Wetlands
DOI 10.1007/s13157-010-0144-5

ARTICLE

Mapping Soil Pore Water Salinity of Tidal Marsh Habitats Using Electromagnetic Induction in Great Bay Estuary, USA

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Received: 7 April 2010 / Accepted: 28 December 2010

Abstract Electromagnetic induction was used to measure apparent conductivity of soil pore water within 15 oligohaline to polyhaline tidal marshes of the Great Bay Estuary in New Hampshire, USA. The instrument was linked to a differential global positioning system via a hand-held field computer to geo-reference data. Apparent conductivity was converted to salinity using a regression derived from field data, and mapped to illustrate spatial salinity gradients throughout the marshes. Plant communities occurring at the study sites included native low marsh, high marsh, and brackish tidal riverbank marsh, as well as communities dominated by native and non-native common reed, *Phragmites australis*. Results revealed mean salinity values were significantly different between each of the community categories sampled within the Estuary. Due to management concerns over expansion of *Phragmites* within the Estuary, we mapped the salinity range for this community and provided graphic and numerical estimates of potential *Phragmites* habitat based on salinity alone (26% of the total acreage surveyed). Electromagnetic induction is an efficient tool for rapid reconnaissance of apparent conductivity and salinity gradients in tidal marsh soils that can be superimposed on aerial imagery to estimate suitable habitat for restoration or invasive control based on salinity ranges.

Keywords Apparent conductivity · Management · *Phragmites australis* · Tidal marsh pools

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Introduction

Salinity is among the most important parameters in determining the structure and maintenance of tidally influenced marsh communities (Odum 1988; Mitsch and Gosselink 2007). Salinity and flood duration can significantly affect seed germination rates and thus community structure in oligohaline marshes (Broome et al. 1995; Baldwin et al. 1996). Salinity can also affect nutrient uptake in both native and invasive tidal marsh plants (Bradley and Morris 1991; Lissner and Schierup 1997), particularly in highly anaerobic soils (Wijte and Gallagher 1996; Chambers 1997; Chambers et al. 1998; Morris and Mendelssohn 2000). Together with restoration of hydrology, salinity is critical in the re-establishment of native plant communities in tidal restoration efforts. Documentation of pore water salinity is necessary to effectively manage and monitor non-native, invasive common reed, *Phragmites australis* (Burdick et al. 2001; Bart et al. 2006). Accordingly, measurement of salinity is widely recommended by published salt marsh monitoring protocols (PERL 1990; Niedowski 2000; Neckles et al. 2002). Commonly

Published online: 16 January 2011

employed methods include the pore water sipper technique, such as that described by Portnoy and Valiela (1997) and use of salinity wells (Burdick et al. 1997). While inexpensive and easy to conduct, these survey methods only show a limited array of soil salinity conditions. Even at a modest scale, they can be labor intensive, particularly for large sites, and may not be representative of conditions more than a few meters from the sampling station. Moreover, soil texture and moisture may affect the ability of these techniques to successfully obtain water for determination of salinity (e.g., clogged sippers and lack of ample interstitial water), often resulting in laborious soil collections and gravimetric analysis or gaps in data sets.

Modern electromagnetic induction instruments (EMI) have been successfully employed to determine in situ apparent conductivity of agricultural soils (Rhoades et al. 1989; Slavish and Petterson 1990; Diaz and Herrero 1992; Nogues et al. 2006; Wittler et al. 2006) as well as to map and infer salinity associated with wetland habitat classifications (Sheets et al. 1994; Paine et al. 2004). Unlike conductivity, which is the measurement of a homogenous medium, apparent conductivity is a volume average conductivity of earthen materials to a specified depth (Greenhouse and Slaine 1983; Doolittle et al. 2001). Various algorithms have been put forward to relate soil apparent conductivity to salinity (Rhoades and Corwin 1981; Slavish 1990; Cook and Walker 1992; Rhoades 1993; Triantafilis et al. 2000; Hendrickx et al. 2002; Wittler et al. 2006). While results have varied, EMI can be calibrated for rapid reconnaissance of soil water salinity (Sheets et al. 1994; Doolittle et al. 2001; Wittler et al. 2006; Morris 2009), especially for sites exhibiting high soil water content (Hanson and Kaita 1997). With advances in GIS-based mapping capabilities and related software, resulting EMI data can now be used to map detailed soil water salinity over the land surface. In this way the EMI techniques hold promise for use in documenting salinity in large brackish to saline tidal marsh areas where traditional methods may be too labor intensive to capture steep gradients and localized areas of low salinity. Such mapping will be valuable for monitoring salinity in response to tidal marsh restoration, sea level rise, or other environmental changes with implications for vegetation management and land use planning.

The goals of this study were to: 1) use EMI to create a detailed GIS-based representation of soil pore water salinity of tidal marshes in the Great Bay Estuary; 2) compare pore water salinity between major marsh community types (low, high, brackish, native and non-native *Phragmites* stands, and unvegetated pools); and 3) map zones vulnerable to non-native *Phragmites* invasion based on the salinity ranges this variety most frequently exploits within the Estuary.

Methods

Study Area and Habitat Categories

The study area included 15 oligohaline to polyhaline tidal marsh sites within the Great Bay Estuary, located in Rockingham and Strafford Counties, New Hampshire (Fig. 1). Four of the sites occur within Great Bay proper, while four occur on the Bellamy River, three on the Cocheco River, three on the Oyster River, and one on the Salmon Falls River. At each site we initially distinguished major plant communities using aerial photo interpretation. Community areas were then confirmed in the field and their boundaries mapped using a differential global positioning system (DGPS). The resulting data were plotted in a GIS environment using ArcMap 9.3 software (ESRI 2009). Communities were simplified into the following five categories, three of which are based on the natural community types described by Sperduto and Nichols (2004): low marsh comprised of native low salt marsh species dominated by *Spartina alterniflora*, high marsh comprised of native high salt marsh species including *Spartina patens*, *Distichlis spicata*, and *Juncus gerardii*, and brackish marsh comprised of native brackish water tolerant species including *Spartina pectinata, Schoenoplectus maritima*, and *Typha angustifolia*. The remaining two habitat categories include *Phragmites*-dominated habitats, including either native or invasive varieties of common reed, *Phragmites australis* (*sensu* Saltonstall et al. 2004), and finally pool habitats consisting of unvegetated water-filled depressions (*sensu* Adamowicz and Roman 2005).

EMI Field Measurements

In the summer of 2007, measurement of apparent conductivity was conducted at each site using a Geonics Model EM38 meter. Alternate current is sent through the meter to generate and receive electromagnetic fields, which consists of two coils 1 m apart and operates at a frequency of 14.6 kHz (Geonics Limited 2006). The EM38 was linked to a hand-held computer (Allegro Field Computer Model CX Field PC) and DGPS (Garmin c176 with CSI Wireless MBX-3S radio beacon receiver), embedding the measured value data, geographic coordinates, date, and time simultaneously in an exportable file. For this study, the instrument was used in the vertical dipole orientation and calibrated at each site, following the manufacturers instructions (Geonics Limited 2006). Calibration was conducted in the marsh habitat, which resulted in the best fit of survey data when compared to calibration in adjacent upland soils. We found the EM38 to be rather responsive as we traversed major habitat types, notably increasing or decreasing in a predictable way. Readings were generally stable within

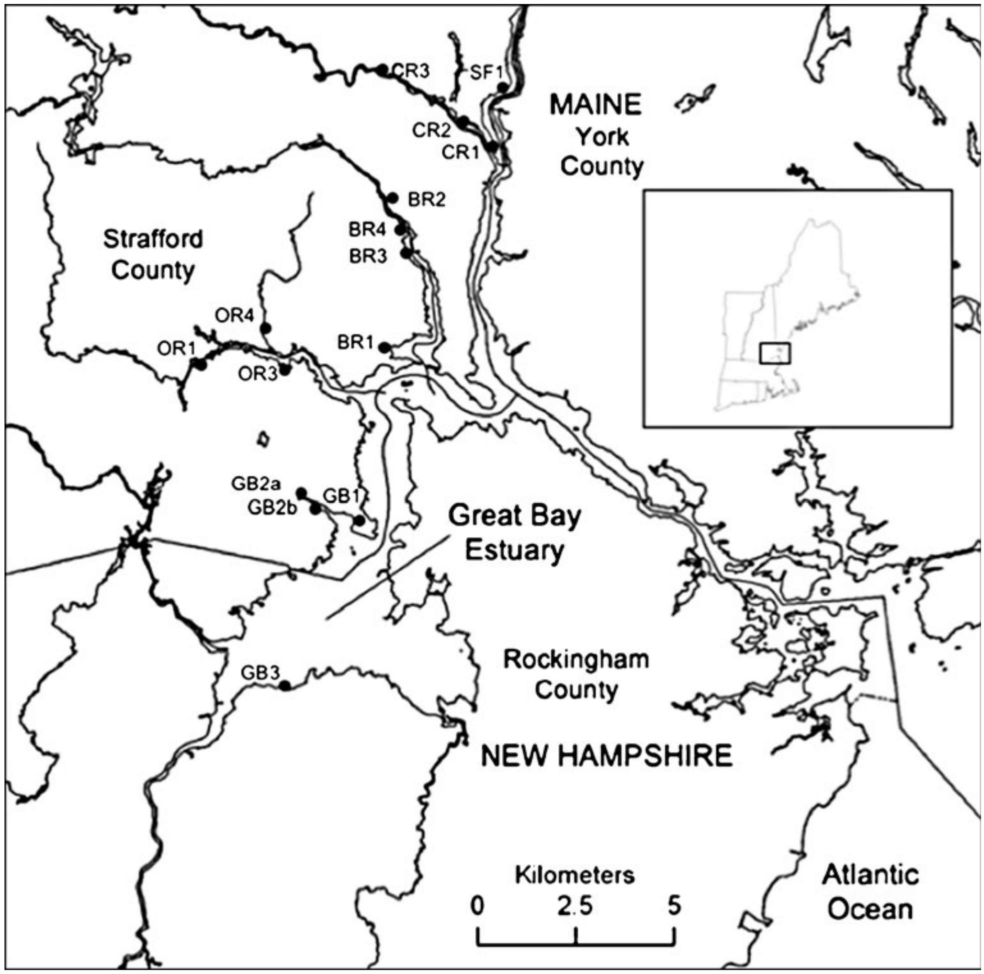

Fig. 1 Study sites within Great Bay Estuary and its tributaries

habitat types, without regard to degree of vegetation. Our observations do not indicate calibration between vegetated community types is required, nor between vegetated and unvegetated habitats.

At each sampling event, the EM38 meter was held 50 cm over the soil and tidal marsh area, and was walked in a zigzag pattern making multiple passes from upper edge to lower edge along the length of the survey area. Each pass was never greater than 20 m away from any adjacent pass, nor closer than 5 m. Linked to DGPS, data were recorded every 3 s and geo-referenced while moving over the marsh surface, resulting in an average of approximately 1,200 samples per hectare. The resulting data were integrated and projected using Geonics DAT38W software v.1.0 (Geonics Limited 2005). Apparent conductivity data from each site were sorted into one of the five defined habitat categories by overlaying the habitat polygons over individual EMI data point locations.

Conversion of Apparent Conductivity to Salinity

Output from the EM38 meter is reported as apparent conductivity in milliSeimans meter^{-1} (EC_a mSm^{-1}); a result of induced current through a maximum penetration depth of 1.5 m of soil, provided the instrument is placed directly on the soil surface (EM38 meter in vertical dipole orientation; Geonics Limited 2006). The strength of the current depends upon ion concentrations in the soil as well as temperature, water content, and soil bulk density (Bork et al. 1998; Brevik et al. 2004; Corwin and Lesch 2005). The relationship between apparent conductivity and soil salinity is strongest when soils have greater than 30% available soil moisture (Hanson and Kaita 1997). For our research, all sites surveyed had uniformly saturated soils of at least 70% due to the nature of tidal marshes in our study area.

While the instrument operation manual suggests conversion of apparent conductivity (mSm^{-1}) to salinity (ppt) by multiplying by a simple correction factor (Geonics Limited 2006), these guidelines were not suited for use across a wide salinity range (J. Doolittle, USDA-NRCS, personal communication), which ranged from 0 to 34 ppt based on actual salinity measures throughout the Estuary. To predict salinity from apparent conductivity values, we sampled apparent conductivity (EC_a using EM38 meter) and salinity (using sipper and refractometer) at locations within each

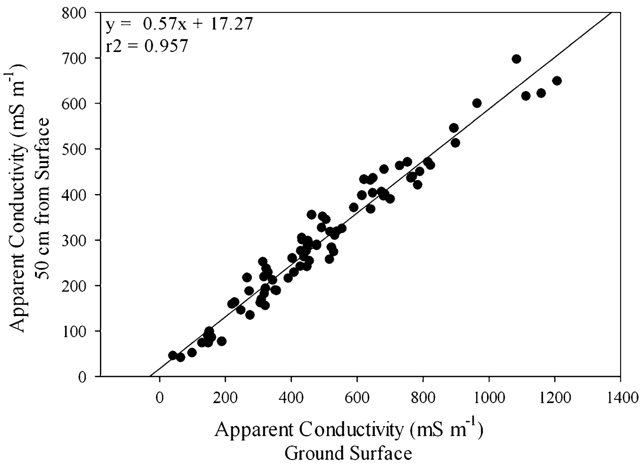

Fig. 2 Comparison of apparent conductivity (EC_a) at different sampling heights (0 and 50 cm) above marsh surface

habitat type at 12 out of our 15 tidal marsh study sites; one additional site also located in Great Bay Estuary was included. Calibration measurements were conducted along an elevation gradient from upland edge to major creek edge to document a representative range of values. On average, six measurements were taken at each marsh, totaling 80 points.

At each point along the gradient, three values of apparent conductivity were recorded for each of three different instrument heights above the marsh surface (i.e., resting on the marsh surface, 10 cm above the marsh, and 50 cm above marsh). These three measures were used to define the optimal instrument height for data quality (Morris 2009). Corresponding pore water salinity measures were obtained at one depth (30–50 cm), using stainless steel sippers and measured using a hand-held temperature-corrected optical refractometer (±2 ppt). When necessary, pore water was filtered in the field using a 0.45 μm cartridge filter. We found that measures of EC_a taken at different instrument heights were highly correlated (range of r^2=0.957–0.988, Fig. 2), but the data set with the instrument height at 50 cm above the marsh yielded the best relationship with salinity collected using the sipper. The 50 cm height allowed for ease of field use, to prevent snagging vegetation or submerging the instrument, and reduces the penetration depth compared to the lower operating heights of 0 and 10 cm. In addition to being the most practical height for safe and efficient use in a challenging saline environment, such a height also captures the maximum live rooting depth of *Phragmites*. We found live belowground biomass of *Phragmites* peaked at 40 cm (*unpublished data*). Thus, pore water sippers at 30–50 cm represented a suitable depth roughly approximating the midpoint of the EM38's effective depth penetration, and the zone of highest biomass for *Phragmites* in area marshes.

Using the data obtained with the instrument 50 cm above the marsh surface, a least squares regression was used to predict salinity from apparent conductivity. Residual analysis indicated a log transform of EC_a values was needed for residuals to exhibit homogeneous variance and normal distribution: $Salinity\,(ppt) = -41.01 + 10.44{*}Ln(EC_a mSm^{-1})$ (r^2=0.50, F=78.6 and P<0.0001). Salinity values reported hereafter are the result of this relationship, which uses salinity alone to account for approximately half of the variability in EC_a (Fig. 3). The other half of the variability is likely due to the limited soil volume sampled by the sipper, error in the refractometer (±2 ppt), and variability in soil moisture, bulk density, and clay content. Variability due to temperature differences was probably minimal at the scale of comparison. Unaccounted variance may also be attributable to apparent conductivity (EC_a) itself, which is an average conductivity horizontally and over depth, unlike conductivity or our relatively more discrete salinity measurements. The size of the three dimensional conductivity structure produced by the meter's electromagnetic field is a function of coil spacing, dipole orientation, and frequency, which also likely affects the relationship with pore water salinity. Salinity means of habitat categories were calculated for each of the 15 marshes included in our study and were analyzed using ANOVA followed by Fisher's Protected Least Significant Difference Test (α=0.05).

Salinity Contour Maps

Once data were converted from apparent conductivity to salinity, a kriging algorithm was used to interpolate values. To display salinity results graphically at each site, we chose to use a regular salinity contour interval of 5 units (e.g., 0–5, 5–10, 10–15 ppt, etc.) through a maximum of 30 ppt

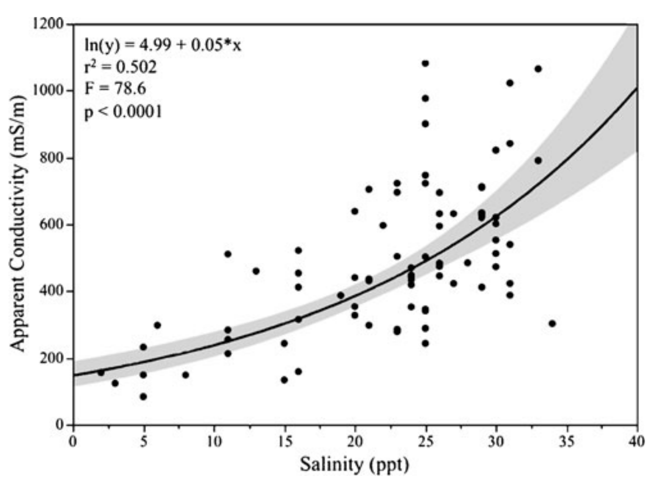

Fig. 3 Regression of apparent conductivity (EC_a) and pore water salinity. EC_a was measured at 50 cm above the marsh surface, whereas salinity was directly measured with a sipper and refractometer. The shaded region represents 95% confidence interval of the regression line

representing the mean salinity range for our sites. However, other break points could be easily used, such as the distinction between oligohaline, mesohaline and polyhaline systems (0–5, 5–18, 18–30 ppt, respectively) as described by Odum (1988). While our results are rough estimates of salinity due to error associated with conversion of EC_a to salinity, we elected to use 5 ppt increments for salinity contour breaks because this depiction facilitates visual interpretation of the data.

Results and Discussion

Apparent Conductivity Contour Maps

Plotting the salinity data from the EM38 measurements as contours revealed diverse conditions between and across marsh sites (Fig. 4), with greater spatial heterogeneity than expected for most sites. The high sampling frequencies recorded at each site resulted in increased resolution of salinity across the marsh than would typically be achieved by conventional approaches (e.g., salinity wells and sippers). However, these salinity contours represent a snapshot of relative patterns rather than stable values due to the potential calibration error and dynamic nature of salinity in estuaries (Fig. 3). Graphic display of the resulting data on this landscape level elucidated two major patterns. One pattern was apparent among sites with prominent drainages (hereafter referred to as "Drainage Sites"), such as those with major creeks, channels and ditches. Drainage Sites demonstrated a relatively linear salinity gradient following elevation and drainage as seen at BR3, and BR4 on the Bellamy River (Fig. 4c and d), and OR1 and OR3 on the Oyster River (Fig. 4m and n). The pattern of salinity at these sites ranged from near zero ppt at the upper edge of marsh, to over 30 ppt in some low marsh and unvegetated mud flat areas. In many ways, Drainage Sites represented a common pattern of salinity in a tidal marsh, following a linear gradient related to elevation.

The second major pattern was found at sites without clearly defined drainage features (hereafter "Poorly Drained Sites"), and was more common in the Estuary. The salinity contours at Poorly Drained Sites were more complex. Salinity contours did not follow a linear gradient, typically exhibiting a proportionally large area of high salinity conditions with lower salinity zones located both up-gradient and down-gradient. Despite this difference from the Drainage Sites, consistent salinity patterns were identified for Poorly Drained Sites and were grouped into two types for further examination: sites with prominent pools, and sites with extensive unvegetated mudflats, typically associated with the low marsh habitat category.

At pool sites, flood waters appear to become trapped and evaporation results in elevated salinity. Sites with pronounced pool habitats included BR1, CR1, GB1, GB3, and particularly at OR4 (Fig. 4a, e, i, l, and o, respectively). Most of OR4 ranged from 20 to 25 ppt, but salinity as high 37 ppt was also measured directly from pore water sampled in isolated locations. Sites with extensive unvegetated mudflats appeared linked to more subtle topographic conditions, such as the presence of natural levees between the main channel and the lower edge of vegetated tidal marsh that result in prolonged periods of flooding and otherwise hinder drainage of tidal waters. These features included seemingly natural microtopography associated with underlying bedrock and soil types, as well as anthropogenic disturbance such as placement of stones, bricks, and other man-made materials associated with historic uses. Sites exhibiting these micro-topographic features included CR2, CR3, SF1, GB2, and to a lesser extent, BR2 (Fig. 4f–h, k–l, and b, respectively). Similar to pool-dominated sites, mudflat areas within the low marsh had a mean soil salinity that was consistently in the 20–25 ppt range. In both cases, the site topography contributed to retention of tidal flood waters, thus presumably the main cause of the elevated salinity values (pool sites) and lack of a simple, linear salinity gradient (pool and mudflat sites).

Comparison of Data by Habitat Category

Grouping the predicted salinity data for all sites, we noted that the vegetated habitats followed a relatively predictable salinity gradient from low salinity brackish habitat to high salinity low marsh. Significant differences were found between each of the five habitat types (Fig. 5). Comparisons were based on analysis of the means for each habitat where the n varied from 6 to 15 sites as explained in Fig. 5. Most notable was the significant difference of salinity found in *Phragmites*-dominated habitat compared to both brackish marsh and high marsh. *Phragmites* occurrences were generally associated with mesohaline conditions, and while some overlap in range (determined by 95% CI; Fig. 5) was noted with adjacent habitat categories, the mean salinity was statistically distinct.

Commonly limited to the landward edge of the marshes sampled, the mean salinity of brackish marsh was lower than all other habitat categories in the study (9.5±1.6 ppt). As an invasive and aggressive colonizer, *Phragmites*-dominated habitat spanned the upland edge through high marsh, with a mean salinity of 15.0±1.2 ppt. This is a typical pattern for invasive *Phragmites* in New England as described by Burdick and colleagues in a set of stands in northern Massachusetts (2001). One stand of native *Phragmites* (*sensu* Saltonstall et al. 2004) was found at GB3 near a non-native stand, but salinity values did not

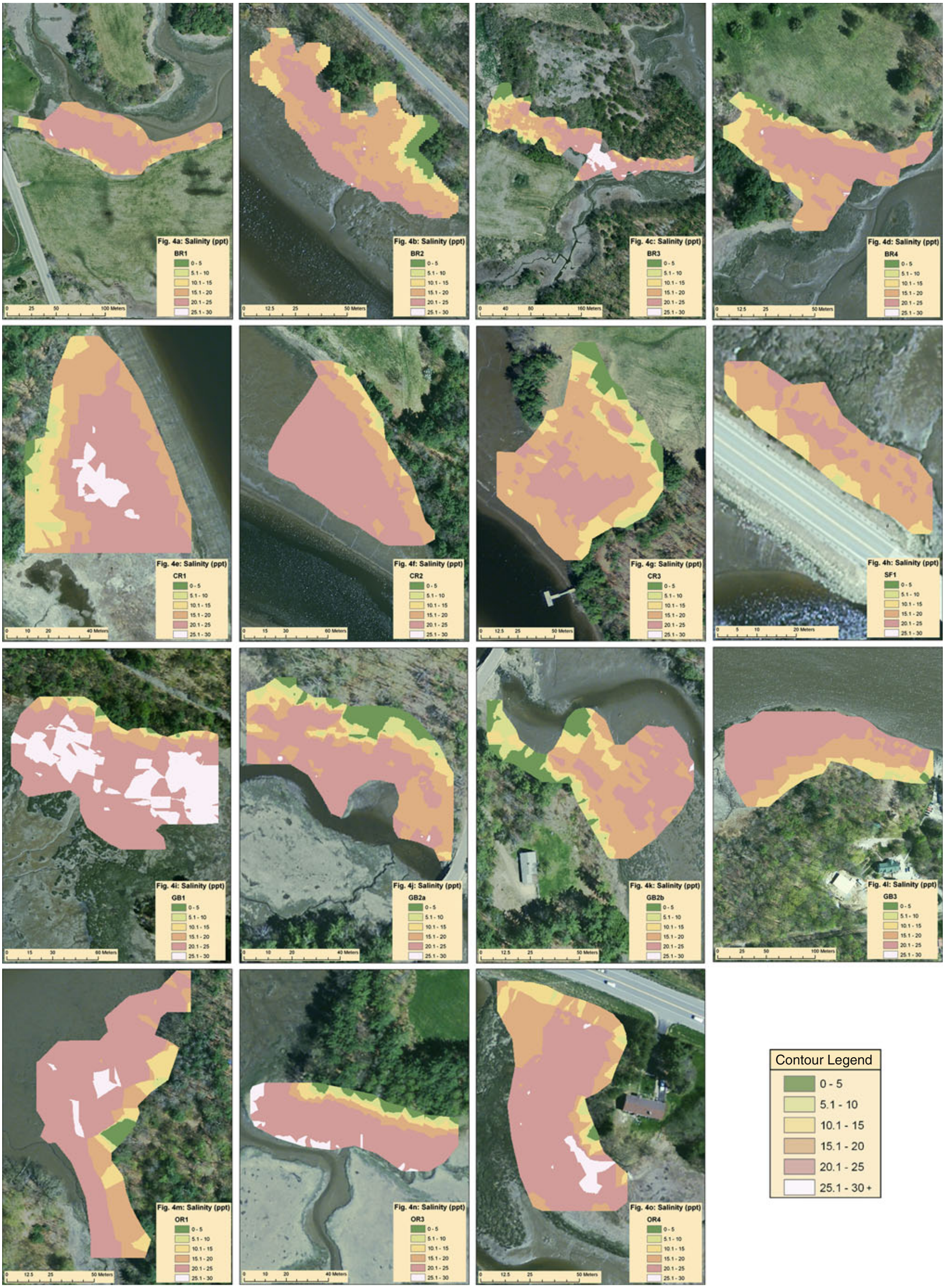

Fig. 4a: Salinity (ppt)
BR1
Fig. 4b: Salinity (ppt)
BR2
Fig. 4c: Salinity (ppt)
BR3
Fig. 4d: Salinity (ppt)
BR4
Fig. 4e: Salinity (ppt)
CR1
Fig. 4f: Salinity (ppt)
CR2
Fig. 4g: Salinity (ppt)
CR3
Fig. 4h: Salinity (ppt)
SF1
Fig. 4i: Salinity (ppt)
GB1
Fig. 4j: Salinity (ppt)
GB2a
Fig. 4k: Salinity (ppt)
GB2b
Fig. 4l: Salinity (ppt)
GB3
Fig. 4m: Salinity (ppt)
OR1
Fig. 4n: Salinity (ppt)
OR3
Fig. 4o: Salinity (ppt)
OR4
Contour Legend
0 - 5
5.1 - 10
10.1 - 15
15.1 - 20
20.1 - 25
25.1 - 30 +

Fig. 4 (**a–o**) Soil pore water salinity (derived from relationship with EC_a) contour maps for 15 sites within Strafford County, NH. BR=Bellamy River, CR=Cocheco River, SF=Salmon Falls River, GB=Great Bay Estuary, OR=Oyster River

differ between the two stands. Means of high marsh habitat averaged 18.3±0.7 ppt and the regularly flooded low marsh averaged 20.3±0.6 ppt. Finally, the non-vegetated pool habitat was distinct in terms of its mean pore water salinity, exhibiting the highest values with a mean of 23.3±0.5 ppt.

Mapping Existing and Potential *Phragmites* Habitat

While it was possible to distinguish habitat categories using our predicted salinity data, it was not our intention to use the EM38 meter to map community categories, *per se*. Major vegetation zone maps can be more accurately constructed at a landscape-scale from aerial photo interpretation coupled with direct field observations as well as with lidar (light detection and ranging) techniques (Paine et al. 2004). Rather, we hoped to define a range of soil pore water salinity associated with invasive *Phragmites australis* and grouping data by habitat category was helpful in generating these ranges. With an empirically derived salinity range defined, we examined the extent of this range within tidal marshes of the Estuary as a broad-scale predictor of the vulnerability of the system to *Phragmites* invasions. Our results identified the presence of *Phragmites* within 6 of the 15 study sites (BR3, CR1, GB3, OR1, OR4, and SF1).

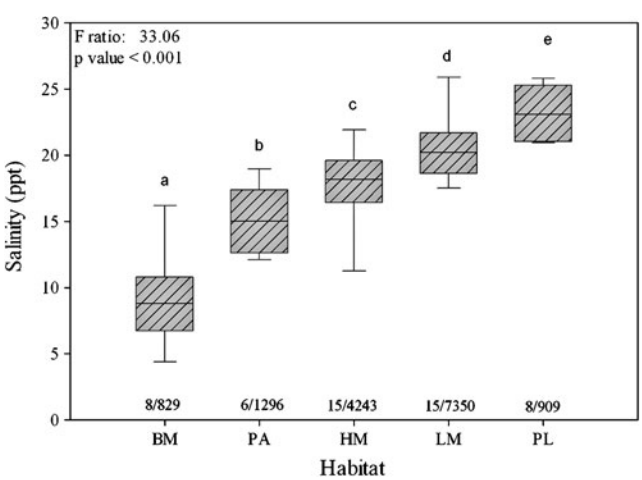

Fig. 5 Comparison of soil salinity (derived from relationship with EC_a) in various marsh habitats as defined by vegetation and surface water (pools). Box and whisker plots show the mean (horizontal line within shaded box), 95% confidence intervals (shaded box), and min/max values for the 15 marsh sites. Means with different letters are significantly different according to a Fischer's Protected F test. Numbers under bars represent the number of marshes containing the corresponding habitat type and the number of data points used to generate the mean for each marsh

When examined on a site by site basis, we noted that the soil pore water salinity of adjacent *Phragmites*-free areas differed little from the *Phragmites*-dominated areas (BR3, GB3 (native stand), OR1, and SF1), while in other cases adjacent areas possessed significantly higher salinity (BR5, CR1, GB3(invasive stand), OR4). While we recognize that additional factors contribute to the presence and success of *Phragmites* in tidal systems, we used our data to map and quantify the total area of exploitable habitat within the marshes surveyed in the study, based on measured soil pore water conditions alone.

We defined invasible condition based on the mean and standard deviation of all data collected within *Phragmites* inhabited areas (n=1,296), resulting in a predicted salinity range of 9.7–19.7 ppt. Of the 11.96 ha sampled, 3.13 ha (or 26.2%) had a mean pore water salinity within this range (Table 1). Moreover, this condition is found at each marsh we sampled, whether *Phragmites* presently exists or not. According to reported values in the literature (Burdick et al. 2001; Chambers et al. 2002), *Phragmites* has the potential to exploit 100% of the tidal marsh area in our study based on the salinity conditions we observed. However, we sought to use the EMI data more conservatively to identify the portions of this total area that may be at the highest risk of invasion.

Electromagnetic Induction in Tidal Marshes

Electromagnetic induction (EM) is a powerful tool to resolve apparent conductivity with multiple applications across a variety of disciplines, including the potential for use in tidal marsh research and management described herein. However, it is clear that a number of considerations and limitations remain before EM can be used reliably in tidal marshes. Instrument calibration, conversion of apparent conductivity (EC_a) to salinity, and the strength of conversion regressions are among the most critical areas requiring further work. From our own challenges in developing this application, we outline several of the specific considerations we encountered, some of which we were able to address, while others remain as items in need of further research.

EM38 Calibration The manufacturers guidelines suggest calibrating the instrument on-site for each use, resulting in a site-specific calibration. We explored the importance of this step by calibrating the instrument in a dry, upland habitat vs. in a salt marsh. Marshes were sampled using both calibration locations and found a discrepancy in the scale of readings when using the upland calibration, resulting in negative EC_a values at and near the landward boundary of the site and an overall compression of the EC_a range. In contrast, resulting EC_a values were within expected ranges

Table 1 Study sites by water body with survey area totals, presence of *Phragmites*, and calculated potential areas of future *Phragmites* expansion

Site code	Total surveyed area (hectares)	Existing *Phragmites* (hectares)	Potential *Phragmites* (hectares)
Bellamy River			
BR1	0.87	0.00	0.22
BR2	0.62	0.00	0.21
BR3	1.74	0.15	0.55
BR4	0.41	0.00	0.12
Cocheco River			
CR1	0.72	0.08	0.20
CR3	0.97	0.00	0.15
CR5	1.17	0.00	0.53
Great Bay			
GB1	1.05	0.00	0.07
GB2a	0.38	0.00	0.09
GB2b	0.56	0.00	0.16
GB3	1.45	0.32	0.41
Oyster River			
OR1	0.94	0.21	0.15
OR2	0.33	0.00	0.03
OR3	0.64	0.03	0.16
Salmon Falls			
SF1	0.11	0.01	0.06
Total	11.96	0.80	3.13

and were relatively repeatable when using marsh calibration. Thus we recommend the calibration be conducted within the marsh itself for each individual site.

Conductivity-Salinity Relationship The most important research needed to further the method should be aimed at improving EC_a-salinity regression. EC_a was rapidly measured using a terrain-conductivity meter (EM38) and subsequently related to direct measurements of salinity. Although our results have found EC_a and salinity to be related (r^2=0.502; n=80), it is likely this relationship would benefit from measuring conductivity, which would diminish error associated with spatial variability in soil properties (e.g., texture, organic matter, water content). Instead of measuring a larger 3D conductivity structure, conductivity would reflect a discrete measurement similar to our salinity data. Commonly used methods to obtain conductivity include soil paste extracts (Williams and Baker 1982) or inversion techniques (Hendrickx et al. 2002; Schultz and Ruppel 2005; Schultz et al. 2007), but are time-consuming and impractical for large-scale, high-resolution ecological applications such as our study, which attempts to generate salinity maps of tidal marshes in and around Great Bay Estuary. Short of employing these rigorous techniques, the empirical relationship might be improved by (1) gathering more detailed soil data to better understand the three dimensional distribution of subsurface structure, (2) obtaining a greater number of calibration points evenly distributed across the range of salinity and using means of subsamples, or (3) through encouraging manufacturers of such instruments to design a model type better suited for higher salinity conditions. In addition, to gain confidence in the accuracy of inferred salinity, spot checks of pore water salinity are recommended.

Applications Within the context of our study results, we feel the EM38 is suitable for rough estimation of salinity inferred from EC_a, particularly at larger sites where collection of pore water for determination of salinity is impractical. This technique is also well suited for lower salinity sites, where the predictive strength of the regression is greater (based on our current regression). In all cases, we found this method equally effective whether sites were vegetated or unvegetated, as well as at sites with common topographic features including berms, banks, and shallow pools at low tides such that surface flooding is avoided.

Given the error we encountered, we feel EM is not suitable for particularly small sites where few samples would be collected, or for spot measurements that could be easily accomplished with traditional techniques such as pore water collection and refractometer measurements. As our work shows, quite a few data points need to be collected to characterize the EC_a and salinity at a location.

Our data also suggest that the EM38 may not be able to discern small differences in salinity, especially at high concentrations in salt marsh applications (Fig. 3) or in habitats where heterogeneous soils exist. However, refinement of the regression could overcome these constraints. Finally, we would also discourage the use of EM during flood tides or in close proximity to man-made structures, the latter of which may interfere with electromagnetic fields (e.g., powerlines).

Conclusion

While the overall area covered in our survey was relatively small, there are approximately 854 ha of tidal marsh in the Estuary as a whole (Odell et al. 2006). Our data suggest that there may be a significant threat for expansion of non-native *Phragmites* broadly within the Estuary, including sites with and without existing *Phragmites* populations.

Notwithstanding the potential error in the calibration curve and the variability of apparent conductivity measurements, we believe electromagnetic induction techniques continue to hold promise for characterizing soil salinity patterns in tidal marsh environments. Several authors have presented arguments for the effective use of EMI techniques for rapid assessment of salinity (Slavish and Petterson 1990; Diaz and Herrero 1992; Sheets et al. 1994; Reedy and Scanlon 2003), but few have shown EMI useful in assessing soil salinity in tidal wetlands (Paine et al. 2004), perhaps due to unexplained variability. In our own experience, the calibration regression to predict salinity was better at the lower salinity range, which may very well be linked to the configuration of the EM38 that was designed for use in lower conductivity ranges of agricultural soils (Triantafilis et al. 2000; Wittler et al. 2006). Future work could identify environmental factors needed to refine the salinity calibration and explore ways to reconfigure the instrument for a wider range of conductivity values.

Building on the work of Sheets et al. (1994) who assessed restoration potential of riparian habitat in New Mexico, our study is the first to utilize EMI to estimate and map soil salinity in tidal marsh environments and to map and predict areas of habitat that are suitable for particular community types or species. The landscape-scale, geo-referenced salinity contour maps that can be produced using EMI can be manipulated and analyzed in GIS format to visualize and quantify relative salinity conditions with applications for land management. For example, salinity mapping may assist in determining optimal timing and location of native plantings within tidal restoration sites as new site hydrology develops. EMI-derived maps can identify particular soil pore water conditions potentially suitable for invasive plant species in tidal marshes (e.g., non-native *Phragmites* or other species of concern). With further study and refinement, application of this technology may aid rapid assessment of soil pore water salinity for planning and monitoring of tidal marsh restoration projects to quantify existing conditions and changes in salinity due to hydrologic disturbance or changing climate.

Acknowledgments We thank James Doolittle of the United States Department of Agriculture Natural Resources Conservation Service National Soil Survey Center for technical assistance with the EM38 instrument and for helpful comments on the manuscript, as well as Don Richard for technical support with modeling data output in GIS. We thank David Shay of Jackson Estuarine Laboratory for tireless field assistance, and four anonymous reviewers for their helpful suggestions to improve the manuscript. This work was supported by the United States Department of Agriculture Natural Resources Conservation Service (Federal Award # 721428–6A380) and the New Hampshire Fish and Game Department. Published as Scientific Contribution Number 501 from the Jackson Estuarine Laboratory and Center for Marine Biology at the University of New Hampshire.

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