2019

The Invisible Flood: The Chemistry, Ecology, and Social Implications of Coastal Saltwater Intrusion

Kate Tully

Keryn Gedan

(...)

Molly Mitchell
Virginia Institute of Marine Science

et al

Recommended Citation

Tully, Kate; Gedan, Keryn; (...); Mitchell, Molly; and et al, The Invisible Flood: The Chemistry, Ecology, and Social Implications of Coastal Saltwater Intrusion (2019). *BioScience*, 69(368), 378.
doi: 10.1093/biosci/biz027

The Invisible Flood: The Chemistry, Ecology, and Social Implications of Coastal Saltwater Intrusion

KATE TULLY, KERYN GEDAN, REBECCA EPANCHIN-NIELL, AARON STRONG, EMILY S. BERNHARDT, TODD BENDOR, MOLLY MITCHELL, JOHN KOMINOSKI, THOMAS E. JORDAN, SCOTT C. NEUBAUER, AND NATHANIEL B. WESTON

Saltwater intrusion is the leading edge of sea-level rise, preceding tidal inundation, but leaving its salty signature far inland. With climate change, saltwater is shifting landward into regions that previously have not experienced or adapted to salinity, leading to novel transitions in biogeochemistry, ecology, and human land uses. We explore these changes and their implications for climate adaptation in coastal ecosystems. Biogeochemical changes, including increases in ionic strength, sulfidation, and alkalinization, have cascading ecological consequences such as upland forest retreat, conversion of freshwater wetlands, nutrient mobilization, and declines in agricultural productivity. We explore the trade-offs among land management decisions in response to these changes and how public policy should shape socioecological transitions in the coastal zone. Understanding transitions resulting from saltwater intrusion—and how to manage them—is vital for promoting coastal resilience.

Keywords: climate change, land use management, salinization, saltwater intrusion, sea-level rise

Saltwater intrusion—the landward movement of seawater—has numerous, complex consequences for coastal ecosystems and communities, which can and should be distinguished from the impacts of flooding due to sea-level rise. Saltwater intrusion may precede tidal inundation of low-lying uplands, and dramatically changes the chemistry of tidal freshwater wetlands. Although there always has been a swath of coastal land adapted to salt, interactions among sea-level rise, climate change and coastal water infrastructure is causing saltwater to reach areas with human and ecological communities that have not experienced or adapted to salinity, leading to novel biogeochemical, ecological, and human land use transitions.

Recent studies have documented increases in storm-driven flooding, and higher-amplitude tidal inundation is associated with sea-level rise, which affects private property and public infrastructure (Moftakhari et al. 2015). Studies also have considered the impacts of coastal inundation on ecosystems and protected lands (Epanchin-Niell et al. 2017). However, the effects of saltwater intrusion, particularly on ecosystem services, have received less attention, perhaps because shifts in water chemistry are invisible to the general public. Marine salt inputs to previously freshwater-dominated systems have profound impacts on ecosystem biogeochemistry, leading to coastal forest loss, species invasions, reductions in agricultural productivity, declines in coastal water quality, and marsh migration. Understanding these biogeochemical and ecosystem changes is critical for predicting the full impacts on society and identifying how people and communities can best adapt.

To improve our understanding of coastal saltwater intrusion, we first describe the climatic and anthropogenic drivers of saltwater intrusion and its global extent, identify the chemical alterations that accompany salinization, and synthesize the ecological consequences. We then examine the social impacts, including how individual landowners and managers may respond, and the role for public policy in shaping socioecological transitions in the coastal zone.

Drivers of saltwater intrusion

Saltwater intrusion and the degree of upland salinization are driven by five main factors: the position of sea-level relative to the land and water table, the frequency and magnitude of storms and tides, the frequency and duration of drought, water use (e.g., surface and groundwater withdrawals for drinking water and irrigation), and hydrologic connectivity (e.g., tide gates, levees, agricultural, diversions, and roadside ditches, and canals; figure 1). Because each of these five factors are themselves variable in space, time, frequency, and duration, the process of ecosystem salinization is extremely dynamic and can occur slowly or quickly, depending on the unique combination of drivers acting on any particular

BioScience 69: 368–378.
doi:10.1093/biosci/biz027

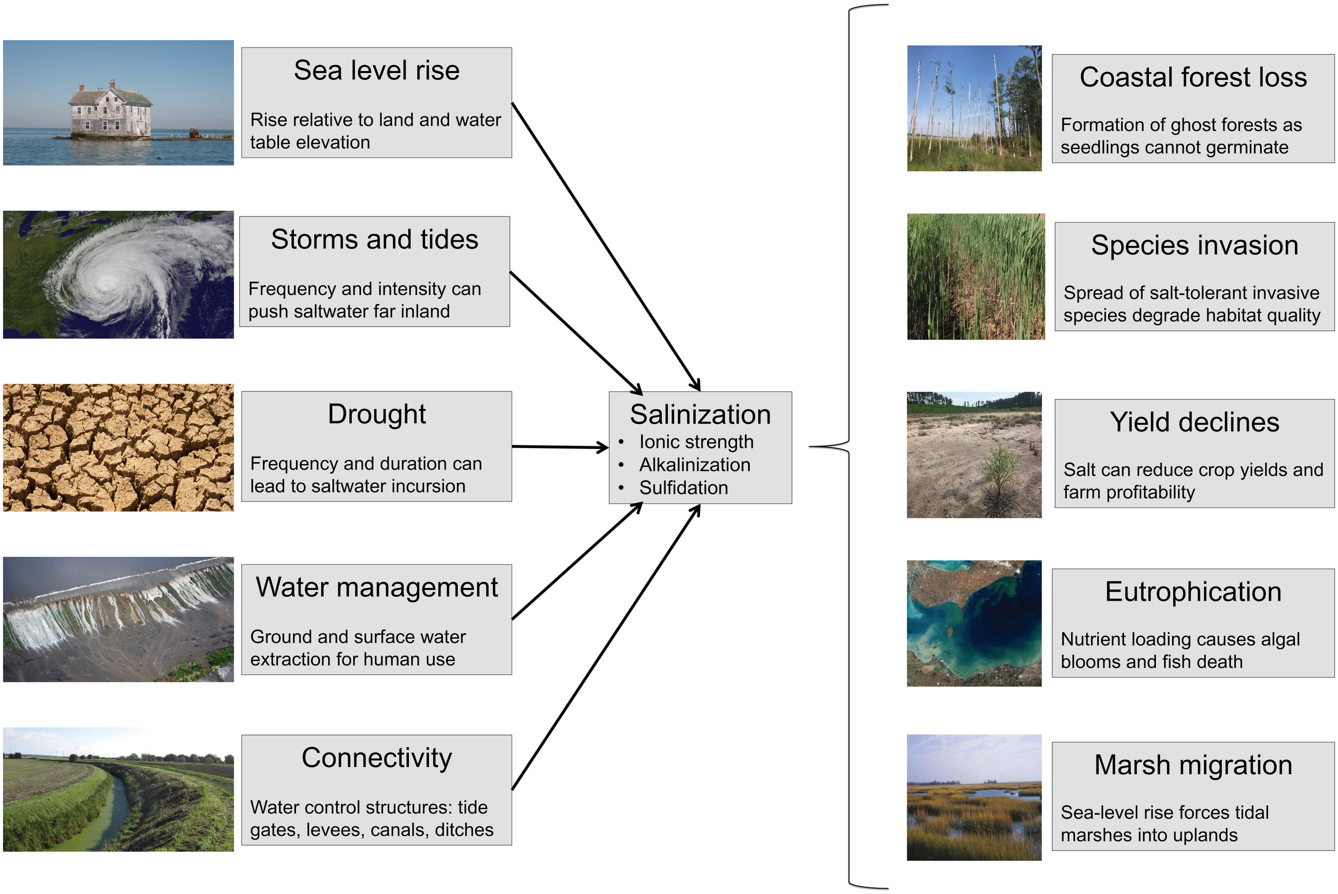

Figure 1. Drivers of salinization in uplands, three primary components of salinization, and their effects on biogeochemistry, plant communities, and ecosystem services.

location. For example, sea salts may be delivered slowly to coastal ecosystems through groundwater exchange, surface water mixing, and tidal pumping or rapidly during extreme events like hurricanes and tsunamis.

The penetration of saltwater into surface and groundwaters is enhanced by relative sea-level rise, which reflects the combined effects of changes in the elevation of the ocean surface (2.8–3.6 millimeters per year globally since 1993; Nerem et al. 2018) and vertical movements of the land surface (e.g., subsidence of 1–3 millimeters per year across North American Coastal Plain; Karegar et al. 2016). Prolonged droughts can increase the extent of saltwater intrusion (sometimes referred to as *incursion*; Ardón et al. 2013) and increase the peak salinity of high tides or of rivers during low flows (Neubauer and Craft 2009). Ground- and surface-water extraction for human consumption, irrigation, or industry can greatly accelerate the otherwise slow rate of saltwater intrusion attributed to sea-level rise (White and Kaplan 2016), and can play a more important role in aquifer salinization than sea-level rise (Ferguson and Gleeson 2012).

The frequency and intensity of storms and tides affect how far saltwater is pushed into uplands and whether water control structures such as levees and tide gates are overtopped (Yang et al. 2018). When overtopped, water control structures can trap marine salts just as effectively as they prevent saltwater intrusion during lower flows. In all cases, ecosystems that are hydrologically connected to saline coastal waters are more likely to have saltwater intrusion (figure 2). Many coastal landscapes that were hydrologically altered to promote drainage or enable navigation are now subject to saltwater intrusion. In Louisiana, for example, thousands of kilometers of canals were cut and dredged by oil, gas, and pipeline companies to allow industrial access to the marsh and now provide a conduit for saltwater to flow into low-salinity regions (Poulter et al. 2008, Bhattachan et al. 2018). Similarly, in northern Italy, agricultural irrigation and drainage channels, constructed to move freshwater to fields, now serve the reverse—and unintended—purpose of facilitating salinization (Antonellini et al. 2008).

Widespread evidence of saltwater intrusion

Saltwater intrusion has been documented in coastal regions across the globe (Barlow and Reichard 2010, Herbert et al. 2015), and its impact on human water resources is a growing threat. Worldwide, about 40 percent of the human population lives within 100 kilometers of a coastline and relies on coastal freshwater aquifers for drinking water (Kummu et al. 2016). Small changes in salinity can render water

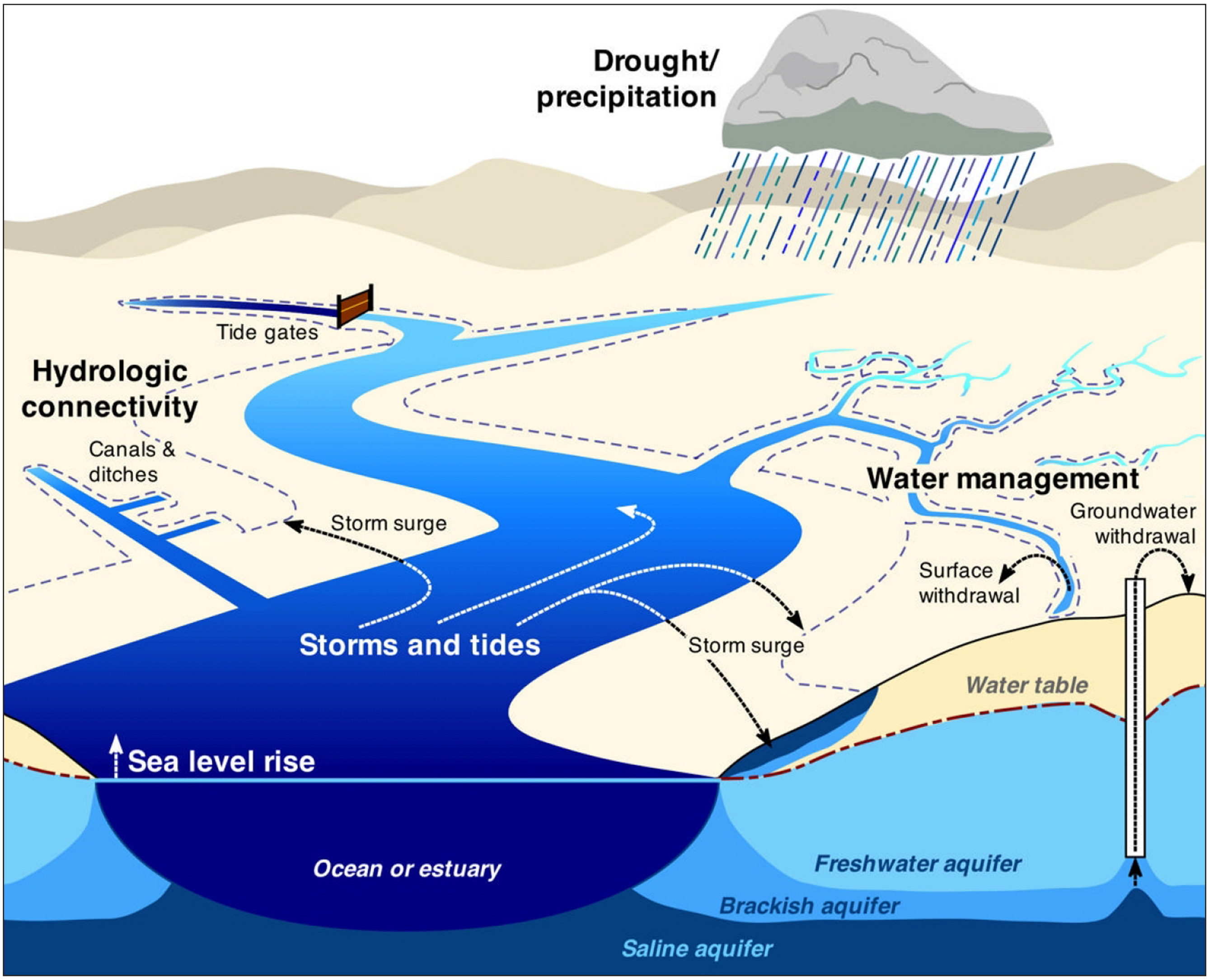

Figure 2. Illustration of shallow coastal system undergoing saltwater intrusion. Coastal saltwater intrusion (dashed lines) is caused by relative sea-level rise, water management (e.g., water withdrawals), the connectivity of creeks and ditches to the source of saltwater, the frequency of rainfall and drought events, and storms and tides.

undrinkable, because drinking chloride concentrations above 250 milligrams per liter (salinity of approximately 0.5 parts per thousand) in drinking water can cause hypertension and stroke (Vineis et al. 2011). Hundreds of groundwater wells in coastal countries such as Cyprus, Mexico, Oman, and Israel have been closed because of saltwater intrusion (Barlow and Reichard 2010). The 2004 earthquake and tsunami that devastated the province of Aceh, Indonesia, salinized drinking water sources both near the coast and inland (Fesselet and Mulders 2006). Bangladesh and Vietnam, the fourth and fifth largest producers of rice in the world, are extremely vulnerable to saltwater intrusion from sea-level rise, freshwater withdrawals, and tropical storms (Mahmuduzzaman et al. 2014) endangering global food security. In the Tropical Pacific, the Carteret Islands have become increasingly uninhabitable because of rising sea levels and saltwater intrusion, which have resulted in declines in agricultural productivity and evacuation to nearby Papua New Guinea (Green 2016). In Los Angeles, water management practices to stave off saltwater intrusion in freshwater aquifers include artificial recharge using imported freshwater (Johnson and Whitaker 2003). In South Florida the Biscayne aquifer must be protected from saltwater intrusion, which otherwise would jeopardize the drinking water of 8 million people in metropolitan Miami (Barlow and Reichard 2010). In other regions affected by coastal saltwater intrusion, people rely on deep wells or reverse osmosis water supply systems to continue to inhabit and farm affected lands (Alameddine et al. 2018). Effects of salinization of drinking water is significant and immediately impactful, whereas other effects of saltwater intrusion are less obvious, because they result from slower biogeochemical and ecological changes that nonetheless

Box 1. Biogeochemical effects of salinization.

Increased ionic strength
- Osmotic stress → plant stress or death
- Ion exchange → nutrient mobilization

Alkalinization
- pH change → phosphorus release from acid soils
- Clay dispersion → prevent drainage
- Cation bridging → change carbon dynamics

Sulfidation
- High sulfate → reduced carbon storage
- Sulfide toxicity → plant stress/death
- Formation of iron-sulfur minerals → phosphorus release

have important ramifications for human society and adaptive responses.

Biogeochemical effects of salinization from saltwater intrusion

The movement of saltwater into terrestrial freshwater systems alters the composition of porewater in upland and wetland soils. Seawater is a complex solution; on average, it has around 400 times more salt than freshwater, with the base cations (calcium, magnesium, sodium, potassium) and sulfate ions accounting for 44% of total marine salts. To better describe how salinization is changing upland chemistry, we detail the process of salinization in three independent but co-occurring chemical alterations: an increase in ionic strength, alkalinization, and sulfidation (box 1).

Increase in ionic strength. As saltwater intrudes into freshwater systems, the ionic strength of surface water or soil porewater increases. Regardless of the specific ionic composition, increases in ionic strength cause osmotic stress for organisms not adapted to saltwater. High salt concentrations essentially draw water out of plant cells. This process is particularly damaging to plant seeds, which imbibe water prior to germination. High osmotic stress prevents tree regeneration in coastal forests (Williams et al. 1999). In soils, an increase in ionic strength increases ion exchange, affecting the release of nutrients such as nitrogen and phosphorus (Weston et al. 2006). For example, multiple studies have shown that saltwater intrusion promotes ammonium liberation from soil particles because of the elevated competition for exchange sites with sodium, calcium, and magnesium (Weston et al. 2010, Steinmuller and Chambers 2018). In nitrogen-limited coastal systems, this leads to eutrophication of estuaries (Ardón et al. 2013).

Alkalinization. The ionic composition of seawater determines the alkalinity, or the *amount* of base cations in solution. The link between salinization and alkalinization has been described in United States stream waters polluted by salts from road deicing, irrigation, and other human activities (Kaushal et al. 2018), and the same general phenomenon acts in coastal lands affected by saltwater intrusion. Saltwater introduces large quantities of calcium and magnesium carbonates, which can change the potential for acid neutralization in soils and affect phosphorus availability (Weston et al. 2006, Chambers et al. 2011). In upland soils, such as those found in agricultural fields, an increase in sodium may also lead to clay dispersion, effectively changing the soil structure by plugging soil pores and impeding water infiltration and drainage (Rengasamy et al. 1984), which can delay planting and dramatically reduce yields.

Finally, the intrusion of alkaline saltwater can induce cation bridging, or bonds between cations and dissolved organic matter, which creates flocculent organic matter. This process reduces the microbial use of dissolved organic carbon and may lead to increased carbon sequestration and burial (Ardón et al. 2016). For example, in the Florida Everglades, flocculant organic matter is a primary detrital energy source, and plays a critical role in food webs (Sanchez and Trexler 2016). Therefore, increased concentrations of base cations through saltwater intrusion can change ecosystem carbon dynamics with consequences for ecosystem productivity and carbon storage.

Sulfidation. Sulfidation is the increase in sulfide stored in ecosystems following saltwater intrusion. Seawater contains relatively high concentrations of sulfate, which is reduced under low-oxygen conditions in the microbial metabolic pathway of sulfate reduction to sulfide. Sulfidation is unlikely after brief episodes of flooding with seawater, but occurs with chronic intrusion (figure 3). The increased availability of sulfate can drive shifts in microbial organic matter mineralization pathways (e.g., from methane production to sulfate reduction) and have major impacts on ecosystem carbon storage (Weston et al. 2011, Neubauer 2013). Furthermore, the reduced form of sulfur produced from chronic intrusion, hydrogen sulfide (H_2S), is toxic to plants and other organisms and can drive shifts in plant community composition and productivity (Lamers et al. 2013). If iron is present in the ecosystem, sulfide can also react to form pyrite and other precipitates, reducing sulfide toxicity and storing the sulfur compounds in the ecosystem (Sjøgaard et al. 2016). Therefore, free H_2S formation is unlikely after brief episodes of flooding with seawater, but occurs with chronic intrusion once the H_2S buffering capacity of the soils is depleted (figure 3). This buffering capacity will vary among systems because of differences in sedimentation rates, iron and organic matter availability, and microbial activity (Heijs and van Gemerden 2000). Nevertheless, saltwater intrusion may eventually consume available iron (Schoepfer et al. 2014), and free sulfides can accumulate in soils or sediments until the system is “fully sulfidized” (figure 3).

Sulfidation can play an important role in other biogeochemical reactions, notably phosphorus cycling and the release of phosphorus to coastal waters. In nonsaline systems, biologically available phosphate is largely bound

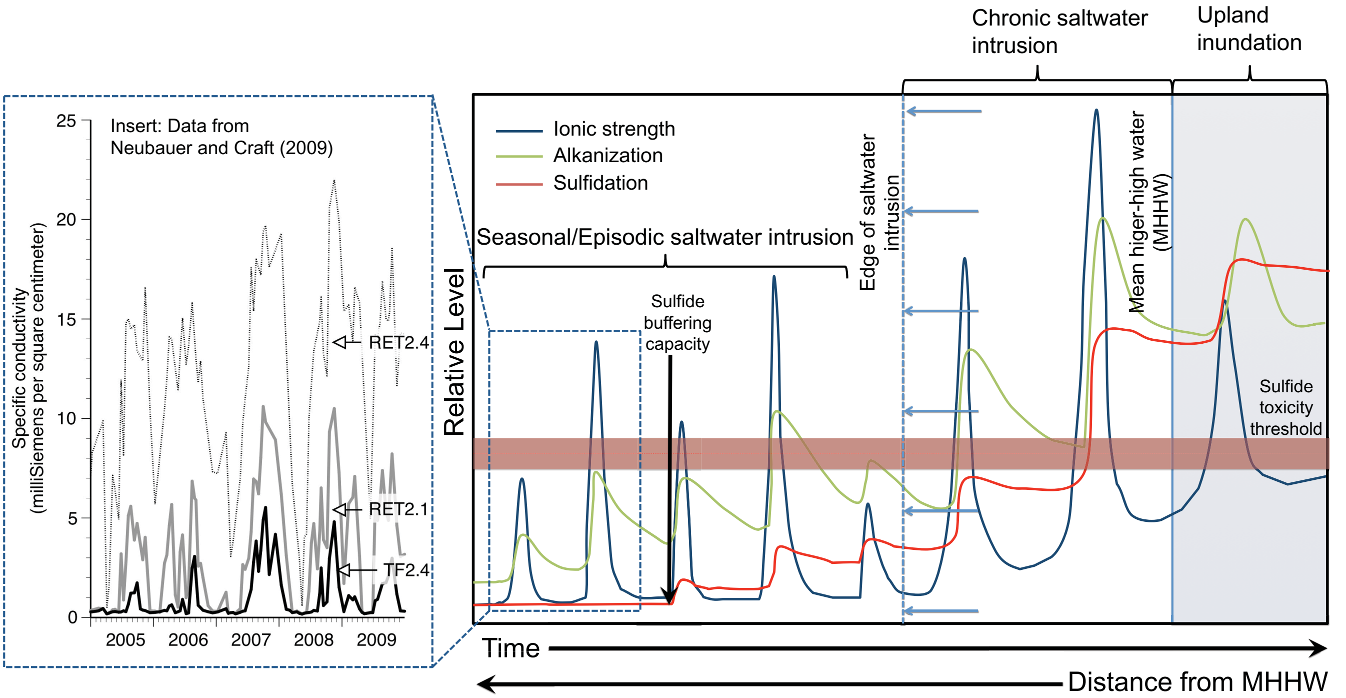

Figure 3. Conceptual diagram of the changes in ionic strength (solid black line in print/blue line online), alkalinization (gray dotted line in print/green line online) and sulfidation (solid gray line in print/red line online) that are likely to occur as systems salinize. Shaded region represents upland regions that experience tidal inundation and the dashed grey line (blue line online) represents the moving edge of saltwater intrusion. The dotted horizontal line represents the point above which a freshwater system experiences sulfide toxicity. Moving along the x-axis can be viewed as changes at a locale over time as sea levels rise or as differences in dynamics across a spatial gradient extending inland from the coast (i.e., distance from mean higher high water, MHHW). Insert (dashed box in print/dashed blue box online) highlights data on salinity from the Potomac River at three stations ranging from seaward (dashed and grey lines, RET2.4 and RET2.1) to landward (solid black line, TF2.4). Similar to ionic strength, salinity returns to zero (data from Neubauer and Craft 2009).

in iron mineral precipitates (Jordan et al. 2008, Hartzell et al. 2017). During sulfidation, iron consumed in reactions with sulfides is no longer available to bind to phosphate, thus increasing water column phosphorus concentrations. Because of the important role of iron as a sink for sulfide, the reactive iron content of soil could be a good predictor of the timescale over which sulfidation will lead to plant community changes and phosphorus loading into waterways.

Dynamics of salinization. We present a conceptual model of expected changes from salinization as the edge of saltwater intrusion moves across a landscape (figure 3). Changes in ionic strength may be pulsed within a given system as fluctuations in freshwater inputs (e.g., via rainfall or river flow) can dilute seawater or drought can concentrate salts (figure 3). However, over time, ionic strength of the system will likely increase, as is evinced by increased salinity in the Delaware River (Ross et al. 2015), James, and Chikahominy Rivers over the twenty-first century (Rice et al. 2012). Alkalinization of a system will likely track increases in ionic strength (especially sodium concentrations), and trends in alkalinity will depend on the initial buffering capacity of the system (Kaushal et al. 2018). Sulfidation will increase more rapidly on reaching the sulfide buffering capacity threshold, although some degree of buffering may continue as a result of organic inputs (Heijs and van Gemerden 2000, Valdemarsen et al. 2009, Duan and Kaushal 2015). As soils periodically dry, some of the sulfides oxidize back to sulfate, partly reacidifying the soil (Luther et al. 1986), leading to interannual variation in pH and dissolved sulfide concentrations (figure 3). Overall, ecosystems may remain more sulfidic and alkaline following pulsed salinity events, even if the ionic strength has returned to (near) freshwater levels. This “memory” may result in more rapid alkalinization and sulfidation following subsequent saltwater intrusion events, such that seasonal saltwater intrusion may have a cumulative impact on ecosystem biogeochemistry, eventually tipping the system into a new state (e.g., forested wetland to brackish wetland).

Ecological consequences of saltwater intrusion

Biogeochemical changes in ionic strength, alkalinization, and sulfidation due to saltwater intrusion cascade into ecosystem changes. Soil microbial communities can shift following salinization (e.g., Herbert et al. 2015, Chambers et al. 2016, Dang et al. 2019 and the citations therein). Although plants vary widely in their tolerances to high sulfide concentrations and salinity, few nonadapted plant species can tolerate waterlogged and saline environmental conditions (Lamers et al. 2013). Therefore, saltwater intrusion can drive inland retreat of upland plant communities and landward migration of coastal plant communities (Brinson et al. 1995) far in advance of regular tidal inundation with seawater (Brinson et al. 1995, Krauss et al. 2009, Antonellini and Mollema 2010). The pace and trajectory of plant community shifts will depend on the rate of saltwater intrusion as well as factors such as soil type, geomorphology, and hydrologic connectivity. In addition, upland ecosystems and land use can either constrain or facilitate community shifts. Below, we examine the transitions in coastal ecosystems occurring in response to saltwater intrusion, to highlight its far-reaching effects and repercussions for coastal ecosystems and ecosystem services.

Coastal forest loss. Salinity stress can lead to coastal tree death. Large areas of forest can be killed by salt deposited by a single storm event (Middleton 2016), but incremental forest retreat also contributes to landscape change (DeSantis et al. 2007). One early sign of salinity stress in trees is reduced sap flow during periods of drought or high salinity (Teobaldelli et al. 2004). In general, forest regeneration is more sensitive to salinity and associated flooding than mature tree mortality (Williams et al. 1999, Kirwan et al. 2007), because seedlings of flood-tolerant trees require moist, but not chronically flooded, soils for germination (Conner et al. 2007). Often, habitat conversion will occur many years after regeneration has ceased in stressed areas, when mature trees are killed by a saltwater intrusion event associated with a drought or large storm (e.g., Hurricane Sandy; Middleton 2016). Colonization by salt-tolerant vegetation follows forest death (Williams et al. 1999).

The rate of forest retreat varied with upland slope, climate conditions, and forest type (Smith 2013). For example, in the Big Bend region of Florida, 148 square kilometers of coastal palmetto, bottomland, pineland, and mixed forest converted to transitional and tidal marsh between the late 1800s and 1995, resulting in a lateral retreat of 4.2 meters per year (Raabe and Stumpf 2015), with forest replacement accelerating during a multiyear drought (Williams et al. 1999, DeSantis et al. 2007). Along tidal tributaries to the Delaware Bay, Smith (2013) observed a lateral retreat in coastal hardwood and Atlantic cedar swamp forests of 1.8 meters per year between 1930 and 2006. In contrast, there was no detectable forest retreat or mortality related to saltwater intrusion from 1974 to 2010 in southern New England, where upland slopes are steeper (Field et al. 2016).

Different sensitivities of tree species to flooding and salinity stresses also result in differential forest retreat. In the subtropics, inland forests of loblolly pine (*Pinus taeda*), winged elm (*Ulmus alata*), and Florida maple (*Acer floridanum*) were more sensitive, whereas juniper palmetto (*Juniperus virginiana* and *Sabal palmetto*) forests were more resistant (DeSantis et al. 2007). Similarly, Smith (2013) found higher rates of migration in Atlantic white cedar (*Chamaecyparis thyoides*) than in hardwood forest (*Acer rubrum*, *Liquidambar styraciflua*, *Nyssa sylvatica*, and *Quercus* spp.). Where a single vulnerable tree species is planted for harvest in plantations, we expect to see more uniform tree mortality relative to more diverse coastal forests.

Freshwater wetland conversion. Salinity is an important ecological structuring agent in estuarine wetland plant communities. In response to saltwater intrusion, downriver and more salt-tolerant tidal wetland plant species can shift upriver, with brackish plants replacing oligohaline plant communities (Schuyler et al. 1993) and oligohaline plants replacing tidal freshwater plant communities (Perry and Hershner 1999). Long-term saltwater intrusion experiments in tidal freshwater wetlands produce a reduction in diversity (Neubauer et al. 2013). Saltwater intrusion reduces tidal freshwater forest tree height and basal area (Krauss et al. 2009) and, ultimately, converts tidal forest to more salt-tolerant emergent marsh over the course of a decade or more (Craft 2012, White and Kaplan 2016).

Expansion of invasive species. Given the habitat preferences of some widespread invasive plant species for moderate salinity (e.g., salt cedar *Tamarix chinensis*; Cui et al. 2010) and the high salinity tolerance of others (e.g., *Schinus terebinth*, Ewe and da Silveira Lobo Sternberg 2007; *Casuarina* spp., Potgieter et al. 2014; *Triadica sebifera*, Conner et al. 2007), and many prominent invasive species are likely to expand their ranges in coastal areas because of saltwater intrusion.

In the eastern United States, the invasive haplotype of *Phragmites australis* is positioned to be a dominant species in salt-affected, retreating coastal ecosystems. It grows at the upland edge of tidal wetlands, is relatively salt-tolerant and is highly opportunistic (Chambers et al. 2008). In Delaware Bay, roughly half of the area of forest retreat now accommodates *P. australis*–dominated marsh (Smith 2013). In contrast, in lawns transitioning because of saltwater along the Connecticut coast, private landowners are inadvertently controlling the spread of *P. australis* through mowing (Anisfeld et al. 2017). During marsh migration, *P. australis* is replacing native high marsh species and contributing to coastal squeeze of native tidal marsh communities. In temperate Atlantic Coast marshes, *P. australis*' expansion will most affect the high marsh dominant *Spartina patens*. A *S. patens*–associated endemic species, the saltmarsh sparrow, *Ammodramus caudacutus*, is predicted to go extinct within the next several decades (Field et al. 2016). In summary, with increasing saltwater intrusion, many salt-tolerant invasive species are poised to colonize new areas and expand their range.

Agricultural productivity decline and field conversion. Very few crops can grow in sustained conditions of greater than 2 parts per thousand salinity (Tanji and Kielen 2002), which is substantially below the salinity levels in many salt-intruded fields (Tully et al. 2019). For example, crops such as corn (*Zea mays* L.) and soy (*Glycine max* (L.) Merr.) have suffered yield declines with saltwater intrusion (McNulty et al. 2015). Because of the episodic nature of saltwater intrusion and the variation in tolerance of crops, coastal croplands can exhibit gradual or sudden declines in productivity (Tanji and Kielen 2002). When farming practices of tillage or herbicide application continue during the initial stages of saltwater intrusion, fields often exhibit large bare areas in which crops cannot tolerate the saline conditions and the farm practices kill colonizing noncrop plants. Abandoned, salinized farmland, which has high nutrient levels (e.g., nitrogen and phosphorus), low organic matter, and a seed bank of opportunistic annuals, can be rapidly colonized by annual graminoids and herbs, including many agricultural weed species (Voutsina et al. 2015). On the arid Pacific Coast of Baja California, for example, abandoned agricultural land is rapidly colonized by the invasive and highly salt-tolerant ice plant, *Mesembryanthemum crystallinum* (Meyer et al. 2016).

Ecological changes from saltwater intrusion in agricultural lands are further shaped by past land use. The presence of tide gates influences the timing of transition for managed uplands. Drainage ditches and tile drainage, installed to divert excess freshwater, often serve as conduits for saltwater as land subsides or sea levels rise (Bhattachan et al. 2018). When impoundments fail, agricultural lands may convert to open water, as they did in Delaware, where over half of tidal wetlands were once impounded (Smith et al. 2017). Even small-scale land management may affect ecosystems exposed to saltwater intrusion. For example, heavy equipment tracks can create microtopography on graded fields, fostering wetter areas that may become salt pannes and pools as abandoned agricultural lands convert to wetlands.

Nutrient leaching and runoff from salt-intruded agricultural land will complicate nutrient management efforts in the coastal zone. Saltwater can extract legacy nitrogen and phosphorus from agricultural soils long after farmlands have been abandoned. Even years after restoring wetlands, legacy nutrients may be transported into ditches connected to tidal creeks draining into estuaries during times of drought and enhanced saltwater incursion (Ardón et al. 2013). Estuarine eutrophication can lead to harmful algal blooms, ecosystem state shifts (e.g., transitioning from oligotropic to eutrophic), and declines in valuable fisheries and is the subject of major regulatory efforts that may be stymied by unanticipated nutrient loading from coastal salt-damaged fields as saltwater intrusion moves across the landscape.

Social implications and adaptive responses to saltwater intrusion

The ecological outcomes of saltwater intrusion are additionally shaped by feedbacks among natural and human-driven processes. Land managers and society are affected by and must prepare for saltwater intrusion, which affects biodiversity, ecosystem services, and human well-being. Responses may differ among land managers, and these adaptation responses can affect the ecological trajectories of coastal systems. The management choice that is in the best interest of a particular land manager often may not provide the greatest benefits to society (e.g., Borchert et al. 2018). For a range of potential adaptation responses to saltwater intrusion, we describe potential costs and benefits that private land managers might consider, as well as the expected returns to society from each choice (table 1). We highlight several such examples below.

Adaptation to saltwater intrusion in coastal forests. Coastal forests may be managed for a variety of values, including timber harvest revenues, biodiversity provisioning, recreation (e.g., hiking, hunting), and aesthetic values (Burkhard et al. 2012). Saltwater can lead to forest death, thereby reducing the range of values described above. In addition, invasive species may invade salt-damaged land (Smith 2013), further reducing biodiversity and recreational values. In contrast to inaction, land managers could enhance one-time timber revenues by harvesting at-risk timber earlier than they otherwise would. Clearing forest in the path of migrating marsh may also provide salt marsh conservation benefits. Therefore, early harvest may promote a range of values provided by wetlands, including enhanced biodiversity, carbon storage, nutrient buffering, and improved water quality, as well as private gains.

Management of invasive species in salt-intruded forestlands could provide additional benefits to society, through improved habitat quality, but might not provide net benefits to a private land manager, depending on the values they derive from the land. A land manager may seek to maximize recreational hunting benefits, which can be reduced by nonnative plant invasions, but they would likely weigh the additional value against his private costs of control. If private incentives are insufficient to induce invasive species control on private lands, but societal benefits would outweigh the costs, then policy intervention may be warranted, which could be achieved through public invasive species control programs or incentives to landowners (Epanchin-Niell et al. 2010).

Adaptation to saltwater intrusion in tidal freshwater wetlands. In the absence of active management, freshwater tidal wetlands affected by saltwater intrusion are expected to transition to oligohaline or brackish tidal marsh. Preferences for particular wetland types may differ among landowners, depending on aesthetic or recreational preferences. Public preferences for freshwater wetlands may depend on the ability of freshwater wetland species to shift inland, the remaining extent of freshwater tidal wetlands, and the particular species that depend on the systems. Options for preserving freshwater wetlands typically involve major changes in the use and flow of freshwater. For example, freshwater diversions have been used to connect or reconnect coastal floodplain wetlands with their river channel to mimic historical flooding and sediment delivery in places such as the Mississippi River delta, China, and Texas, with varying outcomes (White and

Table 1. Public and private costs and benefits of landowner management responses to saltwater intrusion.

Consequence of saltwater intrusion	Management response	Public outcome	Private outcome
Coastal forest loss	Nonaction	• Delay tidal marsh migration • Biodiversity reductions due to species invasions	• Lose land and profit
	Early timber removal	• Promote tidal marsh migration ° Carbon storage ° Nutrient buffering ° Improve water quality	• Avoided harvest value losses
	Control invasive species	• Improved habitat quality • Enhanced biodiversity	• Enhanced recreation or hunting value • Financial costs of invasive control
Tidal freshwater wetland loss	Nonaction	• Transition from freshwater to saline marsh ° In some places, freshwater wetlands are an imperiled habitat • Loss of biodiversity and shift in species distributions	• Change in recreational value, e.g., shifts from freshwater to saltwater birds
	Management of water inflows and outflows	• Conserve freshwater wetlands • Impede tidal marsh migration • Incur upstream consequences of water management ° Water available for power generation, drinking water, etc.	• Sustain recreational value of freshwater wetland • Incur financial cost of constructing water control structures
Crop yield decline	Nonaction	• Eutrophication due to continued or increased fertilizer applications	• Yield declines • Profit loss
	Plant alternative crops (e.g., salt-tolerant)	• Promote rural economy • Potential nutrient remediation • Delay tidal marsh migration	• Sustained yield • Incurred costs of skill or equipment acquisition • Continued investments in agricultural production
	Abandon fields	• Legacy nutrient release • Promote tidal marsh migration and associated benefits	• Yield forfeit • Avoided investment in marginal land
	Enroll land in conservation easements	• Legacy nutrient release • Promote tidal marsh migration and associated benefits	• Yield forfeit • Avoided investment in marginal land • Alternative revenue source • Hunting value

Note: Nonaction scenarios report future trends compared to today. Other management responses are compared to the future nonaction scenario.

Kaplan 2016). Increased stream flow, which may be achieved by modifying water withdrawal or delivery, can also mitigate saltwater intrusion impacts (figure 2; Kaplan et al. 2010). Such changes in water management often require built infrastructure or widespread behavioral changes in water use across communities, which typically cannot be implemented by a single private land manager.

Adaptation to saltwater intrusion in coastal agricultural land. As saltwater intrusion on agricultural lands has led to crop productivity declines, land managers have responded in several ways, including abandoning land or enrolling it in easements, adding water control structures (White and Kaplan 2016), switching to alternative crops (Ventura et al. 2015, Voutsina et al. 2015), adjusting agricultural inputs (e.g., fertilizer and pesticides; Khanom 2016), and applying gypsum to displace sodium ions and improve drainage (Amezketa et al. 2005). Adaptation choices include the types of management changes and the timing of those changes. Some choices may require land managers to weigh costs and benefits only in the short-term (e.g., annual crop or agricultural input choices), whereas others may require evaluation over the long-term (e.g., investments in new farm equipment or participation in a habitat restoration programs).

A farmer facing reduced crop yields because of saltwater intrusion may switch to more salt-tolerant or lower-input crops (e.g., soybeans or sorghum) to increase profitability. This decision would likely weigh differences in expected profitability, accounting for any upfront costs of transitioning to a new crop (e.g., costs of acquiring new skills or equipment). The primary societal benefits of this type of transition likely would stem from support of the rural economy. However, if adoption of low-input crops (e.g., biomass crops, such as switchgrass [*Panicum*] and silvergrass [*Miscanthus*]) could also reduce legacy nutrients in the soils, this could provide potentially large public benefits in terms of reduced impacts from eutrophication, which is estimated to cause $81 billion to $441 billion per year in environmental damages across aquatic systems in the United States (Sobota et al. 2015). Identification of profitable crops that could also provide environmental benefits would help better align private and public interests. Alternatively, a farmer may stop farming salt-intruded fields to avoid economic losses resulting from poor yields. Field abandonment may provide public benefits by allowing landward migration of wetlands, therefore maintaining the ecosystem services and biodiversity benefits wetlands provide (Borchert et al. 2018).

Other options are to enroll abandoned agricultural lands in conservation easements or to restore wetland vegetation and control invasive species. Landowners might be motivated to restore wetland vegetation to obtain tax savings or other financial incentives or to obtain recreational opportunities such as hunting. As the productivity of agricultural land declines with saltwater intrusion, the opportunity cost of engaging in conservation or other land uses also declines, reducing the needed incentives to restore wetlands. However, federal, state, or local governments, or nongovernmental organizations may want to target incentives to enhance the transition of coastal farmland to wetland.

Managing the invisible flood

Understanding feedbacks among biogeochemical factors, ecological processes, and human responses to coastal saltwater intrusion is key for predicting and managing changes in coastal systems as sea levels rise. Adaptation to climate risks including sea-level rise associated inundation and saltwater intrusion can be hindered by knowledge gaps, financial resource constraints, social and cultural barriers, transaction costs, and institutional barriers. Also, because public and private benefits from the implementation of adaptation strategies often differ, incentive programs and public investments may be important for influencing coastal land management to bring about desired social and ecological outcomes. The collaboration of diverse disciplines and stakeholders is needed to improve our understanding of saltwater intrusion drivers and consequences. This understanding is critical for communities that are developing coastal policies that enhance long-term resilience in this era of rapid global change.

Acknowledgments

This collaborative work was supported by the National Socio-Environmental Synthesis Center (SESYNC) under funding received from the National Science Foundation (NSF) grant no. DBI-1639145. Coastal salinization research by EB and TB was supported by NSF Coastal SEES grant no. 1426802. Saltwater intrusion research by KT, KG, and BE-N was supported by a United States Department of Agriculture National Institute for Food and Agriculture Resilient Agroecosystems in a Changing Climate Challenge Area (grant no. 12451226, project accession no. 1015143) and a grant from the Harry R. Hughes Center for Agro-Ecology.

References cited

Alameddine I, Tarhini R, El-Fadel M. 2018. Household economic burden from seawater intrusion in coastal urban areas. Water International 43: 217–236.

Amezketa E, Aragüés R, Gazol R. 2005. Efficiency of sulfuric acid, mined gypsum, and two gypsum by-products in soil crusting prevention and sodic soil reclamation. Agronomy Journal 97: 983–987.

Anisfeld SC, Cooper KR, and Kemp AC. 2017. Upslope development of a tidal marsh as a function of upland land use. Global Change Biology 23, 755–766. https://doi.org/10.1111/gcb.13398.

Antonellini M, Mollema P, Giambastiani B, Bishop K, Caruso L, Minchio A, Pellegrini L, Sabia M, Ulazzi E, Gabbianelli G. 2008. Salt water intrusion in the coastal aquifer of the southern Po Plain, Italy. Hydrogeology Journal 16: 1541–1556.

Antonellini M, Mollema PN. 2010. Impact of groundwater salinity on vegetation species richness in the coastal pine forests and wetlands of Ravenna, Italy. Ecological Engineering 36: 1201–1211.

Ardón M, Helton AM, Bernhardt ES. 2016. Drought and saltwater incursion synergistically reduce dissolved organic carbon export from coastal freshwater wetlands. Biogeochemistry 127: 411–426.

Ardón M, Morse JL, Colman BP, Bernhardt ES. 2013. Drought-induced saltwater incursion leads to increased wetland nitrogen export. Global Change Biology 19: 2976–2985.

Barlow PM, Reichard EG. 2010. Saltwater intrusion in coastal regions of North America. Hydrogeology Journal 18: 247–260.

Bhattachan A, Emanuel RE, Ardón M, Bernhardt ES, Anderson SM, Stillwagon MG, Ury EA, Wright TKBAJP. 2018. Evaluating the effects of land-use change and future climate change on vulnerability of coastal landscapes to saltwater intrusion. Elementa Science of the Athropocene 62: 1–11.

Borchert SM, Osland MJ, Enwright NM, Griffith KT. 2018. Coastal wetland adaptation to sea level rise: Quantifying potential for landward migration and coastal squeeze. Journal of Applied Ecology 327: 29–12.

Brinson MM, Christian RR, Blum LK. 1995. Multiple states in the sea-level induced transition from terrestrial forest to estuary. Estuaries 18: 648.

Burkhard B, Kroll F, Nedkov S, Müller F. 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators 21: 17–29.

Chambers LG, Guevara R, Boyer JN, Troxler TG, Davis SE. 2016. Effects of salinity and inundation on microbial community structure and function in a mangrove peat soil. Wetlands 36: 1–11.

Chambers LG, Reddy KR, Osborne TZ. 2011. Short-term response of carbon cycling to salinity pulses in a freshwater Wetland. Soil Science Society of America Journal 75: 2000–2008.

Chambers RM, Havens KJ, Killeen S, Berman M. 2008. Common reed *Phragmites australis* occurrence and adjacent land use along estuarine shoreline in Chesapeake Bay. Wetlands 28: 1097–1103.

Conner WH, Krauss KW, Doyle TW. 2007. Ecology of tidal freshwater forests in Coastal Deltaic Louisiana and Northeastern South Carolina. Pages 223–253 in Conner WH, Doyle TW, Krauss KW, eds. Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer.

Craft CB. 2012. Tidal freshwater forest accretion does not keep pace with sea level rise. Global Change Biology 18: 3615–3623.

Cui B, Yang Q, Zhang K, Zhao X, You Z. 2010. Responses of saltcedar (*Tamarix chinensis*) to water table depth and soil salinity in the Yellow River Delta, China. Plant Ecology 209: 279–290.

Dang C, Morrissey EM, Neubauer SC, Franklin RB. 2019. Novel microbial community composition and carbon biogeochemistry emerge over time following saltwater intrusion in wetlands. Global Change Biology 58: 1–13.

DeSantis LRG, Bhotika S, Williams K, Putz FE. 2007. Sea-level rise and drought interactions accelerate forest decline on the Gulf Coast of Florida, USA. Global Change Biology 13: 2349–2360.

Duan S, Kaushal SS. 2015. Salinization alters fluxes of bioreactive elements from stream ecosystems across land use. Biogeosciences 12: 7331–7347.

Epanchin-Niell R, Kousky C, Thompson A, Walls M. 2017. Threatened protection: Sea level rise and coastal protected lands of the eastern United States. Ocean and Coastal Management 137: 118–130.

Epanchin-Niell RS, Hufford MB, Aslan CE, Sexton JP, Port JD, Waring TM. 2010. Controlling invasive species in complex social landscapes. Frontiers in Ecology and the Environment 8: 210–216.

Ewe SML, da Silveira Lobo Sternberg L. 2007. Water uptake patterns of an invasive exotic plant in coastal saline habitats. Journal of Coastal Research 231: 255–264.

Ferguson G, Gleeson T. 2012. Vulnerability of coastal aquifers to groundwater use and climate change. Nature Climate Change 2: 342–345.

Fesselet J-F, Mulders R. 2006. Saline wells in Aceh. Waterlines 24: 5–8.

Field CR, Gjerdrum C, Elphick CS. 2016. Forest resistance to sea-level rise prevents landward migration of tidal marsh. Biological Conservation 201: 1–7.

Green M. 2016. Contested territory. Nature Climate Change 6: 817–820.

Hartzell JL, Jordan TE, Cornwell JC. 2017. Phosphorus sequestration in sediments along the salinity gradients of Chesapeake Bay subestuaries. Estuaries and Coasts 40: 1–19.

Heijs SK, van Gemerden H. 2000. Microbiological and environmental variables involved in the sulfide buffering capacity along a eutrophication gradient in a coastal lagoon (Bassin d'Arcachon, France). Hydrobiologia 437: 121–131.

Herbert ER, Boon P, Burgin AJ, Neubauer SC, Franklin RB, Ardón M, Hopfensperger KN, Lamers LPM, Gell P. 2015. A global perspective on wetland salinization: Ecological consequences of a growing threat to freshwater wetlands. Ecosphere **6** (art 206–43).

Johnson TA, Whitaker R. 2003. Saltwater intrusion in the coastal aquifers of Los Angeles County, California. Pages 29–48 in Coastal Aquifer Management, Monitoring, and Case Studies. Boca Raton, FL.

Jordan TE, Cornwell JC, Boynton WR, Anderson JT. 2008. Changes in phosphorus biogeochemistry along an estuarine salinity gradient: The iron conveyer belt. Limnology and Oceanography 53: 172–184.

Kaplan D, Muñoz-Carpena R, Wan Y, Hedgepeth M, Zheng F, Roberts R, Service RR. 2010. Linking river, floodplain, and vadose zone hydrology to improve restoration of a coastal river affected by saltwater intrusion. Journal of Environment Quality 39: 1570–1515.

Karegar MA, Dixon TH, Engelhart SE. 2016. Subsidence along the Atlantic Coast of North America: Insights from GPS and late Holocene relative sea level data. Geophysical Research Letters 43: 3126–3133.

Kaushal SS, Likens GE, Pace ML, Utz RM, Haq S, Gorman J, Grese M. 2018. Freshwater salinization syndrome on a continental scale. Proceedings of the National Academy of Sciences 115: E574–E583.

Khanom T. 2016. Effect of salinity on food security in the context of interior coast of Bangladesh. Ocean and Coastal Management 130: 205–212.

Kirwan ML, Kirwan JL, Copenheaver CA. 2007. Dynamics of an estuarine forest and its response to rising sea level. Journal of Coastal Research 232: 457–463.

Krauss KW, Duberstein JA, Doyle TW, Conner WH, Day RH, Inabinette LW, Whitbeck JL. 2009. Site condition, structure, and growth of baldcypress along tidal/non-tidal salinity gradients. Wetlands 29: 505–519.

Kummu M, de Moel H, Salvucci G, Viviroli D, Ward PJ, Varis O. 2016. Over the hills and further away from coast: Global geospatial patterns of human and environment over the 20th–21st centuries. Environmental Research Letters 11: 034010–16.

Lamers LPM, Govers LL, Janssen IC, Geurts JJ, Van der Welle ME, Van Katwijk MM, Van der Heide T, Roelofs JG, Smolders AJ. 2013. Sulfide as a soil phytotoxin: A review. Frontiers in Plant Science 4: 1–14.

Luther GW, Church TM, Scudlark JR, Cosman M. 1986. Inorganic and organic sulfur cycling in salt-marsh pore waters. Science 232: 746–749.

Mahmuduzzaman M, Ahmed ZU, Nuruzzaman A, Ahmed F. 2014. Causes of salinity intrusion in coastal belt of Bangladesh. International Journal of Plant Research 4: 8-13.

McNulty S, Wiener S, Treasure E, Moore Myers J, Farahani H, Fouladbash L, Marshall D, Steele R, Hickman D. 2015. Southeast Regional Climate Hub Assessment of Climate Change Vulnerability and Adaptation and Mitigation Strategies. Pages 1–61 in Anderson T, ed. United States Department of Agriculture.

Meyer E, Simancas J, Jensen N. 2016. Can we create a sustainable future? Conservation at california's edge the consortium of California herbaria the russian wilderness: A legacy continued. Fremontia 44: 8–15.

Middleton BA. 2016. Differences in impacts of hurricane sandy on freshwater swamps on the Delmarva Peninsula, Mid-Atlantic Coast, USA. Ecological Engineering 87: 62–70.

Moftakhari HR, AghaKouchak A, Sanders BF, Feldman DL, Sweet W, Matthew RA, Luke A. 2015. Increased nuisance flooding along the coasts of the United States due to sea level rise: Past and future. Geophysical Research Letters 42: 9846–9852.

Nerem RS, Beckley BD, Fasullo JT, Hamlington BD, Masters D, Mitchum GT. 2018. Climate-change-driven accelerated sea-level rise detected in the altimeter era. Proceedings of the National Academy of Sciences 115: 2022–2025.

Neubauer SC. 2013. Ecosystem responses of a tidal freshwater marsh experiencing saltwater intrusion and altered hydrology. Estuaries and Coasts 36: 491–507.

Neubauer SC, Craft CB. 2009. Global change and tidal freshwater wetlands: Scenarios and impacts. Pages 253–266 in Barendregt A, Wingham DF, Baldwin AH, eds. Tidal Freshwater Wetlands. PUBLISHER.

Neubauer SC, Franklin RB, Berrier DJ. 2013. Saltwater intrusion into tidal freshwater marshes alters the biogeochemical processing of organic carbon. Biogeosciences 10: 8171–8183.

Perry JE, Hershner CH. 1999. Temporal changes in the vegetation pattern in a tidal freshwater marsh. Wetlands 19: 90–99.

Potgieter LJ, Richardson DM, Wilson JRU. 2014. Casuarina: Biogeography and ecology of an important tree genus in a changing world. Biological Invasions 16: 609–633.

Poulter B, Goodall JL, Halpin PN. 2008. Applications of network analysis for adaptive management of artificial drainage systems in landscapes vulnerable to sea level rise. Journal of Hydrology 357: 207–217.

Raabe EA, Stumpf RP. 2015. Expansion of tidal marsh in response to sea-level rise: Gulf Coast of Florida, USA. Estuaries and Coasts 39: 145–157.

Rengasamy P, Greene R, Ford GW, Mehanni AH. 1984. Identification of dispersive behaviour and the management of red-brown earths. Soil Research 22: 413–431.

Rice KC, Hong B, Shen J. 2012. Assessment of salinity intrusion in the James and Chickahominy Rivers as a result of simulated sea-level rise in Chesapeake Bay, East Coast, USA. Journal of Environmental Management 111: 61–69.

Ross AC, Najjar RG, Li M, Mann ME, Ford SE, Katz B. 2015. Sea-level rise and other influences on decadal-scale salinity variability in a coastal plain estuary. Estuarine, Coastal and Shelf Science 157: 79–92.

Sanchez JL, Trexler JC. 2016. The adaptive evolution of herbivory in freshwater systems. Ecosphere 7: e01414–e01415.

Schoepfer VA, Bernhardt ES, Burgin AJ. 2014. Iron clad wetlands: Soil iron-sulfur buffering determines coastal wetland response to salt water incursion. Journal of Geophysical Research: Biogeosciences 119: 2209–2219.

Schuyler AE, Anderson SB, Kolaga VJ. 1993. Plant zonation changes in the tidal portion of the Delaware River. Proceedings of the Academy of Natural Sciences of Phildelphia 144: 263–266.

Sjøgaard KS, Treusch AH, Valdemarsen TB. 2016. Carbon degradation in agricultural soils flooded with seawater after managed coastal realignment. Biogeosciences Discussions 14: 4375–4389. doi:10.5194/bg-2016-417.

Smith JAM. 2013. The role of *Phragmites australis* in mediating inland salt marsh migration in a mid-atlantic estuary. PLOS ONE 8 (art. e65091–9).

Smith JAM, Hafner SF, Niles LJ. 2017. The impact of past management practices on tidal marsh resilience to sea level rise in the Delaware Estuary. Ocean and Coastal Management 149: 33–41.

Sobota DJ, Compton JE, McCrackin ML, Singh S. 2015. Cost of reactive nitrogen release from human activities to the environment in the United States. Environmental Research Letters 10: 025006–14.

Steinmuller HE, Chambers LG. 2018. Can saltwater intrusion accelerate nutrient export from freshwater wetland soils? An experimental approach. Soil Science Society of America Journal 82: 283–210.

Tanji KK, Kielen NC. 2002. Agricultural drainage water management in arid and semi-arid areas. Pages 1–105. Allex 1. Crop Salt Tolerance Data. Food and Agriculture Organization of the United Nations.

Teobaldelli M, Mencuccini M, Piussi P. 2004. Water table salinity, rainfall and water use by umbrella pine trees (*Pinus pinea* L.). Plant Ecology 171: 23–33.

Tully K, Weissman D, Wyner WJ, Miller J, Jordan T. 2019. Soils in transition: saltwater intrusion alters soil chemistry in agricultural fields. Biogeochemistry 142: 339–356.

Valdemarsen T, Kristensen E, Holmer M. 2009. Metabolic threshold and sulfide-buffering in diffusion controlled marine sediments impacted by continuous organic enrichment. Biogeochemistry 95: 335–353.

Ventura Y, Eshel A, Pasternak D, Sagi M. 2015. The development of halophyte-based agriculture: Past and present. Annals of Botany 115: 529–540.

Vineis P, Chan Q, Khan A. 2011. Climate change impacts on water salinity and health. Journal of Epidemiology and Global Health 1: 5–10.

Voutsina N, Seliskar DM, Gallagher JL. 2015. The facilitative role of Kosteletzkya pentacarpos in transitioning coastal agricultural land to wetland during sea level rise. Estuaries and Coasts 38: 35–44.

Weston NB, Dixon RE, Joye SB. 2006. Ramifications of increased salinity in tidal freshwater sediments: Geochemistry and microbial pathways of organic matter mineralization. Journal of Geophysical Research 111: 41–14.

Weston NB, Giblin AE, Banta GT, Hopkinson CS, Tucker J. 2010. The effects of varying salinity on ammonium exchange in estuarine sediments of the parker river, Massachusetts. Estuaries and Coasts 33: 985–1003.

Weston NB, Vile MA, Neubauer SC, Velinsky DJ. 2011. Accelerated microbial organic matter mineralization following salt-water intrusion into tidal freshwater marsh soils. Biogeochemistry 102: 135–151.

White E Jr, Kaplan D. 2016. Restore or retreat? Saltwater intrusion and water management in coastal wetlands. Ecosystem Health and Sustainability 3: e01258–18.

Williams K, Ewel KC, Stumpf RP, Putz FE, Workman TW. 1999. Sea-level rise and coastal forest retreat on the West Coast of Florida, USA. Ecology 80: 2045–2019.

Yang J, Zhang H, Yu X, Graf T, Michael HA. 2018. Impact of hydrogeological factors on groundwater salinization due to ocean-surge inundation. Advances in Water Resources 111: 423–434.

Kate Tully (kltully@umd.edu) is an assistant professor of agroecology in the Department of Plant Science and Landscape Architecture at the University of Maryland, in College Park. Keryn Gedan is an assistant professor in the Department of Biological Sciences at the George Washington University, in Washington, DC. Rebecca Epanchin-Niell is a natural resource economist and fellow at Resources for the Future, in Washington, DC. Aaron Strong is an assistant professor at Hamilton College, in Clinton, New York. Emily S. Bernhardt is a professor in the Department of Biology at Duke University, in Durham, North Carolina. Molly Mitchell is a marine scientist at the Virginia Institute of Marine Sciences, at the College of William and Mary, in Williamsburg, Virginia. John Kominiski is an associate professor in the Department of Biological Sciences at Florida International University, Miami. Thomas E. Jordan is a senior scientist at the Smithsonian Environmental Research Center, in Edgewater, Maryland. Scott C. Neubauer is an associate professor in the Department of Biology at Virginia Commonwealth University, in Richmond. Nathaniel B. Weston is an associate professor in the Department of Geography and the Environment at Villanova University, in Villanova, Pennsylvania. Todd BenDor is affiliated with the Department of City and Regional Planning, at the University of North Carolina, in Chapel Hill.