HARBORING CHANGE: LEGACY CONTAMINATION, PLANT-METAL DYNAMICS, AND THE FATE OF A BOSTON MARSH

A Thesis Presented

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ABSTRACT

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June 2025

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Coastal wetlands within the Boston Harbor watershed provide numerous ecosystem services and community benefits but are challenged by anthropogenic influence including a legacy of pollution and increasing mean sea level. In this thesis, I will consider the historic and present sources of contamination in Boston Harbor, the role an individual salt marsh species (*Salicornia depressa*) plays in the movement of heavy metals, and how projected sea level rise and urbanization may affect future marsh viability. These complex issues are explored through a case study of Pattens Cove, a 9.6-acre parcel of state-managed land in Dorchester, MA home to a small, fringing salt marsh with metal concentrations exceeding levels of concern. Although our decomposition study found lower than expected metal accumulation in plant tissue, total masses

of select metals remained stable despite overall litter mass loss. This suggested stabilization of heavy metals in marsh plants is one of many ecosystem services offered that may be compromised due to sea level rise. As an urban green space embedded into the fabric of the Savin Hill neighborhood, Pattens Cove has an uncertain fate due to rising tides which we explore through aerial imagery and tidal data. Addressing this uncertainty with informed management and planning is essential to safeguard the longevity of this vital urban resource.

DEDICATION

This thesis is dedicated to my parents, whose sacrifices have granted me endless opportunities, and whose love, support, and perspective give me the courage to seize them, the strength to persevere, and the laughter to enjoy the ride.

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CHAPTER 1

BOSTON HARBOR: A STUDY OF PAST AND PRESENT CONTAMINATION

1.1 Introduction

For over 200 years, the estuaries and harbor of Boston, MA were considered among the most polluted around the world. Once called the "Dirtiest Harbor in America" by President George Bush (1988) and denounced as one of the most flagrant violations of water pollution laws in the country, the harbor of today appears to be a far cry from its polluted past. However, legacies of its contaminated history paired with present urban pressures suggest that Boston Harbor is still recovering from its environmental catastrophe and requires continuing attention for conservation and remediation. Monitoring abiotic and biotic environmental factors provides evidence that, despite relative improvements from the 20th century, contamination remains above thresholds of public health concern at a variety of sites and circumstances, particularly among coastal sediments (Breault et al., 2005). Remediation of these contaminants can be high cost, disruptive, and require oversight at multiple levels of governance and engagement. Continued usage of the land within Boston Harbor watershed for industry, commerce, and high-density housing provides both increased contaminant pressure in the environment and an incentive for addressing lingering pollutants. Coastal wetlands which collect river effluent and tidal deposition have maintained concerningly high levels of contaminants despite infrastructural improvement and regulatory oversight. Wetlands function as a reservoir for these harmful contaminants; an underrecognized ecosystem service provided by the historically discredited ecosystems that should be addressed when considering management, conservation, and remediation efforts. As Boston evolves and adapts to new challenges, anthropogenic pressure will continue to affect the Harbor and the capacity of wetlands to stabilize contaminants.

The contaminant story of Boston Harbor is a human one, not unlike any other urban coastal city. The lands of eastern Massachusetts were inhabited primarily by Algonquian-speaking indigenous tribes including the Massachusett, the Wampanoag, and the Nipmuc until the early 17th century when English colonists settled what they called Boston (Potenza, 2025). In the centuries that followed, mass immigration to the region saw rapid growth of both population and urbanized land use. Alongside this growth, Boston's coastline also underwent monumental change. Nancy Seasholes' seminal work *Gaining Ground* (2003) provides robust coverage of the "land making" that occurred in the harbor region, starting with "wharfing out" in 1630s and continuing until the islands of East Boston were connected in the 1950s (Fig. 1.1). Seasholes' work details how over 5,000 acres of marsh land and tidal flats were brought up in elevation with fill material that ranged from household and commercial refuse, including coal ash, to gravel and tidal dredge. This increase in land acreage was necessitated by booming industries including notable evolutions of transportation (shipyards, railroads, and airports), competition for space and trade, and desire for aesthetically pleasing public space in place of marsh and tidal flats. Population in the mid 1600s fluctuated, but the first census taken in 1722 counted 10,567 residents (Boston Archives). In 1850, after 200 years of land making, Boston's metropolitan area housed over 140,000 which increased exponentially to around 3 million by the time urban land expansion slowed in the mid-20th century.

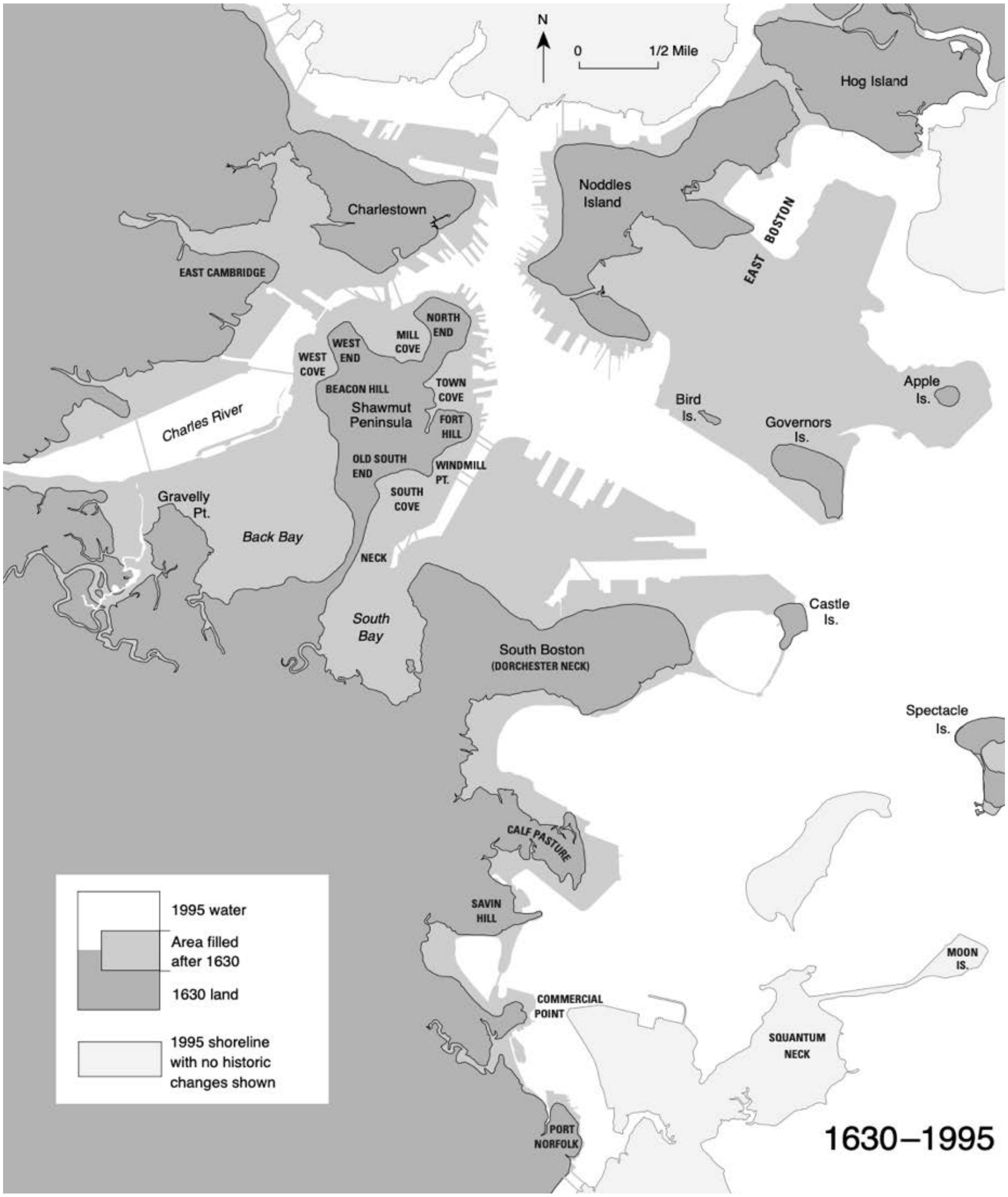


Figure 1.1: Shoreline change in the Boston Harbor from 1630 to 1995 as a result of "land-making" as researched and reviewed by Nancy Seasholes. Image taken from Seasholes' 2003 book *Gaining Ground* and is not for reproduction. © Massachusetts Institute of Technology.

1.2 Urban Waste Management

With people comes waste, and waste unmanaged becomes pollution. In the Boston's early colonial history, disposal of waste, including human biologics and other urban refuse, was managed on an individual family and household basis. The result was the dumping of waste into streets and surface gulleys that led directly to the abutting waterways or indirectly discharged through simple catchment and cesspool systems (Clarke, 1888). The amalgamation of unsanitary conditions on the surface and in the water led to widespread cholera, typhoid, and dysentery outbreaks initiating subsequent mayoral decrees that residents "collect together and place in the street, all rubbish, dirt, filth and decayed or perishable articles….on the following days, in order that the same may be remove [sic] by the City Carts" (Rogers, 1848). This first effort of municipal trash collection was not sufficient to alleviate contamination-driven illness, and by 1884, after a government-sponsored study on pollution, the Calf Pasture Pumping Station and the Moon Island treatment facility had been constructed (Clarke, 1888). This improvement saw the connection of subterranean piping and collection chambers that directed sanitary waste to Dorchester before being pumped to Moon Island just a few miles south in Quincy (Fig. 1.2). Though this system represented the first concerted effort at centralized sewage management and removed waste from the densely populated urban center, the untreated discharge from Moon Island was still high in contaminants and often was carried back into the Harbor by returning tides (Alber et al.,1993).

The move from sewage *management* to sewage *treatment* did not occur until the mid-20th century. Two sewage treatment plants were created in Boston Harbor (Nut Island and Deer Island) which processed approximately 390 million gallons of city sewage per day (Kennison, 1950). These plants utilized large holding reservoirs that removed around half of the total suspended solids before discharging the remaining effluent. This primary treatment method reduced biological oxygen demand (BOD) from raw sewage but was still dispensing 50 tons of biosolids directly into the Boston Harbor per day. By the 1960s, aging infrastructure, diminished resources for utilities, and lack of unified oversight led the system to become overtaxed and direct raw sewage discharges from overflows were commonplace (Reiss and Flynn, 1985).

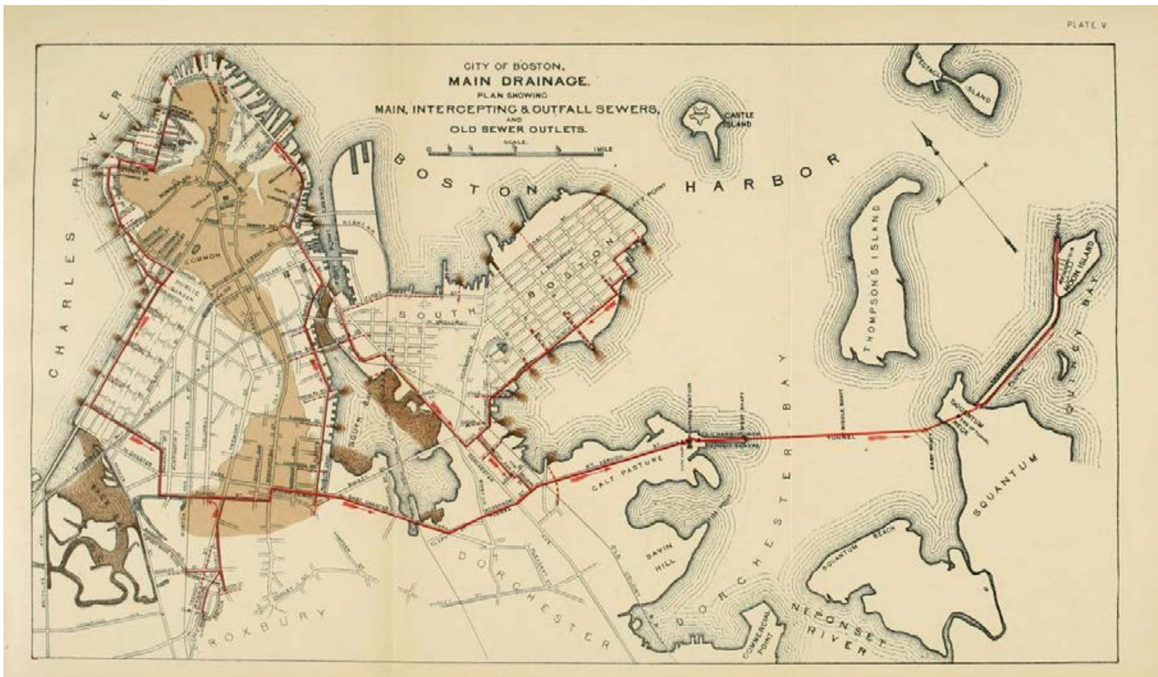


Figure 1.2: Diagram of Boston's original main drainage system of intercepting and gravity-driven sewers first constructed in the 1880s. Drawn by Eliot Clarke, Principle Assistant Engineer for the City of Boston and published by Rockwell and Churchill in 1888.

Aside from being insufficient to handle the volumes of a growing city, it became clear that primary treatment alone was not adequate for addressing human health impacts. Emerging technologies provided methods to quantify a variety of contaminants, expanding notions of pollution beyond aesthetics and bacterial illness (Burian et al., 2000). Additionally, in the 1970s public health concerns and a new appreciation for finite natural resources and their interconnection with societal wellbeing expanded the scope of pollution control to include the environment. The creation of the federal Environmental Protection Agency (EPA) in 1970 and enactment of the 1972 Clean Water Act (CWA) revolutionized accountability for industrial discharge and effluent quality. Mandated by the CWA was secondary treatment of sewage which required bacterial breakdown of sludge to remove at least 85% of both suspended solids and BOD (Alber et al.,1993).

The subsequent years weave a sordid tale of Boston's Metropolitan District Commission applying for waivers from these new federal restrictions and fielding a variety of civil lawsuits that culminated with its replacement with the Massachusetts Water Resources Authority (MWRA) in 1984 (Reiss and Flynn, 1985). What followed became known as the "Boston Harbor Clean Up" and is now considered one of the largest public works projects with $3.8 billion of sewage treatment improvements. Most notable is the construction of the Deer Island Sewage Treatment Plant, designed to apply primary and advanced secondary treatment on over a million gallons of sewage per day before dispensing effluent through an outfall 9.5 miles long in the harbor (MWRA). The result of this incredible overhaul to the sewage treatment system resulted in a decrease by between 80% and 90% of loadings of total nitrogen, total phosphorus, total suspended solids, and particulate organic carbon to the harbor (Taylor, 2010).

1.3 Past and Present Industry

Throughout Boston's expansion and development, concerns over contamination and pollution of the harbor encompassed more than solely human sanitary waste. Boston has a rich history of industry, spanning shipbuilding and ropemaking, textile mills and tanneries, metal working, power generation, car manufacturing, and medical and technology research. From the establishment of Boston as a central figure in shipping and trading around the world to the subsequent expansion into textile milling at the dawn of the U.S. Industrial Revolution each new iteration of industry brought forth a number of environmental impacts (Bowen et al.,2019).

Rivers were dammed as early as 1634, affecting fish migration and trapping sediments, creating mill ponds that later served as repositories of industrial waste. Direct dumping of contaminated discharge included the dyes, soaps, and bleaches of the textile industry but was compounded by the leachate of coal ash piles from energy generation and runoff from paved surfaces. By the 1890s, the textile industry in Lowell discharged approximately 40 million gallons of water per day into the Charles River, over 100,000 times more than contemporary household usage (Tsongas Industrial History Center). Between 1838 and 1988, there were over 100 tanneries and leather finishing operations within the Aberjona Watershed, tributary to the Mystic River, utilizing chemicals found to be toxic, mutagenic, and carcinogenic (Durant et al.,1990). Within the same region, pesticides containing lead and arsenic were synthesized and sprayed heavily at nearby orchards and agricultural fields (Aurilio et al.,1995). Car manufacturing and metal plating including Ford Motor Company's Cambridge Assembly in 1913 and General Motors Framingham Assembly in 1947 contributed to both air and water pollution through the use of heavy metals like iron, copper, and lead, in addition to paint-based solvents and sulfuric acid from required energy production (Melosi, 2010).

Though the Clean Water Act regulated new contributions of these contaminants into the environment, their presence has endured beyond those biological and bacterial concerns related to sanitary waste. Sites with histories of industrial use and contamination exceeding government-set levels are considered a public health hazard and pose a threat to environmental health. Superfunds, sites designated by the federal government as heavily contaminated, and

Brownfields, targeted as sites of concern but managed on the state and municipal level, are prolific in the communities around Boston Harbor. Suffolk, Norfolk, and Middlesex counties have a cumulative 417 sites monitored under EPA Superfund (including the National Priorities List)), Brownfields, and other Authorized Use Limitations designations with contaminants of concern including but not limited to, Polychlorinated Biphenyls (PCBs), Polycyclic Aromatic Hydrocarbons (PAHs), arsenic, lead, asbestos, cyanide, solvents, and Volatile Organic Compounds (VOCs) (Fig. 3). Aside from the legacy of historic industry, present urban uses require monitoring from a variety of government and environmental organizations. The Massachusetts Department of Environmental Protection regulates users of oil and hazardous materials as well as locations where waste clean-up has occurred. These “Activity and Use Limitations” regulations provide critical information about the risks that remain on the site and as of 2025, 185 sites fall within the Boston Harbor Watershed. Additional monitoring of sites of concern is administered through the Massachusetts Water Resources Authority. The MWRA conducts routine monitoring of “Significant Industrial Users” (SIUs) and other locations within its service area that produce with qualifiable discharge; in 2024 they inspected 180 SIUs and issued or renewed 452 industrial permits (MWRA, 2024). In this same monitoring period, MWRA found of the SIUs that 29 were in “significant noncompliance” and issued 110 Notices of Violation or enforcement actions; other regulated industrial and commercial facilities were issued 298 of any type of enforcement actions. These violations occurred at facilities across all sectors with higher than acceptable levels ranging from research and medical lab effluent to poor practices contributing to surface runoff from heavy equipment washing and metal plating.

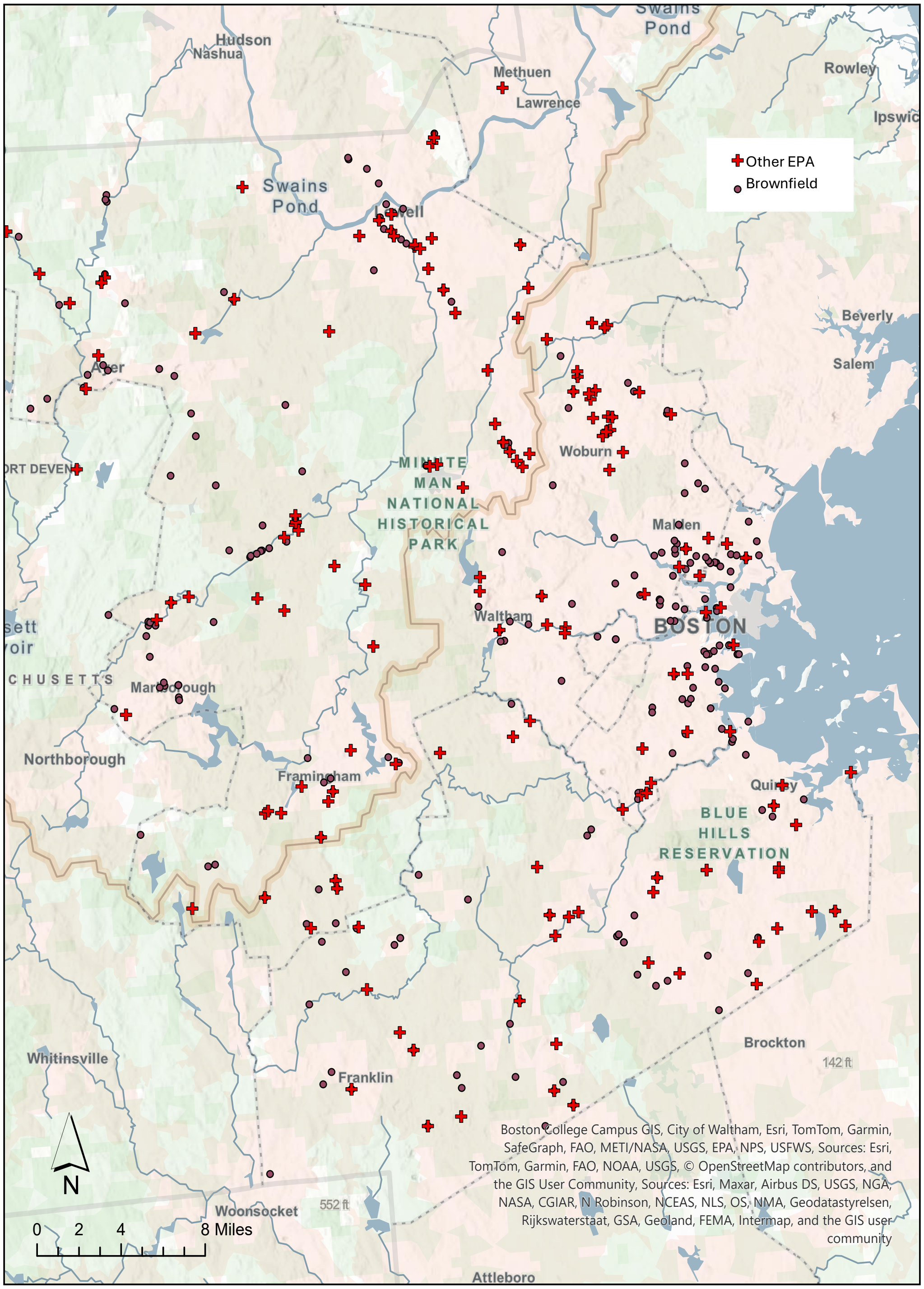


Figure 1.3: Brownfield and EPA-monitored sites in Middlesex, Norfolk, and Suffolk counties. Brownfield data is taken from MassDEP and was most recently updated December 2018. EPA data is taken from the routinely updated EPA Superfund Enterprise Management System database and includes National Priority, Alternative Approach, archived and other site types monitored.

1.4 Urban Pressures

Contemporary pressures of urbanization build on a contaminated legacy. Much of the Boston Water and Sewer Commission (BWSC) infrastructure is outdated with design capacities exceeded by a growing population and increased precipitation and flood events. The resulting direct discharge of mixed sewage and stormwater into Boston Harbor, known as combined sewer overflows (CSOs) and have been accountable for over 616 million gallons of discharge volume in the past 10 years (BWSC, 2024). Additionally, though drinking water supply to the Boston metropolitan area is lead-free from reservoir to distribution pipes, BWSC suspects as many as 2200 lead lines in private buildings may still be connected in their service area.

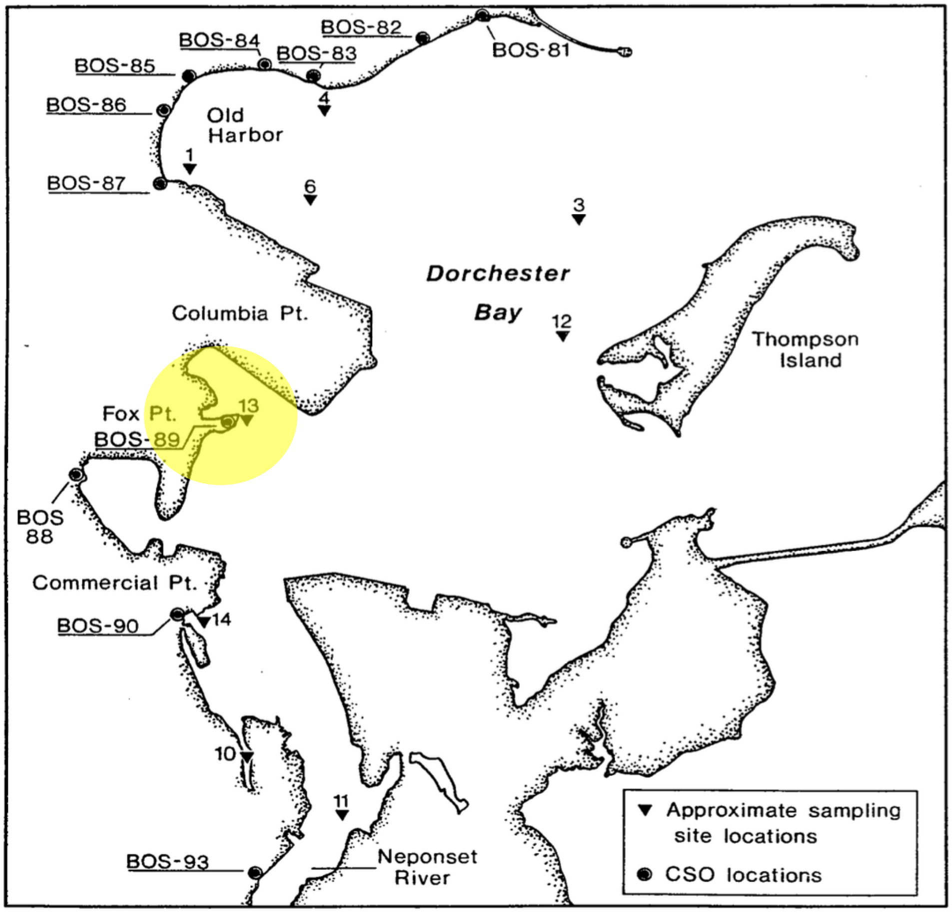
Evolving urban dynamics have also contributed to new sources of contaminants and exacerbated their distribution. Change to land use has increased impervious surface which allows precipitation events to quickly carry contaminants directly into waterways and reduces natural remediation offered by infiltration (Brabec, 2009). Car and other transportation exhausts emit nitrogen oxides, particulate matter, and hydrocarbons (Twigg, 2007) while emerging research reveals that degradation of tires during routine usage contributes to microplastic, heavy metal, and VOCs in road runoff (Tian et al., 2020, Werbowski et al., 2021). High urban pressure and intersection of human use with coastal systems means that incidental exposure, for example car crashes and oil spills, occur frequently (Annear 2014; Thompson, 2025) and can siphon contaminants directly into the environment.

1.5 Monitoring Environmental Indicators

Contaminants of concern from Boston's human and industrial influence are, despite regulation, persistent in the environment. The gem of Boston's rehabilitation story, the sewage treatment upgrade, has without a doubt caused an incredible rebound in both the aesthetics of the waterfront and the quality of the water itself. Monitoring reports comparing previous water quality data to those taken five years after effluent was diverted through the newly constructed outfall showed a significant decrease in nitrogen and phosphorous compounds, chlorophyll-a, particulate organic carbon (Taylor, 2006). The drastic improvement related to nutrient enrichment and bacterial concerns is reflected by the changing attitude toward the Harbor, with increased waterfront development, water-based recreation, and other ecosystem services (Jin et al., 2018). However, monitoring requires consideration of overall environmental quality including ecological indicators and must investigate more contaminants than those related to nutrient enrichment from sewage.

Marine sediment reports prior to the construction of stormwater treatment plants employing higher level treatment painted a dismal picture. A 1994 report commissioned by the MWRA found sediments with elevated levels of not only indicator bacteria *(Clostridium perfringens)*, linear alkylbenzene (LAB), and coprostanol, but also chemical contaminants like PCBs, PAHs, and chlorinated pesticides. The presence of these contaminants, in addition to characteristics like grain size and oxygen availability, directly affect the communities of macroinvertebrates that inhabit the benthic marine environment. Benthic macroinvertebrates who make the marine sediment layer their home are incredibly important bio-monitors and indicators of environmental conditions that have been studied in conjunction with the cleanup of Boston Harbor. Foundational work on the so-called "black mayonnaise" of Boston Harbor marine sediments by Gallagher et al.,(1989, 2002) tracked the changes in benthic invertebrate abundance, species diversity, and community structuring starting in the 1980s and continuing until the early 2000s. Alongside rapid and marked changes to water quality, the shift in the macrofaunal community was subtle yet similarly significant. Prior to the sewage treatment upgrades, the benthic community in Boston Harbor was dominated by pollution-tolerant species associated with low oxygen conditions and high levels of organic enrichment. Sampling that occurred parallel to sewage infrastructure and other CWA improvements found increasing presence of more pollution-sensitive species like *Ampelisca* spp., tube-building amphipods that facilitate oxygenation in sediment layers and in turn create microhabitats for future succession (Gallagher et al., 2002). Diaz et al. (2008) also found strong evidence that between 1992 to 2006, benthic habitats within Boston Harbor shifted from a more anaerobic state to a more aerobic state based on the composition of species surveyed. Recent surveys show this trend has continued with overall habitat condition improvement and increasing species richness (Rutecki et al., 2023).

The changes in quality of marine sediment and associated abundance and diversity of benthic species that indicate improving environmental conditions is largely attributed to the regulation of urban effluent. However, contaminants like heavy metals and legacy chemicals have persisted at concerningly high levels in coastal environments. During the same period that harbor cleanup efforts began, heavy metal contaminants were notably found to be well above thresholds of concern for human health at many sites (Durell, 1994). Further research into sediment metal content in the Harbor has found declining values but are inconsistent and often above levels considered safe for humans and fauna (Zago et al.,2001). Sediment impounded behind historic dams along the Charles, Mystic, and Neponset rivers have also tested at exceedingly high levels of PCBs, PAHs, and metals like mercury and lead (Breault et al., 2005; Breault and Cooke, 2006). In particular, salt marshes in the depositional areas of Boston Harbor have retained high levels of heavy metal contaminants. In fact, of the sites monitored by the MWRA, metal concentrations of marine sediment sampled within coves and inlets where marshes are found are consistently among the highest levels in the Harbor (Fig. 1.4). The MWRA sampling site highlighted, DB13, is a depositional area that received both effluent from the Neponset River and tidally mobilized sediments and has historically been a site of high contamination relative to sources of pollution. This persistence in urban contaminated marshes is supported by a recent survey that have found contaminant levels in the surface sediment of the marsh adjacent to DB13 to be well above threshold effect levels of concern and much higher as compared to less urban sites (Fig. 1.5).



Non-normalized (ug/g, dry weight)*

Site ID	Al (%)	Cd	Cr	Cu	Fe (%)	Ni	Pb	Zn	Mud* (%)
				1994					
DB01	5.70	0.844	157	134	3.72	34.5	152	233	74.4
DB03	4.63	0.228	65.2	27.9	1.99	15.3	54.6	69.5	18.6
DB04	5.47	0.820	139	87.0	2.94	28.2	159	168	60.8
DB06	3.64	0.065	34.8	12.1	1.35	9.10	28.4	33.3	7.4
DB10	6.17	1.69	218	198	4.22	37.8	182	273	87.9
DB12	5.33	1.13	209	98.5	3.14	33.9	130	172	45.6
DB13	6.71	2.12	254	158	3.79	37.1	199	266	85.7
DB14	4.25	1.48	119	126	2.84	25.5	347	344	51.9

Figure 1.4: Map of sites sampled across Boston Harbor (top) and table of marine sediment metal concentrations in 1990 (bottom) compiled by Durell et al.,in a 1995 report prepared for the MWRA. Highlighted is sampling site DB13 close to Pattens Cove and the abutting Fox Point CSO.

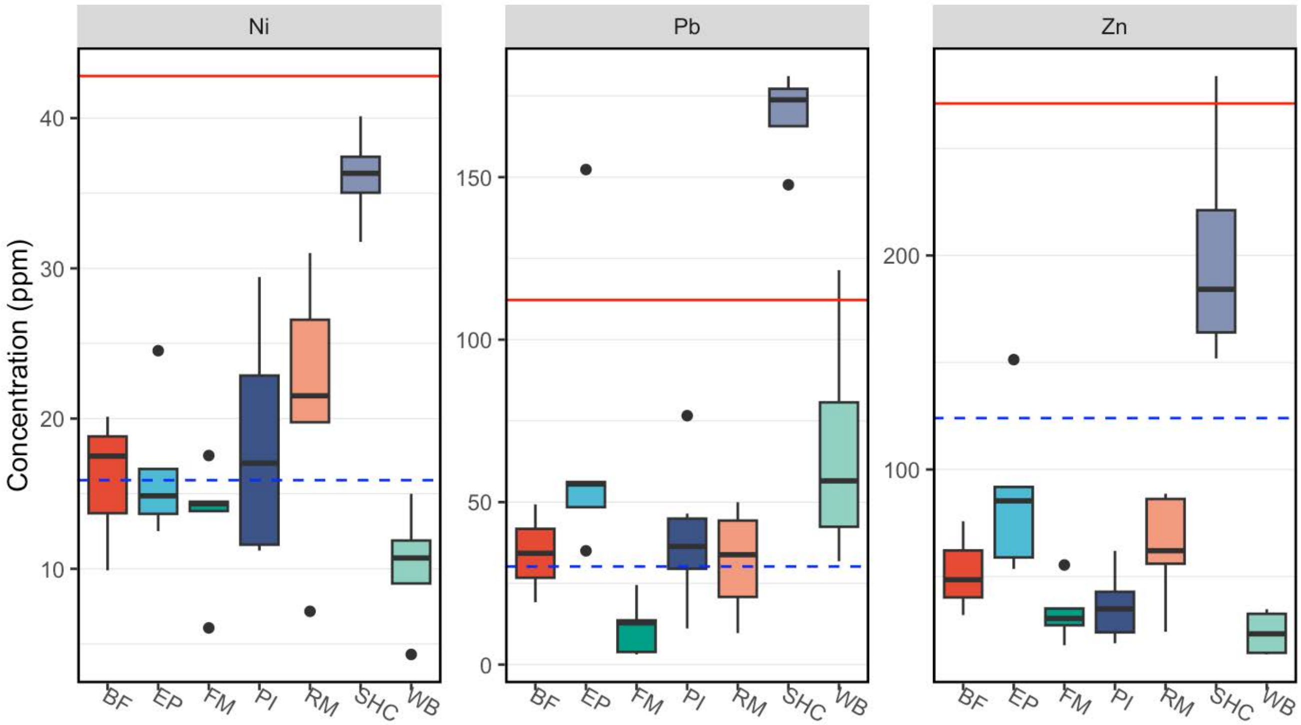


Figure 1.5: Concentrations of New England marsh sediment metal concentrations at seven sites using x-ray florescence of sediment no deeper than 10cm and with intensities converted based on calibrations with known reference material standards (Stefanovic, 2021: UMass Boston). Red solid line is Probable Effects Level (PEL) and blue dotted in Threshold Effects Level (TEL) as described in NOAA SQuiRT tables for inorganics in marine sediment. SHC is Savin Hill Cove in Boston Harbor, MA with a history of industrial contamination while FM and WB are protected estuarine reserves in Nantucket and Falmouth, MA, respectively.

1.6 Remediating the Legacy

Unlike the nutrients and bacteria associated with human sanitary waste that were largely addressed with sewage upgrades, heavy metal contaminants and other so called "forever chemicals" will remain in the environment, particularly in fine grain sediments, without intervention. Positively charged heavy metals like lead, arsenic, and mercury bind tightly with negatively charged organic material in marine sediment which settles into low lying basins, can be disturbed by river or tidal action, or assimilated into coastal wetlands through deposition (Jezycki et al.,2024). These metal contaminants can be bioaccumulated and biomagnified up the food chain, causing lasting harm and reducing viability of species that rely on coastal environments for habitat or nutrition (Chen et al.,2016; Saidon et al.,2024;).

Elimination of these contaminants at their sources would be ideal, but their removal is difficult to orchestrate once deposited. Conventional efforts to remediate these contaminated areas include high-disruption, high-cost methods like mechanical extraction, soil washing, and thermal or chemical treatment (US EPA, 2015). These methods can disturb local ecosystems and require large financial investment and coordination across different regulatory entities and private landowners. Even if projects are undertaken, the question remains of what to do with the extracted contaminated waste and how to prevent re-exposure over time as sediment circulates through the coastal system. While complete removal of harm should be held as the ideal, constraints and challenges of contaminant management suggest turning to alternatives that serve to abate and reduce this harm. In fact, many ecosystems provide services related to contaminant mitigation and should be valued appropriately as nature-based solutions.

Considered through a different lens, depositional areas, and in particular coastal wetlands that retain these metals are currently serving effectively as “sinks.” Applied primarily to the concept of carbon, a natural sink is a reservoir that stores and abates carbon at a rate higher than it releases. Salt marshes, aside from being known as carbon sinks, therefore have effectively been serving as sinks for sediment-associated urban contaminants. Mechanisms by which this occur can include binding of positively charged contaminants with organic-matter rich sediments in addition to phytoremediation. Phytoremediation is any strategy by which plants mitigate contamination—this can encompass phytoaccumulation and subsequent biomass extraction; phytostabilization in which roots immobilize and reduce contaminant bioavailability; phytodegradation in which plants and associated bacteria break contaminants into less harmful substances and the less common phytovolatization in which plants uptake and release contaminants in a less toxic form (Jacob et al.,2018). Phytoremediation has been considered at length within scientific discourse focusing on capacities of individual plant uptake and applications to remediation strategies (Pilon-Smits, 2005), but practical limitations have prevented widespread adoption including the time and labor, biological limitations of plant success in high toxicity settings, and challenges with disposal of contaminated plants. However, less clearly explored is the present, unmanaged existence of phyto-accumulation and -stabilization that is already occurring particularly along the water's edge. It is difficult to begin to quantify such rates due to the complex interplay or abiotic factors (salinity, temperature, oxygenation, accretion and deposition) and biotic factors (bacterial, floral, and faunal behavior) that occur particularly in salt marshes (Bryan and Langston, 1992; Williams et al., 1994). Despite the uncertainty toward explicit values, it is important to recognize and consider how coastal marshes may historically have stabilized contaminants of concern and what role they could play in the future.

1. 7 Conclusion

Boston Harbor, once infamous for its dirty water, has changed monumentally in the past few decades. Despite the herculean efforts to curb the effects of a growing population in an industrial city, the legacy of contamination lives on in the water, sediment, and ecological communities. Though the complete and total remediation of these contaminants remains a challenge, naturally occurring ecosystem services related to stabilizing contaminants as provided by coastal wetlands should not be discounted. Seated at the interface of natural systems and anthropogenic pressures, these wetlands have variable capacities to serve as sinks of pollutants while continuing to provide other notable and critical ecosystem functions. In particular, threats of sea level rise and associated marsh loss may affect contaminant mobility or have other, unexpected implications for contamination in the harbor area. In the ongoing effort to improve Boston Harbor as a place to live, work, and thrive for humans and all other species, more effort is required to understand the extent of the relationship between contaminants and urban wetlands to inform management and remediation approaches.

CHAPTER 2

METAL CONTENT AND LITTER DECOMPOSITION IN THE METAL TOLERANT SALT MARSH PLANT *SALICORNIA DEPRESSA*

Based on a manuscript prepared for submission to *Science for the Total Environment:*

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2.1 Abstract

Coastal wetlands face numerous anthropogenic threats including development, sea level rise, and pollution. In particular, salt marshes along urban New England coastlines have been exposed to heavy metal contamination for over two centuries with historic and present industrial uses. While salt marshes play a critical role in abating and storing the toxicity of these contaminants, not all species perform equally under increased concentrations which can impact community diversity and challenge ecosystem stability over time. Succulent halophytes in the genus *Salicornia* are widely distributed across the northern hemisphere and are known to be hyperaccumulators of such metals, yet little is known about the bioaccumulation capacities and mechanisms of the common New England species *S. depressa*. To study the fate of common heavy metal contaminants within *S. depressa*, we placed recently senesced above-ground

biomass in litterbags at four transects within a fringing Massachusetts urban marsh for 30 weeks. We collected data on remaining biomass and trace metal concentrations for As, Cu, Ni, Pb, and Zn at six-week intervals. Overall, shoot metal concentrations and estimated bioconcentration factors were considerably lower than expected given reported laboratory bioaccumulation rates within the genus. However, the percent of total litter biomass remaining at the end of the study period was notably lower than the percent of total metal mass remaining for Cu, Ni, Pb, and Zn. Retention of metal despite degrading biomass provides evidence that metals are bound to long enduring tissues and/or that adsorption of metals from the environment occurs throughout decomposition. More research is warranted to quantify potential localization of metals to *S. depressa* root tissue and to identify potential biological and physical influences on metal adsorption and retention in salt marsh litter.

2.2 Introduction

Coastal wetlands make up only a fraction of the Earth's surface yet provide invaluable ecosystem services to people and the environment. Aside from fostering diverse plant and animal life, coastal wetlands can sequester high amounts of atmospheric carbon (Drake et al., 2015); act as natural storm and flood protection by reducing erosion and buffering wave action (Shepard et al., 2011; Leonardi et al., 2018) and play an essential role in the storage and purification of water through the cycling of contaminants (Pruell et al., 1990; Li et al., 2022). Despite their clear importance, more than 80,000 acres of coastal wetlands are lost per year primarily from anthropogenic causes including land-use change and sea level rise (US EPA, 2009), with estimates that they have lost a quarter of their historical global coverage (Crooks et al., 2011).

The conservation and restoration of remaining coastal wetlands is critical in preserving not just the services they offer but also the intrinsic value of this complex ecosystem.

In New England and in many other heavily populated or industrialized coastal regions, coastal wetlands like salt marshes face an additional challenge of heavy metal contamination as estuaries at the outlet of industrial activity (Robinson et al., 2003). While salt marshes tend to be tolerant of these pollutants, not all plant and animal species can subsist equally under varying toxin pressures which can suppress productivity and change species diversity over time (Sharifuzzaman et al., 2016). Plants that tolerate higher contaminant loads and even pull heavy metals from the environment play an important role in conserving threatened natural areas and serve as exciting candidates for green-based solutions to environmental pollution. Plants of the genus *Salicornia* are generally extremely tolerant of both salt and heavy metal contaminants (Sharma et al., 2010; Khalilzadeh et al., 2020). Though *Salicornia* are found around the northern hemisphere, the species most widely distributed in North America, *Salicornia depressa,* has rarely been studied. This work is situated within the scope of ongoing research conducted at UMass Boston investigating the breadth of heavy metal concentrations across New England marshes as well as the bioconcentration and partitioning factors of *S. depressa* during the growing season. Metals of interest include Arsenic (As), Nickel (Ni), Copper (Cu), Lead (Pb), and Zinc (Zn), which are regionally known as persistent metals contained within the marsh sediment and that continue to be deposited by tidal deposition, river effluent, and urban runoff. We ask: how much heavy metal is accumulated within fully mature *S. depressa* shoot material; at what rate does *S. depressa* decompose and cycle heavy metals back into the surrounding environment; and do variations in sediment type and site characteristics influence these trends? Answers to these questions play an important role in understanding the full role of *S. depressa* in cycling heavy metals and provide foundational information for how salt marsh ecosystem dynamics and urban communities may be influenced by natural remediators.

2.3 Background:

Heavy metal contamination in salt marshes is the legacy of centuries of manufacturing and continued high-density urban usage. In New England, historic paper and textile mills in conjunction with expanding infrastructure including transportation, sanitation, and urban development contribute to high levels of environmental contaminants like lead, mercury, and arsenic entering waterways (Robinson et al., 2003). These contaminants, transported to low lying coastal areas, persist in the sediment without intervention. Aside from the direct threat to public health from these pollutants, they can be taken up by aquatic plants and animals and accumulate up the food chain causing damage to many levels of biodiversity (Chen et al., 2016; Sharifuzzaman et al., 2016) while also causing direct harm to the microbial community of the rhizosphere (Maiti and Chowdhury, 2013; Mesa et al., 2016).

Marshes of New England, particularly near urbanized locations, are known to have high levels of a variety of metals. Around Boston alone, there are several EPA-designated Superfund sites from historic manufacturing, textile production, milling, and smelting in addition to thousands of continued-use private lead service lines in the water system (US EPA, 2015; BWSC, 2024). From this historic and current influence, metals of human health interest in regional salt marsh environments include arsenic, copper, nickel, lead, and zinc. Arsenic, a well-documented carcinogen, is considered to the most significant chemical contaminant in drinking-water globally (WHO, 2022) while lead exposure is linked to damage of the nervous system and impaired brain functioning, with estimates that half of the U.S population were exposed to harmful levels in early childhood as of 2015 (McFarland et al., 2022). Nickel, known to cause cardiovascular, respiratory, and gastrointestinal diseases, cancers, and distress, has in the past few decades also linked to epigenetic alterations (Genchi et al., 2015). While copper and zinc are both considered essential micronutrients for human health, excess levels can manifest various chronic and acute illnesses (Briffa et al., 2020).

In salt marshes, these metals have broad and varied potential for harm. Arsenic can be taken up by wetland plants and influence microbial communities, potentially altering nutrient cycling and ecosystem function (Fitzgerald et al., 2003). Given the anoxic conditions characteristic of many marsh sediments, arsenic may also be mobilized through redox reactions, increasing bioavailability and transport through the system (Masscheleyn et al., 1991). In marsh ecosystems, lead can persist in sediments for decades, binding to organic matter and sulfides but remaining bioavailable under certain geochemical conditions (Gallagher et al., 2008). Lead and arsenic uptake by marsh vegetation can lead to trophic transfer, impacting herbivores and higher consumers (Saidon et al., 2024). Beyond their individual toxicities, these heavy metals interact in complex ways within marsh environments. Competitive ion exchange, organic matter adsorption, and redox-driven solubility shifts can influence their mobility and bioavailability (Northrup et al., 2018). For example, arsenic and lead both bind strongly to iron and manganese oxides in sediments, but under reducing conditions, arsenic is more likely to be released into porewater while lead remains largely immobile (Smedley and Kinniburgh 2002). Similarly, zinc and nickel, while essential in trace amounts, can disrupt plant and microbial function at higher concentrations (Babich and Stotzky 1982) with zinc often competing with other cations for binding sites in marsh soils (Du Laing et al., 2009).

Despite the differing direct and indirect risk factors of the metals discussed, the magnitude of the risk is tied to their retention and accumulation in environmentally sensitive areas like salt marshes. Recent work by Jezycki (2024) surveying mobilization of metals across marsh soil conditions in the east coast of the United States found similar results and expanded on the relationship between physiochemical environs and metal retention or mobility. They determined that certain metals (including As) had the highest mobility under anaerobic conditions with organic, often saturated saline soils while others (including Pb, Ni, and Zn) were more mobile in well-drained, non-saline conditions and mineral soils. Soil cores reaching back to 1925, despite highlighting an overall decrease in metal concentration within recent decades, confirmed that marshes with high lead loads are correlated with industrial development and that high levels of lead as well as nickel and zinc were correlated with marshes with high accretion and sedimentation rates. The behavior of these metals within marsh ecosystems is facilitated by mechanisms of ion exchange and organic matter mineralization in which the factors of salinity, pH, and sediment type increase metal availability (Chibuike and Obiora, 2014).

Despite regulation of contaminant sources, high retention of metals in marine sediment and new industrial pressures have caused coastal shorelines to maintain levels of metals above threshold effect levels (Durell 1995). Conventional efforts to remediate these contaminated areas include high-disruption, high-cost methods like mechanical extraction, soil washing, and thermal or chemical treatment (US EPA 2015). Both historic and modern contamination burdens in coastal areas could be addressed through less invasive, nature-based solutions. The use of plants to take in and store heavy metal contaminants, known as phytoremediation, has been proven as a lower risk and lower impact method to tackle a persistent problem (Pilon-Smits 2005; Chintapenta et al., 2022). Native salt marsh plants with high tolerance for and high accumulation rates of heavy metals can be used to mitigate contamination in both conservation and restoration settings by accumulating, storing, and stabilizing contaminated sediments throughout their life stages.

One plant with high rates of tolerance and accumulation potential is *Salicornia depressa,* the most common New England species of a genus of halophytic marsh plant that is distributed around the northern hemisphere (Ozturk et al., 2018). *Salicornia* are known by many common names including pickleweed, glasswort, sea bean or asparagus, and samphire. Very few studies have been done on *S. depressa,* with most knowledge of the plant stemming from its European and Canadian relatives *S. europaea* and *S. maritima* but is accepted as an early successional species that exploits disturbance areas in marshes like bare patches created by tidal wrack (Ellison, 1987). *Salicornia* spp. are also known hyperaccumulators of heavy metals and have been found to not only survive but also continue to accumulate at increasingly high rates of contaminant exposure (Table 2.1).

Table 2.1: Literature review of accumulation by Salicornia spp. in greenhouse and field settings with accumulation reported as $mg \cdot g^{-1}$of plant dry weight. Studies include a variety of *Salicornia* spp., experimental designs, treatment methods, plant partitions, and growth stages. Table is intended to show the breadth of accumulation potential of the genus rather than explicit cross comparison of data.

Contaminant	Setting	Accumulation ($mg \cdot g^{-1}$dry wt)	Sources
Arsenic	Greenhouse	0.009 – 4.24	Mesa-Marin et al., 2020; Sharma et al., 2010
Copper	Field	0.018 – 2.635	Nel et al., 2023; Smillie 2015
Lead	Greenhouse	0.381 ± 0.012	Khanlarian et al., 2020
	Field	0.004	Nel et al., 2023
Nickel	Greenhouse	0.016 – 16.82	Sharma et al., 2010
	Field	0.005 – 0.014	Nel et al., 2023
Zinc	Greenhouse	3.559 ± 0.009	Khanlarian et al., 2020
	Field	0.042 – 0.052	Nel et al., 2023; Nel et al., 2023

The established remediation mechanism of *Salicornia* species for most contaminants is accumulation within the plant tissue itself. Research into the mechanisms of *Salicornia* spp. metal accumulation including where within the plant tissue contaminants are stored and at which quantities throughout the growing stage have yielded extremely variable results (Sharma et al., 2010; Khanlarian et al., 2020; Nel et al., 2023). Additionally, metal stability within plant tissues and dynamics during decomposition after senescence has been little studied. Understanding these processes would inform remediation planning (e.g. when plant tissue would need to be harvested and removed from the site for effective remediation) and also provide a better understanding of marsh functioning including insight to successional dynamics potentially facilitated by the pioneer species *S. depressa*. The value of this knowledge is likely to become more useful as marsh migration occurs landward due to tidal range shifts requiring colonization of new ground (Fagherazzi et al., 2019; Osland et al., 2022). Establishment of marshes in new elevational niches will also be complicated by changing weather patterns and increase in severe precipitation incidents (IPCC, 2023), causing pulse events inundating low lying areas with urban runoff. To understand the ecosystem services provided by natural *S. depressa* populations, plan effective remediation management, and strategize for the success of current and future marshes, we must understand the full cycle of its heavy metal accumulation and subsequent decomposition.

2.4 Methods

2.4.1 Study site

The study is situated in the high intertidal zone of *Spartina*-dominated rocky shore and salt marsh at Savin Hill Cove, a highly contaminated depositional area within Dorchester Bay of

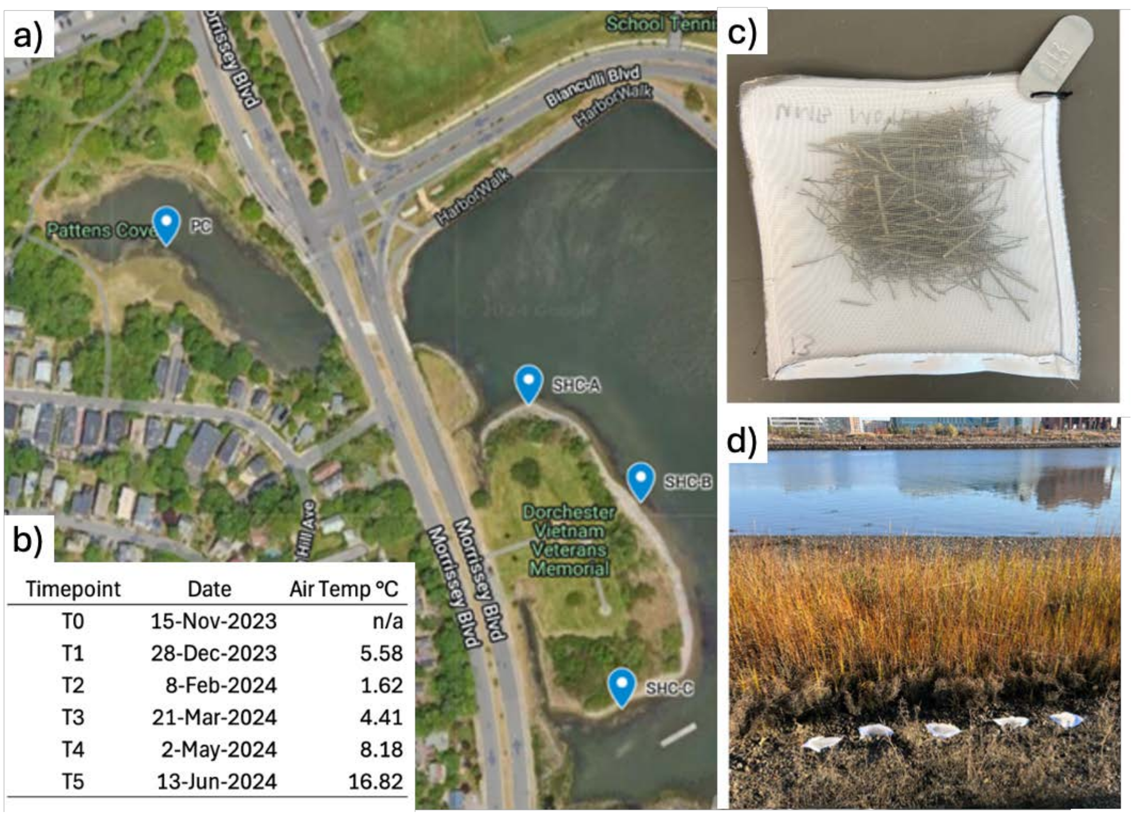
Boston, MA. This location was chosen in accordance due to its notably high levels of known environmental contaminants as compared to other marsh areas in the New England area (Table 2.2). The soils of the survey site are classified by the USDA Natural Resources Conservation Service as wet substratum udorthents with slopes between 0 and 3%. Within the Savin Hill Cove study site, four transects of interest were determined based on their variation in sediment characteristics. Notably, the Pattens Cove (PC) transect is isolated by a constricting culvert over which the nearby major roadway Morrissey Boulevard crosses. Bulk density, reported as g/cm^3, was determined by retrieving two cores 5 cm wide and 8 cm tall at each of the four transects. The sediment was oven dried at 50°C oven for one week prior to measuring mass.

Table 2.2: Metal content in marine and marsh sediment in Boston Harbor. TEL is threshold effect level and PEL is probable effect level. Effect level data is taken from NOAA SQuiRT (Screening Quick Reference Tables) for inorganics in marine sediment. Marine sediment metal concentrations were assessed through x-ray fluorescence of sediment grab samples collected by Battelle as contracted by the MA Water Resources Authority at a depth of 5.5m adjacent to the study site. Marsh sediment metal concentrations were assessed through x-ray florescence of sediment no deeper than 10cm averaged across the study transects. Arsenic was not measured in the 1990 and 1994 samples and was unable to be quantified in 2020 due to spectral interference of high lead concentrations.

	Marine Sediment				Marsh Sediment	
	NOAA Squirt		MWRA		XRF	
μg/g	TEL	PEL	1990	1994	2020	Std. Dev.
As	7.2	41.6	n/a	n/a	n/a	n/a
Cu	18.7	108.2	182	158	48.34	3.97
Ni	15.9	42.8	44	37	36.14	3.43
Pb	30.2	112.2	192	199	169.08	14.79
Zn	124	271	342	266	201.02	58.75

2.4.2 Litterbag preparation and collection

Above-ground biomass of recently senesced *S. depressa* plants from transects along Savin Hill Cove was collected in early fall. The biomass was evenly dried at room temperature and processed to smaller segments (~10 cm) after removing non-*Salicornia* material. Approximately 10 grams of litter, weighed and recorded, were placed in 20 cm^2 0.5 mm mesh litterbags uniquely identified with aluminum tags following procedures outlined by the Long-Term Ecological Research Office (Sexton, 1995). Litterbags were deployed by staking with galvanized landscape staples along four transects each containing three subplots on cleared substrate at the higher intertidal zone where *S. depressa* grows the most densely. One bag was collected from each subplot at six-week intervals until late spring, for a total of 72 samples, including an initial time-zero sample (4 transects x 3 subplots x 6 time points) (Fig 2.1). Litterbags were rinsed with deionized water and foreign materials removed before being placed in a 50ºC oven for at least 5 days.



Timepoint	Date	Air Temp °C
T0	15-Nov-2023	n/a
T1	28-Dec-2023	5.58
T2	8-Feb-2024	1.62
T3	21-Mar-2024	4.41
T4	2-May-2024	8.18
T5	13-Jun-2024	16.82

Figure 2.1: a) Study site within Boston Harbor, MA with transects identified. Google Earth 2024. b) Timepoints and correlated sampling dates. Air temperature data represents an average of the 6 weeks preceding the sampling date listed. Data collected from NOAA station BHBM3 - 8443970 - Boston, MA recorded in 15-minute intervals. c) Individual litterbag prior to field deployment. d) Deployed litterbags at one replicate along transect SHC-B.

2.4.3 Metal content analysis

150mg +/- 5mg of homogenized, ground sample was placed in 50 mL acid digest vessels. 4000 microliters of 70% Aristar Plus trace metal grade nitric acid and 500 microliters of 30% Sigma-Aldrich trace metal grade hydrogen peroxide were pipetted into each digest vessel. The hydrogen peroxide was divided into two aliquots of 250 microliters each added before and after the nitric acid to moderate the exothermic reaction. After one to two hours of digestion, the samples were capped, torqued, and heated to 175°C held for 4:30 minutes using a CEM Mars Xpress Microwave Digestion Oven (EPA 3051_24). Digested samples were then centrifuged at 6000rpm for 10 minutes and the resulting supernatant was diluted 1/20 with Milli-Q water (2mL digest sample to 40 mL Milli-Q).

Following the acid digestion of plant samples, heavy metal concentrations were determined using Inductively Coupled Plasma Mass Spectrometry (ICP-MS). Initial performance checks were performed on the Nu Instrument Attom ES HR-ICPMS with tuning solution and blanks of 2% ultra-pure Optima™ grade nitric acid for optimization. A 6-point calibration curve was created with Science Plasma Cal stock solution for metals of interest at 10 ppb. All blanks, calibration solutions, and samples were spiked with the same internal standard of 1 ppb Indium in 2% nitric acid with 0.05% Triton-X100. Isotopes monitored at mid-resolution were 60Ni, 61Ni, 62Ni, 63Cu, 65Cu, 66Zn, 68Zn, 206Pb, 207Pb, 208Pb while 75As was monitored at high-resolution. Blanks and standards were included throughout each run to monitor for any instrument drift. The concentrations of heavy metals were quantified based on the mass-to-charge ratio (m/z) of each metal isotope, and results were corrected for background interference and reported in mg/kg dry weight of the plant material.

Each round of digestion and spectrometry were completed with additional samples of Standard Reference Material kelp (3232) and tomato leaves (1573a) for quality control and post-process standardization. The maximum absolute difference between the metal concentrations of run one and run two was taken to select by which reference material to adjust to control for experimental variation across runs. A ratio of that reference material was then applied to correct run one by the run two results.

2.4.4 Data analysis

Concentration of each metal was calculated by multiplying the initial ICP-MS parts per billion (ppb) value by the total volume digested, then dividing by the total mass of sample digested before converting to parts per million (ppm). The total mass of metal per litterbag was calculated by multiplying the metal concentration (ppm) by the total end mass on a dry weight basis. Percent mass remaining was calculated by dividing the end dry mass by its initial fresh weight adjusted by the average moisture content of the T0 samples (3.88%). We calculated bioconcentration factor as:

$$BCF = \frac{C_{tissue}}{C_{sediment}}$$

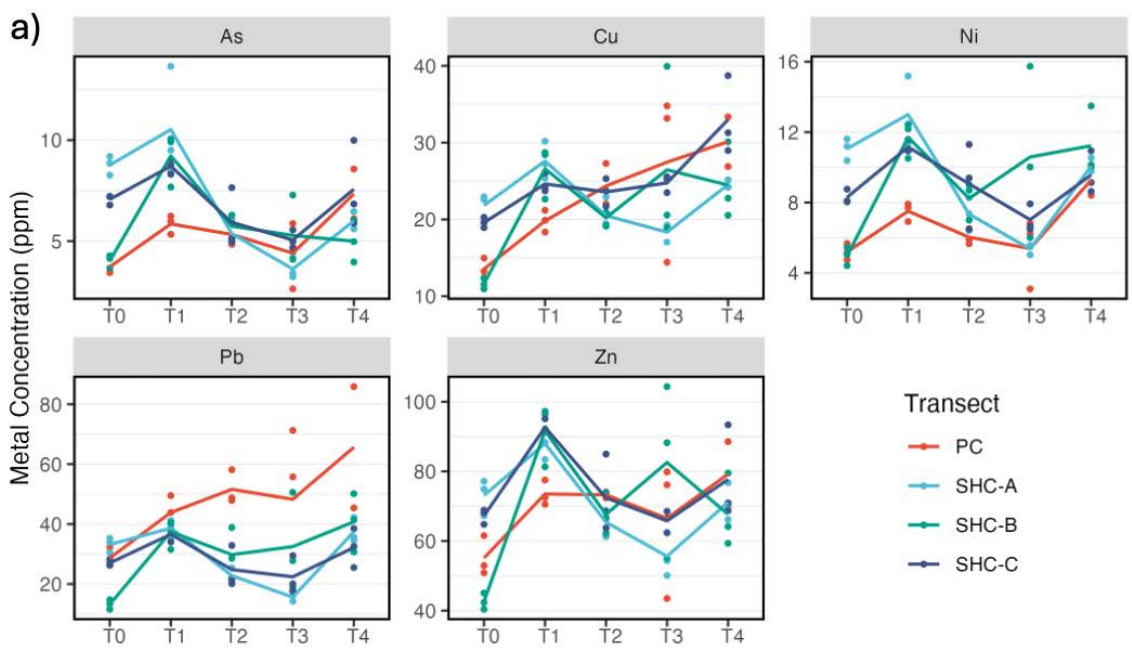
where $C_{organism}$ is concentration (ppm) of trace metals measured with ICP-MS within sampled *S. depressa* aboveground litter and $C_{sediment}$ is concentration (ppm) of trace metals measured using x-ray florescence (XRF). XRF data were collected in 2020 by sampling the top 10 cm of soil at 4 locations across the study site. XRF spectroscopy was used on oven-dried, homogenized sediment after being calibrated on NIST-1646a estuarine sediment standard. Arsenic was not able to be reliably quantified unable to be quantified due to expected spectral interference of high lead concentrations. Comparisons of ICP-MS and XRF data collected from the current litter samples yielded strong correlation within our value ranges suggesting that conclusions can be reliably drawn.

For the final sampling date (T5; June 13, 2024), only one replicate per transect was recovered—we suspect due to either a winter storm or a community shoreline cleanup that occurred shortly before. As a result, T5 was excluded from analyses to maintain consistency and ensure robust statistical comparisons across transects and timepoints. In addition, one sample which had *all five* tested metal concentrations > 1.5 times the interquartile range was removed as an outlier; samples with only two metal concentrations outside of the acceptable range remained in the data set.

Data were analyzed using R 4.3.2 version "Eye Holes" with packages tidyverse (Wickham et al., 2019) and emmeans and multcomp (Hothorn et al., 2008). Change over time to biomass, metal concentration, and total metal mass was determined by fitting linear models including timepoint and transect as dependent variables. Model fit was assessed by inspecting the Akaike Information Criterion (AIC), Bayesian Information Criterion (BIC), and residual sum of squares and selecting that which yielded the lowest values. To determine actual differences among the transects at individual timepoints, pairwise contrasts were conducted with Sidak adjustments to control for multiple comparisons. The statistical significance of the pairwise contrasts was assessed using threshold of 95% confidence; $p<0.05$. To account for the potential of autocorrelation in the data, we visually inspected autocorrelation function (ACF) plots with a maximum lag of 4 for each metal. Only the ACF plot for total mass of arsenic suggested evidence of autocorrelation at certain lags, with some lag lines extending beyond confidence intervals. However, when the residuals of the model were passed to the same ACF analysis, coefficients fell within the acceptable range, suggesting that autocorrelation was not a concern for the final model.

2.5 Results

Metal content analyses were performed for Arsenic, Copper, Nickel, Lead, and Zinc and reported as both concentration (ppm) (Fig. 2.2a) and total metal mass per litterbag (µg) (Fig. 2.2b). Initial T0 metal concentrations within the *S. depressa* litter across all transects averaged 5.90ppm As, 16.64ppm Cu, 7.38ppm Ni, 25.52ppm Pb, and 59.51ppm Zn. T0 metal concentration within the litter varied significantly by transect for all metal types excluding lead, which was lowest at SHC-B by over 50% but just above a 95% significance threshold (p=0.061) (Table 2.3). T0 As, Cu, Ni, and Zn concentrations were highest at SHC-A and SHC-C while the lowest concentrations of those metals occurred primarily at SHC-B followed by PC.



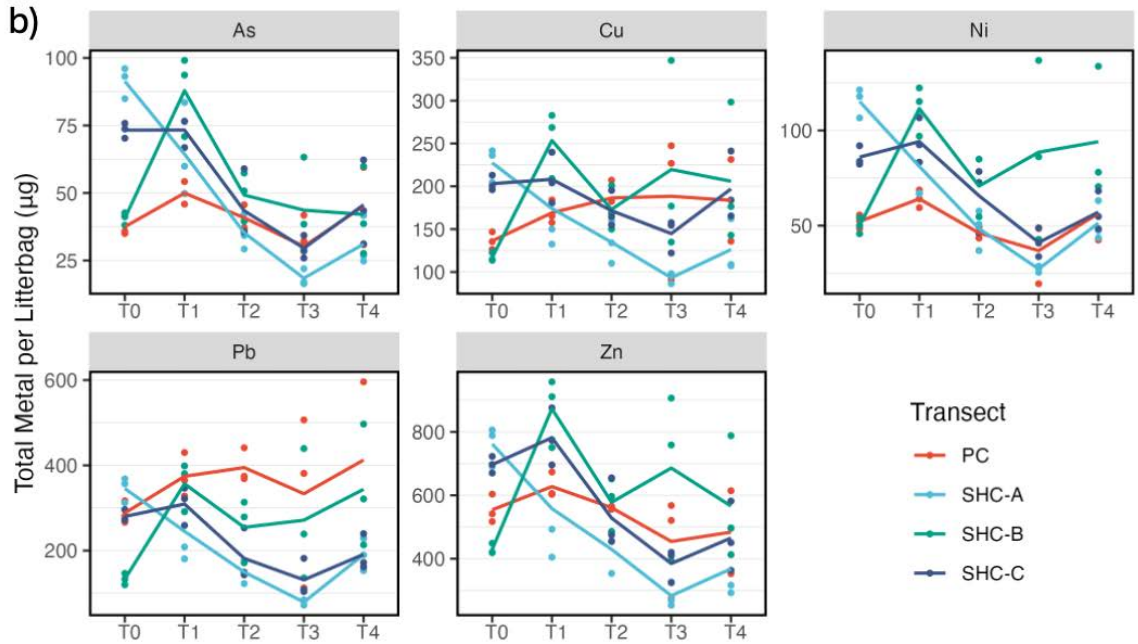


Figure 1.2: Metal concentrations in *S. depressa* tissue. a) Metal concentrations (ppm) in plant above-ground obtained from ICP-MS analysis. Concentrations are shown for each transect (PC, SHC-A, SHC-B, SHC-C) at each timepoint (T0-T4) with solid lines representing the mean and dots individual data points. b) Total metal mass (µg) contained within each litterbag across the same study and methodological parameters.

Table 2.3: T0 metal concentrations in above-ground *S. depressa* tissue representing total plant uptake at the end of the growing season when litter was collected. Metal concentration (ppm) was assessed using a linear model with interaction between transect and timepoint. Estimated marginal means) were calculated and pairwise contrasts were performed to compare the metal concentration between transects. The significance of these differences was determined using Sidak's adjustment controlling for multiple comparisons. The resulting group letters indicate statistically significant differences between transects with distinct letters denoting groups that are significantly different at a 95% confidence level.

	As		Cu		Ni		Pb		Zn	
Transect	ppm	group	ppm	group	ppm	group	ppm	group	ppm	group
PC	3.73	l	13.54	le	5.19	l	28.66	l	55.11	le
SHC-A	8.77	e	21.82	e	11.06	e	33.18	l	73.10	e
SHC-B	4.03	l	11.60	l	4.96	l	13.23	l	42.61	l
SHC-C	7.07	e	19.59	le	8.30	le	27.03	l	67.23	e

Overall biomass loss across all four transect during the six-month study period was estimated to be 37.81%, with a 95% confidence interval ranging 33.65 and 41.97%. Change over time of total metal mass varied based on metal type (Fig. 2.3). Total litterbag metal mass at the final sampling date (T4), expressed as percent metal remaining, was significantly higher than percent biomass remaining for Cu and Pb at $\alpha=0.05$ and Ni and Zn at $\alpha=0.10$ (Table 4).

The *S. depressa* litter remaining at the end of the study period varied by transect and transect:timepoint interactions, ranging from 80.93% mass remaining at SHC-B to only 50.73% mass remaining at SHC-A. Pairwise contrasts revealed that SHC-B consistently and notably retained the most mass at each time point with over 99% confidence ($p<0.005$). Contrarily, the fastest rate of mass loss occurred at SHC-A within the first six weeks. The rate of mass loss at SHC-C and PC was largely similar across the study period. Despite different end mass amounts, the rate of change at all sites was minimal after T3, with no significant loss of mass between T3 and T4 at any transect.

Table 2.4: Percent metal mass remaining compared to percent biomass remaining. For each metal, the difference from mean percent biomass remaining was calculated, and statistical significance was assessed using Z-tests with pooled standard errors. Percent metal mass remaining is held as different from biomass at the 0.05 level.

Type	% Remaining at T4	SE	Difference from Biomass %	p value
Biomass	62.2	2.1	—	—
As	75.2	12	13	0.286
Cu	113.4	17.2	51.2	0.003
Ni	99.8	19.4	37.6	0.055
Pb	127.6	29	65.4	0.024
Zn	82.7	12	20.5	0.093

Across all transects, average metal concentration increased from T0 to T1, with significant increases occurring at a 95% threshold for As, Zn but just below a 95% threshold for Cu, Ni, and Pb. Concentrations of As at all four sampling locations declined to their minimum at T3 followed by a plateau or slight increase at the subsequent T4 sampling. Similar trends are noted for Cu, Ni, and Zn barring concentrations at SHC-B which reached nadir at T2.

Concentrations of Pb remained consistent across the study with the exception of PC which yielded increasingly higher concentrations at each time point.

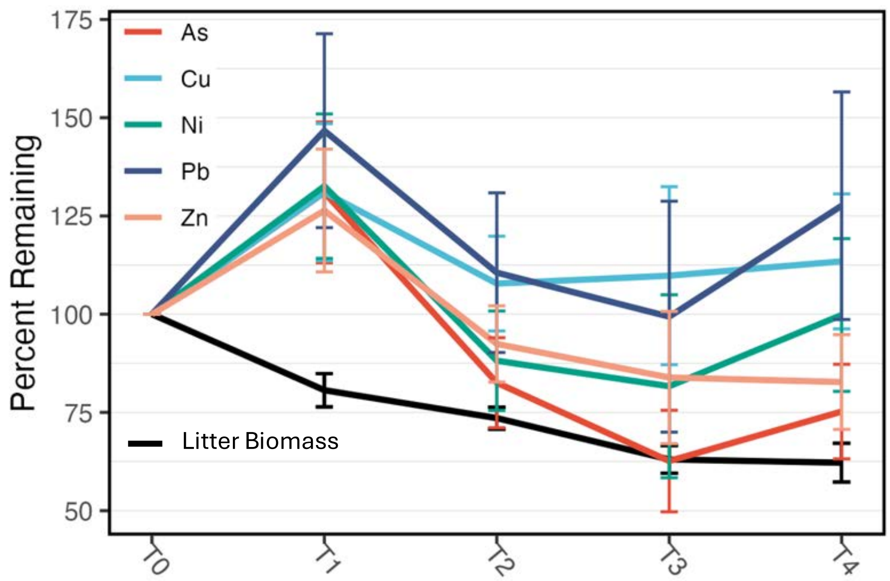


Figure 2.3: Average metal mass and biomass remaining per litterbag throughout the study period expressed as a percent related to initial values. Lines represent means with standard error bars and metal types differentiated by color and litter biomass represented with a black line.

Bioconcentration factor was estimated using the ratio of litter metal concentration with average site environmental sediment metal concentrations as surveyed in 2020 with XRF (see Table 2.2 for details). Plant shoot BCF at T0 was highest for Cu and lowest for Pb while highest BCF values in the study occurred at T1 for Ni and Zn and T4 for Cu and Pb (Table 2.5).

Table 2.5. Bioconcentration factors estimated for above-ground *S. depressa* litter based on background sediment contamination (ppm) measured at the site in 2020. T0 values were calculated by averaging across all replicates and transects while peak BCF was determined by averaging BCF for every sample across all replicates at each individual timepoint and transect before selecting the highest occurrence

	T0		Peak			
Metal	T0 BCF	Std. Dev	BCF	Std. Dev	Transect	Timepoint
Cu	0.34	0.10	0.68	0.11	SHC-C	T4
Ni	0.20	0.08	0.36	0.05	SHC-A	T1
Pb	0.15	0.05	0.39	0.17	PC	T4
Zn	0.30	0.07	0.46	0.01	SHC-C	T1

Cores taken at each transect yielded bulk densities ranging from the highest of 2.063 g/cm^3 at SHC-B, followed by 1.53 g/cm^3 at SHC-C, and finally 1.06 and 1.01 g/cm^3 at SHC-A and PC, respectively.

2.6 Discussion

Plants of the genus *Salicornia* are found in salt marshes distributed throughout the world. While some species are well studied in a laboratory setting and provide a basis for the ecological importance of these halophytes, knowledge gaps exist for the characteristics and functioning of *S. depressa* in the northeastern United States. In particular, research into the role of this species in metal cycling within highly contaminated urban marshes in a practical, field setting is extremely limited. The questions this study sought to answer, including how metal is stored within the plant and efficacy of metal retention during decomposition, are critical for understanding not only the importance of *S. depressa* in within marsh ecology but also how marshes serve as effective contaminant sinks in urban spaces.

Overall, initial metal content from the recently senesced *S. depressa* litter was considerably lower than expected based both on other studies of *S. depressa* accumulation and suspected sediment concentrations of heavy metals. Previous studies have found *Salicornia* spp. able to accumulate orders of magnitude higher than our measured T0 concentrations (>100x for As, Cu, Ni in the most extreme accumulation scenarios). The differences between our study and those previous may be due to variation in plant species or more notably by differences in experimental design. Inundation by high salinity seawater affects the mobility and uptake of trace metals by plants (Singh et al., 2016), but tidal conditions are notoriously difficult to simulate in a greenhouse setting. Also difficult to capture without field studies are the naturally occurring microbe-metal interactions in high organic matter marsh sediment and plant competition that influence plant metal uptake (Mesa-Marín et al., 2020; Khatoon et al., 2024;). An observational study by Nel et al., (2023) of *S. tegetaria* uptake under field conditions yielded ranges of plant accumulation for Cu, Ni, Pb, and Zn within which our results fall.

Additional considerations for why overall metal concentration within the fully mature *S. depressa* biomass is lower than its biological capacities may be due to the tissue type sampled and life stage. Tissues collected in this study were solely above-ground shoot material. Though other studies that monitored uptake of similar species found highest concentrations in shoot tissue, these peaks often occurred during early vegetative growth stages (Smillie 2015). End of life cycle accumulation rates are found to be higher in the root tissue (Nel et al., 2023) which suggests evidence of plant translocation during later growth stages. Studies of non-marsh species have found similar discrepancies in laboratory accumulation potential and uptake in a field setting (Brunetti et al., 2011).

Despite lower than expected metal concentrations within *S. depressa* shoot tissue, results provide evidence that metal is well retained by *S. depressa*. At the end of the six-month study period, biomass decreased by ~38% while total As μg decreased by only ~25% and Zn by ~17%. Ni stayed relatively constant, while Cu and Pb masses were actually measured as increasing to ~113% and ~128% of initial values respectively. The retention of metal content throughout the study provides compelling evidence that *S. depressa* accumulates trace metals in the cellulose and lignin-rich tissues that endure well throughout the winter as opposed to readily degraded labile tissues. Decomposition of salt marsh plant litter occurs in stages, with degradation of non-structural tissues and leaching of soluble compounds occurring within the first month and remaining litter degrading slowly over months and years (Valiela et al., 1985). Supporting the conclusion that metals, once accumulated within the plant, are bound more tightly to more durable tissues is the measured universal increase in metal concentration (ppm) from T0 to T1.

In addition to the evidence that metal was retained and concentrated in the litter, environmental factors appear to have recruited additional metal. Though the plant was no longer alive to actively accumulate trace metals, within the litter total Cu mass was 113.4%± 17.2 and total Pb was 127.6%±29.0 higher than initial values and was retained more than biomass by 51.2% ($p=0.003$) and 65.4%($p=0.024$). This phenomenon has been discussed by Windham et al., (2003) and Giblin et al., (1980) with differing opinions on specific mechanisms for the litter metal enrichment (sorption by metal-rich organics from either water or sediment) but agreed that factors outside of the litter itself are at play. In fact, the estimated BCF of Cu and Pb, metals known to have a high affinity for organic matter and fine marsh sediments, peaked at the final sampling date, nearly six months after deployment and over seven months after harvest. Conversely, the BCF of Ni and Zn, metals known to be more mobile under conditions found in a salt marsh environment, peaked at far earlier at T1 but still after the exposure of the litter in the field.

In our study, one exception to this trend was SHC-A which showed continual decreases in total metal mass despite increasing metal concentration over the same time frame. Our initial hypotheses would suggest variation in transect sediment characteristics would account for this variation. However, measured bulk density values and observed organic matter content at SHC-A fall within the full range spanned by SHC-B (rocky, high bulk, low OM) and PC (silty, well-vegetated, low bulk) which instead showed trends more similar to each other. It is then, perhaps more tied to the overall decomposition rate, which though influenced by sediment characteristics, also encompasses microbial activity and oxygenation. The greatest total biomass loss occurred at

SHC-A during the first six weeks which may have included hardy tissues not decomposed at the other transects. The rapid decomposition could have occurred due to the alignment of sediment organic matter, wave exposure, and continuous aerobic conditions due to lack of biofilm and mud that was noted at SHC-C and PC.

Transect level differences were also observed in initial metal concentrations. Initial metal concentrations, representing metal contained in the plant shoot at senescence, were predominately higher at SHC-A and SHC-C, with consistently lower values measured at SHC-B and to a lesser extent at PC. Low initial concentrations at SHC-B were in line with our hypotheses based on transect characteristics including a high bulk density, rocky shore, lack of organic material, and sediment winnowing which prevent retention of contaminants. However, similarly low concentrations at PC were unanticipated. PC is visibly characterized as a more traditional marsh with dense vegetation, steep marsh bank slopes, and an extremely low bulk density. Characteristics of PC associated with metal retention may also foster conditions that reduce plant uptake, namely binding of positively charged trace metals in the organic-rich sediment or precipitation of metals as sulfides due to anaerobic conditions reducing bioavailability. It is important to note that the distance between transects is minimal (<0.25 mile) and that variations in sediment characteristics are a product of exposure to deposition and erosion rather than different parent material.

Measured litter concentrations and total mass of metal showed an increase at either T3 or T4 across almost all metal x transect combinations. This increase coincided with observed occurrences of sprouting within the litterbags and air temperature increases to above 5°C known for *Salicornia* germination in New England (Doncato, 2025). These results, paired with the literature on the rate of plant uptake during early growth stages, provides a compelling story of metal retention and stabilization by *S. depressa.* Metal concentrations and total metal mass within the litter remained high, despite loss of biomass throughout the winter season. By the time more thorough decomposition of hardy tissues occurs in a warming spring, the next generation of plants is germinating, accumulating, and continuing a closed loop of metal cycling. Research is underway at UMass Boston on *S. depressa* uptake and metal partitioning that will answer key questions about plant translocation and peak accumulation potential in the field.

2.7 Conclusion

Implications from these findings highlight another important ecosystem service provided by salt marshes. The accumulation and immobilization of heavy metal contaminants by halophytic plants like *S. depressa,* clearly continues well-beyond their own growing season. The closed loop of metal cycling by *S. depresssa* and stabilization of contaminated sediment by plant root matrices contribute to salt marshes serving as contaminant sinks in urban environments.

Results from this study and those that follow will also provide important considerations for the role of metal remediating plants in facilitating establishment of primary and secondary successional species. In New England marshes, *Salicornia* spp. in particular function as early colonizers after disturbance events due to their tolerance of harsh conditions (Bertness and Ellis, 1987). Immobilization of trace metals by *S. depressa* well into spring that might otherwise prevent germination and establishment of more abundant marsh species including *S. patens* and *S. alterniflora* is not to be discounted. *S. depressa,* metal cycling, and marsh successional dynamics must also be considered as changes to mean sea level alter the ecological niche marshes are able to occupy. Marsh landward migration as a successful ecological strategy for adapting to sea level rise is highly likely in healthy, well-vegetated marshes that are tolerant of environmental shock (Fagherazzi et al., 2019). With legacy metal pollution, contaminants of emerging concern, increased urban and environmental pressures, understanding the role of each species in fostering successful, resilient marshes is critical for the continued well-being of our ecological and social communities.

CHAPTER 3

PATTENS COVE: FUTURE THREATS AND OPPORTUNITIES FOR AN URBAN SALT MARSH

3.1 Introduction

Wetlands across the United States have historically been undervalued and even seen as wastes of space. Once considered solely to be hosts of nuisance odors and pests that threatened public health, wetlands have been filled in and drained dry since colonial times (Lewis, 2001). It was not until the mid-19th century that new knowledge of habitat provisioning and stormwater management ushered in an appreciation of the ecosystem services offered by wetlands (Dahl and Allord, 1997). The expanded understanding of these ecosystems was codified into law under the provisions of the Clean Water Act in 1972. Though this and subsidiary state and municipal regulations provide robust protections against the removal, filling, dredging or altering of any protected resource area, wetlands have still degraded over time due to industrial contamination, climate change, and other anthropogenic pressures. In the 21st century, perhaps the most pressing threat to coastal wetlands is sea level rise (IPCC, 2023).

Sea level rise as a result of human-influence can be seen in every shoreline around the world. High tides compounded with storm surges and wind and waves are driving water into new places and stressing both infrastructure and communities. With wetlands serving as both mitigator and adapter for the drivers and outcomes of climate change, their continued survival is paramount. This need is heightened when considering urban settings in which anthropogenic pressures are compounded and green space is severely limited and degraded. Current plans for the Boston Metropolitan area attempt to address these challenges but fall short on mechanisms for implementation (City of Boston, 2020). On account of the breadth and complexity of the issue, assessing change and evaluating responses in one urban marsh as an example can provide clarity of these challenges and what the future may hold.

3.2 Sea Level Rise and Marshes

Global assessments conducted by the Intergovernmental on Climate Change have found unequivocal evidence that human influence is changing the ocean. Global mean sea level increased 0.66 ft (0.20 m) between 1901 to 2018 with projections suggesting this rise will continue and increase in the future (IPCC, 2023). This “unavoidable” sea level rise and associated change to the increasing landward extent of seawater will have far reaching impacts on ecosystems and biodiversity in addition to associated human costs. Coastal wetlands, and in particular salt marshes, are vulnerable to rapid sea level rise that surpasses inundation tolerance thresholds for low marsh plants that effectively drown. Salt marsh platforms are comprised of estuarine sediment deposited by river or tidal action, decaying organic material from plant cover and wrack, and the root structures of plants that inhabit the marshes (Bricker-Urso et al.,1989; Friedrichs and Perry, 2001). Marsh survival requires the rate of organic matter assimilation and sediment accretion to match or outpace sea level rise in order to survive (Fagherazzi et al.,2012). These conditions are exacerbated by severe storms and high wave energy that can cause large erosional events and bank sloughing, rapidly eating away the seaward edge of the marsh. Other compounding factors of anthropogenic pressure on marsh environments include eutrophication (Deegan et al.,2012), plant competition and invasive species introduction (Silliman et al.,2009), and changes to sediment circulation (Day et al.,2011), all of which can undermine the stability of the marsh platform.

As tidal range increases with sea level rise, marsh plants may be able to colonize higher elevations that become newly exposed to seawater inundation (Morris et al., 2002). The salt marsh plant community is split into categories based on tolerance to seawater intrusion which creates natural zonation of low and high marsh environments. As high tide increases in elevational reach due to sea level rise, typical New England high marsh plants (*Spartina patens, Juncus gerardi, Distichlis spicatai*) may expand into new ranges allowing continued success of the low marsh. Though the physical characteristics of these new niches may meet baseline plant requirements, their success in establishment and survival is compounded by factors like the rate of sea level rise, competition with resident species, and population genetic diversity (Bertness and Shumay, 1993; Morris et al.,2002; Vahsen et al, 2023). In urban settings, physical obstructions like coastal armoring (bulkheads, sea walls, revetments, berms) and infrastructure (roads, residences, industry) can prevent successful landward colonization squeeze (Torio and Chmura, 2013). Urban pressures like industrial contamination, increasing runoff from impervious surface, and dredge activities may exacerbate these challenges. As a result, from colonial settlement to the 1980s it is estimated that the United States lost 118 millions acres, or over 50% of wetlands (Dahl and Allord, 1997).

3.3 Marshes of Boston

Within close proximity to Boston Harbor, salt marsh area has already undergone significant change. Marsh survival in the region has been affected by land making efforts, dredging, contamination, and other environmental complications. Since the first impacts of colonial times, thousands of acres of marsh and tidal flats were filled in or drained across Boston (Seasholes, 2003) with estimates of up to 81% loss since 1777 (Bromberg and Bertness, 2005). Large swaths of salt marsh that remain (e.g. Belle Isle Marsh, Neponset Marsh, Rumney Marsh) provide critical ecological and public well-being services including habitat for shellfish and migratory birds, stormwater drainage, contaminant filtering, storm surge abatement, and aesthetically valuable urban green space (Fig. 3.1). However, often unaccounted for and understudied are much smaller fringing urban marshes. These so-called “pocket marshes” were inventoried by Bistany (2023) who found over 3 million square feet of these urban fringing wetlands along the Boston metropolitan shoreline. Though afforded the same protection as larger wetlands under the WPA, these small marshes are at greater risk of loss. Often abutting roadways, revetments, and residences, these fringing marshes have less leeway for lateral escape and face high occurrence of coastal squeeze. Additionally, with higher edge-to-interior ratio, these fringing marshes face severe edge effect syndrome that includes erosion and bank sloughing while the marsh plant communities face alternate edge effects such as extreme conditions, invasion and predation, and degradation which undermine overall marsh stability (Gittman et al., 2017; Rippel et al., 2020).

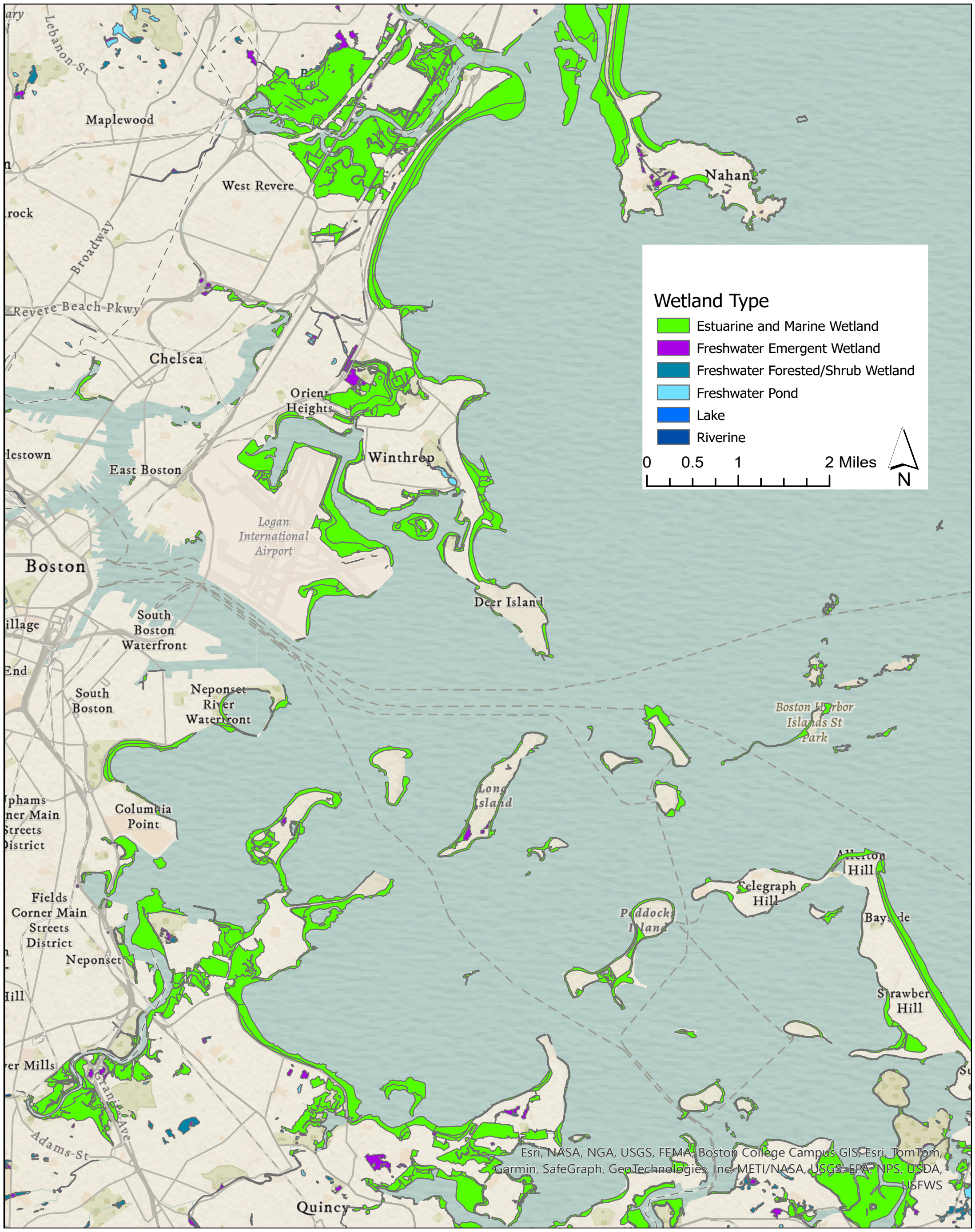


Figure 3.1: 2008 National Wetlands Inventory prepared by the U.S. Fish & Wildlife Service that maps extent of wetlands using interpretations of 1 meter or less digital, true color imagery.

Aside from provisioning many of the same ecosystem services as their larger siblings, fringing marshes provide other key resources in urban settings such as protected green and natural spaces in otherwise developed/developing places. Many marshes and their protected buffer zones around the Boston shoreline serve as publicly accessible green spaces like Pope John Paull II Park, Dorchester Shores Reservation, and the Condor Street Urban Wild. It has been shown that proximity to green space can increase mental health (Nutsford et al., 2013) and raise property values (Tajima, 2016). Therefore, when evaluating challenges that marshes face due to climate change and sea level rise, it is important to consider not only widely accepted ecosystem services but also their role within communities as a public good. Using one Boston Harbor fringing marsh as a case study, we can consider all lenses of change, encompassing physical, biotic, and anthropocentric as the marsh is exposed to sea level rise and other urban pressures.

Pattens Cove, a 9.6 acre protected open space in the Savin Hill Cove neighborhood of Dorchester, MA is owned by the state Department of Conservation and Recreation (DCR) (Boston PRD, 2015). Once primarily tidal flats, Pattens Cove was partially filled in starting in 1925 with the expansion of nearby infrastructure to bridge Savin Hill Cove (Fig. 2). On any given day, one can see residents strolling through, walking their dogs or cutting across to the nearby Morrissey Boulevard. The Savin Hill neighborhood is home to approximately 15,000 residents, with historic homes converted into multifamily living. The homes that overlook Pattens Cove, most valued near or over a million dollars (Zillow) enjoy partial tree cover and easy access to a grassy park. The transition to marsh is, at first glance, hard to discern due to occasional over-mowing, but characterized by plants typical to the region (*S. alterniflora, S. depressa, S, patens*). Water enters the cove from Dorchester Bay through a culvert that passes under Morrissey Boulevard and from a drainage outlet for the nearby parking areas that directs precipitation runoff to the northwesternmost part of the marsh. The semidiurnal tides, often with a range of 10 feet or more, cause high inundation twice a day—during astronomically high tides, many walking access points to Pattens Cove are obstructed with flood water. Pattens Cove has been impacted by many infrastructural changes and expansions in the city including the construction of Calf Pasture Pumping Station, Old Colony Railway, and UMass Boston (Fig 3.3).

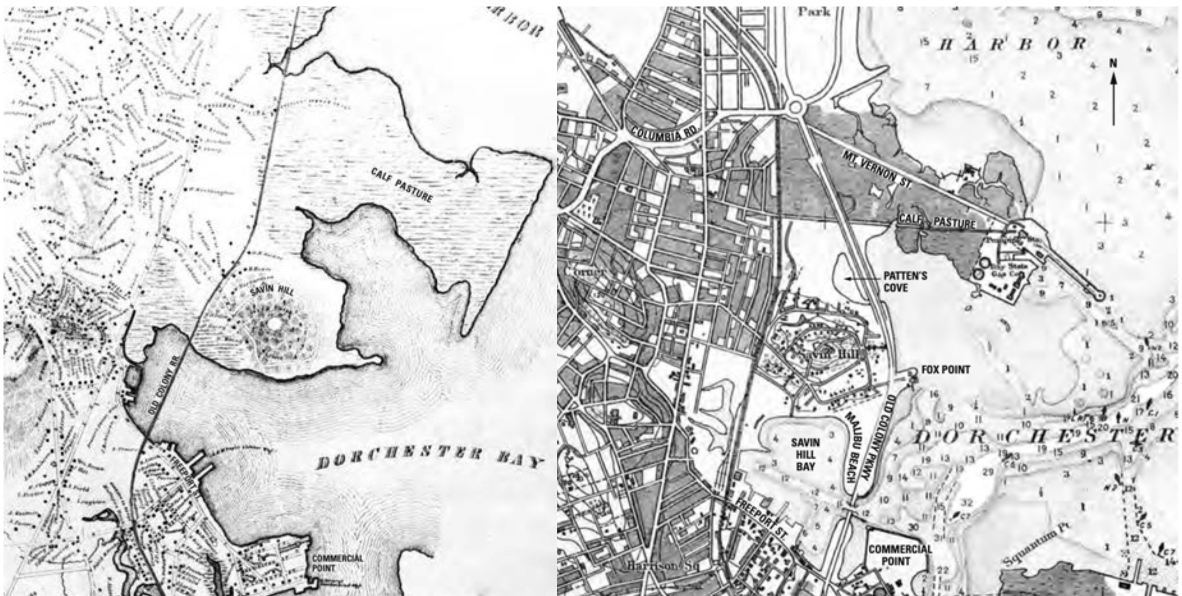


Figure 3.2: Savin Hill historical maps and tidelines. Top left: Detail from 1850 map of Dorchester with Calf Pasture (now Columbia Point) still an uninhabited marsh. Nancy Seasholes, *Gaining Ground* (2003). © Massachusetts Institute of Technology
Top right: Detail from 1934 chart of Boston Harbor that shows the 1920s filling for the Old Colony Parkway (now Morrissey Boulevard) across the Calf Pasture, Patten's Cove, and Savin Hill Bay. Nancy Seasholes, *Gaining Ground* (2003). © Massachusetts Institute of Technology
Bottom left: 2023 aerial imagery of Dorchester and Pattens Cove. Red lines are historic high water derived from city archives by MassDEP to determine presumptive lines of Chapter 91 tidelands jurisdiction.

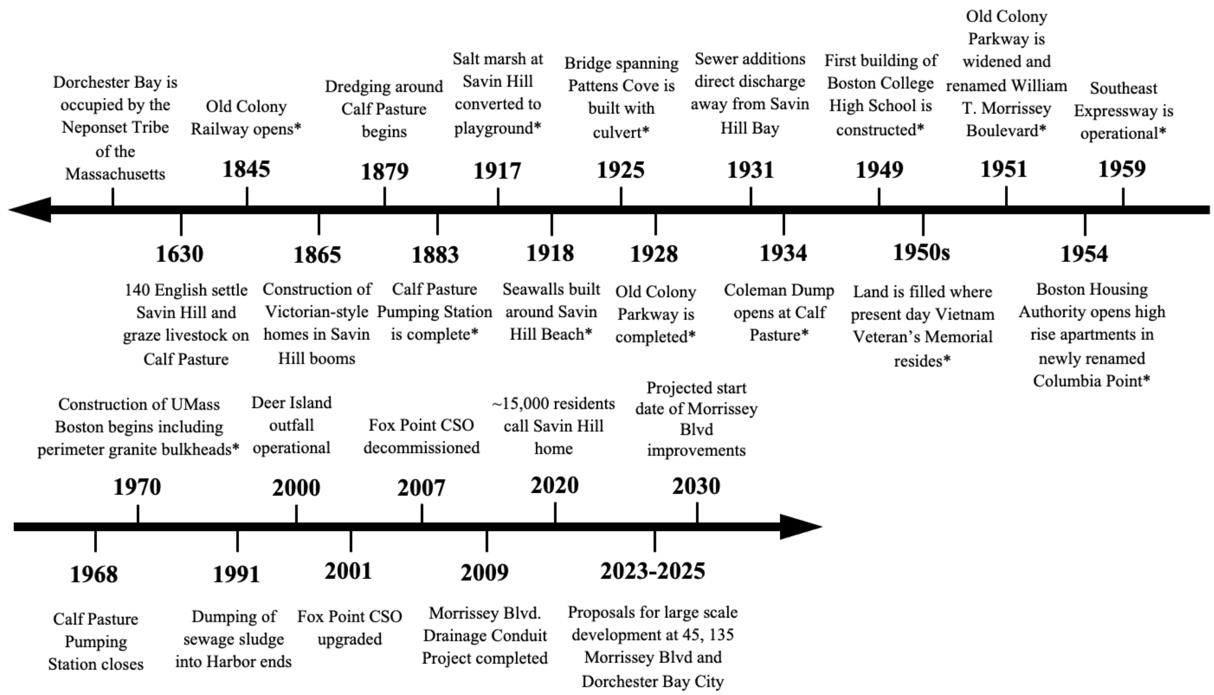


Figure 3.3: Timeline of key industrial and infrastructural changes in Dorchester. Asterisks indicate occurrence or likely occurrence of filling.

3.4 Observations

Boston, in an effort to prepare for and adapt to expected sea level rise, has prepared climate change reports detailing local level impacts and adaptations. These "Climate Ready" plans, based on work by Douglas and Kirshen (2022), estimate that, under intermediately conservative projections, Boston will experience 9 inches of sea level rise by the 2050s and as much as 36 inches by the end of the century. Alongside these projections, the city has modeled associated high tide under these varying conditions that show the expected extent of overland flooding (Fig 3.4). Already, there is strong local evidence that sea level rise is occurring. NOAA observations from the nearby Fort Point Channel have found, with 95% confidence, that mean sea level has increased in the Boston area, on average 2.97 mm or 0.01 ft per year since 1924 (NOAA Tides and Currents, 2024). However, looking more closely at trends of sea level in the past two decades suggests that trend has changed in recent decades. Relative to the current National Tidal Datum Epoch (1983-2001), mean sea level of the past three years (2022-2024) has been 0.53 ft while that of 2002-2004 was only 0.08 ft, suggesting an increase of 0.45 ft over 20 years (Table 3.1). This result is greater than the expected 0.20 ft based on a 0.01ft per year change and supports similar findings that rate of sea level rise is increasing (Douglas and Kirshen, 2022).

Table 3.1: Average mean sea level (ft) in Fort Point Channel, Boston, MA from hourly observations at NOAA buoy 8443970 in using the tidal datum based on the current National Tidal Datum Epoch.

Year	Avg MSL (ft)	Average
2002	0.03	
2003	0.09	0.08
2004	0.11	
2022	0.35	
2023	0.63	0.53
2024	0.59	

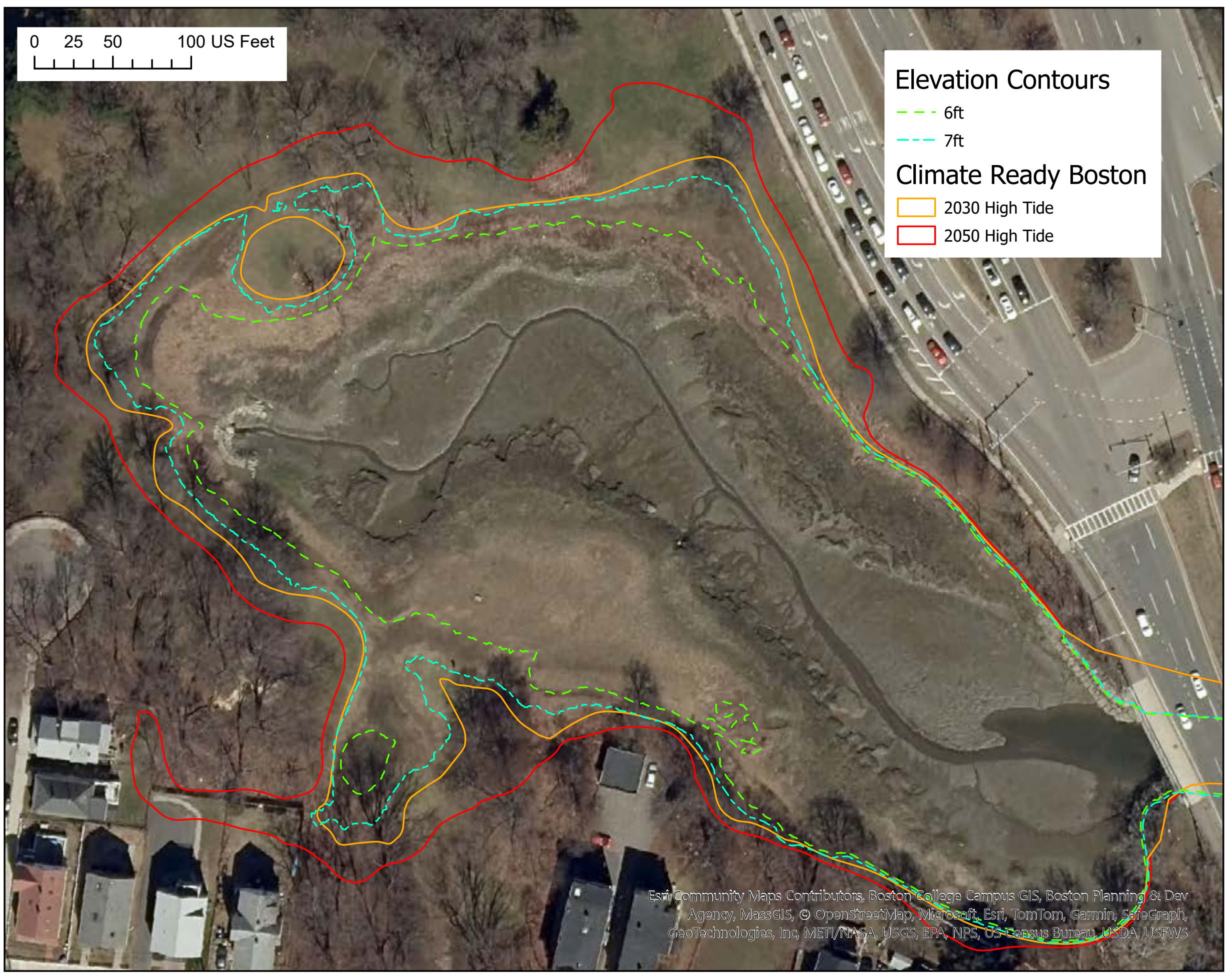


Figure 3.4: Pattens Cove elevational contours of marsh and tidal land with extent of an April 2024 high tide as compared to Climate Ready Boston high tide projections. Elevation contours are constructed from 2021 Mass LIDAR data and Climate Ready Boston high tide projections are based on sea level rise of 9in for 2030 and 21in for 2050 based on a MassDOT-FHWA analysis. Basemap is USDA NAIP aerial imagery from Spring 2021 (15cm resolution).

The impact of this high rate of sea level rise can be seen at Pattens Cove at any given high tide. At 12:25 pm on April 9th, 2024 at the highest tide of the day, photographs were taken of the extent of flooding at Pattens Cove (Fig. 3.5). At the same time, nearby flood sensors at Tenean Beach observed 6.1 ft of overland flooding above MSL. Corresponding 1' elevational contours constructed from MassDEP LIDAR (2021) highlighting 6' and 7' elevations provide an estimate for extent of this high tide under "bathtub" conditions. This high tide flood event, with no precipitation or storm surge, has only one foot less lateral reach than Climate Ready predictions for decades from today. Storm surge and high exceedance flood probabilities would greatly extend the reach of these higher water levels, with potential for compound flooding should a storm event coincide with a high tide. Though it is critical to know the extreme extent of flooding for public safety and urban planning, the amount of time these elevations are submerged in seawater is likely a better driver of marsh plant success. Using the same NOAA mean sea level data for 2002-2004 and 2022-2024, cumulative hours inundated is able to be calculated for each elevational interval (Table 3.2) which provides strong evidence that higher elevations are being saturated for increasingly longer periods of time.

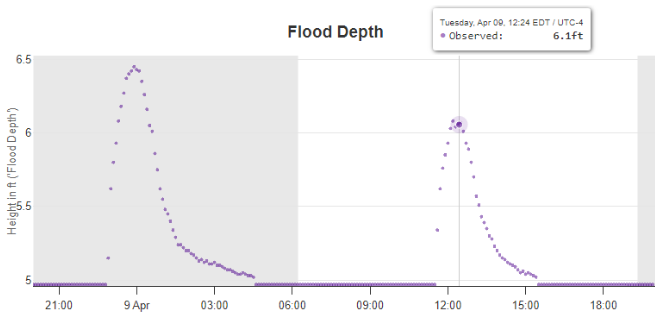


Figure 3.5: Photographs taken of high tide at Pattens Cove at 12:24pm on April 9, 2024 (above). Multiple photos were taken from the same location and aligned to create a panoramic view of high tide flooding. Note the almost completely submerged stone bench which, at other times of day, is dozens of feet from water.
Water level data (feet above MSL) at nearby Tenean Beach collected by Hohonu overland, supersonic flood sensor managed by Woods Hole Group from the same time frame (left)

Table 3.2: Inundation of elevational intervals with seawater for tidally influenced sites around Boston Harbor. Observed water levels are from NOAA buoy 8443970 at Fort Point Channel, Boston, MA and were aggregated in 1-hour intervals measured in feet using the tidal datum of mean sea level based on the current National Tidal Datum Epoch. Hours are cumulative—for example, if the recorded MSL was 7.5 feet, one hour is counted for that elevation and all intervals below it.

	Cumulative Hours Inundated								
Elev.(ft)	2002	2003	2004	Average	2022	2023	2024	Average	20 Year Diff.
0-2	8219	8310	7957	8162	8649	9009	8971	8876	714
2-4	5309	5370	5084	5254	5784	6195	6078	6019	765
4-6	1666	1806	1615	1696	2106	2552	2460	2373	677
6-8	47	63	54	55	83	117	147	116	61
8+	1	1	0	1	2	1	7	3	3

To determine if there has been an associated change in salt marsh footprint, aerial imagery collected by Massachusetts and the USDA provides opportunity for visual comparison of Pattens Cove over a similar time frame (Fig. 3.6). Assessing coastal wetland change from imagery is made challenging by the vast difference in tidal height and plant growth depending on the time of day and year that images were acquired. Inspection of the 2003 and 2021 image metadata reveals that, though they were collected at alternate ends of the growing season, meaningful visual conclusions can be drawn by aligning the images on known static infrastructure. Looking first at the seaward edge of the marsh suggests a widening of the tidal mudflat and, notably, a breakdown and "winnowing" of the bank edges in the 2021 imagery. Evidence of bank sloughing and degradation is particularly evident on the narrowest section of the Pattens Cove marsh channel. It becomes more difficult to assess marsh change at the landward edge due to differences in color scale and pixel quality which is compounded by suspected differences in park vegetation maintenance. Instead of relying solely on aerial imagery of difficult to differentiate vegetation, delineations conducted by surveyors and submitted to the city as required by the Massachusetts Wetlands Protection Act (WPA) can help to clarify if the marsh has expanded landward.

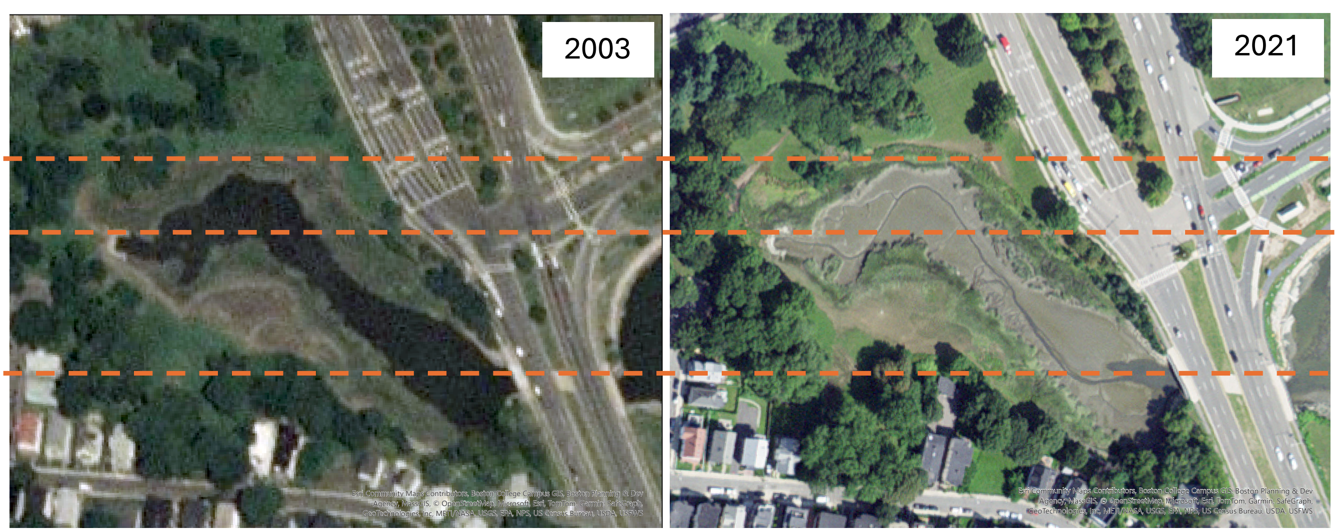


Figure 3.6: Aerial imagery depicting change at Pattens Cove at two different time periods. The 2003 image was taken on August 14, 2003 by the USDA National Agriculture Imagery Program with a 1m GSD. The 2021 image was taken on April 4, 2021 with 15cm resolution funded by MassDOT, the State 911 Department, and the Executive Office of Technology Services and Security. Both photos are projected in NAD 1983 (2011) UTM Zone 19N.

Under the WPA, any residential or commercial project must be permitted by the local Conservation Commission that will impact: a protected wetland resource, the 100 ft buffer around these resources, or any designated flood zone (310 CMR 10.00). As required for approval, plot plans that delineate these resource areas must be submitted and include verification of the boundary through presence of facultative wetlands species, evidence of hydric soils, and hydrologic indicators (U.S. Army Corps of Engineers, 2012). For example, as part of the permitting process for projects at 135 Morrissey Blvd, plot plans were submitted in 2018 that delineate the marsh at Pattens Cove (the buffer of which is outside of their proposed working boundaries). Georeferencing this plot plan to the aerial imagery suggests that the landward extent of the marsh primarily follows along the 5' elevation contour. (Fig. 3.7). 2005 land cover surveys using aerial imagery conducted by the DEP as part of their wetland inventory found the landward edge of the marsh to occur primarily below the 5' elevation contour, although there are some areas of overlap. While the 2005 survey is admittedly limited in its accuracy without ground-truthing, the differences in landward edge of the Pattens Cove marsh is clear between 2005 and 2018.

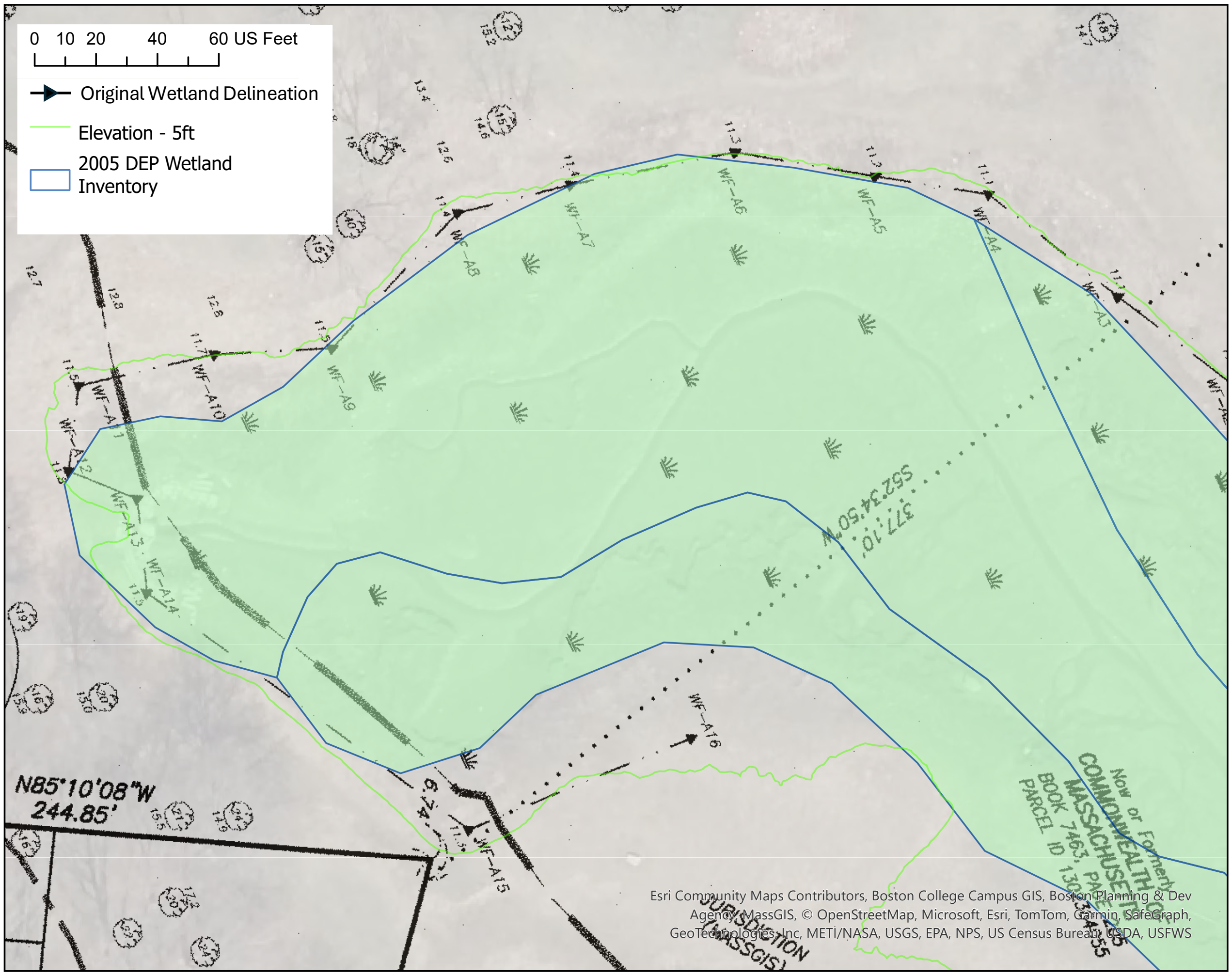


Figure 3.7: Georeferenced wetland delineation prepared by Feldman Land Surveyors on January 8, 2018 and submitted as part of a Notice of Intent to the Boston Conservation Commission and MassDEP on May 23, 2018 for improvements at for 135 Morrissey Blvd. Surveyed marsh delineation was georeferenced in ArcGIS Pro and falls primarily alongside the 5' elevation contour constructed from 2021 Mass LIDAR. This is contrasted with the extent of the 2005 MassDEP wetland inventory which is an available shapefile determined by photographic interpretations.

3.5 Implications and Urban Planning

While it does appear that the marsh is, to some extent, successfully colonizing higher elevations as mean sea level increases, it is unclear how viable this is under increasing rates of change. Coastal squeeze will prevent the marsh from matching sea level rise past the developed areas of Savin Hill, but complications will likely arise well before this occurs. Astronomically high tides and wind-induced storm surge will move water closer to residences and change the physical characteristics of the underlying soil (Douglas and Kirshen, 2022). Localized urban planning and other infrastructural projects taking place in the neighborhood and along Morrissey Boulevard incorporate these emerging challenges into varying degrees of severity and foresight. In particular, the mixed-use commercial lot adjacent to Pattens Cove, known as the BEAT, has expanded its footprint in the past few years. With improvements conducted in 2021 to revitalize the space and increase its function, the property outlets their surface water drainage into Pattens Cove (MassDEP File No. 006-1597). Recent permit requests propose an additional 6-story wing with 305,000 sf of office and laboratory space (BPDA, 2024). While there is limited change to impervious surface as currently proposed, increased density of use and diversion of more untreated runoff into Pattens Cove could affect marsh success. Though the pre-land making, historic high-water line (surveyed well before the industrial era and associated climate impacts) runs directly underneath the BEAT property, new proposals continue to request parking below grade (current plans have construction of three levels accommodating 417 vehicles). Immediately to the north of the BEAT property, filings for a high-density mixed-use development are actively under review. Current schematics include two 18-story residential towers with 754 apartments and 414 parking spaces below grade (BPDA, 2025).

Explicitly discussed in many area project proposals is Pattens Cove, emphasized as a public green space for new tenants to enjoy and as a connection point to the local community. In fact, a "Community Development Agreement" clause within the BEATs' obligations to receive construction approval was to assume a maintenance role over Pattens Cove, valued at $50,000 a year (Fig. 3.8; Smith, 2018). Though Pattens Cove is valued a marketable asset to the community and boon for developments, true consideration of its role in the future is limited. Developers seeking city approval for construction permits at both discussed locations do propose building above legally mandated base flood elevations and adding in stormwater infiltration pathways. However, the scope of these proposals does not suggest true connectivity to the landscape and climate resilience. For example, at the northern site, underground parking lots are proposed to have deployable flood shields and pumping systems to remove floodwaters (Dorchester Reporter, 2025). At a larger urban planning scale, floodwater redistribution has been questioned as a potential environmental injustice, unilaterally directing floodwaters from economically advantaged areas to those without means to implement similar measures (Liao et al.,2019). On a project level scale, emphasis on individual property rather than what will happen to the entire community in a flood scenario could play out in a similar way in this EJ-designated area.

> The Applicant will "adopt" the Department of Conservation & Recreation ("DCR") owned public open space known as Patten's Cove adjacent to the Project for a period of no less than ten (10) years. The Applicant will provide maintenance and general care to the open space that will include turf management, fertilization, and tree pruning. The final scope and duration of the maintenance plan and any improvements shall be determined and specified through an agreement with the DCR and/or other applicable state agencies and the Applicant. A copy of the final agreement governing the maintenance and any improvements shall be provided by the Applicant to the BPDA upon execution.

Figure 3.8. Adoption of Pattens Cove presented in subsection B.6.b. of the Cooperation Agreement between the Boston Planning and Development Agency and the developers of 135 Morrissey Blvd made on October 23, 2018.

At a municipal planning level, the Climate Ready plans for Dorchester proposed by the city provided three main climate change adaptations for the area around Pattens Cove, 1) raise the abutting intersection on Morrissey Boulevard, 2) add a tide gate which could restrict flow of water into the cove, 3) construct a berm between the marsh and the neighborhood of Savin Hill (2020). While raising the roadway was inevitable for continuity of transit and emergency response, addition of a tide gate and berm have yet to occur. A tide gate, able to restrict flow of exceptionally high waters and therefore protecting the homes adjoining Pattens Cove, if operated correctly, could serve the community well in the short term. Contingencies need to be prepared for if and when water reaches levels the tide gate would need to remain closed permanently and potentially cause longer lasting and deeper flood events (Walsh and Miskewitz, 2013). Additional ecological concerns of restricting seawater intrusion include loss of native plant species, marsh degradation, colonization of invasives like *Phragmites australis*, and changes to critical sediment and nutrient fluxes (Johnson and Hutchins, 2023). Such grey infrastructure is also notably undermaintained in the United States and existing structures are in various states of disrepair, with lack of appropriate operations and management plans (Johnson and Hutchins, 2023). The berm, an adaptation proposed on a longer time scale, has limited design specifications. A primary concern with such a berm is how it would obstruct marsh migration. Research has suggested that areas landward of existing earthen berms can collect pools of water that prevent revegetation and can restrict flow of sediment which stifles marsh accretion (Mora and Burdick, 2013).

Challenges posed by sea level rise on marsh success, associated ecosystem services, and potential loss of cherished urban green space need to be addressed on a holistic scale that looks much farther into the future. Marsh restoration and proactive management would see the stability of the marsh at Pattens Cove supported by attempting to raise accretion rate to meet or surpass sea level rise. This could be done through additions of thin sediment layers (TLP) to increase marsh platform elevation (Davis et al., 2022) and by stabilization of marsh edge with revegetation efforts (Brisson et al., 2014; Ganju et al., 2023;). Constraints on municipal budgets could be alleviated by requiring more community/developer agreements that emphasize nature-based solutions. The adoption of maintenance responsibilities at Pattens Cove by the BEAT sets precedent for developer responsibility to green and open spaces beyond their property lines and could be expanded upon. In addition to creating funding pools for climate resilience projects in nearby areas, these agreements could require maintenance of adaptations planned by the city to prevent them from falling into disrepair and failing. Perhaps most important and most impactful would be requiring wetland, stormwater, and climate permitting plans and specifications to be based on environmental conditions expected throughout the lifespan of the project rather than just at its inception. As noted at Pattens Cove, shifting baselines with rising tides and migrating marshes mean that critical delineations of protected zones will change. Sites currently able to claim exemption from mitigation, adaptation, and resilience measures will be within jurisdiction by the end of the century. If not addressed proactively, their environmental protection measures will be similarly outdated and ineffective. For example, rather than defensively diverting water away from these sites, more innovative solutions to infiltration and temporary storage should be considered. The impacts from these projects will be felt far beyond their date of completion and it would be prudent for their current design to be steered by future realities.

3.6 Conclusion

Marshes like Pattens Cove are interspersed throughout Boston, both in the landscape and in the fabric of communities. Ecosystem services they offer range from ecological biodiversity, shorelines stabilization, storm abatement, to less tangible improvements to well-being and quality of life. Anthropogenic pressures are actively undermining marsh success, including historic and continued exposure to contaminants, draining and filling, and perhaps most pressing, climate change. Sea level is, unequivocally, rising at an unprecedented rate. All marshes will be increasingly susceptible to loss as their survival depends on their accretion rate to outpace rising tides. For urban fringing marshes, the threat is severe, with their small size and high ratio of edges increasing exposure to degradation. These challenges are exacerbated by changing landscapes and increasing urban density that occurs despite regulations intended for preservation. Though climate resilience adaptations have been proposed, they fall short in addressing the long-term needs of coastal communities and lack express pathways for implementation. Critical to the adoption of true climate resilience strategies would require integration of forward thinking into existing permitting structures in order to extend protective jurisdiction into expected future extents of resource areas. Development projects seeking permits should be required to take a more innovative and holistic approach to assessing their impacts and implementing climate resilience strategies, rather than merely checking a box. Boston is defined by its shoreline; it is a city that has benefited from access to coastal waters since its inception. Urban marshes, no less intrinsic to the character of the city, require proactive management through both public and private avenues to ensure their continued wellbeing and that of our communities.

CHAPTER 4

FINAL WORDS

Pattens Cove is enmeshed into the fabric of the Savin Hill neighborhood as both an ecologically important salt marsh and as a publicly accessible urban green space. Despite being a protected natural resource, Pattens Cove has been heavily managed and impacted by human activities, like much of the shoreline within Boston Harbor. Evidence of land making and filling in of tidal flats dates back as early as colonial times and was only spurred on by increasing population and efforts to improve sanitation. The drivers of this shoreline change also acted as a source of pollution, contaminating water and sediments that continue to circulate in the environment today. The legacy of the past—unregulated discharge from both industry and sewage management—is compounded by continued high density urban usages of the watersheds connected to the Harbor. In many ways, the decisions of the past about infrastructure and expansion, have shaped how places like Pattens Cove look and function in the present, but also their viability into the future.

Though long gone are the days of unregulated waste management and urban growth, Pattens Cove continues to face emerging anthropogenic threats that determine its very form. Regulations put into place to protect these resources are generally focused on supporting the greater good of public health, safety, and welfare. Wetland and floodplain protections are linked intrinsically to stormwater management and the provision of other physical ecosystem services. Resource areas under jurisdiction of federal, state, and municipal environmental laws are granted buffer zones in which activity must be limited and permitted. This permitting process requires many stakeholders with various skillsets to join together and approve construction, remodeling, and development that might impact places like Pattens Cove. At the city level, even more forward thinking is included with requirements for climate change resilience and sea level rise. However, these additional considerations are triggered based on present jurisdictional boundaries of the environmental resources being preserved. At their core, wetlands like salt marshes are dynamic, with associated plant communities shifting to higher elevations as rising tides change the physical characteristics of the land.

The shoreline of 30, 50, and 100 years from now will likely look very different than it does today. Urban infrastructure and development currently within jurisdiction of wetland protection regulations that are being planned and permitted use assumptions of future water levels for design specifications. These developments are, in turn, changing usage of areas like Pattens Cove as well as affecting area stormwater infiltration. Many large-scale projects within a short distance from Pattens Cove are not within jurisdiction of wetland protection regulations based on delineations of today. Proposals like the new growth at 135 Morrissey and 45-75 Morrissey have lifespans that will extend into futures that project multiple feet of sea level rise. Rather than once again letting actions of the past cause future harm, thinking *much* farther into the future at what Pattens Cove might look like could provide pathways for better management and preservation. At the very least, requiring community and developer agreements that share the burden of preserving public green resources like Pattens Cove could alleviate any financial pressures preventing measures from being undertaken. These relationships between private and public interests are already precedented at Pattens Cove and could lay the groundwork and resources for innovative coastal resilience improvements.

Anthropogenic pressures on urban coastal wetlands like Pattens Cove have a past, present, and future role in defining their success. Management and protections of these resources has changed over time to match the need and as new knowledge is revealed. It is time again to reconsider the relationship and obligations between coast and community so that Boston marshes can thrive as far into the future as the city itself.

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