Restore or retreat? Saltwater intrusion and water management in coastal wetlands

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Abstract.   Coastal wetlands perform a unique set of physical, chemical, and biological functions, which provide billions of dollars of ecosystem services annually. These wetlands also face myriad environmental and anthropogenic pressures, which threaten their ecological condition and undermine their capacity to provide these services. Coastal wetlands have adapted to a dynamic range of natural disturbances over recent millennia, but face growing pressures from human population growth and coastal development. These anthropogenic pressures are driving saltwater intrusion (SWI) in many coastal systems. The position of coastal wetlands at the terrestrial–marine interface also makes them vulnerable to increasing rates of sea-level rise and changing climate. Critically, anthropogenic and natural stressors to coastal wetlands can act synergistically to create negative, and sometimes catastrophic, consequences for both human and natural systems. This review focused on the drivers and impacts of SWI in coastal wetlands and has two goals: (1) to synthesize understanding of coastal wetland change driven by SWI and (2) to review approaches for improved water management to mitigate SWI in impacted systems. While we frame this review as a choice between restoration and retreat, we acknowledge that choices about coastal wetland management are context-specific and may be confounded by competing management goals. In this setting, the choice between restoration and retreat can be prioritized by identifying where the greatest return in ecosystem services can be achieved relative to restoration dollars invested. We conclude that restoration and proactive water management is feasible in many impacted systems.

Key words: *climate change; disturbance; ecosystems; restoration; salinity; saltwater intrusion; watershed management; wetlands.*

Introduction

Environmental stressors impacting coastal wetlands have both natural and anthropogenic sources. Coastal zones are dynamic and subject to changing environmental conditions caused by natural variations in climatic, oceanographic, and ecological processes such as flooding, drought, long-term climate cycles, storm surges, hurricanes, winds, herbivory, and changes in sea level. At the same time, increasing human population and development in coastal areas have increased point source and non-point source pollution (Sampathkumar 2015), habitat destruction (Hefner and Brown 1984, Li et al. 2014), and hydrological alteration (Niemi et al. 2004). Moreover, due to their position in the landscape, coastal wetlands are vulnerable to both sea-level rise (SLR) and changing temperature/rainfall regimes driven by climate change. In human-dominated landscapes, these natural and anthropogenic stressors frequently overlap with synergistic, and sometimes catastrophic, consequences for both natural and human systems. This phenomenon is perhaps best evidenced by the environmental, economic, and social damage caused by Hurricane Katrina as it passed through the severely degraded wetlands of the Mississippi River Delta (MRD; Laska and Morrow 2006, Petterson et al. 2006, Deryugina et al. 2014).

Considering these multiple, interactive stressors, a central question for land managers is how to proactively manage and/or restore coastal wetlands in a future of changing climate, land use, and shoreline modification. Here, we review our current understanding of the state and projected trajectory of coastal wetland change, with a focus on the potential for improved water management to mitigate the worst impacts of saltwater intrusion (SWI) and best prepare coastal wetlands to adapt. First, we review how natural and anthropogenic SWI drivers alter hydrology and water quality in coastal wetlands and describe the subsequent ecological response. Next, we provide a framework for assessing the potential for restoration interventions in systems degraded by SWI and summarize approaches for their restoration.

Drivers of Saltwater Intrusion in Coastal Wetlands

Among the numerous causes of coastal change, SWI is a primary stressor in many coastal wetland systems. SWI refers to landward and/or upward displacement of the freshwater–saltwater interface in coastal aquifers (Knighton et al. 1991) and increased saltwater penetration in estuaries (Barlow 2003). SWI drivers can be natural, anthropogenic, or synergistic. Natural drivers of SWI include storm surges, hurricanes, climatic fluctuations, SLR, and subsidence (Fig. 1). Anthropogenic SWI drivers include land drainage, pumping of coastal freshwater aquifers, reduction in freshwater discharge from dam construction, water withdrawals, or other water diversions, and hydrological/hydraulic structures and land-use changes within watersheds (Fig. 2). While SLR is a natural driver of SWI, anthropogenic climate change exacerbates SLR effects and may also alter the timing and magnitude of climatic cycles and extreme events; these effects are addressed in section *SWI and anthropogenic climate change*.

Natural drivers of SWI

Natural drivers of SWI occur over a gradient of timescales and frequencies (Fig. 1). Chronic and acute drivers of SWI both act to shape the coastal environmental and dictate the direction and rates of change in coastal wetland ecosystems. Chronic drivers of SWI include SLR and geologic uplift and subsidence. These drivers act over millennial timescales, although changes in their relative rates can drive large-scale changes in coastal wetland structure and function (DeLaune and White 2012). Acute drivers of SWI include tsunamis, hurricanes, droughts, and climate oscillations. Importantly, acute
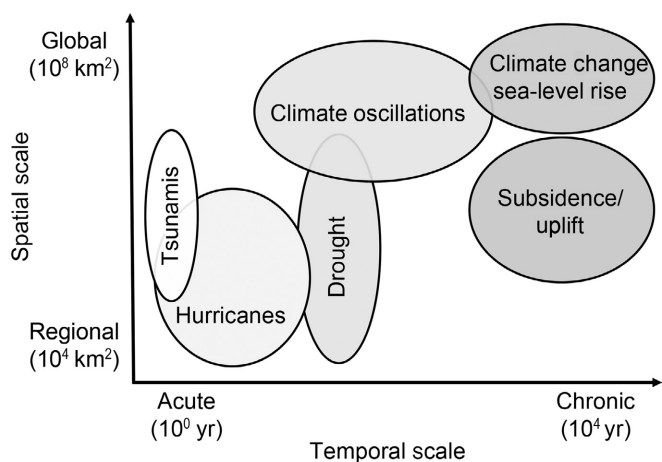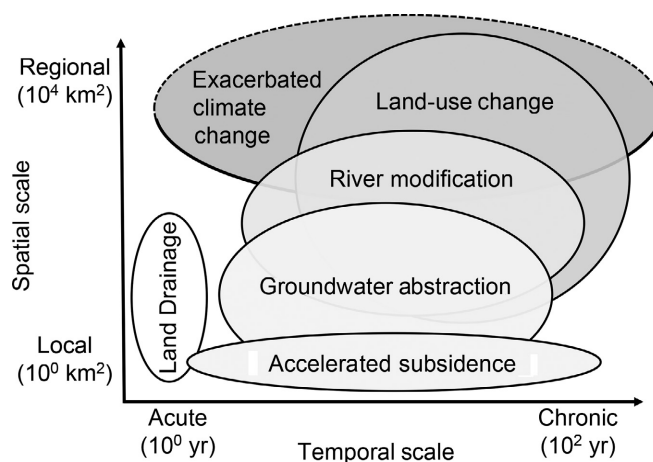

Fig. 2. Schematic representation of the spatial and temporal scales of anthropogenic saltwater intrusion (SWI) drivers. Many anthropogenic drivers act over a wide variety of spatiotemporal scales as a function of development intensity (e.g., local agricultural water abstraction in a small river vs. damming a large river). Note reduction in axis scaling from Fig. 1, indicating that anthropogenic drivers of SWI are generally limited to centennial timescales and regional spatial scales. One exception is anthropogenically exacerbated climate change, which can drive larger spatiotemporal processes (indicated by dashed upper border).

drivers may elicit both short-term and long-term effects on coastal swamps and marshes that can be positive (Baustian and Mendelssohn 2015) or negative (Brisson et al. 2014, Leonardi et al. 2016).

At the shortest timescales, tsunamis can act as physically destructive forces on the coast and carry saline water far inland. For example, the 2011 Tohoku-oki tsunami carried a large load of sediment, nutrient, and salinity 5 km inland (Chagué-Goff et al. 2012). Affected rice paddies and other freshwater wetlands became saline or brackish and remained at highly elevated salinity levels for up to 5 months. Even after surface water salinity decreased, salt concentrations in the soils remained elevated in some areas for the remainder of 2011 (Chagué-Goff et al. 2012). The Indian Ocean tsunami of 2004 illustrated the physically destructive potential of these events on coastal wetlands at a large spatial scale. An assessment by Indonesia's State Ministry of National Development Planning estimated that 25%–35% of coastal wetlands had been destroyed in tsunami-affected areas (Srinivas and Nakagawa 2008).

Like tsunamis, hurricanes are stochastic events that can transport saline water, sediments, and bound nutrients far inland (Rejmánek et al. 1988). Hurricanes have both short-term physical impacts (winds, wave, and surge in the case of hurricanes) and longer-term impacts from subsequent flooding with saline waters. In many regions, coastal salt marshes or mangroves ring the continents and can serve to weaken hurricane wind and wave damage and the associated storm surge (Moller et al. 2014), greatly reducing the potential for damage to

Fig. 1. Schematic representation of the spatial and temporal scales of natural saltwater intrusion (SWI) drivers. Temporal scale varies from fast-acting and low frequency (i.e., acute) to slow-acting and consistent (i.e., chronic). Spatial scale indicates the *size* of the area impacted by SWI.

the coastline and inland freshwater wetlands. However, coastal marshes and mangroves are being lost, decreasing the storm surge attenuation potential and increasing the vulnerability of adjacent freshwater wetlands (Moller et al. 2014).

The magnitude and duration of hurricane surge-induced SWI is strongly driven by hurricane intensity, wind direction, tide, and local hydrological conditions. For example, salinity increased from 4 to 15 parts per thousand (PPT) in a Louisiana brackish coastal marsh during Hurricane Rita in 2005 and remained elevated for more than 3 months. Salinity in the same marsh was unchanged during Hurricane Katrina due to differences in wind direction (Steyer et al. 2007). Vegetation in freshwater and brackish wetlands is not adapted to the high-salinity water-associated hurricane surge and often dies back completely, leaving bare soil that is vulnerable to erosion. After inundation, wetland recovery depends on several factors including post-intrusion salinity and flooding level (Flynn et al. 1995). If saline water is trapped inland by levees or roads, recovery can be extremely slow. Also vital are the location, connectivity, and type of propagules to re-populate the wetland if significant plant mortality occurred (McKee and Mendelssohn 1989). Recent research into community re-assembly after hurricane storm surge points to additional factors, including species-specific reproductive strategies (Middleton 2009), sediment and wrack deposition (Tate and Battaglia 2013), and stochastic community assembly (Guo et al. 2014).

Droughts and yearly-to-decadal climate oscillations are longer-term drivers of SWI (Fig. 1), which can play a significant role in the long-term survival of coastal wetlands. During periods of drought, brackish and freshwater wetlands face stress from reductions in rainfall and freshwater flows with subsequent influences on SWI in surface water and groundwater (Kaplan et al. 2010*a, b*, Kaplan and Muñoz-Carpena 2014). Without relief from saltwater stress, vegetation dies back, eventually converting to salt marsh or open water (Drexler and Ewell 2001). Droughts can have particularly strong effects on coastal wetlands when coupled with SLR-induced SWI, leading to rapid declines in both species richness and regeneration (Desantis et al. 2007). In addition to vegetation effects, drought-induced changes in biogeochemical cycling can cause coastal wetlands to release nitrogen (Scott et al. 2003), with potential effects on marsh productivity, accretion, and coastal eutrophication.

Climate oscillations affect coastal wetlands through impacts to a variety of abiotic factors. For example, the North Atlantic Oscillation (NAO) and Atlantic Multidecadal Oscillation (AMO) affect hurricane frequency and tracks. In the eastern United States, the NAO influences whether hurricanes enter the Gulf of Mexico or travel up the Atlantic Coast (Ting et al. 2009), and the AMO alters sea surface temperatures (SSTs) in the Atlantic Ocean, which also affect hurricane routes

(Knight et al. 2006, Wyatt et al. 2012). AMO, NAO, and El Niño/Southern Oscillation (ENSO) all influence the magnitude and distribution of precipitation and temperature anomalies, impacting coastal and marine ecosystems by changing the coastal freshwater delivery (Karamperidou et al. 2013, Nye et al. 2014). Dry phases of these climate cycles represent long-term droughts, which can cause persistent increases in coastal wetland salinity, driving vegetation mortality, and reorganization (Drexler and Ewell 2001, Tolan 2007, Angelini et al. 2016).

At the longest timescales, chronic stressors include SLR and subsidence, which act over hundreds or thousands of years (Fig. 1). SLR decreases the relative elevation of coastal wetlands and subsidence decreases their absolute elevation. Both drivers act at a slow, but consistent pace, so that the elevation of coastal wetlands is relatively well poised to "keep up" via accretion of organic matter and trapping of inorganic sediments (Morris et al. 2002). Sea level has risen throughout the current interglacial period (USGS 2000) and particularly during the mid- to late Holocene era, though not always at the same rate (Wanless 1989). For example, global sea level rose quickly (2–5 mm/yr) between 3,200 and 6,500 yr before the present, but slowed to approximately 0.4 mm/yr over the next 3,200 yr, allowing for the development of broad coastal wetlands around the continents. More recently, the rate of SLR has increased rapidly to between 2.8 and 3.6 mm/yr (Church et al. 2013). While there are uncertainties about the future rate of ice sheet collapse and glacial melting (Nicholls and Cazenave 2010), anthropogenic climate change is projected to rapidly accelerate SLR over rates observed in recent millennia. The ability of coastal wetland ecosystems to keep up with this acceleration is spatially variable and largely uncertain (see section *SWI and anthropogenic climate change*).

Along with SLR, compaction, subsidence, and uplift are natural processes acting over long timescales in coastal wetlands (Fig. 1). Coastal wetlands are built from surface layers of largely uncompacted peat. Over time, peat at lower depths is compacted and dewatered by the weight of the overlying soil and water (Morton et al. 2002), leading to reductions in elevation. Without accretion of organic and mineral sediments, coastal wetlands become inundated, experiencing SWI in much the same way that is expected with SLR (DeLaune and White 2012). It is estimated that compaction and tectonic-driven subsidence in the MRD account for 80% of relative SLR in that region (Dokka 2006), though this problem is likely exacerbated by anthropogenic activities (Penland and Ramsey 1990, see section *SWI and environmental modification*). Uplift and subsidence are larger-scale phenomena that typically occur in relation to geologic and tectonic activity (Lambeck et al. 2014), and in coastal regions are driven by isostatic adjustment following glaciation, deglaciation, and/or mountain formation. In North America, isostatic rebound following the retreat of the Wisconsin ice sheet is estimated to be from −3 to

3 mm/yr along the U.S. eastern seaboard. In Asia, uplift of the Himalayas is causing subsidence of the Bengal basin, accelerating SWI in that region (Alam 1996).

Anthropogenic drivers of SWI

In contrast to natural drivers, many of the current SWI-driven threats to coastal wetland sustainability have anthropogenic (and/or synergistic) origins (Fig. 2). We identify two primary subcategories of anthropogenic SWI drivers, namely anthropogenic climate change and environmental modification.

SWI and anthropogenic climate change

Coastal wetlands are uniquely vulnerable to landscape-scale ecological change driven by global climate change (Burkett and Kusler 2000, Day et al. 2008). Climate change affects coastal wetlands through changes in the frequency, magnitude, and duration of acute events (e.g., hurricanes, droughts, climate oscillations) and via changes in the rates of chronic stressors such as SLR, all of which play a role in SWI (section *Natural drivers of SWI*). Climate change thus exacerbates natural SWI drivers, yielding accelerated or more extensive effects (Fig. 2).

Climate change is expected to increase hurricane intensity largely due to increased SST (Mendelsohn et al. 2012). This expectation is supported by paleoclimate data from the past 200 yr (Donnelly et al. 2015) and observations over the past 30 yr (Knutson et al. 2010), which show increased hurricane activity and intensity during periods of warm SST. Given current limitations in our understanding of the processes that drive hurricane frequency and intensity (Zwiers et al. 2013), considerable debate remains on how these extreme events will change in the future. Despite this limitation, any increases in hurricane intensity can be expected to produce larger storm surges with concomitant increases in coastal wetland damage from scour, erosion, soil compaction, vegetation burial, and SWI (Lin et al. 2012, Thomas et al. 2015). What is unknown is whether coastal systems, which have adapted to survive periodic disturbance, can survive a future with either more frequent or more severe storms (Michener et al. 1997, Day et al. 2008, Knutson et al. 2010, Leonardi et al. 2016).

Like hurricanes, droughts are expected to increase in intensity as regional temperature and precipitation regimes shift with changing global climate (Trenberth et al. 2014). Generally, these changes will be seen as increasing regional temperatures and increased probability of extreme temperature events. Changes in precipitation regime are less certain and more regionally variable. In general, wet areas are expected to get wetter and dry areas drier (International Panel on Climate Change [IPCC] 2013). As an example, under Representative Concentration Pathway (RCP) 8.5, which represents a future with the highest greenhouse gas emissions, the U.S. Atlantic and Gulf Coasts will see an increase in maximum and minimum temperatures, mean and maximum 1-day precipitation, and the number of consecutive dry days (Wuebbles et al. 2014). Globally, these types of climate projections promise increasing frequency and severity of droughts. For coastal wetlands, less available freshwater may be available to mitigate SWI. While the impacts of anthropogenic climate change on yearly-to-decadal climate oscillations (NAO, AMO, ENSO) are uncertain (Collins et al. 2010, Knutson et al. 2010), climate change is expected to increase the magnitude of these climate cycles (Cai et al. 2014, Mann et al. 2014), meaning that coastal wetlands will likely face larger and more frequent extremes of wet and dry periods.

Most pressingly, current and accelerating rates of SLR exceed those observed in recent millennia. Sea-level rise may eliminate as much as 22% of the world's coastal wetlands by 2100 (Nicholls et al. 1999), though regional impacts would vary (Michener et al. 1997). Under the "best-case" emissions scenario (RCP 2.6), sea level is expected to rise by 0.4–0.6 m by 2100 AD. In contrast, the unmitigated warming scenario (RCP 8.5) has temperature rising by 4.5°C, yielding 0.7–1.2 m of SLR by 2100 (Horton et al. 2014). Either outcome is likely to drive large-scale disturbance and reorganization in many coastal wetlands. It is important to note that SLR rates and projections by IPCC (2013) and Horton et al. (2014) are based on global averages. Other factors such as subsidence, uplift, and accretion rate modify the local rate of SLR in a specific region (Morton et al. 2002). There needs to be better understanding of these local variables to improve early detection of accelerated SLR and enhance wetland survival and resilience at the local level (Haigh et al. 2014).

A primary determinant of how SLR will restructure different coastal wetland communities is accretion rate (Morris et al. 2002), which varies across ecosystem types, latitude, tidal range, suspended sediment concentrations, and local hydrodynamic setting (Morris et al. 2002, Cahoon 2007). While a complete review of these factors is beyond the scope of this work, we briefly summarize coastal wetland accretion rates and how they are expected to respond to accelerated rates of SLR. Tidal freshwater forests accrete 1.3–2.2 mm of soil per year (Craft 2012), less than current SLR rates of 2.8–3.6 mm/yr (IPCC 2013). These forests face the threat of conversion to open water, freshwater marsh, or brackish marsh. Coastal freshwater and brackish marsh both have accretion rates that are higher than the current rate of SLR (Craft 2012). How accretion rates in these systems will respond to increased flooding duration and salinity remains uncertain, but they are expected to be more resilient than tidal forests due to increased carbon sequestration and mineral sediment accumulation (Craft 2012).

The review by Kirwan and Megonigal (2013) seeks to identify threshold levels of SLR beyond which tidal wetlands transition to open water. These authors point to important biophysical feedbacks (e.g., sediment availability, tidal range, marsh productivity, SLR, and

accretion rates) and anthropogenic factors that dictate this threshold rate. Importantly, the ability of salt marsh to maintain its elevation in the face of accelerated SLR is largely dependent on upstream flows of freshwater, nutrients, and sediments. With changing climate and other anthropogenic changes to watersheds, these flows cannot be assumed to be stationary. Some coastal wetlands will be able to keep pace with SLR for 50–70 yr, but will eventually transition to other wetland types or open water. Critically, this type of analysis does not account for added stress from anthropogenic SWI, which likely shortens the survival window in many systems (Thorne et al. 2015).

Finally, anthropogenically exacerbated climate-driven drivers of SWI are often interconnected, which can amplify their effects. For example, more intense hurricanes bring more powerful storm surges, which when coupled to accelerated SLR and local subsidence can drive inundation far more than either driver alone (Yang et al. 2014). Similarly, more frequent droughts coupled with accelerated SLR can push coastal systems beyond their capacity to rebound (Desantis et al. 2007, Angelini et al. 2016), illustrating the perilous situation for coastal wetlands in regions where climate models predict that drought will become more frequent and severe (Karl 2009).

SWI and environmental modification

Beyond impacts to the global climate, humans have also directly modified the coastal environment, with widespread impacts on SWI and coastal wetlands. These anthropogenic modifications act synergistically with the natural and climate change drivers discussed above and include river modification, groundwater abstraction, land drainage for agriculture and urban development, subsidence due to extraction of underground resources, and land-use change (Fig. 2). In many cases, several of these drivers work simultaneously.

River modification causes SWI via construction of canals, changes in bathymetry, and dredging. Canals are used as shipping routes to major rivers and for oil and gas exploration, but also facilitate SWI. For example, simulation modeling of canals in coastal Louisiana showed that, under similar environmental conditions, the 5 and 10 PPT isohalines moved approximately twice as far inland when canals were widened and deepened (Wang 1988). To maintain river widths and depths required to accommodate ever-increasing vessel sizes, natural rivers are also dredged and widened, causing similar SWI and associated ecological impacts. For example, mangrove encroachment was observed after dredging the Tanshui River in Taiwan, and substantial changes to tidal range and SWI occurred after dredging in the Pearl River in China (Liu et al. 2001, Yuan and Zhu 2015). In addition, many rivers have been straightened to improve transportation and flood control (Bechtol and Laurian 2005). These fast, deep, and straight channels act in a similar way to canals in permitting upriver SWI.

In addition to changes in surface water systems, groundwater pumping can cause SWI when aquifer recharge rates are lower than the rate of abstraction. Many coastal wetlands are dependent on the delivery of fresh groundwater from coastal aquifers either directly (i.e., discharge wetlands) or via complex interactions between surface water, groundwater, and soil water in the root zone (Kaplan et al. 2010a, Sánchez-Martos and Sánchez 2013). Aquifer overexploitation can lead to reduced freshwater delivery and increasing groundwater salinity in these systems. Examples of groundwater abstraction-driven aquifer salinization are numerous globally (Sadeg and Karahanoðlu 2001), and in most systems, groundwater use plays a larger role than SLR in driving SWI (Ferguson and Gleeson 2012).

Land-use change, watershed modifications, and hydraulic infrastructure construction in coastal watersheds can also play a large role in driving SWI. Land-use changes can increase impervious surface area, reduce aquifer recharge, and lower freshwater heads and facilitating SWI (Ranjan et al. 2006). Drainage of wetland areas for agricultural and urban development in some coastal areas has also lowered groundwater elevations, leading to SWI in aquifers and surface waters (Holman and Hiscock 1998). Dam construction and surface water withdrawals can also cause reduction or elimination of freshwater flows to the coast, increasing SWI (Bunce et al. 2010). For example, the Aswan High Dam on the Nile River in Egypt has reduced freshwater outflows to the Mediterranean by more than 75%, leading to increased salinity in the wetlands of the Nile River Delta (Johnson 1997).

Finally, accelerated subsidence driven by human activities is a major driver of SWI in many coastal regions. For example, historical rates of subsidence in the MRD region were approximately 3 mm/yr, driven primarily by compaction and isostatic rebound (Wolstencroft et al. 2014). Current rates of subsidence are as high as 23 mm/yr driven by extraction of subsurface hydrocarbons and brine, which destabilizes overlaying sediments (Morton et al. 2002, Ko and Day 2004). Subsidence rates in the delta were highest after extraction peaked in the 1970s and have slowed with decreasing rates of extraction (Morton et al. 2002). Similar trends have been observed in the coastal wetlands of Texas (Morton et al. 2006). Schmidt (2015) illustrated the global scope of this problem with examples from Vietnam, Indonesia, and Thailand, where high rates of natural resource and groundwater extractions are also leading to accelerated subsidence and SWI.

Impacts of SWI on Coastal Wetlands

The impacts of SWI on coastal wetlands are variable in scale as a function of the magnitude, duration, and frequency of the SWI stressors summarized above. These impacts include changes in primary production, community composition, and the provisioning of ecosystem services (Fig. 3).
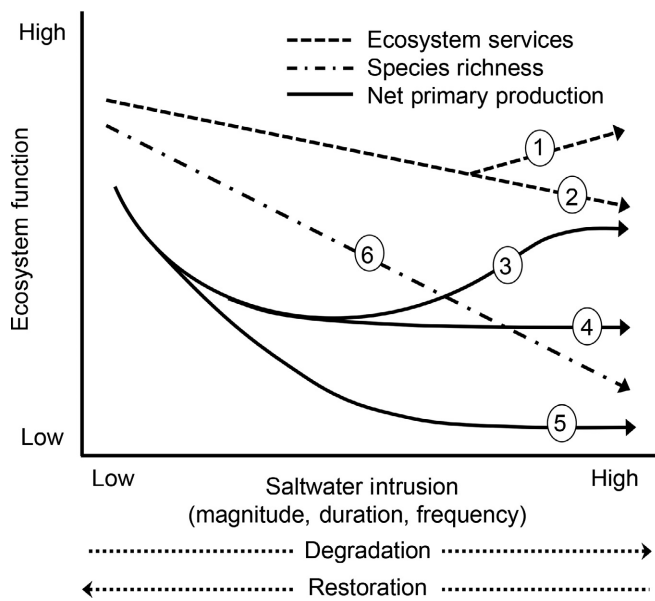
Fig. 3. Schematic representation of the impacts of saltwater intrusion (SWI) on coastal wetland structure and function. In many cases, wetland ecosystem services (ES) are degraded by SWI; however, the overall effect is a function of the spatial and temporal scale of the impact. This situation is illustrated by comparing (1) and (2), where (1) indicates loss of ES with SWI, followed by conversion to another wetland type with equivalent (though perhaps different) ES values, and (2) represents a continued trajectory of ES decline. Similarly, net primary productivity (NPP) is reduced under SWI stress, but may follow several trajectories according to the magnitude and rate of SWI. These pathways are demonstrated by comparing (3), (4), and (5), where (3) represents conversion from one productive wetland type to an equally productive type over time, (4) represents conversion to a less productive wetland system, and (5) represents conversion to mudflat or open water. Finally, wetland species richness (SR) generally decreases monotonically with SWI disturbance, as the vegetated end-member is likely to be salt marsh (dominated by monospecific stands of *Spartina alterniflora*) or mangrove forest (dominated by a small number of mangrove species).

Primary productivity

Coastal wetlands rely on primary production as a source of autochthonous organic matter for soil accretion. While salt marshes can increase productivity and accretion to match contemporary rates of SLR (Cahoon 2006), specific thresholds for marsh survival are locally variable and strongly impacted by human activities (Kirwan and Megonigal 2013). In contrast, SWI is strongly correlated with decreased primary production in coastal freshwater wetlands, where it can instigate increased subsidence and flooding (Fig. 3). Under increasing salt stress, coastal freshwater marshes have reduced above- and below-ground biomass (McKee and Mendelssohn 1989, Neubauer 2011). This trend is also present in tidal forested wetlands, where net primary productivity can be twice as high in forests with low salinity relative to those with high salinity (Pierfelice et al. 2015). At longer timescales, tree mortality in high-salinity forests may open the canopy,

increasing net primary production (Fig. 3). Critically, not only do salt-stressed forests have lower primary productivity, but even sub-lethal salt levels can lead to weaker seedlings (Pezeshki et al. 1990), pointing to the possibility of a change in community composition due to insufficient recruitment (see section *Community composition*).

There is some evidence that salt-stressed coastal forests can return to background primary productivity levels if freshwater is returned to the system (Fig. 3). A managed freshwater pulse intended to keep oil from washing ashore after the 2010 Deepwater Horizon incident increased aboveground productivity in salt-stressed *Taxodium distichum* swamps in the area (Middleton et al. 2015). These results indicate that properly timed freshwater pulses can mitigate the effects of SWI in impacted coastal forests and marshes, representing an important management strategy in systems with the appropriate hydraulic infrastructure (see section *SWI Restoration Goals and Tools*).

Community composition

In addition to changes in the productivity of particular species, SWI can cause wholesale shifts in community composition (Kaplan et al. 2010*a*). In coastal wetlands, plant community is largely structured by abiotic factors, including water level and tidal range, salinity regime, and soil biogeochemistry (Isacch et al. 2006, Mitsch and Gosselink 2015). The response of plants to SWI is a function of the level, duration, and abruptness of exposure to saline water and varies widely across species (McKee and Mendelssohn 1989), but in general, salinity is the most important abiotic factor in determining coastal wetland habitat types (Lin et al. 2012). Under increased salinity and flooding regimes, freshwater wetland soils become more anaerobic and have higher interstitial sulfide concentrations (Flynn et al. 1995). Increased salinity, decreased oxygen, and increased sulfide in these soils disfavor freshwater and brackish plants not adapted to these conditions (McKee and Mendelssohn 1989). Physiological effects of flooding and salt stress include decreased stomatal response and reduced photosynthetic rates; however, the magnitude of the effect is species-specific (Pezeshki et al. 1990).

The magnitude and duration of SWI events is a strong determinant of whether coastal wetland communities will recover or whether a different community assemblage will take its place. If SWI is temporary, impacted wetlands may recover, with recovery rates depending on post-intrusion salinity and inundation (Flynn et al. 1995) and the presence of a propagule source to re-populate the wetland, if necessary. If SWI is sufficiently frequent or extreme, the plant community will transition to a more salt- and/or flood-tolerant wetland type (Webb and Mendelssohn 1996), presuming that a source of more salt-tolerant propagules is available (McKee and Mendelssohn 1989). For example, *Spartina alterniflora* will

colonize bare ground left behind after SWI in brackish and freshwater marshes (Sutter et al. 2015). Once established, *S. alterniflora* is resilient to future SWI, allowing the system to transition to salt marsh. Similar directional transitions have been observed in coastal freshwater forests where SLR and modified water management have caused plant community shifts to salt marsh and mangrove ecosystems (Desantis et al. 2007).

Ecosystem services

Associated with these changes in productivity and community composition, SWI alters the types and amounts of ecosystem services (ES) provided by coastal wetlands (Fig. 3). Ecosystem services derived from coastal wetlands include fisheries production, carbon sequestration, coastal erosion/shoreline stabilization, tourism and recreation, water quality, and biodiversity support (Blair et al. 2015). Understanding how the ecosystem services of coastal wetlands may change with increased SWI is critical for envisioning resilient coastal human communities and prioritizing funding for the management and restoration of the coastal ecosystems (Rugai and Kassenga 2014); however, relatively few studies address this question quantitatively. In a modeling study of coastal wetlands in Georgia, Craft et al. (2009) found that tidal freshwater swamps, tidal freshwater marshes, and salt marshes will decline in areal extent by as much as 34%, 39%, and 45%, respectively, under the maximum IPCC SLR scenario. Taken together, these changes will yield a reduction in nitrogen soil sequestration by 23% and reduce potential denitrification by 25%, a finding supported by Hines et al. (2015). Other ES negatively affected by SWI are carbon sequestration (Neubauer et al. 2013) and storm wave attenuation (Wamsley et al. 2010). It is important to note, however, that conversion from one wetland type to another may change ES provisioning without substantially degrading its overall value (Fig. 3).

One benefit of the ES approach is the ability to convert ES value into economic terms through ES valuation (Costanza et al. 1989, 1997). Valuations put a price tag on consumptive uses, such as fisheries production, and non-consumptive uses, such as carbon sequestration (Coen and Luckenbach 2000). In an ES valuation for the MRD, Batker et al. (2014) found that the system currently provides $12–47 billion of ES annually and investigated how that value would change under three scenarios, including no restoration, prevent future land loss, and large-scale restoration. They found that no restoration would result in a loss of $41 billion in ES. Efforts to prevent future land loss would prevent this loss, but not provide additional value. Large-scale efforts to increase the size of the MRD would prevent the $41 billion loss *and* provide an additional $21 billion of ES. While ES valuation methods are inherently uncertain (Johnson et al. 2012), these dollar amounts highlight the economic importance of coastal wetlands and lead us to a critical question for

coastal resource managers: Should we proactively manage and restore coastal wetlands? If so, how do we set restoration goals, and what approaches are available and most likely to succeed? In the following section, we address these questions in the context of SWI and coastal water management.

SWI Restoration Goals and Tools

Deciding whether and how to pursue restoration interventions in coastal wetlands impacted by SWI requires the assessment of four interrelated questions:

1. What are the drivers of degradation? Are they natural, anthropogenic, or synergistic?
2. Which elements of ecosystem structure and function are degraded?
3. To what ecological condition do we seek to restore the ecosystem?
4. Can we modify the drivers of degradation? If so, what tools are available to do so?

The first two questions are addressed in sections *Drivers of Saltwater Intrusion in Coastal Wetlands* and *Impacts of SWI on Coastal Wetlands* of this review, which summarize the suite of SWI drivers leading to coastal wetland degradation and subsequent impacts on wetland structure and function (Figs. 1–3). In this section, we aim to address the second two questions by briefly reviewing approaches for setting restoration goals and outlining the set of tools available for SWI management in coastal wetlands.

Setting restoration goals for coastal wetlands impacted by SWI

Restoration goals describe the abiotic states and biotic conditions that an ecological restoration effort attempts to achieve. Setting restoration goals is critical for determining project successes and learning from failures (Society for Ecological Restoration [SER] 2004), but can be problematic for coastal ecosystems impacted by natural and anthropogenic SWI. Particularly challenging aspects of restoration are (1) identifying an appropriate reference system to guide project success and (2) assessing the vulnerability and/or resilience of restored sites to present and future environments.

The selection of *n* reference system helps restoration planning by identifying one or more intact ecosystems that the restored site will emulate. Many restoration projects in North America explicitly or implicitly use "pre-European conditions" as a temporal bound for their reference models (White and Walker 1997). This goal is idealistic, but impractical, particularly for coastal ecosystems. Clearly, climate patterns, hydrology, fauna, and other critical drivers in coastal wetlands have changed in the past 500 yr so that using a "pre-European" reference model is inappropriate. Instead of restoring historical

conditions, Choi (2004) argues that we should restore for future conditions based on best available knowledge about projected climate, hydrology, and flora/fauna. This approach recognizes the need to quantify the dynamic range of variation in reference and restored systems. This "dynamic reference model" approach (Hiers et al. 2012, Kirkman et al. 2013) is particularly critical for restoring coastal wetlands impacted by SWI as it enables practitioners to incorporate current and projected SLR and expected increases in storm frequency and intensity into restoration goal-setting efforts.

Restoration planning also requires an assessment of the resilience of the restored system to future environmental conditions (Harris et al. 2006). Ecological resilience is defined as the ability to recover structure or function following a disturbance (Lake 2013). Some coastal wetlands are thus resilient to SLR, but only up to a certain threshold (see section *Drivers of Saltwater Intrusion in Coastal Wetlands*), after which they will transition to another ecosystem type (see section *Impacts of SWI on Coastal Wetlands*). One critical, and often over-looked, element of restored ecosystem resilience is the ability of the restored area to support the regeneration of the species necessary for continued stability or development along a successional trajectory (SER 2004). For example, habitat suitability models based on adult tree survival will underpredict SWI impacts on coastal forests because they overlook life-cycle requirements of seeds and seedlings (Kaplan 2010). It is also important to draw a contrast between resilience and resistance (i.e., the ability of a system to avoid damages) when considering restoration of SWI-impacted coastal wetlands. Many large-scale coastal ecosystem restoration projects rely on heavily engineered solutions to control, limit, or even apply ecosystem disturbances. This "Command and Control" approach (Holling and Meffe 1996) aims to make systems more resistant to disturbance, but often ends up decreasing system resilience (Pittock and Finlayson 2013) by reducing systems' range of natural variation and their capacity to respond to future disturbances (Gunderson 2000, Hilderbrand et al. 2005). Reduced resilience can be particularly problematic for projects with large infrastructural requirements (including some of the coastal water management tools we describe below) and can be a major limitation of complex restoration projects.

Freshwater management to mitigate the impacts of SWI on coastal wetlands

With explicit restoration goals set and an understanding of the likely resilience of restored systems, practitioners may then explore restoration approaches to ameliorate SWI. The potential for improved freshwater management to mitigate the worst effects of SWI is based on the strong inverse relationship between river discharge (i.e., freshwater flow) and salinity in coastal rivers and estuaries (Kratzer and Grober 1991, Kaplan et al. 2010b). Several

authors have leveraged this flow–salinity relationship to explain observed changes in coastal ecosystem function and in support of coastal ecosystem restoration. For example, there was a significant decrease in salinity during periods of high freshwater flow from the Mississippi River in Louisiana and observed improved ecosystem function during this time (Middleton et al. 2015). This relationship has also been used to develop restoration flow thresholds for coastal wetlands impacted by SWI (Kaplan 2010) and to improve freshwater flow management to mitigate severe oyster reef decline tied to SWI (Seavey et al. 2011, Kaplan et al. 2016). In short, upstream flow management may be of fundamental importance in the fight against SWI.

Efforts to ameliorate SWI fall into three overarching categories: (1) modified groundwater and surface water withdrawals, (2) modified groundwater and surface water deliveries, and (3) the use of engineered structures. All approaches rely to some extent on the use of models to develop alternative management scenarios. While our focus is on SWI in coastal wetlands, we also consider SWI in coastal aquifers. In both cases, the strong relationship between freshwater quantity and salinity makes inferences drawn from either type of study useful for understanding how freshwater management can be improved to ameliorate SWI impacts.

Modified water withdrawal

Modified surface water and/or groundwater withdrawals are changes to water use that seek to prevent or minimize SWI and include (1) optimization of the amount, location, and timing of withdrawals and (2) conservation efforts to reduce the magnitude of water diverted from natural systems (Table 1). In the first category, a large body of work has advanced groundwater pumping optimization schemes that reduce the risk of aquifer SWI (e.g., Willis and Finney 1988, Zekri et al. 2015). Also, see reviews in Bear et al. (1999), Cheng et al. (2004), Dhar and Datta (2007), and Werner et al. (2013). These efforts couple spatially distributed groundwater hydrology models with optimization methods to meet a specific management goal (e.g., maximize pumping rate and minimize saline intrusion; Dhar and Datta 2009). While these studies usually focus on the protection of aquifers, the same approach can be used to protect and restore coastal wetlands by including objective functions that address ecosystem requirements (e.g., maintaining freshwater for phreatophytic vegetation or sustaining fresh groundwater delivery via springs or diffuse seepage). A numerical model was developed to understand how past anthropogenic activities and SLR have combined to cause SWI into coastal freshwater forests in the Po River Plain in Italy (Giambastiani et al. 2007). Model results suggested that reduced groundwater withdrawals during periods of low recharge would be required to reduce SWI into this ecologically and historically important ecosystem. Similarly, groundwater modeling on the Kona Coast of

Table 1. Water management strategies, approaches, and tools used to ameliorate the impacts of saltwater intrusion in coastal systems.

Strategy	Approach	Tools	References
Modified water withdrawals	Optimize amounts, locations, and timing	Modeling and scenario analysis	Willis and Finney (1988), Nobi and Gupta (1997), Bear et al. (1999), Cheng et al. (2004), Dhar and Datta (2007), Giambastiani et al. (2007), Duarte et al. (2010), Liu et al. (2001), Werner et al. (2013), Zekri et al. (2015)
	Water conservation	Irrigation efficiency improvements	Laraus (2004)
		Rainwater harvesting	DeBusk et al. (2012)
		Use reduction	Gössling (2001), Kent et al. (2002)
Modified water deliveries	Optimize amounts, locations, and timing	Modeling and scenario analysis	Kaplan (2010), Koch et al. (2015), Zhang et al. (2015)
	Enhance groundwater recharge	Infiltration ponds/basins	Walraevens et al. (2002), Narayan et al. (2003), Dillon (2005)
		Land-use conversions	O'Leary (1996), Scanlon et al. (2005), McLaughlin et al. (2013)
		Artificial recharge	Todd (1959), Dillon (2005), Asano (2016), Werner et al. (2013), Maliva (2014)
Engineered structures	Direct freshwater	Diversions	Ward et al. (2002), DeLaune et al. (2003), Snedden et al. (2007), Cui et al. (2009), Allison and Meselhe (2010), Paola et al. (2011), Das et al. (2012), Kemp et al. (2014), Middleton et al. (2015)
	Impede saltwater	Saltwater barriers and other structures	Applegate (1990), Mulrennan and Woodroffe (1998), Giambastiani et al. (2007), Miloshis and Fairfield (2015)

Hawai'i was used to balance water use and submarine groundwater discharge, which supports important near-shore ecosystems (Duarte et al. 2010). Further application of the SWI modeling approaches in coastal aquifers is an important area of study for the management and restoration of coastal wetlands.

Similar optimization approaches balance the amount and timing of water withdrawals from surface water with the needs of coastal ecosystems. For example, Nobi and Gupta (1997) developed a SWI model for a coupled coastal stream–aquifer system in southwest Bangladesh to demonstrate how planned increases in groundwater pumping would cause undesirable increases in estuarine salinity. Model results showed that prevention of estuarine SWI was possible, but only if plans for expanded groundwater pumping were abandoned or if fresh surface water was diverted from another nearby river. Optimization modeling was also applied to allocate surface and groundwater resources to balance social, economic, and environmental demands in the Pearl River Delta in China, which is experiencing SWI (Liu et al. 2001). The model developed by these authors could be used to propose modified surface water withdrawals during the dry season to limit SWI. In this case, however, the authors proposed water system engineering projects (reservoir storage, river diversions, and distributed pumping stations) to satisfy cumulative water demand.

In the examples, above, models were used to optimize the magnitude and timing of water withdrawals while limiting SWI. Water conservation schemes, in contrast, aim to reduce consumptive use across one or more sectors, leaving a larger proportion of freshwater available for natural systems (or other sectors). Mirroring the discussion above, the literature describing water conservation is expansive (Anderson 2003, Inman and Jeffrey 2006, Pujol 2013), but only a subset of studies address efforts directed specifically to limit SWI. For example, Laraus (2004) points to expansion of irrigated agriculture in driving widespread SWI in the southern and eastern Mediterranean and proposes reductions in agricultural water usage by using drought-tolerant crops and rain-fed production systems. In coastal North Carolina, rainwater harvesting systems provide an alternative irrigation water source and improve aquifer recharge in urban areas, potentially helping to manage SWI that threatens aquifers and ecosystems in that region (DeBusk et al. 2012). Simple conservation efforts to reduce the impacts or likelihood of SWI have been developed in tourist areas where surface and groundwater use outpace supply and/or recharge (Gössling 2001, Kent et al. 2002). While direct study of the impacts of water conservation practices on SWI is limited, opportunities for improved conservation are abundant across the agricultural, municipal, and industrial sectors (Anderson 2003, Low et al. 2015) and have the potential to beneficially augment freshwater flow in coastal systems impacted by SWI.

Modified water delivery

While modified water withdrawals seek to *prevent or minimize* SWI, modified water deliveries usually aim to *treat existing SWI symptoms*. Several studies and projects have sought to develop coastal wetland management or restoration plans by linking water management, water quality, and habitat suitability or threshold models to identify beneficial environmental flow regimes in systems impacted by SWI. Coupled models have been used to provide guidance on the timing and magnitude of freshwater flow required to restore coastal floodplain forests impacted by SWI (Kaplan 2010) and to set flow augmentation goals for restoration of the Florida Everglades (Koch et al. 2015). Regarding resilience, Koch et al. (2015) showed that enhanced freshwater flows

would enhance mangrove peat accumulation, helping these coastal wetlands keep up with SLR and serving to stabilize coastlines and limit SWI. In a novel study of SWI mitigation in the Pearl River estuary in southern China, the design of a coastal wetland network was modeled to optimally store and convey freshwater runoff to the system to reduce river and estuarine salinities during periods of SWI (Zhang et al. 2015).

In addition to augmenting surface water flows, increased freshwater delivery to coastal systems impacted by SWI can also be achieved by increasing recharge. Enhanced recharge may be achieved via changes at the land surface that increase the proportion of precipitation that infiltrates and is then available to surface water or groundwater resources (McLaughlin et al. 2013), or via "artificial recharge" to pump water into the subsurface (Todd 1959). Land surface changes include modifications of surface topography, land use/land cover (LU/LC), or the soil profile to facilitate infiltration. These changes include construction of infiltration ponds and basins (Dillon 2005), LU/LC conversion to more permeable cover types (Scanlon et al. 2005), and soil management to maximize infiltration rates (O'Leary 1996). For example, groundwater modeling was used to assess the coupled use of infiltration ponds and pumping reductions to freshen salinized aquifers and restore native coastal dune vegetation in the Netherlands (Walraevens et al. 2002) and to minimize SWI in aquifers in Australia (Narayan et al. 2003). Where "enhanced recharge" strategies use modification of the surface environment, "artificial recharge" is the direct transfer of water below the surface to increase aquifer storage for later recovery or to serve as a hydraulic barrier to intruding seawater from pumping further inland (Werner et al. 2013, Maliva 2014, Asano 2016). Taken together, enhanced and artificial recharge strategies are collectively referred to as "managed aquifer recharge" (Dillon 2005). This approach can be used to protect both aquifers and coastal ecosystems from SWI, particularly where coastal ecosystems rely on groundwater discharge (Duarte et al. 2010).

Engineered structures

Engineered structures are used widely around the world to modify the delivery of surface water and groundwater in systems impacted by SWI. Engineered solutions can be very effective at preventing or limiting disturbances, though this added protection can decrease system resilience (Pittock and Finlayson 2013) by making the system less adaptable to future disturbances (Gunderson 2000). Structures can also be expensive to design, build, and maintain. Structures reviewed here include freshwater diversions, saltwater barriers, dams, and other hydrological connections and cutoffs (Table 1).

Freshwater diversions are structures that connect, or re-connect, coastal floodplain wetlands to their river channel to mimic historical flooding and sediment delivery. Hydrodynamic and water quality assessment

models can assess the potential for freshwater diversions to be implemented in support of coastal wetland restoration goals in the MRD (RTMRD 2016). The results of studies to assess the potential for these structures to slow wetland loss have been mixed. Increased sediment delivery and salinity reduction in the wetlands of the northern Breton Sound Basin were observed after the opening of the Caernarvon Diversion (DeLaune et al. 2003), suggesting that the diversion has the potential to slow down or reverse wetland loss trends. Snedden et al. (2007) found that this structure provided the largest source of sediments to the sinking wetlands, but that sediment delivery was far lower than historical values and insufficient to support marsh accretion relative to SLR. These authors note that the diversion's primary design goal is to maintain optimal salinity for shellfish production, limiting the amount of freshwater discharge and associated sediments allowed through the diversion. These studies point to the potential utility of diversions in combatting SWI, but also highlight the challenge of using engineered structures to support multiple ecological functions and stakeholders simultaneously.

The MRD is also home to the Davis Pond Diversion (DPD), which is the largest diversion in the world (Das et al. 2012). The DPD connects the Mississippi River to the Barataria Estuary, a complex of lakes, bays, and over 200,000 ha of fresh, brackish, and saline wetlands. Like the Caernarvon Diversion, the DPD was designed to counteract SWI and may reduce site-specific salinity in associated coastal freshwater forests (Middleton et al. 2015). This potential was investigated by Das et al. (2012), who found that despite the large volumes of water passing through the DPD, its effects on salinity were limited. In the upper and lower estuary, salinities were primarily influenced by upstream freshwater and downstream marine waters, respectively, limiting the effect of the DPD. In the central region, diverted flow volumes strongly affected predicted salinity, with modeled differences as high as 10 PPT, suggesting that management of flows through the DPD can be used to manage for specific vegetation communities in specific places. Additional reviews of the current efficacy and future potential for freshwater and sediment diversions in the MRD can be found in Allison and Meselhe (2010), Paola et al. (2011), and Kemp et al. (2014). In general, these studies argue for the utility of large diversions to benefit marshes and suggest continued monitoring to better understand the potential and limitations of this engineered approach.

While the MRD is the world's testing ground for large-scale flow diversions, the technique has also been employed in other regions. Fresh, brackish, and saline floodplain wetlands on the Yellow River (China) severely degraded by SWI are being restored using freshwater flow diversions to re-connect wetlands to the river channel (Cui et al. 2009). The diversion yielded increased wetland hydroperiods, improved surface water and soil

salinity conditions, increased soil nutrient and organic matter, expanded and more diverse vegetation, and an increase in avifauna use relative to the pre-restoration landscape (Cui et al. 2009). A similar project was implemented on the Nueces River in Texas, where freshwater flow reductions had caused severe SWI in the river and its associated wetlands and delta (Ward et al. 2002). River diversions constructed as part of this project increased freshwater flow to the upper Nueces Delta by 700%, restoring a more natural salinity gradient, with positive effects on the abundance and diversity of wetland vegetation and benthic communities.

While river diversions are designed to encourage flow of freshwater and sediments into SWI-impacted areas, saltwater barriers are designed to prevent upriver transport of saline water. Saltwater barriers include gates, dams, dikes, levees, and other structures that physically block the upstream flow of saline water and have been used in river systems around the world. For example, natural resource managers have built concrete and rock "barrages" and earthen blocks to prevent upstream SWI in the Lower Mary River (Northern Territory, Australia) during the dry season and reduce floodplain drainage rates during the wet season (Applegate 1990). These structures have been generally successful in halting the further upstream advance of SWI, which has impacted nearly 20,000 ha of freshwater wetlands over the past 70 yr (Mulrennan and Woodroffe 1998). In an update to this study, Miloshis and Fairfield (2015) developed a scoring system to rank the effectiveness of "traditional" (i.e., engineered) vs. "eco-engineering" and "do-nothing" approaches to manage coastal wetlands in Australia's Northern Territory coastal wetlands. They found traditional engineering approaches (including barrages or levees), were "neither required, nor beneficial" for ecosystem management and that "eco-engineering" (conservation and gentle rehabilitation) was the best management response (discussed further in section *The Path Forward: Restore or Retreat?*). The "Chiaro Pontazzo" water management approach described by Giambastiani et al. (2007) is another example of an engineered artificial embankment. In this case, the structure creates a shallow freshwater "lake" that prevents SWI into a coastal pine forest.

Dams, gates, and other types of mechanical barriers block saline waters in major river systems affected by SWI, though in many cases these are primarily used to reduce flood damage by storm surge in urban areas. While construction of saltwater barriers in coastal rivers is primarily for the protection of drinking and agricultural water supplies, we were unable to find published studies describing their number, distribution, or impacts on water quality and ecosystem integrity. For some engineered structures, it is their removal that best restores natural salinity distributions in coastal wetlands. For example, Yang et al. (2010) investigated the impact of breaching or removing dikes to restore estuarine and coastal ecosystem function. Modeling results suggested

that dike breaching would restore hydrology and salinity in support of four habitat types and substantially increase the area flushed with freshwater. Moreover, by inference, the large number of studies demonstrating how existing dams drive or exacerbate SWI (Ge et al. 2015, Hutchinson 2015, Webber et al. 2015) illustrate the potential for saltwater barrier removal to alleviate this effect.

Finally, we note that where optimizing delivery of existing flows, enhancing recharge, and deploying engineered structures is insufficient to manage SWI, it may be possible to develop new freshwater resources. Several of the examples above implicitly or explicitly include suggestions for new resource development, such as reservoir construction (Kaplan 2010), river diversions and other engineered solutions (Nobi and Gupta 1997, Liu et al. 2001), and non-point source recharge (McLaughlin et al. 2013). Clearly, water is neither created nor destroyed in the global hydrological cycle, so in these cases, it is important to understand not only how the diverted resource reduces SWI stress in the affected system, but also the potential for detrimental effects in the systems from which these "new" resources were diverted.

Other strategies and "best practices" to support coastal wetland restoration

While this review focuses specifically on SWI drivers, effects, and management, there are several other important tools and methodological approaches that support coastal wetland restoration and management. Here, we provide an initial set of references for further investigation in each of these areas. Efforts to facilitate sediment delivery and accretion are central to coastal marshes "keeping up" with SLR. Reviews of sediment dredging, diversions, and other modes of delivery are summarized in Ford et al. (1999), Day et al. (2005), and Tong et al. (2013). In some cases, facilitated or assisted migration of plant and animal species may be necessary to support coastal adaptation in areas with substantial built infrastructure (Smith and Lenhart 1996, Doyle et al. 2010, Minteer and Collins 2010, Dawson et al. 2011). In other cases, the "do-nothing" option may be preferable (Burton 1996, Dolan and Walker 2006, Miloshis and Fairfield 2015), allowing for managed decline and transition via ecosystem self-organization (Odum 1989). In all cases, restoration ecologists, ecosystem managers, and other stakeholders may seek to balance responsive behaviors (i.e., restoration) with proactive behaviors (i.e., improved management and direct climate mitigation) to meet this global challenge (Harris et al. 2006).

We also identified many studies that aimed to summarize "best management practices" for wetland management in the face of climate change (Table 2). Early reviews highlighted the importance of increasing protections for healthy ecosystems and removing existing non-climate stresses from degraded systems to enhance the potential for these ecosystems to adapt to non-stationary conditions

Table 2. Summary of best practices for wetland management under climate change.

Authors	Best Practices
Burkett and Kusler (2000)	Increased protection/remove stresses
	Develop setbacks
	Sediment diversions
	Link fragmented wetlands and waterways
	Use water control structures to enhance particular functions
	Secure water resources for wetland conservation
	Wetland restoration
Erwin (2009)	Significantly reduce non-climate stressors
	Protect coastal wetlands and accommodate SLR (acquisition, setbacks, restoration)
	Monitoring, training, and education
	Incorporate climate oscillations
	Medium- and long-range planning: strategize conservation priorities
Koch et al. (2015)	Integrative & resilience-focused management
	Paradigm that considers coupling of connected terrestrial, freshwater, & marine ecosystems
	Develop comprehensive regional/local governance and planning frameworks
Wiens and Hobbs (2015)	Frame realistic and complementary goals
	Embrace uncertainty
	Enlist public support

(Burkett and Kusler 2000, Erwin 2009). These reviews also suggested land management and policy actions, including developing setbacks to allow for SLR and habitat transition, building water and sediment diversions, restoring connectivity between fragmented wetlands and waterways, securing water allocations for wetlands, using water control structures where needed to secure particular functions, and restoring degraded wetlands. Many of these recommendations directly address the issues of SWI, and applications of these practices are discussed in the sections above. To make coastal *human* communities more adaptable to future conditions, these practices must be combined with improved monitoring, training, and education about coastal systems (Erwin 2009).

More recent reviews advocate for several broad goals, including prioritizing integrative and resilience-focused management as opposed to schemes that seek to resist or avoid disturbance (Koch et al. 2015). This more holistic paradigm considers the coupled terrestrial–freshwater–marine ecosystem and seeks to develop comprehensive regional/local governance and planning frameworks (Koch et al. 2015). Given the constraints of widespread development, environmental degradation, and a non-stationary climate, Wiens and Hobbs (2015) advise ecosystem managers and restoration practitioners to set realistic goals and advocate the use of the dynamic reference concept (Hiers et al. 2012) to help do so, particularly in heavily altered systems. This approach stresses the importance of embracing uncertainty, using the adaptive management framework to adjust goals and objectives if necessary, and enlisting and maintaining public support during this dynamic process (Wiens and Hobbs 2015).

The Path Forward: Restore or Retreat?

In this review, we have shown that where SWI is driven by anthropogenic and/or synergistic drivers, some amount of coastal wetland restoration is likely achievable. The question remains: Is restoration advisable given the number of SWI drivers described in section *Drivers of Saltwater Intrusion in Coastal Wetlands*? The choice between restoration and retreat is not a simple dichotomy. The preferable action will depend on local physiographic setting, the specifics of SWI drivers and impacts, desired ecosystem services, and economic considerations. No single "cookbook" or if-then scenario analysis tool is sufficient to support decision-making for a specific situation. However, in many cases, the best option may be a compromise between restoration and retreat that maximizes the value of ES provided by coastal wetlands while minimizing invested time, money, effort, and risks to human health and the built environment. Performing such an analysis requires an ES valuation of the restored and unrestored systems (section *Ecosystem services*), coupled with an assessment of the time, energy, and money required to pursue restoration. The magnitude of this investment will be driven by the size and proportion of a site that is degraded, the ecological "distance" from the desired state (SER 2004), and the actions required to achieve this state.

From an economic perspective, it makes sense to pursue restoration if the marginal value of the ES provided by the restored ecosystem relative to the unrestored case is greater than the cost of restoration, monitoring, and future adaptive management (i.e., if the benefit-cost ratio [BCR] is >1; Gregory et al. 2006). Retreat is preferable if the opposite is true. In the real world, decision-making is unlikely to be this clear-cut. Some issues that may confound the process include differing objectives of stakeholders, valuations of a given ecosystem service, levels of knowledge regarding science and terminology. Despite these complications, framing the decision in monetary terms can help open the discussion to a broader range of stakeholders (Brauman et al. 2014).

Whether a specific system should be restored can be evaluated by considering both the BCR of restoration actions and the site's current and future susceptibility to SWI. From an ES perspective, BCR is defined as ecosystem services value gained relative to restoration and maintenance costs (Barendregt et al. 1992, Turner et al. 2000, 2007, Dubgaard 2004). While selection of a threshold BCR to justify restoration is arbitrary, values ≫1 indicate that restoration is advisable and values ≪1 suggest that retreat makes more sense. At BCRs closer to 1, the decision to restore or retreat is less clear and may be strongly driven by the issues listed above (largely social), along with other site-specific environmental considerations. For example, restoration of sites with BCR values <1 may be appropriate if restoration activities are tied to the protection of threatened and endangered species, historical/cultural sites, or other non-use values, which

are not necessarily valued as highly in ES assessments (Laurila-Pant et al. 2015).

Threshold BCR values may also be influenced by current and future susceptibility to SWI of a particular site. Current and future SWI impacts include predicted changes in SLR projections, regional climate, future groundwater abstractions, and land use, all of which could lower groundwater recharge or modify freshwater flows. As SWI susceptibility increases, a higher BCR may be required to justify restoration activities given a higher risk and greater potential for future failure. The assessment of a system's susceptibility to SWI requires explicit definition of a time window (e.g., 30 yr in the future) over which to assess BCR. While not a precise methodology, this approach can be used by decision-makers and restoration practitioners to prioritize restoration projects that maximize return on investment and minimize risk.

We close by noting that while changes in global climate that drive SLR and rainfall availability are beyond the scope of local control, choices about the delivery of freshwater to the coast in support of resilient coastal ecosystems are not. This review points to the variety of technical approaches that can be used in these efforts; however in all cases, water management decisions come down to choices and compromises about conservation, allocation, and infrastructural investment. Where this decision-making framework is codified in environmental and water law, communities are likely in a better position to balance water allocation with natural resource protection. For example, Florida state law mandates development of minimum flows to prevent harm to natural resources (s. 373.042, Florida Statutes), and this framework has been used to quantify flow requirements in coastal rivers to prevent SWI into coastal wetlands. While the science of "environmental flows" has advanced greatly in recent decades (Tharme 2003), implementation of environmentally protective flow recommendations via explicit legal frameworks and enforcement of these laws lags behind (Kiwango et al. 2015). As stressors on coastal wetlands continue to increase, wider application and enforcement of environmental laws, which allocate a portion of the water budget to the natural environment, will be critical for maintaining or restoring a resilient coastal environment.

Acknowledgments

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