CHAPTER FIVE

A global review of the impacts of saltwater intrusion on soils and ecosystems

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Abstract

Inland movement of seawater can change soil salinity in coastal landscapes, leading to losses in agroecosystem productivity, reduced resilience of farmland to climatic events, and rapid ecosystem transitions. Over 25 % of arable land globally is damaged and degraded from salinity, resulting in profit loss in billions of US dollars. The inland extent of saltwater intrusion (SWI) is growing over time, exacerbated by sea-level rise (SLR), drought, storm surge, and inadequate land and water management practices. Indeed, SWI represents the leading edge of coastal climate impacts, and understanding its current and future impacts and extent will facilitate better planning and adaptation. There are several ways to tackle this rapidly evolving challenge – resist, adapt or retreat. Resisting would involve continuing with business-as-usual management practices that might result in farmers losing millions of

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US dollars in annual profits. Retreat will involve a multidecadal sequence of actions, planning and considerable financial investment, whereas adaptation will need science-based policy interventions and nature-based solutions. In this review, we synthesize current literature on different aspects of SWI: impacts on soils and ecosystems, global extent along with dominant mechanisms, and mitigation and adaptation strategies.

1. What is saltwater intrusion?

Saltwater intrusion (SWI), sometimes referred to as seawater intrusion, is the landward movement of seawater where saline water migrates into freshwater systems (Allison et al., 1954; Kirwan et al., 2024; O'Donnell et al., 2024). Saltwater is increasingly reaching inland regions that have not adapted to salinity (Tully et al., 2019a), e.g. 40 km upstream of the Po River in Italy (Luo et al., 2024), or 160 km inland in Bangladesh in the dry season (Khanom, 2016), impacting soils in various ways. Salt-affected soils are most commonly found in arid and semiarid climates, such as the Western United States. However, SWI-affected landscapes are different as they tend to be both wet and saline, thus creating some unique challenges for the land managers. Traditionally, SWI was considered to be a process primarily involving groundwater where seawater encroaches into the freshwater reservoir, and moves up the soil layers, thus essentially being a bottom-up process. In this process, the seaward movement of freshwater prevents SWI under natural conditions and creates an interface between seawater and freshwater that resides near the coast or below the surface (USGS, 2019). This zone is known as the zone of dispersion or the zone of transition where the mixing of saltwater and freshwater occurs. However, with variable sea-level rise (SLR) rates across the global coastlines, land subsidence, excessive groundwater pumping, and more frequent and powerful coastal storms (Chen et al., 2020; Maliva, 2020; Siegel, 2020; Zhu et al., 2020), seawater is now increasingly affecting surface water and shallow groundwater (Desantis et al., 2007; Fagherrazi et al., 2019; Sallenger et al., 2012; Sweet et al., 2022), contributing to the top-down aspect of SWI (Kirwan et al., 2024). This two-directional, often simultaneous, process affects soil by changing its composition and structure (a process called soil salinization), and is leading to both short- and long-term ecosystem changes (Fig. 1), including crop health reduction, crop failure, upward marsh migration, freshwater wetland loss and ghost forest formation (Bhattachan et al., 2018; Kirwan and Gedan, 2019; Gedan and Fernandez-Pascual, 2019; Manda and Klein, 2019; Mondal et al. 2023; Small and Nicholls, 2003; Tully et al., 2019a,b; Ury et al., 2021b; Warnell et al., 2022; White et al., 2022; White and Kaplan, 2017).

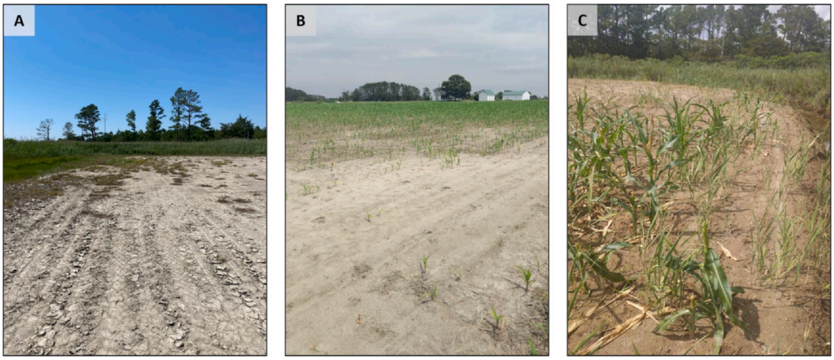

Fig. 1 Visual representation of different stages of salinization observed on farmland. (A) Extreme salinization with visible salt crust on bare soil, rendering it dry and unproductive, with no vegetation growth. (B) Moderate salinization, characterized by a mix of bare soil and sparsely spaced, stunted crops in the foreground, while healthier crops with higher density are seen farther away. (C) Early salinization phase showing sparse crop growth and encroachment of marsh vegetation from the edges of the farmland, indicating the presence of soil salinity.

Salts are ionic mineral compounds naturally present in soil and groundwater. Some salts, such as NaCl, are highly soluble in water, and can easily dissociate into ions in the soil, thus reducing germination and plant growth (Miller, 2021). The most common salts found in saline soil are the anions (e.g. HCO_3^{-1}, CO_3^{-2}, SO_4^{-2}, B^0, NO_3^{-1}, Cl^{-1}) and cations (e.g. K^{+1}, Na^{+1}, Mg^{+2}, Ca^{+2}) (Corwin, 2003; Miller, 2021; Shahid et al., 2018). In addition to seawater, prominent sources of soluble salts in soil include weathering of minerals, plant activities, atmospheric deposition, capillary rise of groundwater, seepage, fertilizer and irrigation (Osman, 2018). Weathering of the primary minerals in soil, such as quartz, micas, and feldspars, can release salt in soil. These salts can either accumulate in regions with low rainfall or low leaching, or can get washed away in regions with high rainfall. Plants can play an important role in removing salt from soil through nutrient uptake. In contrast, they can increase salt concentration through evapotranspiration (i.e. water removal from soil). Atmospheric deposition can happen in the form of salt condensation leading to raindrops or salt spray. This is particularly common near coastal regions where salts can come from the ocean or atmospheric dust (cyclic salts) or salt spray resulting from wave action. This atmospheric mechanism is different from the other coastal mechanisms for soil salinization where saline or brackish water affects land through tidal flooding, stormwater surges, or saline irrigation. Soil salinity can also increase in regions where fossil salts from prior marine sediments exist and are introduced in the system through their movement through groundwater, runoff or irrigation (Cardon et al., 2014; Miller, 2021). Irrigation can change soil salinity beyond crop tolerance in areas with poor drainage or low leaching, even when the salinity of the irrigation water is low. Another pathway for localized SWI impact is the application of fertilizer salts. When coupled with poor drainage, soil with high clay content, or drainage channels acting as conduits of seawater or stormwater, such localized SWI impacts can spread over a larger geographic region. In fact, SWI in coastal regions, e.g. the mid-Atlantic region of the USA, is particularly challenging as many of these mechanisms work in tandem. We further discuss these mechanisms in (Section 2).

While SWI into coastal freshwater resources has been studied for decades, reviews examining all aspects of SWI-impacted soils and resulting consequences for coastal ecosystems are limited. Understanding different stages of SWI and its impacts on the coastal ecosystems can inform us about effective mitigation strategies. As identified by Gibson et al. (2021), coastal lands don't often experience visible signs of SWI in the first stage. With gradual increase in soil salinity, plants start to display stress, with scarce appearances of visible salt patches (Mondal et al., 2023). In this stage, crop choice options start to get limited. Lands in an advanced SWI stage are almost non-productive, and non-profitable without careful crop selection. Agricultural lands might be completely out of production in the last SWI stage and can only support certain marsh species that can withstand regular flooding. Eventually, ecosystems experience a shift to a semi-aquatic system that can support wildlife habitats and thus offer potential for recreational or aesthetic uses and conservation easements. A gradually progressing SWI requires multi-phase landscape planning while also considering feasibility of such planning for long-term sustainability and the choices and preferences of the landowners.

In this review, we synthesize current knowledge on the myriad manifestations of SWI on natural and managed landscapes, along with some effective mitigation and adaptation strategies across the world. We start by briefly describing the primary mechanisms through which SWI impacts soil and ecosystems, then document the current geographic spread of SWI, followed by some of the potential adaptation strategies. This review aims to complement the most recent reviews on SWI in the context of SLR (O'Donnell et al., 2024), feedback regulating SWI (Kirwan et al., 2024; Tully et al., 2019a) and soil salinity assessments using remote sensors (Corwin and Scudiero, 2019).

2. SWI impacts on soils

Salt-affected soils can be mapped into three categories: Saline, sodic, and saline-sodic. Soils are considered saline when salt concentrations are so high that seed germination and plant growth are negatively affected, often leading to salty white crusts on the soil surface (Lamond and Whitney, 1992). Sodic (Alkali) soils are characterized by excess sodium absorbed to the extent that soil structure is negatively affected (Qadir et al., 2001). Saline-sodic soils are soils with high levels of both soluble and exchangeable sodium (Hanay et al., 2004). Saline, sodic, and saline-sodic soils can be classified using electrical conductivity (EC_e) and exchangeable sodium percentage (ESP). Saline soils exhibit high EC_e and low ESP; sodic soils exhibit high ESP and low EC_e; saline-sodic soils exhibit high EC_e and ESP (Wicke et al., 2011). Global predictions of salinity types have changed over the decades, with Squires and Glenn (2004) estimating 40 % saline and 60 % sodic; Wicke et al. (2011) estimating 60 % saline, 26 % sodic, 14 % saline-sodic; and FAO (2021) estimating 85 % saline, 10 % sodic, and 5 % saline-sodic in the topsoil, and 62 % saline, 24 % sodic, 14 % saline sodic in the subsoil.

For primary soil salinization, the sources of soluble salts arise from various factors (Daliakopoulos et al., 2016; Hassani et. al., 2021; Sandhu and Qureshi, 1986). These include rainfall, where oceanic salts are deposited through precipitation; aeolian processes which entail the dry deposition of oceanic salts, and the physical or chemical weathering of parent rock materials. Additionally, the transport of accumulated salts from saline geological deposits by streamflow or shallow underground waters serves as an added contributor to primary salinization. Conversely, in anthropogenic or secondary soil salinization, human activities play a significant role (Rengasamy, 2006; Sandhu and Qureshi, 1986; Thorslund et al., 2021). This involves interventions such as irrigation with brackish or saline water, the rise in water tables due to inadequate land and water management practices, the intrusion of seawater into coastal aquifers owing to rising sea levels or excessive exploitation of freshwater underground resources, and the excessive application of salt fertilizers (Hassani et. al., 2021). The main cause of salt accumulation in agricultural soil is water loss via evapotranspiration, wherein water evaporates from the soil surface and transpires through plants, leaving salt behind.

When saltwater floods freshwater and terrestrial soils, SWI occurs at the top of the soil profile and moves downward. The depth of intrusion during a saltwater flooding event depends on the saltwater volume and the duration of inundation. Wetter soils have less pore space for saltwater to percolate, while dry soils allow more saltwater to move downward. Additionally, the soil properties themselves determine the rate at which water moves through the soil profile. Soils with higher clay content are less permeable than those with high sand content, resulting in slower water movement in clay soils. Saltwater movement can also start at the bottom of the soil profile and move upward in the form of saline groundwater. For example, rising sea levels can impact freshwater aquifers and increase the salinity of the groundwater. Groundwater can move upward through the soil profile through capillary action. As water evaporates at the soil surface, saline groundwater moves upward through the soil profile to replenish the evaporating water. The soil salinization mechanisms are different in the arid and semi-arid areas where salinization typically arises with inadequate irrigation or rainfall for salt leaching, poor drainage, shallow water tables, upslope recharge, downslope discharge, and natural saline sub-soils from marine deposits (Corwin, 2019).

There is limited information regarding how the duration and frequency of saltwater inundation impacts soil salinity. Typically, ecosystems retain 'memory' of salinity events even when pulsed with freshwater inputs through rainfall or river flow. Such memory will increase the ionic strength of a system over time, ultimately increasing salinity (Rice, 2012; Ross et al., 2015) and tipping the ecosystem into a new state, e.g. from freshwater forested wetland to brackish wetland. However, salinized soils may recover more fully as the time between saltwater inundation events and freshwater input increases. The long-term impact on soil productivity is dependent on the duration of inundation, soil texture, and soil moisture conditions before inundation occurs.

Soil texture, saturated hydraulic conductivity, effective cation exchange capacity, and water table fluctuations determine the retention of salts in uplands soils due to saltwater inundation. Soils with higher sand content are likely to have good drainage and retain less salt, while soils with higher clay content are less well-drained and more likely to retain salt. Sandy soils have a lower cation exchange capacity, a lower effective cation exchange capacity, a higher saturated hydraulic conductivity, and typically have reduced impacts from SWI. Saturated hydraulic conductivity is a measure of the rate at which fluid moves through the soil; hence soils with a high saturated hydraulic conductivity drain well and are less susceptible to salinization. If the number of exchangeable cations in the soil is high, the soil more readily retains soluble salts. Thus, clay soils retain salts and drain more slowly than sandy soils due to their low saturated hydraulic conductivity and higher cation exchange capacity. A comprehensive review of the biogeochemical effects of SWI can be found in Tully et al. (2019a).

3. SWI impacts on ecosystems

SWI changes soil chemistry through three co-occurring processes: increase in ionic strength, alkalinization and sulfidation (Tully et al., 2019a). SWI into freshwater systems increases ionic strength irrespective of specific ionic composition, leading to osmotic stress for plants and seeds. The ionic composition of seawater determines the amount of base cations, such as sodium, calcium and magnesium. This can change phosphorus availability, affect ecosystem carbon dynamics, and/or impede water infiltration and drainage, thus reducing plant growth (Ardon et al., 2016; Chambers et al., 2011; Rengasamy et al., 1984; Weston et al., 2006). Chronic SWI increases sulfide through the process of sulfidation and may lead to increased toxicity through H_2S formation, depending on iron availability. Since seawater is a complex solution, SWI impacts on the natural and managed landscapes depend on the dominant mechanism and land use/cover history (Gedan et al., 2020).

3.1 Natural landscapes

The combined threats of global mean SLR rate of 2.8–3.2 mm/year (Church and White, 2011) and far-reaching inundations associated with both SLR (Sweet et al., 2022) and more frequent hurricanes/coastal storms (Donnelly et al., 2015; Knutson et al., 2010; Mendelsohn et al., 2012) are driving ecosystem changes along coastal landscapes (Fagherazzi et al., 2019; Herbert et al., 2015; Kirwan and Gedan, 2019; Mondal et al., 2023; White et al., 2022). The most prominent reminders of such changes are expansive ghost forests surrounded by marshlands and abandoned farmlands now dominated by bare soil and wetland plants, capturing the land cover history of upland to wetland conversion (Kirwan and Gedan, 2019). In addition to land conversion, within-class modification can also result from increased salinization, e.g. tidal freshwater marsh to saltmarsh, often resulting in a change in gross ecosystem production (Neubauer, 2013). Depending on the plant species diversity in a freshwater system, tidal marshes can be resistant and resilient to occasional pulses of brackish water, but might not withstand chronic salinization (Thompson et al., 2024).

While the marshlands that replace these coastal forests are of high ecological importance (Gedan et al., 2009), such land cover conversions are often undesirable to landowners and result in property devaluation (Bin and Polasky, 2005; Field et al., 2017). Yet, such conversions are inevitable in lands that are experiencing higher than average land submergence (Sallenger et al., 2012), such as the mid-Atlantic region stretching from North Carolina to Massachusetts. As such, the Delaware Bay has already witnessed widespread death in hardwood and cedar forests (Smith, 2013), and over 400 km^2 of uplands in the Chesapeake Region have been converted to tidal marsh since the mid-1800s (Schieder et al., 2017). Ghost forests have been reported from Canada and several other US states, including Florida, Georgia, Louisiana, and South Carolina (Conner et al., 2007; Craft, 2012; Langston et al., 2017; Raabe and Stumpf, 2016; Robichaud and Bégin, 1997). Upland forests are particularly vulnerable to the compound effects of salinization and inundation (Barrett-Lennard, 2003; Conner, 1994; Conner et al., 2007; Pezeshki et al., 1990). Of particular importance is soil hydrology that can either alleviate or enhance salinization, thus playing a crucial role in coastal forest resilience in the context of SWI (Nordio et al. 2024). A comprehensive review of salinity-driven changes in upland forests can be found in Kirwan and Gedan (2019).

3.2 Managed landscapes

The most prominent SWI impacts on the managed landscapes appear through reduced crop health or a total crop failure due to a soil salinity level beyond crop tolerance. The timing of salt stress is of particular significance as plants tend to be more sensitive in the early growth stages (Mbarki et al., 2020), even though salt stress can negatively impact plants throughout the growth stages, including germination, seedling emergence and reproduction (Carillo et al., 2011; Katembe et al., 1998). Salinity can adversely affect plant height, leaf weight, green leaf area, and shoot and root growth (EL Sabagh et al., 2021).

Since crops have a range of salt tolerance levels (see Tanji and Kielen, 2002 for the range of salinity tolerance of different crops), SWI impacts on crop health vary across the world. Rice is a popular crop among many coastal countries and is known to have moderately high salt tolerance of EC_e threshold around 3 dS/m (Heenan et al., 1988; Hossain and Li, 2024; Khan et al., 1997a, 1997b; Khanom, 2016; Ologundudu et al., 2014). Yet, crop failure and yield reduction have been reported in highly saline conditions (Dam et al., 2019; Dewi et al., 2022). Especially in Bangladesh, salinity impacts on agricultural losses have been documented for decades (Dasgupta et al., 2014; Karim et al., 1990; Mahmood et al., 2010; Petersen and Shireen 2001; Thomas et al., 2013), often leading to a land use conversion to shrimp farming (Ahmed and Ambinakudige, 2023; Hossain and Li, 2024). Corn (~1.8 dS/m) and some of the bean varieties are known to have lower salt tolerance compared to rice (Farooq et al. 2015; Tanji and Kielen, 2002) and are likely to get negatively impacted in highly saline conditions as well. Salt tolerance of other crops, such as haricot bean, might be determined by their specific cultivars (Tessema et al., 2022). Soybean (5 dS/m), barley (6 dS/m), sorghum (6.8 dS/m), and wheat (6–8.6 dS/m) have higher salt tolerance than rice and corn and are often recommended as alternate crop types in saline condition. However, wet soils and waterlogging that are common in coastal regions will negatively impact certain crops, such as wheat (Makhdum and Ashfaq, 2008). High salinity might also result in discontinuation of certain crops, e.g. oilseed, sugarcane, and jute (Khanom, 2016). In addition to land use conversions (e.g. agriculture to aquaculture), land abandonment is particularly prevalent in highly saline regions across the world (Ardón et al., 2013; Bhattachan et al., 2018; Da Lio et al., 2015; Khanom, 2016; Vanderplank et al., 2014). In suitable conditions, saltmarsh species migrate into these abandoned salinized farmlands (Gedan and Fernandez-Pascual, 2019).

4. Current geographic extent of saltwater intrusion

Soil salinization is a global problem affecting almost every continent to varying degrees (Fig. 2). Estimates of salt-affected soils range from 833 million to 1.1 billion hectares globally (FAO, 2021; Ivushkin et al., 2019; Massoud, 1976; Squires and Glenn (2004); Szabolcs, 1987; Wicke et al., 2011). Li et al. (2014) estimates the extent of salt-affected soils in coastal ecosystems, which is likely driven by SWI rather than mineral weathering as in arid and semi-arid ecosystems. They estimate 230 Mha of coastal salt-affected soils globally with major regions including Central and North Asia along the Black, Caspian, and Mediterranean Sea; Australia; East and Southeast Asia along the Yellow and East China Sea; in South America, Mexico, and Cuba along the Gulf of Mexico; and along large river estuaries such as Congo, Ganges, Mekong, and Indus River. The scope of the problem is rapidly expanding, with Ivushkin et al. (2019) estimating the global extent increasing from 915 Mha in 1986 to 1069 Mha in 2016.

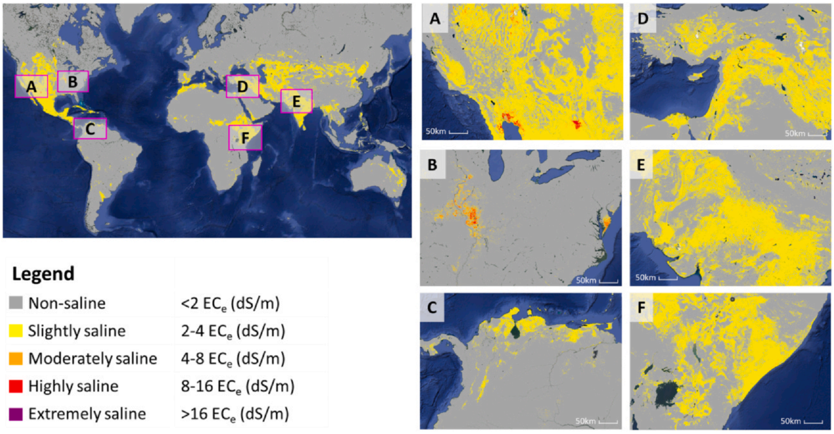

Fig. 2 Global Soil Salinity Maps (GSSM) showing spatial distribution of soil salinity levels in coastal areas. The GSSM dataset was derived by using 7 soil properties, thermal infrared imagery and the ECe point data from the WoSIS database (Ivushkin et al., 2019). The salinity levels are categorized as non-saline, slightly saline, moderately saline, highly saline, and extremely saline based on electrical conductivity (EC) measured in deciSiemens per meter (dS/m). Panels A-F show: (A) West Coast of the United States, (B) East Coast of the United States, (C) Northern coast of South America, (D) Mediterranean Coast, (E) Eastern coast of the Arabian Sea, and (F) Western coast of the Arabian Sea. Map lines delineate study areas and do not necessarily depict accepted national boundaries.

According to the existing literature, Asia has the largest extent of salt-affected soils ranging from 315 Mha to 332 Mha, with North and Central Asia being the most affected (FAO, 2021; Ivushkin et al., 2019; Massoud, 1976; Squires and Glenn, 2004; Szabolcs, 1987; Wicke et al., 2011). Africa is also largely affected with estimates ranging from 80 Mha to 322 Mha (FAO, 2021; Ivushkin et al., 2019; Massoud, 1976; Squires and Glenn, 2004; Szabolcs, 1987; Wicke et al., 2011). Intrusion is the most prominent in coastal countries where intense land development, irrigated agriculture, and overuse of groundwater resources are occurring. A review of local studies providing evidence of SWI, and its impacts are summarized below for affected countries. The omission of a country or region does not necessarily mean that intrusion is not occurring but could be due to the lack of published evidence of intrusion.

4.1 Africa

Saltwater intrusion in Africa has mainly been documented in Northern and Eastern African countries. A review of SWI in Northern African countries concluded that intrusion is occurring along the Atlantic Ocean and Mediterranean Sea at an alarming rate, with Mauritania, Algeria, Morocco, Tunisia, Libya, and Egypt all being affected (Agoubi, 2021). Northern Africa is an arid region prone to salinization because of high incidents of drought and the overuse of limited groundwater resources (Agoubi, 2021). Intrusion has been well documented in Morocco, which is undergoing population and economic growth, leading to increases in irrigated agriculture, tourism, and industry (Arjdal et al., 2024; Ez-zaouy et al., 2022; Ez-zaouy et al., 2023). Development coupled with climate change-induced decreases in rainfall has led to the overuse of groundwater, facilitating the spread of SWI up to 2500 m inland (Ez-zaouy et al., 2022; Ez-zaouy et al., 2023). Algeria, which has 25 % of its population living in coastal areas, has also been severely affected by SWI, with one study finding intrusion up to 1600 m inland (Bouderbala et al., 2016). Similarly, Tunisia has experienced widespread intrusion from the Mediterranean Sea, with the use of saline water for agricultural irrigation contributing to salt-affected soils (Ayari et al., 2023; Zghibi et al., 2022). Intrusion from the Mediterranean Sea has also impacted Egypt due to increased groundwater pumping, spreading saltwater up to 100 km inland leading to increased soil salinity (Abd-Elaty et al., 2021; El-Sayed et al., 2023). Agoubi (2021) notes that the understanding of saltwater intrusion in Mauritania and Libya is limited due to lack of expertise and inadequate data.

A review of SWI in countries along the East African coastline, which border the Red Sea and Indian Ocean, found evidence of intrusion in Kenya, Tanzania, and Djibouti, while noting a lack of studies in Sudan, Eritrea, and Somalia, as priorities are placed more strongly on economic and political stability (Idowu and Lasisi, 2020). In Kenya, population growth has increased water usage, which has led to the unsuitability of some water bodies for irrigation, leading to soil salinization and decreased crop yields (Chalala et al., 2017; Kathuli et al., 2013; Oiro and Comte, 2019). Studies have also found SWI occurring along the Indian Ocean in Tanzania due to overexploitation of groundwater and decrease in rainfall, leading to lower corn and rice production up to 5 km inland (Ligate et al., 2017; Mtoni et al., 2013). While Djibouti has a small coastline with limited SWI reports, evidence of SWI impacts have been documented, caused by high water pumping and low aquifer recharge (Houssein and Jalludin, 1996; Idowu and Lasisi, 2020). The city of Maputo, Mozambique, along Maputo Bay has experienced SWI due to an increasing population and overuse of groundwater due to water shortages (Trasviña and Alberto, 2018). While not directly caused by SWI, increasing soil salinization greatly affects the region of East Africa, specifically Ethiopia, leading to poor water quality and losses in crop production (Tessema et al., 2022). Soil salinity is increasing due to crop irrigation and poor drainage infrastructure, making Ethiopia the country most affected by soil salinization in Africa, and seventh most affected globally (Tessema et al., 2022).

While comparatively less documented in published literature, there is evidence of intrusion in Western African countries. In Western Africa, Nigeria has experienced SWI from the Gulf of Guinea, leading to groundwater challenges and a decrease in crop productivity (Adepelumi, 2008; Callistus et al., 2024). In the coastal town of Limbe, Cameroon, Rigobert and Fru (2020) found that 23 % of the town's drinking water was contaminated by SWI. Multiple analyses of the Western African country Benin have identified the vulnerability of the region to intrusion due to decreasing groundwater levels and proximity to the Gulf of Guinea (Agossou et al., 2022; Hounsinou, 2020). Neighboring Benin to the west, Togo has experienced SWI due to both farming and intrusion of inland saline deep-water aquifers (Akouvi et al., 2008). A study of well-water in Ghana found that intrusion from the Gulf of Guinea was likely impacting the water of 5 of 40 wells sampled, primarily located within 2 km from the coast (Zume et al., 2021). In The Gambia, saltwater intrusion is facilitated by the River Gambia, threatening food production and the livelihood of rice farmers (M'koumfida et al., 2018).

4.2 Asia

A study on the potential impacts of SWI on human health found Southeast Asia to be the most vulnerable region globally (Mueller et al., 2024). The Mekong River Delta in southern Vietnam presents one of the most extreme cases of SWI with soil salinity greatly affecting rice production in the region (Thach et al., 2023). Exploitation of water for shrimp aquaculture, SLR, storm surge from tropical cyclones, the interconnectedness of canals, and dams all contribute to increasing salinization of soils (Xiao et al., 2021). Droughts in 2020 amplified the problem of SWI, leading to salinization up to 110 km inland (Park et al., 2022). SWI is a widespread issue in Southeast Asia, also affecting Indonesia, Thailand, Cambodia, the Philippines, Myanmar, and Malaysia (Phan et al., 2021; Purnama and Marfai, 2012; Samsuddin Sah et al., 2021; Swe and Ando, 2016; Taclan et al., 2024; Vann et al., 2020).

The three coastal countries of Southern Asia, India, Bangladesh, and Pakistan, have all been affected by SWI. Intrusion has been well documented along the 7500 km Indian coastline stretching across nine southern states and bordering the Indian Ocean, Bay of Bengal, and Arabian Sea (Prusty and Farooq, 2020). Rapid urbanization and a large population in India make it the largest user of groundwater globally, with high levels of irrigation, leading to widespread intrusion in these coastal regions (Siddha and Sahu, 2020). Bangladesh is also greatly impacted by SWI, decreasing crop yields and worsening food insecurity (Mahmuduzzaman et al., 2014). The influence of the monsoon rains makes SWI a seasonal problem, reaching a maximum during March and April before the monsoon season begins (Mahmuduzzaman et al., 2014). In Pakistan, SLR has contributed to extensive coastal SWI, leading to the destruction of essential mangrove habitats (Aeman et al., 2023), and loss in wheat productivity (Makhdum and Ashfuq, 2008).

The Arabian Peninsula, Earth's largest peninsula with a population of 153 million people in 2020, bordered by the Arabian and Red Seas, is experiencing damaging SWI due to irrigation and sparse rainfall leading to slow recharge of aquifers. Small-scale farming irrigation in Oman has facilitated intrusion from the Arabian Sea and Gulf of Oman, leading to damaged soils and abandoned farmlands (Grundmann et al., 2016). The United Arab Emirates (UAE) has also resorted to the abandonment of wells and farmlands due to SWI induced by agricultural water demands (Sherif et al., 2012). Yemen, another nation largely dependent on groundwater irrigation for agriculture, has largely been affected due to rapidly decreasing groundwater (Abdulqader and Senosy, 2012; Akhtar et al., 2023). Along the Red Sea, SWI has also been detected in Saudi Arabia due to over-pumping from shallow water wells and geologic conditions such as highly permeable coastal rocks (Ibrahim et al., 2024; Mogren and Mogren, 2015).

Along the Eastern Mediterranean coast, Beirut, Lebanon was found to have groundwater salinity as high as 30,000 parts per million (ppm) of total dissolved solids (TDS). SWI in the area can be attributed to groundwater over-pumping due to rapid urbanization in addition to the abundance of highly permeable fractured limestone (Alameddine et al., 2018). The coastal Gaza Aquifer, which is the sole source of water resources for one of the most densely populated cities, has also experienced salinization due to overexploitation of groundwater and irrigation (Mushtaha and Walraevens, 2023). Use of saline water for irrigation contributes to the degradation of cropland, threatening agriculture in this region (UNEP, 2009). Along the same coastal aquifer, Northern Israel has experienced SWI from the Mediterranean Sea (Sivan et al., 2005). Areas in Northwest Syria, along the Mediterranean Coast, also have increasing salinity due to the use of groundwater for irrigation, development, and tourism, leading to a loss of suitable land for fruit cultivation (Allow, 2011). Northern Iran, along the Caspian Sea has undergone increasing water salinity due to the overuse of water for irrigation, leading to conditions unsuitable for agriculture (Ebrahimi et al., 2016).

Several major cities in Northern China along the Bohai and Yellow Sea have been experiencing SWI as early as 1964 (Guo, 2012; Xuebin et al., 1995). A 2003 study estimated the area of intruded land along the Bohai Sea to be 2457 km^2, up from 937 km^2 in the 1980's, with Laizhou City in Shandong province being the most heavily affected (Shi and Jiao, 2014; Xuebin et al., 1995). Intrusion is high in the area due to the naturally high permeability aquifers, low rainfall, and rapid development (Shi and Jiao, 2014). Severe SWI has also occurred in the Changjiang Estuary, facilitated by discharge from the Yangtze River, threatening drinking water supplies in Shanghai (Xu et al., 2018; Zhu et al., 2020). In South Korea, early SWI studies in the 1980's focused on Jeju Island, south of the mainland, as intrusion posed a major risk to the island's freshwater resources (Jeen et al., 2021; Kim et al., 2003). More recent studies have identified saltwater intruding further inland along the Western Coast bordering the Yellow Sea (Jeen et al., 2021; Park et al., 2005). The Island of Japan has also been affected by SWI, with the 2011 tsunami creating a unique source of intrusion, as 561 km^2 of land up to 1 km inland were submerged by sea water (Liu and Tokunaga, 2019).

4.3 Australia

Australia is a major global SWI hotspot, in part due to a growing population and low rainfall (Werner, 2010). The Murray-Darling Basin in Eastern Australia covering most of New South Wales and some of Queensland and covering 14% of the total Australian landmass has experienced vertical intrusion caused by the widespread removal of trees and perennial grasses (Herbert et al., 2015; Walker et al., 1993). In North Queensland, SWI has been shown to increase with groundwater pumping, which is heavily used in sugarcane farming (Narayan et al., 2003). SWI has also been identified on the Western Coast of Australia in the City of Perth, caused by declining rainfall, land development, increasing water usage, and SLR (Costall et al., 2020). In Northern

Australia, saltwater has intruded into previously freshwater wetlands along expanding tidal creeks, negatively impacting 17,000 ha of vegetation (Mulrennan and Woodroffe, 1998).

4.4 Europe

A 2004 report published by the European Environment Agency found evidence of widespread SWI in Europe due to over-exploitation of freshwater resources (Scheidleder et al., 2004). In this continent, SWI is the most prevalent along the Mediterranean Sea in Greece, Italy, Spain, and Turkiye (Scheidleder et al., 2004). However, evidence of SWI was also found in countries along the Baltic Sea including Denmark, Poland, Latvia, and Estonia (Scheidleder et al., 2004).

Due to freshwater pumping for coastal agriculture, the effects of SWI have been felt across Greece (Apostolakis et al., 2016). Because of its arid climate, the Southern Coast of Greece is the most vulnerable to this process, with intrusion progressing far inland (Daskalaki and Voudouris, 2008). However, intrusion has been reported along the Northern Coast and on Greek Islands such as Crete (Papadopoulou et al., 2005; Petalas et al., 2009). Antonellini et al. (2008) found evidence of SWI along the Po Plain of Italy due to land subsidence below sea level and rivers and canals facilitating sea water encroachment. This process, primarily driven by natural factors and exacerbated by overexploitation of water, gas extraction, and drainage to reclaim farmlands, has significantly reduced crop yields in the Po Plain (Luo et al., 2024). A publication from the European Environment Agency found SWI to be occurring in 58 % of Spain's coastal hydro-ecological units (European Environment Agency, 2016). Southeastern Spain has a semi-arid climate, with little rainfall in the summer months, leading to a reliance on groundwater and facilitating SWI (Pulido-Leboeuf, 2004). Southeastern Spain also has the largest global concentration of agricultural greenhouses, leading to further overexploitation of groundwater and increasing soil salinity (Aznar-Sánchez et al., 2019). SWI has been found along both the Mediterranean and the Black Sea in Turkiye, with saline water used in crop irrigation being a major contributor to saline soils (Arslan and Demir, 2012; Demirel, 2004).

SWI along the Black Sea is also occurring, but less evidenced in the current literature. Intrusion has been reported across Poland due to multiple factors including elevation below sea level, Karst aquifers susceptible to intrusion, reliance on groundwater, and buried valleys which create preferential flow for saline water (Duque et al., 2022). One study found

SWI to be present in the Northern coastal aquifer of Władysławowo, Poland due to overuse of groundwater from a deep aquifer composed of sand and gravel for a fish processing plant between 1964 and 2014 (Pruszkowska-Caceres et al., 2018). A study in Liepāja, along the Western Coast of Latvia found the presence of SWI due to groundwater pumping beginning in 1961 (Bikše and Retike, 2018). Near Tallinn, the capital of Estonia in the North of the country, SWI has been found and hypothesized to be caused by vertical intrusion from an underlying crystalline aquifer (Raidla et al., 2019).

4.5 North America

Coastal regions in Canada, the United States, and Mexico have all experienced SWI to varying degrees. While there have been limited studies in Canada, evidence of minor SWI has been reported in New Brunswick and Prince Edward Island (Carr, 1969; Green, 2012). In the USA, a study on ~250,000 coastal wells found that 15 % along the West coast, 23 % along the Gulf Coast, and 35 % along the East Coast were below sea level, indicating a vulnerability to SWI (Jasechko et al., 2020). One major area of intrusion is along the Delaware Bay and Atlantic Ocean, where intrusion began in the 1890's due to groundwater pumping which lowered groundwater levels by up to 30 m, leading to lateral encroachment (Barlow and Reichard, 2010). The Floridan aquifer system spanning across Florida, Georgia, South Carolina, and Alabama with an area around 250,000 km^2 has also experienced widespread SWI due to vertical intrusion caused by breaching of a permeable saline zone called the Fernandina permeable zone (Barlow and Reichard, 2010). Along the Pacific coast, Central and Southern California have experienced SWI due to groundwater pumping for agriculture and urban development (Barlow and Reichard, 2010). SWI into inland freshwater wetlands has also been widely documented along the East and Gulf coasts, primarily caused by vegetation clearance, irrigation, river regulation, mining, and use of de-icing salts (Herbert et al., 2015). In Mexico, 10 out of 17 coastal states along the Pacific Ocean and Gulf of Mexico were found to be experiencing varying levels of SWI due to overexploitation of groundwater (Cardoso, 1993).

To a lesser extent, SWI has been documented along the Caribbean Sea due to human activities. For example, the ships moving through the heavily hydrologically altered Panama Canal has led to intrusion into Gatun Lake, leading to a potential loss of salt-intolerant biodiversity (Wijsman, 2013). Jamaica, an island heavily used for sugarcane cultivation experienced SWI due to irrigation, leading to the abandonment of many plantations (Howard and Mullings, 1996). The island of Cuba has long been affected by SWI, since the use of irrigation for rice cultivation in the 1950's (Núñez et al., 2004).

4.6 South America

SWI in South American countries, such as Brazil, Ecuador, and Chile, is often facilitated by rivers, with the extent of intrusion controlled by tides, river discharge, and wind (Garcés-Vargas et al., 2020; Mora et al., 2021; Ospino et al., 2018; Toste et al., 2017). Intrusion along rivers can lead to contaminated drinking water and decreased agricultural productivity (Garcés-Vargas et al., 2020; Mora et al., 2021). Intrusion can also be found in wetland ecosystems. Along the Northern Coast of Brazil, SWI has been identified by the inland migration of salt-tolerant mangrove forests since 1984 (Visschers et al., 2022). Intrusion within this primarily non-urban region within the Amazon Rainforest has been attributed to SLR and hydrological alterations caused by buffalo grazing (Visschers et al., 2022). Arid wetlands in Argentina have also experienced SWI due to increasing high tide waters (Alvarez et al., 2014; Pousa et al., 2006). A 2021 report found salinization of agricultural soils caused by irrigation in Venezuela, Brazil, Peru, and Argentina (Lavado, 2021). Agriculture in arid and semi-arid regions, such as Peru and Chile have experienced SWI due to groundwater depletion and low rainfall (Narvaez-Montoya et al., 2022; Vera et al., 2021).

5. Potential mitigation and adaptation strategies

Addressing the challenges posed by SWI and mitigating its impacts requires a multi-faceted approach. It is crucial to recognize that actions aimed at protecting, sustainably managing, and restoring salinized landscapes can effectively address societal challenges, without compromising human well-being and biodiversity (O'Donnell et al., 2024). This can be achieved through various methods, including nature-based solutions (Davis et al., 2024), such as utilizing natural barriers like mangroves, salt marshes, and seagrass meadows, alongside technological, bioengineering, and other human-developed strategies (Barbier, 2014; IUCN, 2016; Tarolli et al., 2024; Temmerman et al., 2013). Since SLR is a significant and long-term contributor to the problem of SWI, adaptation strategies designed to address SLR will also aid in managing SWI (Akter et al., 2022). As such, adapting to SWI along the coasts begins with protecting and managing the first natural barriers that saltwater encounters: wetlands (Gedan et al., 2011; IPCC, 2019; Hilmi et al., 2017; White and Kaplan, 2017). Restoring wetlands is an effective nature-based solution for addressing salinity intrusion in coastal ecosystems. Wetlands naturally retain and filter water, recharge groundwater, and prevent saline water infiltration. They act as natural reservoirs by capturing freshwater during heavy rainfall and gradually releasing it (Johnston and A, 1991), replenishing groundwater aquifers, raising water tables, and developing a counteracting pressure against the intrusion of saline water. For example, in Ben Tre Province in Vietnam, 50% of mangrove forests were cleared between 1998 and 2015, leading to SWI-driven rice to aquaculture conversion (K Veettil et al., 2019). After recognizing the crucial role of mangroves in mitigating salinization, various restoration projects were initiated in the Vietnamese Mekong Delta. These efforts led to an impressive increase of 11,184 ha of mangrove area from 2015 to 2020 (Tinh et al., 2022). By reestablishing mangrove forests and integrating them into agricultural practices, these initiatives aim to restore natural defenses against salinity and enhance agricultural productivity in the region.

Another effective strategy for mitigating potential SWI is careful design of buffer zones around farm fringes and along canals and ditches (Das, 2023). These natural or artificial buffer zones, planted with salt-tolerant native species, act as barriers between freshwater aquifers and saline water bodies, with roots preventing erosion, absorbing excess water, and maintaining the buffer zone integrity. Marsh vegetation will likely prevent further salt accumulation in comparatively less salinized regions (Poulter et al., 2008). However, these salt-tolerant plant species must be carefully selected to match specific climate zones and site conditions. Furthermore, implementation of buffer zone strategy requires well-informed and standardized practices as different regions might have different needs. For example, due to its sensitivity to salinity, the corn-focused agricultural economy is not suitable for many SWI-affected coastal fields across the coastal US states of Delaware, Maryland, and Virginia (Mondal et al., 2023). One potential adaptation strategy is a controlled conversion of these farm fringes into marsh that can support wildlife or act as a barrier to encroaching seawater (Guimond and Michael, 2021). While such transitions are vital to sustainable solutions, the fate of such coastal frontier zones will be shaped by the salinity gradient across these evolving landscapes. In highly salinized regions, halophytes might contribute to further soil salinization through continued and efficient water uptake in brackish soils—an example of a positive feedback loop (Sternberg et al., 2007; Wendelberger and Richards, 2017). The human dimensions of these global environmental changes must also be carefully considered, especially when historically disadvantaged and disproportionately vulnerable communities are involved (O'Donnell et al., 2024). Buffer zones can limit productive farming areas and impact farmers' income due to the resource-intensive maintenance they require. Partial or complete conversion of farmlands to marshlands can also negatively affect nearby property values, thus making it an often-undesirable option among landowners (Field et al., 2017). However, addressing these challenges through subsidies and clear, region-specific guidelines is pivotal to maximize the ecological and agricultural benefits (Tarolli et al., 2024).

In a recent study for the Delmarva Peninsula, Mondal et al. (2023) estimated an annual profit loss of US$39.3 million during 2011–2013, rising to US$70.7 million during 2016–2017, within a 200 m buffer around visible salt patches, for a business-as-usual corn-soybean rotation. This underscores the need for strategies to prevent, manage, and reverse salinization (when applicable). The development of salt-affected soils is generally a gradual process, often taking years before salinity levels become sufficiently high to impair crop growth. Similarly, the reclamation of these soils is a long-term process, requiring sustained efforts over extended periods to restore soil health and productivity. Before implementing adaptation strategies, it is crucial to first understand the nature of salinization, including its source, extent, and the type and concentration of salts in the soil. Once the soil characteristics and salinization history are analyzed, strategies for reclaiming salt-affected lands must be tailored to whether the soil is saline, sodic, or saline-sodic (Cuevas et al., 2019).

To reclaim saline soils and mitigate the impact of salinization, it is essential to manage salt levels in the soil through leaching (Young, 2005). Leaching involves applying large quantities of low to moderately saline water for discharging salts from the upper horizons to the lower soil layers. The reclamation rate depends highly on the amount of water passing through the soil profile out of the root zone, also called the leaching fraction. It is important to ensure adequate drainage in the soil to accommodate an optimal leaching fraction, and to consider the soil's hydraulic properties, particularly in heterogeneous soils where texture significantly impacts water movement and salt distribution (Li et al., 2012). Coarse-textured (sandy) soils require less management for leaching due to high infiltration rates, allowing more water to be applied quickly. Fine-textured (clay) soils need more careful management, as lower infiltration rates limit water application, increasing the risk of water logging and runoff. In addition to adequate drainage, sufficient soil moisture is a crucial aspect of reclamation efforts to avoid re-salinization. Salts raise the osmotic potential of soil water, making it more difficult for plants to absorb water in saline conditions. Consequently, saline soils need to be maintained at higher moisture levels than non-saline soils to provide sufficient water for crops, often requiring more frequent, lower-volume irrigations. Drip irrigation is an effective method for sustaining high soil moisture in saline soils. When salts originate from a shallow water table, it is necessary to lower the water table through drainage before reclamation can occur. In some cases, lowering the water table may not be cost-effective, making alternative crops or land uses a more viable option.

Saline-sodic soils must first be treated as sodic soils, requiring calcium to address sodium issues, followed by leaching to remove salts. High EC irrigation water can help maintain soil structure, improve water infiltration, and prevent sodium dominance, but it generally does not benefit crop production. Many saline-sodic soils result from natural events beyond the landowner's control, and reclamation may not restore productivity. In such cases, using salt-tolerant vegetation (e.g., pasture) is recommended to maintain soil cover.

Sodic soils are often the costliest to reclaim, and full remediation typically requires years or even decades of effective soil and crop management. Reclamation of sodic soils requires addition of soluble calcium, followed by leaching of sodium from the root zone (Dandekar and Chougule, 2010). Gypsum (calcium sulfate) is the primary material for supplying calcium in sodic soil reclamation due to its high calcium content, solubility at high pH, and lack of interfering compounds. While calcium nitrate and calcium chloride can also be used, they are generally more expensive and may negatively affect plant growth and the environment. The sulfate in gypsum poses little risk to crops, even when applied more than plant requirements. Tillage is often needed to break up sodium-rich layers and incorporate amendments into the soil. Coarse organic materials that decompose slowly, such as straw (Pang et al., 2010), cornstalks, sawdust, or wood shavings, can enhance soil structure and infiltration when combined with other reclamation practices.

Various other strategies such as using straw (Li et al., 2023), microbial and biomolecular solution (Ashraf and Aisha, 2009; Etesami and Maheshwari, 2018), crop adaptation (Busoms et al., 2023), organic fertilizers (Xu et al., 2014) and strategic water management (Li et al., 2022) have been adapted globally to mitigate the impact of SWI on agriculture productivity and ecosystem health. Straw incorporation is effective in mitigating soil salinization by increasing soil organic matter, which provides carbon for microorganisms that decompose organic material, release nutrients, and improve soil structure (Liu et al., 2017; Liu et al., 2021). This practice enhances water-holding capacity, reduces salt accumulation in the root zone (Amini et al., 2016), and promotes the formation of stable soil aggregates, aiding in salt leaching and improving root penetration and nutrient uptake (Mengdie et al., 2021). Microbial-based solutions, such as introducing halotolerant microorganisms, can enhance plant growth, nutrient absorption, and stress tolerance in saline soils (Orhan, 2021), but selecting suitable bioinoculants is crucial based on ecological zones (Liu et al., 2021; Ramasamy and Mahawar, 2023). While genetic engineering offers potential for improving salt tolerance, its effectiveness is limited by the polygenic nature of salinity control (Afzal et al., 2020; Arif et al., 2020; Hasegawa, 2013), which involves a coordinated response of multiple traits (Agarwal et al., 2012; Albaladejo et al., 2018; Amirbakhtiar et al., 2019). Organic fertilizers, derived from natural sources like manure, compost, or cover crops, can combat soil salinization by enriching soil organic carbon, improving soil structure, and enhancing nutrient cycling (Naveed et al., 2014). They increase soil organic matter, which reduces salt accumulation, improves water-holding capacity, and prevents nutrient leaching and runoff that can contribute to salinization (Tisdall and Oades, 2006). Crop adaptation strategies, including selecting salt-tolerant crops, implementing crop rotation, and using cover crops, can be effective in maintaining agricultural productivity in saline environments. Salt-tolerant crops thrive in saline conditions, crop rotation disrupts the salt accumulation cycle, and cover crops improve soil structure and reduce erosion (Cuevas et al., 2019).

Effective water management, including rainwater harvesting and efficient irrigation practices, is crucial for adapting to and mitigating soil salinization. Open drainage systems, such as ditches and canals, often have limitations in managing salinity because they can result in uneven drainage and may not effectively remove salts from the root zone. This can lead to salt buildup in the soil, negatively impacting crop yields. Micro Irrigation systems, such as drip and subsurface drip irrigation, offer significant advantages by creating a wetting desalination zone that extends deeper into the soil (Yang et al., 2023), reducing salinity levels and water consumption (Dong et al., 2022). Subsurface drip irrigation is more efficient in reducing evaporation and improving moisture distribution compared to conventional methods (Mukhopadhyay et al., 2020; Wang et al., 2023). These systems provide a more efficient solution by removing excess water and salts from below the soil surface, improving salinity control. Supporting the installation of these systems at the policy level can encourage their adoption, enhancing agricultural productivity and sustainability in irrigated areas (Kotb et al., 2000). Furthermore, the installation of subsurface tile drainage and the use of retention ponds at the end of the system can effectively manage the amount and quality of water, thereby reducing environmental damage from saline drainage downstream (Kitamura et al., 2006).

As discussed above, there are a suite of short-term and long-term strategies to address SWI-related challenges. A large amount of work has focused on improving salinity tolerance in current crops that are mostly salt-sensitive glycophytes in terms of addressing salinity challenges and maintaining productivity on salt-impacted lands. While continued research is needed to document the potential of salt-tolerant crop varieties, efforts should also focus on identifying and implementing nature-based solutions whenever possible. Federal and/or state policies involving economic incentives for landowners could facilitate advanced planning for landscapes experiencing gradually increasing salinization and proactive adaptation in such at-risk landscapes (Haasnoot et al., 2021). In a forested wetland at the risk of salinization, such incentive might provide the landowners with one-time timber revenues, thus encouraging early timber harvesting (Tully et al., 2019a). Proactively clearing forest in highly saline zones would also help with marsh migration ultimately resulting in enhanced ecosystem services (Hansen and Reiss, 2015). Ultimately, a cross-sectoral dialogue involving landowners, community organizations, business sectors, and policymakers would be crucial to protect and sustainably manage coastal landscapes facing the threats of SWI.

6. Conclusion

With highly variable SLR rates across the world, and the compounding effects of climate change, declining freshwater reserves, and increasing groundwater salinity, the coming decades are likely to witness a heightened challenge of salinity in coastal ecosystems. There is no dearth of knowledge on local or regional SWI impacts on soils and ecosystems. A vast number of studies also provide evidence for successful adaptation and/or mitigation strategies along with some of the pitfalls. In this review, we aim to capture the different aspects of SWI, from dominant mechanisms to impacts on soils and ecosystems to some of the adaptation and mitigation strategies, using evidence from all inhabited continents.

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