Modelling the Impacts of Sea Level Rise on Tidal Wetlands

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Abstract

In this century, it is expected that both coastal land development and sea level rise will pose a major threat to tidal wetlands. Historically, tidal salt marshes and mangroves have adjusted to sea level rise, but how they will adjust to the accelerated sea level rise associated with anthropogenic climate change is uncertain. Future adjustments are likely to be limited both by the capacity of the wetlands to accrete, the ability of the vegetation at the seaward edge to tolerate greater hydroperiods and the suitability of inland areas for wetland migration. With the presence of natural and anthropogenic barriers inland, the capacity of wetlands to adjust to sea level rise and the provision of their ecosystem services are likely to be compromised. Using spatially explicit analyses in a geographic information system (GIS), this thesis presents a series of studies modelling magnitude and impacts associated with sea level rise and how these threats will affect two ecosystem services-habitat provision and carbon storage. An index quantifying threats to migration space or 'coastal squeeze' was developed based upon elevation, accretion, slope and degree of imperviousness of intertidal zone. The index was used to rank the threats of coastal squeeze to three marshes at different sea level rise rates. A modification of the coastal squeeze index, using global datasets, was applied to rank the level of threat to North American salt marshes and mangroves. Using a suite of landscape ecology metrics, I examined the impacts of coastal squeeze and different rates of sea level rise on the spatial distribution, size, shape and orientation of wetland patches as they relate to the quality, quantity and availability of fish habitat. The results of different assumptions of accretion rates (i.e., constant rate vs. accretion rate equals sea level rise rate) were compared. Finally, using a spatially and temporally explicit model, I evaluated the sensitivity of carbon storage in a marsh relative to the different rates and trends (i.e., linear vs. non-linear) of sea level rise, spatial variations in vertical accretion, creek expansion, inland migration and topography.

Résumé

Au cours du dernier siècle, les prévisions indiquent que l'augmentation du niveau de la mer et l'intensification du développement dans les zones côtières posera une grande menace sur les milieux humides. Jusqu'à présent, les habitats englobant les marais salants et les mangroves se sont adaptés aux niveaux historiques de la mer, mais la montée des eaux attendue à cause du changement climatique anthropique suscite un avenir incertain pour ces habitats, notamment sur leur capacité d'adaptation qui reste largement inconnu. Dans le futur, cette adaptation pourrait être limitée à la fois par la capacité des zones humides à s'ajuster et la prédisposition des zone intérieures à soutenir la mise en place de la végétation. Ainsi, la présence d'obstacles naturels et anthropiques dans ces zones pourrait compromettre la quantité et la qualité des leurs services écosystémiques. En utilisant des analyses spatiales exclusives dans le domaine du système de l'information géographique (SIG), cette thèse présente une série d'études permettant de modeler l'ampleur et les impacts des menaces indirectes associées à l'augmentation du niveau de la mer sur les zones humides. Particulièrement, ces études permettent à évaluer les impacts des menaces indirectes sur les deux services écosystémiques des zones humides: approvisionnement de l'habitat et stockage de carbone. Tout d'abord, un indice décrivant les menaces sur l'espace de migration ou «indice de la compression côtière» a été développé à partir de l'élévation des sols, du taux d'accrétion, de la pente et du degré d'imperméabilité la zone intertidale. L'indice a été testé pour classer le potentiel de migration dans trois marais sélectionnés avec différents taux d'élévation du niveau de la mer. Ensuite, en utilisant un ensemble de données globales, un indice modifié a été utilisé pour classer les niveaux de menace dans les marais et les mangroves d'Amérique du Nord. En utilisant des indicateurs de l'écologie du paysage, j'ai analysé les impacts de compression côtière et les différents taux d'élévation du niveau de la mer sur la répartition spatiale, la taille, la forme et l'orientation des parcelles des zones humides qui ont trait à la qualité, la quantité et la disponibilité des habitats du poisson. Les résultats des différentes hypothèses du taux d'accrétion (c'est-à-dire que le taux constant par rapport au taux d'accrétion est égal à taux d'élévation du niveau de la mer) ont été comparés. Enfin, en utilisant un modèle spatialement temporellement exclusif, j'ai évalué la sensibilité du stockage de carbone dans un marais par rapport aux différents taux et tendances de l'élévation du niveau de la mer, ainsi qu'aux variations spatiales de l'accrétion verticale, de l'expansion des ruisseaux avoisinants, de la migration intérieure et de la topographie.

Abbreviations and Acronyms

ACCSLSIM	Accretion and Sea Level Simulator
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
CBD	Convention on Biological Diversity
CEC	Commission for Environmental Cooperation
CGVD24	Canadian Geodetic Vertical Datum of 1924
CGIAR-CSI	Consultative Group for International Agricultural Research-Consortium of Spatial Information
C	Carbon
CO_2	Carbon Dioxide
CONABIO	Comision National Para El Conocimiento Y Uso de la Biodiversidad
CSI	Coastal Squeeze Index
DEM	Digital Elevation Model
DIVA	Dynamic Interactive Vulnerability Assessment
ESA	European Space Agency
ESLI	Ecologically-scaled Landscape Index
ESRI	Environmental Systems Research Institute
GEC^3	Global Environment and Climate Change Centre
GEIODE	Geomatics for Informed Decision
GIC	Geographic Information Centre
GIS	Geographical Information Systems
GOMC	Gulf of Maine Council
IPCC	Intergovernmental Panel on Climate Change
KNP	Kouchibouguac National Park
LiDAR	Light Detection and Ranging
MEA	Millennium Ecosystem Assessment
MEM	Marsh Equilibrium Model
MEXSIM	Marsh Extent Simulator
Mg	Megagram
MODIS	Moderate Resolution Imaging Spectroradiometer
MSL	Mean Sea Level
NAMA	Nationally Appropriate Mitigation Actions
NASA	National Aeronautics and Space Administration
NAVD88	North American Vertical Datum of 1988
NERRS	National Estuarine Research Reserve System
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
NSERC	Natural Sciences and Engineering Research Council of Canada
RAMSAR	Convention on Wetlands of International Importance
RCP	Representative Concentration Pathway
RMSE	Root Mean Square Error
SIG	Système de l'Information Géographique
SLAMM	Sea Level Rise Affecting Marshes Model
SRTM	Shuttle Radar Topography Mission
T/MPA	Terrestrial/Marine Protected Areas
TIDEXSIM	Tidal Range Extent Simulator
UNEP-WCMC	United Nations Environmental Programme-World Conservation Monitoring Centre

UNFCCC	United Nation Framework Convention on Climate Change
USGS	United States Geological Survey

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Contribution of Authors

Chapters 3-6 are written manuscript style, with two already published. Due to the overarching question regarding marsh response to sea level rise and the development of spatially explicit models, there is a necessity of some repetition of text within each chapter's introduction

Manuscript #1 (Chapter 3): **"Assessing coastal squeeze of tidal wetlands"** (*Journal of Coastal Research, 2013*). Gail Chmura contributed in choosing the study sites, discussing the import of results, and editing of the manuscript.

Manuscript #2 (Chapter 4): **"Assessing coastal squeeze of North American wetlands"** (*in preparation*). Gail Chmura contributed in choosing the study sites, discussing the import of results, and editing of the manuscript.

Manuscript #3 (Chapter 5): **"Impacts of sea level rise on marsh as fish habitat"** (*Estuaries and Coasts, 2013*). Gail Chmura contributed in choosing the study sites, discussing the import of results, and editing of the manuscript.

Manuscript #4 (Chapter 6): **"Sensitivity of salt marsh carbon to sea level rise"** (*in preparation*). Gail Chmura provided data on marsh accretion rates. Gail Chmura contributed in choosing the study sites, discussing the import of results, and editing of the manuscript

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Chapter 1 Introduction

1.1 The coastal zone

The coast has always been a dynamic place where people and nature interact through time and space. Currently, around 10% of the world's population lives in coastal areas and is projected to increase (Hassan et al. 2005; McGranahan et al. 2007; Wong et al. 2014). Adjacent to coastal settlements are salt marshes and mangroves. These systems provide people with raw materials, protection from storms, enhanced livelihoods, and other ecosystem services (e.g., Barbier et al. 2012). Human pressure on intertidal resources has been so great that large areas of wetlands were degraded or lost. National and international policies have been developed to protect remaining wetlands from further destruction. Although direct human impacts are becoming more irrelevant, tidal wetlands are still being lost and degraded by indirect threats associated with climate change and indirect human impacts. In fact, wetland losses are expected to increase within the century owing to the projected accelerated sea level rise and land development in low lying coastal areas. It is expected that the synergy between these two forces could create physical and socio-economic barriers to inland migration as tidal wetlands respond to sea level rise (e.g., Brinson et al. 1995). The presence of barriers, whether physical or socio-economic, can compromise the resilience, permanence and functional integrity of tidal wetlands. At present, how tidal wetlands will adjust to 21st century threats remains a major knowledge gap.

1.2 Nature of tidal wetlands

Tidal salt marshes and mangroves are ecosystems consisting of and founded by halophytic species adapted to flooded coastal zones (Angelini et al. 2011; Chapman 1974; Lugo and Snedaker 1974; Redfield 1972; Redfield and Rubin 1962). As coastal ecosystems, salt marshes and mangroves are inhabited both by organisms of terrestrial and marine origins (Pennings and Bertness 2001). Both are found on sheltered coasts. Although salt marshes can be found at almost all latitudes, they tend to develop mainly in temperate coastal regions while mangroves are found mainly in the tropics and sub-tropics (Costa and Davy 1992; Pennings and Bertness 2001). On their northern margins, salt marshes are limited by the destructive forces of ice while they are largely replaced by mangroves at lower latitudes (Mitsch and Gosselink 2007; Pennings and Bertness 2001). Salt marsh vegetation may be dominated by graminoids, forbs, or shrubs while trees and palms dominate in mangrove swamps. There is a limited pool of species adapted to survive the stresses of tidal flooding, thus plant diversity in both types of tidal wetlands is low.

A specific elevation range (McKee et al. 2012; Mckee and Patrick 1988) defines the extent of tidal wetlands and its species zones. Although the relationship between elevation and species zonation is complex (Silvestri et al. 2005), it is generally accepted that zonation is an outcome of species competition at the less stressful edge (i.e., higher elevation) and ability to tolerate stress at the more stressful lower elevation edge (Lugo and Snedaker 1974; Mitsch and Gosselink 2007; Pennings and Callaway 1992). This implies that a decreasing elevation increases species exposure to different stressors or an inverse relationship between tolerance to stress and competitive advantage (Levine et al. 1998). Frequent flooding and strong waves are the major stressors in the low elevation wetland zones. To adapt to these stresses, the plant species in this zone have developed aerenchyma tissue for gas exchange and supportive root systems. For example, *Spartina alterniflora*, a species with a robust root system and well-developed aerenchyma (Mendelssohn et al. 1981) occupies the low elevation zones of marshes on the northeast coast of North America. In North American mangroves, the same zones are occupied by *Rhizophora mangle* which have stilt roots highly adaptive to wave action. The middle and upper zones are less frequently inundated and are subjected to desiccation. As such, the species in these zones have developed structures such as dense turf or inwardly rolled leaves to conserve moisture and many species are drought tolerant. On the northeast coast of North America the high marsh platform is typically occupied by *Spartina patens* with *Juncus gerardii* at slightly higher elevations (Mitsch and Gosselink 2007). In North America, *Avicennia germinans* and *Laguncularia racemosa* are found in the upper elevation mangrove zone. These species have pneumatophores or specialized root systems that extend above the soil surface allowing them to respire in highly anaerobic soils. At higher elevation mangrove zones, *Conocarpus erectus* usually dominates; a salt tolerant tree that is sometimes considered a mangrove “associate” (Mitsch and Gosselink 2007).

The persistence of tidal wetlands depends on vertical accretion and inland expansion (Adam 2002; Mitsch and Hernandez 2013; Teal and Teal 1969; Titus and Neumann 2009; Torio and Chmura 2013). Vertical accretion is driven by accumulation of organic matter, primarily through belowground production (Chmura 2013; Darby and Turner 2008), and deposition of sediment (Chmura and Hung 2004; Turner et al. 2004). As the wetland surface builds upward, the wetland edges expand laterally (Feagin et al. 2010). For this to happen, the expansion area should be flat, regularly flooded, relatively sheltered and permeable. In many coastal areas around the world, the intertidal zone has become unsuitable for wetland establishment because of high imperviousness resulting from anthropogenic development. In the face of global change, ensuring the suitability of the future intertidal zones for wetland establishment will be one of the greatest challenges of wetland conservation.

1.3 Ecological functions and ecosystem services

As the ecotone of land and sea (Caffrey et al. 2007; Ewel et al. 2001; Greenberg et al. 2006; Ray 2005; Spurgeon 1998) tidal wetlands link terrestrial and aquatic ecosystems through energy transfer (Turner 2009) thus supporting many interdependent ecological functions (Costanza et al. 1997). High primary production in tidal marshes and mangroves (Kirwan and Megonigal 2013; Mitsch and Gosselink 2007; Tuxen and Kelly 2008) can be exported to coastal waters, where it supports secondary production (Hopkinson Jr 1985; Valiela et al. 2009). The complexity created by the mixture of terrestrial and aquatic components provides habitats to many aquatic and terrestrial organisms at various stages in their life cycles, many of which are endemics (Greenberg et al. 2006). Wetlands improve the quality of coastal waters by trapping toxic substances and uptaking excess nutrients that would otherwise contribute to pollution and eutrophication. Because tidal wetlands provide habitats for fish and crustaceans fisheries are enhanced (Boesch and Turner 1984b; MacKenzie and Dionne 2008).

The structural components and qualities of tidal wetlands provide valuable ecosystem services (Chmura et al. 2003; Koch et al. 2009; Mudd et al. 2009a; Peterson et al. 2008). Mangroves and salt marshes reduce the impacts of storm waves and wind minimizing damages to adjacent human communities (Adam 2002; Chmura et al. 2003; Morgan et al. 2009). In many areas, mangroves are used for fuelwood, forage and building materials. Tidal wetlands are the subjects of increasing ecotourism, recreation, research and education (Barbier et al. 2011).

Tidal wetlands provide global benefits, as well. By sequestering and storing atmospheric carbon dioxide, they mitigate global warming (Koch et al. 2009; Mcleod et al. 2011; Mudd et al. 2009a; Peterson et al. 2008). Recent estimates put the carbon sequestration value of salt marshes and mangroves at \$30.50 ha^{-1} yr^{-1} (Duarte et al. 2013; Mcleod et al. 2011). When other ecosystem services are considered, a hectare of tidal wetland has an annual estimated value between US\$10,000 to US\$60,000 (Barbier 2012; Barbier et al. 2011; Costanza et al. 1997; Feagin et al. 2010).

Demands for ecosystem services are likely to increase in the next decades (Abson et al. 2014; Barbier 2012; MEA 2005). With climate change and declining areas of tidal wetlands, those ecosystem services and ecological functions are at risk of being degraded, lost or replaced. Thus, to ensure sustainable provisioning of ecosystem services, it is crucial to understand where, how and why tidal ecosystems are changing in space and time.

1.4 Threats, response and adaptation

Tidal wetlands are among the most threatened ecosystems on the planet. Since 1900, about 63% of coastal wetlands have been lost globally (Davidson 2014). Much of these losses are in Asia (83.7%), Europe (71%), and North America (36.5%) (Davidson 2014). The remaining areas continue to decrease at an annual average rate of 1-3% (Nicholls et al. 1999; Pendleton et al. 2012; Valiela et al. 2001) despite existing wetland protection policies and vigorous conservation programs. Whereas previous losses were attributed to direct threats such as land conversion, reclamation, vegetation disturbance and habitat destruction, it is now recognized that indirect threats also play a major role in wetland degradation. Specifically, threats such as sea level rise, sediment deprivation, pollution and lack of migration space are already affecting wetlands and wetland losses are likely to increase in the future (Day et al. 2011; Reed 1995; Rybczyk et al. 2012). With climate change, it is expected that the impacts of indirect threats would intensify putting remaining tidal wetlands under increasing vulnerability.

Accelerated rise in sea level can degrade tidal wetlands and stress vegetation (Craft et al. 2009; Moorhead and Brinson 1995). If the rate exceeds their vertical accretion rate, tidal wetlands will be subjected to prolonged inundation or hydroperiods (Cahoon et al. 2006; Cahoon et al. 2009; FitzGerald et al. 2008; Morris et al. 2002). When tidal wetland species are inundated for longer hydroperiods they may no longer produce sufficient root mass, further reducing the capacity for soil vertical accretion (Kirwan and Guntenspergen 2012). Furthermore, prolonged hydroperiods may modify soil chemistry driving changes in species composition or migration of vegetation zones (Reed 1995).

Historically, tidal wetlands have adjusted to sea level rise (Cahoon et al. 2006), but this is likely to change under an accelerated rise in sea level. At present, there is no scientific consensus on how wetlands would respond to accelerated sea level rise (Gutierrez et al. 2009). In order to adjust to sea level rise, tidal wetlands need to constantly build their elevation and expand into newly flooded areas. Depending on accretion dynamics and opportunities for migration, tidal wetlands can either 1) keep pace with sea level rise and migrate upland or 2) drown or be eroded (Cahoon et al. 2006; Day et al. 2011; Gilman et al. 2008; Morris et al. 2002). As such, inland migration space is essential for wetland sustainability with sea level rise. Unfortunately, the increasing number of people wanting to live on the coasts has resulted in intensive development of low lying coastal zones (McGranahan et al. 2007; Nicholls and Cazenave 2010) creating highly impervious barriers to wetland migration.

Biophysical and socio-economic barriers to inland migration could reduce suitable habitats or prevent establishment of new wetland vegetation communities (Fujii and Raffaelli 2008; Hughes 2004; Mander et al. 2007; Mazaris et al. 2009; Morris et al. 2004; Pintus et al. 2009; Schlacher et al. 2007). As sea level rises, the presence of such barriers puts tidal wetlands under the state of "coastal squeeze"(Doody 2004). Constructed urban infrastructure near intertidal wetlands (Bulleri and Chapman 2010; Dausse et al. 2008; Schleupner 2008) could modify water flow, sedimentation and dispersal of propagules. Additionally, infrastructure can increase the steepness of the shore inland to the intertidal wetlands. All these contribute to submergence, isolation and eventual fragmentation of intertidal habitats. Our current understanding of coastal squeeze is still limited despite the fact that it plays a major role in wetland adaptation. Although strategies to address coastal squeeze are slowly emerging in many areas, e.g., coastal realignment (Rogers et al. 2014; Rupp-Armstrong and Nicholls 2009), none have yet quantified coastal squeeze.

1.5 Research context and thesis questions

Amidst emerging 21st century threats associated with sea level rise, what is the future for tidal wetlands? How will they persist and their function change with expected rates of sea level rise over this century? In this century, it is expected that both coastal land development and sea level rise will intensify, posing a major threat to tidal wetlands. So far, tidal salt marshes and mangroves have been adjusting to historical sea levels, but how they will adjust to accelerated sea level rise associated with anthropogenic climate change is largely unknown. Future adjustments are likely to be limited by the capacity of the wetlands to accrete, the ability of the vegetation at the seaward edge to tolerate greater hydroperiods, and the suitability of inland areas for wetland migration. With the presence of natural and anthropogenic barriers inland, the capacity of wetlands to adjust to sea level rise and the provision of their ecosystem services are likely to be compromised by coastal squeeze, limiting wetland accommodation space.

Wetlands under severe coastal squeeze are at high risk of being functionally impaired if they cannot migrate inland. With climate change, indirect threats like coastal squeeze will play an increasingly important role in wetland management. By determining the level of coastal squeeze before any investment in conservation, coastal planners can prioritize efforts and rationally allocate limited resources. This allows them to focus on areas with the highest chance of success. The identification of the limits of resilience is one of the important challenges in predicting how wetlands will do with 21st century sea level rise. The studies developed in this thesis provide help to meet that challenge.

1.6 Thesis objectives and structure

The primary objective of this doctoral study is **to better understand -through spatially explicit modelling- indirect threats and their impacts on tidal wetlands.** I focus on quantifying the threat resulting from the synergy of sea level rise and land development and their impacts on habitat provisioning and carbon sequestration. Chapter 1 provides the overall context, thesis questions and the objective of the study. Here, I provide a general perspective on why indirect threats matters to wetland sustainability and present the overall thesis questions. Chapter 2 reviews the state-of-the-science on sea level rise and coastal squeeze, tidal wetlands as fish habitats and carbon sinks, and vulnerability models. From this review, I highlight the major gaps and basic requirements to assess wetlands under climate change and describe the adopted modeling approach. The succeeding chapters are developed as independent research studies to address those gaps. Each of these chapters with the exception of the overall summary and conclusions (i.e., Chapter 7) are written following a manuscript format intended for publication in a peer-reviewed journal.

Coastal squeeze has not been quantified as a continuous environmental gradient. In Chapter 3, I develop a GIS-based inundation model or procedure to predict the elevation change of tidal marshes and coastal areas flooded by rising sea level in select locations using high resolution digital elevation model (DEM) derived from Light Detection and Ranging (LiDAR) data. First, I used the outputs in a sub-model to map the extent of future intertidal and marsh zones. Second, I develop a sub model to formulate an index of ‘coastal squeeze’ by combining the output from the inundation model and imperviousness data from satellite sensor imagery. Coastal squeeze is assumed to vary continuously over the landscape depending on two major variables: slope and imperviousness. The coastal squeeze index therefore mirrors the degree of impedance along the borders of wetlands and within their potential migration areas. High coastal squeeze means high possibility of restriction. I demonstrate the use of the index to rank selected marshes in the Gulf of Maine and Bay of Fundy.

There is a pressing need to assess the level of threats to tidal wetlands to inform national wetland conservation and climate mitigation policies. In Chapter 4, I adapt the coastal squeeze model and equations developed in Chapter 3 to predict and calculate the coastal squeeze threat over North American coastal zones and rank the threat of coastal squeeze to salt marshes and mangroves in Mexico, Canada and the United States. A global DEM and imperviousness data are used as inputs to the coastal squeeze model. The extent over which the coastal squeeze is calculated is the low elevation coastal zone or coastal area ≤30 m elevation.

Sea level rise and coastal squeeze can modify the spatial structure and configuration of tidal wetlands. To what extent these modifications affect their value as habitat for fish has not been assessed. In Chapter 5, I developed an automated and improved version of the inundation model developed in Chapter 3 by adding a vertical accretion component as a feedback from expanding intertidal creeks. The improved inundation model was used to simulate the change in elevation of a marsh under two assumptions of accretion: constant accretion and accretion equals sea level rise rate. The outputs of the models are used in two sub-models to predict the extent of subtidal and intertidal areas (i.e., marsh) at different sea level rise rates under the two assumptions. These projected extents or areas are used in a landscape ecology analysis to determine quantity, quality and connectivity relative to the habitat requirements of two marsh-dependent fishes.

Tidal wetlands sequester and store huge amounts of carbon. With sea level rise, this function can increase or decrease depending on the limits and opportunities for inland migration. How the three-dimensional properties of tidal wetlands change with time as they relate to carbon sequestration under different sea level rise rates and trends has not been investigated. Chapter 6 deals with an in-depth analysis of the different sea level rise scenarios and trends (i.e., linear versus non-linear) and asses their impacts on carbon storage function of a selected marsh. The improved version of the inundation model developed in Chapter 5 was used to derive yearly marsh elevation change under each sea level rise trend and scenario. The outputs are then used as inputs to a submodel that predicts optimum belowground production and carbon from area and elevation.

Finally, an overall summary, conclusions, limitations and future research direction is provided in Chapter 7.

Chapter 2 Background and literature review

2.1 Sea level rise and the coastal squeeze threat

Since 1900, global sea level has been rising at an average rate of 1.7 mm yr^{-1} (Church et al. 2013a; Wong et al. 2014). However it is very likely that this rate will be exceeded in the next 100 years if greenhouse gas emissions, global temperature, and melting of the polar ice caps continue to increase (Church et al. 2013b; Jevrejeva et al. 2012; Vermeer and Rahmstorf 2009). The Intergovernmental Panel on Climate Change (IPCC) reported that a maximum of 1 m rise in global mean sea level at the end of this century is highly likely (Alexander et al. 2013). On the US coast, Parris et al. (2012) warn against a 2 m sea level rise. Some wetlands can probably survive accelerated sea level rise (Day et al. 2011; Morris et al. 2002; Rybczyk and Cahoon 2002; Rybczyk et al. 2012), but most would probably drown and disintegrate. Because the projected rates of sea level rise are highly variable indices of accelerated climate change, the permanence of intertidal wetlands in changing coastal zones remains uncertain. A minimum sea level rise of 0.6 m by the end of the century (Bindoff 2007; Nerem et al. 2010; Pfeffer et al. 2008; Vermeer and Rahmstorf 2009) will likely drive efforts to protect valuable public and private lands (Feagin et al. 2010) immediately inland of tidal wetlands.

Protection of properties from sea level rise could put tidal wetlands in a state of increasing "coastal squeeze" (Doody 2004; Schleupner 2008; Torio and Chmura 2013) preventing tidal wetlands from migrating inland, a process which would otherwise alleviate some of the loss of wetland area through sea level rise. Tidal wetlands facing coastal squeeze may become more degraded and functionally impaired.

As climate warming causes accelerated rates of sea level rise and development of coastal land intensifies, the sustainability of tidal wetlands decreases (Nicholls et al. 2007). In the past, tidal wetlands vertically accumulated soil in pace with rising sea level, and tidal wetlands migrated inland as sea level rose, e.g., Shaw and Ceman (1999). The accelerated rates of sea level rise accompanying anthropogenic climate change are likely to increase the frequency and duration of flooding beyond the tolerance of the vegetation, which is largely responsible for soil accumulation, e.g., Cahoon et al. (2006) and FitzGerald et al. (2008). As a result, the seaward edge of many wetlands is likely to retreat. At the same time, development of coastal regions and steep gradients in some locations will block migration of tidal wetlands inland [e.g., (Feagin et al. 2010; Gilman et al. 2007)], placing them in what Doody (2004) has termed a "coastal squeeze." This means loss of ecosystem services that tidal wetlands provide, such as buffers to erosion and storm flooding (Anthoff et al. 2010; Jolicoeur and O'Carroll 2007; Schleupner 2008; Sterr 2008), carbon storage [e.g., (Mcleod et al. 2011)], and subsidies to coastal fisheries (Boesch and Turner 1984a). Coastal squeeze might also increase fragmentation of tidal wetlands, reducing their value as habitats (Bulleri and Chapman 2010; Chmura et al. 2012; Mazaris et al. 2009). Multiple studies have mapped the vulnerability of coastal lands to submergence [e.g., (Demirkesen et al. 2008; Gornitz et al. 1994; Vafeidis et al. 2008), but few have considered the risk of coastal squeeze.

Coastal squeeze arises from a combination of factors. Anthropogenic barriers prevent wetlands from migrating inland and steep slopes bordering wetlands stall or completely halt wetland migration (Brinson et al. 1995). Pavement contributes to coastal squeeze by resisting plant colonization (Lu and Weng 2006). I am unaware of any studies that have established the critical slope or imperviousness that prevents marshes from migrating inland, but in a study of rocky coasts, Vaselli et al. (2008) defined steep substrata as those areas with a greater than 40º slope.

Coastal squeeze and its impacts are spatially and temporally variable. On the other hand, existing coastal squeeze are limited in methodology and scale (Doody 2004; Schleupner 2008) to capture these variability. For example, Schleupner (2008) used expert judgment to class coastal squeeze in three categories: low, medium, and high. Rocchini (2008) warns against the use of categorization in mapping landscape patterns because it tends to loose information when data are truncated and aggregated into finite classes. Furthermore, categorical maps offer limited information when used in quantitative analysis (e.g., statistical correlation). As the categorization procedure is very much expert driven, the class boundaries can change depending on who is doing the classification.

In mapping coastal squeeze, the magnitude of the threat could be more important than the threat classes. Also, the factors causing coastal squeeze such as slope and imperviousness vary spatially, temporally and continuously over the landscape. Following these properties, representing coastal squeeze as a continuous gradient rather than discrete classes would be more realistic.

Fuzzy systems (Zadeh 1965) provide an alternative non-discrete mapping logic in a model that integrates the properties of the coastal squeeze variables and produce an index with continuous values. Many studies mapping continuous spatial variations have relied on fuzzy systems because of their robustness. In environmental studies, fuzzy systems have been successfully adopted in mapping qualitative and continuous indicators of climate change (Hall et al. 2007), land suitability (Joss et al. 2008), and environmental risk [e.g., (Mistri et al. 2008; Vrana and Aly 2011)]. Fuzzy logic overcomes the limitations of discrete mapping rules by allowing different degrees of class membership (Hanson et al. 2010). Rather than assigning absolute membership, fuzzy logic uses partial membership in rating variables. Additionally, a fuzzy system is more flexible than categorical ranking when class membership or class boundaries cannot be defined exactly (Burrough and McDonnel 1999). By using partial memberships, conceptual uncertainty associated with coastal squeeze is accounted for in the resulting maps. Unlike categorical maps that are subject to change depending on the expertise, fuzzy logic equations, once developed, can be applied consistently.

Whereas approaches to quantify coastal squeeze are generalized, current discussions to mitigate coastal squeeze impacts are localized (Baily and Pearson 2007; Rogers et al. 2014; Rupp-Armstrong and Nicholls 2009). With the emergence of national policies and strategies for mitigating climate change and biodiversity loss in coastal areas [e.g., United Nation Framework Convention on Climate Change-Nationally Appropriate Mitigation Actions (UNFCCC 2015), Aichi Biodiversity Target 11(CBD 2015), biotic wetland connectivity (Amezaga et al. 2002)] a continent-wide study is needed (Erwin 2009) to assess the overall threat of coastal squeeze to inform such policies and international restoration programs.

2.2 Tidal wetlands ecosystem services

Studies show that demand for ecosystem services is likely to increase in the next decades and in particular, the services provided by coastal systems (Abson et al. 2014; Barbier 2012; MEA 2005). However, the continued ability of tidal wetlands to provide ecosystem functions and services is uncertain. Service provision depends heavily on the quality, stability and resilience of their structural components (Boyd and Banzhaf 2007; Fisher et al. 2008; Fisher et al. 2009; Wallace 2007) and these can be compromised by accelerated sea level rise and coastal squeeze. Ecosystem services assessment and valuation are top priorities in research and conservation (Bennett et al. 2009; Kremen 2005). Despite these, and because of several uncertainties, there are no consistent indicators or metrics of ecosystem services change (Kremen 2005) especially for tidal wetlands. In addition, monitoring of the state of tidal wetland ecosystem services has not been fully integrated in management (Wallace 2007).

2.2.1 Tidal wetlands as fish habitats

Development of a spatial modelling approach brings novel perspectives to our understanding of the response of tidal wetlands to global sea level rise. Whereas quantifying the threat of coastal squeeze provides vital information about the permanence of tidal wetlands with sea level rise, landscape scale properties of the wetland surface could provide quantitative indicators of the quality and quantity of ecosystem services. As such, complementing vulnerability assessment with landscape ecology analysis of tidal wetland change will improve knowledge of wetland adaptation and increase the confidence of decision makers in prioritizing conservation and restoration programs. A demonstration of this is modeling the value of the marsh as fish habitat. Fish are highly dependent upon channels and access from marsh edges to feed, thus are sensitive to changes in marsh configuration and fragmentation expected with sea level rise.

Since 1980, ~48% of the world's tidal wetland area has been lost (Coleman et al. 2008; Lotze et al. 2006; Valiela et al. 2001; Waycott et al. 2009), resulting in a ~69% reduction in the habitat nursery and 63% in the filtering services provided by tidal wetlands (Worm et al. 2006). The rapid increase in sea level rise associated with global warming is expected to further reduce tidal wetland area and its ecological functions (MEA 2005). In addition, urban sprawl along the coast poses a constant threat to the already diminishing tidal wetlands.

The impact of sea level rise on tidal wetlands has been examined with respect to changing marsh area (Craft et al. 2008; Rogers et al. 2014; Traill et al. 2011) and shifts in marsh vegetation (Craft et al. 2008) as well as corresponding shifts in habitat for birds (Brittain and Craft 2012) and mammals (Traill et al. 2011). For example, Craft et al. (2008) developed the Sea Level Rise Affecting Marshes Model (SLAMM) to predict shifts in salinity zones of tidal wetlands and predicted that at the end of this century about 20 to 45% of the total salt marsh area will be lost if sea level rises to the mean and maximum rates projected by IPCC, respectively. They claim that this will significantly reduce biomass production and nitrogen absorption. In subtropical tidal wetlands, Traill et al. (2011) predicted that decreasing marsh area with sea level rise would increase the threat of extirpation and predation risk of a native rodent *Xeromys myoides*. In the USA, a model by Brittain and Craft (2012) indicated that shifting coastal habitats with sea level rise may reduce suitable habitats and populations of coastal birds. These studies relate the loss of ecosystem services mainly with decreasing areas of tidal wetlands. However, I am not aware of any studies that have addressed the impacts of sea level rise and on marsh configuration, particularly as it applies to fish habitat.

Rising sea levels are likely to affect tidal wetland habitat in other ways which have yet to be effectively explored in spatial models. For instance, sea level rise and land development may modify the structure, configuration, spatial distribution, and accessibility of wetlands to fauna. Such landscape-scale modifications may have far-reaching consequences that are just as important as those caused by reduction in area. Earlier works by Dionne and colleagues (Dionne and Dochtermann 2006; Dionne et al. 2006; Dionne et al. 1999) at Wells marsh in Maine (USA) demonstrated that land development surrounding marshes can have positive and negative impacts on fish density and biomass. These impacts are likely to intensify with time and sea level rise. The initial works of Dionne and colleagues have not been followed through using a landscape approach. In other areas, tidal marshes dynamically respond to prolonged hydroperiods from wind and wave action by migrating inland even if there is no sea level rise (Wasson et al. 2013). This demonstrates that as an ecotone, tidal marshes are sensitive to environmental changes. Addition of other stressors such as coastal squeeze (Torio and Chmura 2013), climate warming (Pachauri and Reisinger 2008), and nutrient enrichment (Deegan et al. 2012) will therefore compound the impact of sea level rise on tidal wetland ecological functions.

The functions of a tidal wetland are intricately linked with its spatial and structural characteristics. For example, marshes provide food and refugia in patches of different sizes, location, quantity, and quality. They sustain predator and prey relationships (Boesch and Turner 1984a; Halpin 2000) and link estuaries to open coastal waters (Kneib and Wagner 1994; Peterson and Turner 1994) through exports of secondary production. Species endemic to the marshes use the marsh interior and edges (Greenberg et al. 2006). Proximity and connectivity of patches of emergent vegetation to other intertidal or subtidal environments increase density and biomass of shrimp and fish (Irlandi and Crawford 1997; Pittman and McAlpine 2003) and sustain other species that use the system at some point in their life stages (Kocik and Ferreri 1998). Complexity in patch shape and configuration of wetland edges relate to the quality of wetland refugia and its capacity to sustain interactions between transient and resident species (McGrath 2005; Minello et al. 1994; Peterson and Turner 1994). As in terrestrial ecosystems (Fisher et al. 2011; Holling 1992), the diversity in size of habitat patches may be used by different organisms with different body sizes. To better appreciate its ecological function, we must understand the structural limits of tidal wetlands and its subhabitats (Able et al. 2012; Rountree and Able 2007) and how they are spatially and temporally organized.

Because the impacts of sea level rise and coastal squeeze on tidal wetlands manifest at various scales, it is necessary to adopt a technique that can be used to assess both system-wide and local spatial patterns. A landscape ecology approach (Turner 1989) provides the means to evaluate how spatial configuration and connectivity (Beger et al. 2010a; Beger et al. 2010b; Berkström et al. 2012; Lindenmayer et al. 2008; Sheaves 2009) can be modified by coastal squeeze and sea level rise. Landscape level analyses help us to determine changes in spatial patterns, how habitats are organized, and how their changes affect ecological functions such as provision of food, refugia, and corridors of movement within those habitats (Morris 2012; Turner 1989).

Several landscape metrics are available to quantify spatial patterns and connectivity [(McGarigal and Marks 1995), Table 2-1] . For example, the fractal dimension reveals the shape complexity of a wetland patch, which in turn, affects the ability of organisms to disperse and interact (Forman 1995). With respect to wetlands, a decreasing fractal dimension indicates loss of shape complexity which is usually associated with a landscape heavily modified by humans (O'Neill et al. 1988). In the terrestrial domain, indices of interspersion and edge density have been used to differentiate patterns of land cover patchiness in urbanizing landscapes (Cifaldi et al. 2004). On tidal wetlands, interspersion and edge density could be used to quantify the changes in marsh patch spatial distribution and refuge potential, respectively. Patch carrying capacity or the “ecologically scaled landscape index” (ESLI) is a derivative of patch size which can be used to determine the minimum habitat requirement of a species (Vos et al. 2001). Connectivity metrics such as least cost distance and circuit theory (McRae et al. 2008) have been used to measure landscape “permeability” enabling the identification of priority conservation corridors and sections of the landscape where movement of plants and animals are severely restricted (Beier et al. 2011; Brost and Beier 2011; Carroll et al. 2012). Studies of terrestrial ecosystems show that when habitats are connected and the landscape is permeable, biodiversity at all levels is enhanced (Fahrig et al. 1995; Tischendorf and Fahrig 2000). The application of the landscape ecology approach in estuarine and tidal wetland systems is still in its infancy, and to my knowledge, no studies have used the landscape approach to understand the impacts of coastal squeeze and sea level rise on tidal marshes.

2.2.2 Tidal wetlands as carbon sink

The permanence of tidal wetlands and their functions depend on the stability of their spatial structures. Tidal wetlands are not only important as habitats but also as carbon dioxide sinks that can mitigate or exacerbate climate change (Doughty et al. 2015; Pendleton et al. 2012). In particular, the substrate of mangroves and salt marshes holds huge amounts of carbon (Mcleod et al. 2011). Carbon storage in tidal wetlands is intricately linked to below ground production and accretion. As the wetlands spread laterally and grow vertically, more carbon is stored. This active carbon sink can increase or decrease with sea level rise. With accelerated rates of sea level rise the coastal carbon sink can be lost (Chmura 2013). The carbon stored in their substrate could be released back to the atmosphere. Because of this, carbon sequestration and re-emission have received considerable attention in climate research and mitigation policies in the past five years (Donato et al. 2011; Fourqurean et al. 2012; Mcleod et al. 2011; Pendleton et al. 2012).

The historical response of tidal wetlands has been to expand inland and even seaward as sea level rose [e.g., Redfield (1965)]. Lower rates of sea level rise worked in favor of tidal wetland area and presumably the ecosystem services they provide. Higher rates of sea level rises and modification of the coastal landscape are likely to limit wetland expansion with negative consequences on indirect ecosystem services from tidal wetlands such as primary production and sequestration of carbon dioxide. Decrease in primary production would mean decrease in the value of tidal wetlands as carbon sink as the high amount of carbon stored in tidal wetlands is driven by above and belowground production (Armentano and Menges 1986; Duarte et al. 2013; Mcleod et al. 2011). Marshes and mangroves adjust to sea level rise by building their elevation and in the process, produce root biomass. Through time, the roots are buried in the sediments. As the wetlands accrete and expand, their soils accumulate which, locking up carbon in the soil. If wetlands can adapt to sea level rise, expand and remain intact, the storage function is sustained and the resilience of the tidal wetland as a carbon sink is maintained.

Results of recent wetland carbon modelling studies are conflicting. For instance, Kirwan and Mudd (2012) and Kirwan et al. (2010) project that carbon sequestration in salt marshes will increase in the first half of the century and then slightly decrease in the second half when the rate of sea level rise accelerates and drowns marshes. They report that this shift occurs when sea level rise exceeds 1 m (or a rate 10 mm yr^{-1}) resulting in less productive marshes. The Marsh Equilibrium Model (MEM) output by Morris et al. (2012) makes a more dire prediction that carbon sequestration will decrease at less than a 1 m rise in sea level. These studies however did not account for upland expansion or the variability of future sea level rise trends and the topographically heterogeneous marsh surface. Models that do not account for seaward loss or inland expansion are likely to underestimate wetland response, and hence, carbon sequestration.

Conserving tidal marshes as carbon sinks requires understanding the likely consequences of different sea level rise trends and the barriers of marsh sustainability. Under what sea level rise trend and landscape circumstances could the marsh tip from being an effective carbon sink to a potential carbon source? It is likely that wetland response would vary depending on the characteristics of the landscape. The need to track the changes of a marsh with sea level rise requires robust spatially and temporally explicit models that can be easily parameterized with basic information.

2.3 Models of marsh vulnerability

At present there are about 15 geomorphic models of tidal wetland evolution and vulnerability (Table 2-2). These models are classified as point models, spatial temporal numerical and landscape models, and spatial temporal GIS models. Point models could simulate similar processes as spatial temporal models but differ in assumptions when it comes to spatial variations. Point models [e.g., Chmura et al., 1992, Allen, 1994, Woolnough et al., 1995, Rybzyk and Cahoon 2002, Mudd et al., 2009 and Morris et al., 2012] assume the wetland surface as a single point or cell with uniform or averaged spatial characteristics. Spatio-temporal landscape, numerical and GIS-based models [e.g., Rinaldo 1999, Reyes et al., 2000, Temmerman et al., 2003 & 2004, D'Alpaos et al., 2007, Kirwan and Murray 2007, Hinkel and Klein 2009, Craft et al., 2009, and Strauss et al., 2012] assume a spatially heterogeneous marsh surface. Point models can vary in complexity from simplistic representation of geomorphic processes like sediment accumulation and transport to complex mechanisms of wetland response to inundation, inorganic and organic processes and carbon accumulation, making point models useful in predicting localized changes in a tidal wetland (Mcleod et al. 2010). But because they are not spatially explicit, results from point models are general. There are few attempts to use outputs generated by point models in a spatially explicit setting [e.g., (Schile et al. 2014)]. Even so, point models are far from being a complete and standalone spatially explicit wetland model.

Among the spatial and temporally explicit models, the numerical models are the most complex because they tend to include complex hydrodynamic and biophysical process. Data and expert-driven, these models are expensive to maintain and require dedicated computing infrastructure and programming skills. In addition, the codes of these models are not open source. Although numerical models are useful in research they are far from being useful in management and conservation especially when conservation agencies have tight budgets, limited expertise and work in short timelines (Mcleod et al. 2010). In many conservation initiatives, GIS-based spatio-temporal models [e.g., SLAMM (Craft et al. 2009) add Strauss et al., 2012) are gaining more attention despite their current limitations.

As noted by Fagherazzi et al. (2012) in a recent review, existing spatially explicit models cannot predict whether upland expansion could compensate for the loss at seaward edge and vertical submergence of the wetland surface. Aside from being complicated and expensive, they have limited flexibility with variable input data. Existing models also have limited temporality. For example, SLAMM simulates in large pre-determined time steps of 5 or 25 years (Mcleod et al. 2010; Reyes et al. 2012). When detecting the temporal evolution of important events in tidal wetlands (e.g., surface expansion, primary production tipping points, and creek expansion) models with a flexible and finer temporal resolution (e.g., annual) are more useful. Finally, existing GIS-based spatio-temporal models have limited capability to quantify the importance of expanding channel networks and its feedback to sediment delivery to the marsh interior. This information is crucial in estimating optimum production linked to habitat quality and carbon sequestration.

2.4 Modelling approach

Whether tidal wetlands will survive the 21st century sea level rise is a major conservation and management concern. While it is generally presumed that tidal wetlands will survive if they can build vertically at pace with sea level rise and that there are no bio-physical and socio-economic barriers to inland migration (Feagin et al. 2010; Kirwan and Megonigal 2013; Nicholls et al. 2007), the intertidal landscape is spatially heterogeneous such that magnitude of threats and their potential impacts may vary. Consequently, certain wetlands are more threatened than others and the suitability of their future migration space is not uniform. At present, it is hard to tell which tidal wetlands have the greatest chance of future survival without spatially explicit predictive models.

Because of the functional limitations and complexities of re-engineering current spatio-temporal numerical models (Mcleod et al. 2010), this thesis develops and implements a robust and flexible model in a GIS framework. The modelling approach starts with the development of a simplistic GIS-based model and gradually builds it, incorporating different assumptions of projected rates of sea level rise, feedback mechanisms and basic geomorphic processes such as accretion and belowground production. First, I develop a simple spatially explicit inundation model or procedure to simulate elevation changes relative to a sea level rise projection. This method is a modification of existing inundation modelling methods [e.g., (NOAA 2012a; Poulter and Halpin 2008; Strauss et al. 2012; Weiss et al. 2011)]. The inundation model uses high spatial and vertical resolution elevation data. Then two submodels were built; one that uses the output from the inundation model to determine the extent of marsh and flooded areas and the other to calculate a coastal squeeze index based upon slope and imperviousness using fuzzy logic (Zadeh 1965). Second, I adapt the coastal squeeze equations and models to use coarse resolution elevation and imperviousness data to quantify and rank the threat of coastal squeeze to mangroves and salt marshes of North America. Third, the simplistic inundation model was modified to include an accretion feedback component based upon distance of the marsh surface to the nearest intertidal creek (Chmura and Hung 2004) and designed to run iteratively at a yearly time-step. The accretion feedback component simulates the temporal changes in the width and distance of intertidal creeks and calculates accretion rates based on these variables. This component addresses a critical missing functionality in existing GIS-based and numerical model mentioned by Fagherazzi et al. (2012), Mcleod et al. (2010) and Reyes et al. (2012). Additionally, the new model is developed to address other limitations of existing models by simulating the spatial extent and distribution of inland expansion areas, testing different trends and rates of sea level rise, simulation at desired time-steps, and using variable input data. Fourth, submodels that use the outputs of the inundation model were developed to answer specific questions about ecosystem services change. In the study of fish habitat, variants of extent submodel were built to simulate the extent of the marsh and subtidal areas. The outputs from these two sub-models were used in a landscape ecology analysis to assess habitat fragmentation and connectivity. In the study of carbon sequestration, the inundation model is coupled with an optimum productivity model based upon Kirwan and Guntenspergen (2012). This submodel translates the net elevation to belowground primary production and eventually to carbon using a conversion developed by Gallagher and Plumley (1979) and Roman and Daiber (1984).

Table 2-1 List, description and value range of selected landscape ecology metrics

Landscape Metrics	Description	Range of Values	Reference
Composition Index			
POMA	% Original Marsh Area	0<POMA<max	Computed manually
MPS	Mean Patch Size	A_{min}< MPS <MA_{total}	(McGarigal and Marks 1995)
PSSD	Patch Size Standard Deviation measures the absolute variation in patch sizes	0<PSSD<max	(McGarigal and Marks 1995)
Ecologically Scaled Landscape Index (ESLI)	Ratio between the patch size and the size of the habitat requirement of a species (K) averaged for all the patches	0 < Kavg< Kmax	(Vos et al. 2001), computed manually
Configuration index			
Mean Patch Edge Density (MPE)	Average amount of edge per patch, where edge is the perimeter of patches divided number of patches of a particular class.	MPE > 0	(McGarigal and Marks 1995)
Mean Patch Fractal Dimension (MPFD)	Measure of shape complexity as deviation from simple Euclidian shapes. Mean fractal dimension approaches 1 for shapes with simple perimeters and approaches 2 when shapes are more complex.	1<AWMPFD<2	(McGarigal and Marks 1995)

Table 2-1 continuation…

Landscape Metrics	Description	Range of Values	Reference
		Connectivity index	
Mean Nearest Neighbour (MNN)	Measures the average shortest edge to edge distance between similar patches	0<MNN<MNNmax	(McGarigal and Marks 1995)
Interspersion & Juxtaposition Index (IJI)	Measures patch adjacency of classes as a function of edge length and the number of classes The value approaches 0 when the classes are clumped and 100 when classes are equally distributed and adjacent to each other meaning that each class shares a common border with all other classes.	0≤PC≤100	(McGarigal and Marks 1995); http://ec.europa.eu/agriculture/publi/landscape/ch1.htm
Resistance Index	The index predicts the pattern and the degree of resistance of movement on a landscape with varying obstacles. I calculated the index based on the principles of least-cost modelling and Circuit Theory.	Resistance Index > 0	(McRae et al. 2008) (Carroll et al. 2012)

Table 2-2 Models of tidal wetland processes

Model Reference	Simulation software	Approach	Output or processes modeled
Chmura et al. (1992)	Stella (Beta)	Point model	Sediment accumulation
Allen (1994)	Unknown	Point model	Sediment transport
Woolnough et al. (1995)	Unknown	Point model	Sediment transport and thickness
Rinaldo et al. (1999)	Unknown	Spatial and temporal	Flood discharge in tidal creek networks
Reyes et al. (2000)	Unknown	Spatial and temporal	Wetland elevation, hydrodynamic, soil, productivity and habitat change processes
Rybczyk and Cahoon (2002)	Stella	Point Model	Salt marsh elevation and sea level rise inundation
Temmerman et al. (2003)	Matlab	Spatial and temporal	Marsh vertical accretion and suspended sediment concentration
Temmerman et al. (2004)	Matlab	Spatial and temporal	Sedimentation and elevation change of tidal marsh creeks and natural levees
D'Alpaos et al. (2007)	Unknown	Spatial and temporal based on Rinaldo 1999	Long-term evolution of marsh platform and tidal networks
Kirwan and Murray (2007)	Unknown	Spatial and temporal model	Tidal marsh and channel network evolution
Hinkel and Klein (2009)	DIVA (Stand-alone proprietary software)	Spatial and temporal	Biophysical and socio-economic impacts of sea level rise
Mudd et al. (2009b)	Unknown	Point model	Salt marsh sedimentation, above- and belowground organic processes , growth and decay of organic carbon

Table 2-2 continuation…

Model Reference	**Simulation software**	**Approach**	**Output or processes modeled**
Craft et al. (2009)	Sea Level Affecting Marshes Model (Windows executable)	Spatial and temporal model	Sea level rise, marsh accretion, below-ground production
Morris et al. (2012)	Marsh Equilibrium Model (Web-based)	Point model	Sea level rise, marsh accretion, below-ground production, carbon sequestration
Strauss et al. (2012)	GIS-based	Spatial and temporal	Sea level rise

Chapter 3 Assessing coastal squeeze of tidal wetlands

3.1 Preface

3.1.1 Manuscript detail

This manuscript is co-authored by Gail Chmura. Torio, D.D. and G.L. Chmura. 2013. Assessing coastal squeeze of tidal wetlands. *Journal of Coastal Research* 29:1049-1061.

3.1.2 Context and link to the subsequent chapter

Global sea level is increasing, threatening tidal wetlands worldwide. There are many indirect threats to tidal wetlands associated with sea level rise and human activities but the most emergent and in need of immediate management action is coastal squeeze. The spatially explicit nature of coastal squeeze requires spatially explicit models. However, the constraints posed by input data, the need to capture multi-scale coastal processes and identify priority areas, require that modelling approach needs to be flexible. Spatial and temporal models of wetland vulnerability are crucial in predicting the fate of marshes and mangroves: a critical knowledge for decision makers. Spatially explicit modelling of sea level rise and coastal squeeze addresses the overarching research question: what is the future for tidal wetlands?

This chapter develops an initial model (i.e., inundation model) that simulates marsh elevation change relative to different sea levels in year 2100, and submodels (i.e., 'Extent' or 'mask' model and Coastal Squeeze model) to use the output of the inundation model to predict the extent of future salt marsh and intertidal areas and calculate an index of coastal squeeze, respectively. The coastal squeeze submodel converts slope and imperviousness values into coastal squeeze indices and combine the results into a unified index of coastal squeeze. Also in this chapter, I present an alternative approach based on fuzzy logic to represent coastal squeeze as continuous environmental gradient. Equations to quantify coastal squeeze as indices were developed based upon slope and percent imperviousness. The index is used to rank the threat of coastal squeeze to selected marshes in the Gulf of Maine and upper Bay of Fundy. This chapter aims to compare overall threats amongst wetlands and enable a reserve such as one at Wells, Maine, to manage adjacent uplands to address sea level rise and coastal squeeze. In the next chapter, I adapt the coastal squeeze equations and model to develop continent-wide coastal squeeze indices and rank the threat of coastal squeeze to salt marsh and mangroves of North America.

3.2 Introduction

The coastal landscape is where rising sea levels and expanding land development clash often in the region of tidal wetlands. As climate warming causes accelerated rates of sea level rise and development of coastal land intensifies, the sustainability of tidal wetlands decreases (Nicholls et al. 2007). In the past, tidal wetlands vertically accumulated soil in pace with rising sea level, and tidal wetlands migrated inland as sea level rose [e.g., (Shaw and Ceman 1999)]). The accelerated rates of sea level rise accompanying anthropogenic climate change are likely to increase the frequency and duration of flooding beyond the tolerance of the vegetation, which is largely responsible for soil accumulation [e.g., (Cahoon et al. 2006; FitzGerald et al. 2008)]. As a result, the seaward edge of many wetlands is likely to retreat. At the same time, development of coastal regions and steep gradients in some locations will block migration of tidal wetlands inland [e.g., (Feagin et al. 2010; Gilman et al. 2007)], placing them in what Doody (2004) has termed a "coastal squeeze." This means loss of ecosystem services that tidal wetlands provide, such as buffers to erosion and storm flooding (Anthoff et al. 2010; Jolicoeur and O'Carroll 2007; Schleupner 2008; Sterr 2008), carbon storage [e.g., (Mcleod et al. 2011)], and subsidies to coastal fisheries (Boesch and Turner 1984a). Coastal squeeze might also increase fragmentation of tidal wetlands, reducing their value as habitats (Bulleri and Chapman 2010; Chmura et al. 2012; Mazaris et al. 2009).

Coastal squeeze arises from a combination of factors. Anthropogenic barriers prevent wetlands from migrating inland and steep slopes bordering wetlands stall or completely halt wetland migration (Brinson et al. 1995). I am unaware of any studies that have established the critical slope that prevents marshes from migrating inland, but in a study of rocky coasts, Vaselli et al. (2008) defined steep substrata as those areas with a greater than 40º slope. Pavement contributes to coastal squeeze by resisting plant colonization (Lu and Weng 2006).

Multiple studies have mapped the vulnerability of coastal lands to submergence [e.g., (Demirkesen et al. 2008; Gornitz et al. 1994; Vafeidis et al. 2008), but few have considered the risk of coastal squeeze. An exception is Schleupner (2008), though her approach differed from the one I take here, as she used a probabilistic classification system in her assessment of the coast of Martinique. Specifically, Schleupner (2008) used expert judgment to relate vegetation, sediment budgets, and migration opportunity to the vulnerability of beaches and mangroves to coastal squeeze under three different sea level rise scenarios. The resulting vulnerability measures are three discrete classes of low, medium, and high levels of coastal squeeze. Such assessments are valuable but require site-specific knowledge and local expertise to identify the class boundaries. As such, the method developed by Schleupner (2008) is not easily transferable to other coasts.

Rocchini (2008) warns the use of Boolean logic in mapping landscape patterns because of the tendency to loss information when data are aggregated and reduced into finite classes. Being in such general classes, Boolean maps might contain less information. Additionally, the resulting map could over or underestimate the size of area under threat depending on the method used to decide class the class boundaries. As such, estimates using categorical maps could be less certain and transparent if the class boundaries are not explicitly stated.

In mapping coastal squeeze, the magnitude of the threat could be more important than the threat classes. Also, the factors causing coastal squeeze such as slope and imperviousness vary spatially, temporally and continuously over the landscape. Following these properties, representing coastal squeeze as a continuous gradient rather than discrete classes would be more realistic. Recent studies using Bayesian Network (Gutierrez et al. 2011) and simulation models with decision trees such as SLAMM (Craft et al. 2009) have quantified similar indices (e.g., Coastal Vulnerability or Sensitivity Index); however, these models tend to be complex, data driven, and computationally demanding, whereas categorical ranking [e.g., (Abuodha and Woodroffe 2010; Schleupner 2008)] often neglects uncertainties.

Fuzzy systems (Zadeh 1965) provide an alternate non-discrete mapping logic in a model that integrates the properties of the coastal squeeze variables and produces an index with continuous values. Many studies in mapping continuous spatial variations have relied on fuzzy systems because of their robustness. In environmental studies, fuzzy systems have been successfully adopted in mapping qualitative and continuous indicators of climate change (Hall et al. 2007), land suitability (Joss et al. 2008), and environmental risk [e.g., (Mistri et al. 2008; Vrana and Aly 2011)]. Fuzzy logic overcomes the limitations of discrete mapping rules by allowing different degrees of class membership (Hanson et al. 2010). Rather than assigning absolute membership, fuzzy logic uses partial membership in rating variables. A fuzzy system is more flexible than categorical ranking when class membership or class boundaries cannot be defined exactly (Burrough and McDonnel 1999). By using partial memberships, conceptual uncertainty associated with coastal squeeze is accounted for in the resulting maps.

In this paper, I describe an index that can be used to map the threat of coastal squeeze along the inland border of any tidal wetland, including mangrove swamp, tidal fresh marsh, or tidal salt marsh. My method examines current and future tidal floodplains associated with incremental increases in sea level. Using fuzzy sets I assign to slope and imperviousness standardized, continuous values that correspond to their potential to cause coastal squeeze. Depending on the requirement of the application the index generated using this technique can be represented either as a continuous field or discrete classes.

I employ three case studies to develop the index. Although the examples are based on tidal salt marshes, the techniques I use and the index I developed are valid for all tidal wetlands, including mangroves and tidal freshwater marshes. To assess the relative threats of coastal squeeze, I use fuzzy membership functions to weight the degree to which slope and imperviousness (the proxy for anthropogenic barriers) contribute to coastal squeeze. I combine the results into one index, the Coastal Squeeze Index (CSI), which reflects the magnitude and location of the threat of coastal squeeze with rising sea levels. Using the coastal squeeze index, I determine the portions of current and future marsh areas threatened by squeeze and the factor(s) contributing to the threat for each marsh.

3.3 Methods

3.3.1 Study sites and datasets

I selected three marsh systems that vary in land use, topography, and tidal amplitude and for which appropriate remote sensing imagery and elevation data were available (Figure 3-1). One site is a complex of marshes within the U.S. National Estuarine Research Reserve (NERR) marsh in Wells, Maine, and a town with a population of 9,400 and an average housing density of 56 km^{-2}. To the seaward side, the marshes are sheltered by a sand barrier that separates bordering Webhannet Lagoon from the Gulf of Maine. Farms with gentle to moderate slopes border the northern areas of the marshes, and several houses and minor roads on gentle slopes border the southern areas. At Wells, the range between the highest and lowest astronomical tide is 4.22 m (NOAA 2012a). The low and high marshes consist of *Spartina alterniflora* and *Spartina patens* as dominant species, respectively. Using information provided by Boumans et al. (2002), I determine that the elevation of marsh vegetation ranges from -0.02 m below mean sea level (MSL; i.e., the North American Vertical

Datum of 1988 [NAVD88]) to 1.95 m above MSL (NAVD88).

The second site is a fringing marsh (narrow and elongated strip) along the coast of the Falmouth Estuary near the City of Portland, Maine. Portland has a population of more than 500,000 and an average housing density of 587 houses km^{-2}. The marshes in Portland are surrounded by moderate to steep slopes and subjected to a variety of land uses, such as housing, a multilane interstate highway (I-295), commercial development, and agriculture. Portland has an astronomical tide range of 4.29 m (NOAA 2012a), with marsh vegetation dominated by *S. alterniflora* and a marsh elevation range (utilizing the relationships reported for the Wells marshes) of 0.19 to 2.16 above MSL (NAVD88). The third site is located in Kouchibouguac National Park (KNP) marsh on the coast of the Gulf of St. Lawrence in New Brunswick, Canada, where tides are micro-tidal. The marsh is located on a lagoon protected by a sand barrier. Uplands bordering the marsh are gentle, forested slopes. The dominant marsh vegetation is S. *alterniflora* and *S. patens*. Using information provided by Olsen et al. (2005), I determine that the marsh vegetation ranges in elevation from 1.28 to 1.74 m CGVD28 (Canadian Geodetic Vertical Datum of 1928).

3.3.2 Extraction of physical landscape barriers

The detailed procedure for calculating the coastal squeeze index is presented in Table 3-1. Slope of the land surface was calculated from a light detection and ranging (LiDAR) digital elevation model (DEM) using the average maximum technique algorithm, which takes into account the rate of vertical and horizontal changes in elevation of a 3x3 neighborhood pixel window (Burrough and McDonnel 1999). The DEMs of Portland and Wells are in NAVD88 (with a 3-m spatial resolution and a root mean square error [RMSE] of ±0.39 m for Wells and ±0.78 m for Portland). Both data sets are available from the U.S. National Oceanic and Atmospheric Administration (NOAA) Digital Coast Database (NOAA 2012b), whereas the DEM for KNP is in CGVD28 (with 1-m cell size and RMSE of ±0.1 m). This DEM was provided by Kouchibouguac National Park. I used only the bare ground elevations from LiDAR data, so trees and buildings were eliminated from the analyses. Both the DEM and interpretations from remote sensing imagery (below) were verified with aerial photos and field observations.

The location of anthropogenic features with impervious properties was determined by processing images of the advanced space-borne thermal emission and reflection radiometry (ASTER) sensor (JPL 2012). ASTER has a spatial resolution of 15 to 30 m covering the visible, near infrared, and shortwave infrared and a 90-m cell size in the thermal regions of the electromagnetic spectrum. The sensor has been used in several land mapping applications with results better than most freely available multispectral sensors such as Landsat or MODIS. However, after 2007, ASTER malfunctioned and lost its capability to acquire images in the shortwave infrared regions, leaving only the visible, near-infrared, and thermal bands intact. I chose a single date ASTER image acquired in 2007 with all bands intact. Nine bands that span the visible, near infrared, and shortwave infrared spectrum were selected, re-sampled to pixels with 15-m spatial resolution, and stacked into one image file. The image was corrected for atmospheric effects using the Fast Line-of-sight Atmospheric Analysis of Spectral Hypercubes (FLAASH) radiative transfer model (Cooley et al. 2002), and the pixel values were scaled from radiance to reflectance. I then reduced data dimensionality using a minimum noise fraction (MNF) algorithm (Boardman and Kruse 1994) to transform and select the bands with maximum information. MNF outputs were used to calculate a pixel purity index (PPI), which determines spectrally distinct features, such as vegetation, concrete, soil, and water, on the image. From these features, the percent imperviousness [e.g., (van de Voorde et al. 2010; Weng et al. 2008)] was estimated using a match filter algorithm, which calculates the percent match of the spectral signatures of all pixels in the image to the spectral signatures of known pixels with high proportions of impervious features (Figure 3-2). All the image processing routines were implemented in ENVI software (RSI, Boulder, Colorado).

3.3.3 Developing the Costal Squeeze Index (CSI)

The coastal squeeze index was developed by assigning weights to the slope and percent imperviousness in a series of steps (Table 3-1). First, I develop a procedure or model to predict inundated areas and marsh extent relative to different sea levels in 2100 (i.e., 0.5, 1.0, 1.5, 2.0 and 2.5 m) and use it as a mask (Procedure A). Second, coastal squeeze equations and indices were developed to quantify coastal squeeze slope and imperviousness and combine these indices into one coastal squeeze index using a sub-model (Procedure B, see also Appendix Figure 3-6). Third, a second submodel was developed to simulate the extent of the future intertidal zone iteratively and use the result to produce an intertidal extent confidence image (Procedure C, see also Appendix Figure 3-7).

I assumed that the threat of coastal squeeze generally increases with increasing slope and imperviousness to inflection points beyond which coastal squeeze remains constant, a behavior pattern represented by a sigmoid curve. I also assumed that factors contributing to coastal squeeze vary continuously over the landscape. I initially applied two fuzzy membership models (Table 3-2): the sigmoid and linear fuzzy functions (Burrough and McDonnel 1999; Tsoukalas and Uhrig 1996). Both models level off at an inflection point. I assessed which function and parameter could better estimate the possibility of squeezing or squeeze potential from slope and imperviousness.

The two coastal squeeze scenarios have unknown model parameters, the coastal squeeze by slope and coastal squeeze by imperviousness, which are estimated in a fitting process. During the process, the intensity of squeeze potential, $\mu(x)$ was calculated by finding the spread (f_1) and midpoint (f_2) parameters for each squeezing variable (x). The midpoint represents the minimum crossover value, indicating a 50% possibility of squeeze occurring, whereas the spread parameter determines the magnitude of squeeze around the midpoint and usually is computed on the basis of the relationship between the midpoint and a critical threshold. The full range of slope and imperviousness values was assigned a coastal squeeze rank using the selected membership functions and the parameters obtained from sensitivity analysis. I used aerial photos and 3D rendering of elevations to describe how land use and topography varies within the selected sites. I also conducted limited field reconnaissance to validate slope and imperviousness and the model results on the three study sites.

At each site, slope was classed as flat, moderate, or steep (Table 3-3). The average of the three moderate slope classes, 11.5º, was designated as the midpoint (f_2) in the sigmoid fuzzy function or minimum in the linear fuzzy function, and I assigned it as equivalent to a 0.5 squeeze potential. I used the average, 44.0º, of the three steep slope classes as the maximum slope to compute the spread parameter and assigned a squeeze possibility of 1.0. To determine imperviousness, all natural cover, such as beach sand, bare soil, rock, vegetation, and water, were masked out, leaving only the pixels with different proportions of urban features or impervious materials. Lack of development in the vicinity of the KNP marsh meant that no significant area of impervious surface was detected there. At the Wells and Portland marshes, I designated three classes of imperviousness. The lowest class contained pixels that represented a mixture of pavement, vegetation, and bare soil; the medium class contained pixels corresponding to secondary roads and dark and colored roofs; and the high imperviousness class contained pixels associated with multilane highways, highly reflective pavements, and roofs. An example of these is shown in Figure 3-2. The imperviousness values within the medium and high classes at the three marshes were averaged, and these values (i.e., 15.8 and 45.5, respectively) were used to estimate the parameters of the coastal squeeze in the imperviousness model. Features were verified during field observation and inspection of aerial photographs.

Next, through a sensitivity analysis, the full range of the slope and imperviousness values were assigned a coastal squeeze rank using the selected membership functions and the parameters obtained from previous calculations. This process was implemented in MS Excel (Microsoft Corporation, Redmond, Washington), and the final membership function and parameters that best fit the data and assumption were selected. In cases where the midpoint and maximum threshold parameters do not fit perfectly, I adjusted the values by trial and error or using 'goal seek' function in MS Excel. The final function and parameters were used to estimate coastal squeeze values from slope and imperviousness. I then combined the computed fuzzy values of slope and imperviousness into a single index using a combination rule. I gave equal weights to slope and imperviousness, so I chose a combination rule "Fuzzy OR," (Burrough and McDonnel 1999) that selects the maximum value of a pixel from either the squeeze by imperviousness or squeeze by slope input maps. For example, if a pixel has a squeeze potential of 0.5 in slope and 0.7 in imperviousness, the final squeeze value would be 0.7. Most of the steps involved in producing the coastal squeeze index and combinations using fuzzy models were implemented in ArcGIS using the Spatial Data Modeler (Appendix Figure 3-6), an analysis toolbox developed by Sawatzky et al. (2009).

3.3.4 Uncertainty analysis

At each site, I identified the future tidal floodplains where marshes could migrate at each sea level rise increment of 0.5 m to close match IPCC projected values. First, I subtracted the sea level of interest (e.g., 0.5 m) to the original DEM to obtain a "flooded" DEM. This was implemented using an inundation model (Procedure A, Table 3-1). Inundation models, also known as the "bathtub" models (Gallien et al. 2011; Poulter and Halpin 2008), are simple and robust when complemented by uncertainty analysis (Gesch 2011*, pers. comm.*). Second, I built an uncertainty model (Appendix Figure 3-7) in ArcGIS based upon Holmes et al. (2000) and Wechsler (2007) technique to account for the confidence of locating the marsh and intertidal areas given a DEM vertical error and elevation range. DEM error reported as RMSE is a measure of the absolute difference between the interpolated elevation value and actual ground elevation (Gesch 2007). In the model, I added a random component that generates random fields from the range distribution of the DEM uncertainty and implemented a spatial average filter to enhance autocorrelation of the random errors. The auto-correlated random errors were added to the original DEM, producing a new DEM that included uncertainty and autocorrelation. From this new DEM, all the pixels within the specified marsh elevation ranges were identified and labeled with 1 for marsh and 0 for non-marsh pixels. The process was implemented using map algebra within the model and repeated 100 times. The several realizations produced from the iterations were added together to represent the final probabilities that the pixels belong to the specified elevation range. The modelling process was applied for a series of future sea levels from 0 to 2.5 m, with increments of 0.5 m, to span a range of estimates of sea level rise for the next 100 years (Bindoff 2007; Grinsted et al. 2010; Vermeer and Rahmstorf 2009).

3.4 Results and Discussion

3.4.1 Slope, imperviousness, and robustness of the index

Whether marshes can migrate inland with rising sea level depends on how steep and permeable the areas are surrounding marshes (Figure 3-3). Based on spatial analysis of the slopes of the study sites, I assumed that an 11.5º slope has a 0.5 (or 50%) possibility of causing coastal squeeze. Above this, the squeeze potential increases nonlinearly. From the model results, I estimated that a 1º increase in slope would increase the squeeze potential by nearly a factor of 4 (Figure 3-4A).

Figures 3-3B–G depicts future upper boundaries (i.e., extents) of the Wells marsh that will occur at five future increases in sea level. It shows that the distance between these boundaries decreases with increasing slope of the hinterland. In the area depicted in Figure 3-3B, slope varies from a high of 51.5º (lower bounding box) to 3.8º (e.g., upper bounding box). The close proximity of most of the future upper boundaries of the area of the Portland marsh in Figure 3A reflects the generally steep sloping hinterland around that marsh.

In assigning a squeeze potential or weights to the imperviousness data, I considered the physical characteristics of the actual surface or land cover relative to the spatial and spectral resolution of the imagery. In this case, percent imperviousness does not represent the actual degree of water penetration but the density of urban materials present in a 15 x 15 m area. At this resolution, some pixels contain different proportions of urban features, including low-intensity development and highly urbanized areas. In the final model, areas (i.e., pixels) containing a mixture of pervious and impervious surface with an average imperviousness of 15.8% received a squeeze potential of 0.5, whereas multilane highways, bright pavements, and intensive developments with 45.5% average imperviousness received a squeeze potential of 1.0. Based upon the model assumptions, for every 1% increase in imperviousness, the potential for squeeze increases by a factor of 5 (Figure 3-4B).

3.4.2 Implication of inaccuracies on the index

Neglecting to address uncertainties of the input data can result in misleading model results. Two types of uncertainty are associated with coastal squeeze: conceptual uncertainty as to how coastal squeeze is represented and uncertainty in the elevation data (i.e., RMSE). Both types of uncertainty influence the robustness and validity of the index. High RMSE in the DEM can increase the modeled extent of the intertidal floodplain but decrease the certainty or confidence of mapping its actual location. At Wells marsh, which has the highest uncertainty (0.4 m RMSE), the modeled tidal area is smaller when the RMSE is included in the computation than when it is not, a disparity resulting from selecting only those pixels for which there is more than 90% confidence of being within the intertidal elevation range. If RMSE is not included here, the intertidal extent is larger, but with only 70% confidence. At sites with more accurate DEM, the area calculated with and without uncertainty included is almost equal. Another limitation of DEMs with high inaccuracy is the increased time required for simulation. For example, it took at least 100 iterations to obtain a stable result with the range of the inaccuracy in the DEM used in this study (i.e., 0.1 to 0.78 m). This might take more in DEMs with higher vertical errors.

3.4.3 Coastal squeeze at study sites

The Wells marsh has the highest average CSI (0.17 to 0.29) in all the future sea levels I considered (Figures 3-3B & 3-5), and only about 4% of the migrating Wells marshes will be situated in steep slopes. In Wells, imperviousness by land use is the greatest contributor to coastal squeeze. Goodman et al. (2007) observed that marshes at Wells are undergoing transgression and suggest that the existing marshes would not survive a 0.5 m (4 mm y^{-1}) rise in sea level in the next 100 years. However, the results suggest that there will still be areas with suitable elevation that could accommodate inland migration if sea level rise does not exceed 1.5 m (equivalent to 15 mm y^{-1}); nevertheless if sea level rise accelerates and coastal land development continues, suitable areas could substantially decrease.

The average CSI for the Portland marsh ranges from 0.1 to 0.22 (Figures 3-3A & 3-5). Here, steep slopes are the greater contributor to coastal squeeze. Only a small percentage of the marsh will be threatened by land development regardless of sea levels because most establishments are built on elevated areas. Under a the maximum sea level rise, only a small portion of urban area and U.S. Interstate 295 will be flooded. The KNP marsh has a CSI of 0 in all future sea level rise scenarios examined (Figure 3-5). Only a small portion of the KNP is threatened by steep slopes, and imperviousness is not a factor at any of the sea level considered because there is no development in the vicinity of the marsh.

3.4.4 Further application of the CSI

By applying fuzzy logic, I are able to utilize the incomplete information on the relationship between squeeze (dependent variable) and slope and imperviousness (independent variables); capture the nonlinear behavior and conceptual ambiguity of coastal squeeze; and produce a map of coastal squeeze as a continuous gradient (Figure 3-3). This approach might have a practical advantage over the deliberate generalization into discrete categories because it gives more choices on how to represent coastal squeeze depending on the requirement of the analysis. For example, in its continuous form, the index can be used as a cost surface in cost-distance analyses or can be used in a regression model. Likewise if discrete classes are needed, the CSI can be easily classified in a GIS. The model and parameters (Table 3-4) can be used to calculate a CSI in any coastal area; it can be applied to any tidal wetland, including tidal freshwater marshes and mangrove swamps; and it can be nested with broader assessments, such as the coastal vulnerability index developed by Abuodha and Woodroffe (2010).

The calculation of the index and uncertainty model requires only basic expertise with GIS. However, they do require access to specialized databases. One requirement is elevation data with fine spatial resolution (<5 m) and low vertical uncertainty (<50 cm). Presently, LiDAR surveys are the only source of such data, but they are not available in all areas, or access may be restricted (Chmura 2013). Furthermore, DEM derived from LiDAR must be accompanied by remotely sensed imagery with a spatial resolution that is at least as fine as the DEM and with a suite of spectral bands (at least seven bands, including panchromatic, visible and near infrared, shortwave infrared, far infrared, and thermal infrared) capable of discriminating features found in a typical coastal environments. In this example, I used ASTER, but the failure of its shortwave infrared detectors would limit its future use in similar studies. CHRIS-PROBA (ESA 2012), a sensor similar to ASTER, provides a good alternative, but acquisition and processing is not as easy as that of ASTER. The Worldview-2 sensor (DigitalGlobe 2012), which offers 0.5 m spatial resolution with at least eight spectral bands, can be a better alternative, but it is expensive. There are plans for an improved version of the Landsat sensor through the Landsat Data Continuity Mission (Landsat 8), to be launched in 2013 (USGS 2012). The imagery from this system will likely be useful in future assessments of coastal squeeze. Finally, the ability of a marsh to occupy future tidal floodplains requires knowledge of the elevation range of local marsh vegetation, which can vary with tidal amplitude (Byers and Chmura 2007). This information can be estimated from published studies, but local measurements will increase accuracy of the coastal squeeze indices calculated for a region.

3.4.5 Other threats to tidal wetland persistence

The CSI addresses only limits to wetland migration inland with rising sea level and does not address changes in marsh area from submergence of existing marsh surfaces or retreat of its seaward edge. In addition, sea level rise poses other threats to tidal wetland sustainability not addressed in assessment of coastal squeeze. For example, threats of submergence will vary with the rate of sea level rise and local rates of marsh soil accretion. The latter are driven by suspended sediment supply and belowground marsh production (Cahoon et al. 2006), and the threat increases with decreasing tidal amplitude (Kirwan et al. 2010). If the elevation of the wetland surface decreases relative to mean sea level, vegetation zones might be displaced inland, and at the lowermost elevations, vegetation might not survive. Thus, existing wetland area will be lost, and marsh edges will be exposed to erosion. Because elevation and sea level change are key factors, the DEMs employed to determine the coastal squeeze index can be used to assess the potential loss of wetland on the seaward side or in low-lying interiors. (Such analyses, however, require local data on marsh accretion rates to model these processes adequately – which is outside the scope of the present work.)

Tidal wetlands with a low average CSI could still face serious threats to their persistence. The KNP marsh for example, has few impediments to inland migration, but with low tidal amplitude, its lower elevations are less likely to survive rising sea levels than marshes at Wells or Portland. At present, differences in tidal amplitude were not considered in the development of the coastal squeeze.

Developed landscapes are often associated with high nutrient loading, which can affect the species composition of marshes [e.g., (Bertness et al. 2002) as well as reducing belowground production that is critical to marsh vertical accretion [e.g., (Darby and Turner 2008)]. These effects were not addressed by the CSI. Nevertheless the index could be used in conjunction with other assessments, such as the impact index proposed by Brandt-Williams et al. (2013).

3.5 Conclusions

Preparation of CSIs for a suite of wetlands can help coastal planners prioritize conservation efforts and use of limited funding. For instance, the potential for coastal squeeze should be determined before investing in restoration of a tidal wetland. The index can be used to rank a set of potential restoration sites and prioritize efforts on those with the lowest threat of coastal squeeze. By identifying and ranking the threat of coastal squeeze at locations around a tidal wetland, the application of the index can help direct land use planning to mitigate threats.

Table 3-1. Detailed procedure for developing a coastal squeeze index as implemented in ArcGIS

Coastal Squeeze Development Procedure

Required data or products

1. Bare earth LiDAR Digital Elevation Model (DEM) in meters calibrated to specific vertical datum (i.e., NAVD88 for the US and CGVD24 for Canada). US data available at: http://www.csc.noaa.gov/digitalcoast/data/coastallidar

2. Percent Imperviousness calculated from ASTER image (http://asterweb.jpl.nasa.gov/data.asp) and re-sampled to the spatial resolution and projection of the DEM.

3. Aerial photographs of the study sites.

4. Elevation limit and tidal datum information for each marsh.

For example for Wells marsh:

- Upper marsh elevation limit (at NAVD88) = 1.95 m
- Lower marsh elevation limit (at NAVD88) = -0.02 m
- Highest Astronomical Tide (HAT) = 8.05 m
- Lowest Astronomical Tide (LAT) = 3.84 m
- Mean Higher-High Water(MHHW) = 7.4 m
- MSL(NAVD88) = 6.023 m
- NAVD88 = 5.85216 M
- US Tidal Datum database: http://tidesandcurrents.noaa.gov/data_menu.shtml?stn=8419317%20Wells,%20ME&type=Datums

A) Modelling procedure and ArcGIS operations to derive inundated elevation and analysis mask (i.e., marsh and upland areas) modified from Poulter and Halpin (2008) and NOAA (2012a)

1. Determine the extent of the current marsh. Determine the area below the upper edge of the current marsh from the initial DEM (DEM_0) and assign 1. All other areas are assigned a 0.

$marsh_0 = con(DEM_0 <= 1.95, 1, 0)$

2. Determine the upland that will be flooded at a maximum sea level under study at the end of 100-year period (i.e., 0.5, 1.0, 1.5, 2.0 and 2.5 m). This represents the future potential marsh migration area for that sea level. The extent of this area starts from the upper edge of the current marsh to the upper edge of the 2.5 m tidal floodplain.

DEM $(NAVD88)_{SL}$ = DEM0 (NAVD88)-SL, where SL is the sea level

3. Alternatively , subtract the NAVD88 height from Highest Astronomical Tide (HAT) height to get HAT height in NAVD88 then add the sea level height (e.g., 2.5 m)

- Implement a threshold conditional statement: $marsh_{2.5} = con(DEM_0 <= 4.5, 1, 0)$

4. Extract the areas between the current marsh edge and future tidal floodplain using conditional statement:

upland = con(marsh250 XOR marsh0,1).

I called this area 'upland mask' and used it to define the extent of the succeeding analyses. These procedures are repeated and adjusted according to the upper elevation limit of each marsh, HAT and reference datum.

5. Calculate and classify the slope of the *upland* (area within the future tidal floodplain) using the DEM of the *upland*.

- First get the elevation of the upland. Multiply the *upland* mask with the original DEM: $uplanddem = upland * DEM_0$
- Then calculate the slope in degrees: *uplandslope = slope(uplanddem)*

6. Classify the upland slope using Jenks (1967) classification.

Spatial analysis > Reclassify > uplandslope > classify > Natural Breaks (Jenks) > classes 3

7. Extract the imperviousness of the upland from ASTER imagery and resample it to the same spatial resolution as the DEM

8. Multiply the upland mask with the re-sampled Imperviousness image: *imperviousupland = upland * imperviousness*

9. Classify impervious upland into three bins using Jenks classification

Spatial analysis > Reclassify > imperviousupland > classify > Natural Breaks (Jenks) > classes 3

10. Repeat the above procedure for each marsh

11. Calculate the average slope and imperviousness in each of the three classes.

Tabulate the results of per site and by classes

See Table 3-3

B) Modelling procedure to quantify coastal squeeze (Appendix Figure 3-6)

1. Choose a slope and imperviousness value to use as a midpoint and maximum threshold parameters and assign a squeeze potential of 0.5 and 1.0, respectively.

2. Using the midpoint and maximum threshold parameters I estimated the squeeze potential for the whole range of slope (0.01 to 90) and imperviousness (0.01 to 100) by fitting two fuzzy membership functions and evaluating the results in a sensitivity analysis. The sensitivity analysis was implemented in MS Excel using the following conditional equations:

Equation for the Linear Fuzzy Function (e.g., for slope):

fuzzylinearslope = IF((slope-midpoint)/(maximum-midpoint) < 0, 0, IF((slope -midpoint)/(maximum-midpoint) > 1,1,(slope-midpoint)/(maximum-midpoint)))

Equation for the Sigmoid Fuzzy Function:

fuzzysigmoidslope = 1/ (1+POWER(slope/midpointslope,spreadparameter))

The spread parameter is a value between 1 and 10.

After inspecting the results of the sensitivity and parameterization analyses, I choose the sigmoid fuzzy function.

3. The final parameters for the Sigmoid Fuzzy Function were applied to the slope and impervious data in a GIS. This procedure converts the slope and imperviousness values into coastal squeeze values ranging from 0 to 1. For this procedure I used the Fuzzy Logic Tool in Spatial Data Modeller Toolbox (Sawatzky et al. 2009). Alternatively, this can be implemented manually using the ArcGIS Raster Calculator.

fuzzyslope = 1/ (1+Pow(slope/11.5,-3.95))

fuzzyimperv = 1/ (1+Pow(imperviousnes/15.8,-5.0))

The process yields two output maps of coastal squeeze potential from slope and imperviousness.

4. I combined the resulting map of coastal squeeze into one index -using the 'FUZZY OR' combination rule in Spatial Data Modeller Toolbox. This rule compares the squeeze values from slope and imperviousness and takes the higher value.

coastalsqueezeindex = fuzzyslope OR fuzzyimperve

5. The final coastal squeeze value of each marsh in each sea level was computed by multiplying the coastal squeeze index values with a confidence ratio map of the marsh extents produced in B.

The coastal squeeze index of each marsh extent in each sea level can be calculated in two ways. One is by selecting a confidence threshold and using it to mask coastal squeeze values. For example, select all areas with confidence ratio above 0.9 and assign the value 1 then multiply the resulting map by the coastal squeeze index. The other is by multiplying the confidence map for each sea level by the coastal squeeze index directly. This produces a confidence-weighted coastal squeeze index. I choose the second method to calculate the coastal squeeze index.

a. marshmap50 = con(marshextent50 >= 90, 1)

coastalsqueezeindex50 = coastalsqueezeindex * marshmap50

b. coastalsqueezeindex50 = coastalsqueezeindex * marshextent50

C) Modelling procedure to determine marsh extent using iterative probabilistic method (Appendix Figure 3-7)

The marsh extent map was generated using an iterative process that locates the extent of a marsh in the DEM that reflects its vertical uncertainty. To automate this procedure I developed an uncertainty model (Appendix Figure 3-8) in ArcGIS ModelBuilder (ESRI 2009; Hall and Post 2009) to simulate a probabilistic extent of the tidal wetland. For the optimum number of iterations, the model generates a random field from the distribution range of the DEM uncertainty (i.e., RMSE), adds it to the DEM, and then map the marsh extent using a threshold conditional statement:

con(DEMx < -0.02 | DEMx > 1.95,0,1),

Where DEMx is the DEM at x m sea level of interest and the constants -0.02 and 1.95 are the lower and upper elevation limit of a certain marsh (e.g., Wells Marsh), respectively. DEMx was computed by subtracting the sea level of interest to the original DEM.

The algorithm assigns 0 to areas with elevation lower than the lower marsh limit OR areas higher than the upper marsh limit, all other areas are assigned with 1, meaning a marsh pixel. I repeated this process for several times and found that 100 iterations are sufficient to obtain an optimum result. The iteration calculates 100 versions of a marsh extent on top of each other and adds the result producing a single map with values ranging from 0 to 100. This corresponds to the percent confidence of a pixel classified as a marsh. To have the confidence values on the same scale as the coastal squeeze index, I divided the pixel values by the maximum value resulting in a confidence ratio.

Table 3-2. Candidate fuzzy membership functions used to model the relationship between slope, imperviousness, and coastal squeeze.

Membership Function	Equation	Parameters
Sigmoid Fuzzy Function	$\mu(x) = \dfrac{1}{1 + \left(\dfrac{x}{f_2}\right)^{-f_1}}$	$\mu(x) =$ fuzzy squeeze variable $x =$ raster variable (e.g., slope) f_1= spread f_2 = midpoint
Linear Fuzzy Function	$\mu(x) = \left\{\begin{array}{l} 0, x \leq a \\ \dfrac{x-a}{b-a}, a < x < b \\ 1, x \geq b \end{array}\right\}$	$\mu(x) =$ fuzzy squeeze variable $x =$ raster variable (e.g., slope) a = minimum value b = maximum value

Table 3-3. Classes of slope and imperviousness generated within a GIS using Jenks (1967) classification. The region around Kouchibouguac marsh includes no impervious surface. Average is based on the upper bounds of each class. Mid, low, and high squeeze potential

Class	Wells	Portland	Kouchibouguac	Average	Squeeze Potential
			Slope (°)		
Flat	0–3.8	0–6.9	0–2.3	4.3	0.0
Moderate	3.8–12.1	6.9–17.3	2.3–5.0	11.5	0.5
Steep	12.1–51.5	17.3–44.3	5–36.3	44.0	1.0
			Imperviousness (%)		
Low	0–6.8	0–3.1	NA	4.9	0.0
Medium	6.8–20.5	3.1–10.2	NA	15.8	0.5
High	20.5–66.5	10.2–23.8	NA	45.5	1.0

Table 3-4. Midpoint and spread of parameters used to calculate the coastal squeeze index and the fuzzy logical operation that integrates them into the index.

Variables	Parameters		Equation
	Midpoint(f_2)	Spread(f_1)	
Slope	11.5	-3.95	$\mu(\text{slope}) = \dfrac{1}{1 + \left(\dfrac{\text{slope}}{11.5}\right)^{-3.95}}$
Imperviousness	15.4*	-5.00	$\mu(\text{imperviousness}) = \dfrac{1}{1 + \left(\dfrac{\text{imperviousness}}{15.4}\right)^{-5}}$
Coastal Squeeze Index (CSI)	$\mu(\text{slope})$	$\mu(\text{imperviousness})$	$CSI = \mu(\text{slope})\ OR\ \mu(\text{imperviousness})$

* adjusted value after the sensitivity analysis

Figure 3-1. Location of study sites in the United States and Canada. (1) Wells Marsh, a complex of marshes within the National Estuarine Research Reserve (NERR) in Wells, Maine; (2) Portland Marsh, a complex of fringe Marshes in Presumpscot River estuary between the cities of Portland and Falmouth, Maine; and (3) Kouchibouguac Marsh (KNP), within Kouchibouguac National Park near Richibucto, New Brunswick, Canada

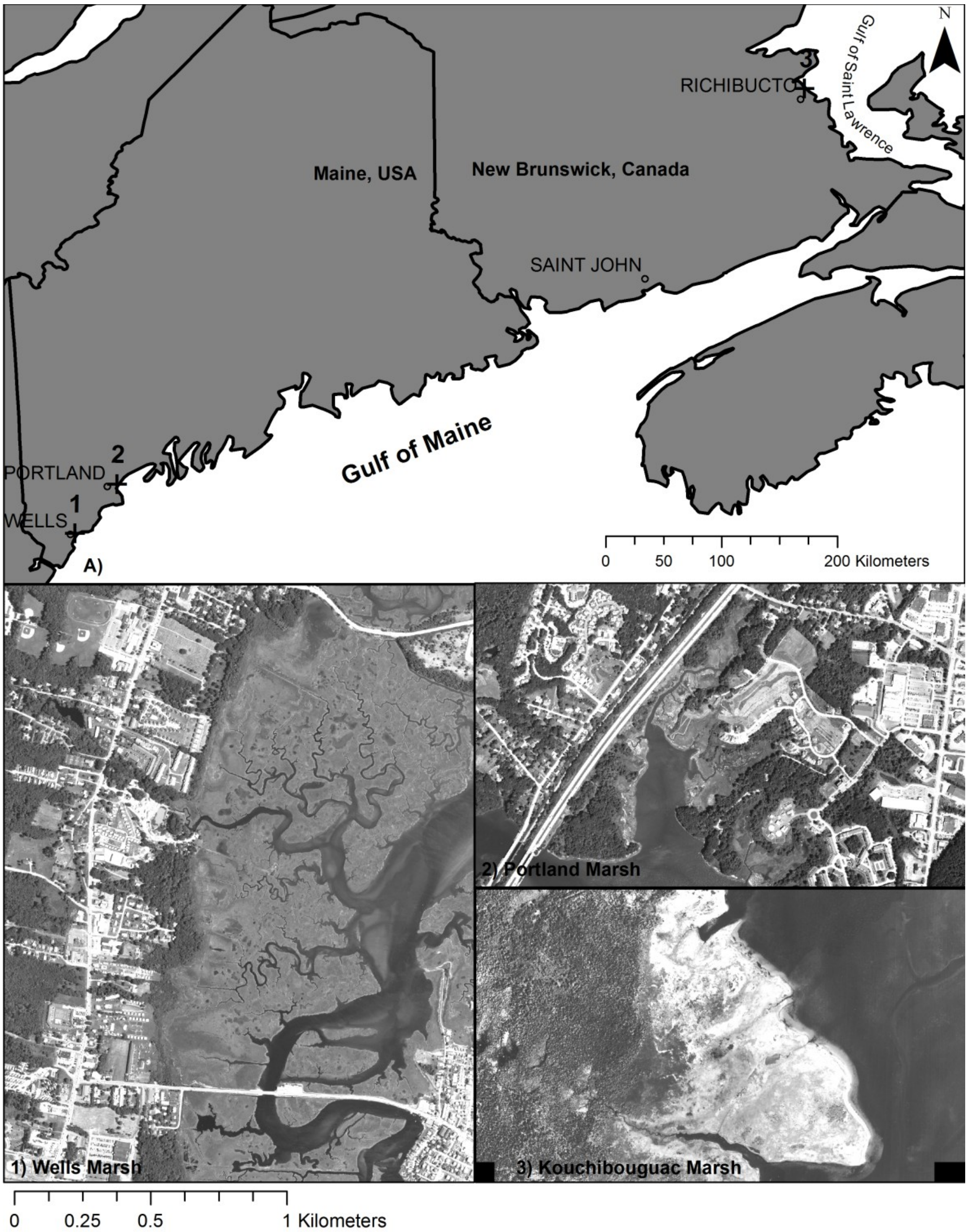


Figure 3-2. Percent imperviousness of a section of Wells Marsh draped over an aerial photograph. Dark areas indicate a high degree of imperviousness.

Figure 3-3. The edge of the 100-year tidal floodplain associated with different sea levels on a selected portion of each marsh in (A) Portland and (B) Wells; (C–G) maps showing the intensity of coastal squeeze at increasing sea levels (i.e., 0.50 to 2.5 m) on a section of the Wells Marsh

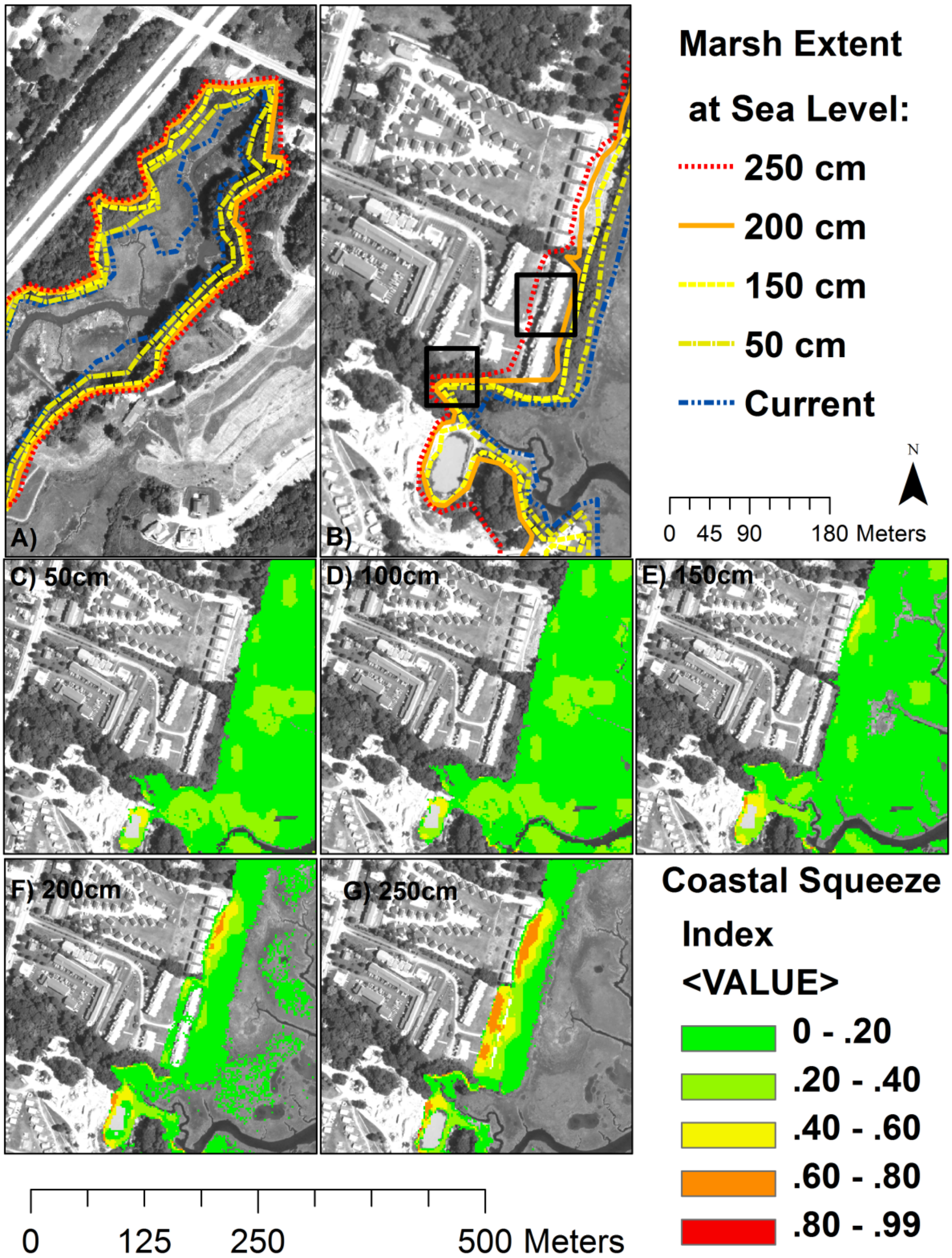
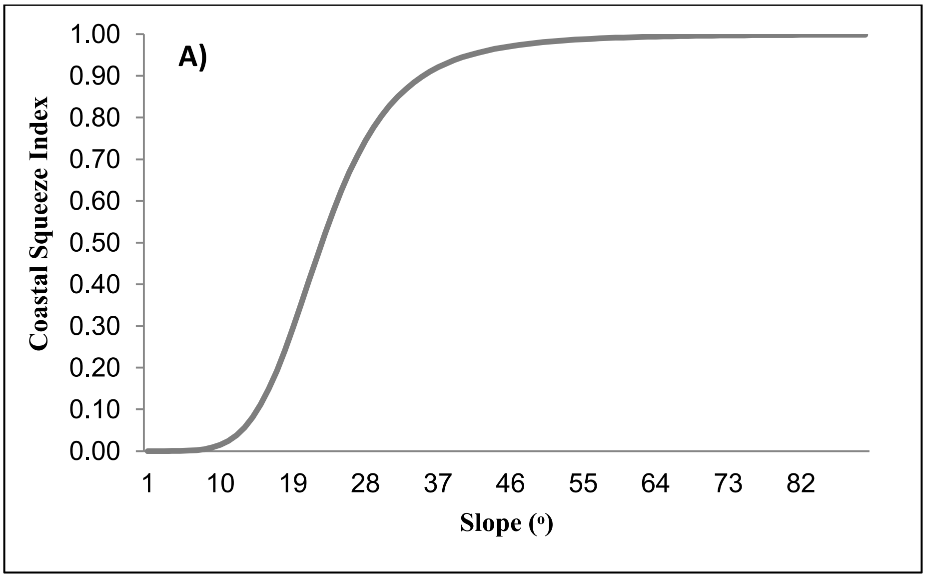


Figure 3-4. The sigmoid curves reveal the modelled relationship between (A) slope and (B) imperviousness with coastal squeeze.



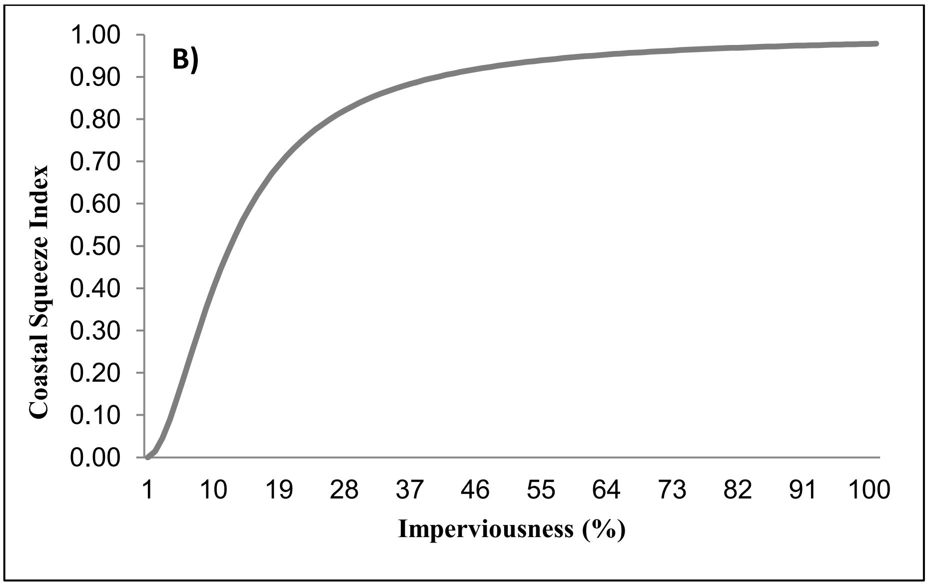
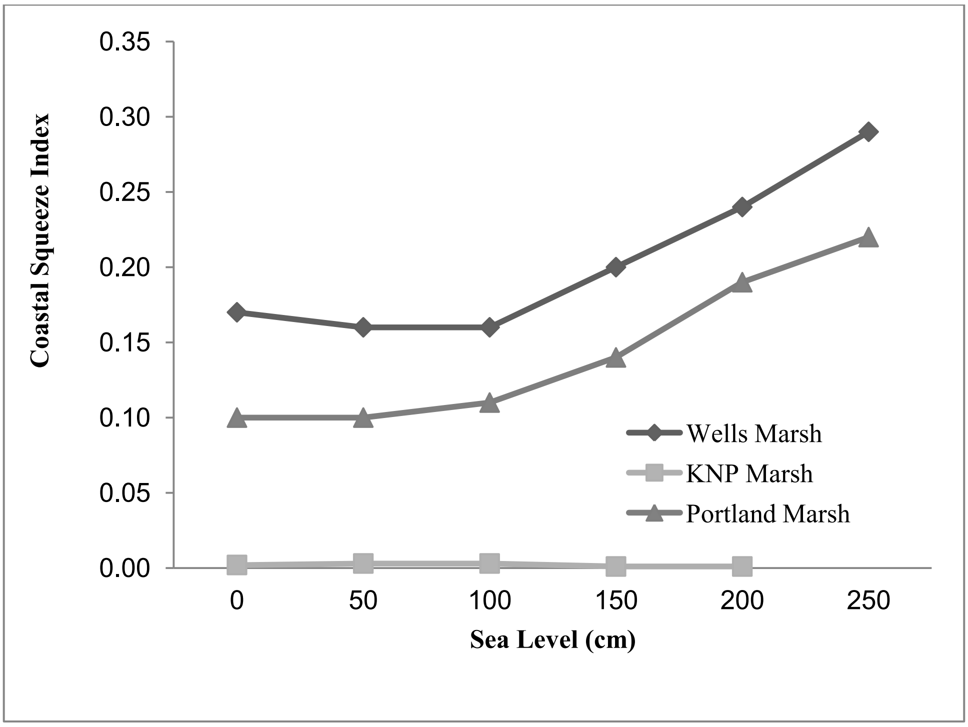


Figure 3-5. The average coastal squeeze index for the three marshes at different future sea levels



3.6 Appendix

Figure 3-6. The implementation of the Coastal Squeeze Index model in ArcGIS ModelBuilder

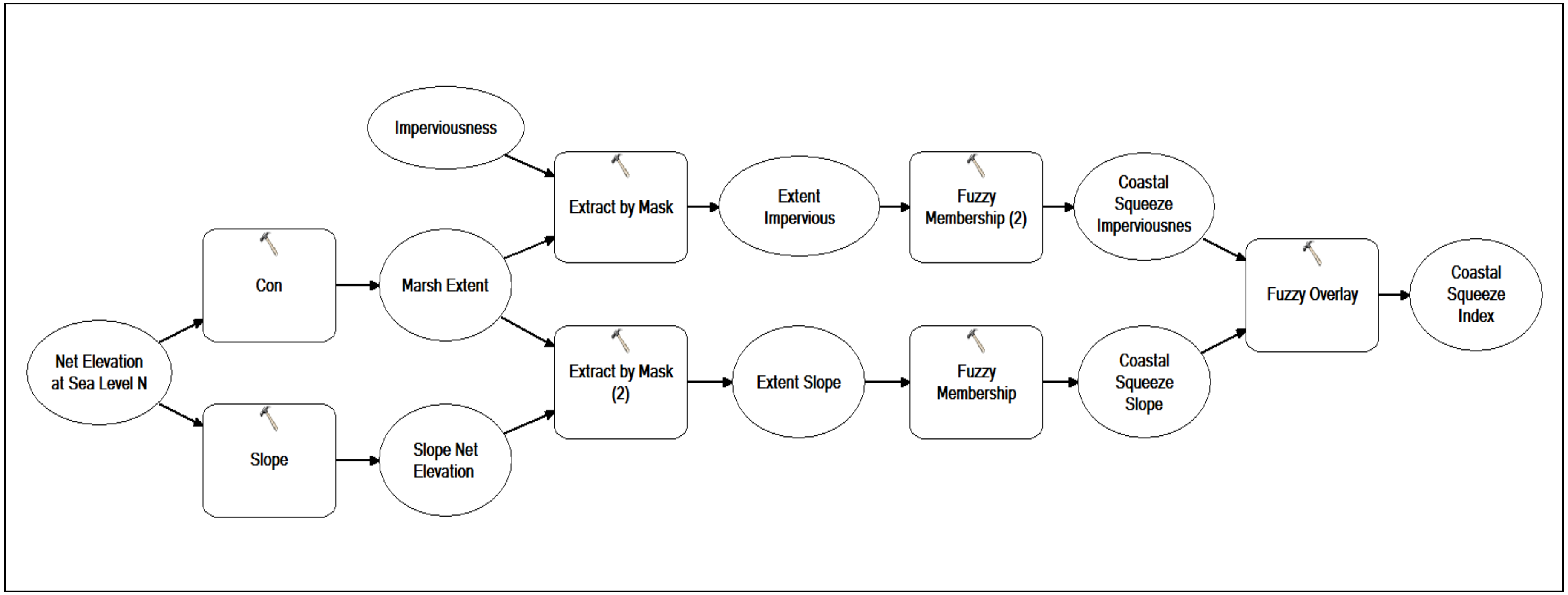
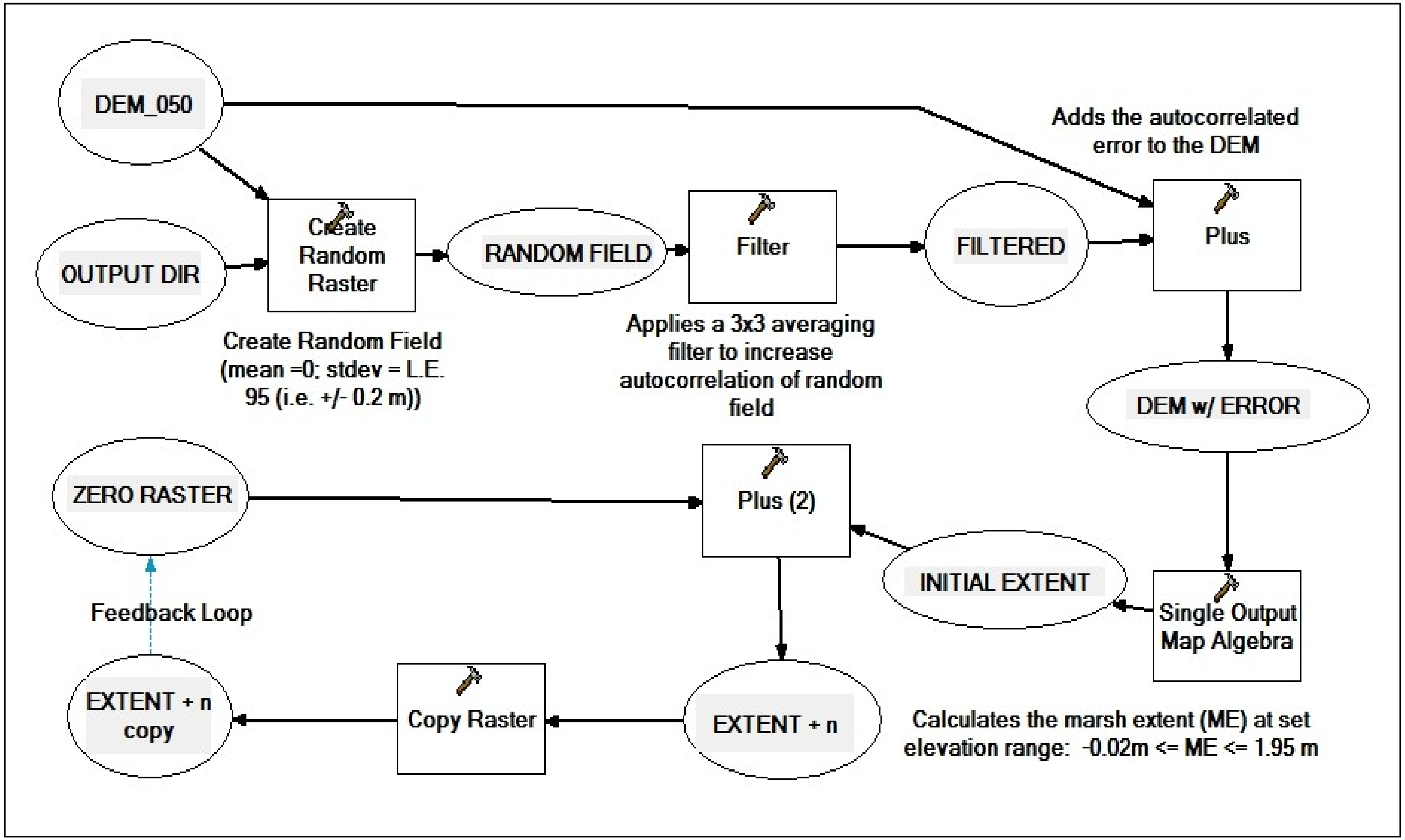


Figure 3-7. Marsh extent probability model implementation in ArcGIS ModelBuilder



Chapter 4 Assessing coastal squeeze of North American tidal wetlands

4.1 Preface

4.1.1 Manuscript details

This manuscript is co-authored by Gail Chmura. It is in preparation for submission to Estuarine, Coastal and Shelf Science.

4.1.2 Context and link to the previous and subsequent chapter

The emerging international and national policies for mitigating climate change and biodiversity loss in coastal areas (e.g., United Nation Framework Convention on Climate Change-Nationally Appropriate Mitigation Actions, Aichi Biodiversity Target 11, biotic wetland connectivity, RAMSAR, National Framework For Canada's Marine Protected Areas, North American Marine Protected Areas Network, Commission for Environmental Cooperation's North American Blue Carbon Project) require information over a broader geographical extent than that available through localized studies. Thus, continental and national scale threat analyses are needed. Additionally, national comparisons will enable environmental international non-governmental organizations to prioritize investments in tidal wetland conservation and restoration.

In this chapter, the coastal squeeze submodel and equations developed in the previous chapter are adapted to apply to a continent-wide scope by parameterizing them with different data sets, i.e., global slope and imperviousness data. The aim is to examine the relative threats of coastal squeeze to mangroves and salt marshes of North America and assess coastal squeeze by slope and imperviousness. In the next chapter, the inundation model and the extent submodel were adapted to be used in the analysis of change in marsh function as fish habitat.

4.2 Introduction

Tidal wetlands are threatened by sea level rise. With sea level rise, tidal wetlands can migrate inland, as they have done in the past. Today, their migration space may be limited by biophysical (e.g., slope) and socio-economic (e.g., coastal development and fortifications creating impervious surfaces) barriers, putting them in a coastal squeeze (Doody 2004; Schleupner 2008; Torio and Chmura 2013). The coastal zones of North America consist some of the most important intertidal wetlands but it is also one of the most developed coastal zones in the world. Together with accelerated sea level rise, these developments create possible barriers to wetland migration. At present, the current level of coastal squeeze of North American wetlands is unknown because coastal squeeze studies are limited to local scale and most management strategies are localized.

With the emergence of national policies and strategies for mitigating climate change and biodiversity loss in coastal areas [e.g., United Nation Framework Convention on Climate Change-Nationally Appropriate Mitigation Actions (UNFCCC 2015), RAMSAR, Aichi Biodiversity Target 11(CBD 2015), National Framework For Canada's Marine Protected Areas, North American Marine Protected Areas Network (CEC 2015a), North American Blue Carbon Project (CEC 2015b)] a North American-wide study is needed (Erwin 2009) to assess the overall threat and permanence of tidal wetlands to identify priority conservation sites. Relative to mangroves and salt marshes, coastal squeeze is one of those threats that require immediate attention to inform national and international conservation programs.

Many ecosystem services and ecological functions [e.g., (Arkema and Samhouri 2012; Chmura et al. 2012; Costanza et al. 1998)] are provided by tidal wetlands. For example, marshes and mangroves provide habitat for crustaceans, fish and other organisms that are harvested. Mangroves and marshes store carbon and trap toxic substances purifying the air and water. Patches of mangroves and salt marshes along migration pathways provide abundant food and refuge for migrating waterfowl and fish. The increasing demand for ecosystem services (Abson et al. 2014; Barbier 2012) requires management to ensure that tidal wetlands can adapt to environmental changes and continue to provide such services. However, it is uncertain whether present conservation strategies are adequately addressing wetland adaptation with sea level rise and potential limitation of inland migration from coastal squeeze.

Decisions regarding tidal wetland restoration seldom are made with the consideration of potential coastal squeeze. In the US, for example, the Gulf of Maine Council has a list of potential salt marsh restoration projects, but does not consider whether some marshes may be immediately threatened be coastal squeeze. In Mexico, CONABIO (Comisión National Para El Conocimiento Y Uso de la Biodiversidad) prioritized conservation sites based on their contribution to biodiversity. Although such criteria are essential to maintain and protect existing wetlands, they do not guarantee long term persistence. With the uncertainty of tidal wetlands surviving accelerated sea level rise, landscape level processes and constraints will become more relevant in restoration. Although restoration programs are slowly leaning towards a more landscape and ecosystem scale approach [e.g., fresh water wetlands (MEA 2005; White and Brown 2005)] salt marsh and mangrove restoration still lags behind in assessing landscape level threats and long term persistence with climate change.

Global efforts to redesign nature reserves are now shifting towards dynamic protected areas as static and geographically fixed reserves are no longer adequate (Hyrenbach et al. 2000) to ensure functional protected area networks. In particular, dynamic Marine Protected Areas (MPAs) are seen as a more effective conservation strategy to ensure resilient ecosystems (Hyrenbach et al. 2006). In some parts of the world, dynamic protected areas have been proposed to protect coral reefs and fisheries (Game et al. 2009; Hobday et al. 2010) because of the increasing threat from climate change, human pressure and the possibility of range shifts (Lewison et al. 2015). Marine Protected Areas for tidal wetlands also should be dynamic because with sea level rise, tidal marshes and mangroves could shift inland. Inland migration could result in tidal wetlands being located outside designated MPAs. If dynamic protected areas will be adopted for tidal wetlands, it is essential to identify which habitats have the most potential to migrate inland or least affected by coastal squeeze.

This study quantifies the threat of coastal squeeze to salt marshes and mangroves of North America by adapting and applying a GIS model developed in Chapter 3 to rank topography and land use imperviousness a continent-wide scale. The aim is to assess the relative threat of coastal squeeze based on the steepness of slope.

4.3 Methods

4.3.1 Data sources

This study considers salt marshes and mangroves in United States, Canada and Mexico (Figure 4-1). Spatial data on tidal wetland extents, protected areas and priority restoration sites were obtained from various sources (Table 4-1). In the USA, most of the wetlands have been mapped as part of the National Wetlands Inventory. Other states like Louisiana have updated wetland maps not included in the National Wetlands Inventory. Maps of marsh distribution were obtained from the provinces of New Brunswick, Nova Scotia, and British Columbia. Maps of Quebec marshes were available from Environment Canada (no coverage is presently available for the province of Newfoundland or the coasts of Hudson and James Bay.) In Mexico, only the marshes in some parts of the Baja coast have been mapped. Mangrove distribution on the coast of the USA (states of Florida, Louisiana and Texas) is available from UNEP-World Conservation Monitoring Centre. Mangrove areas of Mexico were obtained from CONABIO.

Maps of potential restoration sites were obtained for the coast of the Gulf of Maine (USA and Canada) and Mexico. Sites on the Maine coast of the Gulf of Maine were provided by Robert Houston of the US Fish and Wildlife Services. Potential restoration sites on the New Brunswick and Nova Scotia coasts of the Bay of Fundy (the upper part of the Gulf of Maine) were provided by Dr. Danika van Proosdij of Saint Mary's University (Halifax, Nova Scotia). For Mexico, the map of the priority sites was obtained from CONABIO. The maps for the terrestrial and marine protected areas were obtained from the Commission for Environmental Cooperation (CEC) North American Environmental Atlas (CEC 2015c).

4.3.2 Coastal squeeze calculation

Coastal squeeze was calculated in a series of steps and was based upon slope and anthropogenic imperviousness. First, a digital elevation model (DEM) and imperviousness data for North America was obtained from the CGIAR Consortium of Spatial Information (CGIAR 2015) and National Oceanic and Atmospheric Administration (NOAA 2010), respectively. The DEM has a pixel size of 250 m and about ±10 m vertical accuracy and was derived from the Shuttle Radar Topographic Mission (Jarvis et al. 2008; JPL 2015). The percent imperviousness data was derived from VIIRS sensor night time lights and land cover derived from Landsat imagery. It has a pixel size of 1000 m (Elvidge et al. 2007). Because of the differences in resolution, the imperviousness data was re-sampled to the resolution of the DEM (i.e., 250 m). It was assumed that anthropogenic barriers were represented by the degree of imperviousness (Elvidge et al. 2007). Second, subsets of the elevation and imperviousness were extracted for coastal areas with elevations ≤ 30 m. These areas represent low elevation coastal zones (McGranahan et al. 2007) where most of the intertidal wetlands are found and human activities have the most influence on coastal ecosystems. Third, the slope (in degrees) of the delineated coastal zone was calculated using the slope function in ArcGIS 10.2.2 (Burrough and McDonnel 1999; ESRI 2015) from the DEM.

The coastal squeeze index for the slope and imperviousness was calculated using a model developed by (Torio and Chmura 2013). The initial results based on slope underestimated areas so the equation was modified after comparing the results with a LiDAR-based model developed for the marshes in Wells and Portland, Maine as reference sites. The midpoint parameter (f_2) of the coastal squeeze slope model thus was adjusted from 11.5 to 1.5 (equation 1). For the coastal squeeze based on imperviousness (Equation 2), no modification was needed as the results of the validation tests matched the extent of urban areas in the LiDAR-based model.

$$\mu(coastal\ squeeze\ slope) = \frac{1}{1+\left(\frac{slope\ (degrees)}{1.5}\right)^{-3.95}} \quad (1)$$

$$\mu(coastal\ squeeze\ imperviousness) = \frac{1}{1+\left(\frac{imperviousness}{15.4}\right)^{-5}} \quad (2)$$

After computing the two coastal squeeze components, they were combined to produce a cumulative coastal squeeze index in which both slope and imperviousness were assumed to have equal contributions. The resulting coastal squeeze indices (i.e., coastal squeeze slope, coastal squeeze imperviousness and coastal squeeze cumulative) were aggregated into three classes representing three classes of threat with 1 the lowest and 3 the highest threat of coastal squeeze. The upper bound of class 1 is a coastal squeeze index < 0.33 and class 3, coastal squeeze index > 0.66. These intervals were based upon Nature Conservancy threat categories which represent the threshold of habitats (Crain et al. 2008; The Natural Capital Project 2015) under risk of multiple stressors. This is similar to dividing the full range of the threat indices into terciles. After producing the coastal squeeze classes, the resulting raster layer was converted to polygons using the raster to feature tool in ArcGIS 10.2.2. Using a spatial analysis, the mangroves and salt marsh polygons were associated to a threat polygon and each wetland polygon was encoded with a threat level in its attribute table. This was implemented for all the threat classes of slope, imperviousness and cumulative coastal squeeze.

4.4 Results

4.4.1 Cumulative coastal squeeze threat

The analysis indicates that salt marshes are more threatened than mangroves in North America (Figure 4-2A). Of the total salt marsh area (13, 474 km^2) about 9% is under medium threat, and 2% under high threat or about 1,167 and 316 km^2, respectively. Of the mangrove areas only 2% and 1% are under medium and high threat, respectively.

The level of threat of coastal squeeze varies with political jurisdiction. In Mexico the salt marshes in the states of Baja California Sur (40%) and Baja California Norte (70%) are under medium threat (Figure 4-2B). No Mexican marshes are under high cumulative threat, but much of the coast has not been mapped. In Canada, more than 10% of the salt marsh area is under medium threat and at least 5% of the salt marshes in British Columbia, Quebec and New Brunswick are under high threat. In the USA, states with the largest salt marsh areas (>1000 km^2) (i.e., Louisiana, Texas, South Carolina, Florida and Georgia) the percentage of areas under medium and high threat is small. In contrast, the states with the least marsh area, New York, California, Connecticut and Rhode Island stand out by having the greatest areas (>10%) under medium and high threat.

Compared to the marshes, the cumulative coastal squeeze is lower in mangroves (Figure 4-2C). For the Mexican states with the largest areas of mangrove (>1,000 km^2), Yucatan, Quintana Roo and Campeche, less than 1% of their mangrove areas are under medium threat and the rest are under low threat. Conversely, the states with the least mangrove area (<100 km^2), the Baja California Sur and Norte, Colima, Guerrero, Tamaulipas, Jalisco, and Michoacán have larger portions of their mangrove areas under medium and high threat (~5%). In the USA, the trend is reversed. For example, Florida, the state with the largest area of mangrove (~2,000 km^2) has about 3% of its mangroves under high threat compared to Louisiana and Texas which have less area and proportion (<1%) under this category. About 10% of the mangrove areas in Louisiana are under medium threat while all the mangroves in Texas are facing low threat.

Table 4-2 shows the coastal squeeze threat classes of the restoration sites and marine and terrestrial protected areas across North America. About 3% of the Gulf of Maine and 9% of the Bay of Fundy restoration sites are under high threat. In Mexico only about 1% of the country's mangrove restoration sites are under high coastal squeeze threat. Of the 4,358 terrestrial protected areas in North America, 137 sites or about 3% are under high coastal squeeze threat. Likewise, around 5% or 40 sites out of 821 marine protected areas are under high threat.

4.4.2 Coastal squeeze by slope

Steep slopes present natural barriers to inland migration of tidal wetlands, but pose a minor threat along most the coasts of North America I examined. In total, about 7 km^2 of salt marshes and less than 1 km^2 of mangroves are highly threatened by topographic constraints. These represent only about 0.01% and 0.1% of the total North American mangroves and salt marshes, respectively (Figure 4-3A&B). A large proportion of these wetlands, however, are under medium threat.

Limitation to inland migration of salt marshes by natural barriers is most prevalent in Maine (Figure 4-4A), Rhode Island (Figure 4-4B) and Nova Scotia with at least 1% of the marsh areas under high threat. The proportion of salt marsh area in rest of the US states and Canadian provinces at this threat level is lower. All the Mexican salt marshes (that are mapped) are under medium threat with the Baja California Norte state having the largest area under this category. Of the Canadian provinces with salt marshes under high cumulative threat (i.e., British Columbia, Quebec and New Brunswick), only New Brunswick has salt marshes limited by natural barriers.

Only the Mexican states of Guerrero and Quintana Roo (Figure 4-4C) have mangrove areas threatened by natural barriers, although <1% of the area is threatened. As Quintana Roo has extensive area of mangrove the actual area threatened is only about 0.8 km^2. The natural barrier of slope poses a medium threat in Mexican states of Colima, Tamaulipas, Jalisco and Michoacán with smaller mangrove areas that fall under high cumulative coastal squeeze. Unlike the Mexican mangroves the US mangroves are not threatened by slope.

Only few restoration sites and protected areas are threatened by steep slope (Table 4-2). In Maine, of the 33 sites under high cumulative coastal squeeze, 4 sites are in steep slopes. Likewise, only 1 site out of 28 Bay of Fundy restoration sites is in steep slope. There are no priority sites in Mexico under steep slope. For protected areas, only terrestrial protected area is situated in steep slope.

4.4.3 Coastal squeeze by imperviousness

Coastal squeeze by imperviousness threatens more mangroves and salt marshes than coastal squeeze by slope (Figure 4-5A&B). In North America, 83 km^2 of mangroves and 261

km^2 salt marshes are under high threat of imperviousness or roughly about 1% and 9% of the total mangrove and salt marsh areas, respectively. With these figures, the threat of imperviousness surpasses the threat of slope both in percentage area under threat and the number of states affected.

Mexican marshes are not threatened by imperviousness. In Canada, 3-5% of the marshes in New Brunswick, British Columbia and Quebec are under high imperviousness. Likewise in the US, 4-30% of the marshes in New York, California, Connecticut, New Jersey, Rhode Island, Maryland, Virginia and Florida are under the same category. In these areas [e.g., New York (e.g., Jamaica Bay, Figure 4-6A), California (e.g., San Francisco Bay, Figure 4-6B), Florida (e.g., Tampa Bay (Figure 4-6C)] the habitats are surrounded by highly impervious surfaces; characteristics of major urban development.

In Mexico the mangroves in Colima, Jalisco, Michoacán, Guerrero and Tamaulipas are among the highly threatened habitats with at least 2% of their areas under high threat. These states have smaller mangrove areas and under high cumulative coastal squeeze. In one of these states (e.g., Porto Vallarta, Jalisco, Figure 4-6D) the mangroves are surrounded by impervious surfaces from tourism development. With the exception of Quintana Roo, the states with larger mangrove areas like Yucatan, and Campeche have low threat classes. In the Baja region, some of the marshes in the southern Baja are under high threat. Mangroves in Quintana are increasingly threatened by impervious surfaces from tourism development (Pedrozo-Acuña et al. 2015).

Contrary to the threat posed by steep slopes, the threat of imperviousness affects most of the restoration sites and protected areas under high cumulative coastal squeeze threat (Table 4-2). For example, out of the 33 highly threatened Gulf of Maine restoration sites, 29 are under threat of impervious surface. 28 out of 29 highly threatened Bay of Fundy restoration sites are under similar threat. In Mexico, the one site under high cumulative coastal squeeze threat is threatened by imperviousness.136 out of 137 terrestrial protected areas are under high threat of coastal squeeze from imperviousness. All the 40 marine protected areas under high threat are surrounded by highly impervious surfaces.

4.5 Discussion and Conclusions

About 10% of the total North American tidal wetlands are threatened by high coastal squeeze. Of the two coastal squeeze variables; imperviousness threatens more tidal wetlands than slope – providing the potential for mitigation for most of the areas where the threat of coastal squeeze is present. Less than 1% of the total salt marsh and mangrove areas or about 8 km^2 are bordered by steep slopes while 10% or about 344 km^2 are under the threat of imperviousness. Of the two, marshes are more threatened than mangroves. Tidal wetlands in the US and Canada are more threatened than wetlands in Mexico.

Slopes do threaten the migration of small percentage of marshes and mangroves particularly in states that have relatively steep coastal areas such as Nova Scotia Maine, Rhode Island, Guerrero and Quintana Roo. In these areas, there is little that can be done for the threatened tidal wetlands. In cases where mangroves and salt marshes can still provide substantial ecosystem services, enriching them with sediments would be a viable management option if those services outweigh the management costs (Gilman et al. 2007). In contrast, wetlands that are under threat of imperviousness, especially those under low and medium threat, have more potential for permanence.

Tidal wetlands located in Marine and Terrestrial Protected Areas are not protected from rising sea levels and coastal squeeze. As such, static and geographically fixed Protected Areas will not be an effective management tool with climate change and simple protection from direct disturbance will not be enough to ensure their future. We may protect them from direct threats with fixed reserves, however, long term management of Protected Areas must consider the future migration space for wetlands and future reserves should include spatially and temporally dynamic buffers or easements to build a resilient network.

4.5.1 Implication for management decisions

Much of the salt marsh and mangrove areas of North America are found on the Gulf of Mexico and Atlantic coasts, regions that are most vulnerable to sea level rise (Pendleton et al. 2010) and where coastal developments is intensive. Based on relatively extensive areas under medium threat, coastal squeeze will become an increasingly important factor affecting the permanence of tidal wetlands with environmental change. If there are no management interventions to address coastal squeeze now it is highly likely that the areas under high threat would increase in a couple of years. In areas where there is a medium threat of imperviousness, it should be possible to develop incentives to prevent further development in areas immediate to the marshes and mangroves.

Within the USA, Florida and Texas have the largest areas adjacent to tidal wetlands but their populations and proportion of urbanized areas are also high (Figure 4-7A&B). Louisiana, on the other hand, has large low lying areas with less development that are ideal for wetland migration. In Mexico, the states with high potential for wetland migration include Baja California Sur, Campeche, Quintana Roo and Yucatan. In Canada, New Brunswick and Quebec have considerable potential migration areas. The area for Alaska is underestimated because the elevation data does not cover the northern most part of the region, but using a different dataset I calculated an area of about 184,000 km^2 low elevation coastal zone, the largest among the states. However, a large migration area would not always guarantee wetland migration in the future because there might be institutional and socio-economic barriers to migration. As such it would be prudent to reconsider longstanding policies of shoreline hardening and shift to more hybrid approaches (Sutton-Grier et al. 2015). Titus and Neumann (2009) assert that policies to ensure that wetlands are able to migrate are likely to be less expensive if the planning takes place before development.

It is no longer adequate to protect wetlands in their current locations because in the future they may need to migrate inland. Displacement upland will mean that wetlands inside marine protected areas now could be located outside those areas in the future. With sea level rise, it is more sensible to redesign protected areas to include buffers to allow inland migration and prevent further development in those buffers.

4.5.2 Implication for restoration

The level of coastal squeeze of a wetland is an informative indicator of the restorability and permanence of wetlands. As it has both present and future implications, the threat of coastal squeeze should be considered in prioritizing sites for restoration to prevent investments in sites that have a limited future. Wetland areas under low and medium threat with a high potential for migration are ideal for generalized and long term restoration. On the other hand, wetlands under high threat require specific and near term management. In these wetlands, prioritization could be based on their potential to provide ecosystem services (e.g., important to migratory pathways, recreation, or as a storm buffer). Generalized intervention may include preventing further development or keeping the present land use of the migration space undeveloped. Systematic allocation of the coastal zones will also be necessary at some point if the risk of sea level rise is considered. More specific interventions may include restoring hydrological connectivity or enhancing sediment supply for sediment-deprived but potentially resilient habitats.

4.5.3 Implication to ecosystem services provision

Preventing inland migration with sea level rise could lead to degradation of coastal habitats and their functions (Crooks 2004). The degradation and loss of wetlands along migration corridors might endanger population of migratory waterfowl and diadromous fish. If large areas of marshes and mangrove are lost, the small areas remaining may not effectively protect communities from storms. Maintaining coastal water quality is another important function of tidal wetlands which can diminish with limitation of migration space. For many years, tidal wetlands have been storing carbon in their soils, if mangrove and salt marshes are eroded and inundated by accelerated sea level rise, the stored carbon could be oxidized and released back to the atmosphere.

Most of the wetlands under high threat are small in extent so it is possible that coastal squeeze may result to habitat fragmentation. With coastal squeeze, the size, shape and spatial distribution of mangroves and salt marsh patches can be modified in such a way that they may no longer be suitable as habitats for many endemic, birds and fish (Hitch et al. 2011; Tomaselli et al. 2012). In addition, fragmented wetlands may affect the predator-prey dynamics especially between fishes that uses marshes and mangroves (Cooper et al. 2012). A recent study emphasizes the increasing role of mangroves as shelter for reef forming organisms facing thermal stress (Yates et al. 2014), so a reduction of mangrove and salt marsh coverage due to fragmentation and coastal squeeze would diminish this important function. Although habitat fragmentation is recognized a major conservation challenge with rising sea level (Jones et al. 2009) strategies to manage wetland fragmentation are yet to be mainstreamed in mangrove and salt marsh restoration programs.

4.5.4 Limitations and caveat

The robustness of the coastal squeeze index at the regional or global scale is limited principally by the spatial resolution and vertical accuracy of the global elevation. In up-scaling the index to the North America, I found that the elevation data used underestimates or truncates the slope in narrow areas with steep slopes. To illustrate, an area with a slope of 11.5º in the LiDAR-based elevation has a slope of 1.5 º in the SRTM DEM. Consequently, the confidence of estimating coastal squeeze by slope in these areas will be lower and might be underestimated. In British Columbia for instance, the marshes under high threat might be bigger than what is currently estimated. A better elevation dataset (i.e., finer vertical and horizontal resolution) such as those obtained by radar interferometer would improve the analysis. In addition, coarse elevation data does not cover areas below mean sea level or close to the water edge where most of the habitat polygons are situated. This compels us to interpolate the results to assign values to areas with no data. This problem of absence of data on the near-shore area is a persistent one that can be resolved by having seamless topographic and bathymetric data. Incomplete mapping of wetlands also contributes to the limitation of this study. For example, Mexican and Canadian marshes are underrepresented as the mapping was incomplete. In both countries the threat classes remain under-estimated

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Table 4-1. Sources of spatial datasets on habitat and protected areas used in the analysis of coastal squeeze

Habitat Area	Source Database	Creation Method	Temporal Range	Minimum Mapping Unit (m)	Active Link
			Canada		
British Columbia salt marsh	British Columbia ShoreZone 2014	Aerial photo interpretation and satellite image classification	2014	ND	ftp://ftp.gdbc.gov.bc.ca/pub/world/Coastal
New Brunswick salt marsh	GEONB Wetlands	Aerial photo interpretation and satellite image classification	1981-2009	ND	www.snb.ca/geonb1/e/DC/catalogue-E.asp
Prince Edward Island salt marsh areas	2000 Wetland Inventory	Aerial photo interpretation and satellite image classification	to 2000	175	www.gov.pe.ca/gis/index.php3?number=77555&lang=E
Nova Scotia salt marsh areas	Nova Scotia Wetland Inventory	Aerial photo interpretation and satellite image classification	1980; 1990	100--	http://novascotia.ca/natr/wildlife/habitats/wetlands.asp
Quebec salt marsh areas	St. Lawrence Wetlands	Aerial photo interpretation and satellite image classification	1990-2002	4	www.mcgill.ca/library/find/maps/stlwetlands
			Mexico		
Pacific coast salt marsh	United States Geological Survey c/o Ward, David <dward@usgs.gov>	Aerial photo interpretation and satellite image classification			none
Mangroves	CONABIO (ComisiÓn Nacional Para El Conocimiento Y Uso de la Biodiversidad)	Spot 5 image classification	2007-2011	10-500	http://www.conabio.gob.mx/informacion/gis/?vns=gis_root/biodiv/monmang/manglegw
			United States		
Salt marsh*	National Wetlands Inventory	Aerial photo interpretation	1983-2009	< 250	http://www.fws.gov/wetlands/Data/Data-Download.html
Louisiana salt marsh	United States Geological Survey	Aerial photo interpretation	2013	No Data	http://pubs.usgs.gov/sim/3290/
Mangroves	United Nations Environmental Programme-World Conservation Monitoring Centre	Classification of Landsat imagery (Giri et al. 2011)	1997-2000	<30	http://data.unep-wcmc.org/datasets/21

Table 4-1 continuation…

Habitat Area	Source Database	Creation Method	Temporal Range	Minimum Mapping Unit (m)	Active Link
		Protected areas and restoration sites			
Marine protected areas	Commission for Environmental Cooperation North American Environmental Atlas				http://www.cec.org/Page.asp?PageID=924&ContentID=2867
Terrestrial protected areas	Commission for Environmental Cooperation North American Environmental Atlas				http://www.cec.org/Page.asp?PageID=924&ContentID=2979&AA_SiteLanguageID=1
CONABIO (ComisiÓn Nacional Para El Conocimiento Y Uso de la Biodiversidad)					http://www.conabio.gob.mx/informacion/gis/maps/geo/sitpriogw.zip

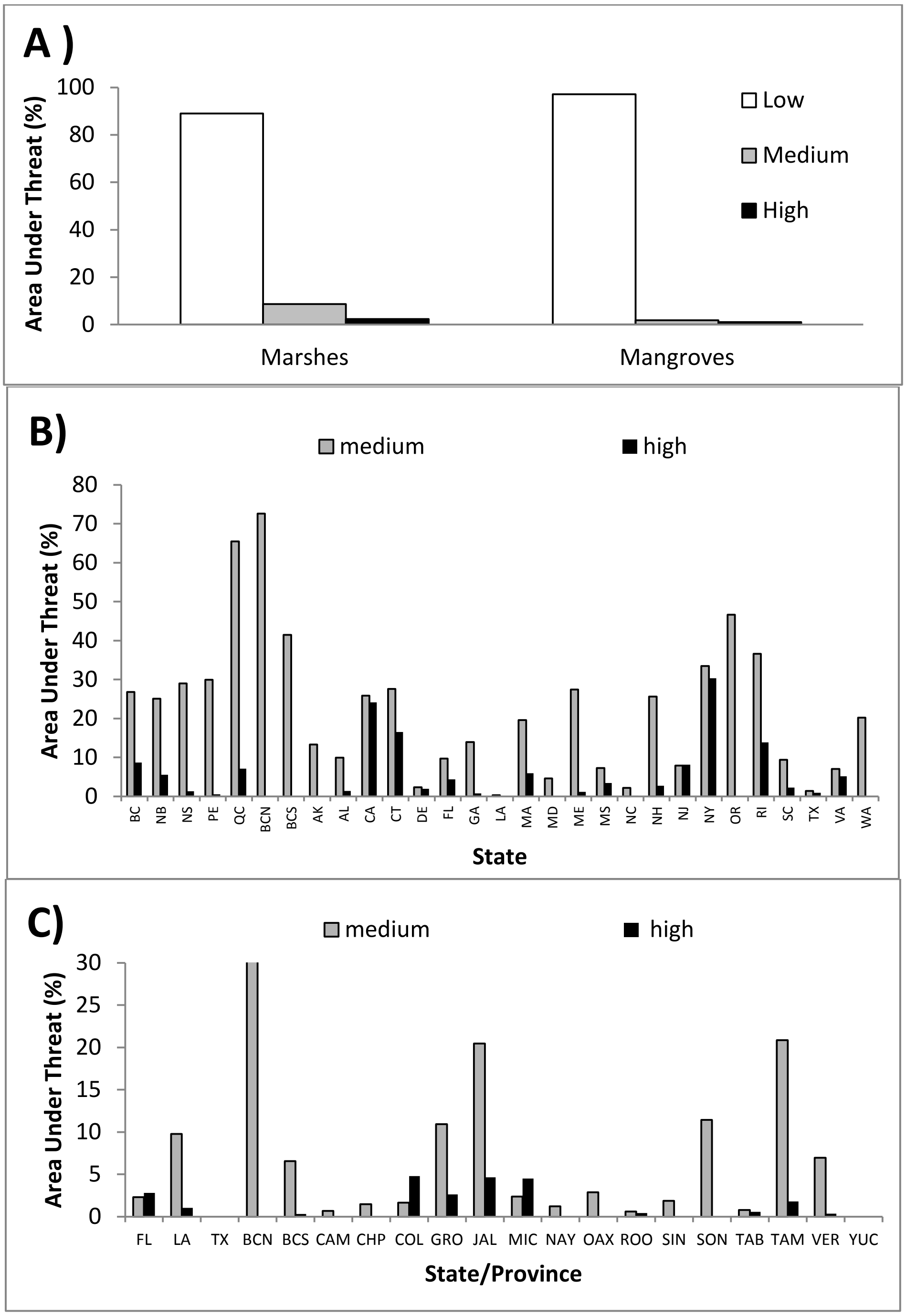
*All states except Louisiana

Table 4-2. Number of sites (i.e., restoration sites, terrestrial protected areas and marine protected areas) under different threats of coastal squeeze

Threat Level	Coastal Squeeze Threat Variables: Slope	Coastal Squeeze Threat Variables: Imperviousness	Cumulative Threat
Gulf of Maine restoration sites (Maine, USA, Total sites = 1084)			
Low	1037	1037	847
Medium	43	18	204
High	4	29	33
Bay of Fundy restoration sites (Nova Scotia and New Brunswick, Total sites = 322)			
Low	302	293	228
Medium	19	2	65
High	1	28	29
CONABIO mangrove restoration sites (Mexico, Total sites = 81)			
Low	79	79	77
Medium	2	1	3
High		1	1
Terrestrial Protected Areas (North America, Total sites=4358)			
Low	4325	4185	3600
Medium	31	75	621
High	1	136	137
Marine Protected Areas (North America, Total sites = 821)			
Low	828	764	584
Medium	12	36	197
High		40	40

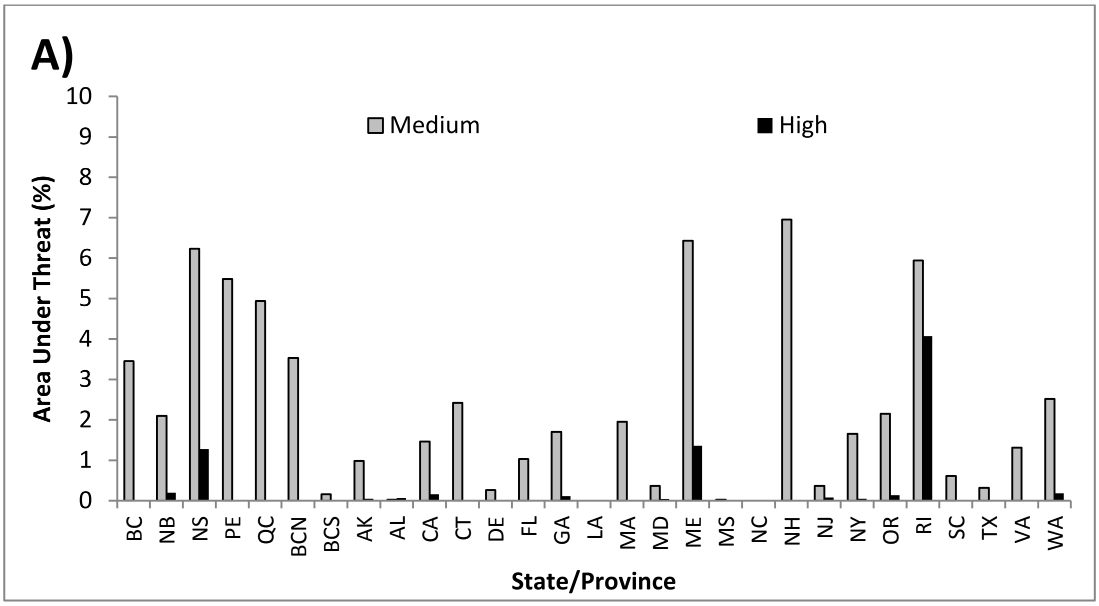
Figure 4-1. Distribution of salt marsh (blue) and mangroves (yellow) in North America. Green represents areas where the two wetlands occur in close proximity.

Figure 4-2. Percentage tidal wetland area under threat of coastal squeeze (A) and the distribution of area under cumulative threat of coastal squeeze in salt marshes (B) and mangroves (C) by state or province.



Note: QC=Quebec, NB=New Brunswick, NS=Nova Scotia, PE=Prince Edward Island, ME=Maine, NH=New Hampshire, RI=Rhode Island, CT=Connecticut, NY=New York, NJ=New Jersey, DE=Delaware, MD=Maryland, VA=Virginia, NC=North Carolina, SC=South Carolina, GA=Georgia, FL=Florida, AL=Alabama, MS=Mississippi, LA=Louisiana, TX=Texas, BC=British Columbia, AK=Alaska, WA=Washington, OR=Oregon, CA=California, BCN=Baja California Norte, BCS=Baja California Sur, TAM=Tamaulipas, VER=Veracruz, TAB=Tabasco, CAM=Campeche, ROO=Quintana Roo, MIC=Michoacán, GRO=Guerrero, OAX=Oaxaca, SIN=Sinaloa, CHP=Chiapas, JAL=Jalisco, COL=Colima, SON=Sonora, SIN=Sinaloa and NAY=Nayarit

Figure 4-3. Percent area under coastal squeeze by slope in salt marsh (A) and mangroves (B). (See Figure 4-2 for explanation of state and province abbreviations.)



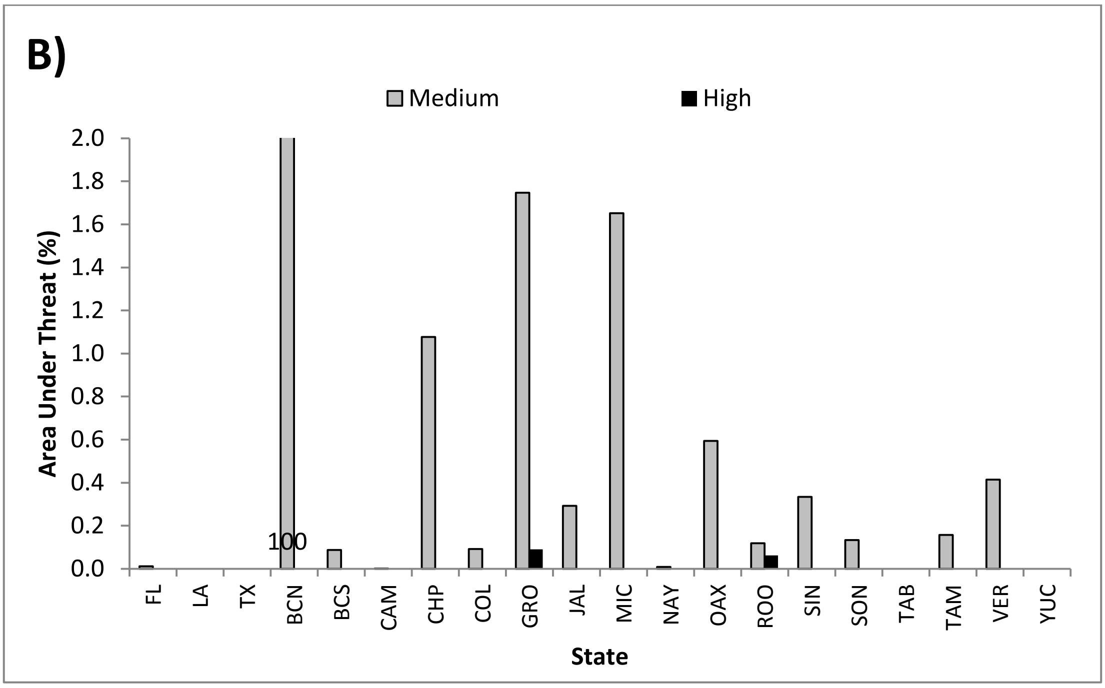


Figure 4-4. Habitats areas (red) threatened by coastal squeeze from steep slope: salt marshes in Quoddy Narrows, Lubec, Maine (A); Pettaquamscutt Cove, Narragansett, Rhode Island (B), USA; and mangroves in Laguna Bacalar, Buena Vista, Quintana Roo, Mexico (C)

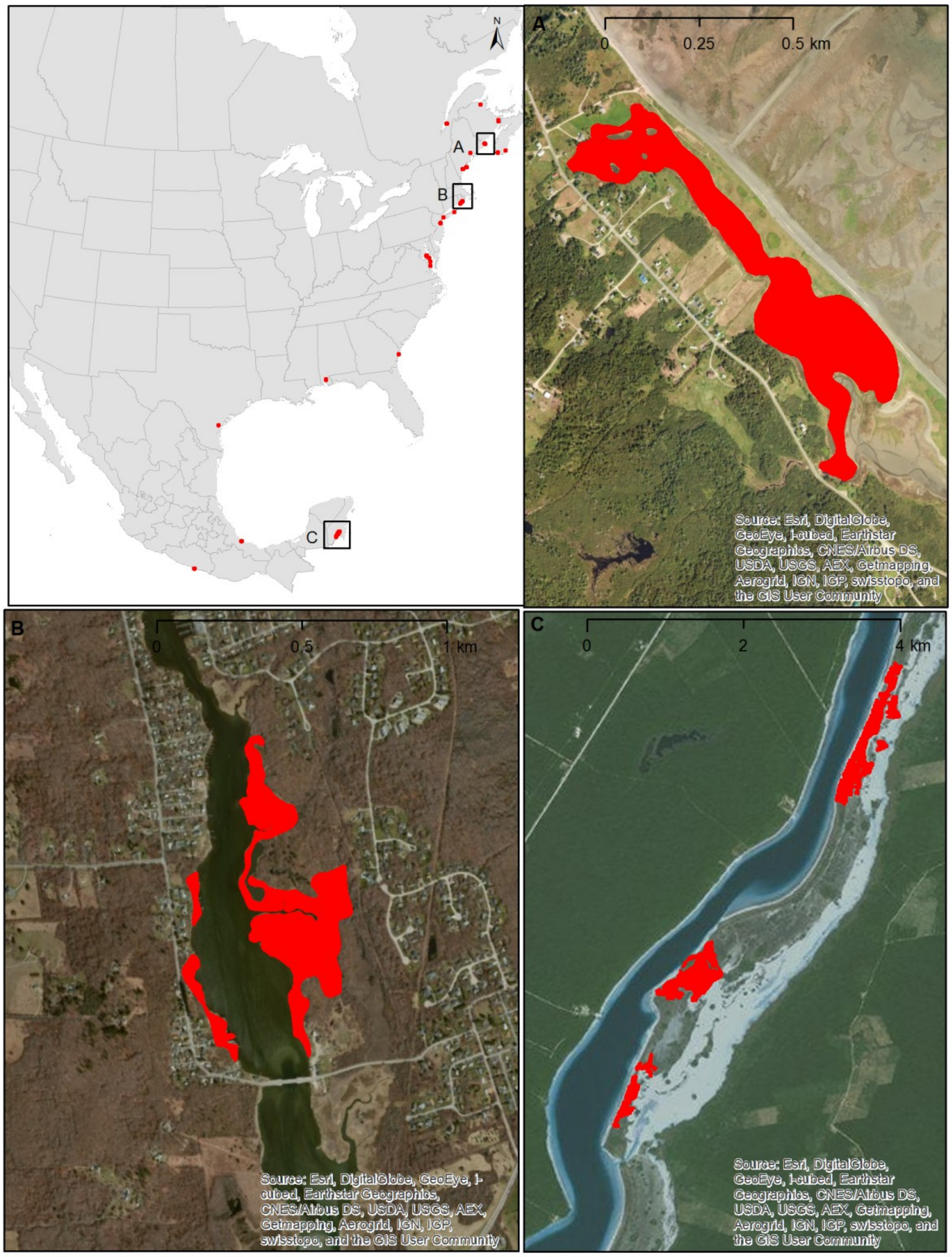
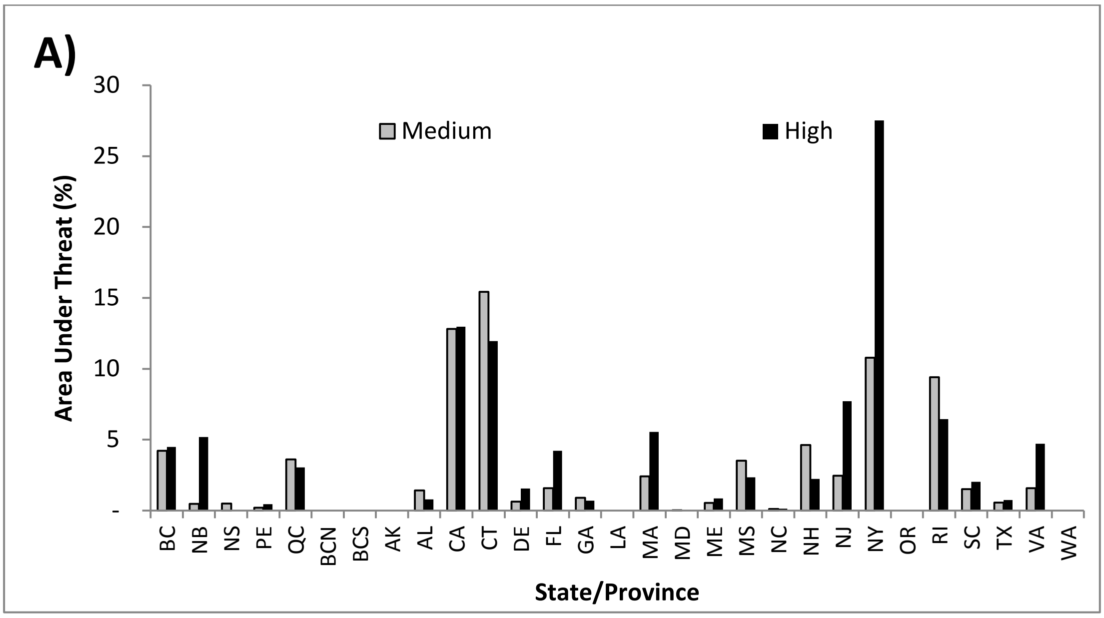


Figure 4-5. Percent area under coastal squeeze by imperviousness in salt marshes (A) and mangroves (B). (See Figure 4-2 for explanation of state and province abbreviations.)



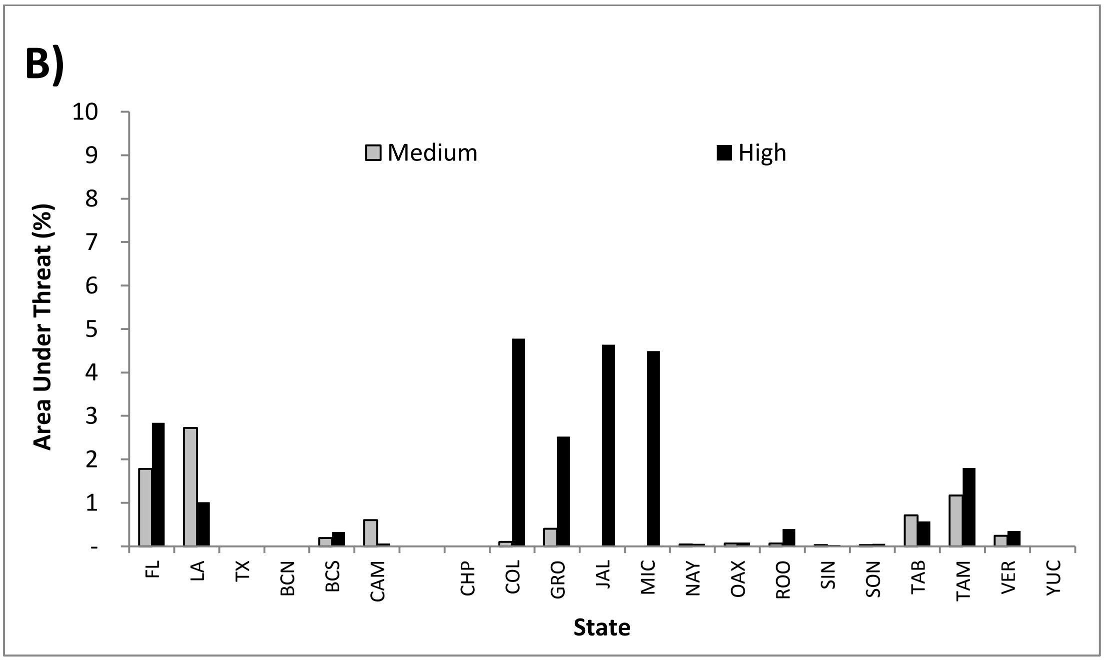


Figure 4-6. Tidal wetland areas (red) threatened by coastal squeeze from imperviousness, e.g., in the USA, salt marshes Jamaica Bay, New York (A); San Francisco Bay, California (B); mangroves in Tampa Bay, Florida, USA(C); and Porto Vallarta, Jalisco, Mexico (D)

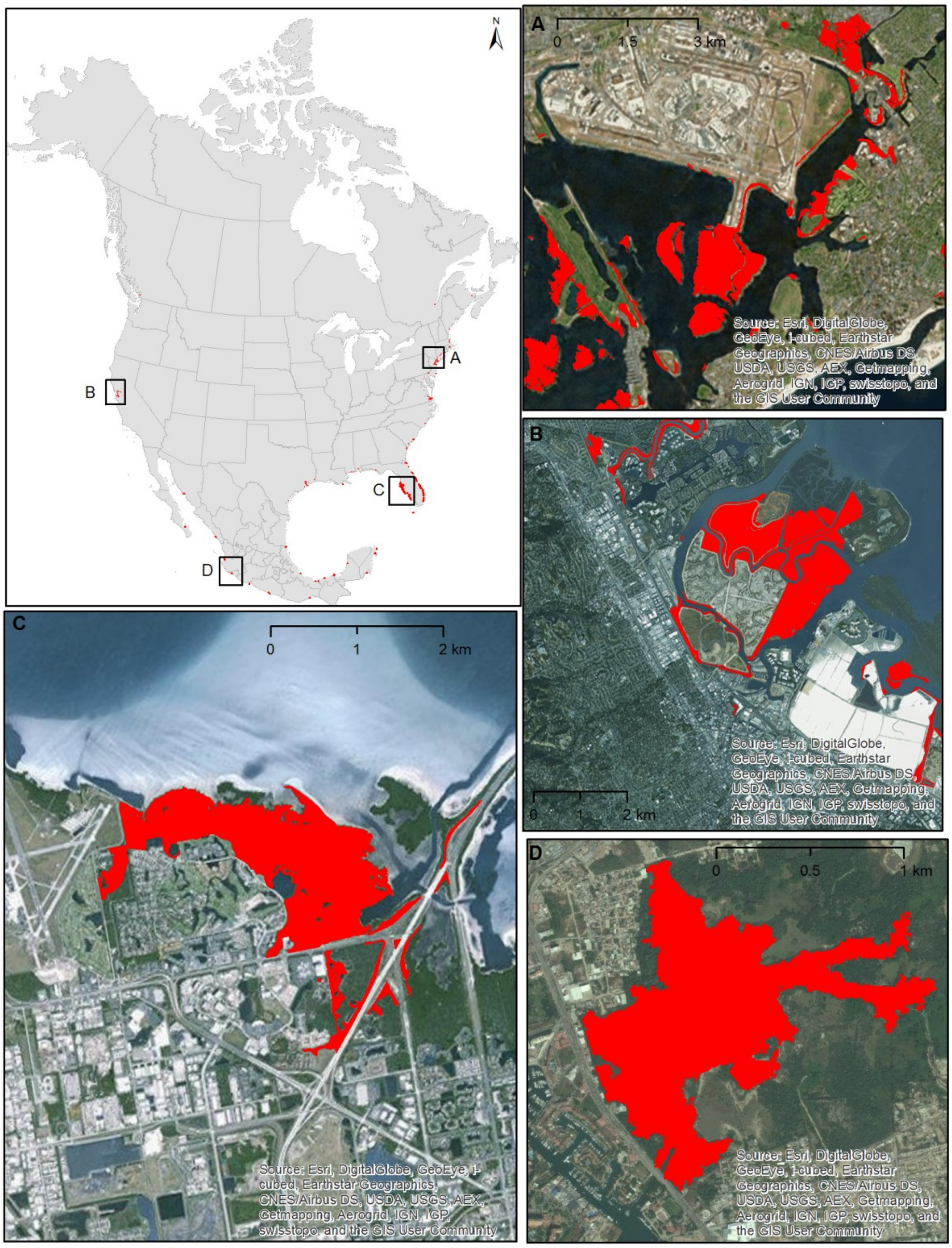
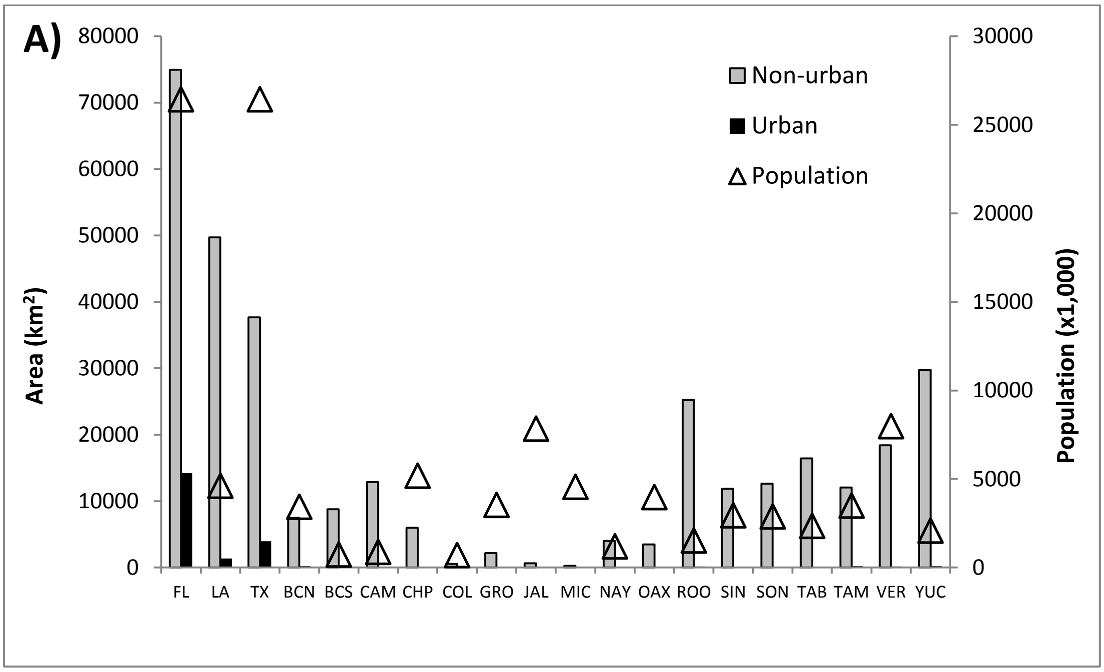
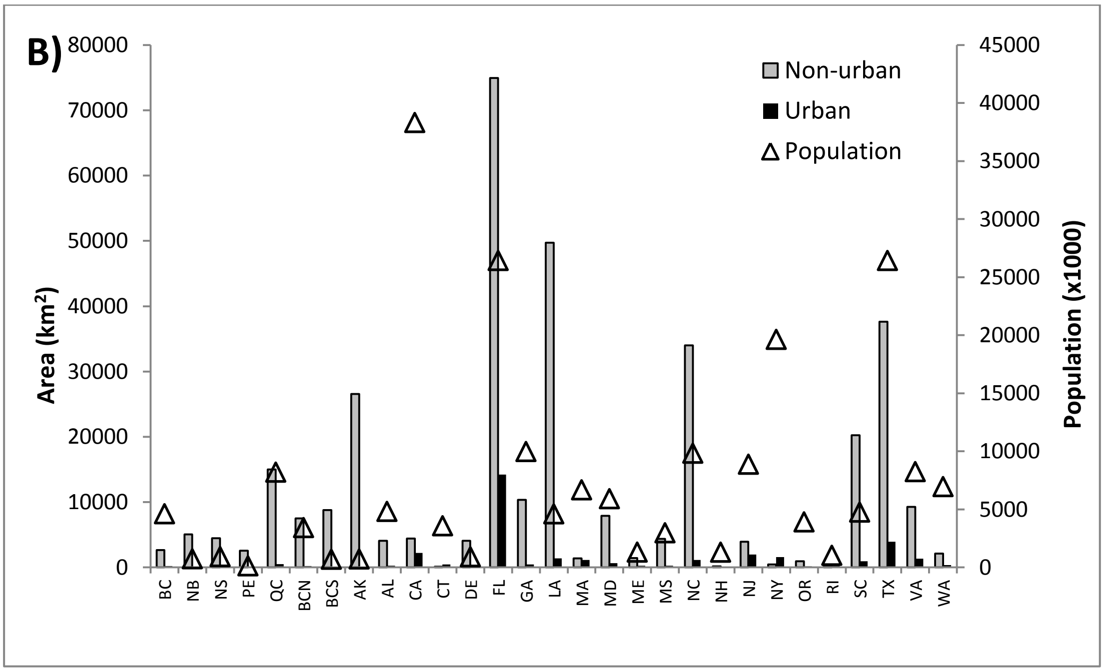


Figure 4-7. Distribution of low lying coastal areas (<30 m elevation) of North America and corresponding population. Low lying areas associated with mangroves (A) and salt marshes (B). (See Figure 4-2 for explanation of state and province abbreviations.)





Source: Population estimates: Mexico (Instituto Nacional de Estadistica Y Geografia 2010), Canada (Statistics Canada 2014), USA (United States Census Bureau 2014)

Chapter 5 Impacts of sea level rise on marsh as fish habitat

5.1 Preface

5.1.1 Manuscript detail

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5.1.2 Context and link to the previous and subsequent chapter

Whereas quantifying the threat of coastal squeeze provides vital information about the permanence of tidal wetlands with sea level rise, landscape-scale properties of the wetland surface could provide quantitative indicators of the quality and quantity of ecosystem services or functions, a major interest in conservation policies. As such, complementing vulnerability assessment with landscape ecology analysis will improve not only the knowledge on wetland adaptation, but will increase the confidence of decision makers in prioritizing conservation and restoration programs. Development of a spatial modelling approach therefore brings novel perspectives to our understanding of the response of tidal wetlands to global sea level rise. A demonstration of this is the value of the marsh as fish habitat. Fishes when compared to other marsh dependent fauna such as birds or insects are highly dependent upon channels and access from marsh edges to feed or use marsh edges as a refugia, thus are sensitive to changes in marsh configuration and fragmentation expected with sea level rise, hence the choice to model their habitat.

The next two chapters delve deeper into the consequences of sea level rise and coastal squeeze at the ecosystem level to answer the thesis question: how will tidal wetland functions fare with expected rates of sea level rise over this century? The last two chapters focused on quantifying coastal squeeze over potential migration areas of tidal wetlands with sea level rise. In the next two chapters, I examine the impacts of sea level rise and coastal squeeze on the structural properties of the wetland surface as they relate to a specific ecosystem function. The modelling approach builds upon insights gained in Chapter 3. In this chapter the inundation model and extent submodel developed in Chapter 3 are modified and applied. The inundation model is modified to include an accretion component and encapsulated in an ArcGIS model. This new model is used to generate modeled elevations under two assumptions of accretion rates: an accretion rate that equals the rate of sea level rise and a constant accretion rate. The model outputs were used as inputs in the modified extent submodel to calculate future mudflat, creeks and marsh extents based on elevation change. The outputs from these submodels were then analyzed using several landscape ecology metrics that evaluate changes in spatial distribution, configuration and connectivity of marsh patches as they relate to fish habitat quantity, quality and accessibility.

5.2 Introduction

Since 1980, ~48 % of the world's tidal wetland area has been lost (Coleman et al. 2008; Lotze et al. 2006; Valiela et al. 2001; Waycott et al. 2009) resulting in a ~69 % reduction in the habitat nursery and 63 % in the filtering services provided by tidal wetlands (Worm et al. 2006). The rapid increase in sea level rise associated with global warming is expected to further reduce tidal wetland area and its ecological functions (MEA 2005). In addition, urban sprawl along the coast poses a constant threat to the already diminishing tidal wetlands.

A minimum sea level rise of 0.6 m by the end of the century (Bindoff 2007; Nerem et al. 2010; Pfeffer et al. 2008; Vermeer and Rahmstorf 2009) will likely drive efforts to protect valuable public and private lands (Feagin et al. 2010) immediately inland of tidal wetlands. As a result, protection of properties from sea level rise could put tidal wetlands in a state of increasing “coastal squeeze” (Doody 2004; Schleupner 2008; Torio and Chmura 2013) preventing tidal wetlands from migrating inland, a process which would otherwise alleviate some of the loss of wetland area through sea level rise. Tidal wetlands facing coastal squeeze may become more degraded and functionally impaired.

The impact of sea level rise on tidal wetlands has been examined with respect to changing marsh area (Craft et al. 2008; Rogers et al. 2014; Traill et al. 2011) and shifts in marsh vegetation (Craft et al. 2008) as well corresponding shifts in habitat for birds (Brittain and Craft 2012) and mammals (Traill et al. 2011). For example Craft et al. (2008) developed the Sea Level Rise Affecting Marshes Model (SLAMM) to predict shifts in salinity zones of tidal wetlands and predicted that at the end of this century about 20 to 45 % of the total salt marsh area will be lost if sea level rises to the mean and maximum rates projected by IPCC, respectively. They claim that this will significantly reduce biomass production and nitrogen absorption. In subtropical tidal wetlands, Traill et al. (2011) predicted that decreasing marsh area with sea level rise would increase the threat of extirpation and predation risk of a native rodent *Xeromys myoides*. In the USA, Brittain and Craft (2012) modelled that shifting coastal habitats with sea level rise may reduce suitable habitats and populations of coastal birds. In a recent study, Kirwan and Mudd (2012) suggest that if marsh areas will decrease with sea level rise, their capacity to store atmospheric carbon will significantly decline. All these studies relate the loss of ecosystem services mainly with decreasing areas of tidal wetlands. However, I am not aware of any who have addressed the impacts of sea level rise and on marsh configuration, particularly as it applies to fish habitat like I had in this study.

Rising sea levels are likely to affect tidal wetland habitat in other ways, which have yet to be effectively explored in spatial models. For instance, sea level rise and land development may modify the structure, configuration, spatial distribution, and accessibility of wetlands to fauna. Such landscape-scale modifications may have far-reaching consequences that are just as important as those caused by reduction in area. Earlier works by Dionne and colleagues (Dionne and Dochtermann 2006; Dionne et al. 2006; Dionne et al. 1999) at Wells marsh in Maine (USA) hinted that land development surrounding marshes can have positive and negative impacts on fish density and biomass. These impacts are likely to intensify with time and sea level rise. In other areas, tidal marshes dynamically respond to prolonged hydro-period from wind and wave action by migrating inland even if there is no sea level rise (Wasson et al. 2013). This demonstrates that as an ecotone, tidal marshes are sensitive to environmental changes. Addition of other stressors such as coastal squeeze (Torio and Chmura 2013), climate warming (Pachauri and Reisinger 2008), and nutrient enrichment (Deegan et al. 2012) will therefore com-pound the impact of sea level rise on tidal wetland ecological functions.

The functions of a tidal wetland are intricately linked with its spatial and structural characteristics. For example, marshes provide food and refugia in patches of different sizes, location, quantity, and quality. They sustain predator and prey relationships (Boesch and Turner 1984a; Halpin 2000) and link estuaries to open coastal waters (Kneib and Wagner 1994; Peterson and Turner 1994) through exports of secondary production. Species endemic to the marshes use the marsh interior and edges (Greenberg et al. 2006). Proximity and connectivity of patches of emergent vegetation to other intertidal or subtidal environments increase density and biomass of shrimp and fish (Irlandi and Crawford 1997; Pittman and McAlpine 2003) and sustain other species that use the system at some point in their life stages (Kocik and Ferreri 1998). Complexity in patch shape and configuration of wetland edges relate to the quality of wetland refugia and its capacity to sustain interactions between transient and resident species (McGrath 2005; Minello et al. 1994; Peterson and Turner

1994). As in terrestrial ecosystems (Fisher et al. 2011; Holling 1992), the diversity in size of habitat patches may be used by different organisms with different body sizes. To better appreciate its ecological function, we must understand the structural limits of tidal wetlands and its subhabitats (Rountree and Able 2007) and how they are spatially and temporally organized.

Because sea level rise and coastal squeeze can reorganize and redistribute tidal wetlands, it is necessary to adopt a technique that can be used to assess both system-wide and local spatial patterns. In this study, I adopt a landscape ecology approach (Turner 1989) which provides the means to evaluate how spatial configuration and connectivity (Beger et al. 2010a; Beger et al. 2010b; Berkström et al. 2012; Lindenmayer et al. 2008; Sheaves 2009) can be modified by coastal squeeze and sea level rise. Landscape level analyses help us to determine changes in spatial patterns, how habitats are organized, and how their changes affect ecological functions such as provision of food, refugia, and corridors of movement within those habitats (Morris 2012; Turner 1989).

Several landscape metrics are available to quantify spatial patterns and connectivity (McGarigal and Marks 1995). For example, the fractal dimension reveals the shape complexity of a wetland patch, which in turn, affects the ability of organisms to disperse and interact (Forman 1995). With respect to wetlands, a decreasing fractal dimension indicates loss of shape complexity which is usually associated with a landscape heavily modified by humans (O'Neill et al. 1988). In the terrestrial domain, indices of interspersion and edge density have been used to differentiate patterns of land cover patchiness in urbanizing landscapes (Cifaldi et al. 2004). On tidal wetlands, interspersion and edge density could be used to quantify the changes in marsh patch spatial distribution and refuge potential, respectively. Patch carrying capacity or the “ecologically scaled landscape index” (ESLI) is a derivative of patch size which can be used to determine the minimum habitat requirement of a species (Vos et al. 2001). Connectivity metrics such as least cost distance and circuit theory (McRae et al. 2008) have been used to measure landscape “permeability” enabling the identification of priority conservation corridors and sections of the landscape where movement of plants and animals are severely restricted (Beier et al. 2011; Brost and Beier 2011; Carroll et al. 2012). Studies of terrestrial ecosystems show that when habitats are connected and the landscape is permeable, biodiversity at all levels is enhanced (Fahrig et al. 1995; Tischendorf and Fahrig 2000). The application of the landscape ecology approach is still in its infancy in estuarine systems, and to my knowledge, there are no studies that have used the landscape approach to understand the impacts of coastal squeeze and sea level rise on tidal marshes.

In this study, I test the hypothesis that sea level rise and accompanying coastal squeeze will drive changes in marsh patch configuration and connectivity which affect the utilization of the marsh by fish. I investigate how sea level rise and coastal squeeze affect the spatial distribution, configuration, and connectivity within tidal marshes in a landscape context. Using spatial analyses, I compare two outputs from a process model: one with constant vertical accretion and the other with a more optimistic prediction that rates of vertical accretion will equal rates of sea level rise. I use landscape metrics to evaluate the changes that occur in a case study based at the marshes of the US National Estuarine Research Reserve (NERR) site at Wells, Maine - a site with considerable available data and where research on coastal squeeze (Torio and Chmura 2013) already has been completed.

The inundation model and extent submodel developed in chapter 1 are adapted and modified to generate the required inputs for landscape ecology analysis. The modifications include addition of accretion component to the inundation model and encapsulation in an ArcGIS model. This new model is used to generate modeled elevations under the two assumptions of accretion rates. The model outputs were used as inputs in the extent submodel elevation to calculate future mudflat, creeks and marsh extents hence the marsh extent submodel developed in chapter 1 was used to simulate marsh extent and modified to a new submodel by changing the conditional threshold statement to simulate subtidal extent.

5.3 Methods

5.3.1 Study site, data, and spatial extent

The study is based on marshes around Webhannet Lagoon and Little River Estuary in Wells, Maine (Figure 5-1). Both estuaries have diadromous fish runs (Dionne and Dochtermann 2006). Like many marshes in the southern Gulf of Maine, those at Wells are fragmented by roads and causeways. Small culverts limit normal tidal flow in and out of the estuary and in some parts of the lagoon (Eberhardt et al. 2011). The uplands bordering intertidal areas have gentle to moderate slopes under different intensities of land use. Both estuaries and the lagoon are rich with transient and resident fauna. Dionne and Dochtermann (2006) recorded 57 species of fish and 15 of these were found to directly use the marshes.

I used multiple sources of spatial data to characterize the study site. For topographic data, I obtained digital elevation models (DEM) from the Coastal LiDAR database (NOAA 2012b). The DEM of Wells has a vertical accuracy in terms of 0.273 m (linear error at 95 % confidence) at 3 m spatial resolution expressed in terms of root mean square error (RMSE). Additional characteristics of the coastal DEM are reported elsewhere (Maune et al. 2007; National Oceanic and Atmospheric Administration 2010; Torio and Chmura 2013). In addition to the DEM, I used high-resolution aerial photos to manually update the boundaries of developed areas and property lines extracted from existing land cover maps of Wells and to verify the results of the modelling processes. All the spatial data were projected in the North American Datum of 1983 Universal Transverse Mercator Zone 19 North (NAD_1983_UTM_Zone_19N).

I limited the extent of the analyses to the present and future intertidal areas and their immediate surroundings. The intertidal area is defined as all areas equal to or below the elevation of the upper edge of the marsh. In Wells, this elevation is equal to 1.95 m (NAVD88) or 0.50 m below the highest astronomical tide (HAT). The smallest spatial resolution of the dataset is 3×3 m and the smallest unit of analyses is a patch (Forman 1995).

5.3.2 Process model development and assumptions

Using ArcGIS Model Builder, I developed three spatial models that take elevation, sea level rise rates, and accretion rates as inputs. For convenience, I called these models ACCSLSIM [Accretion and Sea Level Simulator, a modified version of inundation model developed in Chapter 1 (Appendix Figure 5-9)], TIDEXSIM, modified marsh extent submodel developed in chapter1 [Intertidal Extent Simulator (Appendix Figure 5-10)], and MEXSIM [Marsh Extent Simulator, marsh extent submodel developed in chapter 1, (Appendix Figure 5-11)]. The ACCSLSIM begins with the original DEM and produces a new DEM which together with the original DEM RMSE feeds into TIDEXSIM and MEXIM to calculate the extent of the intertidal zone and marsh, respectively.

I assumed sea level rise to increase linearly within a 100-year period and that vertical accretion rates decrease linearly with distance to the nearest creek as shown in field studies in the Bay of Fundy in the upper reaches of the Gulf of Maine (Chmura and Hung 2004). I used the ACCSLSIM to generate a new DEM at five different rates of sea level rise (no rise, 0.32, 0.50, 1.00, and 1.50 cm year^{-1}, each starting at time 0 and remaining constant throughout the 100 years) and two rates of vertical accretion: one at constant rate of 0.36 cm year^{-1} and the other at rates equal to rates of sea level rise. ACCSLSIM works in the following steps: first, the model maps the location of the creek from the original DEM by assigning a value of 1 to the areas with elevations less than the lowest elevation of the Wells marsh with emergent vegetation. The distance to the edges of creeks was calculated for every pixel and used to calculate vertical accretion as a linear function based upon measurements by Chmura and Hung (2004) for Bay of Fundy marshes. The variable "distance to the nearest creek" in the model will account for the gain or loss in area at the seaward edge as the elevation changes with sea level. Fourth, the accretion values are added to the original DEM before subtracting the sea level at a given rate. The process was repeated for each year within the 100-year period producing a new DEM adjusted to both sea level rise and accretion. The final results are 10 future DEMs representing the five sea level rise rates and two accretion rate assumptions.

I used the original DEM and nine modeled DEMs as inputs in TIDEXSIM and MEXSIM. The DEM RMSE in TIDEXSIM and MEXSIM is assumed to have a normal distribution with 0 mean and standard deviation equal to the RMSE (±0.20) and is auto correlated at a small distance (i.e., ~9 m). Both models create a random field with values taken from the distribution of the RMSE. Then, they apply a 3×3 averaging filter to enhance autocorrelation. The auto-correlated errors are then added to the original DEM. A condition statement classifies the DEM into the desired outputs based on a set elevation range. This is where the two models slightly differ. In TIDEXSIM, the desired output is the intertidal zone extent so a conditional statement assigns 0 to non-intertidal areas and 1 to intertidal areas (i.e., areas below 1.95 m). In MEXSIM, the desired output is a vegetated marsh extent so the conditional statement is slightly modified to assign 1 to marsh areas and 0 otherwise. The process is looped and repeated until a stable result is reached. At the given resolution and uncertainty of the DEM, I found that 100 iterations are optimum. The resulting map after iteration divided by the number of iterations (×100) represents a per-cent confidence.

5.3.3 Mapping intertidal subenvironments

Using the output of TIDEXSIM, I selected and classified the slope and elevation of all areas above 90 % confidence of the intertidal zone extent using an iterative optimization technique developed Brost and Beier (2011) and Jenness et al. (2011). I assumed that the slope and elevation of marshes are separable from the slope and elevation of other intertidal environments such as mudflats and creeks. First, the classification parameters were estimated by importing the elevation and slope in R (R Development Core Team 2010) and using a script (Jenness et al. 2011) to optimally cluster the data. The clustering technique uses a Fuzzy-c mean algorithm (Bezdek 1981; Dimitriadou et al. 2009) to determine the optimum number of classes and calculate the parameters. The parameters were used in a geographic information system to classify the slope and elevation of the intertidal zone into distinct subenvironment at each sea level and accretion scenario using the same confidence threshold. For each of the classified maps, I determine the total area of each subenvironment and the percent marsh area that overlaps with developed areas.

To evaluate its accuracy, I compared the classified map to the map of confidence-weighted similarity index. To calculate the similarity index, the marsh class produced from classification was separated from the rest of the classes and given a value of 1 using a reclassification technique in ArcGIS. Then, the slope and elevation of the marsh class was extracted into a database table using the "Export to R" function in Land Facet Tool (Jenness et al. 2011). Next, I calculated the mean, standard deviation, and a variance/covariance matrix of the slope and elevation in MS Excel and imported these parameters in ArcGIS as vectors to calculate normalized Mahalanobis distances (Clark et al. 1993). The Malahanobis distances were converted to chi-square probabilities and multiplied by marsh extent confidence derived from the MEXSIM. The product is a map of confidence-weighted similarity index. Finally, I extracted and averaged the confidence-weighted similarity values of the marsh class.

5.3.4 Landscape analysis

The marsh classes were selected and converted into polygons with the areas that overlap with developed land use or property lines clipped out. To eliminate the pixilated effect of converting marsh patches from raster to polygon, I smoothed the edges of the resulting polygons using polynomial approximation with exponential kernel (Bodansky et al. 2002) with a tolerance of twice the linear resolution of the pixels. The final polygons were used to compute several landscape metrics.

I reviewed several landscape metrics (Rutledge 2003; Šímová and Gdulová 2012; Tomaselli et al. 2012) and chose those indices that are easy to interpret, scale-invariant, and relevant to the ecosystem and organisms being studied (Table 5-1). I calculated ESLI for two fish known to utilize the Wells marsh using minimum home range sizes of 113 m^2 for the mummichog, *Fundulus heteroclitus* (Lotrich 1975), and 30,000 m^2 for striped bass, *Morone saxatilis* (McGrath 2005).

Culverts severely restrict water and fish movements, creating “pinch points” or bottlenecks in the estuarine system. Circuitscape (McRae et al. 2008), which calculates a resistance index and the degree of decrease in connectivity, was used to provide metrics to compare the severity of pinch points under the different scenarios. The previously computed normalized Mahalanobis distance was used as a resistance surface with marsh edges and subtidal areas having low values. Using the imperviousness data and aerial photos, I located developed land use such as buildings, culverts, and bridges and assigned them with high values. I assigned the width of culverts as 1 pixel and estimated the resistance values as average Mahalanobis distance of the surrounding pixels. With these variables, I simulated the movement of small and larger fishes like mummichogs and striped bass moving randomly on the estuary. Movement is modeled under the principles of circuit theory where a value on a landscape pixel represents the probabilities that a random mover would pass through that cell (Carroll et al. 2012; McRae et al. 2008). For each sea level, I calculated the average resistance index of movement on the estuary with and without the barriers. In addition, I also calculated the width of the estuary at different sea level rise rates using Corridor Designer Tool to determine the location of connectivity “bottlenecks.”

5.4 Results

5.4.1 Mapping marsh patches

The result of the classification indicates that there were three optimum classes corresponding to the three distinct subenvironments on the intertidal areas in Wells, namely, marsh, mudflat, and creek (Figure 5-2). The present area consists of 11 % mudflat, 18 % subtidal, and 71 % marsh with 1.3 % of the marshes inside private lands (Table 5-2). In the constant accretion model, marsh, mudflat, and creek areas increased when sea level reached 0.5 m by 100 years. When there was a 1-m sea level rise by 100 years, the areas of marshes and mudflats decreased while creek area increased. At a 1.5 m sea level rise by 100 years, 40 % of potential marsh areas will overlap with developed lands or exist beyond property lines. In the model where accretion rate equals the rate of sea level rise, the changes are less prominent. Both creeks and mudflats increased steadily while marsh areas showed a small de-crease only at 1.5 m sea level increase. When accretion rate is equal to the sea level rise rate, only 10 % of the marsh areas migrate on developed lands even at the highest sea level increase.

5.4.2 Sea level rise, marsh composition, and configuration

Sea level rise had variable effects on marsh size and shape. Marshes in the model with constant accretion displayed the most drastic change in area (Figure 5-3). In both models, the percent marsh area relative to the original increased (Figure 5- 3a) with increasing sea levels to 0.5 m (Figure 5- 3b). When accretion rates matched sea level rise, there was no reduction in marsh area with further increases in sea level, while marsh area decreased when accretion rates were kept constant. Under constant accretion, up to 40 % of the marsh area is lost when sea level rises between 1 (Figure 5-3c) to 1.5 m (Figure 5-3d).

At constant vertical accretion, the percent marsh relative to the original (Figure 5- 4a) and the average patch sizes (Figure 5- 4b) decreased with increased rates of sea level rise. Notably, at greater than 0.5 cm year^{-1} sea level rise, the average patch size dropped from 0.36 ha (3,600 m^2) to 0.09 ha (900 m^2). When vertical accretion rates were set equal to the rate of sea level rise, the average sizes of marsh patches remain stable. At 1 m sea level, the average patch edge density (Figure 5-4c) or perimeter per unit area increased. At higher sea levels, the edge density did not change in the model with accretion equaling sea level rise while the edge density sharply decreases in runs with constant accretion. As the marsh patches decrease with increasing sea level, the sizes become more uniform (Figure 5- 4d) in the constant accretion model. On the contrary, in the model with accretion equals sea level rise, variability in patch sizes remains constant.

The combination of sea level rise and coastal squeeze modifies the distribution and proximity of patches on the intertidal landscape. The average distance between neighboring patches increased when accretion is held constant while it changes little when accretion keeps up with sea level rise (Figure 5-4e). In both models, the marsh patches are nearly equally distant to creek and mudflats (i.e., high interspersion) at a 0.5 m increase in sea level (Figure 5-4f). With the constant accretion model, interspersion peaks at 1 m sea level rise, then decreases with continued sea level rises. When accretion rate matches the sea level rise rate, the interspersion showed minimal change. Decreasing interspersion results from the expansion of mudflat and creek and subsequent shrinking and isolation of marsh patches.

Patch size is directly proportional to patch carrying capacity (or ESLI) so that the ESLI for mummichog and striped bass increased at moderate sea levels as the patch size increased (Figure 5-5a, b). But at higher sea levels, ESLI decreases more in marshes with constant accretion than in the marshes accreting at the same rate as sea level rise which is about one third of the average size of the marsh patches produced by the model in which accretion equals sea level rise.

Change in patch shape is prominent in marshes with constant accretion. At more than 1 m, sea levels of both the upland (Figure 5-6a, b) and seaward edges (Figure 5-6c, d) of the marsh become less convoluted. This results in decreasing trend in fractal dimension of the landward and seaward edges of the marsh (Figure 5-7a, b) with increasing sea level. Notably, the fractal dimension decreases more in marshes with constant accretion while marshes that accrete with sea level rise maintain higher fractal dimension. At the Wells marsh, sea level rise and fractal dimension are inversely proportional.

5.4.3 Sea level rise, coastal squeeze, and connectivity

Anthropogenic barriers increase the resistance to movement of organisms in the estuary. Figure 5- 8 shows the location of the barriers most influential to movement represented as "hotspots" of movement resistance. In Figure 5- 8a, the entrance of the estuary is steep and narrow (bounding box 1) and a developed parking space (bounding box 2) disconnects two large marsh patches. A road across the estuary (bounding box 3) with a small culvert underneath limits tidal flow between two large sections of the marsh.

With the barriers, the modeled resistance index is high with a range of 0–0.73 (Figure 5-8b). Figure 5-8c shows the estuary without the road with a corresponding decrease in the overall resistance index (i.e., range 0–0.21). Anthropogenic barriers greatly reduced the width of the estuary (Figure 5- 8d). Resistance indices on narrow channels and steep slopes remain high as these cannot be inundated. At all sea levels, the average resistance index in marshes without anthropogenic barriers is lower (Figure 5-8e). Sea level rise may increase the width of the restrictions if these are allowed to be flooded or structurally modified.

5.5 Discussion

The model results support the hypothesis that coastal squeeze and accelerated sea level rise will lead to wetland degradation by modifying marsh spatial structure, subhabitat composition, configuration, and connectivity, all of which have important implications on how fishes use the marsh as habitats. The model results indicate that moderate sea level rise would benefit marshes by expanding their area and maintaining habitat spatial complexity, but higher sea levels coupled with stressors like coastal squeeze are likely to result in more dispersed patches with lesser edge complexity and connectedness. Under these conditions, the function of marshes to provide adequate source of food and refugia for a variety of coastal fishes and even terrestrial organisms is at greater risk of deterioration.

5.5.1 Present marshes expand at moderate sea level rise

At higher sea levels, there is a greater risk that the lower seaward edges of the Wells marsh will be submerged if there is a severe accretion deficit. If vertical accretion is adequate for marshes to maintain elevation in response to sea level rise, loss of marsh area and the effect of coastal squeeze are minimized. The results indicate that at a 0.5 cm annual increase in sea level at the current accretion rate, the Wells marsh will expand at the rate of 1.14 % annually but at more than 1 cm annual increase in sea level, it will deteriorate at an annual rate of 0.5 %. Other studies found a similar trend that marshes expand with moderate sea level rise (Feagin et al. 2010). Marshes under gradual submergence are at the stage similar to a deltaic marsh in a destruction phase (Scruton 1960) with increased secondary production but which eventually deteriorates if a threshold in sea level rise and accretion is exceeded.

5.5.2 Rapid sea Level rise and coastal squeeze induce marsh deterioration

When sea level rise exceeds 1 m and accretion remains constant, while most of the marsh expansion areas inland are blocked, creek expands, replacing marsh patches and the remaining marsh patches become smaller with simplified edge shapes and further apart. All the results from the simulations of high sea levels under constant accretion consistently show a decrease in patch size, carrying capacity, edge density, fractal dimension, interspersion and juxtaposition, and increase in the patch neighborhood distance. These measures of changes in marsh structural complexity provide empirical metrics of marsh fragmentation and deterioration. These metrics can be robust indicators of dynamic response of salt marshes to environmental change and wetland health at local and regional scales-a need raised by Dionne et al. (2006)-without having to resort to purely vegetation characteristics. Tidal marshes as ecotones may maintain their community structure when perturbed and during migration (Wasson et al. 2013), but not their spatial structure as this depends on suitable geomorphology, topography, and land cover.

5.5.3 Implications of marsh deterioration on fish habitats

At the current accretion rate, the carrying capacity of the Wells marsh is unlikely to be affected and may continue to support both resident (e.g., mummichogs) and transient fishes (e.g., striped bass) as long as sea level does not exceed 1 m. Changes in marsh spatial complexity have major ecological implications. Fish production may decrease (Macreadie et al. 2010) if large areas of marshes are lost (Herke et al. 1992; Peterson and Turner 1994). Economically and recreationally important species of marsh-dependent fish are mostly transient and predatory, feeding on marsh-resident fish (Deegan et al. 2002; Peterson and Turner 1994). At higher sea levels, marsh patches may shrink as tidal creeks expand, increasing patchiness and subsequently the risk of predation (Halpin 2000) for small fish and crustaceans that feed on the edges of the marsh, as they have to cover long distances to search for food and refugia (Dunning et al. 1992; Hansen and Quinn 1998; Kneib 1984; Kneib 1987; Schlosser 1995). This may favor the predatory fish species over their tidal marsh prey for a short period, as has been observed in oyster reef habitat where an increase in patchiness resulted in a spatial shift of prey foraging behavior, overexploitation, and a trophic cascade (Geraldi and Macreadie 2013; Macreadie et al. 2010; Macreadie et al. 2012).

The model results indicate that future marsh patches in Wells tend to be distant and less interspersed at higher sea levels and constant accretion. Low interspersion means more isolation of marsh patches. Inversely, high interspersion indicates that the different intertidal subenvironments may be accessible in all locations as they share common boundaries. A highly interconnected system has a high degree of interspersion. A highly interspersed or equally interspersed habitat patches on the landscape are likely to sustain an ecological process (Cowling et al. 2010; Fairbanks and Benn 2000; McKenzie et al. 1989) because they promote interaction between different species with the different subenvironments. In nature, marsh patches of variable sizes are equally interspersed with tidal creeks and mudflats. Interspersion in marsh is the result of complex processes that maintain the marsh such as tidal actions, variable distribution of sediment, differential flooding, and erosion within the dendritic networks of tidal creeks. It follows that a highly interspersed marsh has a high degree of proximity to other marsh patches and other subtidal habitats as indicated by short neighborhood distances. In Wells, the average neighborhood distance among marsh patches increase with rising sea level if accretion is kept constant. If these happen, the Wells marsh is unlikely to support many of the fishes in the future. The proximity and placement of habitat patches is important in sustaining populations of dependent organisms. In Wells, habitat interspersion is maintained with moderate sea level rise but decreases with higher sea level rise and coastal squeeze. When marsh patches are further apart and re-located away from usual migration paths, fishes that use the marsh may not find adequate food and shelter during the critical stages of their development (Irlandi and Crawford 1997; Parris 1989; Saintilan et al. 2007). Some of the fishes in Wells are marsh-dependent and diadromous;

hence, they are likely to be affected. Low interspersion of marsh patches may increase movement cost for endemic fauna moving between the interior marshes, creeks, and mudflat edges. Moderate sea level could maintain patch proximity, but in marshes that cannot accrete with sea level rise, the patches become more secluded as creeks and mudflats dominate the intertidal landscape. Consequently, several habitat types within a marsh system may be lost. In riverscapes, Kim and Lapointe (2011) suggested that spatial variation in habitat type explains the differences in run sizes in Atlantic salmon. Conserving optimum interspersion in marshes may ensure habitat viability and would qualify as another criterion in prioritizing restoration areas that would serve both transient and resident species.

The decreasing trend in edge density and fractal dimension could mean lesser quality refugia in the Wells marshes under rapid sea level rise and coastal squeeze. In turn, this might reduce the density of marsh-dependent fishes in Wells although I lack site- and species-specific evidence to support this. In other areas, Gosselink (1984) found total marsh edge lengths explained about 75 % of the variation in shrimp density in the Mississippi River Delta. Patches of habitats with more convoluted edges provide more food and refugia and, consequently, higher species richness as Corman and Roman (2011) found after comparing nekton population in marsh creeks against linear ditches.

Marshes naturally have convoluted edges such that their shape has a high fractal dimension. The average fractal dimension (D) of Wells marsh both at the landward and seaward edges decreased with increasing sea levels. Land development and slope preventing inland expansion reduce the fractal dimension of the landward edge of the Wells marsh. On the other hand, elevation controls the fractal dimension along the seaward edge. Low fractal dimension on both the landward and seaward edges of the marsh may reduce the habitat quality of the marsh by decreasing its potential for refugia for both terrestrial and aquatic organisms. In fishes, for example, marshes with simpler edges (i.e., low edge density and fractal dimension) could favor lesser species with more exploitative potentials (Doncaster 2001; Skov et al. 2011) such as striped bass and, therefore, might shift the patterns of habitat selection (Craig and Crowder 2002) and marsh utilization among the several categories of marsh users (Peterson and Turner 1994).

Fractal dimension decreases substantially when accretion is kept constant at the present rate while increasing sea levels. Although marshes that accrete at the same rate as sea level are likely to maintain their fractal dimension on their seaward edge, their inland edges become more increasingly susceptible to coastal squeeze by steep slopes and land use such that their average fractal dimension could also decrease. So in marshes that are not highly susceptible to sea level rise, it is still important to conserve their potential expansion areas and vegetated peripheries.

A calculation of fractal dimensions (which indicate the edge complexity) over time will reveal the impacts of anthropogenic perturbations as well as rising sea level. Because fractal dimensions are independent of scale, such analyses place more emphasis on habitat edge complexity than the size of the habitat (Kent and Wong 1982). Fractal shapes may be used as a template in wetland creation and in designing conservation areas that are optimally distributed in space with boundaries that follow the shape of a dominant natural landform. Furthermore, fractal dimension may serve as a surrogate metric for habitat quality of wetlands. In addition, measuring the changes in fractal dimension or evaluating species response to changes in edge complexity (O'Connell and Nyman 2010) may help locate marsh sections that are severely impacted by coastal squeeze and identify restoration areas that can maximize the potential for refugia. Further research is needed to test the link between changes in fractal dimension with change in habitat quality of restored coastal systems.

5.5.4 Potential loss of connectivity

Anthropogenic barriers increase coastal squeeze and the adverse effect of sea level rise. At Wells, I identified the location of physical barriers that squeezed the marsh and the intensity of restriction using the resistance index produced from modelling connectivity. In particular, the small-sized culvert had the most influence because it divided the estuary and restricted normal tidal flow, preventing fishes from accessing the greater part of the marsh. Removal of the barriers will substantially reduce the resistance index within the estuary. There are other ecological consequences of restricted tidal flow. For instance, in marshes disconnected from regular tidal flushing, fishes are found to have more parasites (Dibble and Meyerson 2012). Restoring normal tidal flow could ensure that the marshes are utilized in its full potential by all sorts of coastal organisms. This could mean removing different barriers to connectivity and increasing the permeability of the estuary (by lowering the resistance index) to enhance different forms of movement and interactions. In the model, I demonstrated that by removing one major barrier like a narrow culvert, the resistance index is substantially lowered. Using a similar approach of connectivity modelling, one can test the result of different management options to remove different barriers.

5.5.5 Study limitations

The lagoon at Wells is sheltered by a sand barrier that also will be affected by rising sea level and, in particular, high energy storms that are predicted to occur with warming climate (Meehl et al. 2007). Both will cause the landward migration of the barrier and eventual transgression over the marsh, destroying it before rising sea level does. These changes, obviously, could not be addressed in the model.

I kept the models algorithmically simple while attempting not to compromise repeatability and information content. As such, there were several limitations that might not completely capture the processes on an intertidal system under sea level rise. I adopted a linear trend in sea level rise which is considered to be less uncertain (Bittermann et al. 2013) though some reported that sea level rise is nonlinear (Foster and Rohling 2013; Gasson et al. 2012) and the results will not reflect a situation in which rate of sea level rise accelerates over time. I did not consider local subsidence though it is indirectly accounted for in the accretion rate. Also, I did not explicitly consider erosion though I designed the ACCSLSIM model to account for creek expansion as a function of accretion, elevation, and distance to the nearest creek.

5.5.6 Implications for conservation

I found that the impacts of sea level rise and coastal squeeze on marsh fragmentation and connectivity are nonlinear. In the case study at Wells, sea level rise does not directly translate to habitat loss or deterioration because at moderate sea level rise, the marshes could expand. At moderate sea level rise, there were no substantial changes in marsh configuration and spatial complexity. The relationship among the impacts of sea level rise and coastal squeeze indicates that there is a tipping point at which a maximum sea level that must be exceeded under a certain rate of accretion for a marsh shows some indications of degradation. In brief, the sea level rise must not exceed accretion to a point where severe accretion deficit can occur. At Wells, this threshold is 1 m or equivalent to a rate of sea level increase of 1 cm year^{-1} under a constant regional accretion rate of 0.36 cm year^{-1}. This response is predictable, because there is high tidal amplitude at Wells and the marsh is high in the tidal frame, so that it will not disappear until the rising sea level has submerged it. Currently, sea level is rising at 0.32 cm year^{-1} (Cazenave and Llovel 2009; Nicholls and Cazenave 2010) so the marshes are keeping pace, but recent projections that estimate future sea level to increase as high as 2 m are concerning. At this rate, it is likely that habitat loss will increase.

Coastal squeeze may modify not only the physical composition and spatial arrangement of marshes but its function as habitat. Severely squeezed marshes indicated by reduced patch size, edge density, and fractal dimension may deteriorate more rapidly and may affect spatial partitioning of food source and refugia between the different types of marsh users. For example, simultaneous expansion of creek and decrease in marsh area would increase predation risk on small marsh-resident fishes. If prey populations are extirpated, long-term fishery production might decrease. Aside from reduction in marsh area, coastal squeeze may reduce carrying capacity of marshes affecting several species across different development stages. Change in composition and configuration may redistribute food and refugia in patterns that are unfamiliar with marsh users putting endemic species at higher risk of extirpation. At present, landscape level analyses are not currently built into many conservation and monitoring programs for wetlands. Knowing what indicators to monitor would help resource managers identify the current and future threats to wetland persistence at an ecologically relevant and yet manageable scale. Overall, the results agree with Able et al. (2012) that multiple sub-habitats within the intertidal landscape or seascape should be recognized in any tidal wetland conservation and restoration with climate change.

5.6 Summary

Sea level rise and land use contribute to wetland degradation in the form of coastal squeeze. Coastal squeeze can complement and enhance marsh submergence and degradation particularly in marshes that do not accrete at the same rate as sea level rise. When the marshes are degraded, there are impacts to its functions that are difficult to quantify using conventional metrics. I addressed this challenge by adopting a landscape ecology approach which allowed me to identify two important impacts of coastal squeeze in addition to the loss of area, that is, reduction of habitat complexity and loss of connectivity. Reduced marsh complexity such as edge density, shape, and distribution among other intertidal sub-environments lower the potential of the marsh to provide food and refugia. In particular, reduction in marsh edge fractal dimension on both the landward and seaward edges could reduce the habitat quality of the marsh for both terrestrial and aquatic organisms, respectively. On the other hand, reduced connectivity prevents organisms from efficiently using food resources and refugia. In all, I suggest that without explicitly considering habitat characteristics, a species-centered approach to conservation may be deficient because of the uncertainties related to climate change and constraints imposed by species data. Under these uncertainties, conserving areas that are likely to hold the properties that support future occupants may augment other conservation strategies that adapt to the impacts of climate change.

Table 5-1. List, description and value range of selected landscape metrics

Landscape Metrics	Description	Range of Values	Reference
		Composition Index	
POMA	% Original Marsh Area	0<POMA<max	Computed manually
MPS	Mean Patch Size	A_{min}< MPS <MA_{total}	(McGarigal and Marks 1995)
PSSD	Patch Size Standard Deviation measures the absolute variation in patch sizes	0<PSSD<max	(McGarigal and Marks 1995)
Ecologically Scaled Landscape Index (ESLI)	Ratio between the patch size and the size of the habitat requirement of a species (K) averaged for all the patches	0 < Kavg< Kmax	(Vos et al. 2001), computed manually
		Configuration index	
Mean Patch Edge Density (MPE)	Average amount of edge per patch, where edge is the perimeter of patches divided number of patches of a particular class.	MPE > 0	(McGarigal and Marks 1995)
Mean Patch Fractal Dimension (MPFD)	Measure of shape complexity as deviation from simple Euclidian shapes. Mean fractal dimension approaches 1 for shapes with simple perimeters and approaches 2 when shapes are more complex.	1<AWMPFD<2	(McGarigal and Marks 1995)

Table 5-1 continuation…

	Connectivity index		
Mean Nearest Neighbour (MNN)	Measures the average shortest edge to edge distance between similar patches	0<MNN<MNNmax	(McGarigal and Marks 1995)
Interspersion & Juxtaposition Index (IJI)	Measures patch adjacency of classes as a function of edge length and the number of classes The value approaches 0 when the classes are clumped and 100 when classes are equally distributed and adjacent to each other meaning that each class shares a common border with all other classes.	0≤PC≤100	(McGarigal and Marks 1995); http://ec.europa.eu/agriculture/publi/landscape/ch1.htm
Resistance Index	The index predicts the pattern and the degree of resistance of movement on a landscape with varying obstacles. I calculated the index based on the principles of least-cost modelling and Circuit Theory.	Resistance Index > 0	(McRae et al. 2008) (Carroll et al. 2012)

Table 5-2. Change in areas of the intertidal subenvironments with four rates of sea level rise and accretion rates held constant or equal to the rate of sea level rise

	Sub-environment Area (ha)			% marsh beyond property line
	Creek	**Mudflat**	**Marsh**	
Present Marsh	143	87	560	1.3
Sea level rise rate (cm.yr-1)		**Constant Accretion**		
0.32	148	91	612	4.0
0.50	156	92	638	6.3
1.00	210	85	579	12.8
1.50	610	68	228	40.0
		Accretion Equals Sea Level		
0.32	149	91	612	4.1
0.50	153	93	639	6.0
1.00	167	98	648	9.1
1.50	182	100	639	10.3

Figure 5-1. The location of the study site in Wells, Maine, USA. Wells is located about 50 km southwest of Portland, Maine. The Wells Marsh is a complex of marsh habitats consisting of the Webhannet Lagoon, southern marsh and the Little River Estuary

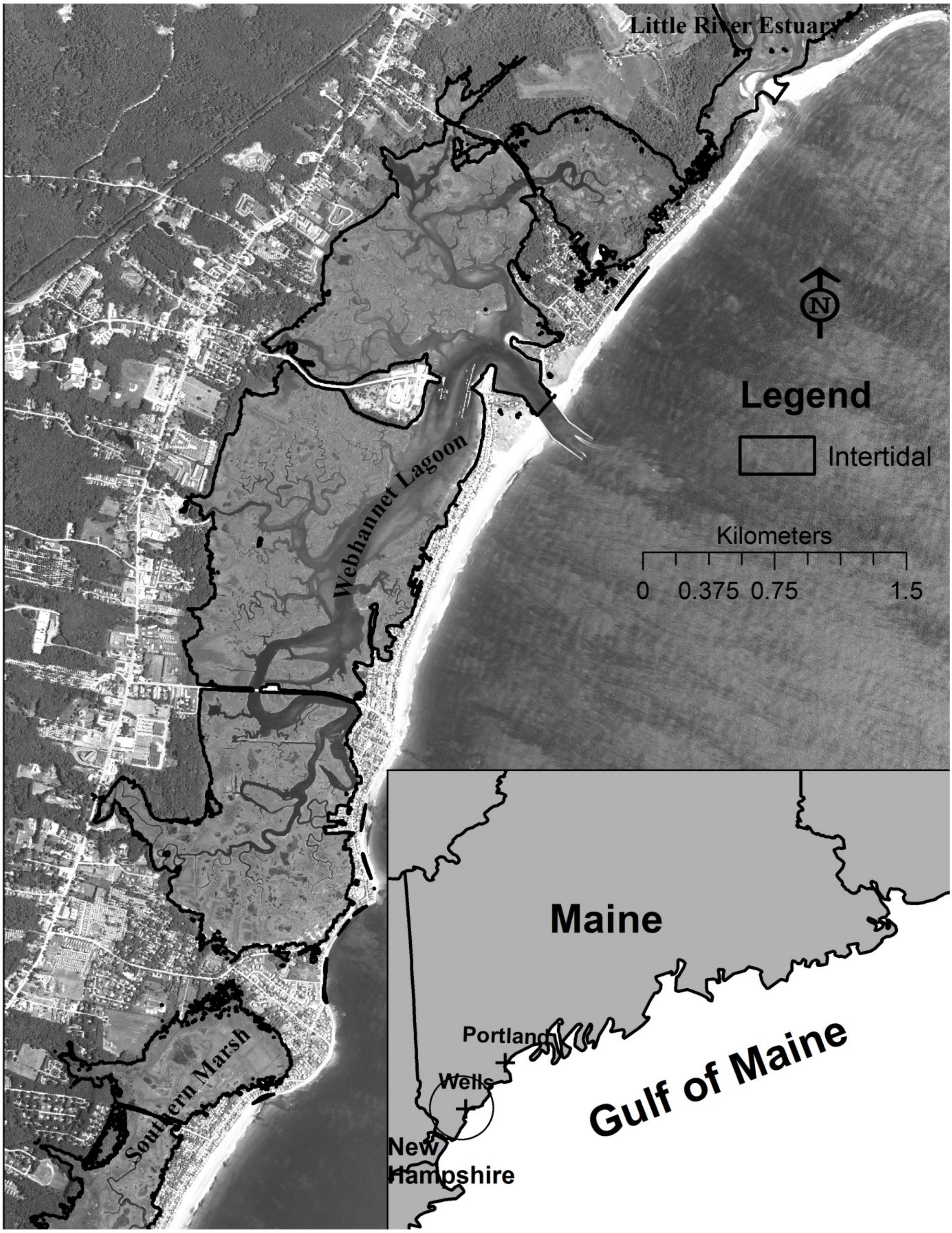


Figure 5-2. Change in the extent of the upland edges of the Wells marsh under different sea levels. The area inside the inset box is magnified (b–d) to show potential migration areas that cross or overlap with developed land use or beyond property lines. At 0.5 m sea level (b), the marsh edges expand over low development and vacant lots and over some portions of the road. As sea level rises further to 1 m (c), more developed areas are occupied including some areas with infrastructures. At 1.5 m sea level, more developed areas are taken over and the lower edge of e marsh connects with the upper edge of the southernmost tip of another marsh

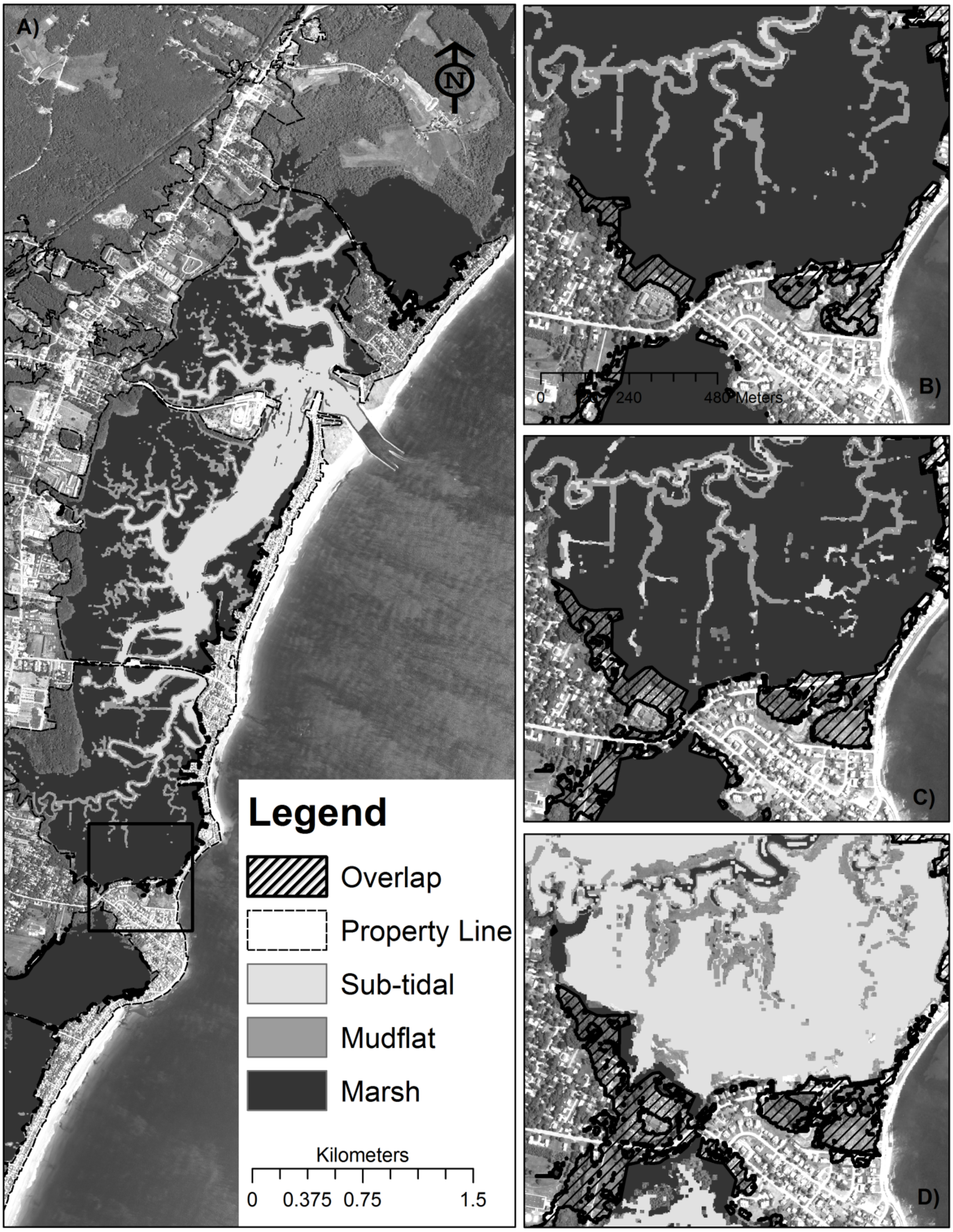


Figure 5-3. Change in area with sea level in marshes under constant accretion. (a) The current extent of the Wells marsh. At 0.5 m sea level (b), there is no substantial change compared to the current area. Marsh area began to decrease at 1 m sea level (c) as more areas are submerged (arrow). At 1.5 m sea level (d), more marsh areas are lost as the seaward edges are submerged, while the upland edges are not able to expand anymore because of steep slope and development

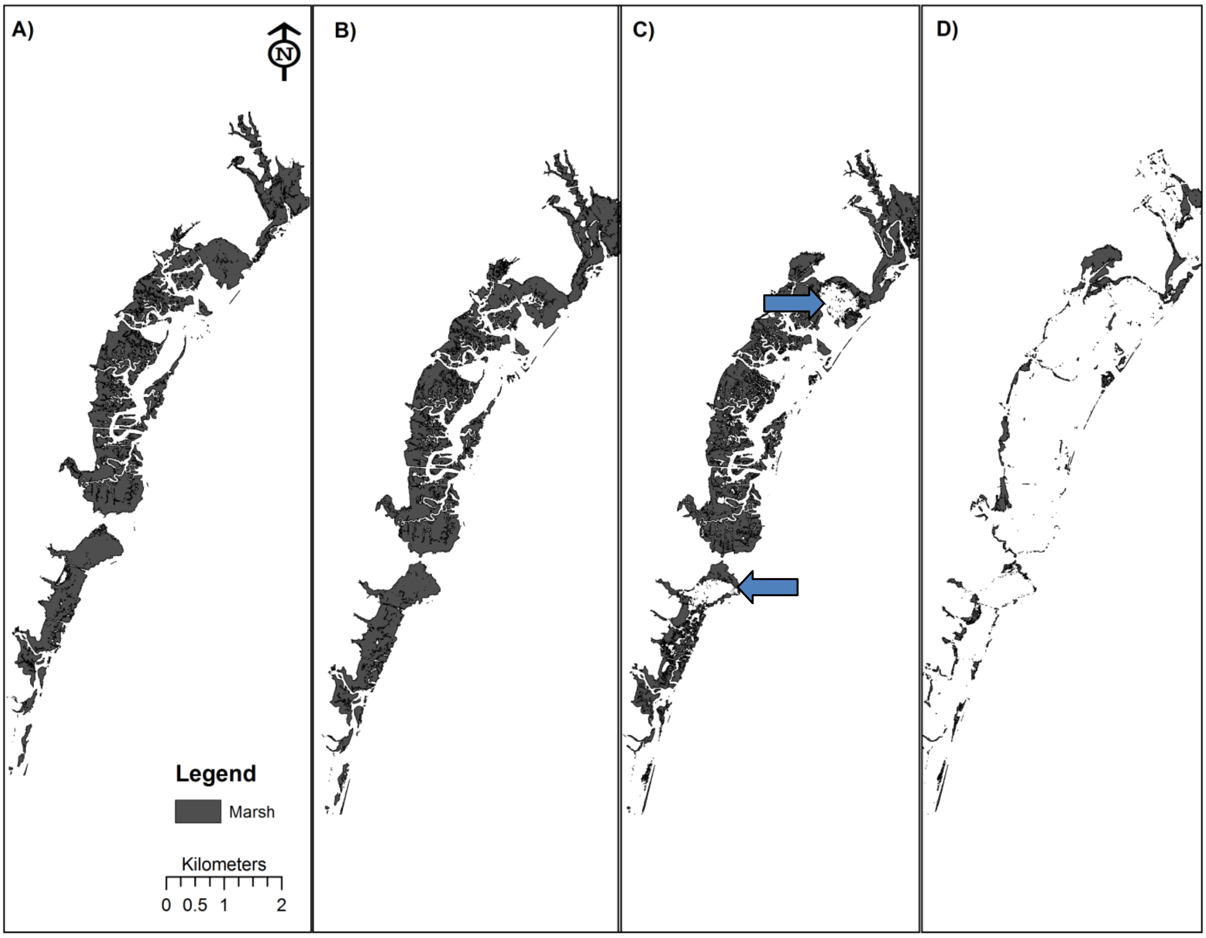


Figure 5-4. Comparison of the landscape metrics in each sea level computed from the outputs of constant accretion and accretion equals sea level rise model. a Percent original marsh, b mean patch size, c mean edge density, d patch size standard deviation, mean nearest neighbour, and f interspersion and juxtaposition index

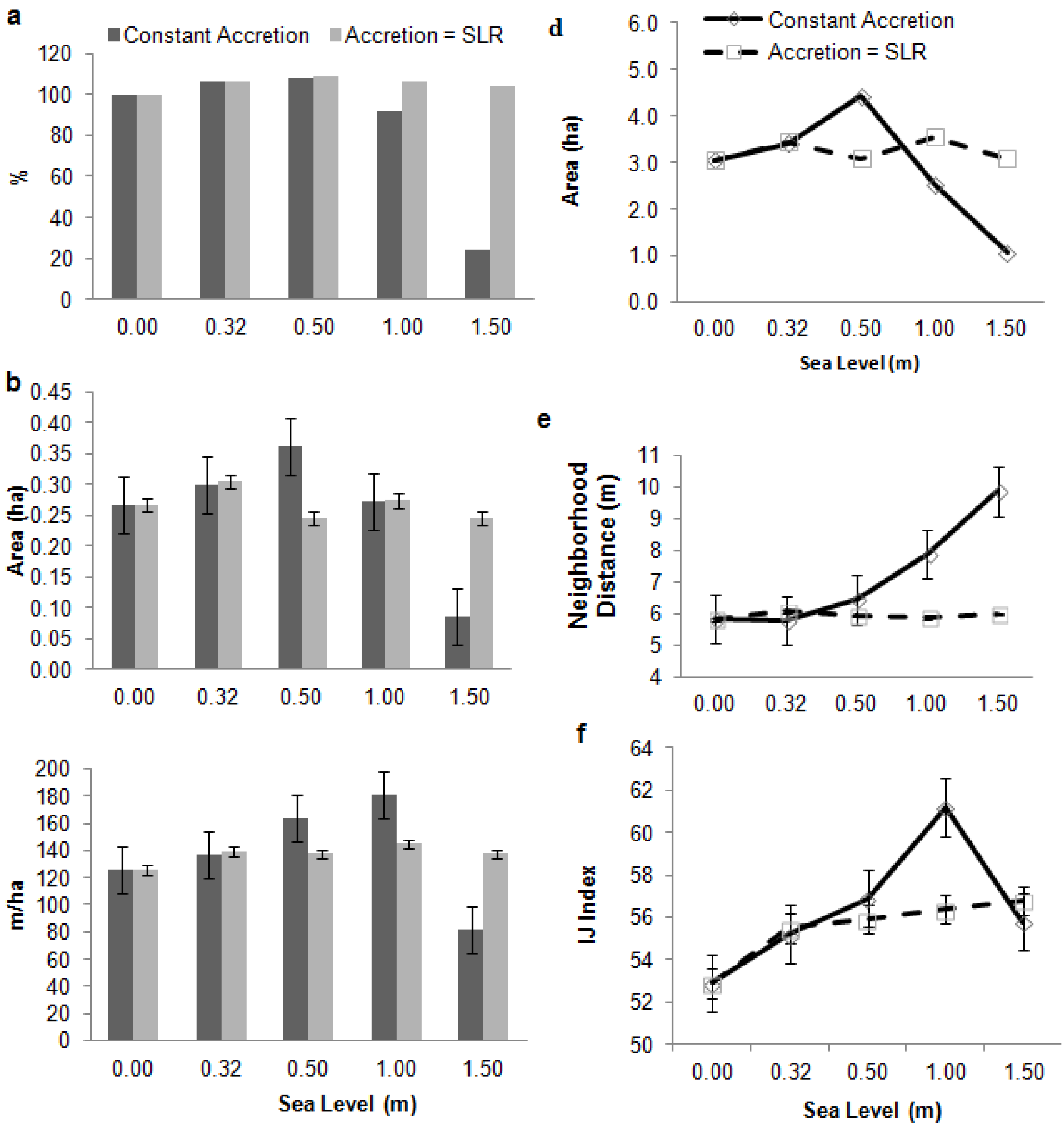


Figure 5-5. Ecologically scaled landscape index (ESLI) computed for mummichog, *Fundulus heteroclitus* (a), and striped bass, *Morone saxatilis* (b), two common fishes found in Wells marshes

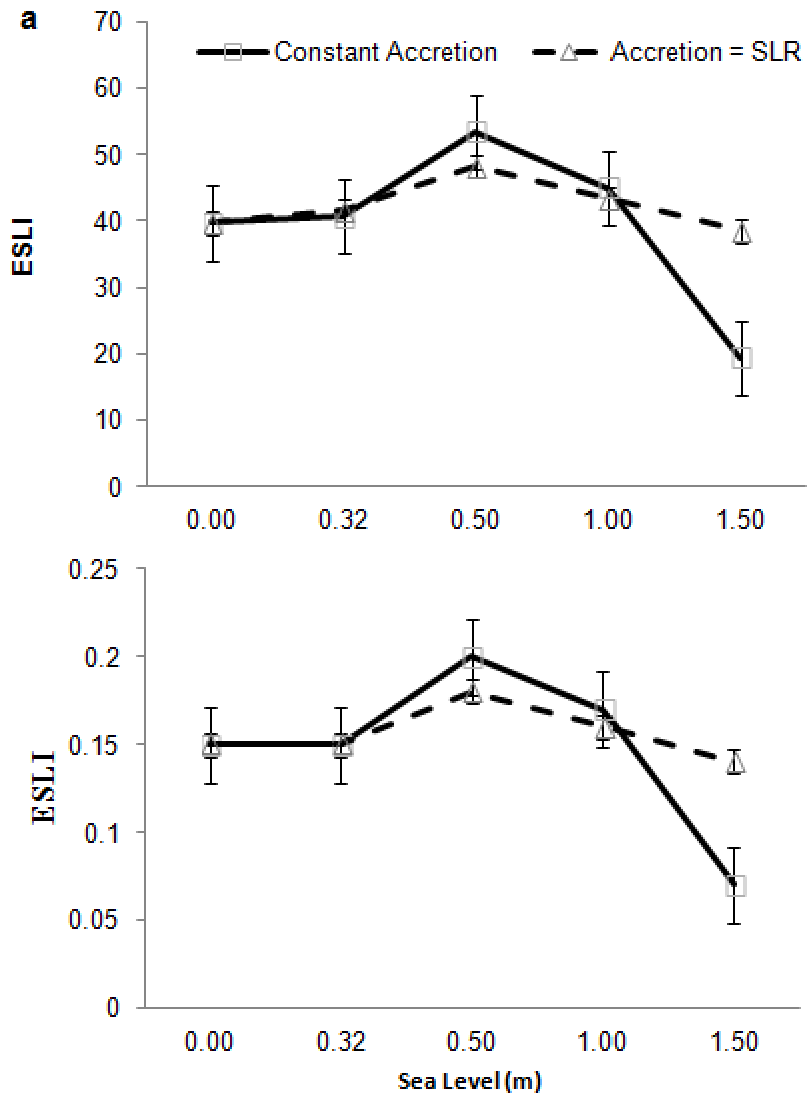


Figure 5-6. Changes in the shape of marsh and subtidal environments under constant accretion: (a) The baseline shape (no rise), in comparison to 1.5 m (b), (c) The current shape of subtidal environments or seaward edge of the marsh (including mudflats and intertidal creek edges). At 1.5 m sea level, these expand and loss their shape complexity (d) as the edges reaches steep slopes and other upland barriers

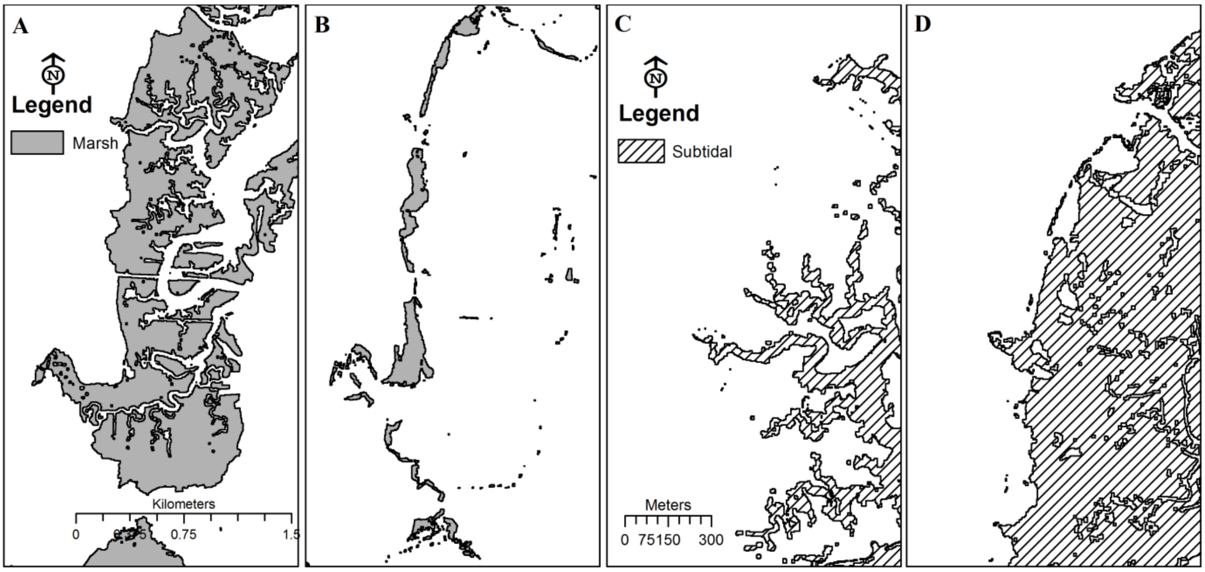


Figure 5-7. Changes in shape as indicated by fractal dimension of landward and seaward edges of the marsh under constant accretion and accretion equals to sea level models. Both accretion models show a decreasing trend but the fractal dimension under constant accretion (a) decreases more. Low fractal dimension at high sea level (i.e., 1–1.5 m) indicates that the upland edges are almost linear. At the seaward edge (b), the fractal dimension did not change if the accretion is equals to sea level. In the constant accretion, fractal dimension drops drastically from 1.53 to 1.44 at 1 to 1.5 m, respectively

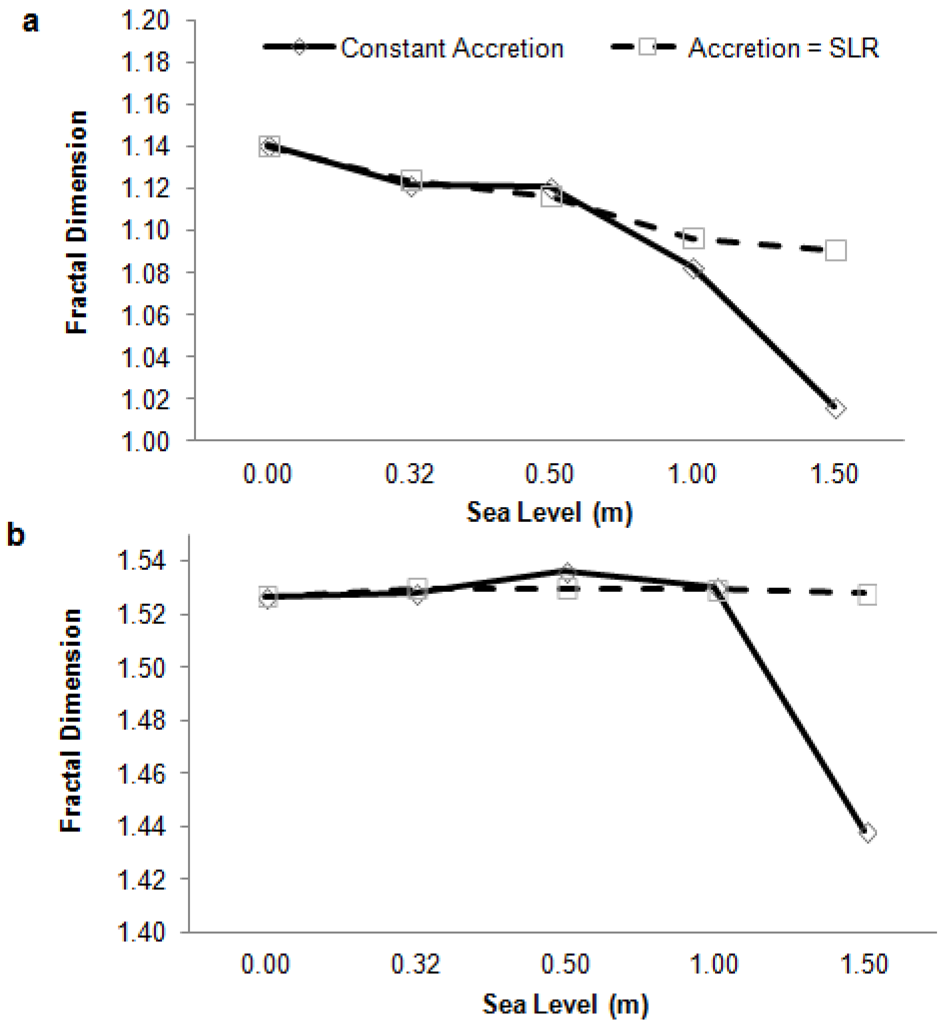
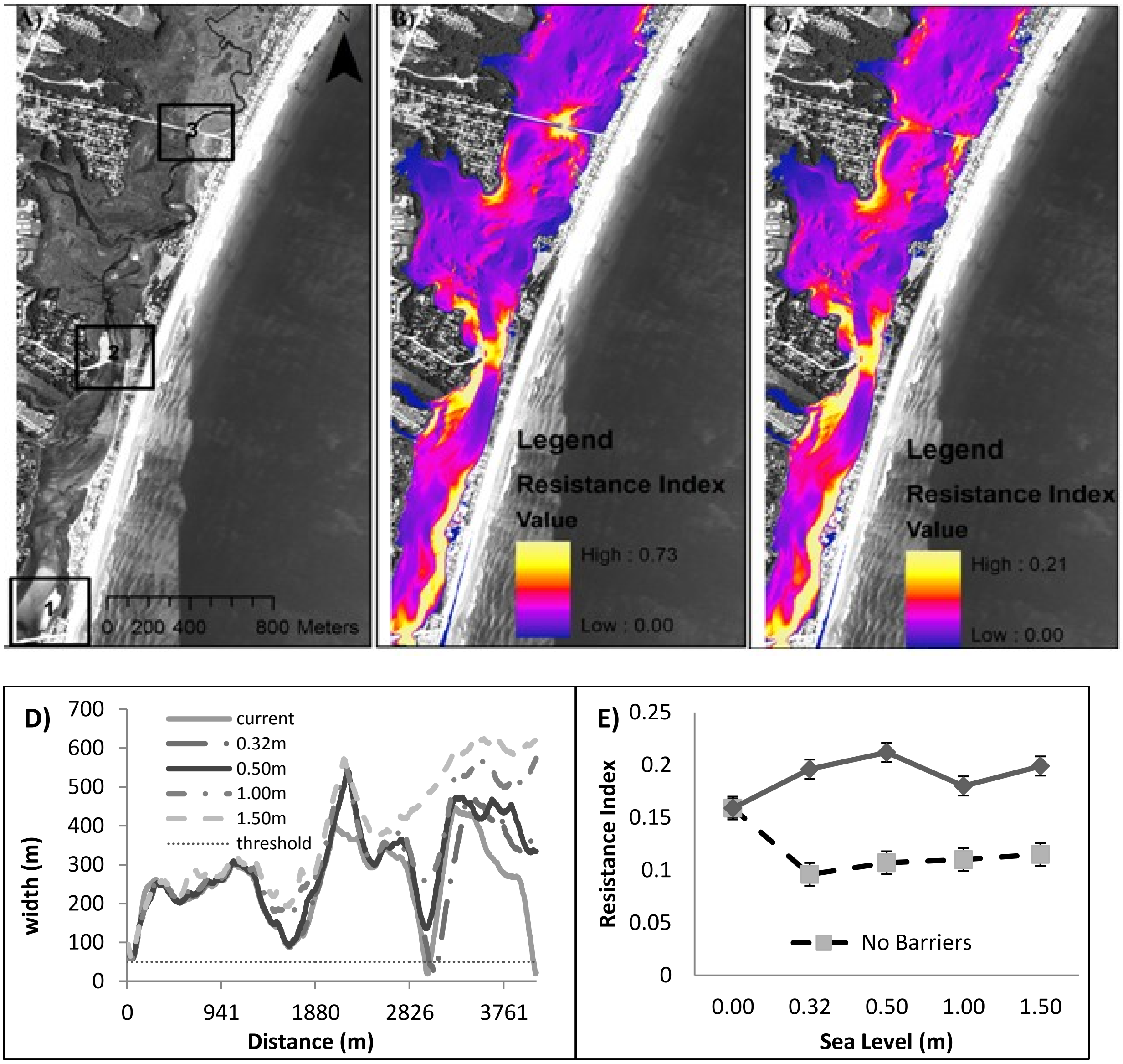


Figure 5-8. Location of existing barriers and modeled resistance index relative to those barriers. (a) Three major connectivity barriers; narrow estuary entrance (box 1), developed parking space (box 2), and small culvert underneath a road (box 3). (b) The modelled movement resistance "hotspots." When the barriers are removed (c), some of the hotspots disappear and the resistance index decreases. (d) The width of the estuary where the narrow areas (dips) correspond to the width with barriers. In comparison, the average resistance index is generally higher in areas with the barriers intact (e)



5.7 Appendix

Figure 5-9. Implementation of Accretion and Sea Level Simulator (ACCSLSIM) in ArcGIS ModelBuilder

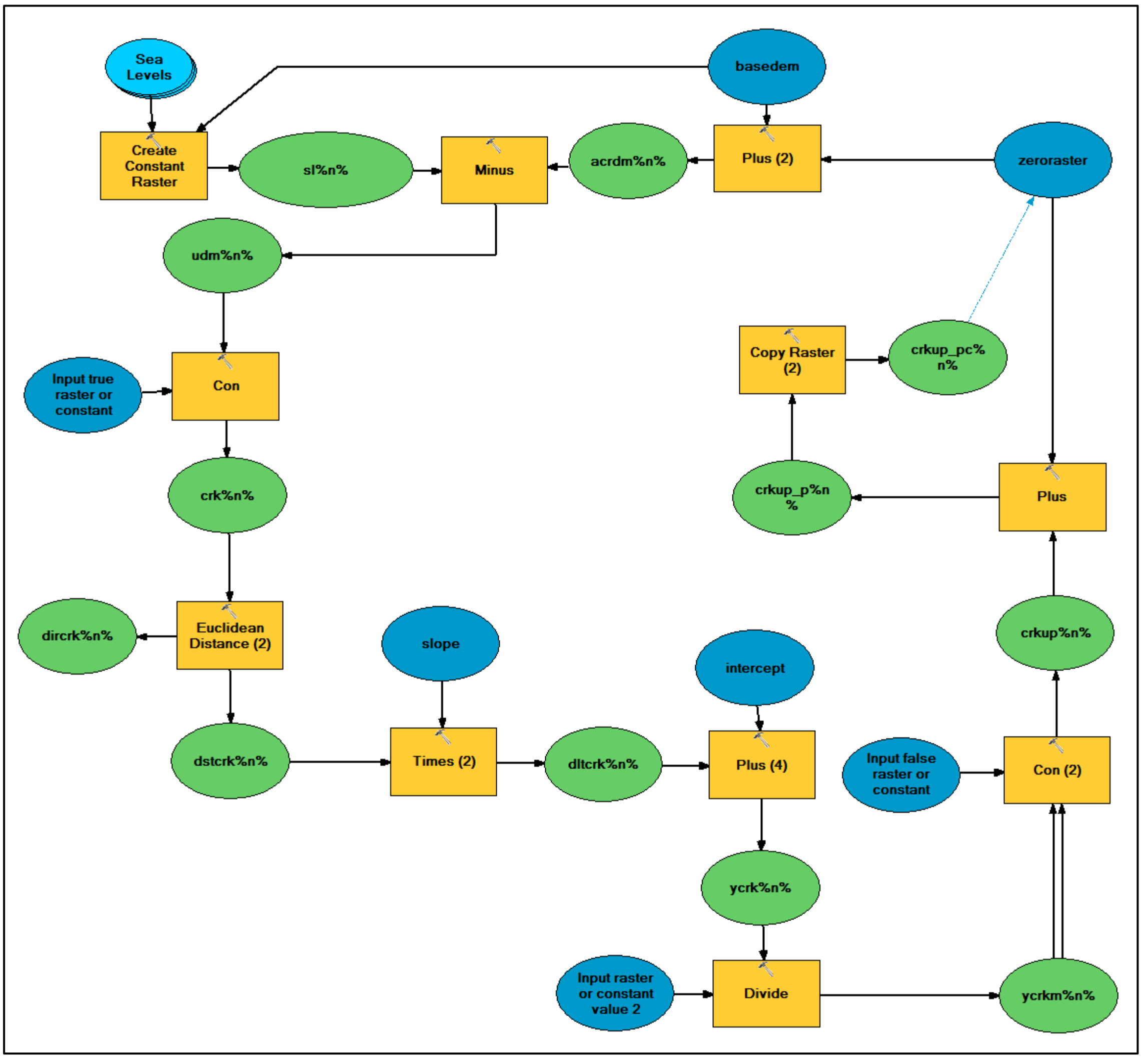


Figure 5-10. Implementation of Tidal Range Extent Simulator (TIDEXSIM) in ArcGIS ModelBuilder

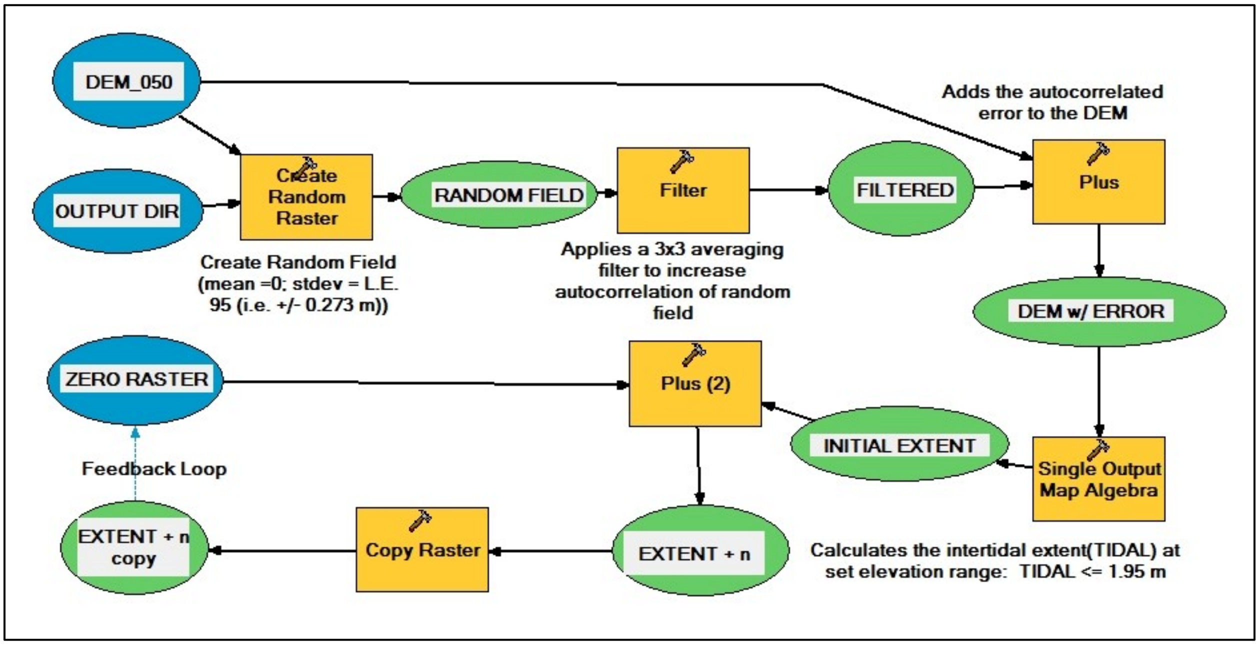
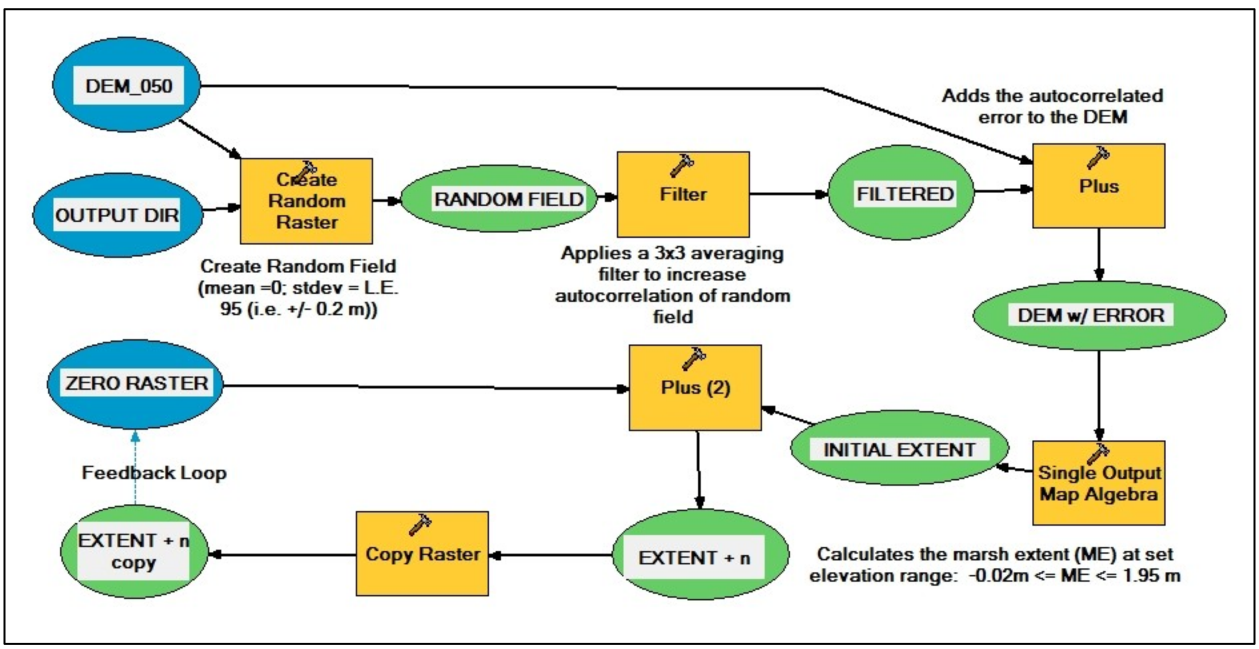


Figure 5-11. Implementation of Marsh Extent Simulator (MEXSIM) in ArcGIS ModelBuilder



Chapter 6 Sensitivity of salt marsh carbon to sea level rise

6.1 Preface

6.1.1 Manuscript details

This manuscript is co-authored by Gail Chmura. It is under preparation for submission to Estuaries and Coasts.

6.1.2 Context and link to the previous chapter

The spatial structures of tidal wetlands are not only important as habitats but also are important in mitigating or enhancing climate change. In particular, the soil of the marsh holds a large amount of carbon. Carbon storage in tidal wetlands is intricately linked to belowground production and accretion. As the wetlands spread laterally and grow vertically, more carbon is stored. This active carbon sink can increase or decrease with sea level rise. With accelerated rates of sea level rise more coastal carbon sink can be lost. Tidal wetlands lost to sea level rise could release the carbon stored in soil back to the atmosphere. Because of this, carbon sequestration and re-emission is receiving considerable attention in climate research and emerging climate mitigation policies.

This chapter investigates the impacts of different rates of sea level rise and trends on carbon sequestration, another important ecosystem function of tidal wetlands that is sensitive to sea level rise. Using the inundation model developed in chapter 5, I determine how carbon sequestration as function of area and elevation varies with the projected rates and trends of sea level rise. Whereas chapter 5 focused on the two-dimensional properties of the marsh surface, this chapter adds a third dimension, i.e., elevation of the actively accreting marsh. Accretion rate is assumed to vary over the surface of the marsh depending on its distance to a nearest creek. The outputs from the inundation model were used in a belowground production submodel to estimate the size of the active carbon sink. Both linear and non-linear sea level rise projected over the century were simulated at a yearly interval, hence, the model produces a yearly estimate of sequestered carbon.

6.2 Introduction

Coastal wetlands have demonstrated the capacity to adjust to historical rises in sea level. Excess in the rates of sea level rise are assumed to have a negative impact. Globally, the historical increase in sea level has followed a linear trend for the last two millennia. The trend is projected to accelerate non-linearly in the next 100 years if greenhouse gas emissions, global temperature, and melting of the polar ice caps continue to increase (Church et al. 2013b). In its 5th Assessment Report, the Intergovernmental Panel on Climate Change (IPCC) projects that a maximum of ~1 m rise in global mean sea level at the end of this century (Alexander et al. 2013) is highly likely. Parris et al. (2012) suggest that a maximum 2 m rise in sea level should be considered for planning along the US coast.

Besides the overall increase, sea level rise could follow a linear or exponential trend. As such, it could have distinctly different impacts on coastal wetlands. Differences in the trend and magnitude of sea level rise can pose a policy dilemma in coastal management, e.g.,(Sallenger et al. 2012). Yet, there has been little research to demonstrate how impacts differ between linear versus exponential trends over time. Understanding how tidal wetlands respond to linear and exponential rates of sea level rise could drive more reasonable future conservation priorities.

The historical response of tidal wetlands has been to expand inland and even seaward as sea level rose [e.g., Redfield (1965)]. Lower rates of sea level rise worked in favor of tidal wetland area and presumably the ecosystem services they provide. Higher rates of sea level rises and modification of the coastal landscape are likely to limit wetland expansion with negative consequences. First, urban infrastructure constructed near wetlands (Bulleri and Chapman 2010; Dausse et al. 2008; Schleupner 2008) creates steep shorelines and impervious surfaces limiting inland expansion areas and putting tidal wetlands under "coastal squeeze" (Doody 2004). Second, if the wetlands are eroded they will no longer protect communities from storms or provide sufficient habitat for fish. Finally, with coastal squeeze and accelerated rates of sea level rise, many of the indirect ecosystem services from tidal wetlands could be lost.

Decreased primary production would decrease the value of tidal wetlands as carbon sink. The high amount of carbon stored in tidal wetlands is driven by belowground production (Armentano and Menges 1986; Duarte et al. 2013; Mcleod et al. 2011). With rising sea level marshes and mangroves build in elevation through accumulation of roots, rhizomes and mineral sediments. However, the projected sea level rise rate for next century threatens marshes with prolonged inundation and erosion. If the rate of peat formation is lower than the rate of sea level rise, the wetland could drown in place converting the vegetated surface to tidal flat or open water and the seaward edge will be prone to erosion. Under these conditions, the carbon stored in the soil could be oxidized and potentially re-emitted back to the atmosphere. If tidal wetlands cannot migrate inland because of coastal squeeze, the area lost at the seaward can never be replaced resulting in a decline in rates of carbon sequestration.

Conserving tidal marshes as carbon sinks necessitates understanding the likely consequences of different sea level rise trends and the barriers to marsh sustainability. Under what sea level rise trend and landscape circumstances could the marsh tip from being an effective carbon sink to a potential carbon source? It is likely that wetland response would vary depending on the characteristics of the landscape. The need to track the changes of a marsh with sea level rise requires spatially and temporally explicit models that can be easily parameterized with basic information.

Results of recent modelling studies are conflicting. For instance, Kirwan and Mudd (2012) and Kirwan et al. (2010) project that carbon sequestration in salt marshes will increase in the first half of the century and then slightly decrease in the second half when the rate of sea level rise accelerates and drowns marshes. They report that this shift occurs when sea level rise exceeds 1 m (or a rate 10 mm yr^{-1}) and results in less productive marshes. The Marsh Equilibrium Model (MEM) output reported by Morris et al. (2012) makes a more dire prediction that carbon sequestration will decrease at less than a 1 m rise in sea level. These studies did not account for upland expansion or the variability of future sea level rise trends and the topographically heterogeneous marsh surface. Models that do not account for seaward loss or inland expansion are likely to underestimate wetland response, and hence, carbon sequestration.

Spatial and temporal models of carbon sequestration are needed to project future changes of carbon stocks with sea level rise. However, tidal marsh evolution models designed to estimate carbon stocks are mostly point based and not spatially explicit. In these models, it is assumed that the marsh surface is spatially uniform while it is spatially heterogeneous. More complex geomorphic [e.g., SLAMM (Craft et al. 2008)] and numerical models [e.g., DIVA (Mcleod et al. 2010)] may or may not be spatially explicit but they do not directly address carbon storage. As carbon sequestration varies spatially and temporally, a model that estimates and predicts wetland-wide changes in carbon stocks should account for variations in topography and expansion.

Here I describe a spatially explicit model that addresses the three limitations that Fagherazzi et al. (2012) have identified in current point-based and landscape models of marsh vulnerability: 1) predicting whether upland expansion could compensate for erosion of the salt marsh edge and vertical submergence of the platform, 2) quantifying the relative importance of an expanding channel network in the delivery of sediment to the marsh interior, and, 3) addressing data deficiency and variability such as different assumptions and rates of sea level rise. The model improves upon previous models [e.g., Torio and Chmura (2015)] developed for the marsh system in Wells, Maine, USA . In addition, I extend the model capability with a submodel to simulate optimum production as a function of variability in elevation, a recent development in the carbon estimation (Kirwan and Guntenspergen 2015; Kirwan and Guntenspergen 2012). This results in a four dimensional (4D) spatially and temporally explicit model that accounts for sea level rise, upland migration, creek expansion, surface accretion, seaward submergence, and optimum production in calculating carbon stock. I run the model at a yearly interval at different rates of sea level rise based upon assumed linear and exponential trends.

6.3 Methods

6.3.1 Study site and data sets

The model is based on the Wells system (Figure 6-1), a complex of marshes that is part of the U.S. National Estuarine Research Reserve System (NERRS) in Maine and is described by Torio and Chmura (2013). Wells is a coastal town with an average density of 56 houses per km^2. Housing density is high with most houses located along the coast, and in some areas, private lands directly border the upland edge of the marshes. Three minor causeways dissect the marsh. To the seaward side, the marshes are sheltered by a sand barrier separating Webhannet Lagoon from the Gulf of Maine. Wells has a meso-tidal range with the highest astronomical tidal range of 4.22 m.

6.3.2 Sea level rise response and change in elevation

Nine future sea levels with endpoints (i.e., the sea level that will occur in the future) in the year 2100 were utilized (Table 6-1). These include; 1) the historical linear trend observed in Wells with an endpoint of 0.5 m; 2) an exponential increase of up to 0.5 m (the IPCC RCP2.6 scenario, see Table 1 for references to sources); 3) a linear increase up to 1.0 m; 4) an exponential increase (the IPCC RCP8.5 scenario) of up to 1.0 m; 5) a linear increase up to 1.5 m; 6) an exponential increase of up to 1.5 m (the mid-range sea level rise projected for the US); 7) a linear increase up to 2 m; 8) an exponential increase of up to 2 m (recommended for coastal planning in the US); and 9) an exponential increase of up to 1.3 m (the relative sea level rise projected for Portland, Maine). Except for the 2.0 m sea level rise endpoint, most of the projected sea level rise trends do not have corresponding equations that can used to estimate yearly sea levels, requiring us to derive them. For the 0.5 m linear sea level rise, historical data published by Gehrels et al. (2002) were used. When no published data existed, the equations were derived by simply regressing the "initial" sea level in 2010 [i.e., 0.19 m (Alexander et al. 2013)] by the individual endpoints.

For the 0.5 and 1.0 m exponential sea level endpoints, the equations were derived from respective projected sea levels under IPCC RCP2.6 and 8.5 scenarios. The equation for the 1.5 m exponential sea level was derived from Hoffman's (1984) mid-range moderate sea levels rise projection and the 2.0 m exponential sea level rise model was based upon Parris et al. (2012) data. For the 1.3 m relative sea level projected for Portland, the equation was derived from data published by Tebaldi et al. (2012). Derived future trends in sea level rise are depicted in Figure 6-2.

6.3.3 Spatial modelling

The marsh and its surroundings have variable topography and land-use, ranging from flat developed lands and farms to the south and moderately sloping forests and grasslands on the north. The marsh surface elevation ranges from -0.02 to 1.95 m relative to the NAVD88 datum. In 2004 the topography of marsh and its periphery were surveyed with LiDAR to generate a high resolution (3 m) and relatively accurate (20 cm RMSE at LE95) digital elevation model (DEM). I used the DEM as input to the model.

A spatial-temporal GIS inundation model (also see Appendix Figure 6-7) was built to predict change in elevation of the marsh surface relative to the change in sea levels and accretion. Under each assumption of sea level rise trends, the yearly net elevation is computed as the difference between the elevation with accretion and sea level. The model was initialized to 0 at year 2004; the year when the LiDAR survey was conducted. In each year, the surface area of the marsh and its corresponding elevations were calculated and extracted using an extent 'mask' or the extent of the Wells marsh at a given elevation range (i.e., -0.02 m to 1.95 m).

In the model the accretion rate is calculated based on the distance of the marsh surface relative to the edge of the nearest lagoon or channel within the marsh. It is assumed that accretion rate decreases with distance to the nearest channel. This relationship is based upon an accretion model reported by Chmura and Hung (2004) for marshes on the coast of eastern Canada.

6.3.4 Carbon sequestration and potential storage

From the spatial model I could then calculate rates of carbon sequestration. Annual carbon sequestration per unit area was estimated by multiplying the marsh surface area by the annual accumulation rate of 167 g CO_2 m^{-2} y^{-1}; the average sequestration rates of the Gulf of Maine and Bay of Fundy marshes (Chmura et al. 2003). As most carbon sequestered by the marsh is stored belowground, I computed the optimum belowground production based on elevation by applying an equation developed by Kirwan and Guntenspergen (2012) to the modeled marsh elevations. The model assumes that root production is optimum at elevations of 10 to 50 cm. At this elevation range, root productivity follows a humped-shaped curve which fits into a quadratic equation: y (g) = -6.0*elevation^2 + 375.7*elevation -3315.8, $r^2 = 0.71$ $p = 0.004$). The estimated belowground production was converted to carbon mass by multiplying a factor of 0.38, the average of C content of *S. patens* roots and rhizomes reported by Gallagher and Plumley (1979) and Roman and Daiber (1984).

6.4 Results and Discussion

6.4.1 Carbon sequestration and marsh area

If I consider C accumulation rates as constant over the marsh area, as is done in point models (Morris et al. 2012) then marsh area alone would drive estimates of future carbon storage potential. When the MEM model was parameterized with variables used in the model the MEM model resulted in decreasing carbon storage even at a very low sea level rise. As such the carbon estimated by MEM is lower. In MEM, the size of current and future marsh area is not accounted for.

With the present area of the Wells marsh (6.79 km^2), the vegetation is fixing ~133 Mg CO_2 annually belowground. Output from all runs results in a non-linear response in marsh area, with area increasing up to the 1.0 m-endpoint, then decreasing (Figure 6-3A). With exponential rates of sea level rise, the loss of area is generally greater. At the 0.5 m sea level endpoint (the historical trend at Wells) the marsh area increases by 10% in both linear and exponential models and at 1 m it increases to 16% and 13%, respectively. Using the projected relative sea level rise for Maine (i.e., exponential 1.3 m), the marsh area increases by 7%. Under the 1.5 m-endpoint linear scenario, there is a gain of 6% while under the exponential rate, the marsh losses ~9% of its original area. At the maximum sea level rise of 2 m, the marsh loses more than 50% of its original area whether sea level rise follows a linear or exponential trend (Figure 6-3B). Considering these results, I expect that the Wells marsh would survive if sea level rise continues as the observed historical trend (linear or exponential) or if the expected sea level of 1.3 m occurs. The 1.3 m rise, however, would pose an increasing threat to the marshes when anthropogenic coastal squeeze is considered (Torio and Chmura 2013).

Model runs at sea level rise rates associated with endpoints at 0.5 m and 1.0 m (whether linear or exponential) and 1.3 m (exponential) produce a continuous gain in marsh area from the initial year to the end of the simulation in 2100 (Figure 6-4A to E). In the linear model, marsh area decreased after 2087 with 9% of the marsh area lost (Figure 6-4F). Runs with exponential 1.5 m-endpoint sea level rise produce abrupt declines in marsh area in the middle and northern sections of the marsh (Figure 6-4G). These areas are bordered by steep slope. With the 2.0 m sea level rise endpoint, both the linear and exponential trend result in an area loss of ~50% (Figure 6-4H). In the scenarios that resulted in reduced marsh area, losses occur at different time periods. With the 1.3 m-endpoint (and exponential rise in sea level) the marsh area begins to decrease in the year 2092. With the endpoint at 1.5 m, marsh area declines in 2073 with the linear trend while this happens in 2084 with the exponential trend. With the 2 m-endpoint sea level, marsh area declines in 2052 in the linear rate and 2067 in the exponential rate.

The linear rate of sea level rise enhances upland expansion by allowing flat areas to be flooded earlier as the sea level is higher than in non-linear scenarios throughout the run. However, when coastal squeeze becomes significant, the marsh can no longer expand at the upland edge and the overall area starts to decline at the seaward edge. Marsh area loss in the linear models area earlier but the rate of decrease is slower. Conversely, in the exponential models, marsh area loss occurs later but at a more rapid rate.

Under optimistic sea level rise projections such as the IPCC RCP 2.6 and 8.5, the expanding marsh would sequester about 1200 Mg CO_2 at the end of the century. Under the 1.3 m-endpoint exponential rate projected for Maine, the 7% additional area could fix approximately the same amount. At 1.5 m sea level, the amount of sequestered carbon dioxide is almost equal to the baseline. At 2.0 m sea level, the Wells marsh could only sequester half (500 Mg) of its original potential whether sea level rise trend is linear or exponential.

6.4.2 Carbon sequestration, elevation and belowground production

Belowground carbon has a variable response to sea level rise (Figure 6-5A). At sea level rise rates of $\leq$ 1 m belowground C increases with sea level rise. With the 0.5 m-endpoint sea level, both the linear and exponential rates have the same response: increasing carbon sequestration with similar magnitude. At the end of the simulation, the marsh sequesters about 20 Mg C with both rates. With the 1.0 m-endpoint sea level, 18% more C is sequestered with the linear rate than with the exponential rate or around 215 Mg C and 40 Mg C, respectively. With the 1.3 m-endpoint exponential rate, production starts increasing later in the century around 2080 and a peak of 857 Mg C is reached in 2099. With sea level endpoints >1.3 m C sequestration has a parabolic relationship with sea level rise, that is, marsh belowground production rapidly increases then rapidly decreases after reaching an inflection point. This non-linear trend is consistent whether the rate of sea level rise is linear or exponential. At the 1.5 m sea level endpoint, production with the linear rate increases more rapidly than with the exponential rate, yet both reach peak carbon sequestration in ~2094 with similar amounts of C, ~901 Mg C and 871 Mg C, respectively. Similar to the response with the 1.5 m sea level rise endpoint, carbon sequestration with the 2.0 m-endpoint is more rapid with the linear rate than the exponential rate. With the linear rate, the peak sequestration occurs in 2066 at 894 Mg C, while with the exponential rate produces a similar peak of 871 Mg C, but 12 years later in 2079. Both the 1.5 m and 2 m endpoints resulted in a substantial loss of the active C sink by the end of the century (Figure 6-5B). These results imply that the Wells marsh will remain at optimum elevation if the threshold sea level of 1.3 m is not exceeded. In fact, optimum root production is highest at the projected accelerated sea level rise of 1.3 m. If sea level rise exceeds this threshold, root production could decrease, which could lower the stability of the marsh. If sea level rise follows the historical trend and does not exceed 1.3 m by 2100, the Wells marsh will be a net carbon sink for the next 100 years. This holds whether the rate of sea level rise is linear or exponential. However, if sea level rise exceeds 1.5 m, more than half of the Wells marsh will be submerged and lost as an active carbon sink. The fate of the carbon in the submerged marsh is uncertain. If the submerged peat is oxidized and returned to the atmosphere as CO_2 then the Wells marsh could shift from a from carbon sink to a carbon source within 100 yr.

These results support those of Kirwan and Guntenspergen (2012) who suggest that acceleration in the rate of sea level rise will lead to enhanced root growth, organic accretion and carbon sequestration. However, they did not consider the spatial and temporal variation in these variables that may arise from the spatially heterogeneous nature of an actual marsh. The model used for Wells advances earlier models by applying optimum production rates (i.e., optimum elevation at which highest production occurs) to account for the change in elevation of the entire marsh surface subjected to different trends and rates of sea level rise and accretion. The results are more optimistic than previous point based modelling studies. For example, Kirwan and Mudd (2012) report organic matter accumulation accelerating at the first half and then decelerating at the end of the century under the IPCC B1 Scenario (equivalent to the 1.0 m linear and non-linear rate). The results indicate no deceleration, but increasing belowground productivity, hence carbon sequestration, at these rates. Similarly, they report marsh drowning in year 2080 and 2085 under the A1F1 and A2 scenarios, respectively. At similar rates (1.5 exponential and linear, 1.3 exponential in the model) my results show that marsh drowning starts at 2094 and 2099, respectively. The differences in the results imply that marsh loss tends to be exaggerated in models that ignore topographic heterogeneity and creek expansion.

A novelty of my work lies in the use of spatially explicit model to map the variability of belowground carbon in a salt marsh landscape at certain time period and sea level rise trend (Figure 6-5). Compared to other models I believe that opportunity for variable C accumulation rates over the marsh surface makes the response of mine more detailed and sophisticated. By mapping the variability of carbon storage I show that the impacts of sea level are not uniform across a marsh.

6.4.3 Change in active carbon sink from losses at the seaward edge

Most of the losses in the active carbon sink in both area and optimum production are due to the losses of the seaward edge. Under the exponential sea level rise trend of >1.3 m, marsh area increases initially due to inland and seaward expansion and as inland expansion stops, the seaward edge is submerged resulting in an overall decrease in marsh area. This happens when the rate of submergence overrides the rate of accretion and inland expansion is blocked or under coastal squeeze. Similarly, belowground carbon sequestration initially increases then decreases but it does this at a later period. The initial increase in area and production would allow the marsh to store more carbon in a shorter period. With accelerated rate of sea level rise the marsh peat at the seaward edge that stores the carbon could be lost in an equally short period. As such coastal squeeze can turn the positive feedback between sea level rise and productivity (Kirwan and Guntenspergen 2012) to a negative one. Such a trend is likely to occur in S. *patens* -dominated marshes with belowground production and carbon sequestration becoming sub-optimal under accelerated sea level rise and limited supply of sediments as in the case of the Wells marsh. Since *S. patens* occupies a higher optimum production elevation, the sections at the seaward edge dominated by the species will become less resilient to frequent flooding resulting in a less stable vegetation zone. Given this, it is possible that the zone could be completely washed away or other species with lower optimum production elevation like S. *alterniflora* may take over.

6.5 Conclusions and limitations

In the spatially explicit model the decrease in marsh area occurs earlier than the decrease in belowground production. For example, with the 1.3 m-endpoint, the marsh area begins to decrease in 2092 while the belowground production decreases slightly after 2100. Similarly, with the 1.5 m-endpoint, the marsh area starts to decline by 2080 while a decrease in production lags, occurring around 2094. With the 2 m-endpoint marsh area decreases around 2050 while belowground production decreases around 2060. The lag time between the decrease in area and decrease in belowground production suggests that estimating the impacts of sea level rise on carbon storage based on area alone could be misleading.

Under a 0.5 m sea level endpoint, whether the rate is linear or non-linear, the Wells marsh would survive and even increase its potential to sequester carbon. However, it will be at increasing risk of coastal squeeze under higher rates of sea level rise. At the maximum rate projected for Maine (i.e., 1.3, m exponential) the Wells marsh will be more productive and will remain as a significant carbon sink throughout the century but its productivity might decrease after 2100.

Moderate sea level rise will enhance marsh accretion and expansion and therefore carbon storage. If, however, sea level rise exceeds a threshold, marsh stability can be compromised making it a potential carbon source. In this study I found that a threshold of 1.3 m sea level rise could tip the Wells marsh from being a carbon sink to a carbon source. Similar marshes near or past this threshold are unlikely to survive accelerated 21st century sea level rise. The highest sea level rise projected for Maine follows a non-linear trend with an endpoint of 1.3 m at the end of the century so it is near or within the threshold. If sea level rise stays between the historical rate (i.e., endpoint at 0.5 m) and the state-wide projected rate, the Wells would continue to be a net carbon sink at least within the century.

Whether the sea level rise trend is linear or non-linear, its effects on marsh persistence and carbon sequestration are always non-linear. Through time, marsh area and its potential to sequester carbon will increase and at some point decrease once inland expansion becomes limited and optimum surface elevation decreases. At the landscape scale, salt marsh loss (or gain) will not be spatially uniform as the proximity and nature of inland barriers differs, as demonstrated by this spatially explicit model.

Thresholds exist in wetland response to sea level rise. At a certain rate of sea level rise, the marsh can increase in productivity but eventually lose production in a short period if there is coastal squeeze. Because of this, I suggest a more precautionary approach in managing wetland adaptation. For example, in areas with projected sea level rise of >1 m, it will be prudent to base management strategies on an exponential trend as this would compel planners to plan under a worst case scenario context. In areas with projected sea level rise of < 1 m, adopting a linear trend could be more politically acceptable as it may dispel fears about exaggerated climate change impacts from an exponential trend.

Identification of the limits of salt marsh resilience is one of the important challenges in predicting how they will fare with 21st century sea level rise. By using a spatially explicit model, I am able to project the impacts of different sea level rise trends on a tidal marsh, map the variability of carbon storage in a marsh landscape, and identify critical thresholds when shifts are likely to take place. Finally, restoration that does not account for upland migration will be short-sighted. Long term tidal wetland conservation and restoration strategies should therefore include locating and maintaining inland areas suitable for wetland migration.

By using spatially and temporally explicit GIS models, this study was able to highlight the spatial and temporal variability of carbon storage in tidal wetland which are not currently addressed in existing models. Despite these, there are several limitations of the modelling approach that merit further research and improvement. First, the assumption of vertical accretion is simplistic, that is, there is negligible or no vertical accretion at a certain distance from the creek. Consequently, it is assumed that there is no or little belowground production in areas far from creeks. To improve existing accretion models or estimates, they should be complemented with data or model estimates of rates based on the distance of the marsh surface to upland edges or immediate sources of sediments. This would provide a more realistic estimate of vertical accretion and belowground production over the entire elevation range of the tidal wetland. Including a more realistic estimate of accretion rates in the current GIS model would greatly improve estimates of belowground production and carbon storage. Second, because the model was only implemented on a meso-tidal marsh it is not known to what degree the results can be generalized to other marshes or mangroves in macro- or micro-tidal regimes.

Table 6-1. Equations used to estimate yearly sea levels in both linear and non-linear trends based upon projected sea levels in the year 2100.

2100 Sea Level (m)	Equation	Basis
Linear trend		
0.5	SL = 0.00288*YEAR-5.45	Historical trend (Gehrels et al. 2002; Gehrels 1999)
1.0	SL = 0.009*YEAR-17.9	Global trend (Alexander et al. 2013)
1.5	SL = 0.014*YEAR-29.067	Global trend (Alexander et al. 2013)
2.0	SL = 0.02011*YEAR-40.23	Global trend (Alexander et al. 2013)
Exponential trend		
0.5	$SL = 4E\text{-}80*YEAR^{23.8}$	IPCC RCP2.6 Maximum (Alexander et al. 2013)
1.0	$SL = 2E\text{-}109*YEAR^{32.66}$	IPCC RCP8.5 Maximum (Alexander et al. 2013)
1.5	$SL = 4E\text{-}187*YEAR^{56.17}$	Global Trend (Hoffman et al. 1983)
2.0	$SL=0.0017*(Y-1992)+1.56E\text{-}04*(Y-1992)^2$	US Trend (Parris et al. 2012)
Non-linear relative sea level projection (Portland Maine)		
1.3	$SL = 9E\text{-}246*YEAR^{73.798}$	Tebaldi et al. (2012)

Figure 6-1. The salt marsh complex (in hatch marks) in Wells. Inset map shows location of Wells, Maine, USA.

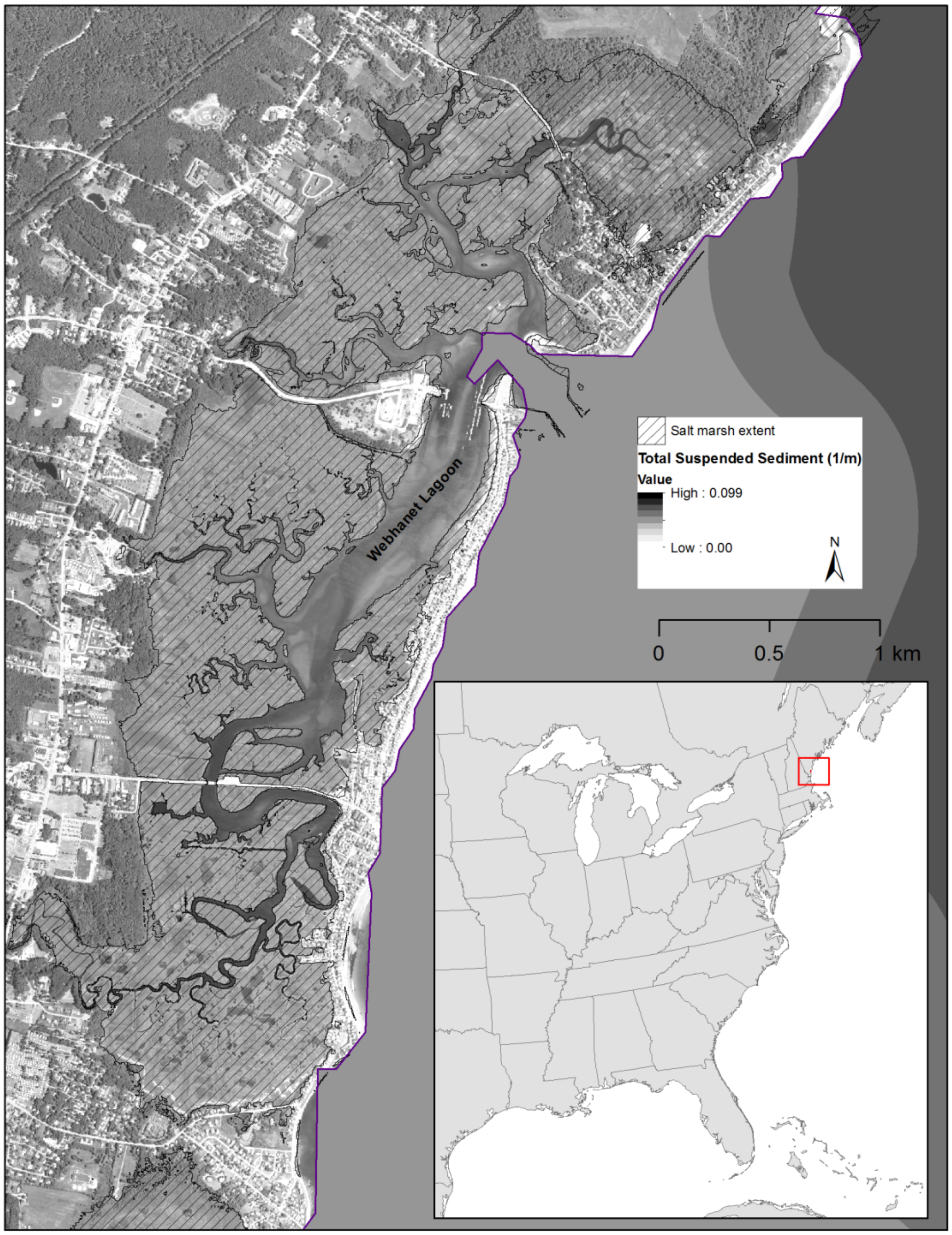


Figure 6-2. Linear (A) and exponential sea level rise trends (B).

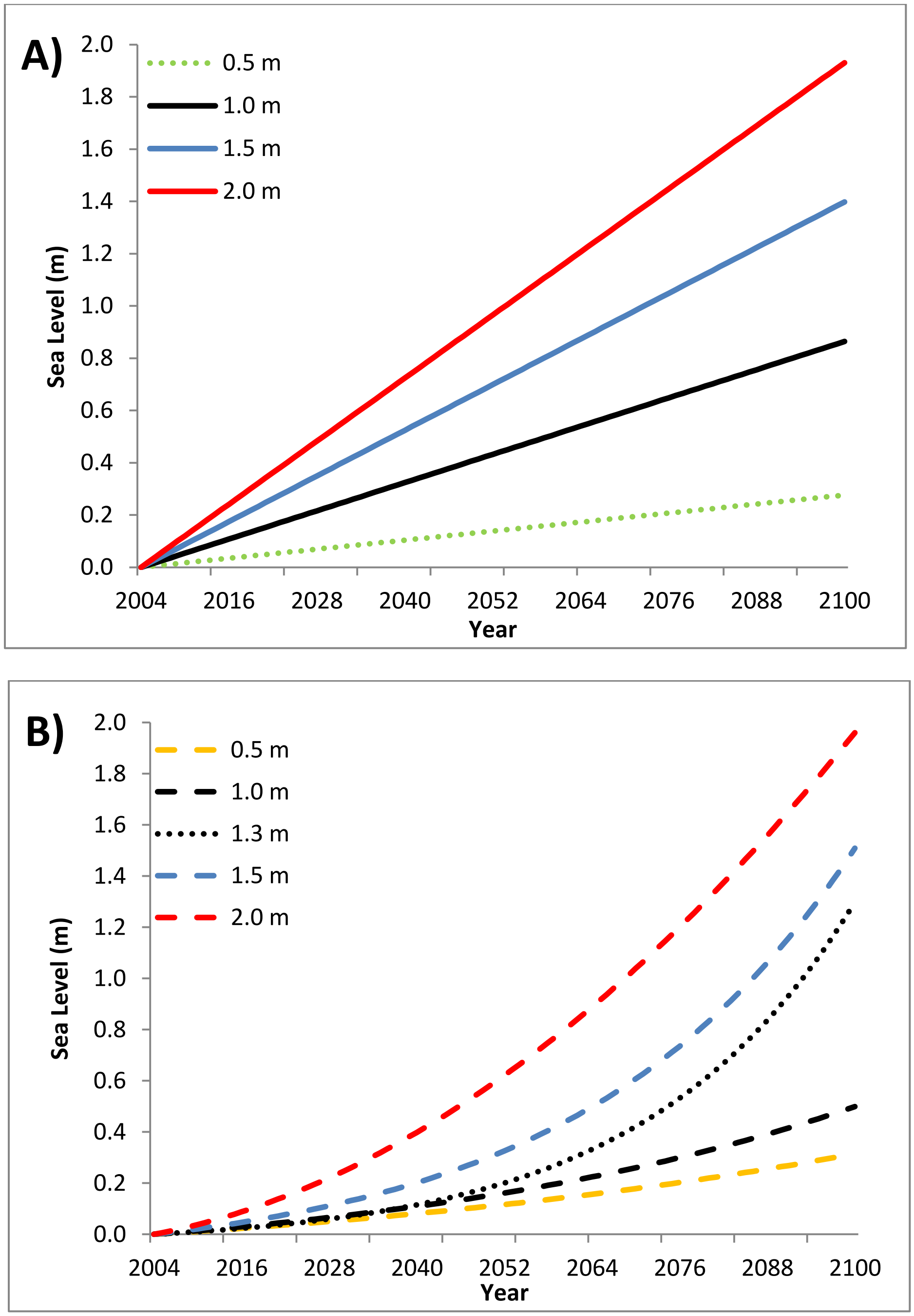


Figure 6-3. Area (A) and percent change in area (B) of the Wells marsh under different sea level rise endpoints (m). The highlighted endpoint corresponds to the projected future relative sea level rise in Portland, Maine, 48 km from Wells.

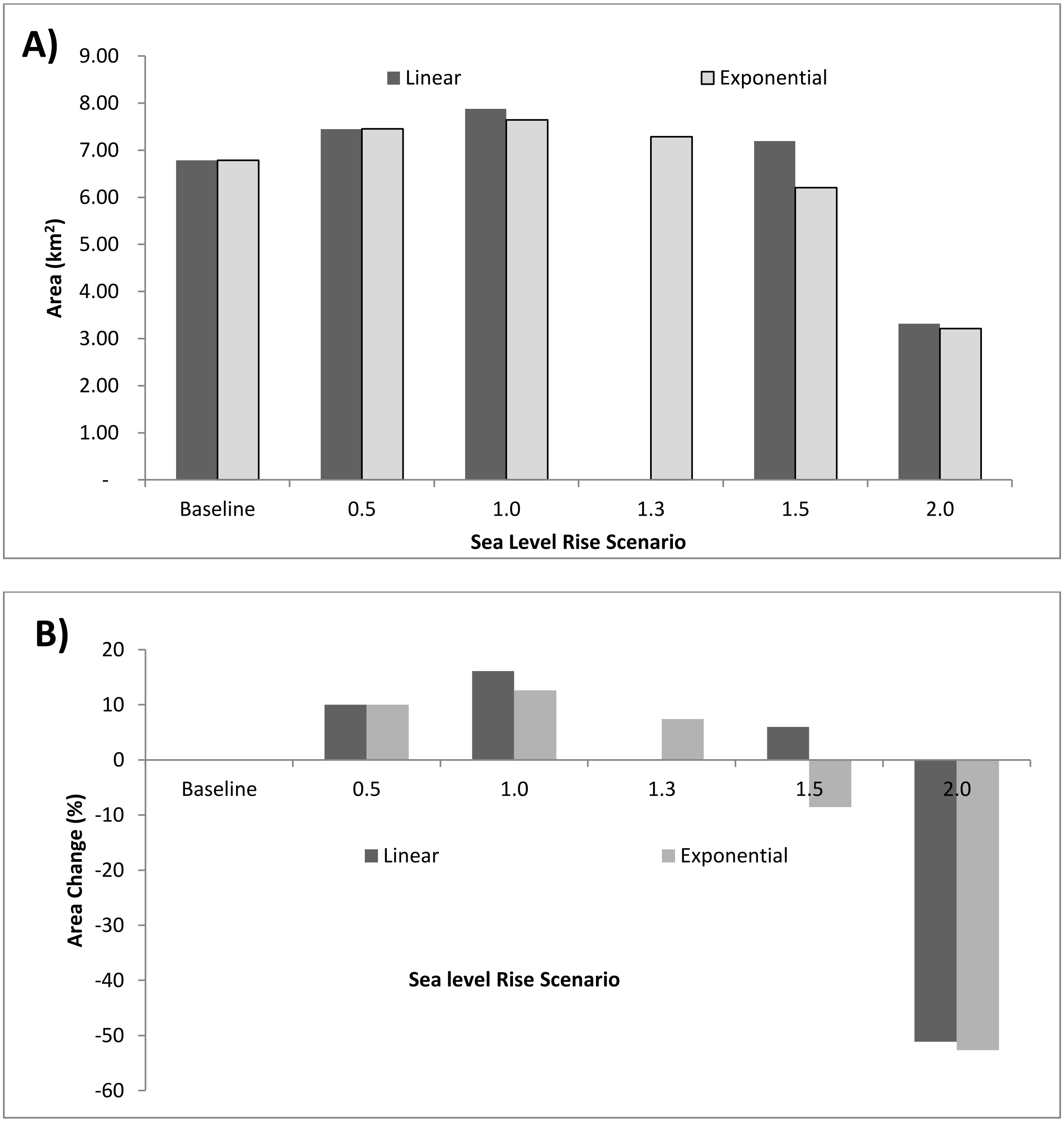


Figure 6-4. Change in Wells marsh are after 100 years under different rates of projected sea level rise: 0.5 m linear and exponential (A & B, respectively); 1.0 m linear and exponential (C & D, respectively); exponential 1.3 m relative sea level rise for Maine (E); 1.5 m linear and exponential (F&G, respectively), red boxes indicate areas where changes are most prominent.

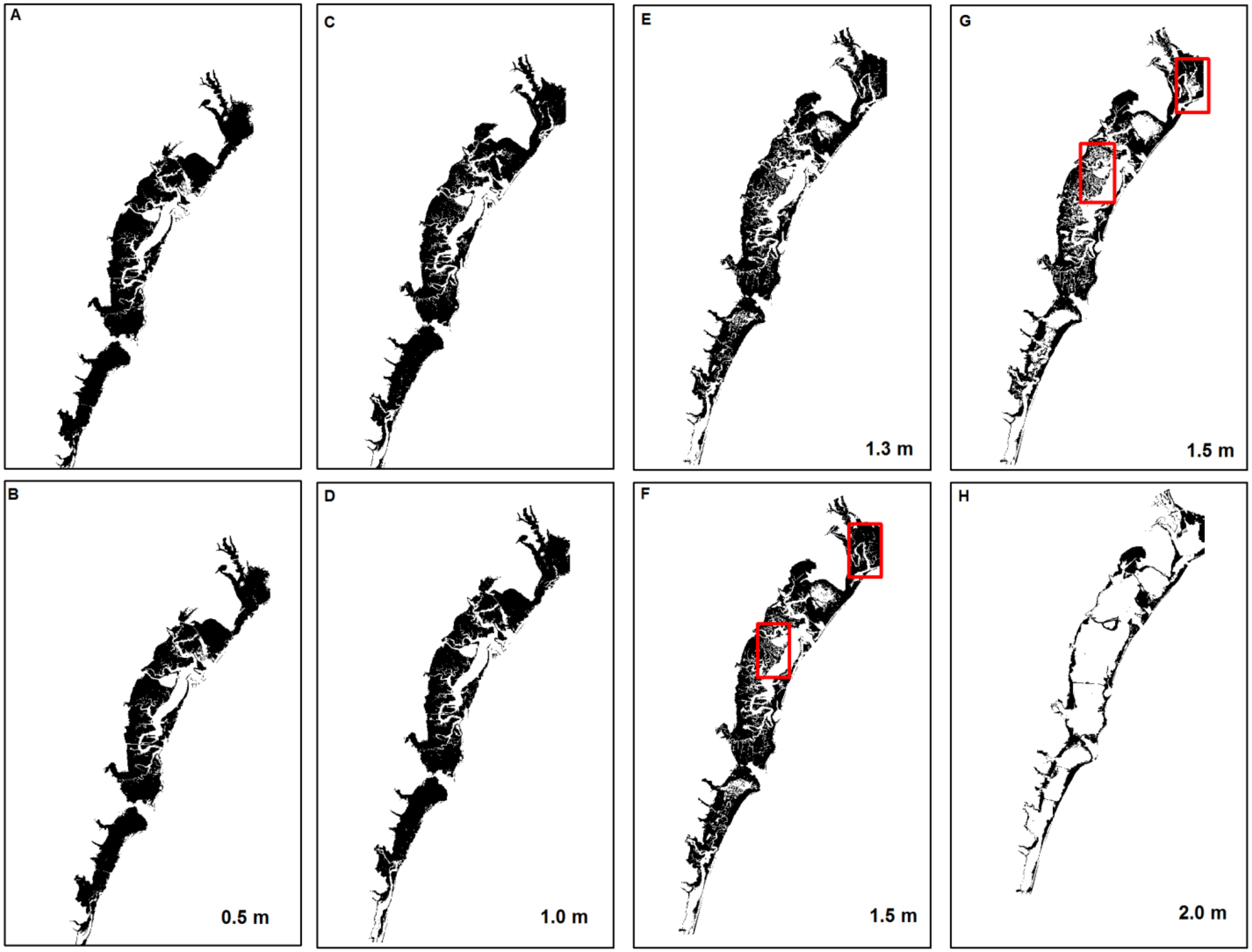


Figure 6-5. Yearly optimum belowground C accumulation under projected linear and exponential rates of sea level rise (m)

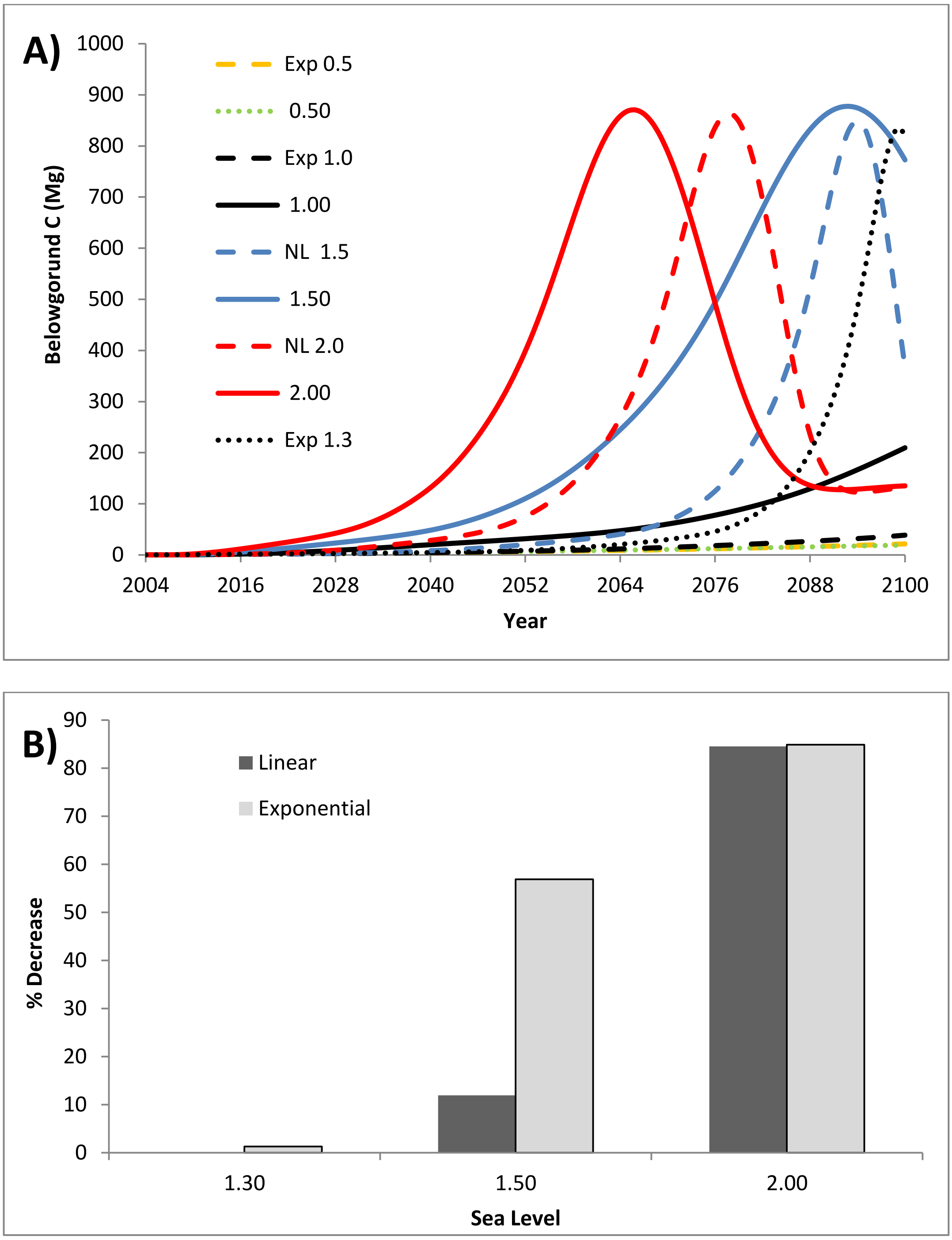


Figure 6-6. Variability in the spatial distribution of belowground carbon in Wells marsh at 1.3 m (A), 1.5 m (B) and 2 m (C) in 2100

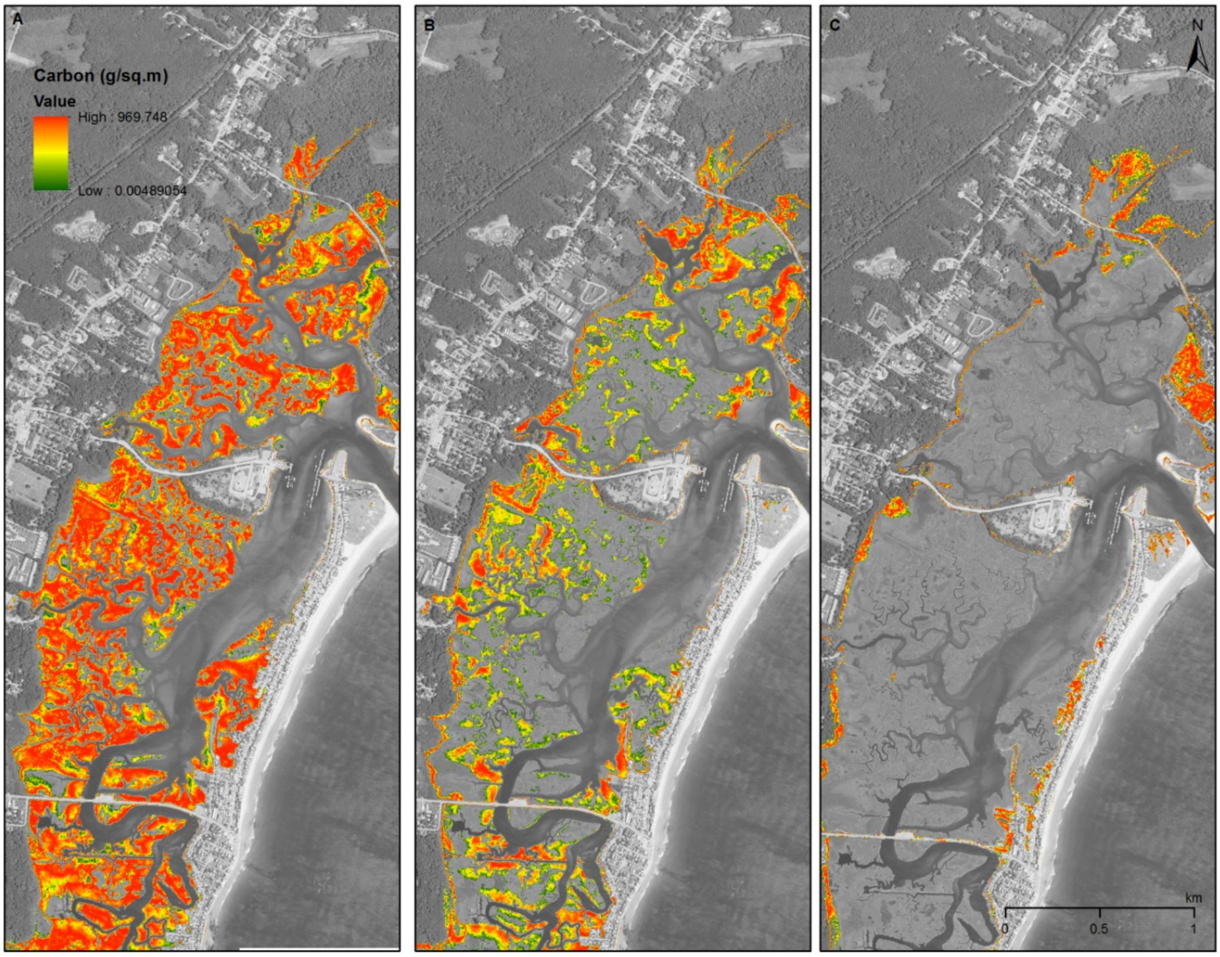
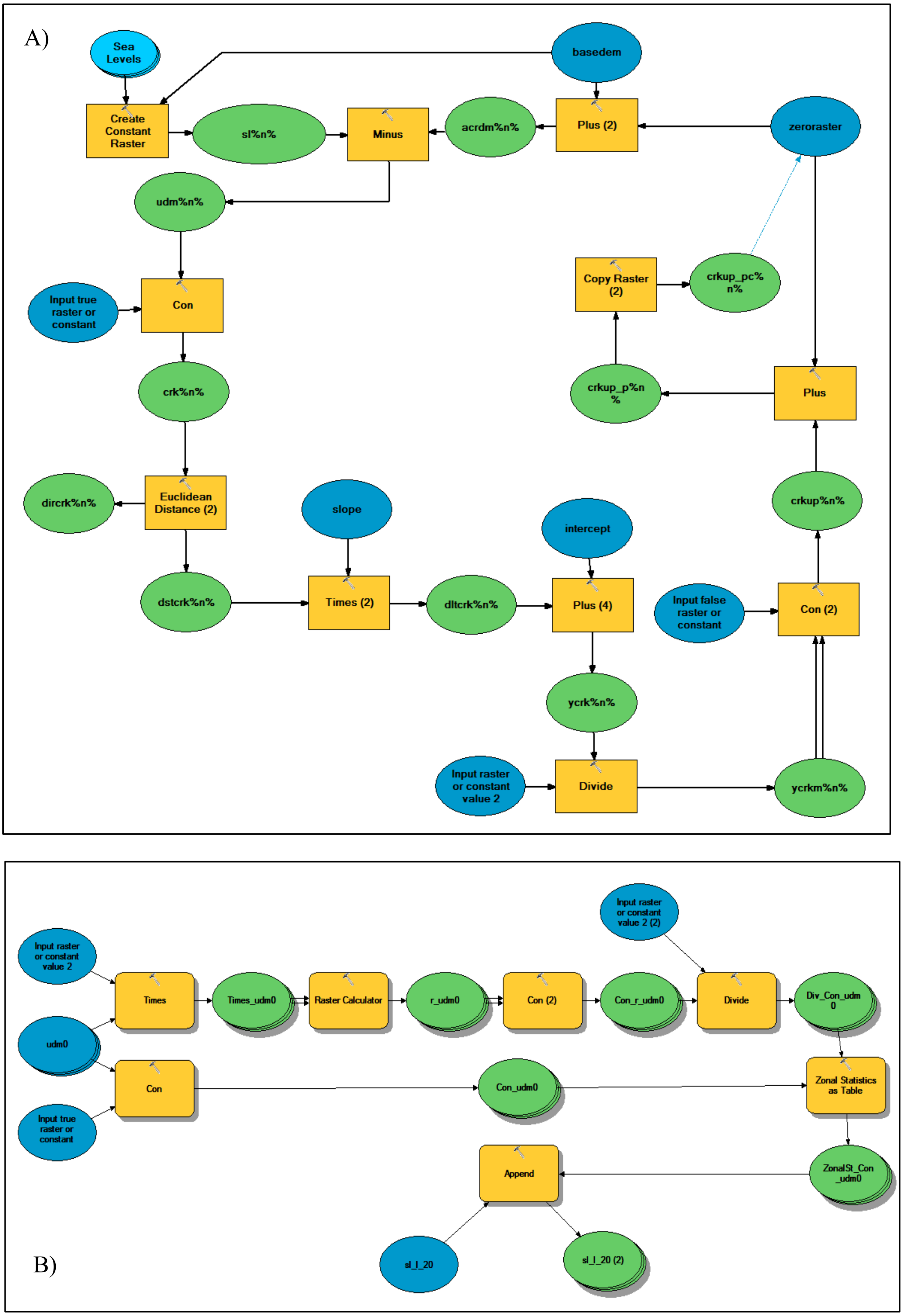


Figure 6-7. ArcGIS implementation of the marsh inundation (A) and optimum production model (B)



6.6 Appendix

6.6.1 Modelling steps:

1. Obtain the yearly elevation generated from an inundation model
2. Multiply the net elevation by 100 to convert meter to centimetres
3. Perform conditional statement to map the marsh zone from the a range of net elevation (assign 1 if 1.95 ≥net elevation ≥ -0.02)
4. The net elevation (in cm) is converted to belowground production [g/m3] using Kirwan & Guntenspergen (2012) equation [y (g.m-3) = -6x2+375.7x-3315.8), where x = net elevation in cm]
5. Select all positive belowground production using conditional statement
6. Convert the positive belowground production values to $Mg.m^3$ by dividing it by 10^6
7. Sum the belowground production over the surface of the marsh using the zones identified in step 2.
8. Run the model for all the years

Chapter 7 Summary, conclusions, limitations and future research

Through spatially explicit modeling, this study addressed two fundamental questions about the response of tidal wetlands to projected accelerated sea level rise. Amidst emerging 21^{st} century threats associated with sea level rise, what is the future for tidal wetlands? How will they persist and their function change with expected rates of sea level rise over this century? Chapters 3 and 4 answered the first question and addressed several limitations from previous modelling efforts. Using fuzzy logic, coastal squeeze- an important emerging threat- was also quantified as a continuous environmental gradient. The index of coastal squeeze was used to measure the magnitude of coastal squeeze and its impacts on salt marshes and mangroves at local sites and continent-wide. Chapters 5 and 6 answered the second research question reporting on the development and implementation of more detailed and sophisticated models to understand the impacts of sea level rise and coastal squeeze to two fundamental functions of tidal wetlands, provision of habitat for fishes and carbon sequestration. The modelling approach progressed from simple models that simulate single wetland process under simple sea level rise assumptions to models that simulate wetland change under different rates and trends of sea level rise. Results of this thesis improve our understanding of the threats posed by sea level rise.

7.1 Summary of findings

7.1.1 Assessing coastal squeeze of tidal wetlands

An index of coastal squeeze has been developed from two main parameters: slope and imperviousness. Using a fuzzy logic approach, equations were formulated to quantify coastal squeeze by slope and imperviousness as continuous environmental gradients and an ArcGIS model was developed to combine individual indices to a unified Coastal Squeeze Index. In this study I found that 11.5^{o} is the threshold slope for coastal squeeze. Above this, an additional 1^{o} increase in slope will increase coastal squeeze potential nearly fourfold. Similarly, the imperviousness threshold is 15.4%. Above this threshold, an additional 1 % increase will increase coastal squeeze potential nearly five times. Below the thresholds, coastal squeeze is negligible.

Perhaps the greatest utility of the index is in ranking the threat of coastal squeeze at locations around a tidal wetland – or among tidal wetlands to inform land use planners, guide prioritization of conservation efforts, and help allocate limited funding to areas with higher potential for permanence (i.e., low coastal squeeze threats). Alternately, the index can be used to allocate future intertidal zones to non-intensive uses. Among the three sites studied, the Wells marsh has a higher coastal squeeze mainly from imperviousness. The Portland marsh has moderate coastal squeeze rank, with slope having more influence. The KNP marsh is least threatened by coastal squeeze and have substantial expansion area inland.

7.1.2 Assessing coastal squeeze of North American tidal wetlands

This is the first attempt of a continent-wide coastal squeeze threat assessment. Equations and models of coastal squeeze were developed and applied on a continent-wide analysis of coastal squeeze threat to rank marshes and mangroves in Canada, United States and Mexico. The results could inform national and international policies to mitigate climate change, conserve biodiversity and build resilient networks of Marine and Terrestrial Protected Areas.

About 10% of North America's tidal wetlands are threatened by high coastal squeeze. Of the two coastal squeeze variables; imperviousness threatens more tidal wetlands than slope – providing the potential for mitigation for most of the areas where the threat of coastal squeeze is present. Less than 1% of the total salt marsh and mangrove areas are bordered by steep slopes while 10% are under the threat of imperviousness. Of the two habitats, marshes are more threatened than mangroves. Tidal wetlands in the USA and Canada are more threatened than wetlands in Mexico.

Tidal wetlands located in Marine and Terrestrial Protected Areas are not protected from rising sea levels and coastal squeeze. As such, static and geographically fixed Protected Areas will not be an effective management tool with climate change and simple protection from direct disturbance will not be enough to ensure their future. We may protect them from direct threats with fixed reserves, however, long term management of Protected Areas must consider the future migration space for wetlands and future reserves should include spatially and temporally dynamic buffers or easements to build a resilient network.

7.1.3 Impacts of sea level rise on marsh as fish habitat

Drastic changes could happen to a marsh if it does not accrete sufficiently with sea level rise (i.e., constant accretion). If sea level rises to ≥ 1 m with no significant increase in marsh accretion one can expect creeks to expand while marsh area decreases. This would result in smaller and isolated patches with less carrying capacity to support marsh-dependent fish. Upland and seaward edges will become less convoluted or have a decreased fractal dimension. Low fractal dimension may favor predatory species and decrease marsh provision of refugia for small prey species like mummichogs. As such, sea level rise may shift habitat selection and utilization. At Wells marsh, fractal dimension and sea level of more than 1 m are inversely proportional. The projected year 2100 sea level for Maine is 1.3 m. Model results demonstrate that sea level rise would be a significant factor contributing to marsh change if it exceeds the marsh accretion rate of 0.36 cm yr^{-1}.

My model results support the hypothesis that coastal squeeze and accelerated sea level rise will lead to wetland degradation by modifying marsh spatial structure, subhabitat composition, configuration, and connectivity, all of which have important implications on how fish use the marsh as habitat. The model results indicate that moderate sea level rise would benefit marshes by expanding their area and maintaining habitat spatial complexity, but higher sea levels coupled with stressors like coastal squeeze are likely to result in more dispersed patches with lesser edge complexity and connectedness. Under these conditions, the function of marshes to provide adequate source of food and refugia for a variety of coastal fishes and even terrestrial organisms is at greater risk of deterioration. The result also lend support to the hypothesis that marshes under gradual submergence are at the stage similar to a deltaic marsh in a destruction phase (Scruton 1960) with increased secondary production but which eventually deteriorates if a threshold in sea level rise and accretion is exceeded. The relationship among the impacts of sea level rise and coastal squeeze indicates that there is a tipping point at which a maximum sea level that must be exceeded under a certain rate of accretion for a marsh shows some indications of degradation.

A calculation of fractal dimensions (which indicate the edge complexity) over time will reveal the impacts of anthropogenic perturbations as well as rising sea level. Because fractal dimensions are independent of scale, such analyses place more emphasis on habitat edge complexity than the size of the habitat (Kent and Wong 1982). Fractal shapes may be used as a template in wetland creation and in designing conservation areas that are optimally distributed in space with boundaries that follow the shape of a dominant natural land-form. Furthermore, fractal dimension may serve as a surrogate metric for habitat quality of wetlands. In addition, measuring the changes in fractal dimension or evaluating species response to changes in edge complexity (O'Connell and Nyman 2010) may help locate marsh sections that are severely impacted by coastal squeeze and identify restoration areas that can maximize the potential for refugia.

7.1.4 Sensitivity of salt marsh carbon to sea level rise

A spatial-temporal GIS inundation model was developed to predict change in elevation of the marsh surface relative to different sea level rise rates and spatially varying accretion rates to estimate the yearly carbon sequestration of an active carbon sink. The model included optimum production component to estimate belowground carbon sequestration and evaluate the linear and non-linear assumptions of sea level rise rates. These improved upon the limitations of existing point based and spatially enabled numerical models.

The linear rate of sea level rise enhances upland expansion by allowing flat areas to be flooded earlier as the sea level is higher than in non-linear scenarios throughout the run. However, when coastal squeeze becomes significant, the marsh can no longer expand at the upland edge and the overall area starts to decline at the seaward edge. Marsh area loss in the linear models area earlier but the rate of decrease is slower. Conversely, in the exponential models, marsh area loss occurs later but at a more rapid rate.

Moderate sea level rise will enhance marsh accretion and expansion and therefore carbon storage. If however sea level rise exceeds a threshold, marsh stability can be compromised making it a potential carbon source. In this study I found that a threshold of 1.3 m sea level rise could tip the Wells marsh from being a carbon sink to a potential carbon source. Similar marshes near or past this threshold are unlikely to survive accelerated 21st century sea level rise. The highest sea level rise projected for Maine follows a non-linear trend with an endpoint of 1.3 m at the end of the century so it is near or within the threshold. If sea level rise stays between the historical rate (i.e., endpoint at 0.5 m) and the state-wide projected rate, the Wells would continue to be a net carbon sink at least within the century.

Whether the sea level rise trend is linear or non-linear its effects on marsh persistence and carbon sequestration is always non-linear. Through time, marsh area and its potential to sequester carbon will increase and at some point decrease once inland expansion becomes limited and optimum surface elevation decreases. At the landscape scale, salt marsh loss (or gain) will not be spatially uniform as the proximity and nature of inland barriers differs, as demonstrated by the spatially explicit model.

Thresholds exist in wetland response to sea level rise. At a certain rate of sea level rise, the marsh can increase in productivity but eventually losses production in a short period if there is coastal squeeze. Because of this, I suggest a more precautionary approach in managing wetland adaptation. For example, in areas with projected sea level rise of more than 1 m, it will be prudent to base management strategies on an exponential trend as this would compel planners to plan under a worst case scenario. In areas with projected sea level rise of less than a meter, adopting a linear trend could be more politically acceptable as it may dispel fears about exaggerated climate change impacts from an exponential trend.

A novelty of the work lies in the use of spatially explicit model to map the variability of belowground carbon in a salt marsh landscape at certain time period and sea level rise trend. Compared to other models I believe that opportunity for variable C accumulation rates over the marsh surface makes the response of the models more detailed and sophisticated. By mapping the variability of carbon storage I show that the impacts of sea level are not uniform across a marsh.

7.2 General conclusions

7.2.1 Coastal squeeze threatens tidal wetland sustainability

Coastal squeeze may result in degradation and loss of salt marshes and mangroves. In North America coastal squeeze is largely anthropogenic with salt marshes more threatened than mangroves. Because the threat is anthropogenic, management strategies can address coastal squeeze by ensuring that there is enough suitable inland space to accommodate wetland migration. This means implementing policies that prevent further development of inland coastal areas especially conversion to impervious surfaces. Existing policies of protecting tidal wetlands by establishing them as protected areas have limited value if the migration buffers are not adequately protected. Currently, coastal squeeze is not included as criteria in designing many marine protected areas for tidal wetlands. It is no longer adequate to protect wetlands on their current locations. To ensure the resilience and permanence of tidal wetlands under increasing sea level rise, their migration areas need to be protected as well.

Knowing the location and level of coastal squeeze can help in setting conservation or restoration priorities. For example in marshes and mangroves under low to medium threat identified through the coastal squeeze index described in this thesis, a more generalized and long term approach can be implemented and priorities can be based upon potential for migration. For tidal wetlands under high threat a more specific near term approach might be necessary based upon the potential to provide the most ecosystem services (e.g., carbon sequestration, recreation, and storm buffer). Generalized intervention may include preventing further development, keeping the present non-intensive land use of the migration space and systematic allocation of the coastal zones while more specific interventions may include tidal regime restoration or enhancing sediment supply for sediment-deprived habitats. More work is needed to assess which of the approaches are likely to succeed under the prevailing biophysical and socio-economic of a particular management scale.

7.2.2 Sea level rise and coastal squeeze create dysfunctional wetlands

The landscape ecology approach applied in this thesis shows that coastal squeeze contributes to wetland degradation by amplifying submergence, fragmentation and eventual degradation of the wetlands. This could happen when marshes and mangroves do not vertically accrete at the same rate as sea level rise. Wetlands under prolonged inundation have reduced complexity in edge density, shape, and spatial distribution of wetland patches. In particular, reduction in marsh edge fractal dimension on both the landward and seaward edges can lower the potential of the habitat to refugia for both terrestrial and aquatic organisms as the convoluted edges of the wetland patch are simplified into straight line features reducing their potential as hiding places. For small marsh dependent fishes like mummichogs, the far distance between fragmented patches could prevent efficient use of the resources in the marshes and mangroves and could enhance the risk of predation. In all, my findings suggest that a species-centred approach to conservation should be complemented with conservation of habitats. Conserving areas that are likely to hold the properties that support future occupants could ensure not only the resilience but also the permanence of the populations in the midst of environmental change.

Carbon sequestration is one of the important roles played by tidal wetlands. I found that this function is sensitive to sea level rise and coastal squeeze. In an ideal setting, sea level rise does not necessarily mean marsh loss. Marshes may still persist if sea level continues to rise at recent historical rates, but if sea level rises at rates beyond what the plants can tolerate and migration space is limited, the magnitude of the active carbon sink in tidal marshes and mangroves will be compromised because of submergence of the seaward edges. Allowing marshes to expand inland can compensate for seaward losses. For example, if sea level does not exceed 1.3 m, the Wells marsh will persist, expand and continue to store carbon over this century. In this case, a sea level rise rate of ≤ 1.30 cm yr^{-1} even contributes positively to carbon storage by increasing marsh area and belowground production without substantial submergence towards the end of the century. At present, the Wells marsh is a stable marsh.

7.2.3 There is a tipping point and lag in sea level rise and wetland change

The model simulation in this thesis provides evidence on the existence of ecological thresholds and tipping points in marsh carbon sequestration and accelerated sea level rise. At sea level rise rates of >1.3 m, exponential trends are likely to bring about a tipping point and wetland change. At rates corresponding to a 1.3, 1.5 and 2.0 m rise, the tipping points would occur around the years 2099, 2086 and 2066, respectively. Possible regime shifts need to be anticipated and included in the management of marshes as carbon sinks. This means that coastal planning legislations for sea level rise that exceeds 1.3 m should adopt a precautionary principle while those below could be flexibility but keeping in mind ecological and social implications.

An important advantage of having the spatial component included in a model is that its output reveals a lag between the loss of marsh area and loss of belowground production. In the example of the Wells marsh, a decrease in belowground production occurs at an average of 12 years after loss of marsh area at sea level rise rates of ≥1.3 cm yr^{-1}. Point-based non- spatially explicit models cannot capture these trends as they provide a more simplistic or averaged projection of marsh change.

7.3 Limitations and future research

By using spatially explicit models, this study was able to highlight important trends of tidal wetland change with sea level rise and the utility of flexible GIS models in tidal wetland assessment. Despite these, there are several limitations of the study that merit further research. First, the assumption on vertical accretion is simplistic, that is, there is negligible or no vertical accretion at a certain distance from the creek. To improve existing accretion models or estimates, they should be complemented with data or model estimates of rates based on the distance of the marsh surface to upland edges or immediate sources of sediments. This would provide a more realistic estimate of vertical accretion and belowground production over the entire elevation range of the tidal wetland. Including a more realistic estimate of accretion rates into the current GIS model would greatly improve estimates of belowground production and carbon storage. Second, because the carbon model was only implemented on a meso-tidal marsh it is not known whether the results can be generalized to other marshes or mangroves in the macro or micro tidal regimes. Third, with climate change it is uncertain if the lost functions of tidal wetlands from potential shift in their current distribution and coastal squeeze would be replaced by other ecosystems (e.g., mud flats) with different yet still important functions. As such, studies on trade-offs of ecosystem functions and services are needed to inform management. Further research is also needed to test the link between changes in fractal dimension with change in habitat quality of restored coastal systems and validate the results with empirical data on fish population and species distribution and predator-prey dynamics. Fourth, with sea level rise, it is certain that some wetlands will be submerged and eroded but knowing the fate of the carbon stored in these wetlands remains incomplete. And finally, there is a need to upscale local studies of wetland landscape ecology (e.g., Chapter 5) into regional and global extents. With climate change the global and regional distribution of marshes and mangrove will become more important in sustaining population of migratory species.

Some data and methodology gaps were also noted but, beyond the scope of this study. First, the equation of coastal squeeze slope is sensitive to vertical and horizontal resolution of elevation data. At present, the only global elevation models that are publicly available have coarse vertical and horizontal resolutions. This necessitates adjusting the equation on coastal squeeze slope. With higher resolution data, the adjustment might not be necessary. Second, the model did not account for stochastic events which may influence wetland permanence with climate change. For example, the lagoon at Wells is sheltered by a sand barrier that can be affected by rising sea level, and in particular, high energy storms that are predicted to occur with warming climate (Meehl et al. 2007). Both will cause the landward migration of the barrier and eventual transgression over the marsh, destroying it before rising sea level does. Lastly, salt marshes in Mexico are underrepresented such that their threat levels were underestimated.

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