Opportunities & Limitations of Coastal Wetlands in Reducing Coastal Risk in Light of Projected Sea-Level Rise

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ABSTRACT

Sea level rise (SLR) is a global problem that will have catastrophic effects on coastal wetlands, which not only house nearly half of the world's population but also provide essential ecosystem services (ESS). SLR will increase the vulnerability to coastal risk, by making coastal flooding, erosion, and storm surges more likely. Ecosystem-based adaptation (EbA) is a cost-effective strategy to protect low-lying coasts from SLR by enabling the wetlands to fulfil their adaptive potential. Most coastal wetlands have inherent strategies for coping with SLR, such as inland or vertical movement, a regulating ESS they provide. Additionally, coastal wetlands provide a number of provisioning, supporting, and cultural ESS, which would be lost if the wetland is drowned by SLR. To be able to provide these services, wetlands require certain conditions that are not always given. This is a great opportunity for EbA strategies that protect the wetlands, and therefore the ESS they provide to stakeholders. The potential of coastal wetlands to reduce coastal risk in light of SLR is limited by a number of physical and socio-economic factors. The opportunities and limits of coastal EbA measures are discussed and evaluated, resulting in the recommendation that well-planned measures must utilize a multidisciplinary approach.

Contents

1. Introduction

Coasts are home to nearly half of the world's population but occupy only about 7% of the land area on Earth (Sterzel, *et al*., 2020). Among these populations, more than 600 million people live 10 meters or less above sea-level, and over 2 billion people living within 100km of a coastline (UN, 2017). Coupled with the wide range of climate and environmental changes predicted to occur throughout the remainder of the 21st century, coastlines present a critical point of interest in understanding the impacts beginning to emerge on a global level. Increased sea surface temperatures (SST), sea-level rise (SLR), changes in oceanic circulation patterns, atmospheric circulation patterns, salinity variability all have the ability to fundamentally alter the way that coastal wetland ecosystems operate both on micro and macro scales (IPCC, 2019). Some of the impacts which have already been observed include increased frequency of extreme flooding events (EFE), increases in magnitude, duration and frequency of tropical cyclone (TC) activity, droughts, heatwaves, destruction of wildlife and habitats, threatened or destroyed ecosystems, radical fluctuations in El Niño/La Niña years and coral bleaching. In addition to these, many pristine coastal wetland areas have been severely modified or destroyed entirely to serve human needs; this threatens a wide range of complex, natural processes which work not only to sustain the ecosystems and biodiversity which thrive within these regions but also endanger the ability of coastal wetlands to provide a buffer for coastal communities around the world (Oppenheimer, *et al*., 2019).

The sheer variety in types of ecosystem services (ESS) provided by coastal wetlands has made them a region of interest for many sectors including agriculture, real estate, risk reduction and tourism. Rates and severity of degradation, however, exceed those of almost any other ecosystem, specifically relative to the impacts of SLR (Millennium Ecosystem Assessment, 2005). Consequently, the necessity of preserving these valuable ecosystems has become increasingly apparent. With growing public awareness and concern pertaining to climate change and related risks, topics such as carbon sequestration and flood mitigation have risen to the forefront of coastal research. This paper will examine some of the contemporary efforts in reducing risk in coastal communities around the world; and attempt to argue how, perhaps, we may find solutions to adapt to a changing climate in preserving and protecting sustainable wetlands.

2. Background

2.1 Coastal Wetlands

Definitions of coasts vary throughout scientific literature, but coastlines can be generally defined as the area of land which extends from a large water body inland, until which point the land is no longer affected by coastal processes (Parish, *et al*., 2007). Perhaps the reason for such a vague definition can be explained by the numerous types of coasts present around the world. In this paper, the focus highlights coastal wetlands, a more specific variety of coastline in which a number of anaerobic processes are continuously at work. According to the Ramsar definition wetlands, “are areas of marsh, fen peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters,” (Article 1.1). The Ramsar definition is one which was adopted in 1971, during the Ramsar Convention on Wetlands of International Importance in Ramsar, Iran when 170 countries signed a treaty, “for the conservation and sustainable use of wetlands,” (Ramsar, 2010).

Highly diverse and unique ecosystems, wetlands are home to an array of hydrophytes, or aquatic plants, that have adapted over millions of years to the unique properties of wetland soils (Joosten, *et al*., 2002). Individual features of particular wetland types depend on the presence, duration of presence, type (either saltwater, freshwater or brackish water) and the amount of water in a particular region. Certain characteristics may also arise in wetland ecosystems which experience seasonal flooding, versus regions which experience more sustained water levels as well as amongst wetlands which inundate with daily tides and those of non-tidal environments (Graham, *et al*., 1980). Swamps, marshes, bogs, and fens are the four main types of wetlands found around the world; and are home to smaller, more specific sub-categorized wetlands like mangroves or floodplains.

In recent decades, researchers have found that despite having specialized types of soils and plants with the ability to adapt to flooded conditions or withstand fluctuating levels of water and salinity, conditions related to climate change, such as SLR and SST increases are beginning to threaten the biological stability of these ecosystems (Parish, *et al*., 2007). Threatened coastal ecosystems directly contribute to a number hardships including disruption in economic activity dependent upon wetland resources, access to clean water, decreased tourism, damage to local infrastructure and destabilized social livelihoods. In 2005 the UNMEA found that wetlands experience a higher level of environmental degradation than any other ecosystem (Ramsar, 2010). Subsequently, conservation efforts have placed a heavy focus on the plethora of ESS which wetlands provide, in order to secure funding and educate communities for coastal wetland protection efforts.

2.2. Ecosystem Services

Ecosystem services are comprised of four main types; regulatory, provisioning, supporting and cultural (Millenium Ecosystem Assessment, 2005). Each of these types will be examined, briefly, in this section and further expanded upon in following sections as a suggested and recommended ecosystem-based adaptation (EbA) measure. Wetlands are major contributors to a number of important ESS that have been estimated to be valued around $14 trillion USD, annually (Ramsar, 2010). Wetland management and policy decisions are derived largely through examining, “the development, trends and limitations of wetland ecosystem services (WES),” (Xibao, *et al*., 2020). Covering nearly 4 million square kilometers of land worldwide, coastal wetlands can be found in nearly all climate zones and in a variety of temperatures, making them responsible for capturing the largest amount of atmospheric carbon, as compared to any other ecosystem in the world (Leadley, *et al*., 2014; Buckmaster, *et al*., 2014; Lindsay, 2014). Through this type of regulatory service alone, the significance of maintaining robust wetlands becomes apparent. It is, however, only one example.

As a major contributor of global renewable freshwater supplies, wetlands also endow critical access to a variety of provisional services to both coastal and inland communities, worldwide. This includes access to clean water for up to 3 billion people, according to the Millennium Assessment for the Ramsar Convention and the future of wetlands (MEA, 2005). However, the loss of wetlands for agricultural or commercial land development, water extraction/abstraction, over-exploitation, eutrophication and pollution places enormous pressure on current ESS, while, “demand for these same services is projected to increase,” (Ramsar, 2010). In terms of quantity, water is the dominant resource wetlands offer, it is one of many resources which wetlands contribute to communities worldwide. Timber, fish, rice cranberries and blueberries are also extracted from wetlands and offer a substantial contribution to the economy each year through supporting services (Vallecillo, et al., 2019).

The stability provided by wetlands either via ESS or by direct interaction (i.e. tourism, economic livelihoods and/or improvements in well-being/quality of human life), reinforces the importance of environmental stewardship as a means to maintain not only their functionality but also the sustainability of the many aspects of society and nature which stand to benefit from them (Jiang, *et al*., 2015). Cultural services foster an integral relationship between communities and the environment; enabling a direct link between nature and human well-being to be established and sustained. In recent years, researchers have begun emphasizing quantitative assessments of ESS, in an attempt to improve tradeoff calculations, both in terms of potential and actualized damage observed in coastal wetlands (Barbier, et al., 2008; Buckmaster, et al., 2014; Rezaiel, et al., 2020; Xiabo, et al., 2020). Valuation studies investigate various methods of quantifying ESS, where ESS are defined as, “a useful tool that provides relevant information on the role of ecosystems in delivering services, and the society benefiting from them,” (Vallecillo, *et al*., 2019). Examining the complex mechanisms by which WES can remain intact and valuable is a crucial step in developing a holistic and systematic approach that can take full advantage of the opportunities, while minimizing the limitations of reducing SLR-related risk in coastal/wetland communities.

2.3 Sea Level Rise

When examining the phenomenon of Sea Level Rise (SLR), a clear distinction between regional and global trends needs to be made, since they are driven by different forces. Steric, isostatic and eustatic changes, whose impacts are spread unevenly across the globe, together with storm surges and tidal forcing, influence regional sea--levels and cause them to differ from the global mean. Roughly 90% of Global Mean Sea Level (GMSL) rise is driven by ice melt (eustatic) and ocean thermal expansion (steric) (Palanisamy *et al*., 2014). Ice melt has been identified as the main source of GMSL rise (Oppenheimer *et al*., 2019). Regional SLR, on the other hand, is largely driven by ocean circulation changes, which affected local salinity (steric) and temperature (steric). Isostatic changes, which are external changes in sea-level due to uplift or subduction of land, also only play a role on the regional scale. Isostatic changes can, for example, be caused by the melting of sea ice, which leads to a vertical uplift of the land formerly covered in ice, but can lead to subduction of land near the equator due to SLR. These isostatic changes are often caused directly by human activity, e.g. groundwater extraction, and have become the dominating source of SLR in many deltas (Palanisamy *et al*., 2014; Oppenheimer *et al*., 2019). Due to these spatially varying influences, RSL is expected to vary ±30% around the GMSL rate (Oppenheimer *et al*., 2019). Since the 1970s, these sources of SLR are predominantly caused by anthropogenic forcing (Oppenheimer *et al*., 2019).

Understanding and predicting the SLR at a particularly vulnerable coastal wetland location is thus rather complicated, as the sum of GMSL rise and the regional variability, as well as anthropogenic activity, need to be taken into account (Palanisamy *et al*., 2014; Oppenheimer *et al*., 2019). Nevertheless, it has long been recognised that the effects of SLR need to be understood since it severely threatens coastal wetlands, making coastal erosion, storm surges, and coastal flooding more likely (Nicholls *et al*., 1999). Furthermore, coastal wetlands are “intimately linked” with sea level (SL), as they depend on being partially submerged and can even benefit from periodic flooding (Nicholls *et al*., 1999). SLR is predicted to have catastrophic effects on coastal wetlands even under a ‘best case’ climate change scenario. The most recent IPCC predictions state that GMSL will rise between 0.43m and 0.84m, depending on the Representative Concentration Pathway (RCP) scenario, by the end of the century (Oppenheimer *et al*., 2019). It is clear that even if climate change is mitigated to moderate warming only (RCP 2.5), GMSL will still rise significantly. SLR is an inevitable consequence of climate change, and it will continue for centuries due to the nature of thermal expansion and the losses of the world’s major ice sheets (Oppenheimer *et al*., 2019). In light of this, adaptation strategies urgently need to be developed to protect low-lying coastal wetlands, which will - and are - most affected by SLR.

3. Review of Methodologies

3.1 Quantitative Analyses - Valuation Studies

In order to analyze the severity of SLR effects on coastal wetlands, a variety of methods have been explored by researchers in recent decades. As a broad subject, the subsequent literature spans a wide variety of sectors and makes use of an array of tools. The very mechanism which allows wetlands to flourish, the act of flooding, is ironically one of the major threats to the sustainability of healthy wetlands through sea-level rise (Lindsay, et al., 2014). The disturbance of coastal wetland health attributed to SLR has been well documented (Williams, *et al*., 1999; Joosten, et al., 2002; Lindsay, et al., 2014; EPA, 2020; Arkema, et al., 2013; Ayers, et al., 2014; Doyle, *et al*., 2010); although the breadth of these changes is constantly in flux and dependent upon local variability in hydrological as well as geomorphological gradients. The tolerance of hydrophytes in saltwater environments, for example, becomes challenged as SLs continue to rise (Hossain, *et al*., 2018). Changing SLs can also accompany changing levels of salinity. While varying degrees of salinity are required for specific plant species in upland areas and throughout marsh-upland boundaries, this highly sensitive ratio can destabilize, creating a wide range of unsustainable ramifications (Fagherazzi, *et al*., 2019). Following flooding events and major storms, damage to wetland forests may be severe enough to, "halt regeneration because seedlings can be more sensitive to environmental change and variability," (Donovan *et al*., 2010; Fagherazzi, *et al*., 2019). The unique qualities of such consequences have created the demand for numerical modelling to assist in the quantification of SLR-induced wetland degradation. Fagherazzi, *et al.*, (2019) propose a conceptual model for monitoring the transitional shifts of coastal forests and marsh ecosystems. Represented in Figure 1, the *Ecological Ratchet Model* examines the hydrological exceedance probability for a site, where water is measured as mean sea level (MSL) and Pr{h>z} represents the probability that water levels will exceed elevation (Z) in a given year.

The value of calculating a given commodity's availability in the future, relative to its existence today is also suggested by Fagherazzi *et al*., as an important ESS indicator. The 'discount rate' references the diminishing value of a product in the future, relative to today (Fagherazzi, *et al*., 2019). Discount rate is represented as the inverse of an interest rate; or rate of return on investment (minus rates of inflation). This rate can be calculated with the following formula;

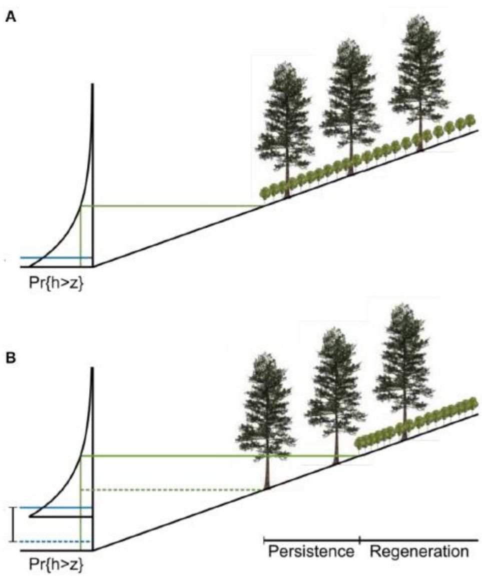

Figure 1: Demonstrating the Ecological Ratchet Model proposed by Fagherazzi, et al., 2019 in which changes in water levels due to SLR can be measured. A) Illustrates an initial forest scenario and B) depicts change relative to A, where a threshold of persistent flooding may surpass into debilitating potential for regeneration.

$$C_t = C_0 \left(1 - \frac{r}{n}\right)^{nt} \qquad (1)$$

Where C_t represents the value at time t in the future and r is the discount rate charged at term n. The r value can also be replaced with a standardized rate of inflation (3% in the US) if consumer price index, tax appraisal or other detailed data exists for a given commodity. Differences in land cover type (private vs. public, in this case) have demonstrated a significant diversion in terms of discount rate over time. Using this equation, two lots of contrasting land types (public/private) possessing equal value today, but differing discount rates (3% and 6%, respectively) amount to a two-fold difference within a period of 25 years, favoring private ownership. Difference of discount rate can be reduced, in this example to 1% and sustains a 10% increase of value difference (in private upland marshes) in a period of only 10 years (Fagherazzi, *et al*., 2019). These type of rate differential calculations may shed light on favoring privately-owned, protected coastal wetlands, as Fagherazzi, et al., describes, "...at the expense of the public trust," if such equations continue to produce numbers similar to those in this study (Fagherazzi, et al., 2019). It should be noted, however, that private land ownership is not synonymous with protected land.

The privatization of upland marsh land, in and of itself, does not protect wetland environments or their ESS, however. The ambiguity of this loosely defined approach in fact hinders broader, more holistic implementation of such measures. Figure 2 shows a concept for the implementation of engineered barriers as a means of protecting upland marshes from the encroachment of seawater. As the demand for coastal property continues to increase, while supply decreases, interest rates for private land continues to rise. The solution suggested in this paper, however, does not specify barrier type to be implemented or the efficacy of such measures. As Sterzel, et al. (2020) points out, often the presence of engineered coastal protection techniques presents a number of issues to the wildlife of wetland and coastal ecosystems. In a study conducted in 2020, Sterzel, et al., found stability rates of fish colonies decrease with the installation of engineered (sometimes referred to as 'hard') shoreline protection.

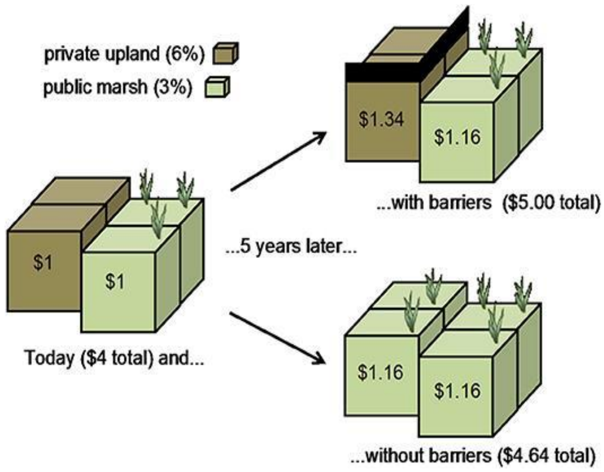

Figure 2: Privately owned upland versus public marsh both valued at $1 when t=0 but when t=5, the differential net value, demonstrated by the private land + with barriers, is notable. (Fagherazzi, et al., 2019).

3.2 Qualitative Analyses

As shown, coastal wetlands provide provisioning, regulating, cultural, and supporting ESS to their surroundings. The cultural ESS, in particular, are difficult to assess using quantitative analyses, which is where the emerging field of qualitative ESS assessments comes to play as a complementary method to existing quantitative assessments (Oppenheimer *et al.*, 2019). The 2019 SROCC report defines five key 'human' dimensions to SLR assessments, which are often overlooked: Power asymmetries, politics, and the prevailing political economy; Gender Inequality; Loss of Indigenous/Local Knowledge; Social Capital; and Risk Perception (Oppenheimer *et al.*, 2019). All of these can obstruct EbA measures, and social barriers to adaptation are already encountered in many parts of the world (Oppenheimer *et al.*, 2019). Highlighting the cultural ESS that coastal wetlands provide, and the danger that SLR poses, is a crucial step in any effective EbA approach. Coastal wetlands are "dynamic environments where multiple interests meet, converge, and sometimes clash", making clear the need for stakeholder involvement (O'Donnell, 2019).

Qualitative assessments are mostly conducted via questionnaires and interviews, allowing stakeholders to express their risk perception and worries, but also their attitude towards potential EbA measures. In many cases, these attitudes are "harmful" to the protection and preservation of coastal wetlands (Xie *et al*., 2010). Often, a coastal wetland is only perceived as "the beach", positing it entirely "in relation to settler human activity and enjoyment", which shows that cultural ESS are taken for granted (O'Donnell, 2019). The other ESS are often not perceived at all, even where residents are faced with the realities of SLR, EFEs, and TC activity daily (O'Donnell, 2019). Only in extremely low-lying coastal areas, such as the Solomon Islands, have respondents reported "fear and worry on a personal and community level" due to SLR (Asugeni *et al.*, 2015). Contrastingly, residents of coastal Australian towns are indifferent to SLR, stating that "it might not happen in our lifetime" (O'Donnell, 2019). Inhabitants of northern Tianjin, China, are quoted to "care little about the wetland protection" if their livelihoods are negatively affected "since crop farming activities [are] forbidden in the natural reserve" (Xie *et al*., 2010). There is thus a time component to risk perception, where many will deny the risk of SLR until it is either too late, or they face the loss of land and property. Additionally, risk perception is informed by "historical and current knowledge" of local weather and climate, but also the belief - or disbelief - in climate change science (O'Donnell, 2019).

Stakeholder opinions of EbA measures must be assessed, as their implementation can "raise equity concerns about marginalising those most vulnerable and could potentially spark or compound social conflict" (Oppenheimer *et al*., 2019). As the field is still emerging, a critical lack of qualitative studies concerning the risks SLR poses to coastal communities persists. To avoid failure of EbA measures due to social barriers, qualitative assessments must always be taken into account when planning an EbA measure for coastal wetland protection.

4. Opportunities

The threats posed by SLR to coastal wetlands have the potential to significantly affect the ESS they provide; the scope of which is being rigorously documented (e.g. Field, 2012; Nicholls, et al., 2010; Traill, et al., 2011; Epanchin-Niell, et al., 2015). Concerns around initial stages of degradation are only one aspect, indirect and consequential ramifications of manifested SLR-related impacts are another (Epanchin-Niell, et al., 2015). Second-stage effects stand to further increase the likelihood of hazard occurrence, lowering economic productivity, decreasing access to resources and escalating probability of risk for surrounding communities, as well as for communities who rely on wetlands for resource extraction/livelihoods (Arkema, et al., 2013). This is a dangerous aspect to overlook, as encroached upland marshes become increasingly salinized, a number of novel issues will arise.

As literature in SLR, climate-impact studies, EbA methods and related policy efforts become more nuanced, a number of opportunities for increased environmental and social resilience have become apparent (Leadley, et al., 2014; Fagherazzi, et al., 2019; Buckmaster, et al., 2014; Barbier, et al., 2008, 2011 and 2014; Jiang, et al., 2015). Understanding which areas are ripe for investigation and improvement is critical in the continuation of environmental degradation and risk analysis research. One such example can be observed in an overview of past research and protection efforts which aimed, largely at making structural modifications through top-down methods. Resulting engineering approaches left many of the dynamic socioeconomic and ecological systems overlooked, entirely (Ayers, et al., 2014). Moving forward, it is clear that contemporary and future efforts must incorporate more comprehensive, bottom-up perspectives that expose and address root causes as well as solutions and outcomes.

Epanchin-Niell, et al., (2015) and Hossain, et al., (2018) highlight the lack of cohesion amongst various coastal protection efforts and suggest solutions for a more holistic approach.

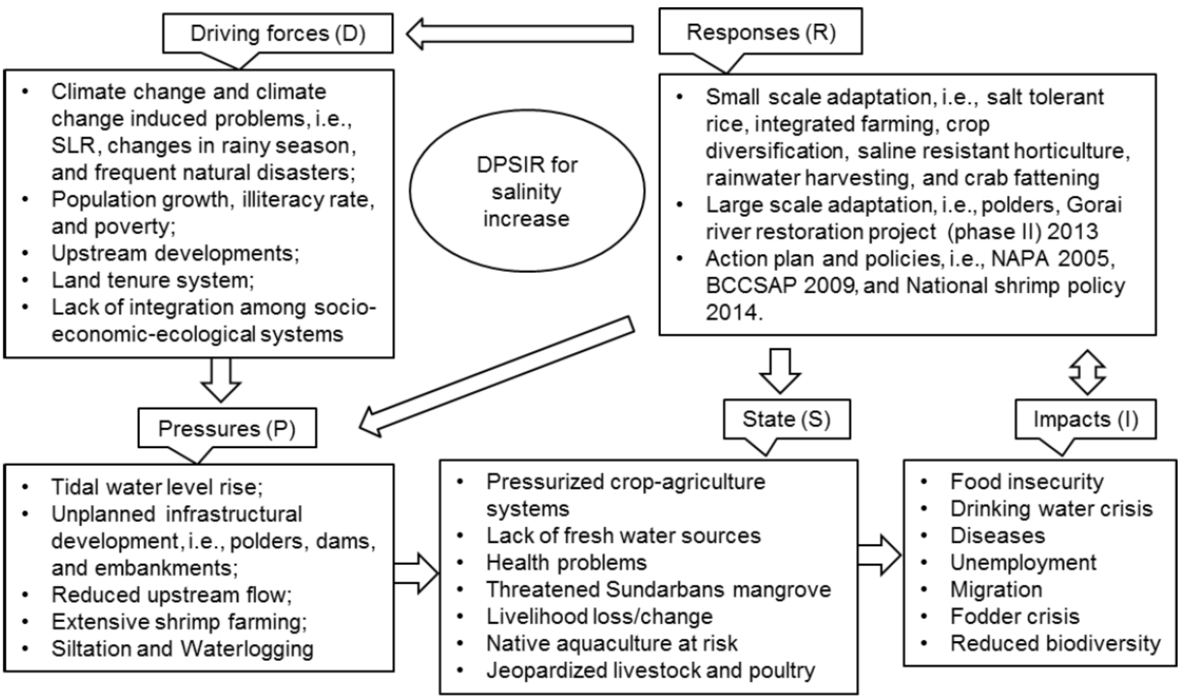

Figure 3: Suggested theoretical framework examining intersections and relations between coastal wetland environments and varying levels of society. Driving forces, Pressures, State, Impacts and Response (DPSIR), National Adaptation Programme of Action (NAPA), ad Bangladesh Climate Change Strategy and Action Plan (BCCSAP) Source: Hossain, et al., 2018.

The need for techniques that can surpass temporal and spatial scales, as well as remain relevant for different land cover types and interface each level of government, is by no means a modest endeavour. It does, however allow for a multidisciplinary synergy to develop amongst researchers and creates the type of pathways necessary for addressing SLR-induced risk throughout coastal wetland communities. Figure 3 demonstrates a model proposed by Hossain, et al., (2018) for the investigation of relation between society and environment; in an attempt to address the complexities present within and amongst different key-players. The Driver-Pressure-State-Impact-Response (DPSIR) model builds on the interconnectivity between social, ecological, governmental influences, points of critical failure, their impacts and subsequent response measures either suggested or implemented. The acknowledgment of socio-ecological driving forces (D) is a highly beneficial perspective and can be adapted to work with other scenarios such as SLR, flood risk and extreme hazard events. Such models contribute an effective framework from which further research can apply specific attributes to inform adaptation and response efforts. It should also be noted that this model includes stakeholder participation, namely local community members.

4.1 Case Study

The argument that EbA can, and perhaps should, as a rule, begin with environmental protection has been proposed by many, and is a narrative adopted by this paper as a means of evaluating potential SLR-induced effects on ESS and secondary hazards (Arkema, et al., 2013; Bamlford, et al., 2002; Barbier, et al., 2011 & 2014; Bell, 1997; Gedan, et al., 2011; Epanchin-Niell, et al., 2015). Regardless of land ownership, protected wetlands and the ESS they offer are nevertheless vulnerable to potential loss due to SLR; this includes but is not limited to wave and storm attenuation, erosion control, water purification, fisheries maintenance and carbon sequestration (Gedan, et al., 2011; Barbier, et al., 2011). Land use and ownership does, however, play a role in protection efforts and the securing of funds (Epanchin-Niell, 2015).

Recent studies have begun to confirm, by way of natural wetland preservation, coastal resilience can be strengthened; specifically, with regards to flood mitigation and storm surge attenuation (Barbier, et al., 2011; Epanchin-Niell, Glass, et al., 2015; 2015; Rezaie, et al., 2020; Wamsley, et al., 2009). Indirect values or services include enhancing flood reduction capacities, decreasing risk for surrounding populations and securing property values. Estimation of these values has been achieved with the use of a number of different types of valuation studies. Through the implementation of hydrodynamic models or flood and loss models, for example, researchers have been able to calculate flood protection service values (Barbier, et al., 2008). The findings of Rezaiel, et al. (2020) make use of an approach which utilizes the precision of numerical modeling and a holistic integration of natural variability. This was comprised of a coupled version of the Advanced Circulation model (ADCIRC) and Simulating Waves Nearshore model (SWAN); where storm surge modelling for both historical flood data and hypothetical/synthetic storms was examined. A third model, the Sea Level Affecting Marshes Model (SLAMM) was also applied specifically to integrate the potential variation in natural habitats as a direct result of SLR. The results point to a general reduction of 5.4% of marsh area (comprised of transitional, regularly and irregularly flooded salt marsh areas) where each land cover type experiences a slightly varying degree of change (Table 1). The results also show that 23.1% fewer residential parcels become inundated when natural habitats are intact. Flood depth estimations, measured in meters and percentage of change, relative to today's conditions is illustrated in Table 2. Finally, Table 3 documents the estimated property damages and estimated avoided damages due to protected habitat presence, in USD.

Land Cover Type (SLAMM)	Tidal Category	Current Scenario (km^2)	Future Scenario (km^2)	Change (%)
Developed Dry Land	Non-Tidal	1,030.95	1,029.92	-0.10%
Swamp	Freshwater Non-Tidal	375.56	375.50	-0.02%
Cypress Swamp	Freshwater Non-Tidal	0.01	0.01	0.00%
Inland Fresh Marsh	Freshwater Non-Tidal	15.37	15.34	-0.23%
Tidal Fresh Marsh	Freshwater Tidal	0.82	0.78	-4.80%
Transitional Salt Marsh	Transitional	3.20	2.00	-37.67%
Regularly-flooded Marsh	Saltmarsh	23.54	88.18	274.65%
Mangrove	Transitional	0.02	0.02	-0.59%
Estuarine Beach	Low Tidal	2.65	2.69	1.28%
Tidal Flat	Low Tidal	0.53	13.71	2,506.14%
Ocean Beach	Low Tidal	4.93	5.00	1.31%
Inland Open Water	Open Water	21.17	18.96	-10.46%
Riverine Tidal	Open Water	2.73	1.19	-56.36%
Estuarine Open Water	Open Water	290.91	294.66	1.29%
Open Ocean	Open Water	135.90	135.90	0.00%
Irregularly Flooded Marsh	Transitional	212.54	137.10	-35.49%
Inland Shore	Freshwater Non-Tidal	0.72	0.72	0.00%
Tidal Swamp	Freshwater Tidal	16.56	16.43	-0.76%

Table 1: (above) Categorized land cover types, simulated by the SLAMM model, calculated to demonstrate square km of land change due to SLR. Pathways of transition or change vary by land cover type; for example, irregularly flooded marshes and transitional salt marshes are expected to experience nearly 40% change, each and become regularly flooded salt marshes. Tidal flats, on the other hand are expected to increase significantly. These findings imply that SLR are causing wetlands to migrate inland (Source: Rezaiel, et al., 2020). Table 2: (below) Estimated monetary loss and avoided damages in USD- directly related to the presence and/or absence of preserved wetland habitats. Used in part to improve understanding per-unit-area benefits of preserved ecosystems, results show nearly 14% total property value reduction under current conditions and 6.1% of future scenarios.

	Property Damage (Habitat Present)	Property Damage (Habitat Absent)	Avoided Damage (Flood Protection Value)	Percent Change (From Habitat Present)
		Current Scenario		
25-year storm	$82,062,657	$91,894,099	$9,831,442	11.98%
50-year storm	$94,888,388	$107,972,822	$13,084,434	13.79%
Sandy	$2,322,731,031	$2,331,067,963	$8,336,932	0.36%
		2050 SLR Scenario		
25-year storm	$125,436,468	$126,980,226	$1,543,758	1.23%
50-year storm	$329,190,819	$349,122,514	$19,931,695	6.05%
Sandy	$2,562,559,835	$2,594,648,892	$32,089,057	1.25%

4.2 Evaluation

Research themes present in WES research including ES valuation, driving force analysis, policy & management, review & opinion, trade-off analysis, modelling methods and, 'others' (Xu, et al., 2020). These strategies offer a wide range of opportunities for researchers, policy makers and stakeholders to become involved in understanding the value of protecting coastal wetlands as an EbA technique. Established research provides a structure from which future studies can operate, monetary and change percentages as a way of communicating the severity of successful or unsuccessful efforts. Inclusion of socio-ecological frameworks will also be essential, moving forward, in order to ensure a comprehensive context is maintained. As research continues, SLR and related impacts are also at work and it is imperative that the continuous observation and documentation of manifested changes are taken into account. As pointed out by Rezaiel, et al. (2020), “it is important to develop ecosystem-based flood protection approaches that will protect [communities] and protective ecosystems.” (Rezaiel, et al., 2020).

5. Limitations

There are biophysical and socioeconomic limits to the adaptation potential of coastal wetlands in light of SLR, and both will be discussed here. Analysis of historical records has revealed that coastal ecosystems have responded well to past SLR with a combination of inland and vertical (i.e. ‘upward’ migration of the wetland’s biota by means of sediment accretion) movement on the landscape (Enwright *et al*., 2016). Schuerch *et al*. (2018) conservatively estimate that “wetland gains [i.e. expansion of wetland area] of up to 60% (...) are possible, if more than 37% (...) of coastal wetlands have sufficient accommodation space, and sediment supply remains at present levels”. If the wetlands are provided with enough physical space to move horizontally or vertically in response to SLR, they are able to fulfil their adaptation potential as well as provide the ecosystem services that coastal populations can benefit from (Schuerch *et al*., 2018). Sufficient accommodation space can therefore be identified as the first - and perhaps primary - biophysical limiting factor to the risk reduction potential of coastal wetlands in light of projected SLR. The required accommodation space for vertical and/or inland movement is highly site- and wetland type-specific and the potential for movement itself is constrained by certain biophysical factors.

As mentioned by Schuerch *et al.* (2018), sediment supply is a key factor in determining the success of vertical accretion of coastal wetlands. Kirwan *et al.* (2013) note that past response to SLR is an “imperfect model” since human activity is continually changing sediment delivery rates, as well as climate and water quality. The ability of coastal wetlands to withstand SLR, by means of vertical accretion, is highly dependent on sediment ability. So much so, that some argue that the threshold rate of SLR in terms of marsh accretion is a function of sediment availability (Kirwan *et al*., 2013). While not necessary for lateral movement, sediment is the key driver of vertical accretion and its unavailability along many coastal wetlands is largely due to anthropogenic influence, such as dams, which prevent 20% of global sediment load from reaching the coasts (Kirwan *et al*., 2013). Sediment availability should be considered another key biophysical limiting factor to the risk reduction potential of coastal wetlands in light of projected

SLR. A lack of sediment supply will in turn lead coastal wetland to respond to SLR with increased inland migration (Schuerch *et al*., 2018). Vertical accretion is further constrained by the rate of SLR itself, as coastal wetlands “must build soil elevation at a rate faster than or equal to the rate of sea-level rise to survive in place” (Kirwan *et al*., 2013). Initially, wetlands will respond to SLR with vertical accretion at increasingly fast rates, using up more and more sediment, until they reach their peak productivity. If SL continues to rise after this point, they drown, as shown in Figure 4. While the accretion process can happen rapidly due to sophisticated feedback loops, once the wetland has ‘peaked’ its inundation will also happen rapidly, due to fast-acting feedbacks (Kirwan *et al.*, 2013). The SL at which a coastal wetland may drown is highly site-specific since it depends on vegetation type, sediment availability, and human activity (Kirwan *et al.*, 2013). Mangroves, for example, are able to withstand more SLR than saltmarshes (Seddon *et al*., 2020). Evidently, the factors that limit vertical accretion potential are highly interconnected and must be investigated holistically.

The strategy of inland migration, should vertical movement not be possible, is also difficult in many coastal wetlands globally, since the physical space needed for this strategy to work is becoming increasingly constrained due to anthropogenic structures built along coastlines, as well as topography (Enwright *et al*., 2016). This includes buildings, but also - paradoxically - flood-protection infrastructure. Careful spatial analysis can determine ‘migration corridors’ that are unobstructed and marked by low-lying topography that allows for inland migration of a coastal wetland. These would have to be designated conservation lands and, if applicable, cleared of all man-made structures, with regulation put in place to minimize urban expansion into these areas. The maintenance and protection of these corridors is expensive, but their benefits, to urban areas, in particular, may outweigh the cost (Enwright *et al*., 2016). Inland migration is not always possible, especially where the land is too anthropogenically modified or the geomorphological characteristics constrict movement (Schuerch *et al*., 2018).

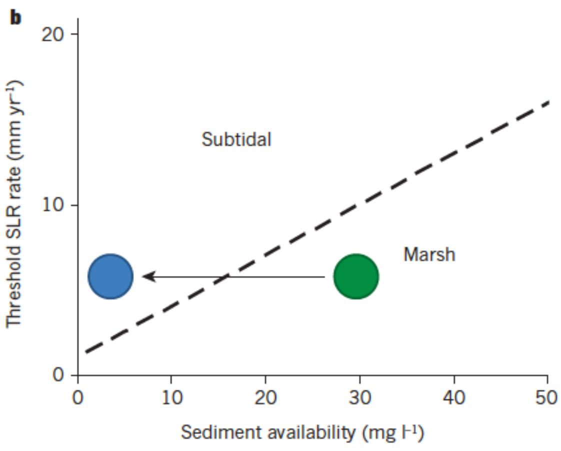

Figure 4: The threshold rates of SLR beyond which a marsh (green) drowns (blue) as a function of sediment availability. Source: Kirwan et al., 2013.

On this note, another limitation to the success of inland migration must be considered. The landward movement of coastal ecosystems in response to SLR can often lead to salinity intrusion in existing inland freshwater ecosystems, involving the replacement of local biota with halophytic organisms and an increase in soil and groundwater salinity (Enwright *et al.*, 2016; Fagherazzi *et al*., 2019). Saltwater intrusion can cause the loss of farmland and forests, or at least a replacement of halophobic tree species, as well as salinization of fresh groundwater resources, which again affects farming as well as water security. This 'knock-on effect' is often overlooked when examining the natural response of coastal wetlands to SLR (Oppenheimer *et al*., 2019). This is especially true when the rather sudden inward movement of a saline coastal wetland may endanger an ecosystem inland that also provides valuable ESS. In this situation, a careful cost-benefit analysis needs to be conducted that weighs the consequences of allowing the coastal wetland to move inland.

As mentioned in 3.2, stakeholder attitudes towards EbA measures can constitute social barriers to adaptation, and even be "harmful" to wetland protection (Oppenheimer *et al*., 2019; Xie *et al*., 2010). This is a socioeconomic limiting factor that often remains under-investigated and overlooked, despite its potential to lead to the failure of an EbA measure (Oppenheimer *et al*., 2019). Nalau *et al.* (2018) find that "stakeholders might hold negative perceptions about particular types of EbA strategies". These perceptions can be affected by worldviews, risk perceptions, and sense of place (Nalau *et al*., 2018). Stakeholder attitudes are also often negatively affected if the cost of EbA measures is perceived to be too high, which will in turn influence the decision of whether to implement the measure at all. Additionally, land ownership can arise as an issue, where stakeholders refuse to participate in EbA measures on their land, due to economic concerns (Nalau *et al*., 2018).

While biophysical limits to EbA strategies may be reached in the 21st century, socioeconomic ones will arise "well before the end of the century" due to societies being "neither economically able nor socially willing to invest in coastal protection" (Oppenheimer *et al*., 2019). The main 'selling point' of EbA measures is their cost-effectiveness compared to technocratic approaches. There are prevailing issues with assessing this cost-effectiveness, leading to a critical lack of investment (Seddon *et al*., 2020). Especially where governance structures are rigid, engineered interventions will often still be preferred (Seddon *et al*, 2020). This constitutes institutional and governmental constraints on the effectiveness of EbA, which is reflected in a lack of coping capacity, potential corruption, and miscommunication between stakeholders (Nalau *et al*., 2018). These issues are again connected to the socioeconomic factors of risk perception and perceived cost-effectiveness. The danger of underestimating socioeconomic limits to EbA becomes clear.

5.1 Trade-Offs

Arguably, this is where the concept of 'trade-offs' comes to play. Not all ESS can always be guaranteed to the same extent at all times, and the cultural services (e.g. aesthetic enjoyment of "the beach") will have priority for many stakeholders who are not directly confronted with the need for adaptation to SLR (O'Donnell, 2019). This would be an example of a trade-off between the cultural and regulating services. Alternatively, the regulating services can shift into the focus of an EbA strategy aimed at wetland conservation for protection from EFEs, which can see supporting services - and thus biodiversity and ecosystem health - being levied against strengthened regulating potential (Morris *et al*., 2018). EbA strategies aimed at responding to SLR raise a number of trade-offs between economic development, safety, and conservation, but also between public and private interests, as well as short- and long-term goals (Oppenheimer *et al*., 2019). Any successful EbA assessment should therefore consider and carefully negotiate the trade-offs between the different ESS, and model which services are most crucial to maintaining the ecosystem as well as the benefits it provides. The occurrence of trade-offs is inevitable, and the consequence of neglecting one aspect of the ecosystem can lead to a failure of the EbA strategy or maladaptation (Seddon *et al*., 2020).

It goes without saying that any EbA measure should have the goal of minimizing trade-offs while maximizing ESS benefits for all. But it must be noted that "all measures have their limits" and not all coastal wetlands can be feasibly protected in the light of global SLR, which will not only force local populations to eventually retreat but also cause a tremendous loss to global ecosystem biodiversity (Oppenheimer *et al*., 2019). Lastly, many coastal wetlands that human populations rely on for protection are already degraded or anthropogenically altered to some extent, which sets a lower baseline for potential EbA approaches (Nalau e*t al*., 2018).

5.2 Case Study

The mangrove reforestation project along the coast of Gujarat, India, provides a good example of the limits of the risk reduction potential of coastal wetlands in light of projected SLR. In an effort to stabilize coasts, a government initiative was started in the 1990s, which has been successful in more than doubling the coastal mangrove cover since then - from 876.36 km^2 in 1990 to 1693.88 km^2 in 2013 (Das, 2020). A recent geospatial analysis of the plantation area conducted by Das (2020), however, found that the mangroves have not been able to meet their EbA potential everywhere along Gujarat's coast. For this analysis, the area was broken into 28 plots, 11 of which saw a net decrease in mangrove cover from 1990 to 2013. Of these, 4 also showed net sediment loss (i.e. erosion). Overall, it was found that mangroves had an insignificant effect on erosion rates in Gujarat state, but a net increase in mangrove cover was shown to have a strong relationship with net accretion rates (Das, 2020). Nevertheless, these results show that the EbA measure was not fully successful in Gujarat. In fact, the mangroves "seem to be either not surviving if wave actions are strong or aggravating erosion to some extent if they survive" (Das, 2020). Das (2020) hypothesizes that the planting of mangroves in discontinuous patches along the coast, instead of as one connected mangrove forest, may be to blame as it enables erosion in between the forested patches of land. South Gujarat has the area's healthiest and most diverse mangrove population, and this is reflected in the net accretion rates observed for this area (Das, 2020).

This example shows the importance of implementing site-specific EbA measures to fully realize the potential of coastal wetlands to mitigate coastal risk. As shown in section 5, the factors that limit the vertical and inland movement of coastal wetlands are spatially heterogeneous and a 'one size fits all' approach simply is not feasible if effective protection is to be achieved. While the mangrove plantations in Gujarat were conceived with an EbA mindset, the implementation was not fully thought through, as evident from the apparent failure - and even drowning - of the mangroves along some areas of the coast.

6. Recommendations and Next Steps

To optimize possible EbA measures, the prevailing literature recommends a number of steps. In general, there is a consensus that more research is needed to assess the potential of ecosystem-based approaches, and that these are best developed via an interdisciplinary approach (e.g. Das, 2020; Fagherazzi *et al*., 2019; Oppenheimer *et al.*, 2019). Importantly, the spatial variability in SLR and the consequences for wetlands has to be considered (Fagherazzi *et al.*, 2019; Oppenheimer *et al*., 2019). There is a need to strike the right balance between paying attention to site-specific characteristics while also upscaling successful strategies (Schuerch *et al*., 2019).

Kirwan *et al.* (2013) suggest conducting large scale spatial analyses to infer where wetland movement is possible and where it is obstructed. This can be critiqued as a 'top-down' approach, but the aim of it is to improve the global database of coastal wetlands. Once this spatial analysis is performed, individual EbA strategies can be developed in a site- and context-specific manner. It has become clear that there cannot be an 'umbrella' approach for EbA strategies, as their success depends on how well they can be integrated into the physical, but also societal, landscape. Therefore, socioeconomic and biophysical factors need to both be taken into account for the strategy to have a higher chance at success - hence the call for more interdisciplinarity. In this sense, Kirwan *et al*. (2013) also stipulated that "new socio-economic research examining perceptions of wetland value is needed to fully understand coastal sustainability". The engagement of stakeholders, often via qualitative methodologies, is a crucial step towards a more widespread societal acceptance of EbA measures. After all, there is still the need for a more complete paradigm shift to embracing EbA and other nature-based solutions (Seddon *et al*., 2020).

As previously mentioned, the breadth of research which, to date, has allowed for promising EbA measures to enter implemental stages and suggest further recommendations should be acknowledged (SOURCES). Through the integration of multidisciplinary perspectives, many, if not all branches of society stand to participate as well as benefit from the efforts currently underway. The techniques outlined in this paper offer a method for stakeholder participation which is adaptable to a wide variety of specific community and wetland types. As suggested, the importance of a comprehensive EbA measure cannot be overstated and is, perhaps the greatest asset EbA and environmental degradation/conservation studies can build on to strengthen climate adaptation worldwide.

With global SLR posing a significant threat to the world's coasts and thus the coastal populations residing on them, there is a pressing need for solutions. Ecosystem-based approaches, that maximize benefits and minimize trade-offs, offer not only a cost-effective but also an environmentally friendly alternative to investing solely in artificial defences (Ferrario *et al.*, 2013). However, since nature-based approaches are an emerging field still, Fagherazzi *et al*. (2019) also rightfully call for more research on the consequences of inland migration of coastal wetlands. A good understanding of knock-on effects, which can be an intended or unintended consequence of an EbA strategy, needs to be part of the monitoring and evaluation process. To fully realize their potential, however, the EbA measures must be well-planned, implemented with care, and developed with a site-specific focus.

7. Conclusion

This paper has shown that strengthening coastal wetlands with EbA measures opens up many opportunities to reduce coastal risk, especially in light of SLR. Evidently, both quantitative and qualitative methodologies play a key role in the assessment of these opportunities, but also of the biophysical and socio-economic limitations that EbA measures may encounter. There is a critical lack of incorporation of qualitative assessments of coastal wetlands and coastal risk, as well as a notable inconsistency among quantitative methodologies.

Additionally, coastal wetlands are often already in a degraded state due to anthropogenic influence, which reduces their adaptation potential significantly (Nalau *et al*., 2018). Coupled with the fact that current rates of degradation will also impact future the base point from which hazards affect wetlands, the necessity of improved valuation estimation modeling, risk analysis and disaster preparedness is clear. Nevertheless, coastal wetlands are naturally resilient to SLR and can play a key role in protecting coastal communities and ecosystems from flooding, EFEs, TCs etc. The threshold beyond which a coastal wetland is unable to adapt to SLR needs to be well-understood in order to develop targeted and efficient adaptation strategies (e.g. Cooper & Lemckert, 2012).

Coastal wetlands offer a cost-effective, sustainable, and efficient way to protect low-lying areas from SLR and the associated risk and yet the development of EbA strategies to protect and maintain is often not seen as a priority. A shift in risk perceptions, but also a more holistic approach to risk management are arguably needed to fully utilize the opportunities coastal wetlands have to offer, while operating within their specific limitations.

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