘temperate oceanic climate’ with no distinct dry season. The summers are cool and the average temperature of the warmest month does not exceed 22°C. Morrissey, et al., (2010) have identified Cfb and Cfa\(^{200}\) (humid subtropical climate) as the most relevant climate types for studies on temperate mangroves.

Searching of the global literature started with using different keyword combinations in the ‘Web of Science’ and ‘Google Scholar’ (Table A2-2). This search identified 657 studies. Abstracts of these

\(^{200}\) Cfa refers to ‘humid subtropical climate’ and is characterised by hot, usually humid summers and mild to cold winters (Belda, et al., 2014)
studies were then reviewed for quantitative estimates of CS&S in sediment (soil) or biomass (above- and below-ground) of mangroves and saltmarshes. Studies were only selected if they had reported CS&S data for mangrove and saltmarsh species similar to those in Auckland and in Cfb climate. However, data from studies in Cfa climate were only used if no or limited data (one or two) was reported for a given mangrove and saltmarsh species in Cfb climate. A number of the selected studies did not specify the subspecies of *Avicennia marina*. Given the geographical distribution of this mangrove species (as mentioned in Section 3.3.1.2), only data from studies in South East Australia were considered for further review.

Table A2-2. Keywords used to search ‘Web of Science’ and ‘Google Scholar’ for global studies on CS&S by the selected species of mangroves and saltmarshes (1990 to 2015, Search period: December 2014 to December 2015)

<table>
<thead>
<tr>
<th>Subject</th>
<th>Database</th>
<th>Search statement</th>
</tr>
</thead>
<tbody>
<tr>
<td>Studies on CS&amp;S of <em>Avicennia marina subsp australasica</em></td>
<td>Web of Science</td>
<td>TITLE: (Mangrove) AND TOPIC: (Carbon) AND TOPIC: (Australia)</td>
</tr>
<tr>
<td></td>
<td>Google Scholar</td>
<td>with all of the words: (Carbon); with the exact phrase: (<em>Avicennia marina subsp australasica</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (Carbon); with the exact phrase: (<em>Avicennia marina subsp australasica</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (Carbon); with the exact phrase: (<em>Avicennia resinifera</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (Carbon) (sequestration) (accumulation); with the exact phrase: (<em>Avicennia marina var. resinifera</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (Australia) (mangrove) (Carbon) (sequestration) (accumulation) (store); with the exact phrase: (<em>Avicennia marina</em>)</td>
</tr>
<tr>
<td>Studies on CS&amp;S of saltmarsh species similar to those in Auckland</td>
<td>Web of Science</td>
<td>TOPIC: (<em>Juncus kraussii</em>) AND TOPIC: (carbon)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TOPIC: (<em>Sarcocornia quinqueflora</em>) AND TOPIC: (carbon)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TOPIC: (<em>Apodasmia similis</em>) AND TOPIC: (carbon)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TOPIC: (<em>Plagianthus divaricatus</em>) AND TOPIC: (carbon)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TOPIC: (<em>Triglochin palustris</em>) AND TOPIC: (carbon)</td>
</tr>
<tr>
<td></td>
<td>Google Scholar</td>
<td>with all of the words: (carbon) (sequestration) (storage); with the exact phrase: (<em>Juncus kraussii</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (carbon) (sequestration) (storage); with the exact phrase: (<em>Sarcocornia quinqueflora</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (carbon) (sequestration) (storage); with the exact phrase: (<em>Apodasmia similis</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (carbon) (sequestration) (storage); with the exact phrase: (<em>Plagianthus divaricatus</em>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with all of the words: (carbon) (sequestration) (storage); with the exact phrase: (<em>Triglochin palustris</em>)</td>
</tr>
</tbody>
</table>

The screening process, after duplicates were removed, resulted in 68 observations, of which 67 were reported for sediment CS&S by temperate *Avicennia marina*, *Spartina alterniflora*, *Spartina anglica*, *Sarcocornia quinqueflora* and *Juncus kraussii* (Table A2-3) and one on biomass carbon storage of *Avicennia marina subsp. Australasica*. None of the reviewed studies reported quantitative estimates of CS&S by saltmarsh species including *Apodasmia similis*, *Plagianthus divaricatus* and *Triglochin palustris* in temperate climates. Sediment carbon density and accumulation rates for *Spartina alterniflora* and *Spartina anglica* saltmarsh species were derived from the dataset provided by Ouyang and Lee (2014).

R-5
Table A2-3. Summary of the results of the literature review about studies on the CS&S of temperate mangrove and saltmarsh habitats

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Number per climate</th>
<th>Location</th>
<th>Number per climate</th>
<th>Location</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Avicennia marina</em></td>
<td>15 (6 Cfa/ 9 Cfb)</td>
<td>Australia, New Zealand</td>
<td>5 (4 Cfa/ 1 Cfa-Cfb)</td>
<td>Australia</td>
</tr>
<tr>
<td>Saltmarshes</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Total</em>:</td>
<td>24 (19 Cfa/ 5 Cfb):</td>
<td>Australia, USA, Europe</td>
<td>23 (17 Cfa/ 5 Cfb/ 1</td>
<td>Australia, USA, Europe</td>
</tr>
<tr>
<td><em>Spartina alterniflora</em></td>
<td>14 (Cfa)</td>
<td>USA</td>
<td>14 (Cfa)</td>
<td>USA</td>
</tr>
<tr>
<td><em>Spartina anglica</em></td>
<td>4 (Cfb)</td>
<td>Europe</td>
<td>4 (Cfb)</td>
<td>Europe</td>
</tr>
<tr>
<td><em>Sarcocornia quinqueflora</em></td>
<td>4 (3 Cfa/ 1 Cfb)</td>
<td>Australia</td>
<td>4 (3 Cfa/ 1 Cfb)</td>
<td>South east Australia</td>
</tr>
<tr>
<td><em>Juncus kraussii</em></td>
<td>2 (Cfa)</td>
<td>Australia</td>
<td>1 (Cfa-Cfb)</td>
<td>South east Australia</td>
</tr>
</tbody>
</table>

Given the very limited data for biomass carbon storage, only estimates on ‘sediment’ CS&S in temperate mangrove and saltmarsh species were used in this research. The dataset including 67 data (as explained below) is presented in Tables A2-4 and A2-5:

- 15 estimates of sediment carbon storage by temperate *Avicennia marina* in south east Australia and New Zealand with Cfa and Cfb climates (Table A2-4). However, only four estimates were reported for *Avicennia marina subsp. Australasica* in New Zealand. Nine out of 15 estimates were reported for *Avicennia marina* in Cfb climate and therefore were used in this research to estimate an approximate carbon storage potential for mangrove forests in the Auckland region.

- 24 observations on sediment carbon storage for *Spartina alterniflora, Spartina anglica, Sarcocornia quinqueflora* and *Juncus kraussii* in USA (Cfa climate), Europe (Cfb climate) and South East Australia (Cfa – Cfb climate) (Table A2-4). Given the lack of carbon storage data for *Spartina alterniflora* and *Juncus kraussii* in Cfb climate and insufficient similar data for *Sarcocornia quinqueflora*, carbon storage values reported for these saltmarsh species in Cfa climates were used in estimating the average carbon storage values.

- 5 estimates on sediment carbon sequestration rates for *Avicennia marina* in South East Australia (Cfa - Cfb climate) (Table A2-5). Since no data of carbon sequestration by *Avicennia marina* in Cfb climate was found, the data from areas with Cfa and Cfa-Cfb climate were used in this research.

- 23 estimates on sediment carbon sequestration rates for *Spartina alterniflora, Spartina anglica, Sarcocornia quinqueflora* and *Juncus kraussii* in USA (Cfa climate), Europe (Cfb climate) and South East Australia (Cfa – Cfb climate) (Table A2-5). Given the lack of carbon sequestration data for *Spartina alterniflora* in Cfb climate and insufficient similar data for *Sarcocornia quinqueflora*, carbon sequestration values reported for these saltmarsh species in Cfa climates were used in estimating the average carbon storage values.
Similar to the study by Lawrence, et al., (2012) and following IPCC protocol for tracking changes in carbon stocks (IPCC, 2007a), the amount of carbon is expressed in terms of potential CO₂ equivalent, obtained by multiplying carbon stocks by 3.67, which is the molecular weight ratio of CO₂ to Carbon. Most studies, particularly those related to mangroves, reported carbon storage (stock) data for the top metre of sediment. In order to ensure consistency between mangrove and saltmarsh carbon data and following the approach by Murray, et al., (2011), Pendleton, et al., (2012) and Nicholas Institute for Environmental Policy Solutions201, carbon stock values are provided for the top metre of the sediment. For data reported for sampling points less than 1 metre depth, carbon density is assumed to be uniform to a depth of 1 metre (Howe, et al., 2008).

Table A2-4. Carbon stock in sediment of mangrove (*Avicennia marina*) and saltmarshes (*Spartina alterniflora*, *Spartina anglica*, *Sarcocornia quinqueflora*, *Juncus kraussii*) across temperate climate (Cfa/Cfb)

<table>
<thead>
<tr>
<th>Specific Location</th>
<th>General Location</th>
<th>Dominant Species</th>
<th>Climate</th>
<th>Tidal range (m)</th>
<th>Soil depth (cm)</th>
<th>Organic/Inorganic Carbon</th>
<th>Sediment carbon stock in top meter (Mean) (tCO$_2$e ha$^{-1}$)</th>
<th>Method</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. MarraMarra Creek, Hawkesbury River, Sydney</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>Max 2</td>
<td>0-100</td>
<td>ns</td>
<td>1258</td>
<td>S, δ$^{13}$</td>
<td>Saintilan et al. 2013$^a$</td>
</tr>
<tr>
<td>2. Berowra Creek, Hawkesbury River, Sydney</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>Max 2</td>
<td>0-100</td>
<td>ns</td>
<td>1045</td>
<td>S, δ$^{13}$</td>
<td>Saintilan et al. 2013$^a$</td>
</tr>
<tr>
<td>3. Kooragang Island, Hunter River estuary, Newcastle</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>Max 2</td>
<td>0-100</td>
<td>ns</td>
<td>957</td>
<td>S, δ$^{13}$</td>
<td>Saintilan et al. 2013$^a$</td>
</tr>
<tr>
<td>4. Kooragang Island, Hunter River estuary, Newcastle</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>Max 1.75</td>
<td>100</td>
<td>Organic</td>
<td>1588</td>
<td>S</td>
<td>Howe et al. 2008$^b$</td>
</tr>
<tr>
<td>5. Kooragang Island, Hunter River (Site1), Newcastle</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>1.9</td>
<td>100</td>
<td>Organic</td>
<td>1038</td>
<td>S</td>
<td>Howe et al. 2009$^b$</td>
</tr>
<tr>
<td>6. Kooragang Island, Hunter River (Site2), Newcastle</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em> (Forsk.)</td>
<td>Cfa</td>
<td>1.9</td>
<td>100</td>
<td>Organic</td>
<td>1701</td>
<td>S</td>
<td>Howe et al. 2009$^b$</td>
</tr>
<tr>
<td>7. Currambene Creek, Jervis Bay, Wollamia</td>
<td>SE Australia</td>
<td><em>Avicennia marina var. australisca</em></td>
<td>Cfb</td>
<td>Microtidal</td>
<td>0-100</td>
<td>ns</td>
<td>92</td>
<td>S, δ$^{13}$</td>
<td>Saintilan et al. 2013$^a$</td>
</tr>
<tr>
<td>8. Cararma Inlet, Jervis Bay, Wollamia</td>
<td>SE Australia</td>
<td><em>Avicennia marina var. australisca</em></td>
<td>Cfb</td>
<td>Microtidal</td>
<td>0-100</td>
<td>ns</td>
<td>884</td>
<td>S, δ$^{13}$</td>
<td>Saintilan et al. 2013$^a$</td>
</tr>
<tr>
<td>9. Stony Point, Westernport Bay, Victoria</td>
<td>SE Australia</td>
<td><em>Avicennia marina var. australisca</em></td>
<td>Cfb</td>
<td>2.15$^c$</td>
<td>0-100</td>
<td>ns</td>
<td>532</td>
<td>S</td>
<td>Livesley and Andrusiak, 2012$^b$</td>
</tr>
<tr>
<td>10. Jacks Beach, Westernport Bay, Victoria</td>
<td>SE Australia</td>
<td><em>Avicennia marina var. australisca</em></td>
<td>Cfb</td>
<td>2.15$^c$</td>
<td>0-100</td>
<td>ns</td>
<td>542</td>
<td>S</td>
<td>Livesley and Andrusiak, 2012$^b$</td>
</tr>
<tr>
<td>11. Yaringa Marine National Park, Victoria</td>
<td>SE Australia</td>
<td><em>Avicennia marina var. australisca</em></td>
<td>Cfb</td>
<td>2.15$^c$</td>
<td>0-100</td>
<td>ns</td>
<td>520</td>
<td>S</td>
<td>Livesley and Andrusiak, 2012$^b$</td>
</tr>
<tr>
<td>12. Waitemata Harbour, Auckland, Core 1</td>
<td>New Zealand, North Island</td>
<td><em>Avicennia marina var. resinifera</em></td>
<td>Cfb</td>
<td>2</td>
<td>0-76</td>
<td>Organic</td>
<td>307</td>
<td>S, δ$^{13}$</td>
<td>Yang et al. 2013$^d$</td>
</tr>
<tr>
<td>13. Waitemata Harbour, Auckland, Core 2</td>
<td>New Zealand, North Island</td>
<td><em>Avicennia marina var. resinifera</em></td>
<td>Cfb</td>
<td>2</td>
<td>0-109</td>
<td>Organic</td>
<td>360</td>
<td>S, δ$^{13}$</td>
<td>Yang et al. 2013$^d$</td>
</tr>
<tr>
<td>14. Waitemata Harbour, Auckland, Core 3</td>
<td>New Zealand, North Island</td>
<td><em>Avicennia marina var. resinifera</em></td>
<td>Cfb</td>
<td>2</td>
<td>0-105</td>
<td>Organic</td>
<td>539</td>
<td>S, δ$^{13}$</td>
<td>Yang et al. 2013$^d$</td>
</tr>
<tr>
<td>15. Waitemata Harbour, Auckland, Core 4</td>
<td>New Zealand, North Island</td>
<td><em>Avicennia marina var. resinifera</em></td>
<td>Cfb</td>
<td>2</td>
<td>0-93</td>
<td>Organic</td>
<td>307</td>
<td>S, δ$^{13}$</td>
<td>Yang et al. 2013$^d$</td>
</tr>
<tr>
<td>Specific Location</td>
<td>General Location</td>
<td>Dominant Species</td>
<td>Climate</td>
<td>Tidal range (m)</td>
<td>Sediment carbon density (g C cm(^{-3}))</td>
<td>Soil depth (cm)</td>
<td>Organic/Inorganic Carbon</td>
<td>Sediment Stock in top meter (Mean) (tCO(_2)e ha(^{-1}))</td>
<td>Method</td>
</tr>
<tr>
<td>-------------------------------------------------------</td>
<td>------------------</td>
<td>------------------</td>
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<td>---------------------------------------------</td>
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<td>--------------------------------------------------</td>
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</tr>
<tr>
<td>16. Aransas, Texas</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.11</td>
<td>0.040</td>
<td>0-40</td>
<td>Organic</td>
<td>1467 S</td>
<td></td>
</tr>
<tr>
<td>17. San Bernard, Texas</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.22</td>
<td>0.033</td>
<td>0-40</td>
<td>Organic</td>
<td>1210 S</td>
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</tr>
<tr>
<td>18. Biloxi Bay, Mississippi</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.47</td>
<td>0.027</td>
<td>0-50</td>
<td>Organic</td>
<td>990 S</td>
<td></td>
</tr>
<tr>
<td>19. Old Oyster Bayou, Louisiana</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.31</td>
<td>0.019</td>
<td>nd</td>
<td>Organic</td>
<td>697 nd</td>
<td></td>
</tr>
<tr>
<td>20. Bayou Chitigue, Louisiana</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.31</td>
<td>0.016</td>
<td>0-30</td>
<td>Organic</td>
<td>587 S</td>
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<tr>
<td>21. Old Oyster Bayou, Louisiana</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.31</td>
<td>0.021</td>
<td>0-30</td>
<td>Organic</td>
<td>770 S</td>
<td></td>
</tr>
<tr>
<td>22. Bayou Chitigue, Louisiana</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.31</td>
<td>0.019</td>
<td>0-30</td>
<td>Organic</td>
<td>697 S</td>
<td></td>
</tr>
<tr>
<td>23. Lafourche Parish, Louisiana</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.31</td>
<td>0.019</td>
<td>0-2</td>
<td>Organic</td>
<td>697 S</td>
<td></td>
</tr>
<tr>
<td>24. Ogeechee River, Georgia Coast</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>2.09</td>
<td>0.019</td>
<td>0-30</td>
<td>Organic</td>
<td>697 S</td>
<td></td>
</tr>
<tr>
<td>25. Altamaha River, Georgia Coast</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>2.19</td>
<td>0.022</td>
<td>0-30</td>
<td>Organic</td>
<td>807 S</td>
<td></td>
</tr>
<tr>
<td>26. Satilla River, Georgia Coast</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>2.10</td>
<td>0.021</td>
<td>0-30</td>
<td>Organic</td>
<td>770 S</td>
<td></td>
</tr>
<tr>
<td>27. Oregon Inlet, North Carolina</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.61</td>
<td>0.022</td>
<td>0-30</td>
<td>Organic</td>
<td>807 S</td>
<td></td>
</tr>
<tr>
<td>28. Oregon Inlet, North Carolina</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.61</td>
<td>0.023</td>
<td>0-30</td>
<td>Organic</td>
<td>843 S</td>
<td></td>
</tr>
<tr>
<td>29. SA4, Little Assawoman Bay, Delaware</td>
<td>USA</td>
<td>Spartina alterniflora</td>
<td>Cfa</td>
<td>0.47</td>
<td>0.062</td>
<td>0-30</td>
<td>Organic</td>
<td>2273 S</td>
<td></td>
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<tr>
<td>30. St. Annaland, Netherlands</td>
<td>Europe</td>
<td>Spartina anglica</td>
<td>Cfb</td>
<td>4.48</td>
<td>0.041</td>
<td>0-50</td>
<td>Organic</td>
<td>1503 S</td>
<td></td>
</tr>
<tr>
<td>31. Scheldt, Netherlands</td>
<td>Europe</td>
<td>Spartina anglica</td>
<td>Cfb</td>
<td>4.48</td>
<td>0.029</td>
<td>0-60</td>
<td>Organic</td>
<td>1063 S</td>
<td></td>
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<tr>
<td>32. Scheldt, Netherlands</td>
<td>Europe</td>
<td>Spartina anglica</td>
<td>Cfb</td>
<td>4.48</td>
<td>0.020</td>
<td>0-60</td>
<td>Organic</td>
<td>733 S</td>
<td></td>
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<tr>
<td>33. Stiffkey Marsh, Norfolk, UK</td>
<td>Europe</td>
<td>Spartina anglica</td>
<td>Cfb</td>
<td>4.08</td>
<td>0.041</td>
<td>0-50</td>
<td>Organic</td>
<td>1503 S</td>
<td></td>
</tr>
<tr>
<td>34. Kooragang Island, Hunter River, Newcastle (Site1)</td>
<td>SE Australia</td>
<td>Sarcocornia quinqueflora</td>
<td>Cfa</td>
<td>1.9</td>
<td>0.040</td>
<td>100</td>
<td>Organic</td>
<td>1467 S</td>
<td></td>
</tr>
<tr>
<td>35. Kooragang Island, Hunter River, Newcastle (Site2)</td>
<td>SE Australia</td>
<td>Sarcocornia quinqueflora</td>
<td>Cfa</td>
<td>1.9</td>
<td>0.064</td>
<td>100</td>
<td>Organic</td>
<td>2347 S</td>
<td></td>
</tr>
<tr>
<td>36. Kooragang Island, Hunter River, Newcastle</td>
<td>SE Australia</td>
<td>Sarcocornia quinqueflora</td>
<td>Cfa</td>
<td>1.75</td>
<td>0.062</td>
<td>100</td>
<td>Organic</td>
<td>2273 S</td>
<td></td>
</tr>
<tr>
<td>37. Cararma Inlet, Jervis Bay, Woollamia</td>
<td>SE Australia</td>
<td>Sarcocornia quinqueflora</td>
<td>Cfb</td>
<td>Microtidal</td>
<td>nd</td>
<td>0-100</td>
<td>ns</td>
<td>403 S, δ(^{13})(^C)</td>
<td></td>
</tr>
</tbody>
</table>
Carbon stock values are reported in tonnes of carbon per hectare (t C ha\(^{-1}\)). To convert this to tonnes of carbon dioxide equivalents per hectare (t CO\(_2\)e ha\(^{-1}\)), the carbon stock values are multiplied by 3.667 (Based on the atomic weight of carbon and oxygen).

Sediment carbon stock is calculated through the following formula:

\[
\text{Sediment carbon stock (t CO}_2\text{e ha}^{-1}) = \text{Carbon density (g C cm}^{-3}\) \times 100 \times \text{Depth (cm)} \times 100 \times 3.667
\]  
(Formula 1)

Carbon density at 100 cm depth is calculated as:

\[
\text{Carbon density at 100 cm depth} = (\text{Mean carbon density at 20 cm}) - (\text{Mean carbon density at 20 cm} \times 0.015)
\]  
(Formula 2)

(According to the NSW Department of Primary Industries (2008), reported by Howe et al. (2009), soil carbon density at the site declines linearly to around 1.5% at a depth of 1 m)


t Carbon stock values are calculated through multiplying carbon store values (Kg C m\(^{-2}\)) by 36.67 (10*3.667)

t Information on tidal range, sediment carbon density and type of carbon (organic/inorganic) for saltmarsh case studies in USA and Europe are derived from the database provided by Ouyang and Lee (2014). Information on soil depth and method is derived from the original citations. Sediment carbon stock for these studies is estimated through the above-mentioned formula (1).

S: means soil carbon density is estimated using sediment cores, identifying soil bulk density (BD) and carbon concentration (%C)

\(\delta^{13}\): Carbon isotopic composition is measured to distinguish between organic and inorganic carbon

ns: It is not specifically indicated whether the carbon refers to organic carbon.

nd: no data

Note: values are rounded to the nearest one.
Table A2-5. Carbon sequestration in sediment of mangrove (*Avicennia marina*) and saltmarshes (*Spartina alterniflora*, *Spartina anglica*, *Sarcocornia quinqueflora*, *Juncus kraussii*) across temperate climate (Cfa/Cfb)

<table>
<thead>
<tr>
<th>Specific Location</th>
<th>General Location</th>
<th>Dominant Species</th>
<th>Climate</th>
<th>Tidal range (m)</th>
<th>Sediment accretion rate (SAR) (mm yr(^{-1}))</th>
<th>Organic/Inorganic Carbon</th>
<th>Sediment carbon sequestration (tCO(_2)e ha(^{-1}) yr(^{-1}))</th>
<th>Method For SAR estimation</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Mangrove case studies (5)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Moreton Bay, Brisbane</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>1.19&lt;sup&gt;a&lt;/sup&gt;</td>
<td>1.72&lt;sup&gt;b&lt;/sup&gt;</td>
<td>ns</td>
<td>2.8</td>
<td>M</td>
<td>Lovelock et al. 2014&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>2. SE Australia (Victoria, NSW)</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa/Cfb</td>
<td>Microtidal to 3 m</td>
<td>nd</td>
<td>ns</td>
<td>9.4</td>
<td>M</td>
<td>Saintilan et al. 2013&lt;sup&gt;d&lt;/sup&gt;</td>
</tr>
<tr>
<td>3. Kooragang Island, Hunter River, Newcastle (Site1)</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em> (Forsk.)</td>
<td>Cfa</td>
<td>1.9</td>
<td>3.66</td>
<td>Organic</td>
<td>3.8</td>
<td>M</td>
<td>Howe et al. 2009&lt;sup&gt;d&lt;/sup&gt;</td>
</tr>
<tr>
<td>4. Kooragang Island, Hunter River, Newcastle (Site2)</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em> (Forsk.)</td>
<td>Cfa</td>
<td>1.9</td>
<td>1.89</td>
<td>Organic</td>
<td>3.3</td>
<td>M</td>
<td>Howe et al. 2009&lt;sup&gt;d&lt;/sup&gt;</td>
</tr>
<tr>
<td>5. Kooragang Island, Hunter River, Newcastle</td>
<td>SE Australia</td>
<td><em>Avicennia marina</em></td>
<td>Cfa</td>
<td>1.75</td>
<td>3.67</td>
<td>Organic</td>
<td>5.9</td>
<td>M</td>
<td>Howe et al. 2008&lt;sup&gt;d&lt;/sup&gt;</td>
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<tr>
<td><strong>Saltmarsh case studies (23)</strong></td>
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<td></td>
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<td></td>
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<td></td>
</tr>
<tr>
<td>6. Aransas, Texas</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.11</td>
<td>4.5</td>
<td>Organic</td>
<td>6.5</td>
<td>137 Cs</td>
<td>Callaway et al. 1997&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>7. San Bernard, Texas</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.22</td>
<td>6.2</td>
<td>Organic</td>
<td>7.4</td>
<td>137 Cs</td>
<td>Callaway et al. 1997&lt;sup&gt;e&lt;/sup&gt;</td>
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<tr>
<td>8. Biloxi Bay, Mississippi</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.47</td>
<td>5.7</td>
<td>Organic</td>
<td>5.6</td>
<td>137 Cs</td>
<td>Callaway et al. 1997&lt;sup&gt;e&lt;/sup&gt;</td>
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<tr>
<td>9. Old Oyster Bayou, Louisiana</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.31</td>
<td>4.4</td>
<td>Organic</td>
<td>3.1</td>
<td>nd</td>
<td>Chmura et al. 2003&lt;sup&gt;e&lt;/sup&gt;</td>
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<td>10. Bayou Chitigue, Louisiana</td>
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<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.31</td>
<td>32.3</td>
<td>Organic</td>
<td>18.9</td>
<td>M</td>
<td>Day et al. 2011&lt;sup&gt;e&lt;/sup&gt;</td>
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<tr>
<td>11. Old Oyster Bayou, Louisiana</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.31</td>
<td>28.5</td>
<td>Organic</td>
<td>22.1</td>
<td>M</td>
<td>Day et al. 2011&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>12. Bayou Chitigue, Louisiana</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.31</td>
<td>35</td>
<td>Organic</td>
<td>24.5</td>
<td>M</td>
<td>Day et al. 2011&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>13. Lafourche Parish, Louisiana</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.31</td>
<td>9.9</td>
<td>Organic</td>
<td>6.8</td>
<td>M</td>
<td>Cahoon and Turner, 1989&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>14. Ogeechee River, Georgia Coast</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>2.09</td>
<td>2.2</td>
<td>Organic</td>
<td>1.8</td>
<td>137 Cs</td>
<td>Loomis and Craft, 2010&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>15. Altamaha River, Georgia Coast</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>2.19</td>
<td>1.2</td>
<td>Organic</td>
<td>1.0</td>
<td>137 Cs</td>
<td>Loomis and Craft, 2010&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>16. Satilla River, Georgia Coast</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>2.10</td>
<td>2.3</td>
<td>Organic</td>
<td>1.6</td>
<td>137 Cs</td>
<td>Loomis and Craft, 2010&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>17. Oregon Inlet, North Carolina</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.61</td>
<td>2.7</td>
<td>Organic</td>
<td>2.2</td>
<td>137 Cs</td>
<td>Craft et al. 1993&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>18. Oregon Inlet, North Carolina</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.61</td>
<td>0.9</td>
<td>Organic</td>
<td>0.8</td>
<td>137 Cs</td>
<td>Craft et al. 1993&lt;sup&gt;e&lt;/sup&gt;</td>
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<tr>
<td>19. SA4, Little Assawoman Bay, Delaware</td>
<td>USA</td>
<td><em>Spartina alterniflora</em></td>
<td>Cfa</td>
<td>0.47</td>
<td>2.5</td>
<td>Organic</td>
<td>5.6</td>
<td>137 Cs</td>
<td>Elsey-Quirk et al. 2011&lt;sup&gt;e&lt;/sup&gt;</td>
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<td>20. St. Annaland, Netherlands</td>
<td>Europe</td>
<td><em>Spartina anglica</em></td>
<td>Cfb</td>
<td>4.48</td>
<td>6.8</td>
<td>Organic</td>
<td>10.2</td>
<td>137 Cs</td>
<td>Callaway et al. 1996&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
</tbody>
</table>
Tidal range data come from http://www.bom.gov.au based on the nearest tidal gauge to the study site within 70Km (Brisbane Port Office, QLD) (Accessed on 10 Feb 2015)

Sediment accretion rate is the average value of surface elevation rates for Avicennia marina at Western Moreton Bay (Lovelock et al. 2014)

Carbon sequestration rates are calculated using the above-mentioned formula (B)

Sediment carbon sequestration is calculated through multiplying sediment carbon sequestration (t CO$_2$e ha$^{-1}$ yr$^{-1}$) by 3.667.

Information on tidal range, sediment accretion rate, type of carbon (organic/inorganic) and sediment carbon sequestration are derived from the database provided by Ouyang and Lee (2014). Information regarding the method are derived from the original citations. Sediment carbon sequestration for these studies is estimated through the following formula:

\[
\text{Carbon sequestration rates (t CO}_2\text{e ha}^{-1}\text{yr}^{-1}) = \text{Carbon sequestration rates (g C m}^2\text{yr}^{-1}) \times 10^2 \times 3.667
\]

The carbon sequestration rates reported in Saintilan et al. (2013) were not specified by location, so the author was contacted to get location specific information.

M: represents marker horizons method used for measuring vertical sediment accretion and calculation of carbon sequestration

$^{137}$Cs and $^{210}$Pb: Isotope marker techniques for estimation of vertical sediment accretion.

ns: It is not specifically indicated whether the carbon refers to organic carbon.

nd: no data

Note: Values are rounded to the nearest tenth.
Appendix 3: Steps for content analysis

The framework for content analysis in this research is schematically illustrated in Figure A3-1. Although the arrows show a step-wise process, in practice, these steps are not necessarily sequential and the analysis in one step may require referring back to previous steps to reconsider and refine the analyses in the preceding steps. The various steps of this framework are discussed below.

Formulating the questions

The framework starts with formulating a number of specific questions to be answered through the content analysis. The questions are framed as below and intended to eventually provide input into answering the main research questions. These questions point to the key concepts related to climate change benefits of coastal wetlands and are formulated to identify how those concepts are accounted for in the current policy documents in New Zealand and Auckland. Where required through the analysis, similar questions are formulated for other concepts.

- Are coastal wetlands and their services clearly defined in the policy documents and plans?
- Are climate change services of coastal wetlands (CS&S, coastal protection) specifically referred to in the current policies?
- Where there are specific references, what policy documents make the reference and at what level (national/regional; statutory/non-statutory) are they specified?
- How do current policies relate climate change to coastal wetlands?
- What aspects (specifically benefits, services) of coastal wetlands are given higher priority?
- What aspects of coastal protection are linked to coastal services, and how?
- Do policies consider and link sea level rise and coastal hazards to coastal wetlands, and if so how?
- Do policies consider and link mitigation and adaptation benefits of coastal wetlands, and if so how?

Setting the context: this includes the legislative framework that governs management of coastal resources and the roles and responsibilities of central and local government in New Zealand and Auckland as discussed in Chapter 6 (Section 6.2).
Selecting relevant policy documents: The policy documents (at both local and central levels) that were reviewed and analysed in this research are provided in Section 3.3.4. Those regulations and policy documents were selected because they represent the main documents that address the research questions. The selected national legislation and local statutory documents are also the ones that affect decision making regarding management and protection of coastal wetlands. Technical material and guidance manuals were selected because they are non-statutory documents prepared by central government and aim to assist local governments in planning and decision making.

Identifying key terms: A number of key terms were identified and selected based on the focus of the research and the theoretical framework discussions in Chapter 2. The keywords were used to search and identify relevant information within the policy documents. They include ‘coastal wetlands’, ‘coastal environment’, ‘coastal natural resources’, ‘management and protection of coastal wetlands’, ‘climate change services of coastal wetlands’, ‘sea level rise’ and ‘coastal hazards’, ‘resilience’, ‘benefits’ and ‘co-benefits’, ‘carbon sequestration’, ‘carbon storage’, ‘mitigation’, ‘adaptation’, ‘coastal protection’, and the concepts related to policy instruments, initiatives and strategies including ‘ecological compensation’ and ‘biodiversity offset’.

Analysing documents: The information from the review of the policy documents using the selected keywords were categorised under the two main themes of ‘management of coastal resources including coastal wetlands’ and ‘responses to climate change and coastal hazards’ (at both national and regional scales) and were analysed based on a number of key elements (or aspects) that the analysis needed to
focus. Those aspects were identified based on the questions and purpose of the analyses and are listed below:

- Presence or absence of the key terms
- Explicit or implicit wording/articulation of policies
- Definitions (whether they are implicit or explicit)
- Context (legislation and governance structure and relations)
- Priorities/preferences (whether objectives/outcomes are conflicting)
- Barriers/constraints (where the information was available, the analysis considered the issues in relation to potential barriers such as public pressure, lack of knowledge, other policies, lack of mandate, lack of national or international driver, theoretical e.g. avoiding an action if uncertain, lack of transparency, lack of adequate support, financial, technical, time etc.)
- Consistency between policy propositions across documents

**Discussion:** Once the policy and planning documents are reviewed through the previous step, findings are interpreted against findings of the global case studies and the theories discussed in Chapter 2. The discussions are primarily focused to identify the overall position of the current policies about the management of coastal wetlands in relation to their climate change services and what that means from a broader policy and planning perspective within the current legislative, governance and planning context.
Appendix 4: Questionnaire survey

- **Process to identify and contact the potential participants**

  The questionnaire was accompanied by a Participant Information Sheet (PIS) that contained background information about the subject under question, the purpose of the survey, how the questionnaire was structured, privacy of personal information, where and how results will be published and a clear description of the participant's rights to participate or decline. The participants also received a PIS form prior to sending the questionnaires.

  To identify appropriate participants for the purpose of this survey, information from relevant peer-reviewed and grey literature, online resources (e.g. websites of relevant companies, organisations and review of the information about background and experience of staff, where available) and conference proceedings was used. The following three categories were also identified to facilitate selection of participants for the survey:

  - **Category A (Practitioners from the Auckland Council):** This group consists of professional individuals (Manager, Principal and Senior) who have been or are currently involved in policy development, planning and research in the fields related to coastal ecosystems.
  - **Category B (Practitioners from relevant consultant companies and organizations):** This category included individuals who have experience in policy making and research activities regarding coastal wetlands and mangrove management in Auckland.
  - **Category C (Academics, NGO members and researchers from research institutes):** This consists of individuals who are preferably involved in policy advisory and research activities regarding coastal wetlands and mangrove management in Auckland.

  In total, 30 individuals including 28 practitioners from the Auckland Council, consulting companies, Landcare Research, NIWA, Environmental Defence Society (EDS), National Wetland Trust, Auckland University and Landcare Trust and two independent individuals were selected as the potential participants for the survey. However, from the 30 people contacted, only 15 people accepted to complete the questionnaire and participate in the survey. From 15 questionnaires sent, only eight were returned that means a return rate of 53%.

  There is no standard for the acceptable response rate and various studies reported different response rates. A meta-analysis by Manfreda, et al., (2008) found that on average, the response rate in web surveys has been approximately 11% lower than that of other survey methods. A range between 25% to 75% is reported as good or adequate response rate in e-mail surveys (Biersdorff, 2009). Minimal acceptable response rates of 50% (Babbie, 1990), 60% (Fowler Jr, 2013) and 75% (Bailey, 1987) are also reported. For a small sample size, similar to this research, a high response rate (80% or higher) would be ideal (Fryrear, 2015); however, 53% return rate can be also considered as an acceptable rate. In the next step, the participant’s responses to each question were entered into spreadsheets and were separately analysed.

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202 Seven people from category A, three people from category B and five people from category C.

203 Five questionnaires were received from participants in category A, one questionnaire from participants in category B and two questionnaires from participants in category C.
### Structure and content of the questionnaire

**Questions:**

1) How do you evaluate importance of the following ecosystem services\(^1\) of coastal wetlands (Mangroves, saltmarshes and salt meadows) in Auckland? For each service, please specify your own perception of its importance and the degree of importance it has been given in the Proposed Auckland Unitary Plan (i.e. how does the Unitary Plan take account of ecosystem services of the coastal wetlands?)

M: Mangroves, SM: Saltmarshes, SG: Seagrasses

<table>
<thead>
<tr>
<th>(i) Degree of importance</th>
<th>(ii) Degree of importance</th>
</tr>
</thead>
<tbody>
<tr>
<td>(As you would assign to each service)</td>
<td>(Given in the Unitary Plan)</td>
</tr>
<tr>
<td>Very High</td>
<td>High</td>
</tr>
</tbody>
</table>

(A) Provisioning Services: Production of food/medicine for human 

<table>
<thead>
<tr>
<th></th>
<th>M</th>
<th>SM</th>
<th>SG</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

(B) Regulating services: Carbon Storage and Sequestration \(^1\)

<table>
<thead>
<tr>
<th></th>
<th>M</th>
<th>SM</th>
<th>SG</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
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<td></td>
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</tbody>
</table>

(C) Regulating services: Storm Protection

<table>
<thead>
<tr>
<th></th>
<th>M</th>
<th>SM</th>
<th>SG</th>
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</thead>
<tbody>
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<td></td>
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</tbody>
</table>

(D) Regulating services: Erosion Control

<table>
<thead>
<tr>
<th></th>
<th>M</th>
<th>SM</th>
<th>SG</th>
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</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(E) Supporting Services (biogeochemical cycles, e.g. Nutrient cycling, water cycling)</td>
<td>M</td>
<td>SM</td>
<td>SG</td>
</tr>
<tr>
<td>(F) Supporting Services (e.g. Support threatened plant and animal species, supporting biodiversity)</td>
<td>M</td>
<td>SM</td>
<td>SG</td>
</tr>
<tr>
<td>(G) Cultural Services (e.g. Social, religious, recreational)</td>
<td>M</td>
<td>SM</td>
<td>SG</td>
</tr>
<tr>
<td>(H) Other (Please specify):</td>
<td>M</td>
<td>SM</td>
<td>SG</td>
</tr>
</tbody>
</table>

Please use the following box to add your comments, if any:
How do you evaluate the level of knowledge of the ecosystem services of coastal wetlands by the following groups in Auckland, with respect to mitigation (i.e. carbon sequestration capacity) and adaptation (i.e. flood control and coastal protection against e.g. sea level rise, storm surges) services of these ecosystems?

<table>
<thead>
<tr>
<th>Groups</th>
<th>Mitigation services</th>
<th>Adaptation services</th>
</tr>
</thead>
<tbody>
<tr>
<td>(A) Councilors</td>
<td>High</td>
<td>Do not know</td>
</tr>
<tr>
<td></td>
<td>Medium</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Do not know</td>
<td></td>
</tr>
<tr>
<td>(B) Planners (Council officers)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(C) Council Controlled Organisations</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(D) Developers</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(E) Relevant academics and researchers</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(F) Relevant consultants</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(G) Public</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(H) Harbour care or Wai care groups¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(I) Other Non-Governmental Organisations</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(J) Other (Please specify)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Please use the following box to add your comments, if any:
3) How do you rank the following ecosystems in terms of the amount of carbon stored in their biomass and soil and the rate of carbon sequestration (biomass/soil)?
(Please select your answers from the following categories A to G and write it in the table)

A. Natural forest: Including tall indigenous forest; self-sown exotic trees, such as wilting pines and grey willows, broadleaf hardwood shrubland, mānuka–kānuka (Leptospermum scoparium–Kunzea ericoides) shrubland and other woody shrubland (≥ 30 per cent cover, with potential to reach ≥ 5 metres at maturity in situ under current land management within 50–40 years) (MFE, 2015, New Zealand’s GHGs Inventory 1990-2013)

B. Planted forest: Including radiata pine (Pinus radiata), Douglas fir (Pseudotsuga menziesii), eucalypts (Eucalyptus spp.) or other planted species (with potential to reach ≥ 5 metre height at maturity in situ) (MFE, 2015, New Zealand’s GHGs Inventory 1990-2013)

C. Cropland (Annual and Perennial): Including annual crops, orchards and vineyards; cultivated bare ground, linear shelterbelts associated with annual/perennial cropland (MFE, 2015, New Zealand’s GHGs Inventory 1990-2013)

D. Vegetated non-coastal wetlands

E. Mangroves

F. Saltmarshes

G. Seagrass

<table>
<thead>
<tr>
<th></th>
<th>First place</th>
<th>Second place</th>
<th>Third place</th>
<th>do not know</th>
<th>Not enough information for:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic carbon content in biomass</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td></td>
<td>A, B, C, D, E, F, G</td>
</tr>
<tr>
<td>Soil organic carbon</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td></td>
<td>A, B, C, D, E, F, G</td>
</tr>
<tr>
<td>Carbon sequestration rate (in biomass)</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td></td>
<td>A, B, C, D, E, F, G</td>
</tr>
<tr>
<td>Carbon sequestration rate (in soil)</td>
<td>...</td>
<td>...</td>
<td>...</td>
<td></td>
<td>A, B, C, D, E, F, G</td>
</tr>
</tbody>
</table>

Please use the following box to add your comments, if any:

4) To what extent do you agree with the following statement?
“Compared to other ecosystems, in terms of their size and climate change services, coastal wetlands in the Auckland region are important elements of the Auckland urban landscape”

Strongly agree  Agree  Neutral  Disagree  Strongly disagree  I do not know

4.1) What ways, if any, do you think the planning process should take account of climate change services of coastal wetlands?

4.2) To what extent should a response be at a national, regional or local level?
5) To what extent do you agree that an effective climate change mitigation and adaption policy in the Auckland region would include controlling land use in connection with protecting and/or restoring coastal wetlands?

- [ ] Strongly agree
- [ ] Agree
- [ ] Neutral
- [ ] Disagree
- [ ] Strongly disagree

5.1) If you agree, is it because: (You can choose more than one answer)

- [ ] Coastal wetlands effectively contribute to climate change mitigation and adaptation
- [ ] Future changes in land use will threaten these ecosystems and makes them more vulnerable
- [ ] Both A and B; however, A is more important than B
- [ ] Both A and B; however, B is more important than A
- [ ] Both A and B are equally important
- [ ] Other (Please specify)

Please use the following box to add your comments, if any:

5.2) If you disagree, is it because: (You can choose more than one answer)

- [ ] Protection is not necessary as the coastal wetlands particularly mangroves have been naturally expanding in the region
- [ ] There is a growing demand for coastal land development that needs to be given higher priority
- [ ] Auckland's coastal wetlands do not have significant contribution to climate change mitigation and adaptation
- [ ] Engineered flood control infrastructure is more effective than coastal wetlands
- [ ] Other (Please specify)

Please use the following box to add your comments, if any:

6) Which one of the following policies is given higher importance in the “Proposed Auckland Unitary Plan”? (Please select one answer)

- [ ] Land use change policies that allow further land development in coastal areas
- [ ] Policies that encourage private ownership of the coastal wetlands
- [ ] Protection of the coastal ecosystems as significant habitat for indigenous flora and fauna, and as important sites for Mana Whenua
- [ ] Other (please specify)
- [ ] Other (please specify)

Please use the following box to add your comments, if any:

7) On a scale of 1 to 5 (1 is lowest and 5 is highest score), how do you rank the following key threats to the coastal wetlands in Auckland? (Please add other threats, if any)

- [ ] Eutrophication and increased sedimentation rates due to land use change in their catchments
- [ ] Water pollution and reclamation
- [ ] Heavy Stock grazing and trampling, especially by cattle
- [ ] Invasive weeds and introduced predators
- [ ] Vegetation removal for recreation and amenity
- [ ] All of the above
- [ ] Other (Please specify)

Please use the following box to add your comments, if any:

R-15
8) Please select from the following the parameters that need to be considered in development of coastal areas and their degree of importance. For each parameter also specify whether and to what degree of importance it has been considered in the Proposed Auckland Unitary Plan.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Degree of Importance (As you would suggest)</th>
<th>Degree of Importance (Given in the Unitary Plan)</th>
<th>Not considered in the Unitary Plan</th>
</tr>
</thead>
<tbody>
<tr>
<td>(A) Vulnerability of low-lying coastal areas to the effects of climate change (Risk assessment approach)</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
</tr>
<tr>
<td>(B) Land value and public interest in coastal areas</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
</tr>
<tr>
<td>(C) Other (please specify) ..................................................................</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
</tr>
<tr>
<td>(D) Other (please specify) ..................................................................</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
</tr>
<tr>
<td>(E) Other (please specify) ..................................................................</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
<td>☐ ☐ ☐ ☐</td>
</tr>
</tbody>
</table>

Please comment below if you have any suggestions for a suitable mitigation-adaptation policy:

9) Which one of the followings do you consider as the most fundamental factor influencing intention and capacity of the local government to contribute to the climate change mitigation? (You can choose more than one answer)

(A) Central government management of greenhouse gas emissions initiatives ☐ ☐ ☐ ☐
(B) Legislative change .................................................................................. ☐ ☐ ☐ ☐
(C) Lack of independence to take action at local, regional and national levels ☐ ☐ ☐ ☐
(D) All of the above .................................................................................. ☐ ☐ ☐ ☐
(E) Other (please specify) .......................................................................... ☐ ☐ ☐ ☐
(F) Other (please specify) .......................................................................... ☐ ☐ ☐ ☐
(G) Other (please specify) .......................................................................... ☐ ☐ ☐ ☐

Please use the following box to add your comments, if any:

Are you interested in receiving the results of this survey? ☐ Yes ☐ No

Thank you so much for your time
UNIVERSITY OF AUCKLAND HUMAN PARTICIPANTS ETHICS COMMITTEE (UAHPEC)

15-Mar-2015

MEMORANDUM TO:
Dr Stephen Knight-Lesihan
Architecture & Planning

Re: Application for Ethics Approval (Our Ref. 013770): Approved with comment

The Committee considered your application for ethics approval for your project entitled Climate Compatible Green Infrastructure Planning in the Auckland Region.

Ethics approval was given for a period of three years with the following comment(s):

1. On all PISs:

   a. Please remove the wording: “This form will be kept for a period of six years”, because the PIS is kept by participants and not stored by the researchers.

   b. Please also add the UAHPEC Chair contact details on the PISs (For any queries regarding ethical concerns you may contact the Chair, The University of Auckland Human Participants Ethics Committee, The University of Auckland, Research Office, Private Bag 92019, Auckland 1142. Telephone 99 373-7999 extn. 83711. Email: ro-ethics@auckland.ac.nz).

The expiry date for this approval is 15-Mar-2018.

If the project changes significantly you are required to resubmit a new application to UAHPEC for further consideration.

In order that an up-to-date record can be maintained, you are requested to notify UAHPEC once your project is completed.

The Chair and the members of UAHPEC would be happy to discuss general matters relating to ethics approvals if you wish to do so. Contact should be made through the UAHPEC Ethics Administrators at ro-ethics@auckland.ac.nz in the first instance.

All communication with the UAHPEC regarding this application should include this reference number: 013770.

(This is a computer generated letter. No signature required.)
Secretary
University of Auckland Human Participants Ethics Committee

c.c. Head of Department / School, Architecture & Planning
Ms Behnaz Khodabakhshi
Dr Marjorie van Roon
Ms Elizabeth Aitken Rose

Additional information:
1. Should you need to make any changes to the project, write to the Committee giving full details including revised documentation.

2. Should you require an extension, write to the Committee before the expiry date giving full details along with revised documentation. An extension can be granted for up to three years, after which time you must make a new application.

3. At the end of three years, or if the project is completed before the expiry, you are requested to advise the Committee of its completion.

4. Do not forget to fill in the ‘approval wording’ on the Participant Information Sheets and Consent Forms, giving the dates of approval and the reference number, before you send them out to your participants.

5. Send a copy of this approval letter to the Awards Team at the Research Office if you have obtained funding other than from UniServices. For UniServices contracts, send a copy of the approval letter to: Contract Manager, UniServices.

6. Please note that the Committee may from time to time conduct audits of approved projects to ensure that the research has been carried out according to the approval that was given.
## Appendix 6: Responses to the questionnaire

### Questions:

1) How do you evaluate importance of the following ecosystem services\(^1\) of coastal wetlands (Mangroves, saltmarshes and salt meadows) in Auckland? For each service, please specify your own perception of its importance and the degree of importance it has been given in the Proposed Auckland Unitary Plan (i.e. how does the Unitary Plan take account of ecosystem services of the coastal wetlands?)

M: Mangroves, SM: Saltmarshes, SG: Seagrasses

<table>
<thead>
<tr>
<th>(i) Degree of importance (As you would assign to each service)</th>
<th>(ii) Degree of importance (Given in the Proposed Auckland Unitary Plan)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Very High</td>
<td>High</td>
</tr>
</tbody>
</table>

### Provisioning Services: Production of food/medicine for human

| M | - | 1 of 8 | 2 of 8 | 3 of 8 | 2 of 8 | - | - | - | - | 3 of 8 | 5 of 8 | - |
| SM | - | 1 of 8 | 1 of 8 | 4 of 8 | 2 of 8 | - | - | - | 1 of 8 | - | 1 of 8 | 5 of 8 | 1 of 8 |
| SG | - | 2 of 8 | 1 of 8 | 4 of 8 | 1 of 8 | - | - | - | 1 of 8 | - | 1 of 8 | 5 of 8 | 1 of 8 |

### Regulating services: Carbon Storage and Sequestration\(^2\)

| M | 1 of 8 | 5 of 8 | 2 of 8 | - | - | - | - | - | - | 1 of 8 | 2 of 8 | 4 of 8 | 1 of 8 |
| SM | - | 3 of 8 | 2 of 8 | 2 of 8 | 1 of 8 | - | - | - | 1 of 8 | 1 of 8 | 1 of 8 | 4 of 8 | 1 of 8 |
| SG | - | 3 of 8 | - | 4 of 8 | - | 1 of 8 | - | - | 1 of 8 | 1 of 8 | 1 of 8 | 4 of 8 | 1 of 8 |

### Regulating services: Storm Protection

| M | 4 of 8 | 3 of 8 | - | 1 of 8 | - | - | - | 1 of 8 | 3 of 8 | 1 of 8 | 2 of 8 | 1 of 8 | - |
| SM | 1 of 8 | 3 of 8 | 1 of 8 | 2 of 8 | 1 of 8 | - | 1 of 7 | 1 of 7 | 1 of 7 | 1 of 7 | 2 of 7 | 1 of 7 | 1 of 7 |
| SG | 1 of 8 | 2 of 8 | - | 3 of 8 | 2 of 8 | - | - | 1 of 7 | 1 of 7 | 1 of 7 | 3 of 7 | 1 of 7 | 1 of 7 |

### Regulating services: Erosion Control

| M | 4 of 8 | 1 of 8 | 3 of 8 | - | - | - | - | 1 of 8 | 3 of 8 | 1 of 8 | 2 of 8 | 1 of 8 | - |
| SM | 2 of 8 | 2 of 8 | 2 of 8 | 1 of 8 | 1 of 8 | - | 1 of 8 | 2 of 8 | 1 of 8 | 1 of 8 | 2 of 8 | 1 of 8 | - |
| SG | 1 of 8 | 2 of 8 | 1 of 8 | 2 of 8 | 2 of 8 | - | - | 2 of 8 | 1 of 8 | 1 of 8 | 3 of 8 | 1 of 8 | - |

### Supporting Services (biogeochemical cycles, e.g. Nutrient cycling)

<p>| M | 4 of 8 | 3 of 8 | 1 of 8 | - | - | - | - | - | - | 1 of 8 | 3 of 8 | 4 of 8 | - |
| SM | 2 of 8 | 3 of 8 | 1 of 8 | 1 of 8 | 1 of 8 | - | - | 1 of 8 | 2 of 8 | 4 of 8 | 1 of 8 | - |</p>
<table>
<thead>
<tr>
<th></th>
<th>M</th>
<th>SM</th>
<th>SG</th>
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</thead>
<tbody>
<tr>
<td>water cycling</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(F) Supporting</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Services</td>
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<tr>
<td>(e.g. Support</td>
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<td></td>
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<tr>
<td>threatened plant</td>
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<td></td>
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<tr>
<td>and animal</td>
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<tr>
<td>species,</td>
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<td>supporting</td>
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<tr>
<td>biodiversity)</td>
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<td></td>
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<tr>
<td>(G) Cultural</td>
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<tr>
<td>Services</td>
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<td>(e.g. Social,</td>
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<tr>
<td>religious,</td>
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<tr>
<td>recreational)</td>
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<td></td>
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<tr>
<td>(H) Other</td>
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<td></td>
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<tr>
<td>(Please specify)</td>
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<td></td>
<td></td>
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<tr>
<td>juvenile fish</td>
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<tr>
<td>nursery</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>(I) Other</td>
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<td></td>
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<tr>
<td>(Please specify)</td>
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</tbody>
</table>
2) How do you evaluate the level of knowledge of the ecosystem services of coastal wetlands by the following groups in Auckland, with respect to mitigation (i.e. carbon sequestration capacity) and adaptation (i.e. flood control and coastal protection against e.g. sea level rise, storm surges) services of these ecosystems?

<table>
<thead>
<tr>
<th>Mitigation services</th>
<th>Adaptation services</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>High</td>
</tr>
<tr>
<td>(A) Councillors</td>
<td>-</td>
</tr>
<tr>
<td>(B) Planners (Council officers)</td>
<td>2 of 8</td>
</tr>
<tr>
<td>(C) Council Controlled Organisations</td>
<td>1 of 8</td>
</tr>
<tr>
<td>(D) Developers</td>
<td>-</td>
</tr>
<tr>
<td>(E) Relevant academics and researchers</td>
<td>7 of 8</td>
</tr>
<tr>
<td>(F) Relevant consultants</td>
<td>3 of 8</td>
</tr>
<tr>
<td>(G) Public</td>
<td>-</td>
</tr>
<tr>
<td>(H) Harbour care or Wai care groups</td>
<td>2 of 8</td>
</tr>
<tr>
<td>(I) Other Non-Governmental Organizations</td>
<td>2 of 8</td>
</tr>
<tr>
<td>(J) Other (Please specify): iwi</td>
<td>1 of 1</td>
</tr>
</tbody>
</table>

3) How do you rank the following ecosystems in terms of the amount of carbon stored in their biomass and soil and the rate of carbon sequestration (biomass + soil)?

<table>
<thead>
<tr>
<th>First place</th>
<th>Second place</th>
<th>Third place</th>
<th>Not enough information for:</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Organic carbon content in biomass</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural forest (4 of 6)</td>
<td>Mangroves (1 of 5)</td>
<td>Vegetated non-coastal wetlands (1 of 6)</td>
<td>Mangroves (2 of 5)</td>
</tr>
<tr>
<td>Vegetated non-coastal wetlands (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 5)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Vegetated non-coastal wetlands (1 of 6)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Soil organic carbon</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural forest (3 of 6)</td>
<td>Planted forest (1 of 6)</td>
<td>Vegetated non-coastal wetlands (2 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td>Vegetated non-coastal wetlands (2 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td>Do not know (1 of 6)</td>
<td></td>
<td>Mangroves (1 of 6)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Carbon sequestration rate (in biomass)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Planted forest (2 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (2 of 4)</td>
</tr>
<tr>
<td>Vegetated non-coastal wetlands (2 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td>Mangroves (1 of 5)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td>Croplands (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td></td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Vegetated non-coastal wetlands (1 of 6)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Carbon sequestration rate (in soil)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Vegetated non-coastal wetlands (3 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Planted forest (2 of 5)</td>
<td>Mangroves (1 of 5)</td>
</tr>
<tr>
<td>Planted forest (1 of 5)</td>
<td>Mangroves (1 of 6)</td>
<td>Natural forest (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
</tr>
<tr>
<td>Natural forest (1 of 6)</td>
<td>Do not know (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td></td>
</tr>
<tr>
<td>Do not know (1 of 6)</td>
<td>Do not know (1 of 6)</td>
<td>Mangroves (1 of 6)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

R-21
4) To what extent do you agree with the following statement?

"Compared to other ecosystems, in terms of their size and climate change services, coastal wetlands in the Auckland region are important elements of the Auckland urban landscape."

[ ] 5 of 7  [ ] 2 of 7  [ ] Neutral  [ ] Disagree  [ ] Strongly disagree  [ ] I do not know

4.1) What ways, if any, do you think the planning process should take account of climate change services of coastal wetlands?

Classify as significant natural areas or some other form of protection rules

Anyway

Anyway

Many areas in wider Auckland will be flooded with sea level rise, so the region needs to understand wetland services to take advantage of these

Anyway

By protecting these

By protecting them

The protection of coastal wetlands is acknowledged in planning documents but seldom given effect by the rules. Often a lack of integration across different rules e.g. zoning and ecological protection overlays result in poor protection of these areas. A lack of a clear biodiversity mitigation and offset guideline also inhibits best practice for compensating loss of these ecosystems.

Anyway

These areas need to be mapped, potential at-risk and expansion zones identified, and rules should protect these areas (with consent assessment criteria related to these functions)

Anyway

By delineating a wide "no go" zone around the entire coastline where long-term, private property rights are restricted, the land is kept in public ownership and the regeneration of natural ecosystems and processes is actively supported and encouraged. Obviously, this is a long-term project!

One of my great concerns is that the very rich (who tend to own property along the coast) will begin to force the rest of us to pay for very expensive and ultimately (King Canute like) doomed to failure 'solutions' to protect their private assets...

Anyway

After all a cliff is just an eroding hill. Why build your house at the top of one and expect the hill to stop eroding?

4.2) To what extent should a response be at a national, regional or local level?

It should be at a regional level, to take account of different coastal processes, geology and soil types, and development pressures. But the requirement to do this should come from a national level

Anyway

Should relate to regional rules and district consent processes

Anyway

A mixture depending on the issue to be addressed by policy

Anyway

A response needs to be at all of these levels, that includes a non-response (allowing inundation to occur)

Anyway

All levels have a role to play with clear direction at national and regional levels and action at local level

Anyway

National policy, local and regional implementation

R-22
5) To what extent do you agree that an effective climate change mitigation and adaption policy in the Auckland region would include controlling land use in connection with protecting and/or restoring coastal wetlands?

<table>
<thead>
<tr>
<th>3 of 7</th>
<th>4 of 7</th>
<th>Neutral</th>
<th>Disagree</th>
<th>Strongly disagree</th>
</tr>
</thead>
<tbody>
<tr>
<td>Strongly agree</td>
<td>Agree</td>
<td>Neutral</td>
<td>Disagree</td>
<td>Strongly disagree</td>
</tr>
</tbody>
</table>

5.1) If you agree, is it because: (You can choose more than one answer)

(A) Coastal wetlands effectively contribute to climate change mitigation and adaptation
(B) Future changes in land use will threaten these ecosystems and makes them more vulnerable
(C) Both A and B; however, A is more important than B
(D) Both A and B; however, B is more important than A
(E) Both A and B are equally important
(F) Other (Please specify)

5.2) If you disagree, is it because: (You can choose more than one answer)

(A) Protection is not necessary as the coastal wetlands particularly mangroves have been naturally expanding in the region
(B) There is a growing demand for coastal land development that needs to be given higher priority
(C) Auckland's coastal wetlands do not have significant contribution to climate change mitigation and adaptation
(D) Engineered flood control infrastructure is more effective than coastal wetlands
(E) Other (Please specify)

6) Which one of the following policies is given higher importance in the “Proposed Auckland Unitary Plan”?

(Please select one answer)

(A) Land use change policies that allow further land development in coastal areas
(B) Policies that encourage private ownership of the coastal wetlands
(C) Protection of the coastal ecosystems as significant habitat for indigenous flora and fauna, and as important sites for Mana Whenua
(D) Other (please specify)
(E) Other (please specify)
7) How do you rank the following key threats to the coastal wetlands in Auckland?

(A) Eutrophication and increased sedimentation rates due to land use change in their catchments
   - Very high (4 of 6)
   - High (1 of 6)
   - Moderate (1 of 6)
   - Low (1 of 6)

(B) Water pollution and reclamation
   - High (3 of 6)
   - Very high (1 of 6)
   - Moderate (1 of 6)
   - Very low (1 of 6)

(C) Heavy Stock grazing and trampling, especially by cattle
   - Very high (2 of 6)
   - High (2 of 6)
   - Low (2 of 6)

(D) Invasive weeds and introduced predators
   - Low (2 of 6)
   - Very low (1 of 6)
   - Moderate (1 of 6)
   - High (1 of 6)
   - Very high (1 of 6)

(E) Vegetation removal for recreation and amenity
   - Very low (2 of 5)
   - Moderate (1 of 5)
   - High (1 of 5)
   - Very high (1 of 5)

(F) All of the above
   - 1 response
8) Please select from the following the parameters that need to be considered in development of coastal areas and their degree of importance. For each parameter also specify whether and to what degree of importance it has been considered in the Proposed Auckland Unitary Plan.

<table>
<thead>
<tr>
<th>Degree of importance (As you would suggest)</th>
<th>Degree of importance (Given in the Unitary Plan)</th>
<th>Not considered in the PAUP</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td>(A) Vulnerability of low-lying coastal areas to the effects of climate change (Risk assessment approach)</td>
<td>6 of 7</td>
<td>1 of 7</td>
</tr>
<tr>
<td>(B) Land value and public interest in coastal areas</td>
<td>1 of 7</td>
<td>5 of 7</td>
</tr>
<tr>
<td>(C) Other (Please specify): Ability to retreat</td>
<td>1 of 1</td>
<td>-</td>
</tr>
<tr>
<td>(D) Other (Please specify): Servicing issues</td>
<td>-</td>
<td>1 of 1</td>
</tr>
<tr>
<td>(E) Other (Please specify)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

9) Which one of the followings do you consider as the most fundamental factor influencing intention and capacity of the local government to contribute to the climate change mitigation? (You can choose more than one answer)

(A) Central government management of greenhouse gas emissions initiatives 2 of 9
(B) Legislative change 2 of 9
(C) Lack of independence to take action at local, regional and national levels 0
(D) All of the above 3 of 9
(E) Other (please specify) General lack of public interest and action in forcing their leaders to change things 1 of 9
(F) Other (please specify) Agency capture by the development community 1 of 9
(G) Other (please specify)
Appendix 7: Systematic literature survey to identify case studies

The cases of interest in this research included the jurisdictions where at least one of the climate change services of coastal wetlands have been considered within their local climate change strategies and plans. Selection of the case studies started with a preliminary search of the peer-reviewed journal articles in the ‘Google Scholar’ using the search terms in Table A7-1. The search returned a number of references that included information about coastal wetlands restoration/protection activities undertaken at local scale specifically with a view to address climate change. These included restoration of coastal wetlands in New York City as a viable soft measure for shoreline protection (Rosenzweig, et al., 2011), integration of flood protection and coastal wetland restoration activities in New Orleans under the approach of ‘multiple lines of defence’ to enhance resilience to the impacts of climate change (Kazmierczak & Carter, 2010; Moore & Stone, 2009) and community-based mangrove restoration projects in Gazi Bay, Kenya and South Sulawesi, Indonesia.

Table A7-1. Keywords used to search ‘Google Scholar’ for studies on coastal wetlands protection/restoration for climate change mitigation/adaptation (Timespan: 1990 – May 2016, Date of search 12 May 2016)

<table>
<thead>
<tr>
<th>Subject</th>
<th>No.</th>
<th>Combination of search terms</th>
</tr>
</thead>
<tbody>
<tr>
<td>Studies on restoration/protection of coastal wetlands for climate change mitigation/adaptation</td>
<td>1.</td>
<td>“climate change” + “restoration of coastal wetlands” + “mitigation” + “adaptation”</td>
</tr>
<tr>
<td></td>
<td>2.</td>
<td>“climate change” + “protection of coastal wetlands” + “mitigation” + “adaptation”</td>
</tr>
<tr>
<td></td>
<td>3.</td>
<td>“climate change” + “restoration of tidal wetlands” + “mitigation” + “adaptation”</td>
</tr>
<tr>
<td></td>
<td>4.</td>
<td>“climate change” + “protection of tidal wetlands” + “mitigation” + “adaptation”</td>
</tr>
</tbody>
</table>

The initial review also found a recent publication by Mega (2016) which included references to a number of global (transnational) partnerships and alliances working in the field of climate change adaptation and mitigation responses by local governments. These include the World Mayors Council on Climate Change (WMCCC); the C40 Cities Climate Leadership Group (C40); 100 Resilient Cities (100 RC); the International Council for Local Environmental Initiatives or Local Governments for Sustainability (ICLEI) and the Compact of Mayors. C40 also includes a smaller network called Connecting Delta Cities (CDC) consisting of a number of delta cities that are active in the field of climate change adaptation. Review of the information provided by these initiatives found references to the use of green measures and ecosystem services in New Orleans, New York, Ho Chi Minh City, Jakarta and Rotterdam that aimed to create more resilient and adaptive urban environments to climate change.

Since the above initiatives were mainly focused on climate change adaptation, a Google search and a review of grey literature were conducted to identify any initiatives related to climate change mitigation. The review found a number of initiatives including Blue Carbon Initiative; GRID-Arendal (the Blue Carbon Portal); the Livelihoods Carbon Fund; Mangroves for the Future (MFF); the Ocean Foundation; Global Climate Change Alliance (GCCA); Mangrove Alliance; and Global Environment Facility (GEF) Blue Forest Program. Review of the information provided by these initiatives found references to a number of local coastal wetland restoration (mainly mangrove restoration) projects that were primarily aimed to generate carbon credits for improving local livelihood mainly in developing countries. The

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projects were located in Kenya (Mombasa city), Dominican Republic (Monte Cristi province), Senegal (Casamance and Sine Saloum regions), Indonesia (Provinces of Gorontalo, North Sumatra and Central Java), India (States of Tamil Nadu and West Bengal), Thailand (Samut Songkhram province) and Singapore.

The cases selected through an initial review included the cities of New York, New Orleans, Rotterdam, Ho Chi Minh City, Jakarta, Mombasa and Singapore; states of Tamil Nadu and West Bengal; provinces of Gorontalo, North Sumatra, Central Java, Samut Songkhram and Monte Cristi and regions of Casamance and Sine Saloum. A search within peer-reviewed and grey literature using the keywords in Table A7-2 was conducted to identify climate change related plans/policy documents of the cases. The purpose was to review the climate change related plans or policy documents of these cases to identify the cases that have included provisions for restoration/protection of coastal wetlands within their climate change response or urban development plans.

The search found no climate change related plans or policy documents for Mombasa, Monte Cristi, Casamance, Sine Saloum, Gorontalo, Samut Songkhram, North Sumatra and Central Java. Review of the climate change related plans or policy documents of the remaining cases indicated that the cities of New York, New Orleans and Jakarta and the states of Tamil Nadu and West Bengal have included initiatives or strategies for restoration of coastal wetlands (including mangrove and saltmarsh ecosystems) within their climate change response- and urban development-plans/strategies.

Table A7-2. Keywords used to search databases for climate change related plans/policy documents of cases selected from the initial review (Timespans: 1990 – Feb 2016, Date of search: 26 Feb 2016)

<table>
<thead>
<tr>
<th>Database</th>
<th>Search statement</th>
</tr>
</thead>
<tbody>
<tr>
<td>Google</td>
<td>Name of the city- climate change legislation</td>
</tr>
<tr>
<td></td>
<td>Name of the city- climate change policy</td>
</tr>
<tr>
<td>Google Scholar</td>
<td>“Name of the city” + “climate change” + “legislation”</td>
</tr>
<tr>
<td></td>
<td>“Name of the city” + “climate change” + “policy”</td>
</tr>
<tr>
<td>ScienceDirect</td>
<td>ABSTRACT (Name of the city) and ABSTRACT (climate change)</td>
</tr>
<tr>
<td>Web of Science</td>
<td>TOPIC:( Name of the city) AND TOPIC: (climate change) AND TOPIC: (policy)</td>
</tr>
</tbody>
</table>

There was no explicit reference to coastal wetlands within Rotterdam’s climate change response plan; however, the plan, under the ‘building with nature’ approach, includes provisions that encourage the use of ecosystem services for adaptation to climate change. The city of Rotterdam is also playing a leading role in developing innovative climate change adaptation strategies and supporting development and implementation of the adaptation measures across a number of other delta cities such as Ho Chi Minh City and Jakarta (Connecting Delta Cities, 2014).

Therefore, the cities of New York and New Orleans in the US, the city of Jakarta in Indonesia, the state of Tamil Nadu in India and the city of Rotterdam in the Netherlands were identified as the case studies for this research. The state of West Bengal was not considered as a case study because it contains the world’s largest mangrove forest (i.e. Sundarbans mangrove forest) which makes it a special case and not comparable with the extent of mangroves in the Auckland region.
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