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Implementing the NERR Sentinel Sites Program at the Narragansett Bay Research Reserve to Track Salt Marsh Responses to Climate Change Stessors

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1. Introduction

Salt marshes protect coastlines from storms, filter nutrients, provide habitat for economically and ecologically valuable fauna, and sequester large amounts of carbon (Valiela and Teal 1979, Boesch and Turner 1984, Gedan et al. 2011, Mcleod et al. 2011). Over 50% of this valuable habitat has already been lost in Rhode Island due to human activities (Bromberg and Bertness 2005) and the remaining marshes face threats from sea level rise, global climate change, invasive species, tidal restrictions, adjacent human development, eutrophication, and overgrazing. Fortunately, most salt marsh habitat within the Narragansett Bay National Estuarine Research Reserve (NBNERR or Reserve), has escaped major impacts from tidal restrictions, elevated nutrient inputs from surrounding developed land, and other direct human activities. However, these marshes are now showing signs of stress in the form of altered nekton communities and increasing occurrences of marsh vegetation die-off events (Raposa and Durant unpublished data). It is possible that these and other changes are the result of elevated nutrient inputs from estuarine sources in the middle and upper portions of Narragansett Bay (Rhode Island Sea Grant 2005) and/or climate change-associated stressors such as sea level rise.

The National Estuarine Research Reserve (NERR) Sentinel Sites program was developed in order to determine how salt marshes are affected by climate change and sea level rise (NERRS 2012). This program grew organically from ongoing salt marsh monitoring efforts in the NERRS (e.g., emergent vegetation bio-monitoring [Moore 2009]) and was later included as an Application Module within the broader NERR System-wide Monitoring Program (NERRS 2011). Its major field components include emergent vegetation structure, wetland surface elevation change, water quality, and meteorology monitoring coupled with the development of a vertical reference system for water level measurements and the ability to detect elevation change. As of 2012, most NERR reserves were implementing one of more core components of Sentinel Sites, but only 7 reserves had achieved full Sentinel Sites status. The NBNERR began emergent vegetation monitoring in 2000 as part of a local salt marsh restoration (Raposa 2008) and continued building other Sentinel Sites infrastructure while participating in the NERR Reference Site project (Raposa and Weber 2009, Raposa and Weber 2011, Dionne et al. 2012). By 2012, all that remained for the NBNERR to achieve full NERR Sentinel Sites status was to install surface elevation tables (SETs) and permanent water level logging platforms and tie them all in to the local vertical control network.

The overall goals of this project were to 1) continue NERR emergent vegetation and elevation monitoring in the Coggeshall and Nag salt marshes in the NBNERR, and 2) install and tie in SETs to attain full NERR Sentinel Sites functionality (water level loggers were also installed and tied in to the vertical network in 2012, but not as part of this project). Achieving these goals will allow us to improve our ability to evaluate salt marsh ecosystem responses to climate change and sea level rise stressors. More specifically, it will allow us to quantify temporal changes in vegetation community composition and determine if the extent of vegetation die-off events is increasing over time. It will also allow us to build additional monitoring infrastructure to measure marsh surface platform elevations and determine where these wetlands sit within the tidal frame. Thus, the proximate benefits include improved assessments of salt marsh vegetation, vegetation die-off events, and marsh surface elevations in the Reserve's marshes; the overarching benefit is full Sentinel Site implementation and the subsequent ability to assess and track salt marsh responses to climate change and sea level rise.

2. Methods

2.1 Study Sites

This study was conducted in the Coggeshall and Nag salt marshes on Prudence Island, RI within the NBNERR (Fig. 1). Unlike most salt marshes in Narragansett Bay, Coggeshall and Nag marshes remain relatively un-impacted by common anthropogenic stressors such as tidal restrictions, invasive species (i.e., the common reed *Phragmites australis*) or elevated terrestrial anthropogenic nutrient inputs. Both marshes are comprised of networks of tidal creeks, ponds, pools and historic mosquito ditches, and emergent vegetation communities are dominated by typical New England salt marsh species (e.g., *Spartina alterniflora*, *Spartina patens*, *Distichlis spicata*, *Juncus gerardii*, *Iva frutescens*, etc.) predictably distributed along an elevation gradient (Niering and Warren 1980). A more complete description of both marshes can be found in Raposa and Weber (2011).

Coggeshall and Nag marshes have been the focus of a considerable amount of NERR-related research and monitoring efforts. Both marshes have not only been studied extensively by Reserve staff, but also by researchers from Brown University, the Environmental Protection Agency, and many other institutions. Habitat types in each marsh were mapped in 2003 in accordance with a draft NERR habitat classification system (Kutcher et al. 2004). From 2000-2004, Coggeshall Marsh served as the reference site when the Reserve restored tidal flow to the nearby Potter Pond salt marsh (Raposa 2008). Coggeshall and Nag marshes both served as reference marshes during the 2008-2010 Reference Site Project collaboration between the NOAA Restoration Center and five National Estuarine Research Reserves (Raposa and Weber 2009, Raposa and Weber 2011; Dionne et al. 2012). Nag Marsh has also served as one of the Reserve's SWMP water quality monitoring sites since 2002. Prior to this study, these combined efforts had already produced baseline datasets for emergent vegetation, marsh elevations, sediments, and hydrology in both marshes. Maps of both marshes showing the locations of all Sentinel Site monitoring infrastructure (including those components that were installed during this 2011-2012 study) are shown in figures 2-3.

2.2 Emergent Vegetation

Emergent vegetation in both marshes was monitored using the NERR Tier 2 standardized bio-monitoring protocols (Moore et al. 2009), which are largely based on National Park Service salt marsh vegetation monitoring protocols (Roman et al. 2001). The NERR Tier 2 monitoring protocols involve annual marsh vegetation sampling at permanent plots located along a series of randomly-located transects that encompass water-to-upland gradients. Three permanent transects and monitoring plots were previously established in Coggeshall Marsh in 2000 (21 plots) and three transects were established in Nag Marsh in 2008 (24 plots). In this study, all plots in each marsh were sampled at the peak of the growing season (August/September) in 2011 and 2012. Sampling involved 1) placing a 1-m^2 PVC quadrat directly on each plot, 2) visually surveying the entire plot to identify all vegetation species present, 3) quantifying the percent cover of all vegetation species in the 1-m^2 quadrat using the point-intercept method (Elzinga et al. 1998) at 50 grid points within the quadrat, and 4) taking digital pictures of each plot to provide a permanent photographic history of vegetation at each location.

The heights and stem densities of target species/species of concern were also measured in each quadrat to augment the datasets that were collected during the 2000-2004 Potter Pond restoration study and the 2008-2010 NERR Reference Site project. Target species include *S. alterniflora*, *S. patens*, *D. spicata*, *J. gerardii*, and *Schoenoplectus americanus*. When these species were present, the heights of up to twelve random stems were measured per plot (as described in Roman et al. 2002). This is different from the primary method used during the NERR Reference Site Project where the three tallest stems of each species were measured in each quadrat. In our experience, the latter only worked well for erect, rigid species such as *S. alterniflora* and *P. australis*. For most other species, we found it difficult to find and measure the three tallest individuals in a plot. For this reason, and to ensure compatibility with the 2000-2004 height data collected from Coggeshall Marsh, we decided to use the Roman et al. (2002) method for measuring vegetation heights in this study and for all of our future monitoring efforts. Finally, the number of stems of each target species (when present) was counted from within a sub-quadrat (0.0625 m^2 for tall-form *S. alterniflora* [i.e., growing along creek banks and generally greater than 75-cm in height] or 0.01 m^2 for shorter *S. alterniflora* and all other species) that was placed in a standardized location within the larger 1-m^2 plot (the far right corner when facing the upland). These sub-quadrat stem counts were then converted to densities by extrapolating to the larger 1-m^2 quadrat.

2.3 Elevation mapping with real-time kinematic GPS

2.31 Vegetation plots and shallow water wells.

We captured elevations of vegetation monitoring plots and PVC hydrology wells in 2012 using real-time kinemetic (RTK) GPS as described previously in Raposa and Weber (2011). These data augment similar data that were collected in 2009 (Nag Marsh) and

2010 (Nag and Coggeshall marshes). In each vegetation monitoring plot, elevations were measured at a single point in the estimated center of the plot, while hydrology well elevations were collected from the uncapped top of the PVC well pipe. All RTK elevation data were collected from June 6-15, 2012.

2.32 Elevation profiles

Elevation profile data were collected with the RTK along all monitoring transects in 2012 following the methods described in Raposa and Weber (2011). Elevation data were taken at 0.5-1.0 m intervals stretching from the beginning of the transect (typically a tidal creek) into the marsh/upland border (typically the *I. frutescens*/*Baccharis halimifolia* shrub zone). These data complement similar profile datasets that were collected in 2009 and 2010 in Nag Marsh and in 2010 in Coggeshall Marsh. All elevation profile data were collected from June 12-20, 2012.

2.33 Habitat transitions

Transitions between adjacent habitat types were mapped with the RTK along all transects in both marshes in 2012 following the methods described in Raposa and Weber (2011). When coupled with similar habitat transition data that were collected from both marshes in 2010, these new data allow us to examine how salt marsh habitat composition may have changed over time. In the field, we mapped the locations where dominant vegetation species changed (e.g., from *S. alterniflora* to *S. patens*, from a *S. patens*/*D. spicata* mix to an *I. frutescens*/*D. spicata* mix, etc.). We then converted these species into logical and ecologically meaningful overarching habitat types. The definitions we used for these habitat types are (in alphabetical order):

- <u>Creek/ditch</u>
    - Any intertidal or subtidal creek or mosquito ditch. Mosquito ditches include those that remained functional, allowing for the transport of tidal water and remnant ditches that have begun to fill in with sediment and colonizing *S. alterniflora*;
- <u>Grazed creek bank</u>
    - Any area along a tidal creek where herbivores have clearly grazed away all vegetation and exposed bare peat;
- <u>*Iva frutescens*</u>
    - Any section of the upper marsh/upland border where the high tide bush *I. frutescens* is present;
- <u>Panne</u>
    - Panne definitions vary widely; in our study we defined a panne as a previously vegetated area that is now completely or near-completely bare due to vegetation die-off events;
- <u>Pool</u>

    - A pool is any isolated depression with relatively steep sides within the vegetated marsh platform that generally contains water even at low tide (Adamowicz 2005);
- *S. alterniflora*
    - Any area where the only dominant species is cordgrass *S. alterniflora*. No attempt was made to differentiate between tall and short forms of *S. alterniflora*, since our prior data show that two distinct life forms of this species do not clearly exist in our study marshes and *S. alterniflora* height instead progresses along a continuum;
- *S. alterniflora*/lower salt meadow
    - Any area dominated by any combination of *S. alterniflora*, *S. patens*, and *D. spicata*;
- *S. alterniflora*/*Salicornia* spp.
    - Any area where *S. alterniflora* and *Salicornia* spp. are co-dominant. No attempt was made to differentiate between annual and perennial life forms of *Salicornia* spp.;
- *S. alterniflora*/upper salt meadow
    - Any area where *S. alterniflora* and *J. gerardii* are co-dominant or where *S. alterniflora*, *J. gerardii*, and *D. spicata* are co-dominant;
- *Salicornia* spp.
    - Any area where the only dominant species is *Salicornia* spp. No attempt was made to differentiate between annual and perennial life forms;
- Lower salt meadow
    - Any area where *S. patens* and *D. spicata* are either sole or co-dominants;
- Unknown
    - Any area of marsh where the habitat could not be identified by field personnel;
- Upper Brackish meadow
    - Any area of marsh near the upland border that includes brackish species (e.g., *Schoenoplectus americanus*);
- Upper salt meadow
    - Any area where *J. gerardii* is either the lone dominant or is co-dominant with *D. spicata*;
- Walking trail
    - Any area of marsh that is clearly trampled by either human or animal foot traffic;
- Wrack
    - Any area of marsh that is covered by wrack (generally composed of dead *S. alterniflora* stems).

Most of these habitat types were defined *a priori* based on our experience working in southern New England salt marshes; others were added in the field when we encountered a new habitat that was not already defined (e.g., grazed creek bank). The minimum mapping size of a habitat was 0.25 m in length. Mapping was always conducted while moving away from the estuary towards the upland. All RTK habitat transition mapping was conducted from June 12-20, 2012.

2.34 Mapping die-off patches

We also originally proposed to select 6 locations in each marsh in 2012 where we would use the RTK to compare elevations of healthy vegetated salt marsh and elevations where marsh vegetation had died-off. Although we were not able to accomplish this as proposed due to time constraints, we were able to quantify the elevations and extent of die-off areas (i.e., pannes) using the elevation profile and habitat transition data collected with the RTK as described above in sections 2.32 and 2.33. With the habitat transition data, we were able to record the specific locations of all die-off boundaries along each of the 6 monitoring transects. With the combined profile and habitat transition data, we were then able to quantify the elevations of die-off areas compared to other vegetated habitat types as described below in section 2.64.

2.4 Elevation mapping with digital leveling equipment

We used the NERR Leica DNA03 digital leveling equipment to determine the elevations relative to NAVD88 of each of the 12 new surface elevation tables (SETs) that were installed in the two marshes in 2012 (see section 2.5 below). Detailed methods on how we use this equipment for determining elevations are described in Raposa and Weber (2011). At each SET, elevations were measured at the highest point on the uncapped metal SET receiver (this high point on the receiver was determined with a level and marked with a permanent marker). All leveling of SETs was conducted from December 4-12, 2012.

2.5 Surface Elevation Tables

In 2012, we installed 12 SETs and associated marker horizons that will allow us to measure micro-topographic changes in marsh platform elevation and accretion rates in response to sea level change over time (Webb et al. 2013). Teams of 4-5 people installed 6 SETs and feldspar marker horizons in each marsh following the protocols established by the US Geological Society (Cahoon et al. 2002 a, b). All SETs were installed from July 31-August 29, 2012 along each of the permanent monitoring transects. Along each transect, one SET was randomly established in stunted *S. alterniflora* habitat, and another was randomly established in *S. patens* and/or *D. spicata*-dominated salt meadow habitat (i.e., lower salt meadow). These two habitats currently dominate the salt marsh platform in southern New England and we wanted to be able to track how elevation is changing over time in each habitat. All SETS were established within 5 m of their associated monitoring transect. Initial baseline readings were taken from each SET from October 10-11, 2012.

2.6 Data analysis

2.61 Emergent vegetation

We used cluster analysis and non-metric multi-dimensional scaling (MDS) to identify and visualize in two-dimensional space distinct groups of monitoring years based on our point-intercept data. Hierarchical clustering was performed using group-average linking of Bray-Curtis similarities calculated on square-root transformed data. MDS was also performed using Bray-Curtis similarities from square-root transformed data. We then used analysis of similarity (ANOSIM) to statistically compare vegetation community composition among the year-groups identified with cluster analysis. Similarity percentages (SIMPER) was then used to identify species that contributed to community similarity within each year-group and community dissimilarity between any year-groups that were found to be significantly different with ANOSIM. All ANOSIM and SIMPER analyses were performed using square-root transformed data and Bray-Curtis similarities. All cluster analyses, MDS, ANOSIM, and SIMPER tests were run separately for Coggeshall and Nag marshes because the data were collected over different periods of time, and all analyses were run using PRIMER version 6.1.2 (Clarke and Warwick 2001, Clarke and Gorley 2006). In addition, temporal trends in individual species percent cover and target species heights were examined in each marsh using best-fit linear and non-linear regression analyses in SigmaPlot version 12 and SigmaStat version 3.5.

2.62 Elevation profiles

In order to graphically display elevation profile data along the same transect from different years we standardized all data point locations to a single starting location at the beginning of each transect. The permanent starting location for each transect was created using ARCGIS version 10.0. It was necessary to standardize the data post-hoc because transect profiles were not always started from the exact same location in the field each year. Additionally, all of our elevation data were recorded in two dimensional space (x and y coordinates), while it was necessary to plot it along a one dimensional axis for this analysis. This approach allowed us to visualize transect geomorphology and identify any erroneous or outlying data that may have been collected. To examine changes in marsh platform elevation over time, we averaged all of the elevation data across each individual transect each year. For each transect, elevation data were clipped to the shortest profile length, since field staff did not always start or end the transects in the exact same locations each year.

2.63 Habitat transitions

All habitat transition data were standardized to a single starting location on each transect as described above (section 2.62). For each individual transect, all data from 2010 and 2012 were color coded by habitat type and data were plotted side-by-side to visualize how habitats changed during this time period along the lengths of each transect. Two-way ANOSIM was used to compare habitat community composition between marshes and years. We then used SIMPER to identify species that contributed to habitat community similarity within each marsh and community dissimilarity between marshes and/or years that were found to be significantly different with ANOSIM. All ANOSIM and SIMPER analyses were performed using fourth-root transformed data and Bray-Curtis similarities. Finally, at every meter interval along each transect we determined the habitat that was present in 2010 and then again in 2012. In this way we were able to quantify temporal habitat dynamics at every location along each transect. This included quantifying specific shifts from one habitat to another and the stability of each habitat type across the two-year period.

2.64 Profile/Transition combinations

We combined the profile and transition data to quantify elevations for every habitat type in the marshes. All data that had already been standardized relative to the beginning of each transect were used, and we clipped the elevation profile data using the boundaries (i.e., transitions) of each habitat type. We then used those data pooled across the 2 marshes to calculate mean elevations for each habitat type. We only used data from 2012 for this analysis because of the issues encountered with some of the 2010 elevation data in Nag Marsh (see section 3.22 below).

3. Results and Discussion

3.1 Emergent Vegetation

We successfully sampled emergent vegetation percent cover, heights, and stem densities from all monitoring plots in both marshes in both 2011 and 2012. Community composition and percent cover of all species from all available monitoring years (including 2011 and 2012 from this study) are shown in Tables 1 (Coggeshall Marsh) and 2 (Nag Marsh). In general, very few species were found in either marsh, and both marshes remain overwhelmingly dominated by only five species (*S. alterniflora*, *S. patens*, *D. spicata*, *J. gerardii*, and *I. frutescens*). Aside from these five dominant species only four additional vegetation species were found in Nag Marsh in 2011-2012, while only three additional species were found in Coggeshall Marsh.

In Coggeshall Marsh, three groups of monitoring years were identified at the 88% similarity level using cluster analysis. Group 1 included the three earliest years of monitoring data (2000, 2003, and 2004), group 2 the first two years of data collected during the NERR Reference Site project (2008 and 2009) and group 3 the remaining year of Reference Site project data and the two years of data collected during this study (2010, 2011, and 2012). Two-dimensional spatial relationships of these three groups of data were then visualized using MDS (Fig. 4). Based on ANOSIM, overall vegetation community composition differed significantly over these three periods of time (Global R=0.038; p=0.007) due to a significant shift in vegetation composition between 2000-2004 and 2010-2012 (pairwise test; R statistic=0.056, p=0.004). Further, the relative importance of individual species that typified Coggeshall Marsh also changed over time based on SIMPER. For example, the overall contribution of *S. alterniflora* to community similarity within each year-group increased from 49% (2000-2004), to 55% (2008-2009), to 62% (2010-2012) (Table 3). At the same time, the contribution of *S. patens* decreased steadily (34%, 29%, and 16%, respectively). Bare ground did not rank among the cover types contributing to a cumulative total of 90% similarity during the first two year-groups, but by 2010-2012, bare ground contributed 6% to overall community similarity, highlighting the increasing prevalence of this habitat type in recent years.

In Nag Marsh, two groups of monitoring years were identified at the 92% similarity level using cluster analysis. Group 1 included data from 2008 and 2009, while group 2 included data from 2010, 2011, and 2012. Two-dimensional spatial relationships of these two groups of data were then visualized using MDS (Fig. 4). Vegetation community composition did not change in Nag Marsh over time across the two year groups (ANOSIM, Global R=-0.015, p=0.69). However, contributions of *S. alterniflora* and *S. patens* to overall Nag Marsh community similarity did change over time between the two year-groups based on SIMPER. In 2008-2009, community similarity was mostly defined by *S. patens* (47%), followed by *S. alterniflora* (34%) (Table 4). The opposite was observed during the 2010-2012 period, when *S. alterniflora* and *S. patens* contributed 39% and 38% to community similarity, respectively. This highlights the increasing prevalence of *S. alterniflora* relative to *S. patens* in Nag Marsh over time.

The temporal vegetation patterns in both marshes revealed with SIMPER are supported by changes in *S. alterniflora*, *S. patens*, and bare ground percent cover over time. In Coggeshall Marsh, we documented a significant nonlinear increase in *S. alterniflora* and a significant nonlinear decrease in *S. patens* over time (Table 1; Fig. 5). A significant nonlinear decrease in *S. patens* was also documented in Nag Marsh, but no change in *S. alterniflora* cover was detected in this marsh over time. Foreword projections indicate that if these patterns continue the salt meadow foundation species *S. patens* will be lost from both marshes by approximately 2018. In the background of these dominant changes in *S. alterniflora* and *S. patens*, small yet steady increases in bare ground were observed in both marshes over time (though statistically significant only in Coggeshall Marsh); this is largely a reflection of an increasing amount of unvegetated panne habitat (see below).

Using data dating back to 2000, we found no change in mean *S. alterniflora* height over time in Coggeshall Marsh (linear regression, $p>0.05$; Table 5). We did not feel that it would be meaningful to statistically compare means from only 2 years of data for *S. alterniflora* in Nag Marsh or for the remaining 4 species in either marsh. We also did not statistically compare mean stem densities of any species across only 2-3 years of data (Table 6). Nevertheless, these additional height and stem density data provide a solid baseline that can be used to assess change over time with future monitoring. Of some concern, however, is the occasional disconnect between stem density and percent cover data. For example, in Nag Marsh between 2010 and 2012, *J. gerardii* cover remained stable, but stem densities decreased from 346 to 0 stems m^{-2}. We feel this is a problem associated with collecting stem density data with a small sub-quadrat placed in a fixed location within the larger 1-m^2 cover quadrat. In some cases for some species (e.g., for *J. gerardii*), cover estimates were high within the larger quadrat, but because of small-scale within-quadrat patchiness, this same species might not be present at all in the far right-hand corner where the sub-quadrat was always placed. However, at this point we cannot recommend a better or alternative approach. We do not recommend a larger-sized sub-quadrat or more replicate sub-quadrats as both of these options would be too time consuming. We also do not see any merit with allowing field personnel to randomly place the sub-quadrat within the larger quadrat since small-scale patchiness would remain an issue. Hopefully, long term stem density monitoring will eventually dampen the extreme variability encountered when using small sub-quadrats to sample sometimes very patchy species.

Overall, our vegetation monitoring data clearly show that salt marsh vegetation is changing over time in both marshes. The most dramatic and conspicuous change in both marshes is the rapid loss of the salt meadow foundation species *S. patens*. This loss is actually accelerating over time to the point where if these rates of loss continue *S. patens* may be completely lost from both marshes as early as 2018. Other vegetation changes differed somewhat between Coggeshall and Nag marshes. For example, community composition changed significantly in Coggeshall Marsh over a 13-year period, but no changes were observed in Nag over a smaller 5-year period. This highlights the importance of collecting standardized monitoring data over the long-term. It may very well be that vegetation in Nag Marsh is changing in a way similar to Coggeshall Marsh, but our dataset is not long enough to detect it. Only continued vegetation monitoring will allow us to determine whether community composition is actually changing in Nag Marsh.

The loss of high marsh habitats due to encroachment by low marsh *S. alterniflora* has been documented previously and attributed to increased tidal flooding from sea level rise (Donnelly and Bertness 2002) and elevated nitrogen inputs from adjacent human development (Bertness et al. 2002). In our marshes, the loss of high marsh *S. patens* over time is likely due to issues associated with tidal flooding rather than nitrogen inputs from adjacent shoreline development. In our study, we documented a continuing loss of *S. patens* in both marshes even while habitats adjacent to and surrounding both marshes remain protected within the NBNERR and are therefore undeveloped. Further, nitrogen inputs to the marshes from estuarine sources have likely been reduced since Bertness et al. (2002) conducted their study because nitrogen inputs to the Bay from waste-water treatment facilities have decreased during this time (Krumholtz 2012). Instead, the changes we found are likely due to increasing rates of sea level rise as postulated by Donnelly and Bertness (2001). Around the time of their study, the long-term rate of sea level rise from the Newport RI tide gauge was 2.6 mm yr^{-1}; during our study, the long-term rate remained essentially the same (2.7 mm yr^{-1} from 1930-2011) although recent short-term rates are much higher (3.6 mm yr^{-1} over the last 20 years; Boothroyd, unpublished data). Donnelly and Bertness (2001) also predicted overall marsh loss if sea level rise increased dramatically; the increased occurrence and prevalence of die-off/unvegetated panne areas shown with our data suggest that this may now be occurring.

A major difference between the two marshes is that stunted *S. alterniflora* is replacing *S. patens* in Coggeshall Marsh, while bare ground (i.e., unvegetated pannes) is replacing some *S. patens* in Nag Marsh. Inter-marsh variability in temporal vegetation patterns highlights the need for more spatial replication by expanding our long-term vegetation monitoring into additional marshes in the Reserve and elsewhere in Narragansett Bay. It is recommended here that vegetation monitoring of percent cover, height and stem densities continue annually at both Nag and Coggeshall marshes to further build the long-term datasets at both sites. It is further recommended that permanent vegetation transects and plots be established in at least 2-3 additional salt marshes. Specific marshes could be identified from condition scores determined from ongoing RI salt marsh rapid assessments (Cole, unpublished data), with the 2-3 additional marshes spanning the full gradient in marsh condition scores.

3.2 Elevations and Transitions

3.21 Vegetation plots and shallow-water wells.

In 2012, we were not able to collect elevation data from all vegetation monitoring plots or from all hydrology wells using the RTK because of multiple factors. We successfully collected elevation data for all vegetation plots in Nag Marsh but only for plots along 2 of the 3 transects in Coggeshall Marsh; plots on the third transect could not be sampled because multiple vegetation plot stakes were missing at the time of the elevation survey. We did not collect elevation data for any hydrology wells in Nag Marsh in 2012 since Aquatroll water level loggers were deployed in 3 of the wells throughout the summer of 2012; elevations were only collected for 4 of 7 wells in Coggeshall Marsh since 3 could not be located by field staff. This highlights the need to better mark all vegetation monitoring plots and wells so that seasonal field personnel can find them at all times. Nevertheless, when coupled with previous RTK elevation data collected in 2009 and 2010, these additional 2012 data allow us to further build our vegetation plot and hydrology well elevation database (Tables 7-8). In addition, once we derive tidal datums for each marsh from ongoing water level monitoring, these elevation data will allow us to quantify inundation patterns for each of the vegetation monitoring plots. Continued monitoring will also allow us to track how the elevation of each monitoring plot changes over time, which we will then be able to relate back to any changes that we might observe in vegetation cover.

In most cases, there was excellent agreement among individual plot elevations across multiple years (Figs. 6-7), and hydrology well elevations were even more consistent across multiple years (Fig. 8). However, for some vegetation plots, elevations varied dramatically across years (e.g., Coggeshall Marsh plot 2-7, Nag Marsh plot 3-5). This variability suggests that taking only one elevation measurement from the estimated center of a vegetation plot with the RTK is not enough to accurately determine plot elevation. This is further complicated because the actual plot is not permanently marked in the field. Instead, we follow the Roman et al. (2001) protocols and locate the plot by offsetting specific x and y distances from a permanent marker stake. We therefore recommend that the four corners of every vegetation monitoring plot be permanently marked in the field using short sections of PVC stakes to ensure that the exact same 1-m^2 patch of marsh is monitored each sampling season. We further recommend a supplemental project to determine the number of within-plot replicates that is needed to accurately estimate plot elevation. This could be accomplished by collecting a large number of within-plot replicates (e.g., 12 or more) in multiple plots and using simple sample size estimators. By coupling these additional elevation data with vegetation composition data from the same plots, we might also be able to determine if within-plot elevation sample size differs depending on the number and types of vegetation present (e.g., in a plot with only *S. alterniflora* versus a plot with multiple habitat or vegetation types).

3.22 Elevation profiles

We successfully collected elevation profiles along each of the 3 permanent monitoring transects in each marsh in 2012, thereby augmenting similar profiles that were collected in 2009 and 2010. These elevation profiles allow us to visualize the geomorphology of the marsh platform from water to upland along each transect (Figs. 9-14), and by collecting profiles over time we can observe changes in marsh surface geomorphology should they occur. From a visual inspection of these profile plots, it is clear that the elevation data collected from Nag Marsh transects 2 and 3 in 2010 are consistently high relative to the 2009 and 2012 data. These data are clearly not accurate and at this point we have not been able to determine the source of this error (we always correct all appropriate elevation data when we improve the accuracy of a benchmark in a given year; we corrected these particular 2010 data, but this was not the root cause of the error). We therefore did not use any 2010 Nag Marsh profile data in any further analyses (although the northing and easting data appear accurate from visual inspections; we therefore used these data for habitat transition analyses; see below).

We also used elevation profile data to quantify mean elevation across the length of each transect and by tracking how this changes over time we can derive estimates of marsh accretion rates. With this approach, we documented a mean increase (i.e., accretion rate) of 3.81 mm yr^{-1} across all habitats and transects, although this varied widely (-5.0 to 9.3 mm yr^{-1} range among transects). These estimates should be interpreted cautiously for multiple reasons. First, only 2 years of accurate data were available from each site (Nag Marsh 2009 and 2012; Coggeshall Marsh 2010 and 2012). In addition, these estimates integrate across all habitats that are present along the transects, although we can tease apart elevation for specific habitat types by combining the profile and transition data (see below). Regardless, these data provide an initial estimate of marsh accretion rates that agree well with other estimates in the region (Bricker-Urso et al. 1989; Donnelly and Bertness 2001; Carey, unpublished data), and correlate well with recent local rates of sea level rise (3.6 mm yr^{-1}, Boothroyd, unpublished data). This suggests that using elevation profiles generated with an RTK could be an effective tool for tracking changes in marsh elevation over time.

The variability in marsh accretion rates observed among transects further indicates that changes in elevation might vary not only among different marshes but also among different areas within the same marsh. This further illustrates the need for a large enough sample size in elevation monitoring within a given marsh to account for this variability. For this reason, it may be more effective and more efficient to use RTK elevation profiles to track changes in marsh platform elevation over time than establishing an expensive network of SETs (assuming one has access to an RTK and has established vertical control at the study site). The continued long-term monitoring of transect elevation profiles and SETs at NBNERR will eventually allow us to compare marsh accretion estimates based on these two techniques.

3.23 Habitat transitions

We successfully mapped the locations of all habitat transitions along all 6 monitoring transects in the two marshes in both 2010 and 2012 (as stated previously, 2010 elevation data from Nag Marsh transects 1 and 2 appear in error; however, we are not using these data in this analysis). We identified 16 different habitats in the two marshes, but when combined across marshes and years, the overwhelmingly dominant habitats were *S. alterniflora*, lower salt meadow, and *S. alterniflora*/lower salt meadow (Table 9). The amounts of some habitats changed markedly between 2010 and 2012, but the changes were not always consistent between the two marshes. For example, we observed an 85% increase in *S. alterniflora* habitat over two years in Coggeshall Marsh and a concomitant 49% decrease in lower salt meadow habitat. In contrast, these habitat changes were dampened at Nag Marsh, where *S. alterniflora* habitat increased only slightly (16%) and lower salt meadow habitat decreased much less dramatically (-15%). Instead, the most dramatic change in Nag Marsh across the two years was a complete loss of upper brackish meadow habitat and its concomitant replacement by upper and lower salt meadow. The most conspicuous change that was consistent between the two marshes was a dramatic increase in unvegetated panne habitats in 2012; combined across marshes, this habitat increased in extent from 0.8 m in 2010 to 24.4 m in 2012 (an increase of 2963%). Other noteworthy changes occurred only in Coggeshall Marsh, including an increase in recovering pannes (i.e., pannes with colonizing *Salicornia* spp. and stunted *S. alterniflora*) and the first occurrence of grazed creek banks, with the latter presumably due to overgrazing by the herbivorous nocturnal marsh crab *Sesarma reticulata* (Coverdale et al. 2012).

Based on ANOSIM, overall habitat community composition differed significantly between the two marshes (Global R=0.278; p=0.04), but not between 2010 and 2012 when combined across both marshes (Global R=-0.056; p=0.6). Based on SIMPER, both marshes were typified in descending order by *S. alterniflora*, lower salt meadow, and *S. alterniflora*/lower salt meadow habitats, while the dissimilarity between marshes was primarily due to more *I. frutescens* at Coggeshall Marsh, and more upper salt meadow, creek/ditch, and panne habitats at Nag Marsh (Table 10). The lack of a significant change in habitats between 2010 and 2012 is likely due to the small sample size (N=3 transects per marsh each year) across a short time-frame (two years), and to the fact that some major habitat changes differed in degree between the marshes (e.g., *S. alterniflora* habitat increased dramatically at Coggeshall Marsh, but increased only slightly at Nag Marsh).

Since all of our habitat transition data are georeferenced to specific locations relative to the beginning of each transect, we were also able to visualize and quantify any changes in habitat at every location along each transect (Figs. 15-20). For example, we can clearly see the change from *S. alterniflora* to panne habitat at distances of 35-48 m from the beginning of Nag Marsh transect 2 (Fig. 16). Similarly, we can also see the dramatic increase in habitat heterogeneity and fragmentation along almost the entire length of Coggeshall Marsh transect 1 (Fig. 18). After pooling all habitat transition data from the 6 transects, we documented changes in habitat type along 250 m of the entire 653 m (i.e., 38%) combined length of the transects (Table 11). Within these 250 m of changes, the dominant habitat changes were from *S. alterniflora*/lower salt meadow to pure *S. alterniflora* (35% of all habitat changes), from lower salt meadow to *S. alterniflora*/lower salt meadow (26%), and from *S. alterniflora* to panne (6%). Also noteworthy was the shift from upper brackish meadow to both upper salt meadow and lower salt meadow in Nag Marsh in 2012, possibly a result of more saline conditions from increasing tide levels.

Another way to consider patterns in habitat change is by quantifying all of the ways in which a given habitat changed (or did not change) from 2010 to 2012. For example, of all the *S. alterniflora* habitat that was present in both marshes in 2010, 83% remained as *S. alterniflora* in 2012 indicating that this habitat is very stable over short periods of time (Table 12). However, 9% of *S. alterniflora* habitat was converted into panne during this same time period. Lower salt meadow was less stable as only 59% of this habitat remained unchanged over the 2-year period. Another 31% of lower salt meadow was invaded by *S. alterniflora* and was thus classified as *S. alterniflora*/lower salt meadow habitat in 2012, and yet another 5% of this habitat degraded into panne by 2012. In contrast, *S. alterniflora*/lower salt meadow habitat was clearly a transitional phase as 66 m of lower salt meadow was converted into this habitat and a separate 87 m of *S. alterniflora*/lower salt meadow was converted into pure *S. alterniflora* habitat in 2012.

Finally, we found that 52% of the pannes that were present in 2012 formed from former *S. alterniflora* in 2010, while the remaining formed from lower salt meadow (37%) and *S. alterniflora*/lower salt meadow (7%).

Combined, these results from habitat transition monitoring demonstrate that ongoing changes in marsh habitat composition are not necessarily the same among individual marshes. This further suggests that our habitat transition monitoring should be expanded to additional marshes and conducted over a longer period of time to more comprehensively document broad patterns in salt marsh habitat change in Narragansett Bay (a similar approach to habitat transition monitoring at additional marshes in Narragansett Bay is currently ongoing as part of the RI salt marsh assessment project [Cole, unpublished data]). The data produced from this study clearly demonstrate that this approach would be a relatively rapid and cost-effective way to quantify short- and long-term changes in salt marsh habitat composition. However, we recommend some minor methodological changes moving forward. For example, we recommend modifying the habitat definitions so that we can differentiate between functional ditches (e.g., unvegetated with flowing water) and those that have been colonized by *S. alterniflora*. We combined both types of ditches into the single creek/ditch habitat category, which prevented us from including any *S. alterniflora* that was growing in former ditches in habitat elevation analyses (see below). Even so, for the purposes of this report, very few *S. alterniflora*-colonized ditches were encountered and the results presented here would not have changed much. We also recommend marking off the beginning and end of every transect with permanent markers or stakes in the field to ensure that the exact same length of transect is monitored each year.

3.24 Profile/Transition combinations

We combined the profile and transition data from 2012 to quantify the mean elevation and elevation range of each habitat type (Fig. 21). These data show a generally predictable pattern in habitats across the elevation gradient, ranging from creeks at the lowest elevation to the *I. frutescens* zone at the upper marsh border. These data also show that habitats with *S. alterniflora* as at least a co-dominant occur across an elevation range of approximately 76 cm (from -0.02 m to 0.74 m above NAVD88). This indicates that under current conditions, *S. alterniflora* is able to occupy a very broad range of elevation on the marsh surface platform, enabling it to occupy virtually any area of the marsh including the lower portions of the *I. frutescens* zone. In contrast, pannes were only found within a very narrow elevation range (from 0.51 m to 0.69 m). This elevation range virtually mirrors the upper portions of the pure *S. alterniflora* and *S. alterniflora*/lower salt meadow zones and the lower portion of the lower salt meadow zone. This, coupled with the fact that 52% of new pannes present in 2012 formed from previously pure *S. alterniflora* habitat and a further 37% formed from lower salt meadow, suggests that pannes are forming at the elevation overlap between *S. alterniflora* and lower salt meadow habitats.

3.3 Surface Elevation Tables

We successfully installed all SETs and marker horizons in each marsh without any issues. In total, 6 SETs and feldspar plots were established in each marsh as proposed. Descriptive data for each of the SET locations are provided in Table 13. Initial readings of all SETs were taken in the fall of 2012 and provide a baseline to which future measurements can be compared. Currently, we intend to measure all SETs annually during the fall season beginning in 2013 and measure feldspar plots after a few years. Finally, all 12 SETs were successfully tied into the local vertical network and to NAVD88 using the NERR Leica digital leveling equipment. These elevation data for each SET are also provided in Table 13.

3.4 Summary and Conclusions

During this project, we were able to continue emergent vegetation monitoring at our two marshes for two additional years. We were also able to conduct additional elevation mapping using RTK and digital leveling equipment and establish 12 SET/feldspar plots and tie them into local and national vertical reference networks. Since all of this work was part of the original proposal, this project can be considered a full success.

The major findings of this study are:

- Based on point-intercept sampling in Coggeshall Marsh, percent cover of *S. alterniflora*, *J. gerardii*, and bare ground all increased significantly over time, while *S. patens* cover decreased significantly;
- *Spartina patens* cover also decreased significantly over time in Nag Marsh based on point-intercept sampling, but no changes were observed for any other species or cover type;
- If current rates of loss continue, *S. patens* may disappear from both marshes as early as 2018;
- Vegetation community composition in Coggeshall Marsh (from point-intercept sampling) changed significantly over time, due primarily to an increase in *S. alterniflora* and a concurrent decrease in *S. patens*;
- Vegetation community composition did not change over time in Nag Marsh, although this may be in part due to a shorter sampling period than in Coggeshall Marsh; even so *S. alterniflora* played an increasing role in defining Nag Marsh community similarity over time at the expense of *S. patens*;
- Mean *S. alterniflora* stem height in Coggeshall Marsh did not change significantly from 2000-2012;
- Elevation profiles along each transect from RTK mapping can be used to visualize changes in marsh geomorphology over time;

- Elevation profiles are also potentially useful for estimating changes in marsh accretion rates; we calculated a mean increase in marsh surface elevation of 3.8 mm yr^{-1} from our profile data;
- Graphical displays of habitat transition data are an excellent way to visualize changes in habitat composition over time at specific locations along monitoring transects;
- Based on habitat transition monitoring, the extent of *S. alterniflora* habitat increased while lower salt meadow extent decreased in both marshes between 2010 and 2012; panne habitat increased by 2963% during this time;
- We documented dominant changes in habitat in direct contrast to the classic salt marsh development pathway; in our marshes lower salt meadow habitat largely converted to *S. alterniflora*/lower salt meadow, which in turn largely converted into pure *S. alterniflora*; portions of each of these habitats ultimately converted into unvegetated pannes;
- *Spartina alterniflora* was able to occupy a 0.76-m elevation range that extends from the lowest levels of the marsh surface platform up to the lower portion of the *I. frutescens* marsh-terrestrial border;
- Unvegetated marsh pannes were found in a narrow elevation range that stretched from the upper elevations of stunted *S. alterniflora* monostands to the lower elevations of the *S. patens/D. spicata* lower salt meadow zone;
- If these trends continue, NBNERR marshes (and likely marshes throughout Narragansett Bay) will soon be dominated by monostands of stunted *S. alterniflora* interspersed with sometimes extensive unvegetated pannes and die-off areas.
- This project helped the NBNERR achieve full NERR Sentinel Site status and continued Sentinel Site monitoring will help us identify the specific factors that are causing the dramatic changes in vegetation and habitats documented in this study.

Our major recommendations from this study are:

- Continue point-intercept vegetation monitoring in Nag and Coggeshall marshes annually;
- Continue monitoring stem densities and heights of target species using the Roman et al. (2002) approach where the heights of up to 12 random individuals per quadrat are measured;
- Expand emergent vegetation monitoring to at least 2-3 additional marshes;
- Permanently mark the four corners of each vegetation plot with PVC stakes to ensure that the exact same plot is sampled each year;
- Determine how many within-plot replicates are required for deriving accurate estimates of vegetation plot elevation with the RTK;
- Continue long-term elevation profile mapping and compare estimates of marsh accretion rates from this method to estimates from SET monitoring;

- Continue habitat transition mapping over time and expand it into additional marshes to capture broader temporal and spatial patterns in marsh habitat change;
- Modify habitat transition definitions to differentiate between functional ditches and those that are colonized by *S. alterniflora*;
- Mark the beginning and end of every monitoring transect in the field to ensure that the exact same length of each transect is monitored for profiles and transitions each year.

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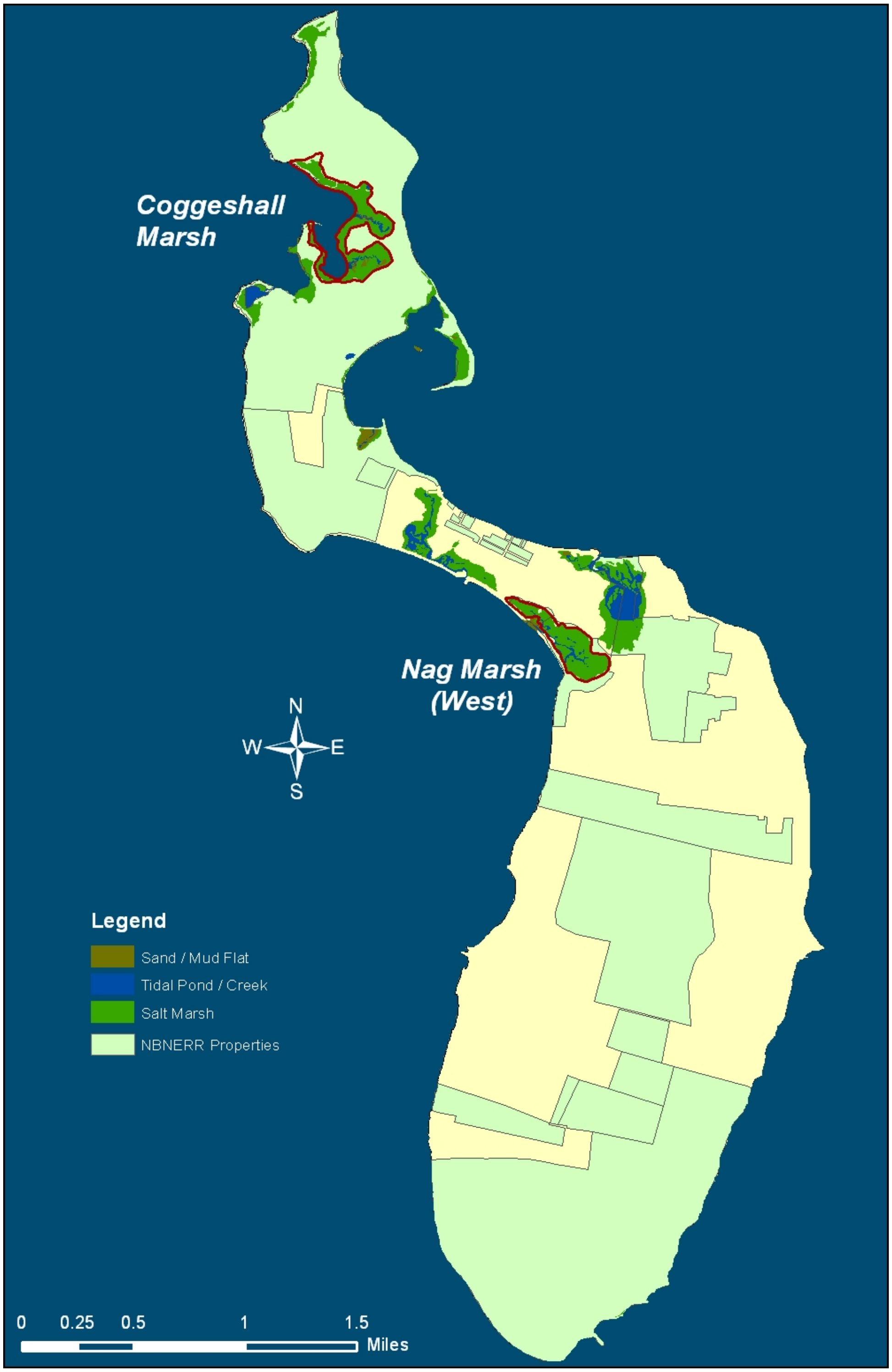


Figure 1. Map showing the location of the Nag and Coggeshall Sentinel Site salt marshes on Prudence Island relative to NBNERR properties.

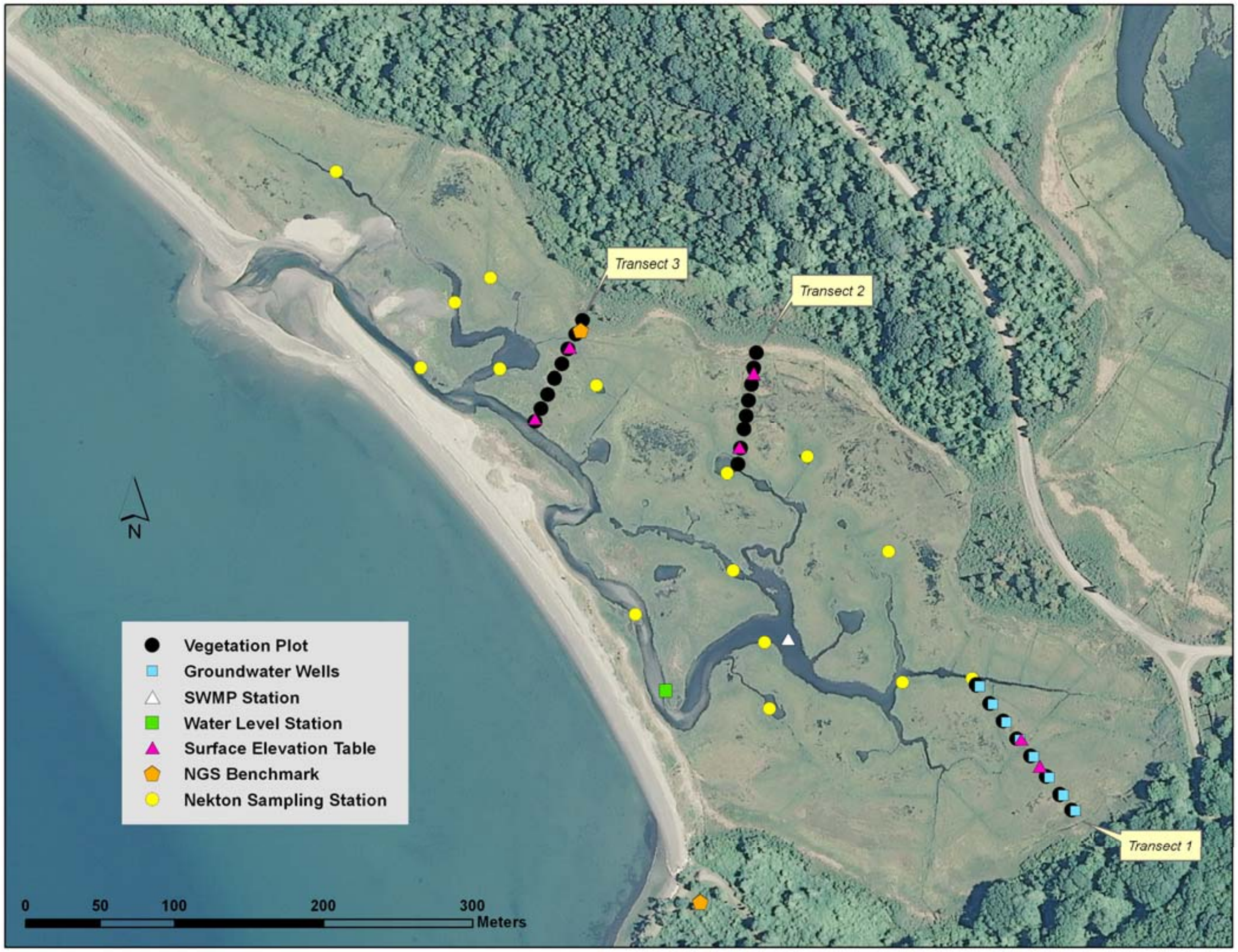


Figure 2. Locations of Sentinel Site and other monitoring infrastructure in Nag Marsh.

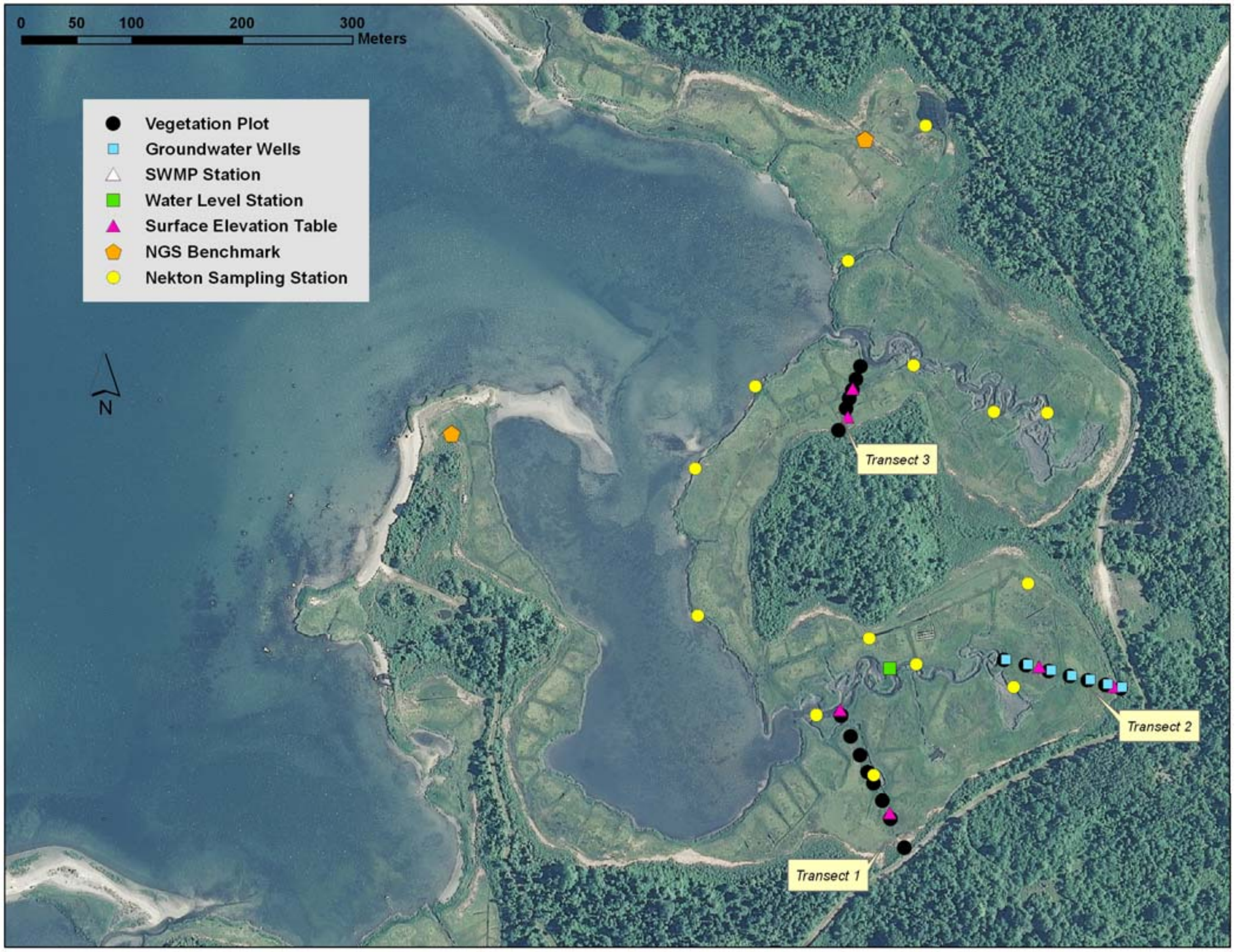
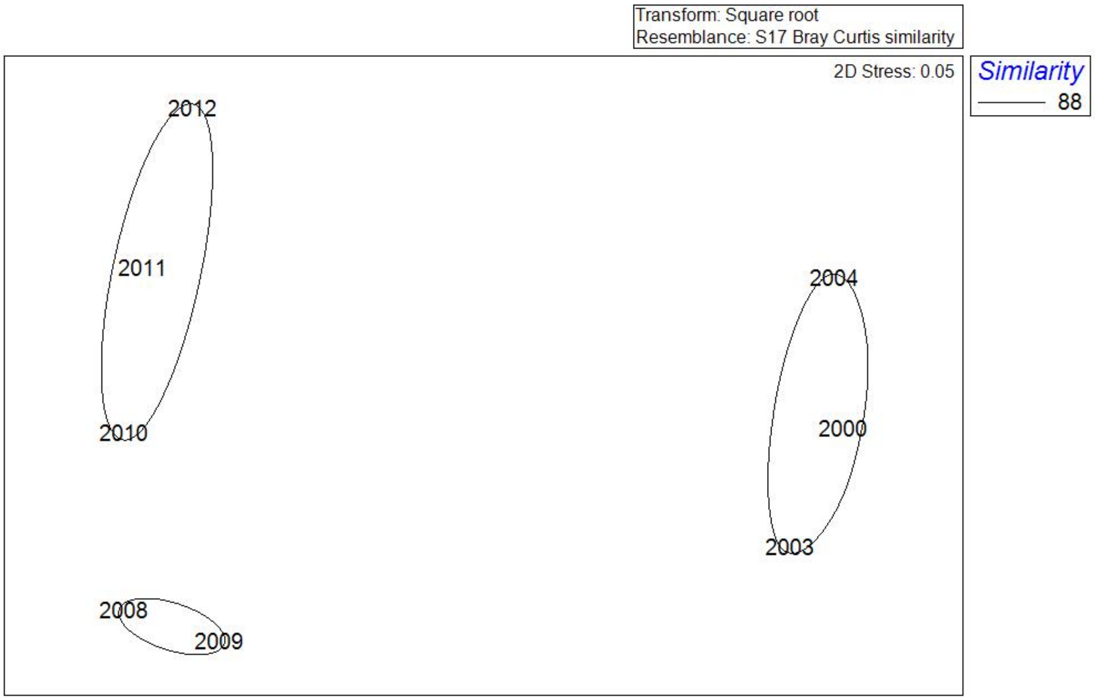


Figure 3. Locations of Sentinel Site and other monitoring infrastructure in Coggeshall Marsh.



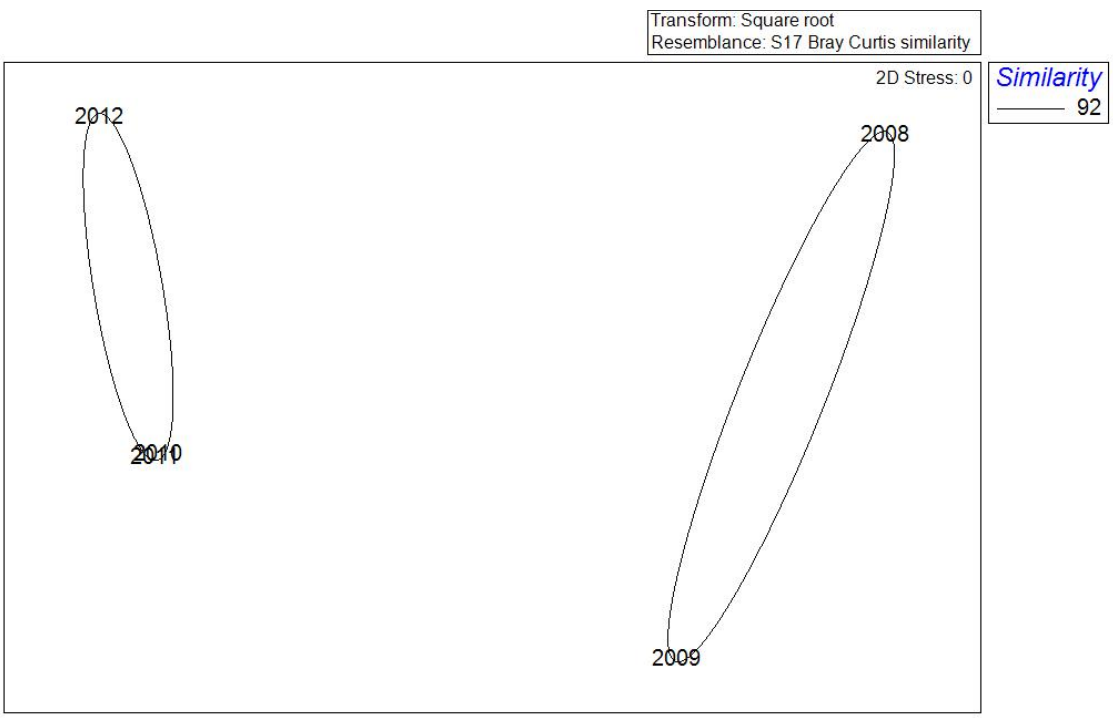
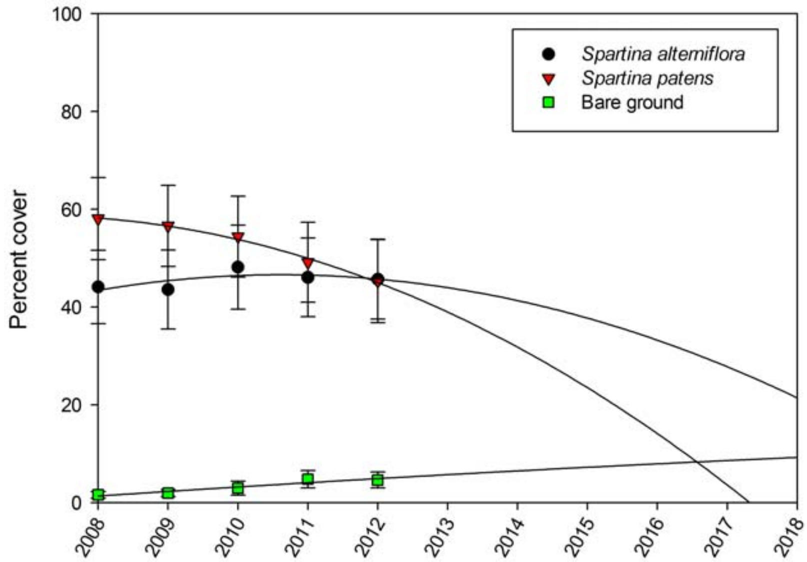


Figure 4. MDS ordinations of monitoring years in Coggeshall (top) and Nag (bottom) marshes based on square-root transformed data and Bray-Curtis similarities. Groups identified with cluster analysis at the 88% (Coggeshall) and 92% (Nag) similarity levels are also shown.



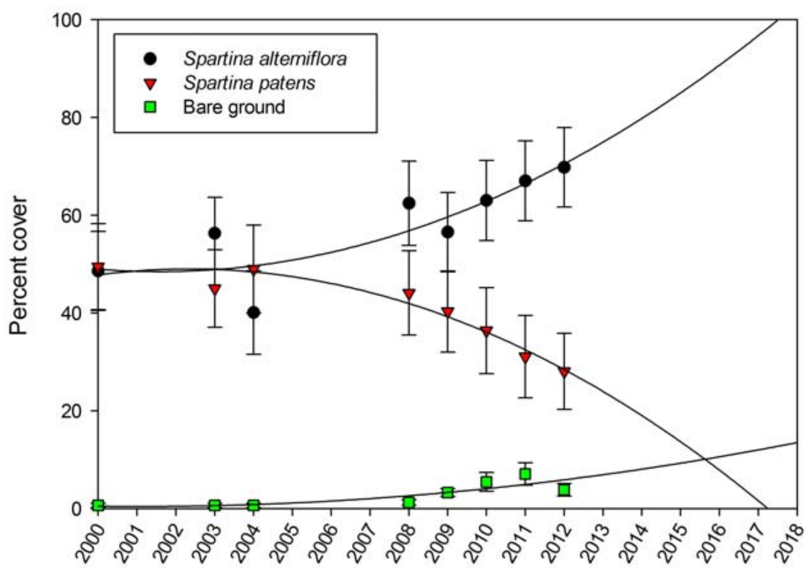


Figure 5. Changes in percent cover of *S. alterniflora*, *S. patens* and bare ground over time in Nag (top) and Coggeshall (bottom) marshes. Error bars are 1 SE and curves were fit with best-fit regression.

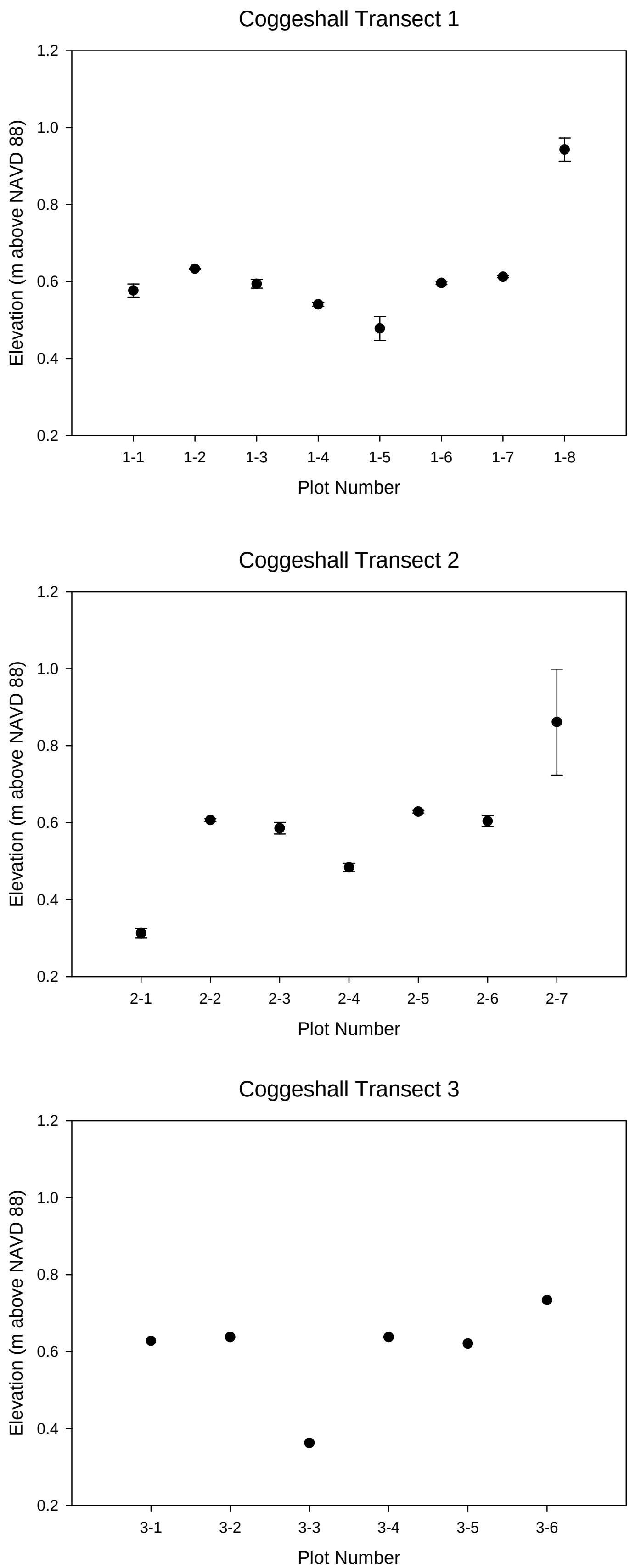


Figure 6. Elevations of vegetation plots in Coggeshall Marsh. Error bars are 1 SE.

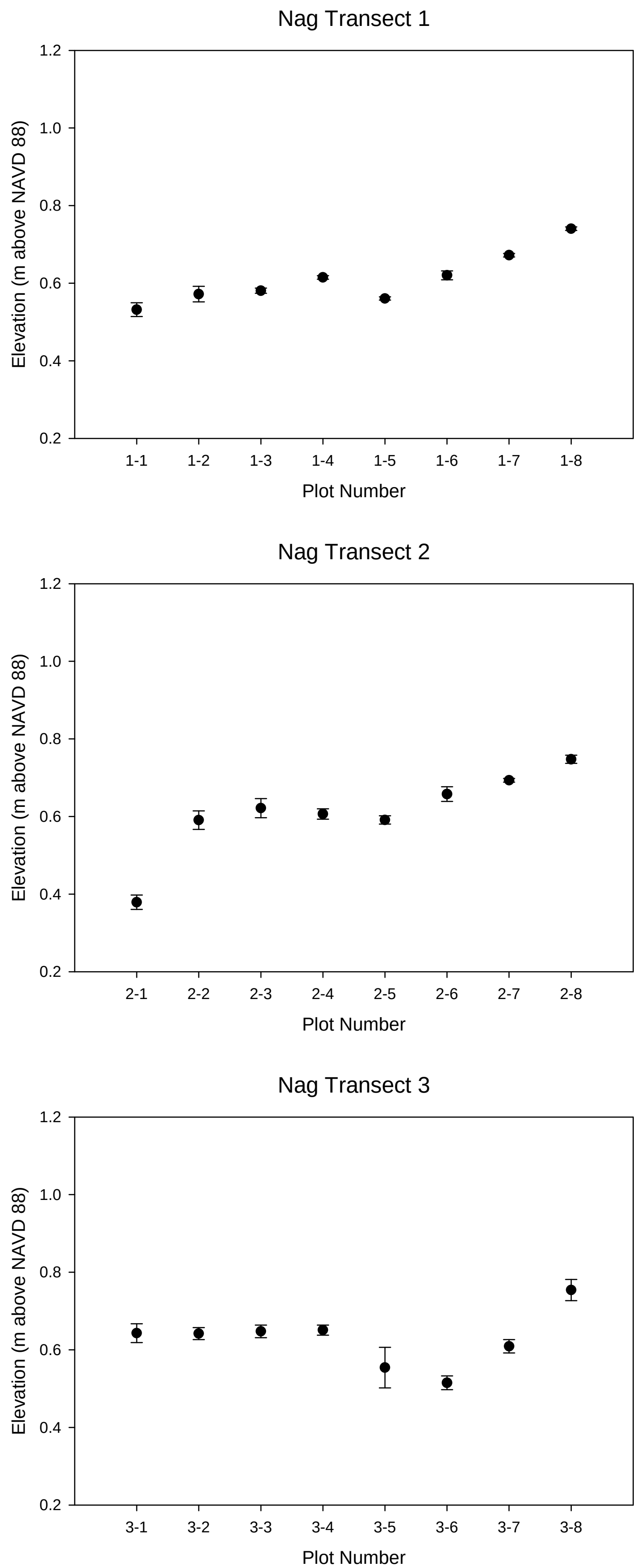
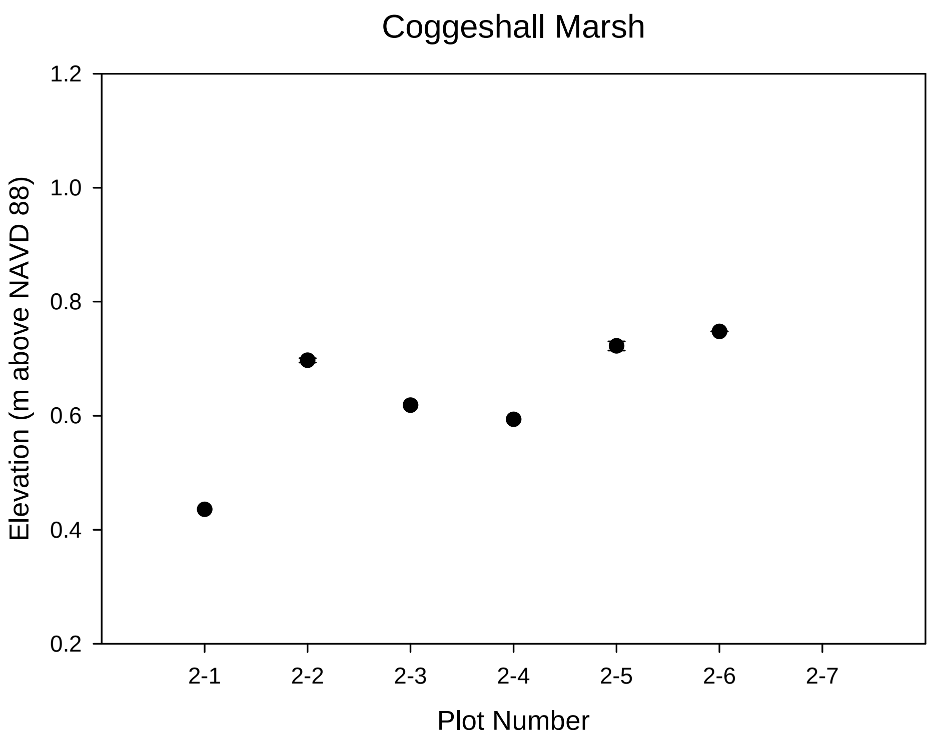


Figure 7. Elevations of vegetation plots in Nag Marsh. Error bars are 1 SE.



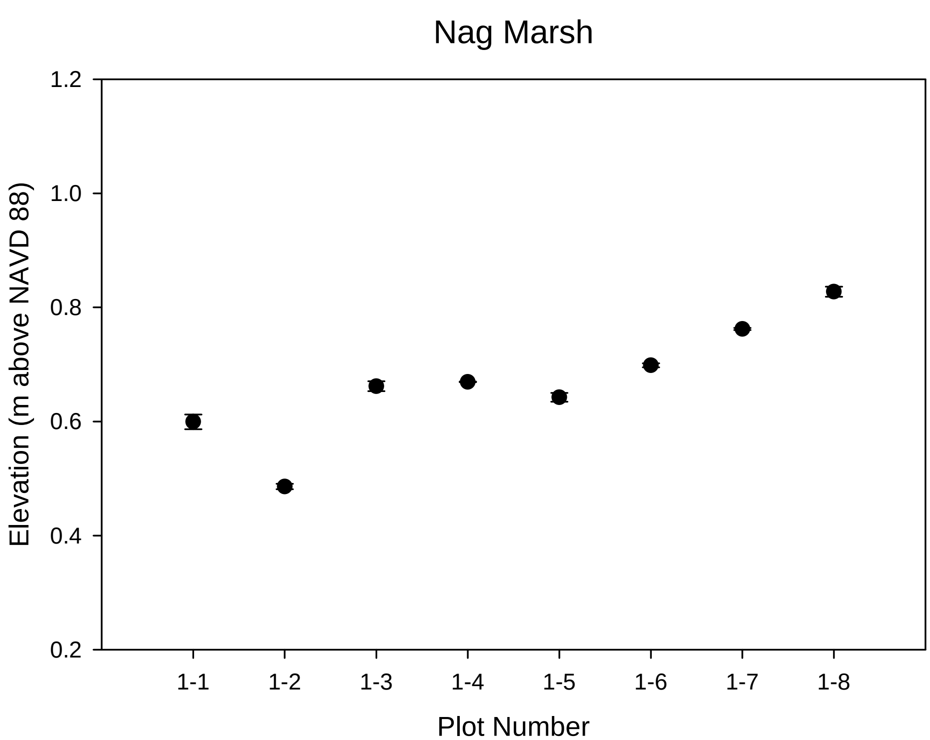


Figure 8. Elevations of shallow PVC hydrology wells in Coggeshall and Nag marshes. Error bars are 1 SE.

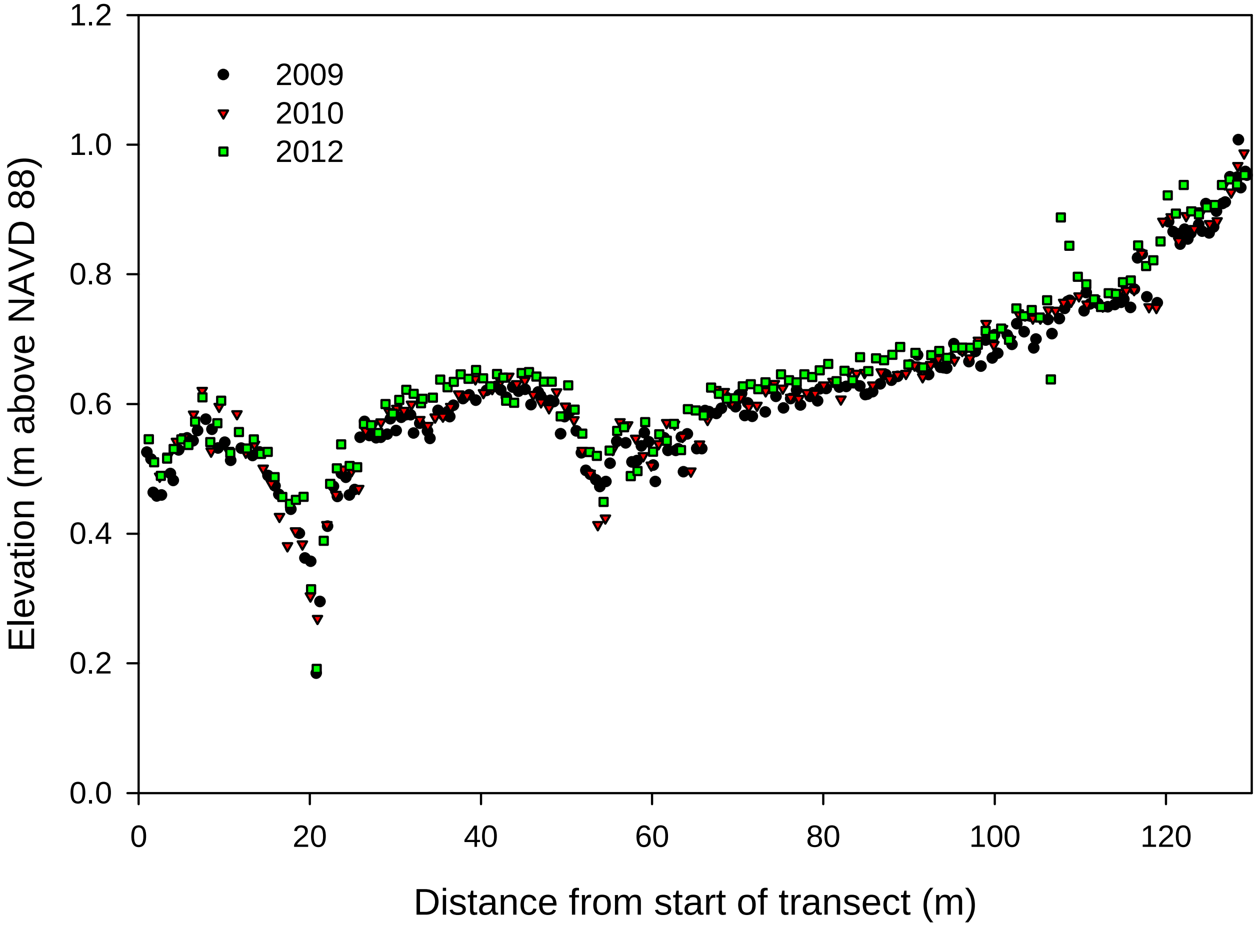


Figure 9. Multi-year elevation profiles along transect 1 in Nag Marsh.

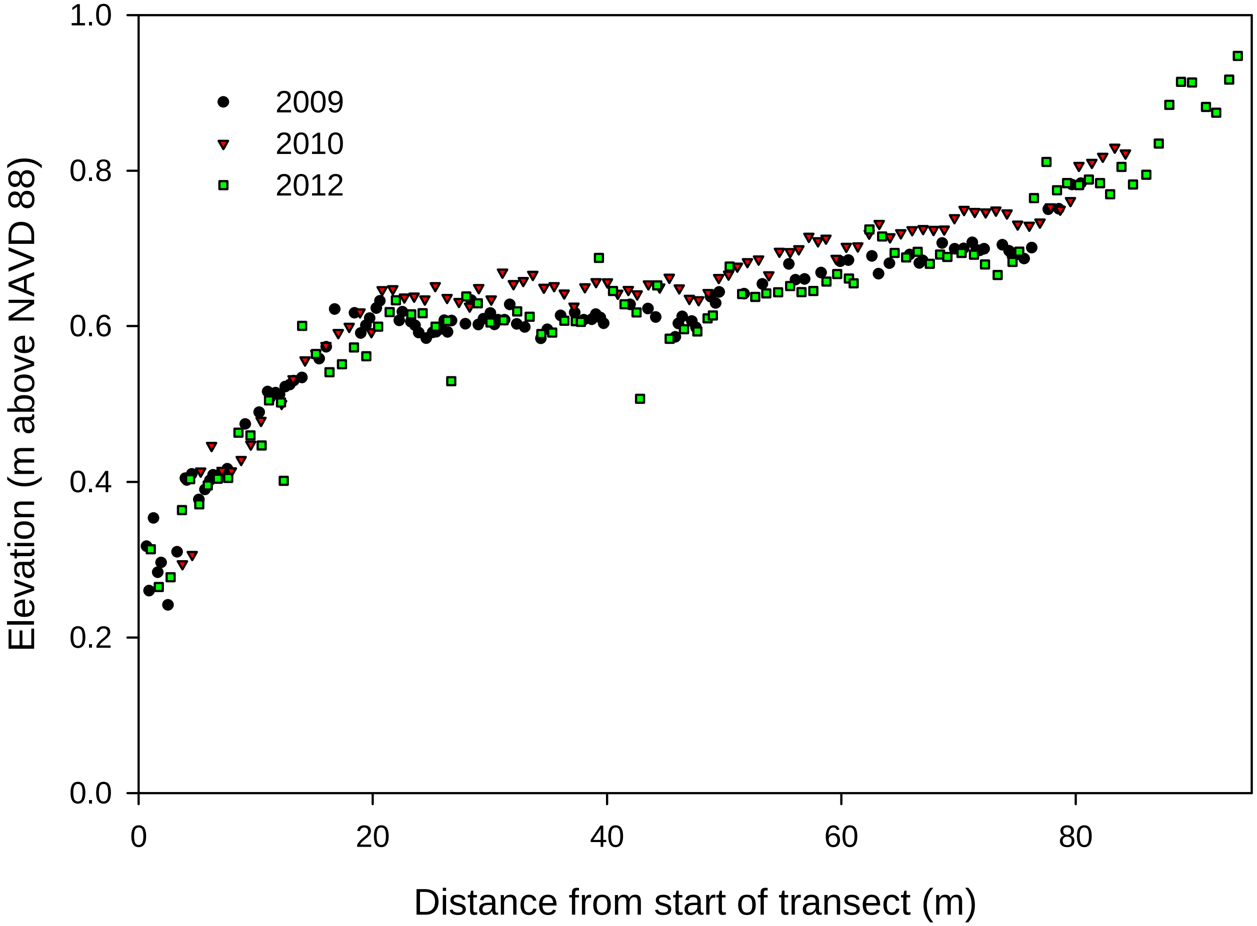


Figure 10. Multi-year elevation profiles along Transect 2 in Nag Marsh. Elevation data from 2010 are likely inaccurate (see text on page 13).

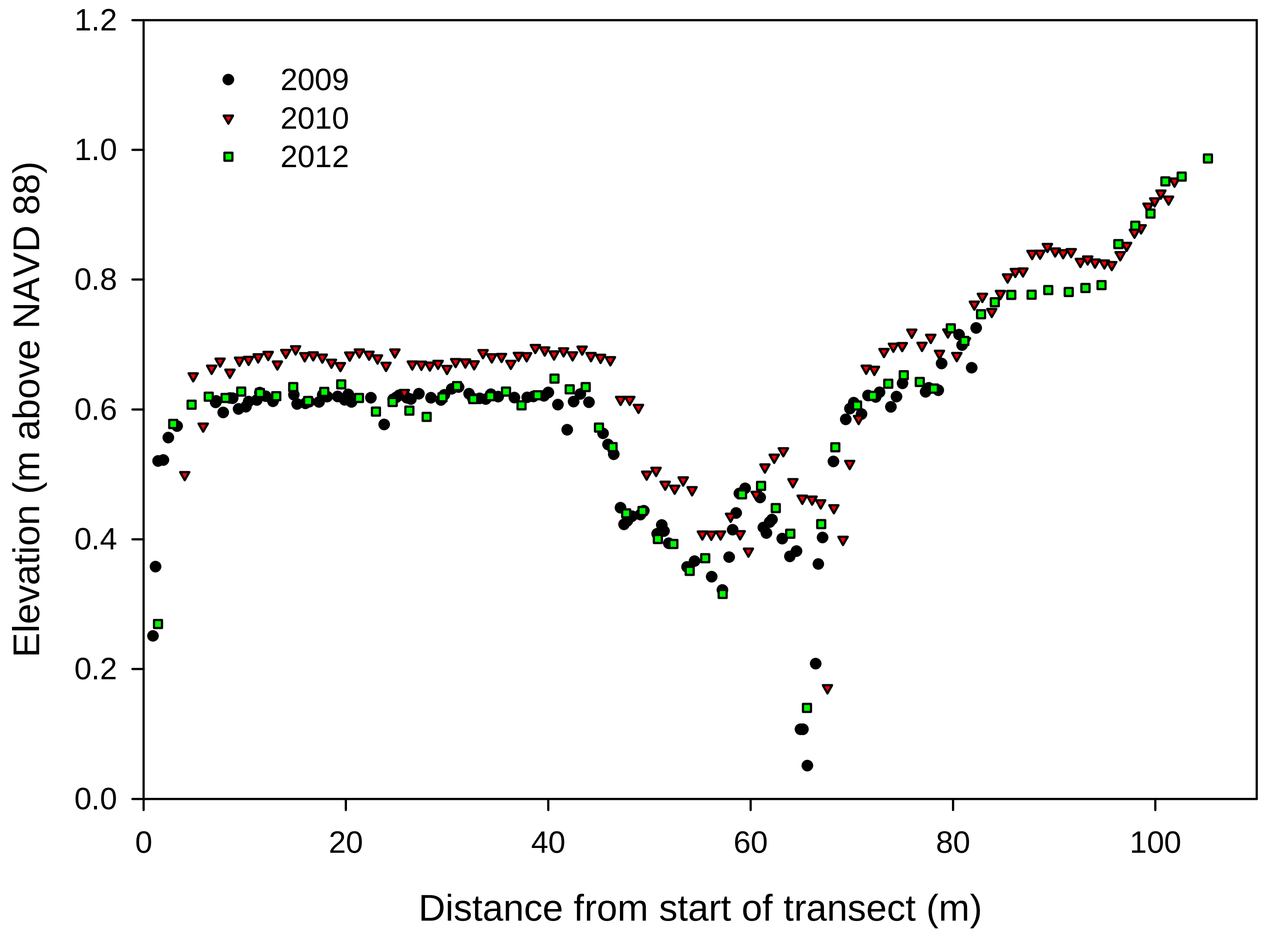


Figure 11. Multi-year elevation profiles along Transect 3 in Nag Marsh. Elevation data from 2010 are likely inaccurate (see text on page 13).

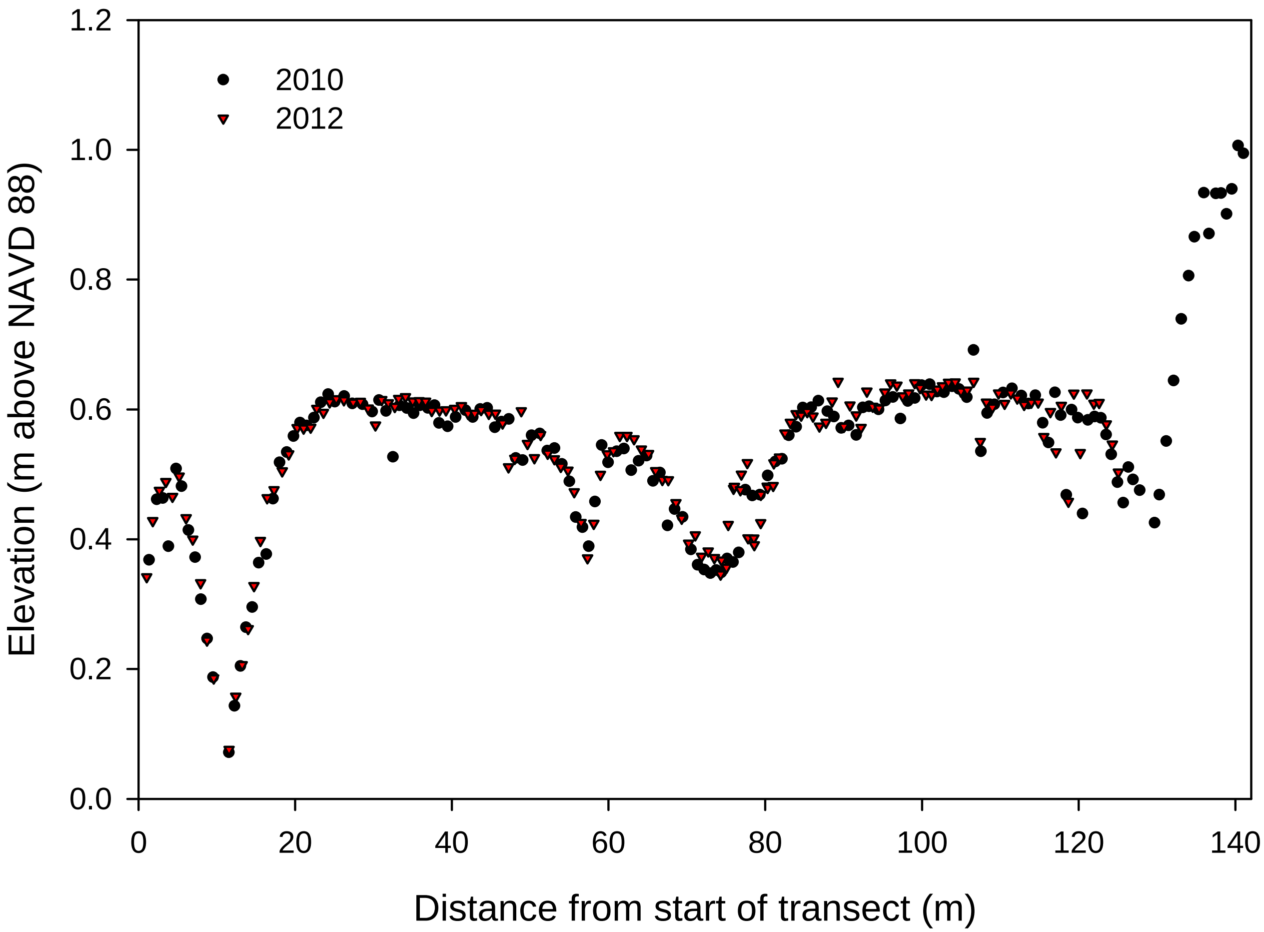


Figure 12. Multi-year elevation profiles along transect 1 in Coggeshall Marsh.

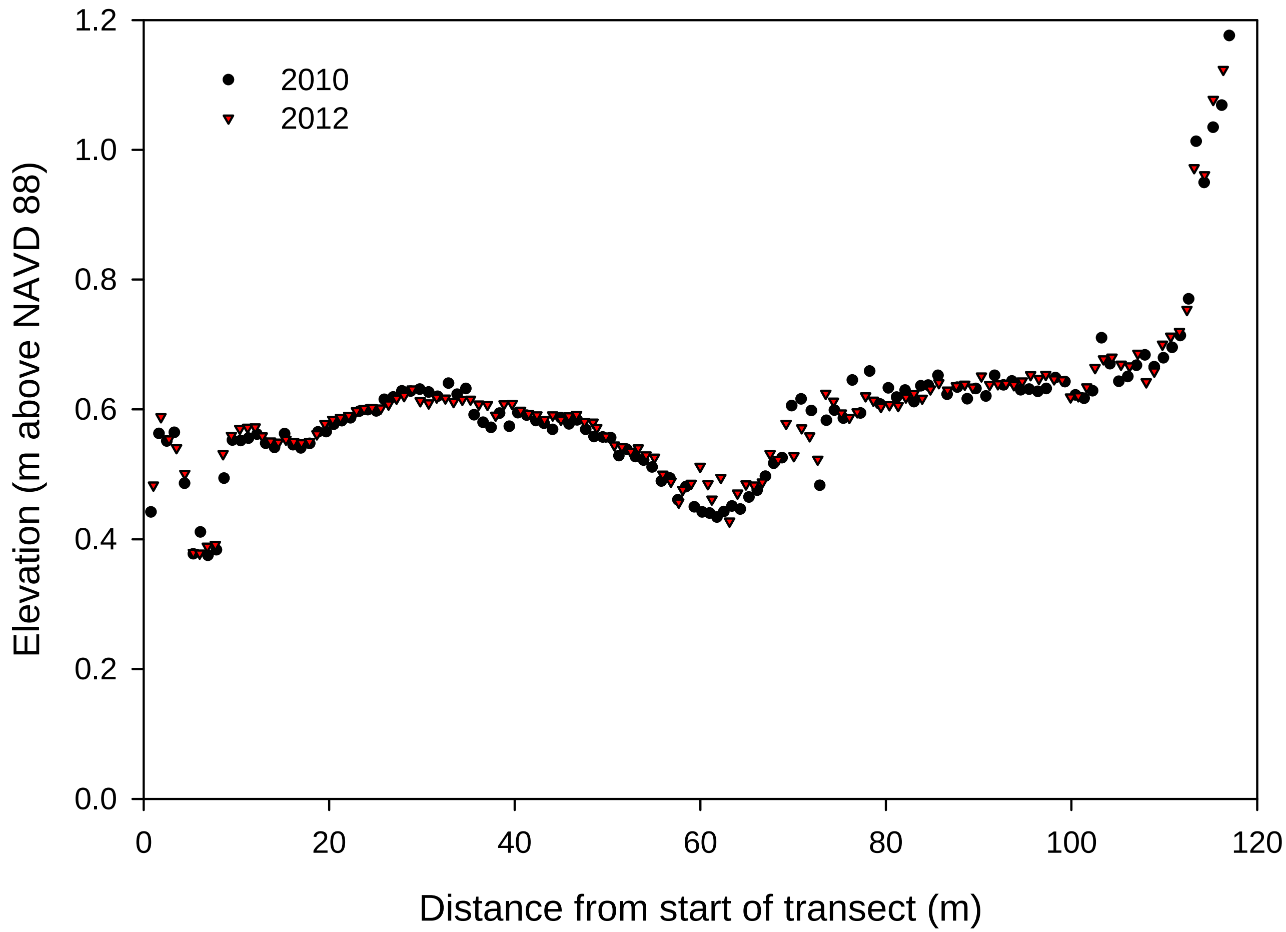


Figure 13. Multi-year elevation profiles along transect 2 in Coggeshall Marsh.

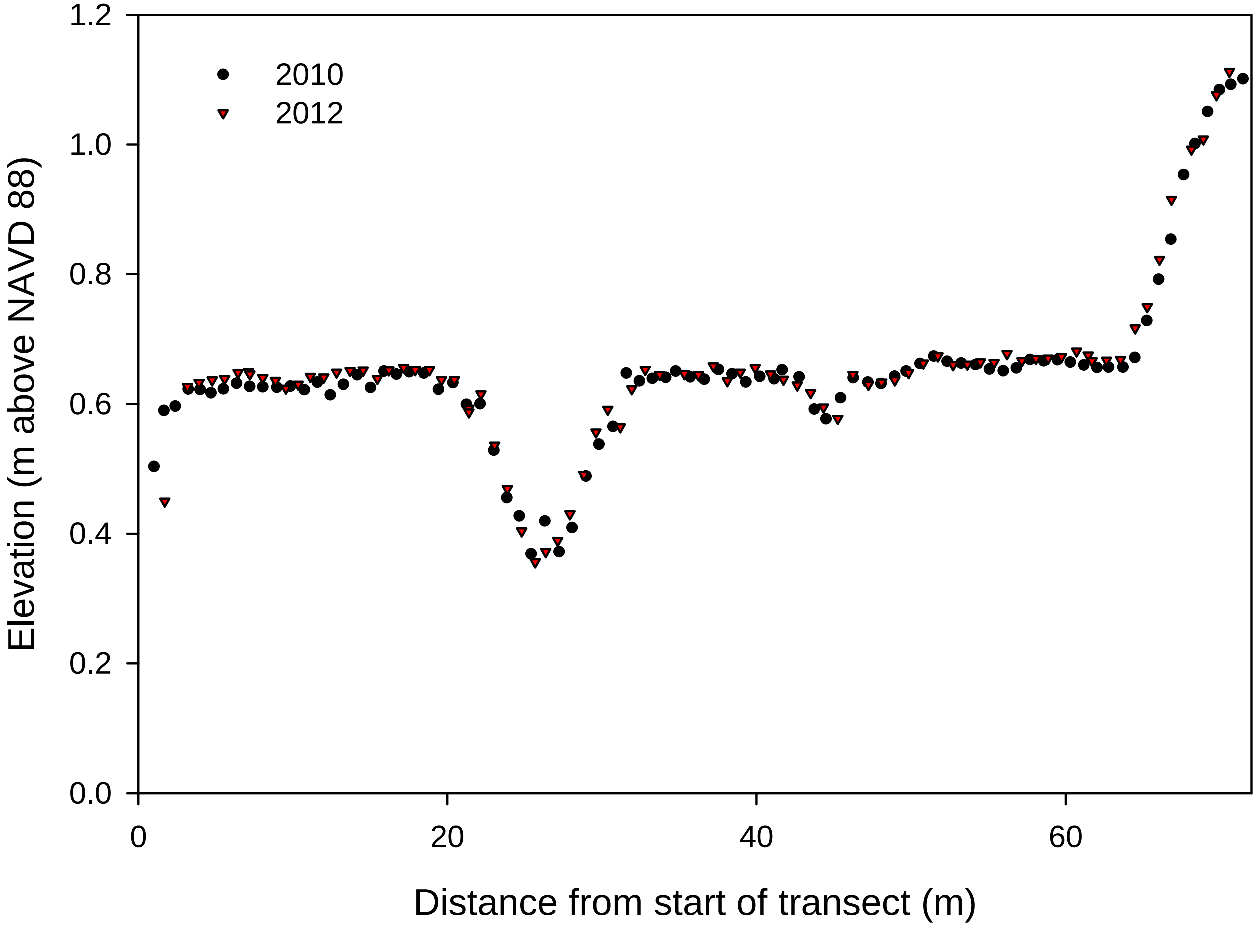


Figure 14. Multi-year elevation profiles along transect 3 in Coggeshall Marsh.

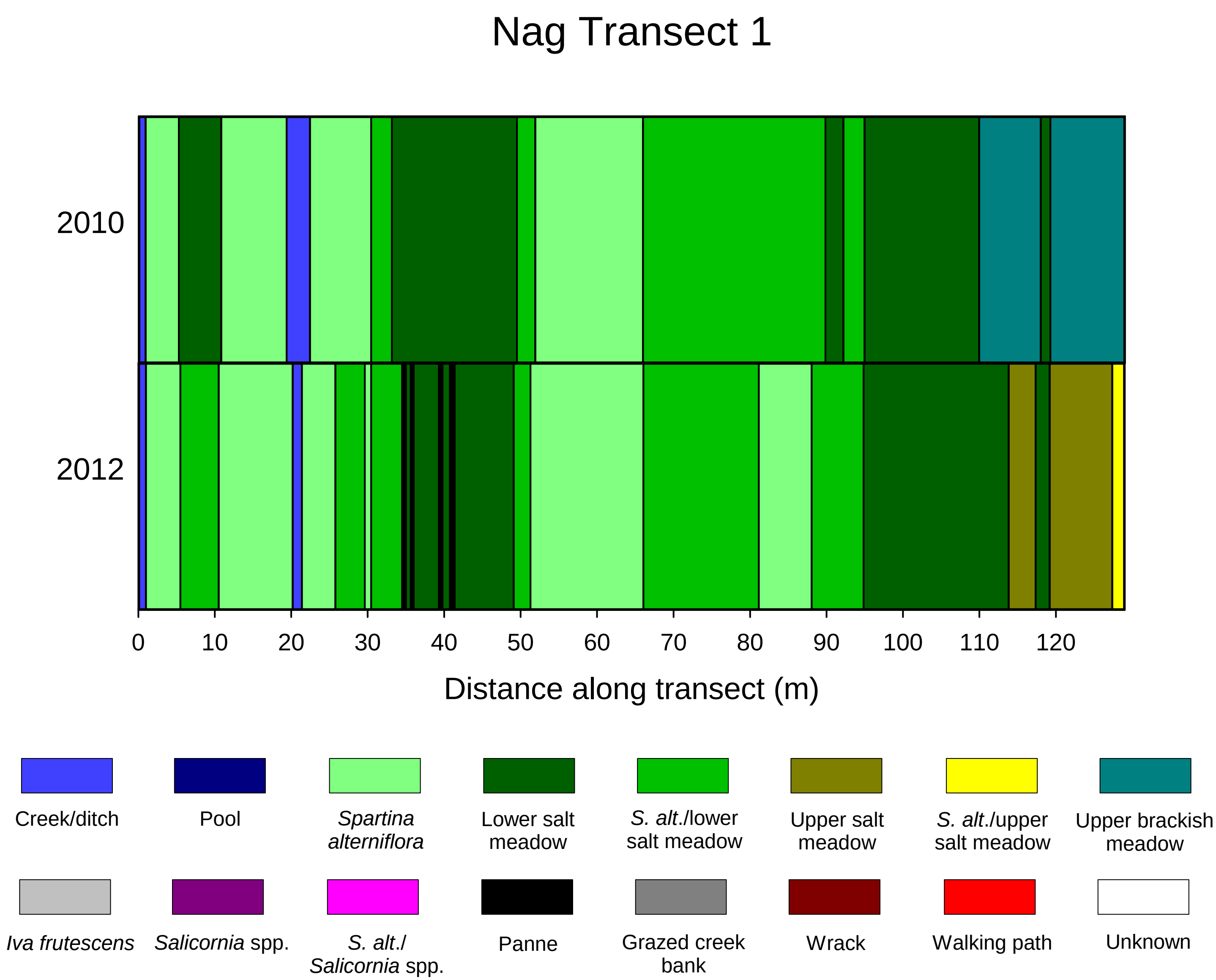


Figure 15. Distribution and extent of salt marsh habitats from habitat transition mapping along transect 1 in Nag Marsh.

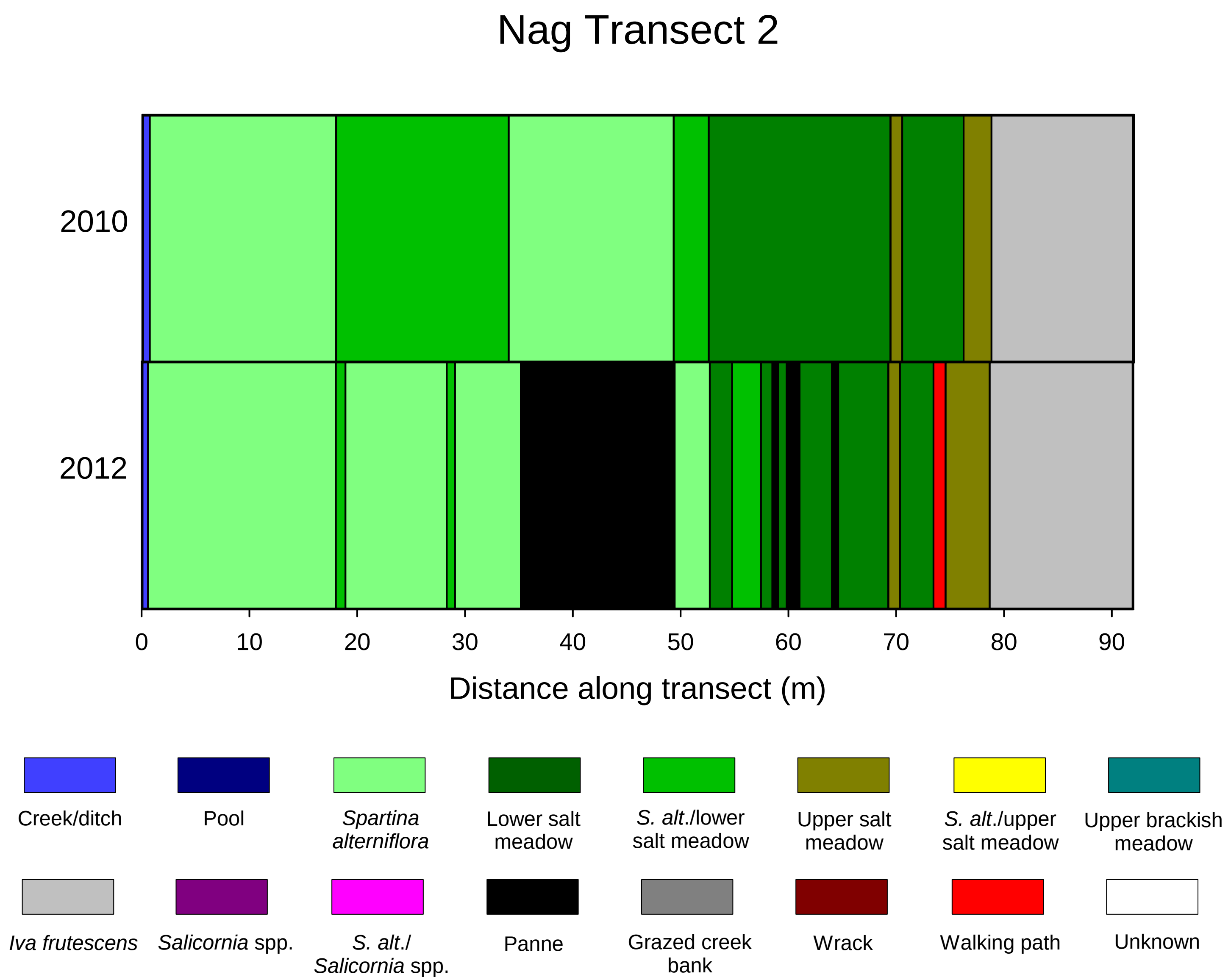


Figure 16. Distribution and extent of salt marsh habitats from habitat transition mapping along transect 2 in Nag Marsh.

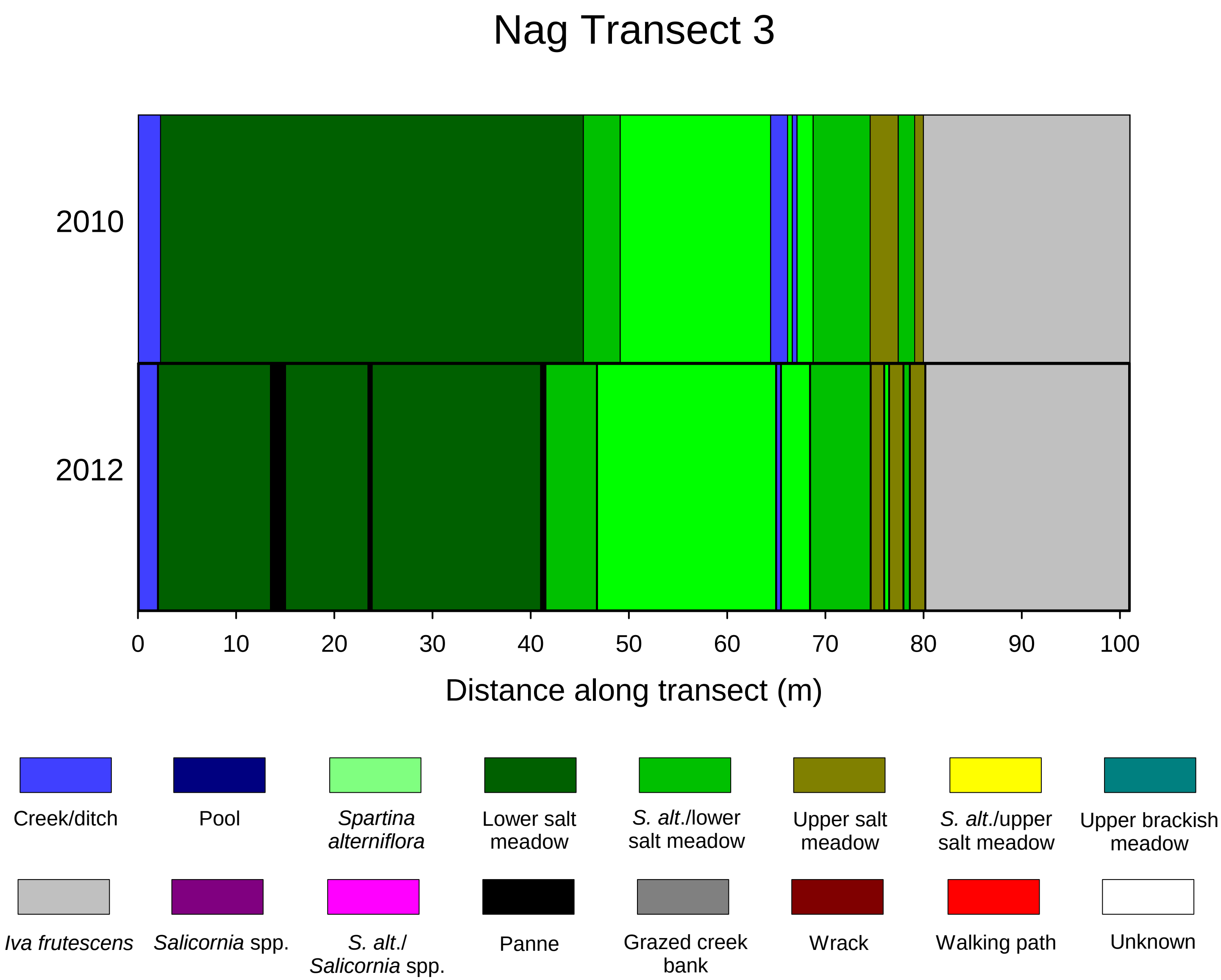


Figure 17. Distribution and extent of salt marsh habitats from habitat transition mapping along transect 3 in Nag Marsh.

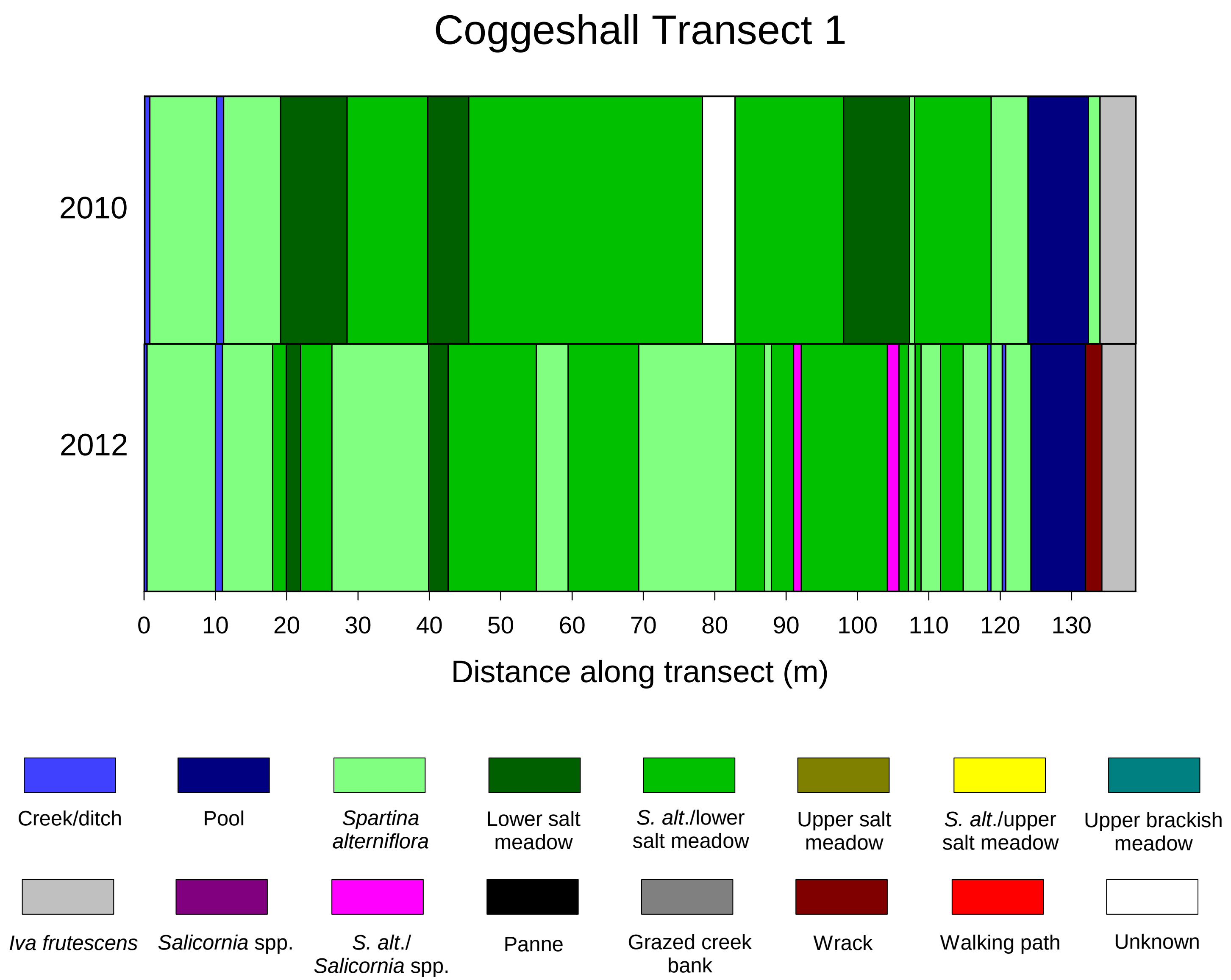


Figure 18. Distribution and extent of salt marsh habitats from habitat transition mapping along transect 1 in Coggeshall Marsh.

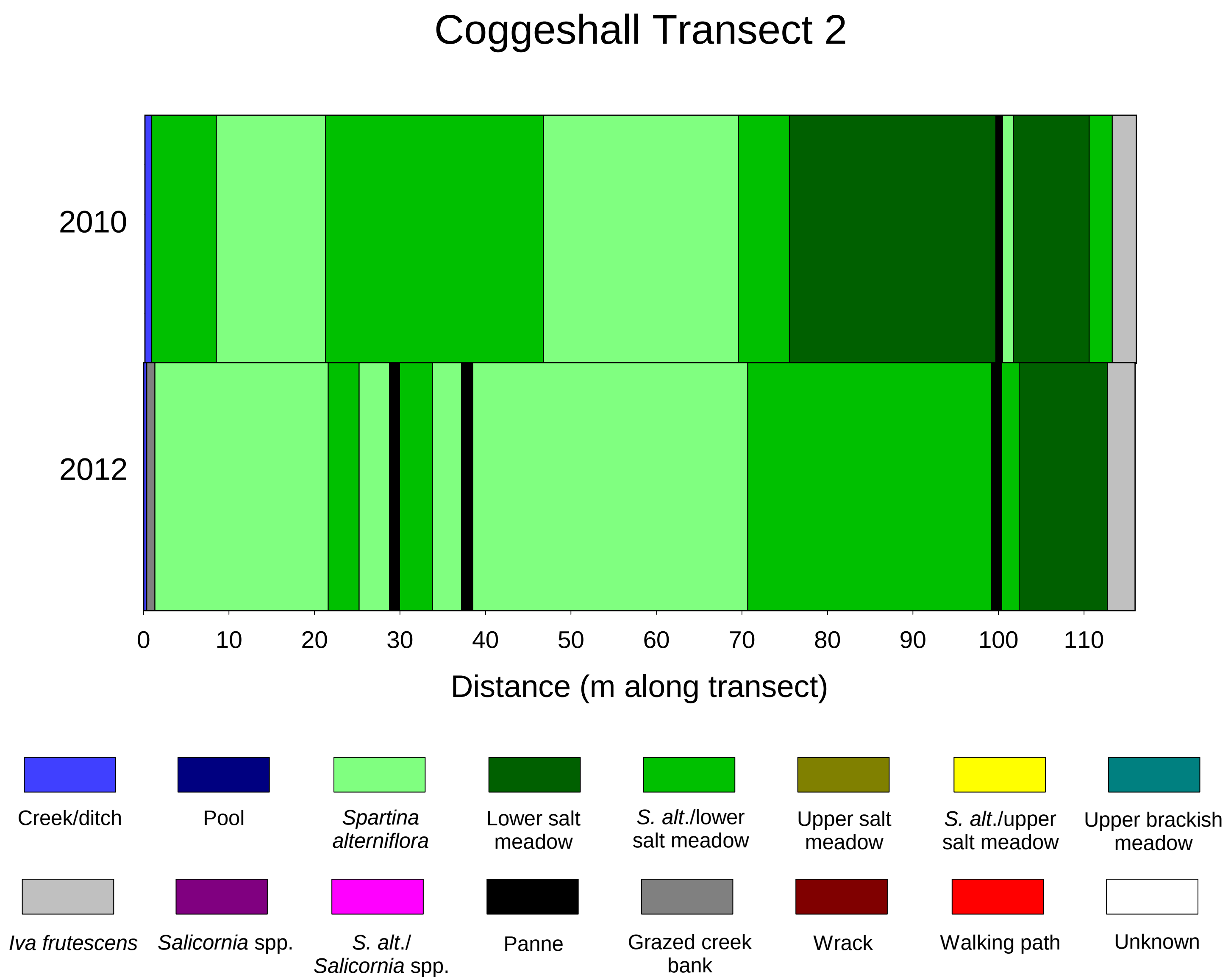


Figure 19. Distribution and extent of salt marsh habitats from habitat transition mapping along transect 2 in Coggeshall Marsh.

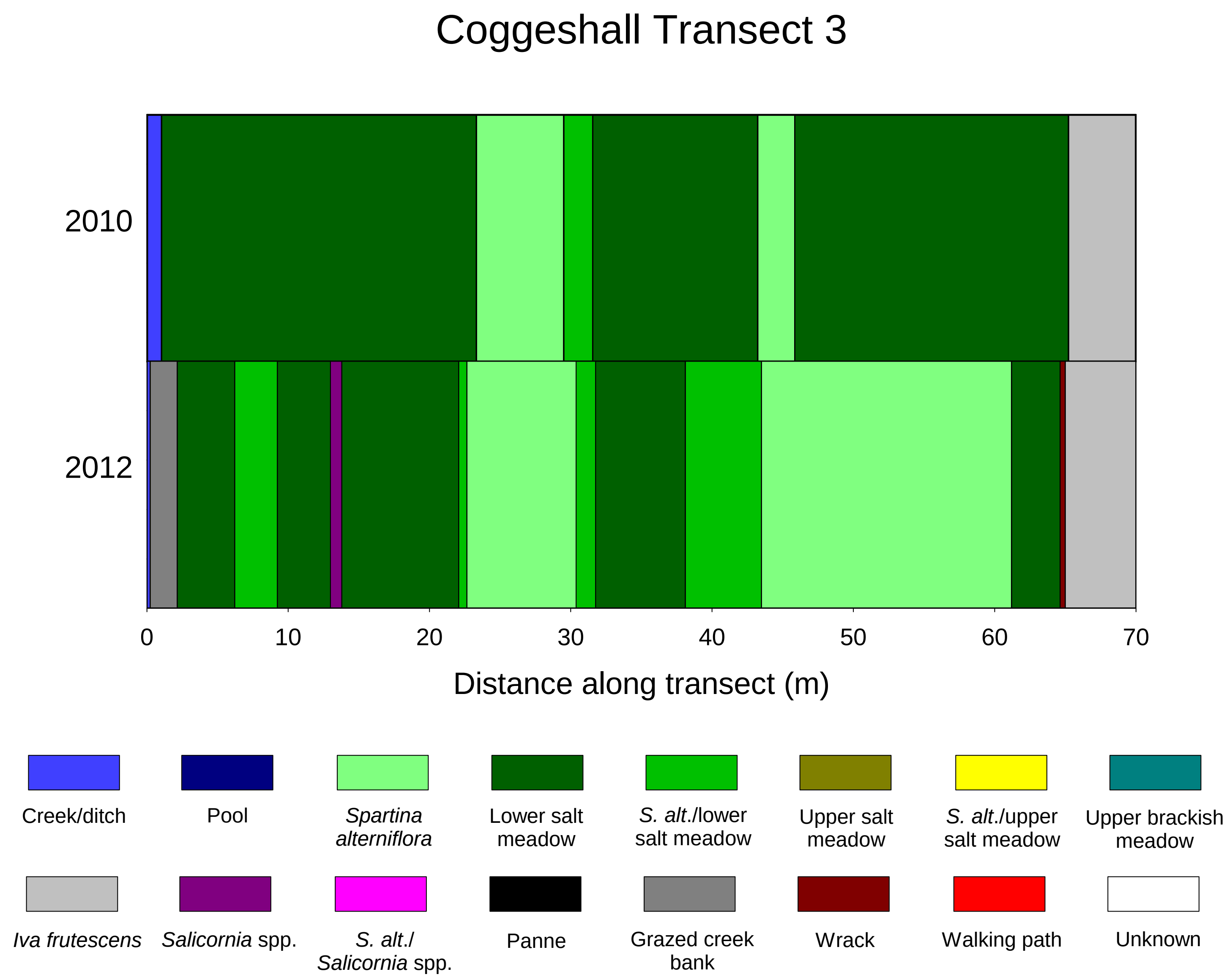


Figure 20. Distribution and extent of salt marsh habitats from habitat transition mapping along transect 3 in Coggeshall Marsh.

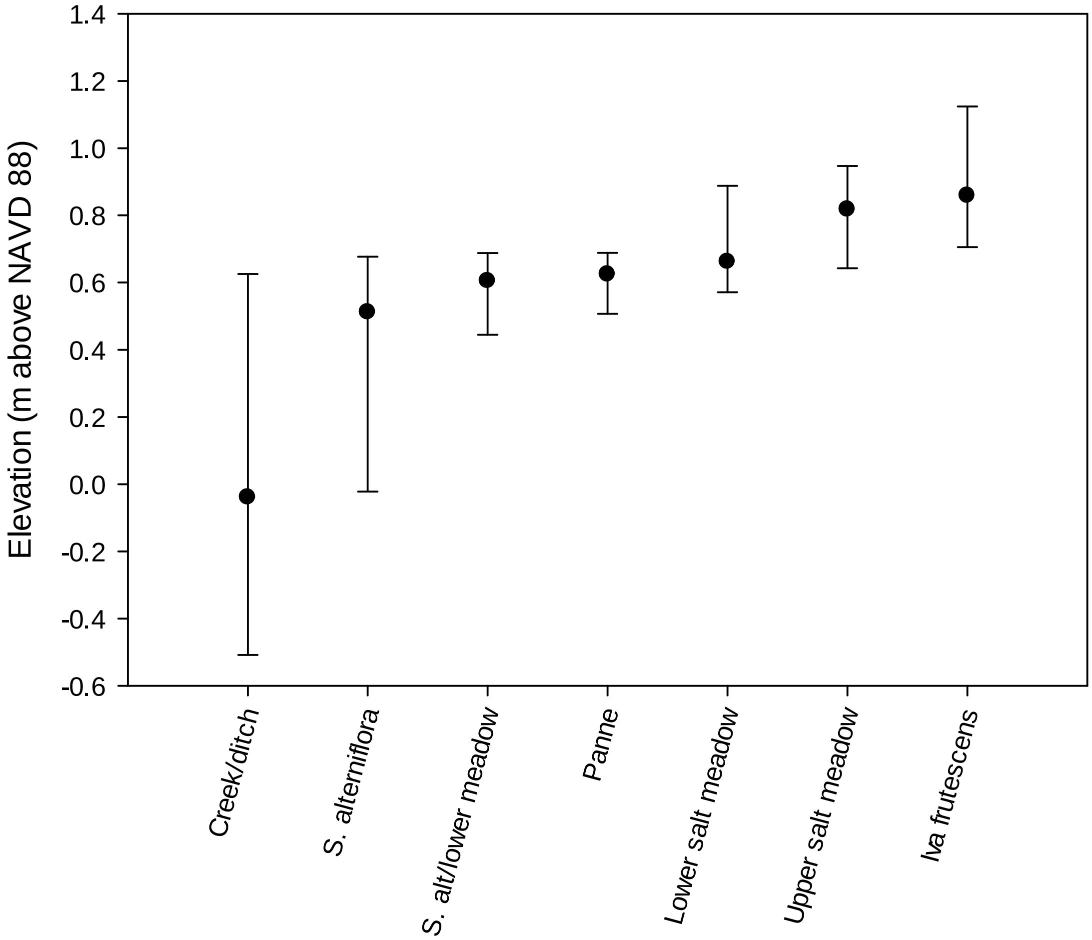


Figure 21. Mean elevations of common habitat types in Nag and Coggeshall marshes. Data are from 2012 only and are derived from profile and habitat transition mapping along 6 monitoring transects. Error bars are maximums and minimums.

Table 1. Percent cover of vegetation species and other cover types in Coggeshall Marsh from 2000-2012. Data for each year are means and standard errors derived from replicated monitoring plots. Asterisks next to a species name indicate significant changes in cover over time based on best-fit regression (linear for *J. gerardii*, non-linear for all others; *=p<0.05; ***=p<0.001). Species are listed in order of decreasing mean cover across all sampling years.

Species	2000	2003	2004	2008	2009	2010	2011	2012
Spartina alterniflora *	48.62 (8.12)	56.33 (7.34)	40.10 (8.58)	62.48 (8.62)	56.57 (8.10)	63.05 (8.20)	67.05 (8.17)	69.81 (8.15)
Spartina patens ***	49.48 (8.81)	45.00 (7.95)	49.00 (9.02)	44.10 (8.63)	40.29 (8.29)	36.38 (8.82)	31.05 (8.42)	28.05 (7.79)
Distichlis spicata	18.00 (6.09)	14.57 (5.44)	10.00 (3.66)	13.62 (3.72)	13.05 (3.63)	19.33 (5.37)	16.33 (5.36)	17.52 (5.41)
Juncus gerardii *	4.57 (4.52)	3.33 (3.33)	5.52 (4.27)	12.48 (6.87)	12.10 (6.65)	12.29 (6.75)	7.90 (4.89)	11.52 (6.40)
Iva frutescens	6.76 (4.37)	7.81 (4.31)	11.14 (6.38)	11.05 (6.15)	7.62 (4.48)	12.38 (6.80)	9.71 (5.82)	10.19 (5.78)
Bare ground *	0.62 (0.42)	0.62 (0.44)	0.62 (0.46)	1.24 (0.58)	3.24 (0.81)	5.43 (1.94)	7.05 (2.30)	3.81 (1.29)
Limonium nashii	0.38 (0.21)	1.10 (0.53)	1.24 (0.61)	4.00 (2.13)	2.10 (0.94)	0.67 (0.25)	0.48 (0.24)	0.95 (0.60)
Salicornia spp.	1.86 (1.06)	3.05 (1.19)	3.81 (1.00)	0.00 (0.00)	0.19 (0.13)	0.48 (0.19)	2.10 (0.61)	8.48 (3.42)
Stone	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	1.71 (1.71)	1.24 (1.24)	1.05 (1.05)	1.05 (0.95)	0.00 (0.00)
Wrack	0.24 (0.24)	0.86 (0.86)	0.00 (0.00)	0.00 (0.00)	1.14 (0.87)	0.29 (0.29)	0.00 (0.00)	0.00 (0.00)
Panicum virgatum	0.43 (0.43)	0.38 (0.38)	0.33 (0.33)	0.00 (0.00)	0.00 (0.00)	0.19 (0.19)	0.00 (0.00)	0.00 (0.00)
Solidago sempervirens	0.33 (0.33)	0.38 (0.38)	0.10 (0.10)	0.10 (0.10)	0.00 (0.00)	0.10 (0.10)	0.10 (0.10)	0.29 (0.29)
Baccharis halimifolia	0.00 (0.00)	0.00 (0.00)	0.43 (0.43)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Unknown grass	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.10 (0.10)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Setaria spp.	0.33 (0.33)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Myrica pennsylvanica	0.00 (0.00)	0.29 (0.29)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Cyperus spp.	0.00 (0.00)	0.05 (0.05)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)

Table 2. Percent cover of vegetation species and other cover types in Nag Marsh from 2008-2012. Data for each year are means and standard errors derived from replicated monitoring plots. Asterisks next to a species name indicate significant changes in cover over time based on nonlinear best-fit regression (**=p<0.01). Species are listed in order of decreasing mean cover across all sampling years.

	2008	2009	2010	2011	2012
Spartina patens **	58.09 (8.41)	56.61 (8.28)	54.42 (8.26)	49.17 (8.20)	45.33 (8.53)
Spartina alterniflora	44.09 (7.52)	43.57 (8.08)	48.17 (8.60)	46.08 (8.07)	45.67 (8.15)
Distichlis spicata	20.18 (5.25)	20.17 (4.82)	23.08 (4.70)	22.00 (5.65)	21.08 (5.48)
Juncus gerardii	5.82 (4.23)	7.57 (5.23)	7.83 (5.43)	7.58 (5.26)	7.75 (5.36)
Iva frutescens	4.18 (3.50)	4.26 (3.53)	3.75 (3.34)	3.25 (3.25)	3.50 (3.50)
Bare ground	1.55 (0.71)	1.91 (0.72)	2.92 (1.45)	4.75 (1.81)	4.58 (1.63)
Schoenoplectus americanus	1.91 (1.91)	3.22 (3.22)	1.25 (1.25)	2.75 (2.75)	0.42 (0.42)
Salicornia spp.	3.55 (1.34)	0.00 (0.00)	0.83 (0.45)	0.50 (0.28)	2.83 (1.41)
Wrack	2.09 (1.68)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Limonium nashii	0.45 (0.45)	0.52 (0.38)	0.00 (0.00)	0.08 (0.08)	0.50 (0.50)
Agalinis maritime	0.45 (0.45)	0.61 (0.61)	0.08 (0.08)	0.00 (0.00)	0.00 (0.00)
Aster tenuifolium	0.00 (0.00)	0.00 (0.00)	0.08 (0.08)	0.08 (0.08)	0.17 (0.12)

Table 3. SIMPER results for comparisons of vegetation community composition among groups of years in Coggeshall Marsh. Top three tables show species contributing to within-group community similarity; bottom table shows species contributing to community dissimilarity between the 2000-2004 and 2010-2012 year groups. Av=average; SD=standard deviation.

2000-2004 (average similarity = 51.35)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative%
Spartina alterniflora	6.02	25.05	1.07	48.79	48.79
Spartina patens	5.61	17.66	0.87	34.4	83.19
Distichlis spicata	2.61	5.5	0.74	10.7	93.9

2008-2009 (average similarity = 51.44)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative%
Spartina alterniflora	6.81	28.28	1.14	54.99	54.99
Spartina patens	5.2	14.55	0.86	28.29	83.28
Distichlis spicata	2.56	4.76	0.6	9.26	92.53

2010-2012 (average similarity = 50.37)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative%
Spartina alterniflora	7.26	31.34	1.2	62.22	62.22
Spartina patens	4.01	8.38	0.63	16.65	78.87
Distichlis spicata	2.85	4.71	0.57	9.35	88.22
Bare ground	1.48	2.94	0.47	5.84	94.06

2000-2004 v 2010-2012 (average dissimilarity = 51.49)

Species	2000-2004 Av.Abundance	2010-2012 Av.Abundance	Av.Dissimilarity	Dissimilarity/SD	Contrib%	Cumulative%
Spartina patens	5.61	4.01	12.82	1.23	24.9	24.9
Spartina alterniflora	6.02	7.26	10.55	1.18	20.49	45.39
Distichlis spicata	2.61	2.85	8.25	1.22	16.03	61.42
Iva frutescens	1.08	1.22	4.65	0.56	9.04	70.46
Bare ground	0.25	1.48	4.33	0.83	8.41	78.87
Salicornia spp.	1.09	1.06	3.77	0.99	7.32	86.19
Juncus gerardii	0.54	1.21	3.66	0.48	7.1	93.29

Table 4. SIMPER results for year-group comparisons of vegetation community composition in Nag Marsh. Both tables show species contributing to within-group community similarity. Av=average; SD=standard deviation.

2008-2009 (average similarity = 51.98)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative%
Spartina patens	6.57	24.39	1.12	46.93	46.93
Spartina alterniflora	5.35	17.59	0.82	33.84	80.77
Distichlis spicata	3.44	8.49	0.84	16.33	97.1

2010-2012 (average similarity = 49.58)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative%
Spartina alterniflora	5.49	19.58	0.79	39.48	39.48
Spartina patens	5.8	18.75	0.91	37.82	77.3
Distichlis spicata	3.56	8.59	0.81	17.32	94.62

Table 5. Heights of target salt marsh vegetation species over time. Data for all target species were collected in 2011 and 2012; additional data were collected for *S. alterniflora* from earlier years as well. Data are means and standard errors.

	Coggeshall				
	2000	2003	2004	2011	2012
Distichlis spicata				35.32 (3.51)	33.05 (2.24)
Juncus gerardii				30.14 (1.06)	21.85 (2.91)
Schoenoplectus americanus					
Spartina alterniflora	40.66 (3.84)	41.32 (3.77)	32.98 (2.45)	36.18 (3.90)	33.76 (3.15)
Spartina patens				27.09 (2.23)	24.94 (1.58)

	Nag				
	2000.00	2003.00	2004.00	2011.00	2012.00
Distichlis spicata				29.78 (1.51)	27.21 (1.26)
Juncus gerardii				38.50 (10.17)	25.56 (3.33)
Schoenoplectus americanus				68.25	61.67
Spartina alterniflora				44.26 (4.16)	34.90 (3.26)
Spartina patens				28.49 (2.06)	26.14 (1.91)

	Combined				
	2000	2003	2004	2011	2012
Distichlis spicata				31.96 (1.70)	29.50 (1.27)
Juncus gerardii				33.48 (3.86)	23.33 (2.12)
Schoenoplectus americanus				68.25	61.67
Spartina alterniflora	40.66 (3.84)	41.32 (3.77)	32.98 (2.45)	40.22 (2.89)	34.38 (2.25)
Spartina patens				27.93 (1.51)	25.65 (1.28)

Table 6. Stem densities of target salt marsh vegetation species from 2010-2012. Data are means per m^2 and standard errors.

	Coggeshall		
	2010	2011	2012
Distichlis spicata	n/a	109.52 (47.26)	138.10 (71.21)
Juncus gerardii	390.48 (256.80)	14.29 (14.29)	33.33 (18.69)
Schoenoplectus americanus	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Spartina alterniflora	1268.95 (264.57)	1146.67 (228.93)	1024.57 (187.90)
Spartina patens	1890.48 (636.92)	1138.10 (393.49)	490.48 (217.44)

	Nag		
	2010	2011	2012
Distichlis spicata	n/a	354.17 (146.58)	233.33 (104.20)
Juncus gerardii	345.83 (243.39)	16.67 (16.67)	0.00 (0.00)
Schoenoplectus americanus	20.83 (20.83)	0.00 (0.00)	0.00 (0.00)
Spartina alterniflora	425.83 (129.94)	1205.00 (703.55)	624.00 (165.11)
Spartina patens	3141.67 (839.58)	1787.50 (667.53)	1916.67 (615.61)

	Combined		
	2010	2011	2012
Distichlis spicata	n/a	240.00 (82.47)	188.89 (64.45)
Juncus gerardii	366.67 (174.70)	15.56 (10.99)	15.56 (8.96)
Schoenoplectus americanus	11.11 (11.11)	0.00 (0.00)	0.00 (0.00)
Spartina alterniflora	819.29 (153.55)	1177.78 (386.18)	810.93 (126.48)
Spartina patens	2557.78 (539.83)	1484.44 (399.32)	1251.11 (356.63)

Table 7. Northing, easting, and elevation (m above NAVD88) data for vegetation monitoring plots in Coggeshall and Nag marshes.

		2009			2010			2012			Means		
Site	Plot #	N	E	elevation	N	E	elevation	N	E	elevation	N	E	elevation
Coggeshall	1-1	n/a	n/a	n/a	62992.76	113077.33	0.59	62992.49	113077.22	0.56	62992.62	113077.28	0.58
Coggeshall	1-2	n/a	n/a	n/a	62974.30	113085.81	0.63	62974.32	113085.64	0.63	62974.31	113085.73	0.63
Coggeshall	1-3	n/a	n/a	n/a	62956.88	113094.16	0.58	62957.01	113093.60	0.61	62956.94	113093.88	0.59
Coggeshall	1-4	n/a	n/a	n/a	62941.46	113101.02	0.54	62941.64	113100.89	0.55	62941.55	113100.96	0.54
Coggeshall	1-5	n/a	n/a	n/a	62931.36	113106.53	0.45	62930.06	113106.32	0.51	62930.71	113106.43	0.48
Coggeshall	1-6	n/a	n/a	n/a	62915.64	113114.09	0.60	62915.81	113113.81	0.59	62915.73	113113.95	0.60
Coggeshall	1-7	n/a	n/a	n/a	62899.14	113121.58	0.61	62899.33	113121.24	0.61	62899.24	113121.41	0.61
Coggeshall	1-8	n/a	n/a	n/a	62873.23	113134.24	0.91	62873.09	113133.70	0.97	62873.16	113133.97	0.94
Coggeshall	2-1	n/a	n/a	n/a	63043.15	113224.74	0.30	63042.74	113225.00	0.32	63042.95	113224.87	0.31
Coggeshall	2-2	n/a	n/a	n/a	63038.18	113244.66	0.60	63037.92	113244.46	0.61	63038.05	113244.56	0.61
Coggeshall	2-3	n/a	n/a	n/a	63033.05	113265.56	0.57	63032.96	113265.44	0.60	63033.01	113265.50	0.59
Coggeshall	2-4	n/a	n/a	n/a	63028.40	113284.39	0.47	63028.46	113284.67	0.49	63028.43	113284.53	0.48
Coggeshall	2-5	n/a	n/a	n/a	63024.39	113301.01	0.63	63024.31	113301.18	0.63	63024.35	113301.09	0.63
Coggeshall	2-6	n/a	n/a	n/a	63020.56	113316.86	0.59	63020.38	113316.89	0.62	63020.47	113316.88	0.60
Coggeshall	2-7	n/a	n/a	n/a	63017.37	113330.47	1.00	63018.64	113328.04	0.72	63018.00	113329.25	0.86
Coggeshall	3-1	n/a	n/a	n/a	63309.64	113094.63	0.63	n/a	n/a	n/a	63309.64	113094.63	0.63
Coggeshall	3-2	n/a	n/a	n/a	63297.54	113090.63	0.64	n/a	n/a	n/a	63297.54	113090.63	0.64
Coggeshall	3-3	n/a	n/a	n/a	63287.98	113087.36	0.36	n/a	n/a	n/a	63287.98	113087.36	0.36
Coggeshall	3-4	n/a	n/a	n/a	63280.65	113084.58	0.64	n/a	n/a	n/a	63280.65	113084.58	0.64
Coggeshall	3-5	n/a	n/a	n/a	63271.76	113081.63	0.62	n/a	n/a	n/a	63271.76	113081.63	0.62
Coggeshall	3-6	n/a	n/a	n/a	63251.68	113074.77	0.73	n/a	n/a	n/a	63251.68	113074.77	0.73
Nag	1-1	60121.89	114769.35	0.53	60121.66	114769.51	0.56	60122.06	114769.01	0.50	60121.87	114769.29	0.53
Nag	1-2	60109.72	114778.50	0.53	60109.73	114778.12	0.58	60109.82	114778.15	0.60	60109.76	114778.26	0.57
Nag	1-3	60097.77	114787.25	0.58	60097.45	114787.29	0.59	60097.83	114787.20	0.57	60097.69	114787.25	0.58
Nag	1-4	60085.82	114796.46	0.61	60085.75	114795.95	0.62	60085.90	114796.20	0.61	60085.82	114796.20	0.61
Nag	1-5	60074.14	114805.75	0.55	60074.38	114805.19	0.56	60074.10	114805.65	0.57	60074.21	114805.53	0.56
Nag	1-6	60060.35	114816.49	0.60	60060.24	114815.83	0.64	60060.31	114816.21	0.62	60060.30	114816.18	0.62
Nag	1-7	60048.29	114825.46	0.67	60048.16	114824.96	0.67	60048.11	114825.21	0.68	60048.19	114825.21	0.67
Nag	1-8	60038.03	114833.57	0.74	60037.92	114833.00	0.74	60037.81	114833.30	0.75	60037.92	114833.29	0.74
Nag	2-1	60270.42	114609.60	0.39	60271.96	114610.29	0.41	60270.51	114609.12	0.34	60270.96	114609.67	0.38
Nag	2-2	60281.36	114611.43	0.59	60283.12	114612.46	0.63	60281.28	114610.90	0.55	60281.92	114611.59	0.59

Nag	2-3	60294.24	114613.64	0.62	60295.83	114614.61	0.67	60294.00	114613.12	0.58	60294.69	114613.79	0.62
Nag	2-4	60303.15	114615.00	0.60	60305.29	114616.02	0.63	60303.18	114614.58	0.59	60303.88	114615.20	0.61
Nag	2-5	60313.86	114616.60	0.58	60315.74	114617.74	0.61	60313.83	114616.20	0.58	60314.48	114616.85	0.59
Nag	2-6	60324.75	114618.24	0.64	60326.75	114619.39	0.70	60324.42	114618.04	0.64	60325.31	114618.56	0.66
Nag	2-7	60335.61	114619.77	0.70	60337.57	114621.10	0.70	60335.42	114619.61	0.68	60336.20	114620.16	0.69
Nag	2-8	60345.49	114621.30	0.76	60347.09	114622.52	0.76	60345.58	114621.33	0.73	60346.06	114621.72	0.75
Nag	3-1	n/a	n/a	n/a	60301.27	114473.39	0.67	60299.40	114472.67	0.62	60300.33	114473.03	0.64
Nag	3-2	60308.62	114476.21	0.63	60310.35	114477.18	0.67	60308.23	114476.71	0.63	60309.07	114476.70	0.64
Nag	3-3	60318.62	114480.88	0.63	60320.50	114481.86	0.68	60317.84	114481.27	0.63	60318.99	114481.34	0.65
Nag	3-4	60328.65	114485.43	0.64	60330.43	114486.40	0.68	60328.50	114485.78	0.63	60329.19	114485.87	0.65
Nag	3-5	60338.76	114489.92	0.45	60340.62	114490.99	0.61	60338.06	114490.79	0.60	60339.15	114490.57	0.55
Nag	3-6	60348.50	114494.63	0.49	60350.43	114495.70	0.55	60348.06	114494.92	0.51	60349.00	114495.08	0.51
Nag	3-7	60358.44	114499.30	0.63	60360.05	114500.21	0.62	60358.21	114500.12	0.58	60358.90	114499.87	0.61
Nag	3-8	60368.76	114503.89	0.75	60370.30	114504.90	0.81	60367.85	114504.79	0.71	60368.97	114504.53	0.75

Table 8. Northing, easting, and elevation (m above NAVD88) data for shallow PVC wells in Coggeshall and Nag marshes.

		2009			2010			2012			Mean		
Site	Transect-plot #	N	E	elevation	N	E	elevation	N	E	elevation	N	E	elevation
Coggeshall	2-1	n/a	n/a	n/a	63043.71	113226.55	0.44	n/a	n/a	n/a	63043.71	113226.55	0.44
Coggeshall	2-2	n/a	n/a	n/a	63039.03	113246.49	0.69	63039.03	113246.50	0.70	63039.03	113246.49	0.70
Coggeshall	2-3	n/a	n/a	n/a	63033.86	113267.36	0.62	n/a	n/a	n/a	63033.86	113267.36	0.62
Coggeshall	2-4	n/a	n/a	n/a	63029.15	113286.35	0.59	n/a	n/a	n/a	63029.15	113286.35	0.59
Coggeshall	2-5	n/a	n/a	n/a	63025.28	113303.02	0.71	63025.28	113303.04	0.73	63025.28	113303.03	0.72
Coggeshall	2-6	n/a	n/a	n/a	63021.27	113318.74	0.75	63021.26	113318.75	0.75	63021.27	113318.74	0.75
Coggeshall	2-7	n/a	n/a	n/a	63018.56	113331.67	1.25	63018.53	113331.64	1.25	63018.55	113331.66	1.25
Nag	1-1	60120.98	114771.25	0.59	60120.99	114771.24	0.61	n/a	n/a	n/a	60120.99	114771.24	0.60
Nag	1-2	60109.12	114780.15	0.48	60109.11	114780.15	0.49	n/a	n/a	n/a	60109.12	114780.15	0.49
Nag	1-3	60097.17	114789.10	0.65	60097.18	114789.10	0.67	n/a	n/a	n/a	60097.18	114789.10	0.66
Nag	1-4	60085.26	114798.38	0.67	60085.29	114798.36	0.67	n/a	n/a	n/a	60085.27	114798.37	0.67
Nag	1-5	60073.47	114807.64	0.63	60073.48	114807.65	0.65	n/a	n/a	n/a	60073.47	114807.64	0.64
Nag	1-6	60059.50	114818.18	0.70	60059.50	114818.20	0.69	n/a	n/a	n/a	60059.50	114818.19	0.70
Nag	1-7	60047.63	114827.52	0.76	60047.65	114827.54	0.76	n/a	n/a	n/a	60047.64	114827.53	0.76
Nag	1-8	60037.21	114835.44	0.84	60037.25	114835.46	0.82	n/a	n/a	n/a	60037.23	114835.45	0.83

Table 9. Changes in habitat composition based on RTK transition analyses. Habitats are listed in descending order based on the combined 2012 raw data.

	Coggeshall			Nag			Combined		
Habitat zone	2010	2012	% Change	2010	2012	% Change	2010	2012	% Change
Spartina alterniflora	70.49	130.76	85.49	84.98	98.96	16.45	154.78	229.71	48.41
S. alterniflora/lower salt meadow	113.77	101.51	-10.78	62.23	53.38	-14.22	176.00	154.89	-11.99
Lower salt meadow	110.59	56.42	-48.99	100.57	85.38	-15.10	211.17	141.80	-32.85
Iva frutescens	16.11	16.06	-0.32	33.18	34.64	4.39	49.29	50.70	2.85
Panne	0.80	3.66	358.83	0.00	20.76	inf	0.80	24.42	2963.10
Upper salt meadow	0.00	0.00	0.00	7.44	21.27	186.00	7.44	21.27	186.00
Pool	8.45	7.63	-9.72	0.00	0.00	0.00	8.45	7.63	-9.72
Creek/ditch	1.00	1.94	93.40	5.25	1.70	-67.73	6.26	3.63	-41.92
Grazed creek bank	0.00	2.88	inf	0.00	0.00	0.00	0.00	2.88	inf
S. alterniflora/*Salicornia* spp.	0.00	2.68	inf	0.00	0.00	0.00	0.00	2.68	inf
Wrack	0.00	2.66	inf	0.00	0.00	0.00	0.00	2.66	inf
S. alterniflora/upper salt meadow	0.00	0.00	0.00	0.00	2.12	inf	0.00	2.12	inf
Walking trail	0.00	0.00	0.00	0.00	1.13	inf	0.00	1.13	inf
Salicornia spp.	0.00	0.80	inf	0.00	0.00	0.00	0.00	0.80	inf
Unknown	4.58	0.00	-100.00	0.00	0.00	0.00	4.58	0.00	-100.00
Upper brackish meadow	0.00	0.00	0.00	23.77	0.00	-100.00	23.77	0.00	-100.00
Total length of transects (m)	325.79	326.99		317.42	319.33		643.22	646.31	

Table 10. SIMPER results for within-marsh similarity (top and middle) and between-marsh dissimilarity (bottom) in habitat composition in Coggeshall and Nag marshes. Data are combined from 2010 and 2012. Av=average; SD=standard deviation.

Coggeshall Marsh (average similarity = 69.39)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative.%
Spartina alterniflora	3.09	18.42	6.42	26.54	26.54
Lower salt meadow	2.94	18.41	2.95	26.53	53.08
S. alt/lower salt meadow	3.05	16.62	4.22	23.96	77.03
Iva frutescens	2.03	13.33	6.59	19.21	96.25

Nag Marsh (average similarity = 75.42)

Species	Av.Abundance	Av.Similarity	Similarity/SD	Contrib%	Cumulative.%
Spartina alterniflora	3.14	19.28	9.38	25.56	25.56
Lower salt meadow	3.14	19.15	7.03	25.39	50.95
S. alt/lower salt meadow	2.71	15.43	4.57	20.45	71.41
Upper salt meadow	1.70	8.11	1.29	10.75	82.16
Iva frutescens	1.81	5.63	0.64	7.46	89.62
Panne	0.99	4.84	0.91	6.42	96.04

Coggeshall Marsh v Nag Marsh (average dissimilarity = 32.84)

Species	Coggeshall Av.Abundance	Nag Av.Abundance	Av.Dissimilarity	Dissimilarity/SD	Contrib%	Cumulative%
Upper salt meadow	0.00	1.70	5.61	2.09	17.08	17.08
Iva frutescens	2.03	1.81	3.81	1.64	11.59	28.67
Creek/ditch	0.49	0.99	3.08	1.19	9.38	38.06
Panne	0.52	0.99	3.02	1.04	9.19	47.25
S. alt/lower salt meadow	3.05	2.71	2.92	1.64	8.90	56.15
Pool	0.76	0.00	2.25	0.68	6.85	63.00
Upper brackish meadow	0.00	0.50	1.80	0.43	5.47	68.47
Spartina alterniflora	3.09	3.14	1.79	1.53	5.44	73.91
Lower salt meadow	2.94	3.14	1.59	1.34	4.84	78.75
Grazed bank	0.49	0.00	1.57	0.68	4.77	83.52
Wrack	0.45	0.00	1.34	0.67	4.09	87.61
Unknown	0.33	0.00	1.03	0.43	3.13	90.75

Table 11. Habitat changes at specific locations along all 6 transects in Coggeshall and Nag marshes. For example, 14 m of habitat that was *S. alterniflora* in 2010 changed to panne habitat in 2012.

From (2010)	To (2012)	Amount (m)	% of all changes	Cumulative %
S. alterniflora/lower salt meadow	*Spartina alterniflora*	87	34.80	34.80
Lower salt meadow	*S. alterniflora*/lower salt meadow	66	26.40	61.20
S. alterniflora	Panne	14	5.60	66.80
Upper brackish meadow	Upper salt meadow	12	4.80	71.60
Upper brackish meadow	Lower salt meadow	10	4.00	75.60
Lower salt meadow	Panne	10	4.00	79.60
S. alterniflora	*S. alterniflora*/lower salt meadow	8	3.20	82.80
Unknown	*S. alterniflora*	4	1.60	84.40
S. alterniflora	Creek/ditch	4	1.60	86.00
Creek/ditch	*S. alterniflora*	4	1.60	87.60
S. alterniflora/lower salt meadow	*Iva frutescens*	3	1.20	88.80
Lower salt meadow	*S. alterniflora*	3	1.20	90.00

Table 12. Targeted changes in select habitat types from 2010 to 2012. For example, 126 m of lower salt meadow habitat that was present in 2010 remained the same in 2012, while another 10 m converted into pannes.

From (2010)	To (2012)	Amount (m)	% of all changes	Cumulative %
Lower salt meadow	Lower salt meadow	126	59.43	59.43
	S. alterniflora/lower salt meadow	66	31.13	90.57
	Panne	10	4.72	95.28
S. alterniflora	*S. alterniflora*	128	82.58	82.58
	Panne	14	9.03	91.61
	S. alterniflora/lower salt meadow	8	5.16	96.77
S. alterniflora/lower salt meadow	*S. alterniflora*	87	49.43	49.43
	S. alterniflora/lower salt meadow	79	44.89	94.32
	Iva frutescens	3	1.70	96.02
Upper brackish meadow	Upper salt meadow	12	50.00	50.00
	Lower salt meadow	10	41.67	91.67
	S. alterniflora/lower salt meadow	2	8.33	100.00
S. alterniflora	Panne	14	51.85	51.85
Lower salt meadow		10	37.04	88.89
S. alterniflora/lower salt meadow		2	7.41	96.30

Table 13. Descriptive information for surface elevation tables (SETs) in Coggeshall and Nag marshes. Elevations are in meters above NAVD88.

Site	SET #	Transect #	Latitude (DMS)	Longitude (DMS)	Elevation	Habitat	Rod depth (m)	Date installed	Date 2012 reading
Nag	1	1	41 37 27.568 N	71 19 20.942 W	0.82014	Salt meadow	10.18	8/14/2012	10/10/2012
Nag	2	1	41 37 26.585 N	71 19 20.139 W	0.83332	Short-form *S. alterniflora*	9.75	8/3/2012	10/10/2012
Nag	3	2	41 37 33.413 N	71 19 28.747 W	0.71987	Short-form *S. alterniflora*	9.51	7/31/2012	10/10/2012
Nag	4	2	41 37 35.276 N	71 19 28.284 W	0.89396	Salt meadow	11.80	7/31/2012	10/10/2012
Nag	5	3	41 37 34.740 N	71 19 34.420 W	0.80696	Salt meadow	17.07	7/31/2012	10/10/2012
Nag	6	3	41 37 35.846 N	71 19 34.020 W	0.72406	Short-form *S. alterniflora*	13.41	8/3/2012	10/10/2012
Coggeshall	7	1	41 39 00.762 N	71 20 34.374 W	0.78019	Short-form *S. alterniflora*	11.47	8/15/2012	10/11/2012
Coggeshall	8	1	41 38 58.865 N	71 20 33.272 W	0.79722	Salt meadow	7.32	8/15/2012	10/11/2012
Coggeshall	9	2	41 39 02.889 N	71 20 27.037 W	0.76617	Short-form *S. alterniflora*	7.91	8/29/2012	10/11/2012
Coggeshall	10	2	41 39 02.443 N	71 20 24.222 W	0.88114	Salt meadow	4.67	8/15/2012	10/11/2012
Coggeshall	11	3	41 39 11.178 N	71 20 34.319 W	0.64889	Short-form *S. alterniflora*	7.55	8/29/2012	10/11/2012
Coggeshall	12	3	41 39 10.217 N	71 20 34.800 W	0.85955	Salt meadow	8.27	8/29/2012	10/11/2012