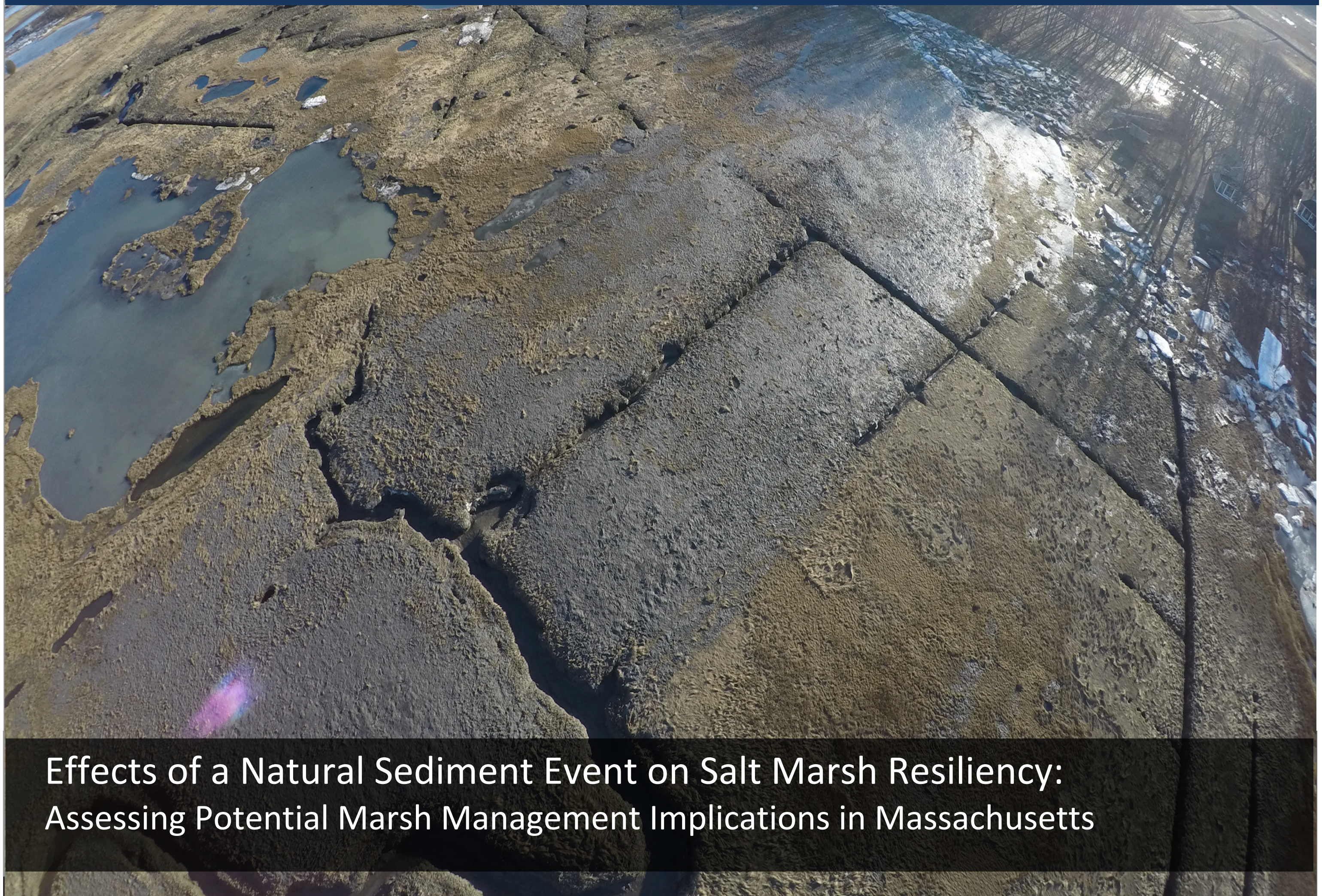

Effects of a Natural Sediment Event on Salt Marsh Resiliency:
Assessing Potential Marsh Management Implications in Massachusetts

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FINAL REPORT

Effects of a Natural Sediment Event on Salt Marsh Resiliency: Assessing Potential Marsh Management Implications in Massachusetts

On the cover: Overview of accretion event showing extent of ice-rafted sediment in a salt marsh in Essex Massachusetts, January 2018. *Photo by G.E. Moore.*

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EXECUTIVE SUMMARY

In mid-winter 2018, a combination of an extended period of extreme cold followed by northeast winds and a storm surge, resulted in a series of natural sediment deposition events of unprecedented size on the surface of salt marshes on the North Shore of Massachusetts (MA), and also noted in New Hampshire and southern Maine. These large-scale events were presumably caused by freezing seawater on exposed flats and rafting of sediment-laden ice over marsh areas during a period of astronomical tides coinciding with a significant storm. In MA, particularly expansive sediment events were noted in Great Marsh Estuary including Essex Bay (Lowes Island), Ipswich Bay (Jeffrey's Neck) and Newbury (Plum Island). The natural, incremental process of sediment addition is critical to building and maintaining the marsh platform, yet the magnitude of this event is extremely uncommon in this region. We used this natural event as opportunity to study effects of large-scale sediment additions over marsh areas on a scale that is ecologically significant and difficult, if not impossible, to recreate given environmental regulations in the state.

Unmanned aerial vehicle (UAV) imagery and field surveys were used to estimate the cumulative footprint of the events, calculating the areal extent at approximately 29.2 hectares. Field measured sediment thickness was measured directly after the event (winter 2018) and again in later summer of the same year at fixed plots. Overall, winter (30.1±2.1mm) and summer (31.3±2.0mm) sediment thickness did not differ significantly, suggesting sediment is not migrating and therefore, is effectively building marsh capital. Sediment characteristics (*i.e.*, grain size, texture, bulk density) differed somewhat between sites, but reflected those of control samples collected at each site's respective mudflat. These findings support the suggestion that deposited materials resulted from ice rafting from adjacent flats. Finally, comparison of percent cover revealed a significantly greater vegetated area ($p<.0001$) in sediment-free controls (93.1±1.6%) versus sediment addition plots (75.6±2.3%) after one growing season. When sediment addition plots were sorted by thickness ranges, *e.g.*, low: 1-19mm, medium: 20-39mm, high: 40-90mm, differences between experiment and control plots weakened, but remained significant ($p<.0001$) such that all thickness ranges had less plant cover than controls, and the high range had less plant cover than the low and medium ranges when blocked by habitat type..

These results suggest at least an initial reduction in plant cover after sediment addition in salt marshes of the North Shore of Massachusetts, but perhaps more notable is how minor this reduction appears to be given the significance of sediment coverage. Based on sediment area and thickness, this winter storm delivered roughly ~8,000m^3 of sediment to these marshes. Given the average accretion rate in New England marshes is ~2mm per year, the sediment event documented herein could represent over 15 years-worth of marsh building deposition at once with relatively minimal reduction in plant coverage in the first growing season. While monitoring over subsequent growing seasons is warranted, the initial results suggest that this natural sediment event will likely benefit marsh resilience to sea level rise over the short term by building the marsh platform. Opportunities to study natural sediment events build better understanding of the processes that contribute to marsh health and the potential management implications of marsh nourishment.

INTRODUCTION

Salt marshes are coastal wetlands that are recognized for their ability to protect properties from flooding, remove nutrients, capture carbon, and provide habitat for fish, mollusks, birds, and others (Costanza *et al*., 1997). Marshes undergo constant change as they respond to both hydrological and biological components; tidal flooding supplies marshes with sediment (Stumpf, 1983), while plants help to trap sediment (Morris et al. 2002) and build peat through the accumulation of organic matter (Bricker-Urso, 1989; Boyd and Sommerfield, 2016; Nyman *et al.* 2006). Salt marshes can be thought of as poised systems reflecting a balance of sediment accretion and peat development on one side versus erosion, compaction and oxidation of sediments on the other (Morris *et al.* 2002; Burdick and Roman, 2012). Human actions, from local to global in scale, have disrupted this balance, resulting in marsh loss and conversion of high marsh to low marsh due to increased flooding (Donnelly and Bertness, 2001; Warren and Neiring, 1993).

From a perspective of geologic time (*e.g.*,100,000 years) tidal salt marshes are ephemeral features as the ever-changing coastal landscape responds to 100-meter changes in sea level associated with glacial expansion) and retreat (Kelley, Gehrels, and Belknap, 1995; Fagherazzi, 2013). Over the past 4,000 years, sea level has been rising slowly, about 1 mm/year in New England (Keene 1971; Kelley, Gehrels, and Belknap, 1995). As average water levels rise, the potential for sediments suspended by flooding tides to deposit on the marsh increases, and the potential for sediments to dry out and oxidize decreases. Over an extended period of time (months to years) this sediment, combined with accumulation of organic matter leads to elevation gains on the marsh surface. The depth of peat in older New England marshes is up to 4 meters deep, which reflects both elevation growth and compaction over time (Kelley, Gehrels, and Belknap, 1995). The pace of sea level rise is critical to marsh peat development and is, in fact, a requirement for healthy marsh maintenance. However, if the rate of relative sea level rise surpasses the rate of marsh elevation gain, the frequency and duration of flooding will increase.

In New England, the majority of tidal marshes are classified as high marsh, comprised of wide flat plains dominated by salt hay (*Spartina patens*) that are dissected by tidal creeks lined with tall cordgrass (*Spartina alterniflora*) and interspersed with shallow pools. Tidal marshes in New England were formed during a period when sea level rise was fairly slow, and marshes played significant roles in support of coastal birds, fisheries, and storm protection. Although New Englanders have benefited from salt marsh ecosystem services since colonial times, they have only recently become aware of marsh values now that upwards of one third of the marshes have been lost to human actions (Bromberg and Bertness, 2005). Recognizing that further losses will have negative impacts to the economy, all the New England states have enacted laws and regulations that protect marshes from direct human impacts. Unfortunately, a variety of indirect impacts (*e.g*., loss of sediment availability, rapid sea level rise) still pose a major threat to marsh survival (Raposa *et al*., 2016; Watson *et al*., 2017).

Given the growing concern of marsh loss from human impacts, conversion of high marsh to low marsh with elevated sea levels (Donnelly and Bertness, 2001; Warren and Neiring, 1993), and impediments to sediment supply processes that help build marsh resilience (Weston, 2014), natural events that supply sediment to the marsh surface are particularly valuable. Sediment deposition from Hurricanes has been well-documented, particularly in southern marshes (Baustian and Mendelssohn, 2015; Mckee and Cherry, 2009). Deposition resulting from storm events can build the elevation of submerging marshes and increase plant growth (Baustian and Mendelssohn, 2015; Mckee and Cherry, 2009), but storm waves can also erode marsh edges (Priestas and

Fagherazzi, 2011). Because of the positive effects of sediment deposition on marsh plants, the process has been replicated artificially through thin-layer deposition (TLP), which typically involves jet-spraying a slurry of dredged material onto the marsh (Ford, Cahoon, and Lynch, 1999). While few TLP projects have been tried in New England, deposition from hurricanes and severe storms has been shown to build marsh elevation in the region (Roman *et al.,* 1997).

Another important mechanism for sediment deposition in northern marshes is ice rafting (Argow, Hughes, and FitzGerald, 2011; Dionne, 1989; Wood, Kelley, and Belknap, 1989). Deposition through ice rafting can occur when ice forms on exposed mudflats at low tide, and the rising tide transports the ice sheets to the adjacent marsh, along with mudflat sediment attached to the bottom of the ice (Argow, Hughes, and FitzGerald, 2011). When the ice melts, any sediment contained within or beneath is left on the marsh surface. This process can be repeated throughout the winter, resulting in multiple layers of sediment accumulating on the marsh surface (Argow, Hughes, and FitzGerald, 2011).

In New England, where marshes sit high in the intertidal zone, the phenomena known as "marsh drowning" is being increasingly observed (Watson et al. 2016, Raposa et al. 2016, Smith 2009, Smith 2015) in systems prone to subsidence, especially where sediment availability has been reduced (Day et al. 2007, Kirwan and Guntenspergen 2012). Too much flooding, especially when coupled with poor drainage, and marshes drown. As plants die, erosion rates increase and mud flats expand. Factors that can contribute to marsh loss due to drowning may include erosion, loss of sediment availability, subsidence and marsh collapse. Sediment supply is a key part of the marsh lifecycle and keeping the addition and removal of that supply in balance is important in helping it to maintain its resiliency. Marshes already have a limited amount of sediments supplied to them within the estuaries where they are found. River damming, diversions, navigation channels, and construction of jetties and marinas can alter sediment dynamics so they become unavailable to marshes for building as sea-level rises, ultimately leading to large-scale marsh loss (Gateway National Recreation Area 2001, Kirwan et al. 2011, Fagherazzi et al. 2013). Dredging in estuaries has removed sediments needed for marshes to build in elevation as well as created sediment sinks where fine grained sediments that would normally be captured in the marsh surface now settle in the dredge footprint (as in Jamaica Bay in New York).

Thin layer placement, or TLP, has been employed successfully in the US to offset net losses in marsh platform accretion. Present day use of TLP as a restoration tool occurs in some south-Atlantic and Gulf states where marsh loss and degradation has been rapid and extreme, particularly in the low-lying Mississippi Delta of Louisiana. It has also been used in middle-Atlantic states like New Jersey and New York to restore marsh that was devastated by Superstorm Sandy in the fall of 2012. A New England state whose marshes were hit particularly hard by the fierce wave and wind action of Sandy was Rhode Island. It is the only state in New England that has taken TLP beyond the experiment phase and applied it to marshes on a larger scale to restore functions that have become heavily degraded over time but to help those marshes build up resiliency to future sea-level rise. However, under current environmental regulations, TLP is not an accepted practice in Massachusetts. In 1963 Massachusetts became the first state to adopt wetland regulations meant to protect and preserve the state's 48,000 acres of wetlands (Commonwealth of Massachusetts 2018). Massachusetts General Laws (MGL) Chapter 131, Section 40, The Wetlands Protection Act is the regulatory authority in the state that oversees the 310 Code of Massachusetts Regulation (CMR) 10.00, Wetlands Protection Act Regulations (known hereafter as the Act). The responsibilities within 310 CMR 10, effective as of October, 2014) are carried out by the

Massachusetts Department of Environmental Protection (DEP). The purpose of the Wetlands Act Regulations are to regulate activities that may affect Areas Subject to Protection (which includes wetlands and marshes) in order to "*contribute to the following interests: protection of public and private water supply, protection of ground water supply, flood control, storm damage prevention, prevention of pollution, protection of land containing shellfish, protection of fisheries, protection of wildlife habitat*" (310 CMR § 10.01(2) 2014). "Protection" of the many valuable ecosystem services that wetlands and marshes provide, such as protection of the water supply, habitat and fisheries, is the primary purpose of the Act. If empirical evidence could support the benefit (and lack of long-term impacts), TLP could contribute to these protections. The Act in Massachusetts defines "fill" as "to deposit any material so as to raise an elevation, either temporarily or permanently" (310 CMR § 10.04 2014). Section 40 of G.M.L. 131 states that "*No person shall remove, fill , dredge or alter any bank, riverfront area, fresh water wetland, coastal wetland, beach, dune, flat, marsh, meadow or swamp bordering on the ocean or on any estuary, creek, river, stream, pond, or lake, or any land under said waters or any land subject to tidal action, coastal storm flowage, or flooding, other than in the course of maintaining, repairing or replacing, but not substantially changing or enlarging, an existing and lawfully located structure or facility used in the service of the public…*" (131 GML § 40). The Act's language regarding fill is of critical importance to overcoming the presumptions of the statute. An applicant for TLP must file a Notice of Intent as well as meeting the requirements of the 314 C.M.R. 9.00: 401 Water Quality Certification standards, neither of which guarantee issuance of a permit – particularly when the presumption that placement of material on the marsh will cause no quantifiable impacts may not be met. Under the Water Quality standards, any alterations to a salt marsh are also prohibited. The Massachusetts Surface Water Quality standards, 314 C.M.R. 4.06(2), defines wetlands as "Outstanding Resource Waters" and discharge of fill into these designated waters is also prohibited (314 CMR 4.00).

There is also significant burden of proof required for permitting when it comes to TLP. Discussions with regulators in the state have pointed out that TLP is still a relatively new and untested approach in Massachusetts. There is a sentiment that not enough scientific research has been conducted in order to support TLP as a tool, whether it is used by itself or in conjunction with other approaches, for improving marsh resiliency. While more science is necessary for Massachusetts regulators to evaluate TLP as an effective approach, the wetland regulations for Massachusetts make no allowances for conducting even small-scale scientific studies. There is a high burden of proof required by the regulations but no way to develop evidence required to build that platform of proof. Climate change and sea-level rise are currently not within the purview of the Act in Massachusetts. The Executive Office of Environmental Affairs (EEA) addresses climate change and rising sea levels via the Massachusetts Environmental Policy Act (MEPA) which reviews the environmental impact of potential projects within the state. MEPA does not make decisions on permits, rather, via its processes, provides opportunities for public review and comment on projects that may have an environmental impact. MEPA also requires that state agencies use measures deemed feasible to "avoid, minimize, and mitigate damage to the environment" when analyzing and/or permitting or funding projects (Mass.gov 2018).

Restoration activities are permissible in the state and under the Act via the Ecological Restoration Project provision however those activities shall only be permitted if it can be proven that there will be no negative impact on the marsh as a result of the activity. That brings us back to the required burden of proof. And while there are accommodations for ecological restoration projects that remove fill, the placement of fill is still not allowed. Feedback received from regulators and ecologists in Massachusetts pointed out that marshes in the state are highly varied in that there are numerous tidal environments that exist along its large coastline. An intervention that works well in one marsh, may not be appropriate for another. When it comes to TLP, we were reminded, a one-size-fits-all solution is not appropriate. According to the experts we spoke to, this is also the case when it comes to determining the level of degradation being experienced by individual marshes. One may argue that there are still too many unanswered questions about whether some of the degradation being witnessed today is irreversible and as a result of sea-level rise or if it is a natural event that will eventually correct itself with time. The unknowns may outweigh the possibilities of testing TLP based on a lack of empirical data in a strictly regulated state like Massachusetts.

However, in mid-winter 2018, a series of natural sediment deposition areas of unprecedented size were observed on the surface of salt marshes on the North Shore of Massachusetts. These deposition events were also noted on a smaller scale in New Hampshire and southern Maine. This deposition followed a period of abnormally low temperatures combined with high tides and strong northeast winds, leading us to hypothesize ice rafting as the mechanism of sediment deposition. While overwash events from severe storms have been known to deposit sediment on marsh surfaces at rates exceeding typical tidal deposition processes (Roman *et al*., 1997), and ice rafting can float slabs of calving marsh peat onto the high marsh platform from time to time, the authors know of no such documented large-scale natural sediment deposition event caused solely by ice rafting in Massachusetts. Moreover, it is unclear how large-scale deposition affects marsh plant communities in New England. Ultimately, this large-scale natural event provided a perfect opportunity for studying the effects of variable thickness sediment deposition on marshes on the north shore of Massachusetts.

Accordingly, the goals of this study are to 1) document the scale and distribution of this natural sediment event in salt marshes on the north shore of Massachusetts; 2) map the area, thickness and volume of the sediment deposited, and use these data to track changes or redistribution of sediment on the marsh surface over time; and 3) examine the effect of sediment deposition on marsh plant community structure and resilience over the range of sediment thicknesses across the three marsh sites. To meet these goals, we used a combination of aerial drone surveys and field surveys, as well as plant assessments on the marsh surface and pore water chemistry analysis in the marsh peat, as detailed below.

METHODS

Study Site

The study sites occur in three individual salt marsh systems on the north shore of Massachusetts that experienced a large-scale sediment deposition event in early January – February of 2018. The marshes are located at Lowes Island, Essex (EB), Jeffrey's Neck, Ipswich (JN) and Jeffrey's Neck, Newbury (PI) in Essex County, Massachusetts (Figure 1). Each site occurs within the same watershed and are collectively part of the Great Marsh Estuary, the largest continuous area of saltmarsh on the north shore of the Commonwealth.

Aerial Imagery and Spatial Analysis

Visible light (RGB true color) geo-tagged imagery of the sediment event areas was acquired at an altitude of 60m at an effective 1.67cm ground resolution using *MAPIR Survey3* camera flying on a *3DR Solo* UAV platform. Flight planning and control for UAV flights completed for this survey used *3DR Tower* software. Programed survey flight lines insured 70-80% overlap with neighboring images to increase resolution and facilitate subsequent photomosaic production using *Pix4DMapper* software (version).

Once the imagery was mosaicked for RGB, a supervised classification was completed to identify and quantify sediment coverage using *ArcGIS* software interactive classification methods. This classification used forty-two training sites within each site's photomosaic image to capture unique signatures in the sediment deposition areas. Additional refinements to the classification were completed with ARIS Grid & Raster Editor, a third-party extension for ArcGIS, ArcMap software (version).

Study Plot Establishment and Sediment Measures

A series of randomly located, linear transects were established at each site. Plots were situated along each transect approximately 20m apart to avoid sampling bias as the sediment thickness varied throughout each event, with scattered patches within the sediment footprint receiving no sediment at all. Google Earth historical imagery was used to identify plots that fell upon areas that were unvegetated or partially vegetated before the sediment event (e.g., pools and ditches), and these plots were omitted from analysis. At each plot, sediment thickness was measured by placing a rigid meter stick into the overlaying sediment down through to the point of refusal (*i.e.*, either the original marsh surface or a layer of ice) during the winter sampling. If a plot corresponded with a non-sediment deposition patch, thickness was recorded as a zero. Thickness was recorded to the nearest millimeter and the samples location was marked using a Leica GSSN Rover model GS14 Real Time Kinematic (RTK) GPS (±2cm accuracy). Summer sampling in August employed the same method, navigating to the original sampling locations using the RTK waypoints. Sediment cores were obtained during the winter sampling event. Coring was completed at 20m intervals along each transect using a 3.5cm diameter piston coring device. Cores were placed in pre-labelled plastic bags (ZipLock™) and stored at 4°C until analysis. Additional plots were established outside the zone of sediment influence to serve as control plots (no sediment deposition).

Pore Water Analyses

Soil pore water was sampled using the sipper method (Portnoy and Valiela, 1997) which extracts water trapped in pore spaces using a 1mm diameter stainless steel tube fitted with a 60cc plastic syringe. Pore water was sampled at a uniform depth of ~25cm at 20m fixed intervals along the transects (corresponding with soil core collection locations). At each sampling station, we measured pore water salinity, redox potential, pH and sulfides. Salinity was determined in the field using a Thermo Scientific Orion Star A329 Portable pH/ISE/Conductivity/RDO/DO/Temperature Multiparameter Meter with DuraProbe conductivity cell, while redox potential and pH were obtained using a platinum electrode and a Ross Sure-Flow temperature corrected pH triode, respectively. Because pH varied greatly between plots (3.0 – 7.3), redox potential was adjusted to pH 5 using the equation $Eh5=redox+[(pH - 5)*59]+244$ (Megonigal, Patrick, and Faulkner 1993).

Vegetation Metrics

Vegetation sampling was completed in late July through early August to represent peak plant growth. Sampling plots corresponding with fixed plots sampled for sediment and pore water metrics, situated at 20m intervals along transects. Species richness and percent cover were recorded using visual estimation within 0.5m^2 plots at each station. Vascular plants were identified to the species level in the field. Bare/unvegetated ground was accounted for in the total percent cover estimate and the predominant habitat type was noted (*i.e.*, high marsh, low marsh).

Data Analysis

ANOVA was used to analyze multivariate effects of sediment deposition on pore water chemistry and plant cover and Tukey's HSD post-hoc test was used to determine differences between treatments. The Shapiro-Wilk goodness of fit test was used to determine whether residuals met the assumption of normality. The following transformations were made to satisfy assumptions of parametric tests: log(salinity), log(-pH + 8), and log(-percent cover + 101). When transformations were ineffective, Wilcoxon rank sum was used for univariate tests (salinity, percent cover) and Kruskal Wallis was used instead of one way ANOVA (effect of site on Eh). All statistical analyses were performed in JMP Pro 14.

Results

Table 1: Comparison of area, sediment thickness and volume by site.

		Thickness (mm)		
Site	**Area (ha)**	**Winter**	**Summer**	**Volume (m3)**
Essex Bay (EB)	1.7	19.1+/-2.2	20.9+/-2.4	35530
Jeffreys Neck (JN)	3.2	38.5+/-3.3	39.6+/-2.8	126568
Plum Island (PI)	24.3	25.6+/-2.9	26.5+/-3.8	642845
Total	**29.2**	**30.1+/-2.1**	**31.3+/-2**	**804944**

Sediment Estimates

A total of 29.2 ha of naturally deposited sediment was calculated from the orthomosaics and supervised classifications (Figures 2-3(a-c)). This total is comprised of 1.7 ha at EB, 3.2 ha at JN, and 24.3 ha at PI (Table 1). The estimates do not account for the entire event, but they do capture the most densely covered areas and thus the majority of deposition at each site.

Winter measures of sediment thickness (30.1±2.1 mm) did not differ significantly from summer measures (31.3±2 mm) (Figure 4a). However, thickness appeared to differ between sites in both winter and summer (Figure 4b), with JN being virtually double the thickness of EB in both cases. When data were sorted by thickness range classes, there was a significant difference in sediment thickness change ($p<.05$). For the low thickness plots, the sediment compacted or eroded whereas for higher thickness levels the sediment expanded (Figure 4c) Based on an average summer thickness of 31.3±2.0 mm, the affected areas received a total volume of 804,944 m^3 of sediment. The summer thickness measure was chosen in this calculation because this is the period of time most significant to plant growth.

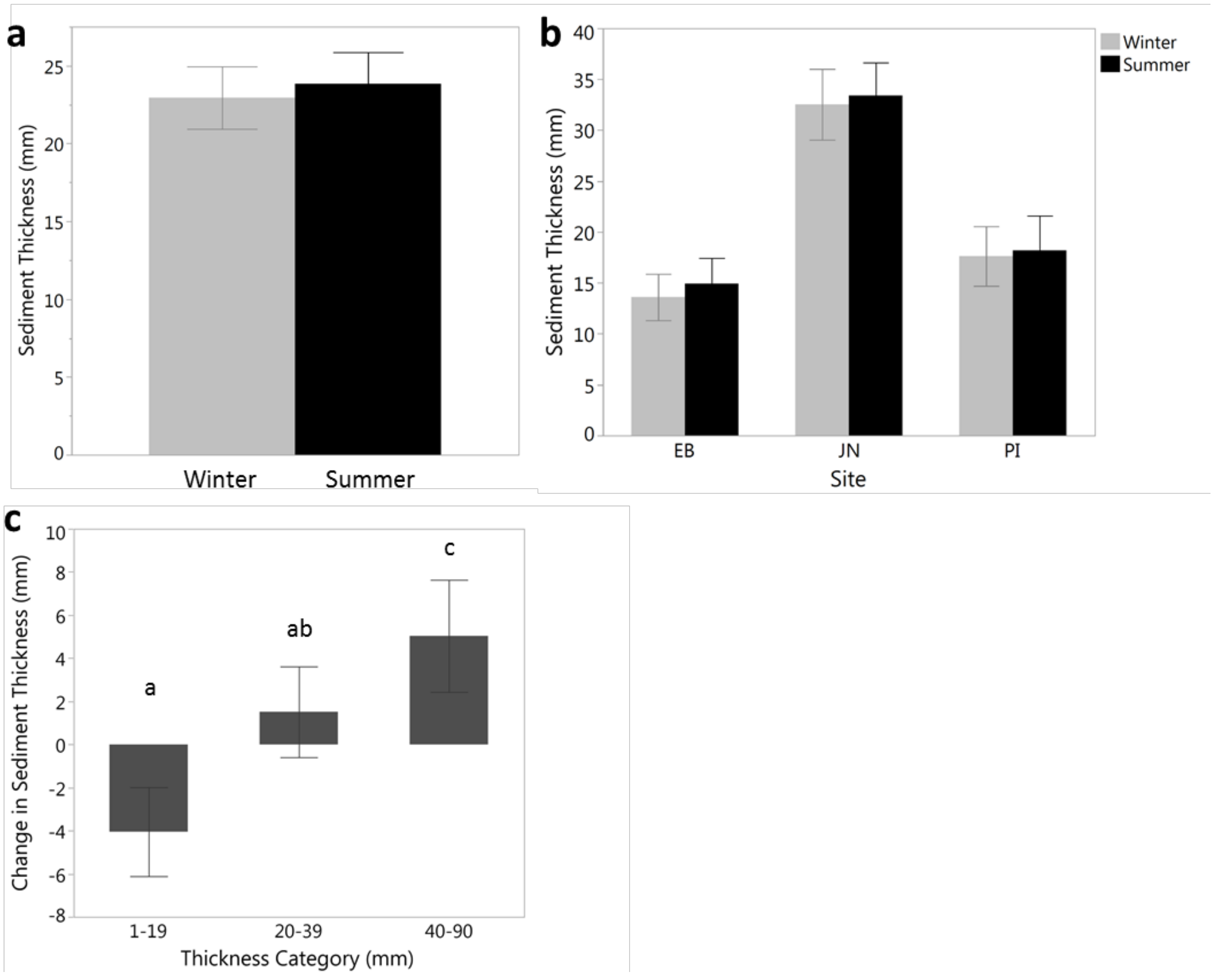

Figure 4. a) Thickness of deposited sediment in winter and the following summer. b.) Sediment thickness at each site for winter and summer. c.) The change in thickness of deposited sediment (summer – winter) for each thickness category. Different letters denote significant differences between thickness categories. Error bars show standard error.

<u>Pore Water</u>

Salinity – Sediment deposition, regardless of thickness was not found to elevate salinity (Figure 5a-b). Even if only high marsh data are examined, control salinity does not differ from sedimented areas. However, high marsh areas had significantly higher salinity than low marsh areas ($p<.05$).

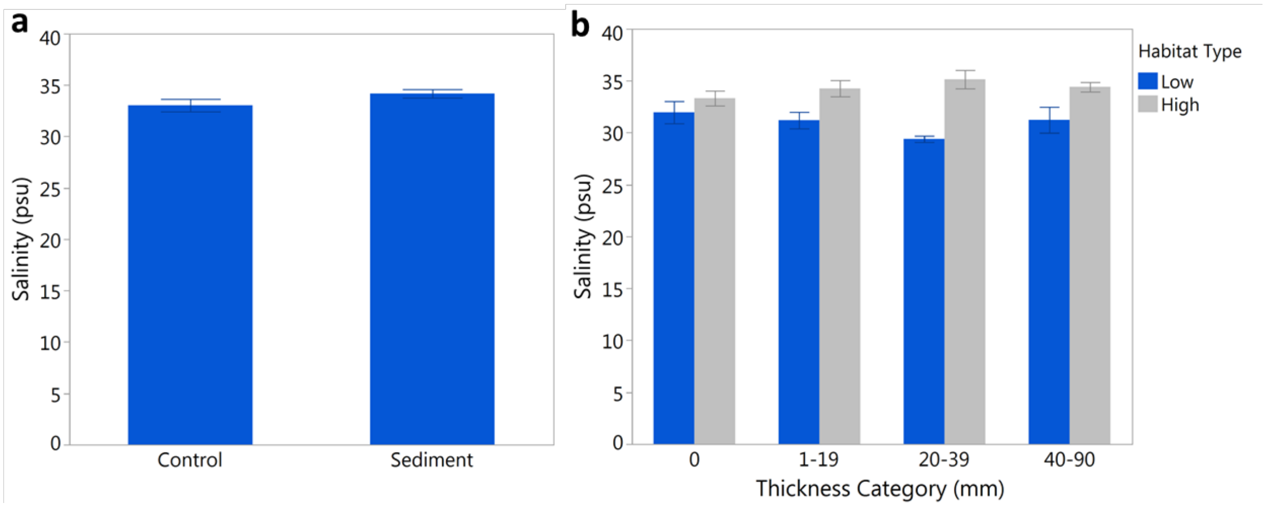

Figure 5. a) Salinity comparison between control areas and areas that received sediment. b) Salinity for each thickness category with low marsh and high marsh separated. Error bars show standard error.

pH – Similar to salinity, pH did not differ between treatment and control, but did vary considerably by site (Figure 6) with JN showing significantly higher pH than EB and JN (p<.0001). pH was especially low at PI, with an average of 5.8±.2.

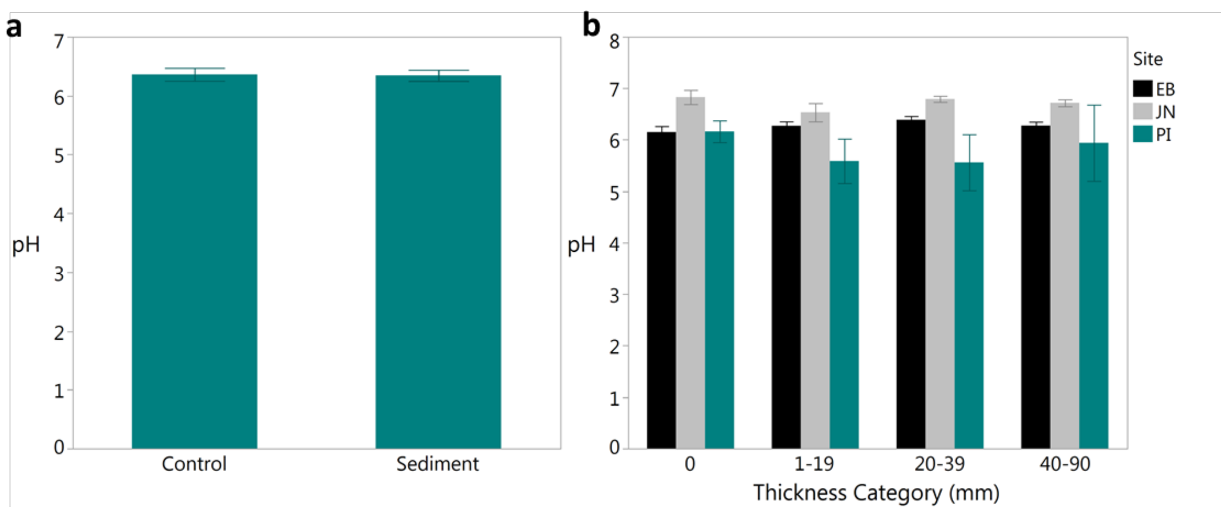

Figure 6. a) Comparison of pH between control areas and areas that received sediment. b) pH at different thickness categories separated by site. Error bars show standard error.

Redox Potential – Mirroring pH, redox potential did not differ between treatment and control, but did vary considerably by site (p<.0001). The Refuge site (PI) had higher but variable redox (138.5±16.7 mV) than EB and JN which were consistently anaerobic (12.3±3.8 mv and 40.5±5.6 mV, respectively). However, redox potential did not differ based on sediment thickness, with averages for each thickness category ranging from 49 to 68 mV.

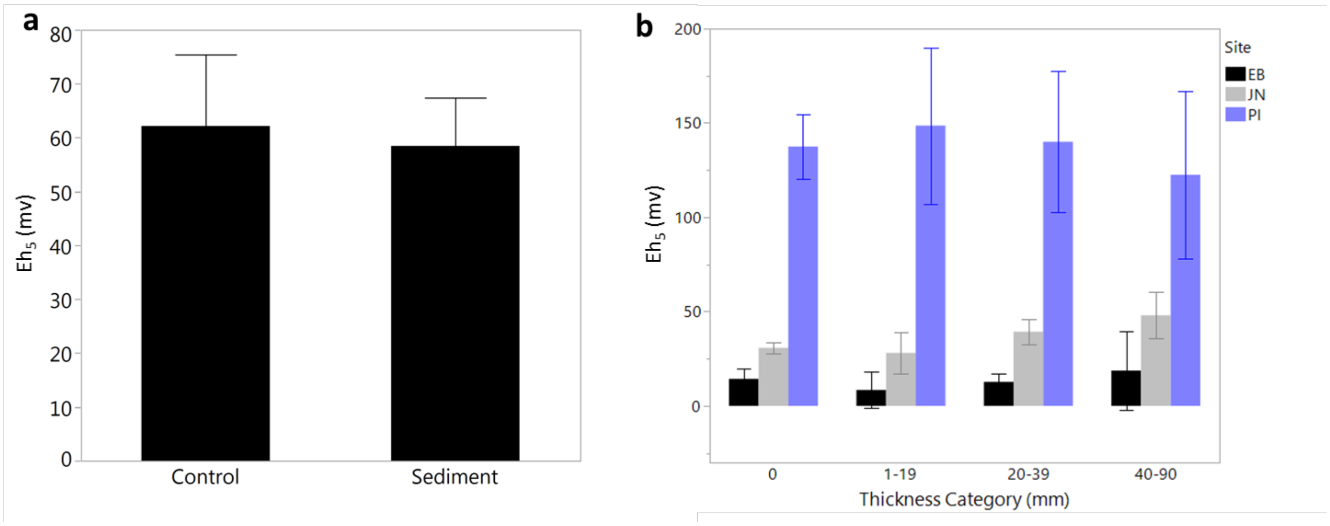

Figure 7. a) Soil redox potential (Eh) for controls vs. areas that received sediment. b) Redox separated by thickness category and site. Eh was adjusted for a pH of 5. Error bars show standard error.

Vegetation Response

The native marsh community appeared to be influenced by sediment additions. While species composition showed no effect between control and sediment addition, percent cover was significantly reduced after one growing season ($p<.0001$). Control plots had an average percent cover of 93.1±1.6 while the sediment plots had 75.6±2.3 (Figure 7a). There was a significant effect of sediment thickness ($p<.0001$), with the thickest sediment impairing vegetation more than the thinnest sediment. All thickness levels had lower cover than controls. High marsh areas had lower cover than low marsh areas ($p<.01$), and this difference appeared to be higher for thicker sediment levels, though the habitat type x sediment thickness interaction was not significant.

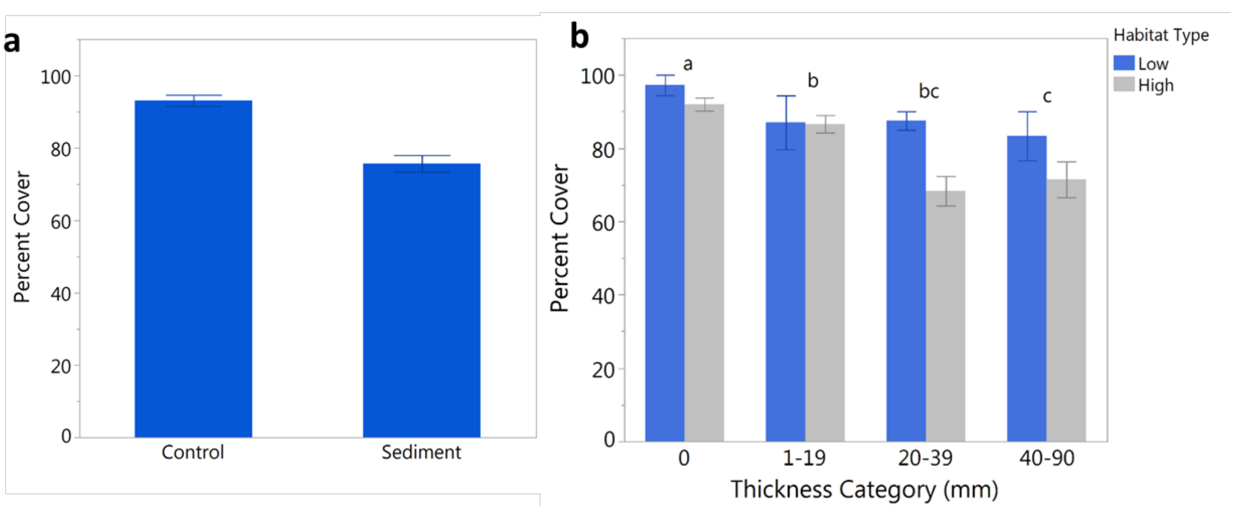

Figure 7. a) Cover (%) of marsh plants in control areas and areas that received sediment. b) Cover (%) of marsh vegetation for categories of sediment thickness with low and high marsh separated. Different letters denote significant differences between thickness categories. Error bars show standard error.

DISCUSSION

Storm overwash events can significantly contribute to marsh accretion (Mckee and Cherry, 2009; Stumpf, 1983), but deposition of this magnitude has not been previously documented in New England. Episodic sediment events have been recorded in New England, such as the 1938 Hurricane, evidence of which is preserved within the peat profile of many area marshes (Boldt *et al.,* 2010). These events are significant to the ability of a salt marsh platform to build with fluctuations in sea level over geologic time scales, but do not seem to be a regular process documented with frequency in our area. The suggestion that natural sediment events can benefit marshes underscores the premise for anthropogenic sediment placement strategies (TLP) to build marsh resilience to SLR. However, the potential ecological benefits of sediment placement must be balanced against the potential for unintended consequences. The goal of most studies of this approach therefore must focus on determining the presence and extent of ecological impact – or alternatively to demonstrate evidence that such impacts are minor or insignificant. The present study attempts to consider a subset of parameters that are known to represent or influence salt marsh health, including vegetation and pore water chemistry.

Other studies have shown that ice rafting plays an important role in marsh accretion in northern New England (Argow, Hughes, and FitzGerald, 2011; Wood, Kelley, and Belknap, 1989) and Canada (Dionne, 1989), but marsh-wide deposition of this size due to ice rafting has not been recorded. Dionne (1989) documented a total volume of ice-rafted peat clumps of up to 99 m3 in a 2500 m2 area, while Wood, Kelley, and Belknap (1989) showed that ice-rafted material contributes from 0-100% of the accretion in Maine marshes. These studies measured discrete patches of deposition in study areas whereas our study measured marsh-wide deposition at three marshes.

The deposited sediment documented in this study did not change in area or thickness during the 6-7 month study period. This stability suggests that little to no erosion occurred on the marsh surface, and the sediment did not compact over time. A review of elevation change following storm events in the southern U.S. showed that accretion usually outweighs erosion on the marsh platform (Cahoon, 2006), but subsidence may occur following sediment additions as the new sediment consolidates and compresses the underlying peat (Cahoon *et al*., 2018; Corno and Sadro, 2002; Mckee and Cherry, 2009). Although we did not measure overall elevation change, there was no compaction of the deposited sediment. The sediment was composed of coarser grain material than what is typically found in salt marsh soils, which may reduce the compaction potential (Mckee and Cherry, 2009). Additionally, since the mechanism of sediment addition in our study was ice rafting instead of storms or TLP, the weight of the overlying ice could have already compressed the deposited sediment, preventing further compaction from occurring.

Porewater measurements showed that soil conditions were similar between sediment areas and controls, despite a significant difference in elevation caused by the added sediment. Sediment addition often results in higher redox potential and lower sulfide levels due to a lower frequency of flooding (Mendelssohn and Kuhn, 2003; Stagg and Mendelssohn, 2010). Although we found a nearly-significant difference in salinity between sediment areas and controls, salinity was only higher in sediment areas by about 2 psu. This lack of difference in pore water characteristics suggests that sediment areas and controls were equally hospitable for marsh plants.

Our results suggest large-scale sediment deposition may have negative impacts on plant communities in the short term, but the impacts are relatively small after one growing season. The lower cover found in deposition areas was likely caused by the energetic cost to plants from having to penetrate the added layer of sediment, and in some cases the sediment may have been too dense for shoots to break through. Since the deposition event occurred during the winter, burial of winter shoots could have also prevented diffusion of oxygen into the roots, resulting in further plant stress or mortality (Wijte and Gallagher, 1991).

While the long-term effect of sediment deposition on New England marshes is still uncertain, benefits to plants are well-documented in southern marshes. Studies on *S. alterniflora* marshes in the southern U.S. have shown that TLP results in greater aboveground growth compared to submerging areas that did not receive sediment (Croft *et al*., 2006; DeLaune *et al*., 1990; Pezeshki *et al*., 1992; Ford, Cahoon, and Lynch, 1999; Mendelssohn and Kuhn, 2003; Slocum, Mendelssohn, and Kuhn, 2005; Tong *et al*., 2013). Marshes that were not clearly submerging have also been shown to benefit from sediment addition through improved aboveground growth (Baustian and Mendelssohn, 2015; Tong *et al*., 2013; Walters and Kirwan, 2016). Benefits to plant growth from TLP are attributed to higher redox potential and lower sulfide levels, which both indicate a reduction in flooding stress (Mendelssohn and Kuhn, 2003; v). Sediment addition can also temporarily boost growth through the sudden infusion of nutrients contained in the sediment (Mendelsohn and Kuhn, 2003; Slocum, Mendelssohn, and Kuhn, 2005). Although sediment addition can reduce aboveground growth in the short-term, plants have been shown to recover within one year (Mckee and Cherry, 2009; Ford, Cahoon, and Lynch, 1999).

The effect of sediment addition on plants may vary depending on the sediment thickness (DeLaune *et al*., 1990; Slocum, Mendelssohn, and Kuhn, 2005; Stagg and Mendelssohn, 2010). While percent cover in all of our treatments was lower than in controls, the lowest thickness (1-20 mm) had higher percent cover than the other thicknesses. In a 6-month long mesocosm experiment in Virginia, Walters and Kirwan (2016) found that burial of 5 cm had the greatest benefit to *S. alterniflora* growth, with greater depths resulting in a diminishing effect or even plant mortality at depths greater than 30 cm. Plants may have benefited more from sediment in their study because it was conducted only in the low marsh, where flooding stress is higher. Similarly, Reimold *et al*. (1978) found that burial at depths greater than 21 cm killed *S. alterniflora*, leaving the slower process of colonization through seeds as the only means for recovery.

The sediment deposition documented in this study could improve marsh resilience to sea level rise in the long-term. SET measurements indicate that nearby marshes in New Hampshire are losing elevation relative to sea level rise at a rate of ~2 mm/year, and the average rate of accretion in these marshes is ~3 mm/year (Payne, Burdick, and Moore *In Press*). Based on these rates, the average sediment thickness we found (~30 mm) amounts to about ten years of normal annual accretion. If subsidence is minimal and plant communities continue to recover, the added sediment could reduce flooding stress as sea levels rise by raising the marsh elevation. However, continued erosion of the mudflat could also increase erosion of the marsh edge (Mariotti and Fagherazzi, 2013). This high-deposition anomaly also underscores the importance of collecting long-term accretion data since short-term datasets may fail to capture such an important deposition event. Other studies have shown that ice rafting plays an important role in marsh accretion in northern New England (Argow, Hughes, and FitzGerald, 2011; Wood, Kelley, and Belknap, 1989), but it is unclear how climate change will affect this mechanism of sediment delivery. Larger temperature swings could increase the number of freeze-thaw cycles (cite?), creating more opportunity for ice rafts to form and melt on the marsh. However, higher temperatures could also decrease the number of ice rafts that form.

The results of this study could provide insight as to how marsh plants in Massachusetts could respond to a modest range of TLP. Our data suggest that plants would recover to 75% of the cover found in untreated areas within one growing season if sediment was applied to the marsh at an average thickness of 30 mm. However, longer time periods are needed for full recovery, and it is unknown whether TLP can prevent conversion of high marsh to low marsh or overall marsh loss.

CONCLUSIONS

Ice rafting likely plays a large and perhaps underappreciated role in building marsh elevation in New England. Although rare in Massachusetts, large-scale sediment deposition through ice rafting results in a sudden influx of elevation capital and could account for up to ten years of typical accretion over just one winter, as shown in this study. Our results show plant cover was slightly reduced by burial in the short-term, but plants should fully recover within 1-2 growing seasons, based on previous studies. Similar impacts could be expected from TLP, but the different method of sediment delivery (jet spray of dredged material) could influence plant responses. Further monitoring is needed to determine how large-scale sediment deposition, both natural and artificial, will affect long-term marsh resilience to sea level rise in New England.

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FIGURES

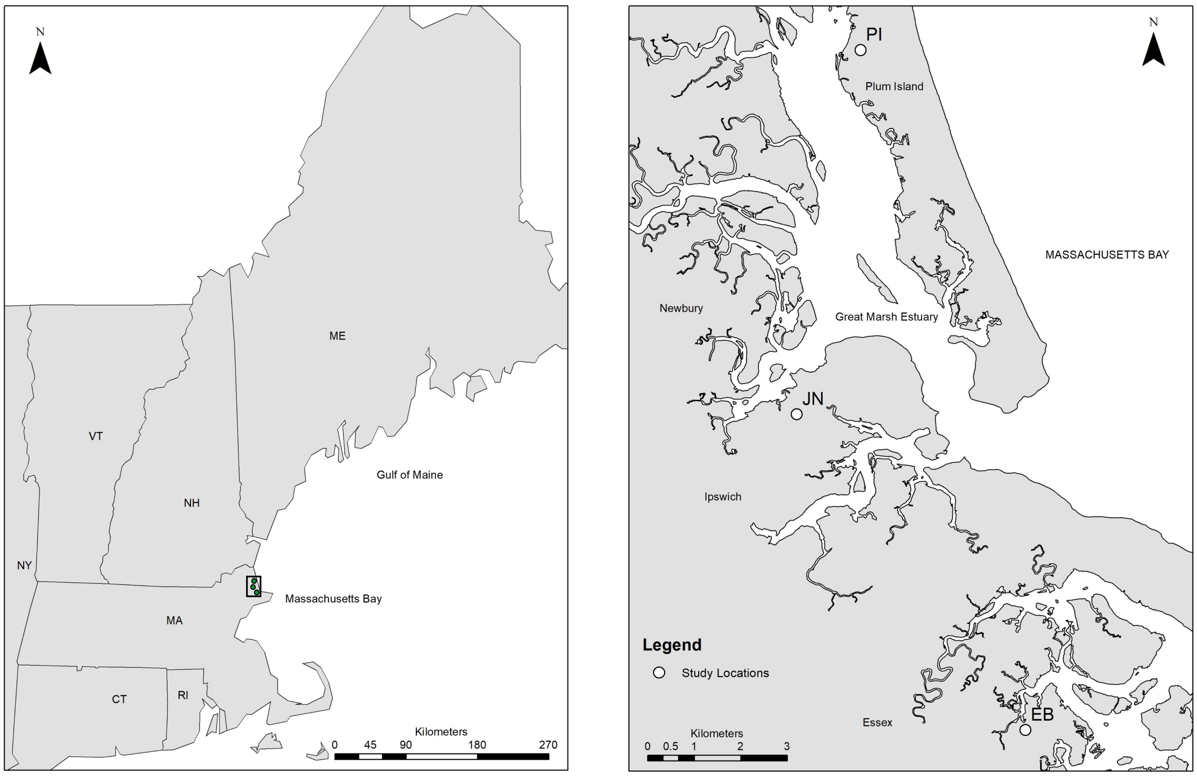

Figure 1: Study Area

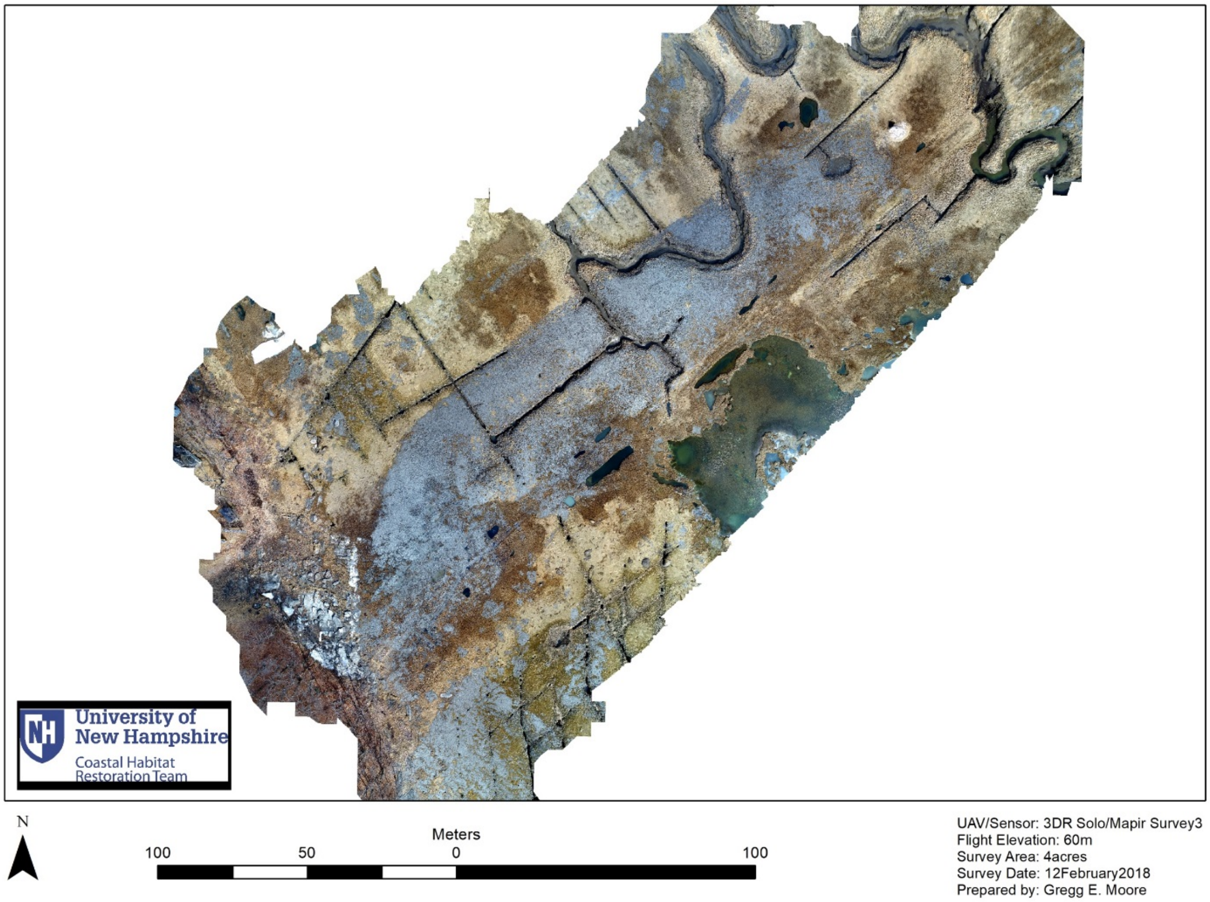

Figure 2(a): Sediment area orthophoto maps

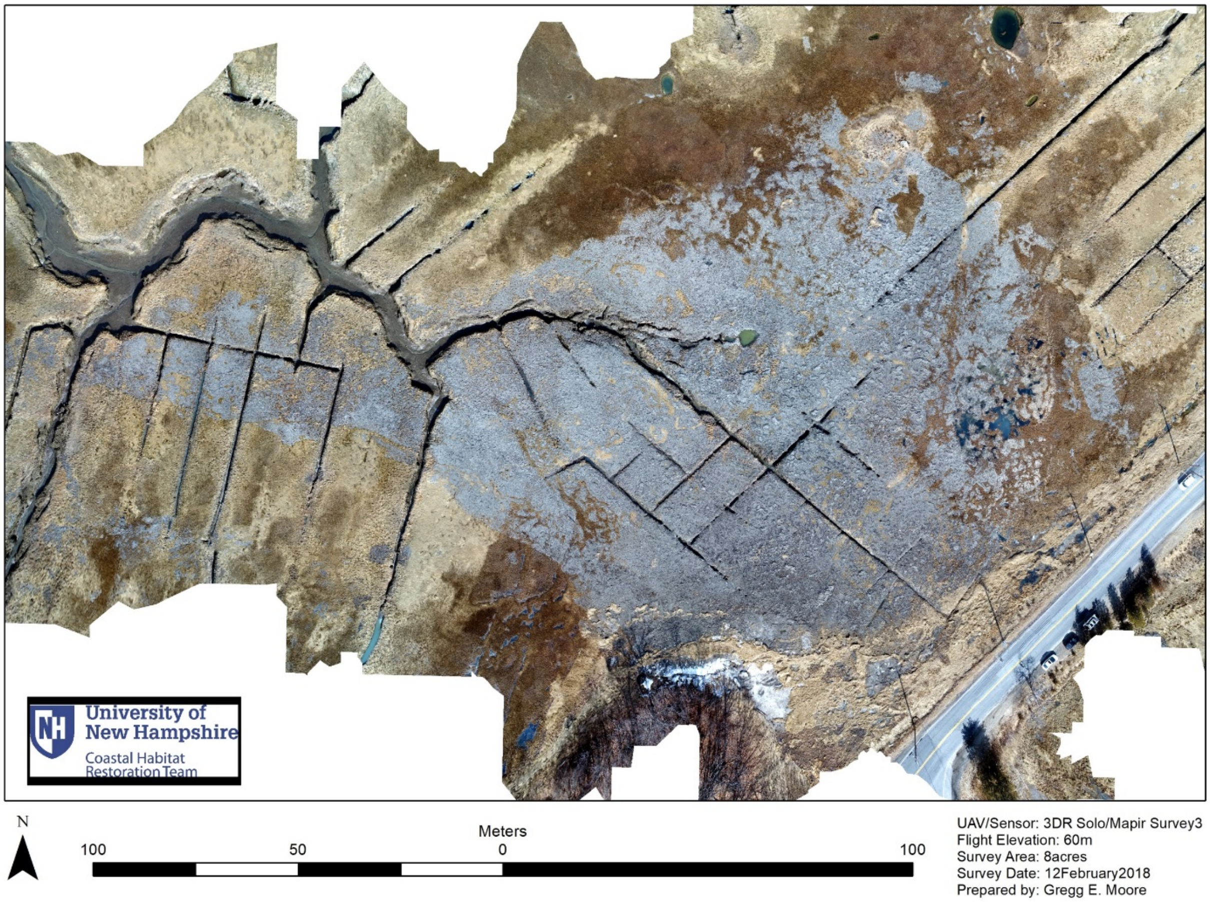

Figure 2(b): Sediment area orthophoto maps

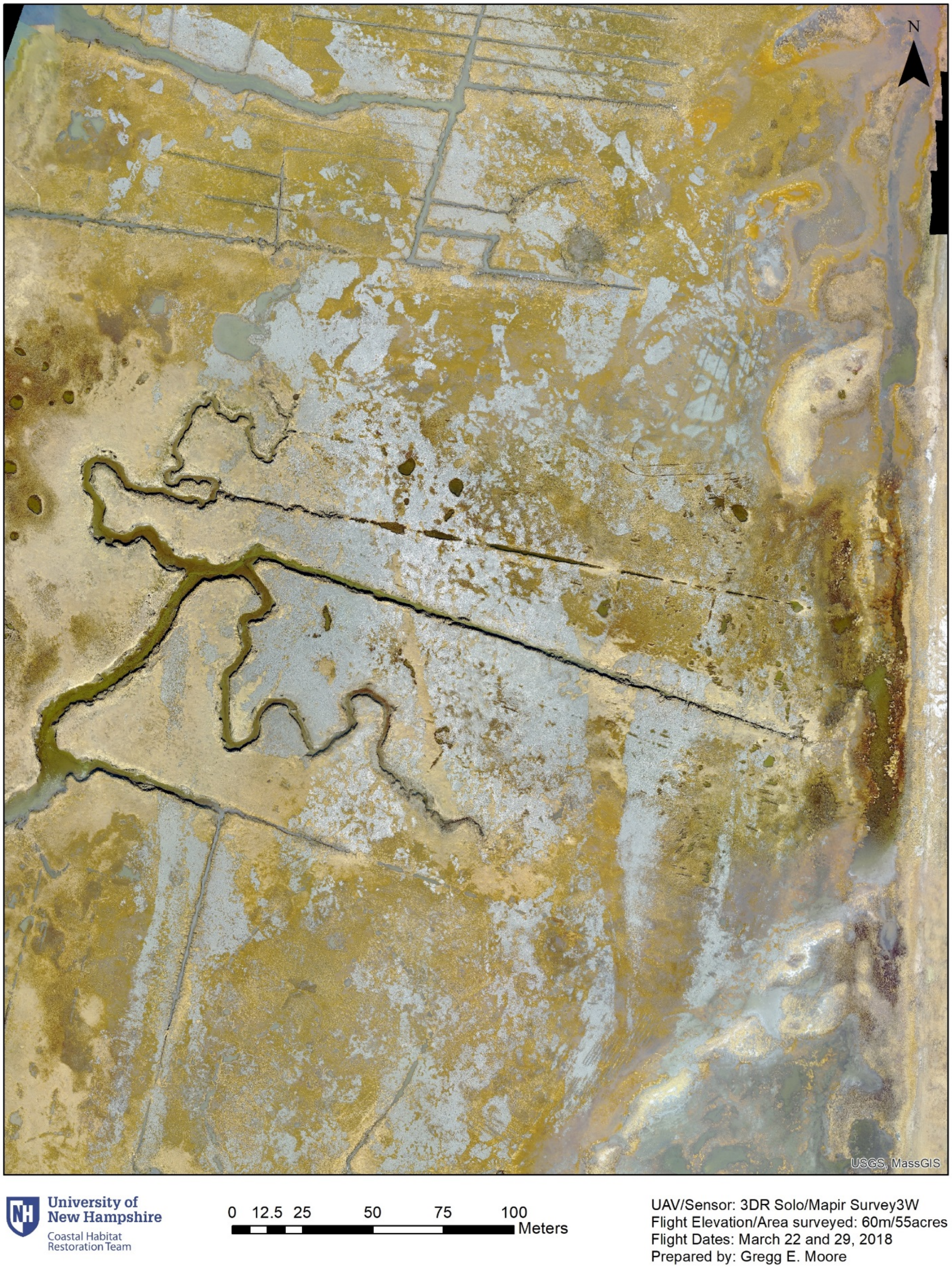

Figure 2(c): Sediment area orthophoto maps

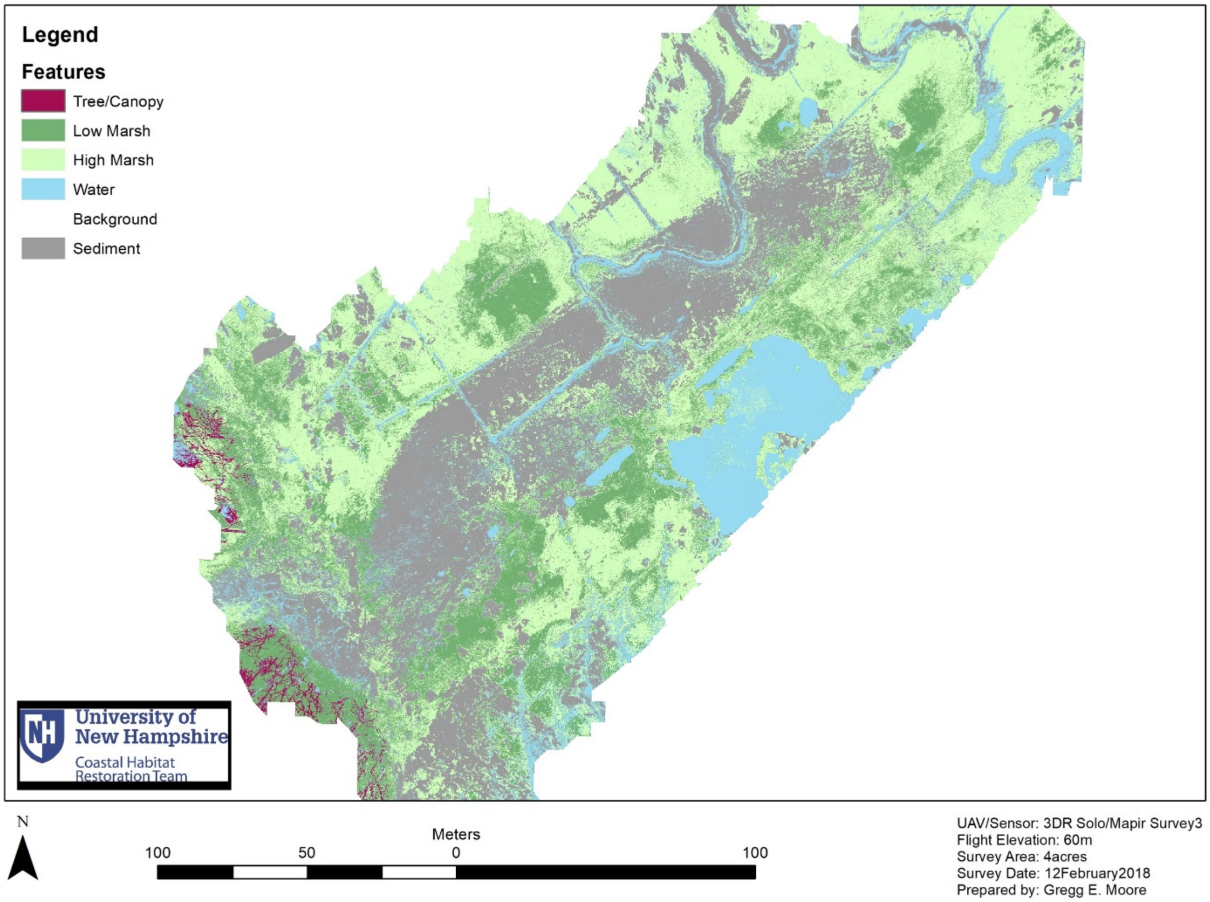

Figure 3(a): Sediment area supervised classification maps

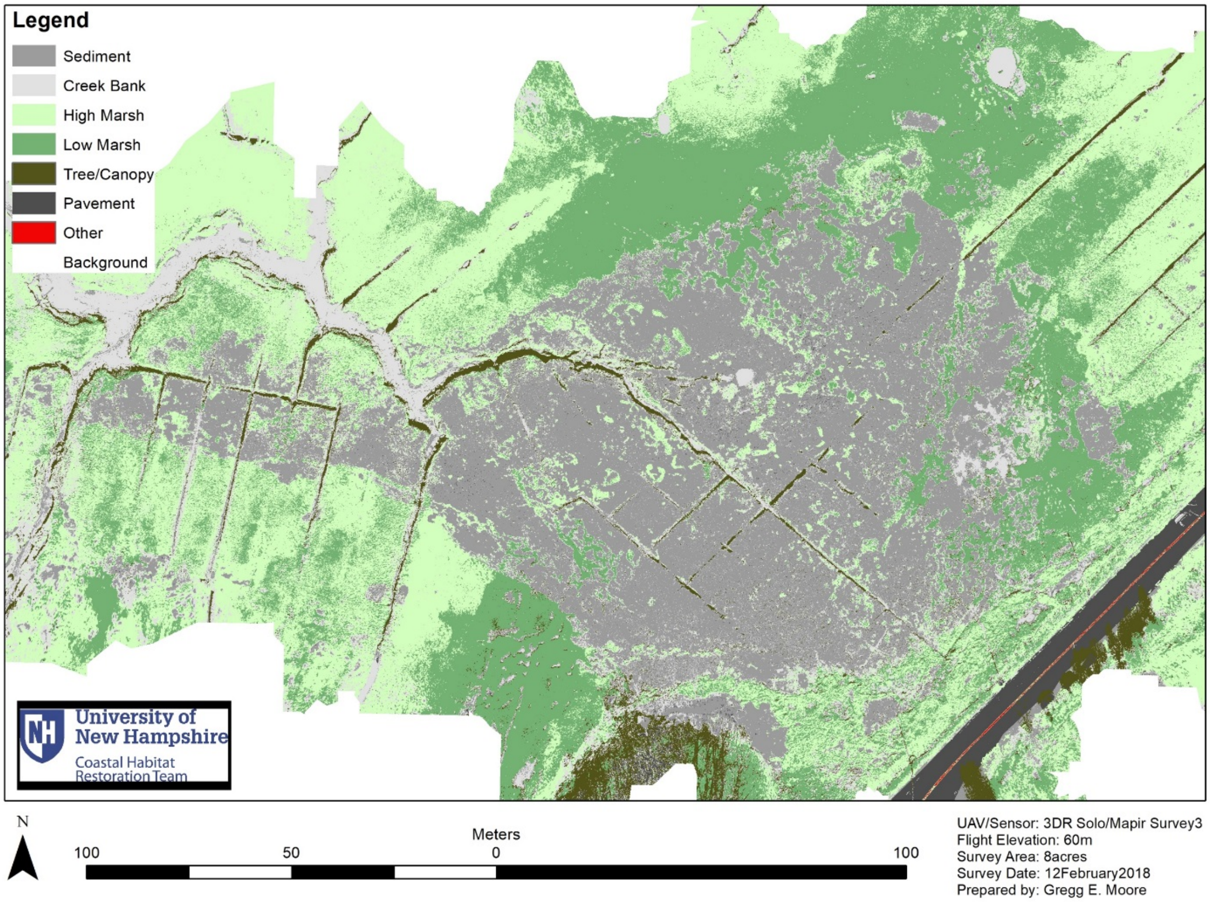

Figure 3(b): Sediment area supervised classification maps

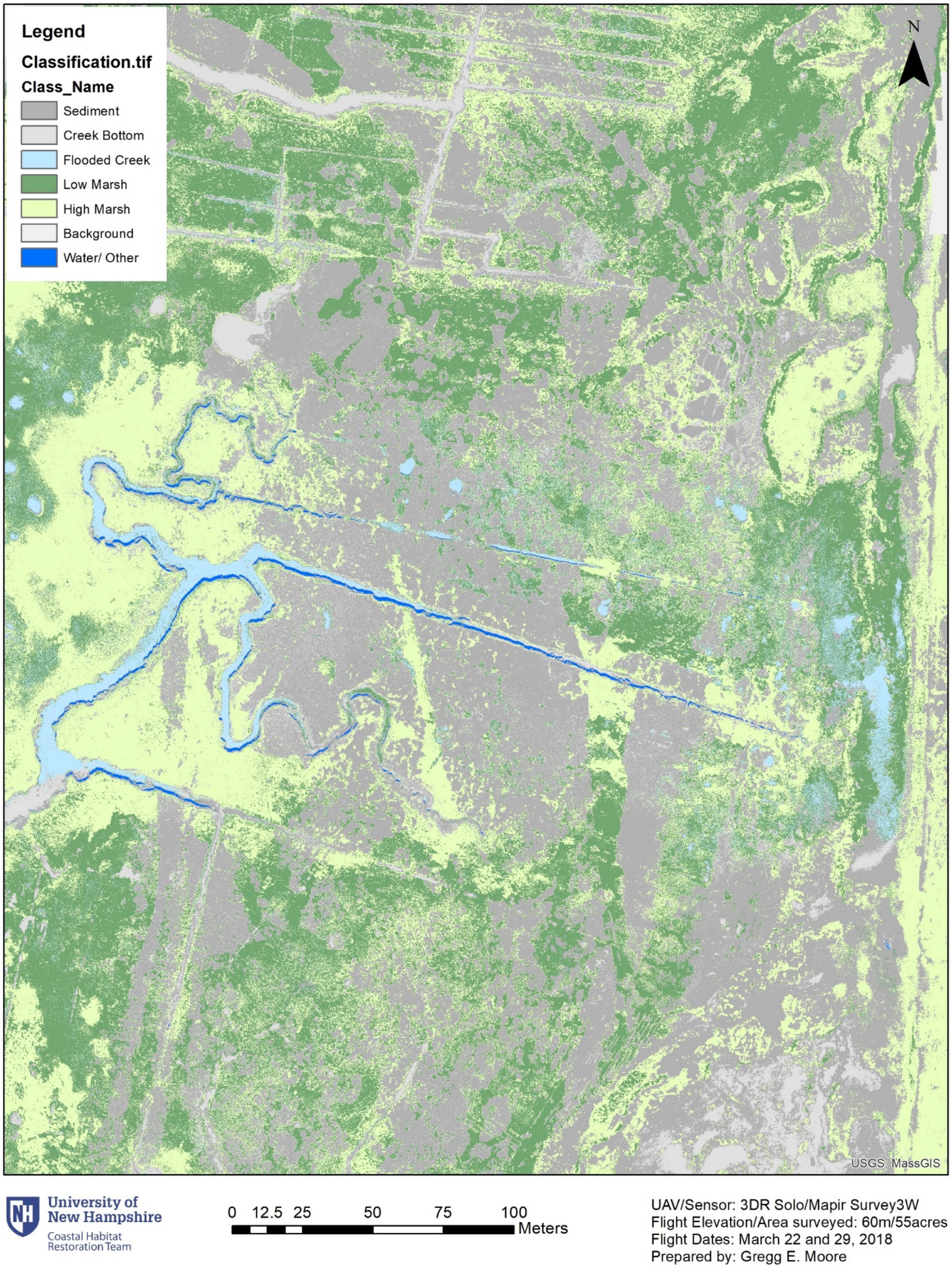

Figure 3(c): Sediment area supervised classification map