Mapping vegetation composition on Rhode Island coastal salt marshes for habitat quality assessment

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A MAJOR PAPER SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF MASTER OF ENVIRONMENTAL SCIENCE AND MANAGEMENT

UNIVERSITY OF RHODE ISLAND

July 8, 2011

MAJOR PAPER ADVISOR: Peter Paton

MESM TRACK: Conservation Biology

Abstract

Salt marshes provide some of the most fertile and productive habitats in coastal environments. This foundation is primarily based on the tightly integrated plant communities that exist within these ecosystems. A historical overview of salt marsh development and vegetation composition is discussed to introduce the ecological benefits a healthy marsh system provides to coastal wildlife and humans. However, it is understood that these natural processes have been disrupted since the 18th Century due to a multitude of anthropogenic impacts. The vegetation mosaic found on salt marshes today has been influenced by hydrological manipulations such as ditching and draining. In addition, eutrophication from agriculture and excessive development along the coast has encouraged the spread of the invasive plant species, *Phragmites australis*. The consequences to these anthropogenic changes have proved to impose negative impacts on higher trophic levels. Avian surveys in Rhode Island showed a drastic population decline over 25 years in the Seaside Sparrow (*Ammodramus maritimus*), a salt marsh-dependent songbird. This prompted the use of a multifunctional ecological assessment model to evaluate the habitat value of salt marshes. Most the data for the model variables were organized in Arc Map software (version 10.0, Environmental Systems Research Institute, Redlands, CA), using available Rhode Island Geographic Information System (RIGIS) datasets. In addition detailed vegetation maps were created through photointerpretation and ground truthing using the 2008 E911 Digital color orthophotography, available from the Rhode Island Digital Atlas Map Services. Although the vegetation maps were initiated for the respective model, their value extends to other aspects of environmental science. One underlying theme presented in the paper highlights that the maps provide a detailed and current baseline reference for vegetation composition of RI salt marshes. This can be applied to many on-going monitoring studies including sea-level rise and rate of *Phragmites* invasion. To make future conservation research projects possible, the maps are available through the University of Rhode Island's (URI) Master of Environmental Science and Management (MESM) web site (http://edc.uri.edu/mesm/docs/Majorpapers/mn/).

Introduction

Salt marsh communities are highly productive ecosystems that provide a transition zone between terrestrial and marine environments. The vegetation communities within salt marshes provide the structure for the marsh and are the basis for critical wildlife habitat and ecosystem services valued by society. Although there are only a handful of plant species that occupy the harsh environmental conditions of salt marshes, they are recognized as being fairly stable in the environment over a period of 100 years (Bertness and Ellison 1987).

However, it is unlikely that the vegetation composition of New England salt marshes we currently observe reflects the patterns of natural evolutionary processes (Bertness et al. 2002). Coastal wetlands have undergone extreme manipulations that suited the needs of an early society. Prior to understanding the ecological value of salt marshes, society saw these areas as commodity rich areas ready for the taking. *Spartina patens* (salt hay) harvesting, commercial fishing, waterfowl hunting, and numerous recreational activities currently impose on estuarine systems all over the world. As a result, the estuarine ecosystem has been altered, and impacts from anthropogenic changes have been proven to negatively influence the ecology of today's salt marshes.

Despite these anthropogenic changes, marsh formation, sediment accretion rates and plant assemblage has been extensively studied over the years. This fundamental knowledge can help us determine adaptive management strategies in the face of increased coastal development and climate change. The current management trend for coastal areas has moved away from exploitation of resources to sound environmental practices concerned with conservation and restoration in efforts to regain value for coastal ecosystem services. Without a basic understanding of salt marsh development, restoration efforts are likely to be ineffective, reducing the value of salt marshes to society and wildlife alike.

Natural Processes

Marsh Development

Understanding salt marsh development is critical for assessing the current ecological trends in marsh ecosystems. Redfield (1972) investigated these processes over a 12-year study period at Barnstable Marsh in Massachusetts. Through extensive core sampling, Redfield (1972) estimated that the marshes in Barnstable had developed over 4,000 years ago. He determined that this process occurred after the most recent glaciation, the Quaternary period, when the estimated sea level was 5.5 meters lower than in 1972.

The original topography of the Barnstable marsh was gently sloping, which promoted marsh development. Sediments were deposited from land during natural shoreline erosion processes, such as floods and wind, as well as sedimentation from the ebb and flow of tides. Through his study, Redfield (1972) recognized five stages of colonization in the low marsh, which is dominated by *Spartina alterniflora* (smooth cordgrass); 1) Colonization, 2) Juvenile marsh, 3) Panne marsh, 4) Slough marsh and 5) Transition to high marsh.

In the first stage, seedlings of *S. alterniflora* colonized mudflats creating clumps of grass through rhizomes, which eventually established the juvenile marsh, which is a uniform stand of *S. alterniflora* that averages 1.2 meters in height. At this point in marsh development there are no significant bare patches or drainage channels. The even distributions of plants eventually cause uneven deposition of sediments, creating pannes that hold water at low tide in the second stage. Eventually sediment deposition and vertical accretion from dead plant material exacerbate the depressions and ridges, which are identified as the slough stage. The ridges formed during this stage hold optimum conditions for *S. alterniflora* where it grows up to 2 meters tall (Redfield 1972). The pannes, or isolated depressions above mean sea level (MSL), can reach salinities of 40-60 ppt, which is too saline for *S. alterniflora*, so they remain bare (Neiring and Warren 1980). The final stage is the transition into high marsh. Sedimentation continues and ridges start to spread around the more shallow pannes and *S. alterniflora* will decrease in height as a result of increased elevation (Redfield 1971).

Redfield's (1972) work emphasized the importance of organic matter as a function of vertical accretion. Other studies have since indicated that inorganic sediment and minerals are also influential to the overall accretion rates of salt marshes (Bricker-Urso et al. 1989). Inorganic sediment from terrestrial runoff can provide nutrients that promote primary production which increase organic matter deposition (Bricker-Urso et al. 1989). In the case of Narragansett Bay, accretion rates for the low marsh proved to be a combination of organic and inorganic sedimentation. In the high marsh, accretion rates were a result of organic matter deposition (Bricker-Urso et al. 1989).

Redfield (1972) also explained the abilities of salt marshes to develop in times of relatively slow sea-level rise. Over the 4,000 years in which the Barnstable marsh developed, sea level increased 7.6 meters relative to land (Redfield 1972). In fact, current salt marshes have developed during periods of slow sea level rise (1mm/year) (Neiring and Warren 1980, Donnelly and Bertness 2001). However, radiocarbon peat core samples indicate that relic marshes developed when the average sea level rise was 16mm/year (Bricker-Urso et. al. 1989, Nixon 1980). This appears to be that maximum rate of sea level rise in which salt marsh development can 'keep up' (Bricker-Urso et al. 1989). Therefore, continuously assessing accretion rates against the relative sea level rise is important in order to strategically plan for future salt marsh conservation efforts.

Plant Zonation

Salt marshes exhibit strong vertical plant zonation due to hydrology, micro relief, and available nutrient and oxygen levels (Neiring and Warren 1980). The factors of plant composition are described in general to give a broad idea of the dynamic processes that occur on salt marshes. Experimental studies of plant zonation show that physical environmental stressors are major factors that determine species distribution in the low marsh, whereas interspecific competition is the causal factor in plant patterns on the high marsh (Bertness and Ellison 1987).

The low marsh typically is regularly flooded and dominated by *S. alterniflora* (Bertness 1999). *S. alterniflora* has developed an aerenchyma system, which serves as a pathway for the diffusion of atmospheric oxygen from the leaves to the roots (Mendelssohn and Postek 1982). The oxygen not consumed in the roots during respiration diffuses into the anoxic mud surrounding the roots and forms an oxidized rhizosphere. Not only does this rhizosphere aerate the soil substrate, it is also capable of metabolizing toxic chemicals into relatively harmless compounds. This adaptation responds to the harsh coastal conditions and functions to support primary productivity in the regularly flooded zone (Mendelssohn and Postek 1982).

Tall-form *S. alterniflora*, which defines the low marsh for the purpose of this study, occurs along the seaward edge of the low marsh in the irregularly exposed class of the intertidal zone (Cowardin et al. 1979, discussions with URI faculty). *S. alterniflora* has the capability of growing to heights of almost 2 meters due to several factors. First, oxygen levels improve in the low marsh with the ebb and flow of tides, which in turn allows for constant salinity levels (Niering and Warren 1980). The high marsh generally experiences infrequent flooding which can result in a high concentration of salts due to evapotranspiration (Bertness and Ellison 1987, Mitch and Gosselink 2007). In addition, the tidal flux simultaneously provides the most nutrients as well as toxic substance relief to the fringing areas of the low marsh (Mitch and Gosselink 2007, Niering and Warren 1980). This can also be seen along estuarine streambeds in the irregularly exposed zones (Mendelssohn and Postek 1982). These factors allow for higher productivity, which promotes vertical accretion resulting in waterlogged conditions landward of the stream edge (Mendelssohn and Postek 1982). Therefore, increased oxygen and nutrient circulation are factors that directly affect the height of *S. alterniflora.*

S. patens is generally the dominant species on the high marsh in New England (Bertness 1999, Niering and Warren 1980). Although *S. alterniflora* will grow in the well-drained soils of the high marsh, the thick, turf-like growing characteristics of the *S. patens* outcompetes *S. alternifora* just landward of mean high water (MHW) (Bertness and Ellison 1987). In addition to *S. patens,* other species including *Juncus gerardi* (saltmeadow rush), *Salicornia europaea* (glasswort) and *Disticlis spicata* (saltgrass) find their niche in the high marsh. The latter two are associated with being the first to colonize naturally disturbed, bare areas (Bertness and Ellison 1987). During storm or moon tide events, the velocity of the tidal waters can deposit clumps of dead plant material, or wrack, from the low marsh to higher ground (Niering and Warren 1980, Bertness and Ellison 1987). This process covers vegetated areas, creating bare patches that develop high soil salinities. *Salicornia* and *Disticlis* are well adapted to these conditions and facilitate the colonization of other species by diffusing the salts through their vascular system and by shading the soil substrate to reduce evaporation (Bertness and Ellison 1987). *S. patens* and *Juncus* will eventually take over that isolated area when the soil chemistry has reached optimal conditions (Bertness and Ellison 1987).

In areas of even higher elevation within the marsh, woody species such as *Iva fructescens* (marsh elder) and *Baccharis halimifolia*(eastern baccharis) grow. Much like mangrove species, they have adapted their root system to extend slightly above ground to allow for more oxygen exchange. In addition, *Phragmites australis* (common reed) is common on the upland edge of marshes. Although, since increased development and human disturbance, *Phragmites* has been able to invade seaward areas of the high marsh, outcompeting the natural vegetation mosaic described above (Bertness et al. 2002, Meyerson et al. 2009).

Ecosystem Services of Salt Marshes

Plants are an integral part of the salt marsh ecosystem. They essentially design and maintain this unique landscape, which provide many ecological functions. Not only do they accrete and stabilize sediments that are the basis for marsh formation, the plant community is subsequently involved with protecting the sea and landscapes on either side. Storm waves approach the shore with intense velocity, all the while transporting sediment and nutrients from the sea. The vegetation on the marsh acts as a buffer to attenuate storm surge. In addition, the sediments are trapped by the vegetation, which has direct influences on the speed of the water. Wave energies can be reduced up to 26% on the seaward edge, and reduced further as the wave travels over the marsh (Bertness 1999). In cases where water is funneled to land via a narrow bay or river, the intensity can increase resulting in major destruction on land (Bertness 1999). Not only does this have significant value for the ever-growing human populations along the coast, but it also acts to prevent shoreline erosion from terrestrial runoff.

In the same way that salt mashes protect the land from destructive waves, they also protect the adjacent water body from freshwater runoff and flood events (Bertness 1999, Mitch and Gosselink 2007). As stated before, water velocities decrease as it passes over the vegetation, which limits the water's ability to hold sediments. This prevents particulates from entering the subtidal zone of the estuarine system. Many of the nutrient and pollutant particles in the sediments are taken up by the plants root system. In cases of heavy metal deposition, the anoxic substrate itself is able to break down and store the metal complexes (Bertness 1999, Mendelssohn and Postek 1982, Mitch and Gosselink 2007). This functions to maintain water quality of the estuary and reduce turbidity, inhibiting photosynthesis of submerged plant material (Bertness 1999, Mitch and Gosselink 2007). The previous examples show the importance of salt marshes as a buffer system between terrestrial and marine environments with respect to sediment transfer and pollutant abatement.

Sedimentation from the ocean and upland along with plant decomposition provides the basis for the estuarine food web (Bertness 1999). The rate of primary production of *S. alterniflora* has a significant influence on the overall productivity of the coastal ecosystem. Eutrophication is shown to have major effects on the production of marsh plants; including increased plant density, which in turn has a positive influence on herbivory, detritus and overall ecosystem productivity (Bertness 1999, Gedan et al. 2009). Excess nitrogen and nutrients from point and non-point sources can present negative issues such as altered species distribution that will be discussed later in the paper.

Salt marshes are detrital-driven systems, where dead plant material is converted to nutrient rich organic matter by fungi and bacterial processes, which provides food to benthic organisms (Bertness 1999). Filter feeders, such as mussels, attach to the roots of *S. alterniflora* to feed on the dispersed nutrients from the marsh. This symbiotic relationship aids in stabilizing the vulnerable seaward edge of the marsh as well as maintaining clarity of the water column (Bertness 1999). Clear estuarine waters are an important attribute providing crucial habitat for larval and juvenile stages of other fish and invertebrates (Bertness 1999, Nixon 1982). Without water transparency, submergent life forms, such as eelgrass (*Zostera marina)*, would not be able to photosynthesize, thus degrading the quality of nurseries. Many nekton species such as mummichogs (*Fundulus heteroclitus)*, flounder (*Pseudopleuronectes americanus)* and herring (*Clupea harengus)* rely on eelgrass and other aquatic plant life for sustainability. These nurseries are an integral part in the food web and provide a range of linkages from primary producers to predators (Bertness 1999).

Salt marshes in New England support at least 31 bird species (Shriver et al. 2003). Few are salt marsh obligates including Willet (*Catoptrophorus semipalmatus*), Seaside sparrow (*Ammodramus maritimus*) and Saltmarsh sparrow (*Ammodramus caudacutus*) utilizing the marsh for breeding, foraging, and social displays (Shriver et al. 2003). Other bird species utilize the marsh for nesting and foraging, including the Clapper rail (*Rallus longirostris*), Red-winged black bird (*Agelaius phoeniceus*) and the American black duck (*Anas rubripes*) (Shriver et al. 2003). However, the majority of the bird species use New England salt marshes for foraging (Shriver et al. 2003).

In RI, there are only four species that breed on salt marshes. They include the Saltmarsh and Seaside sparrow, the Red-winged black bird and the Marsh wren (*Cistothorus palustris*). The latter two also use freshwater wetlands for breeding as well (Stoll and Golet 1983). In the case of the Seaside Sparrow and other obligate and facultative marsh fauna, their survival depends on the quantity and quality of food as well as the size and composition of the marsh (Stoll and Golet 1983, Marshall and Reinert 1989). Although passerine birds are not associated with the charismatic mega fauna such as egrets or herons, observations made on abundance can help to assess marsh habitat quality as in the case of the two pertinent studies for this paper (Stoll and Golet 1983, Berry et al. 2009).

Anthropogenic Changes

Physical Change

Modifications to New England salt marshes reflect the direct mechanical abuse by early society. Ditching and draining of marshes was popular in colonial times when harvesting *S. patens* for animal feed and bedding (Bertness 1999, Gedan et. al. 2009). This manipulation escalated to widespread ditching projects for mosquito control to prevent the spread of malaria and yellow fever after the Civil War. By the depression era, nearly all of the salt marshes in the northeast were ditched in some fashion. The ditched material was usually placed adjacent to the ditch to create a levee, which immediately changes the micro topology of the marsh landscape (Bertness 1999). Even slight elevation changes will reconfigure tidal flow on the marsh. Drastically changing the hydrology of the salt marsh in this way will inevitably create more high marsh conditions conducive to supporting patches of *S. patens, Juncus* and *Disticlis*, which will eventually replace *S. alterniflora* (Bertness 1999, Niering and Warren 1980). These changes in estuarine habitat have negative effects on the abundance of obligate salt marsh birds that use the marsh for breeding. However, most generalists bird species that utilize salt marshes for behaviors other than breeding are relatively unaffected by hydrologic changes (Pepper 2008).

These direct anthropogenic changes have caused New England to loose about 80% of their coastal salt marshes (Bertness et al. 2002). Environmental changes indirectly caused by humans are also a threat to the future of salt marshes. Climate change and sea level rise are inevitable and immediate efforts are needed to restore and protect these valuable habitats. Many agencies are looking at ways to assess the environmental and social values of salt marshes to better manage them. Detailed, current vegetation maps provide these organizations with a necessary tool in expediting these efforts.

Eutrophication

Historic nutrient loading is a result of wastewater and agricultural runoff, atmospheric deposition and groundwater discharge from high-density populations. With the Northeast being the most heavily populated region of the Atlantic Coast, this puts the region's salt marshes at major risk. Salt marshes are naturally adapted to deal with sedimentation and nutrient transfers; however excess amounts will cause the system to malfunction. In a natural setting, ground water filters through the organic substrate of the salt marsh, the process of denitrification removes land-generated nitrogen before reaching estuarine waters (Bertness 1999, Mitch and Gosselink 2007). Increased nutrient loading results in increased macroalga, phytoplankton blooms and seagrass epiphyte densities. These organisms restrict necessary light to eelgrass inhibiting photosynthesis causing severe degradation to critical habitat for fish and invertebrates (Bertness 1999, Gedan et. al. 2009).

Since the 18th century, the land cover and land use has dramatically changed in the region due largely to agricultural and industrial practices. Extensive land clearing preceded the American Industrial Revolution, which introduced local salt marshes to many contaminants, such as lead, chromium and copper (Roman et. al. 2000). Development since this time period has continued to fragment and degrade the forest and shrub buffer habitats adjacent to coastal wetlands, which are designed to intercept terrestrial runoff and reduce overall stress to coastal areas. Without the natural vegetated buffer, freshwater runoff is accelerated as well as the rate of pollution (Silliman and Bertness 2004, Mitch and Gosselink 2007, Roman et al. 2000).

Eutrophication can have significant effects on the distribution of plant species. While there is conflicting evidence that heavy nutrient loads influence the distribution of the native grass species found on the marsh (Bertness et al. 2002, Niering and Warren 1980), there is solid evidence eutrophication promotes the colonization and spread of *Phragmites*, a robust invasive grass species (Bertness et al. 2002, Gedan et al. 2009, Meyerson et al. 2009, Silliman and Bertness 2004). The removal of buffer zones creates disturbance and allows *Phragmites* establishment (Meyerson et al. 2009). In addition, it decreases marsh salinity by allowing more freshwater runoff to be displaced to the marsh without interruption (Bertness 1999). The native grasses that occupy the high marsh (*S. patens, Juncus, Disticlis*) compete with each other for nutrients (Bertness et al. 2002, Bertness and Ellison 1987). When nutrients are readily available, light becomes the competing factor (Bertness and Ellison 1987). The three native species previously mentioned are all relatively the same height at maturity, so the initial mosaic of the high marsh is unaltered in this situation. However, *Phragmites* can reach heights of almost 6 meters (Meyerson et al. 2009), giving it an advantage in the competition for light and enabling the plant to invade substantial areas of the marsh. Most of the salt marshes of interest in this project currently have relatively low Phragmites coverage relative to the size of the marsh, however, *Phragmites* propagates rapidly through underground rhizomes (Meyerson et al. 2009) and therefore it is an important factor when assessing the health of the salt marsh.

History of Study

Golet and Stoll (1983) conducted a preliminary study of the abundance of Seaside Sparrows. The study assessed the effects of mosquito ditching in salt marshes for avian habitat. Surveys were done at 23 sites along the South Shore and Narragansett Bay, RI. Sites were chosen based on previous sparrow sitings in the state. This comprehensive study not only provided an overall account of species abundance, it revealed important ecological preferences in nesting, feeding and social habits of the seaside sparrow relative to marsh size and composition.

The study also concluded that because RI is located in the northern boundary of the range of Seaside Sparrows, it would inevitably display sporadic abundance. However, it was also noted that due to major loss of marsh habitat through anthropogenic activities, such as filling and dredging, the integrity of this habitat has been compromised especially to salt marsh specialist like the Seaside Sparrow (Stoll and Golet 1983).

In efforts to assess the impacts of physical land use and land cover surrounding RI marshes, the Atlantic Ecology Division (AED) of the Narragansett Environmental Protection Agency (EPA), conducted a similar avian survey at the same 23 sites throughout Narragansett Bay and the South Shore of RI (Figure 1) (Berry et al. 2009). The EPA conducted the survey twice during 2007 and 2008. The results of the two EPA surveys were similar, which showed a 50% average decrease in Seaside Sparrow population indices from 2007-08 to 1982 surveys (Berry et al. 2009). Counts of Seaside Sparrows decreased in 2007 at 13 of 14 sites were there were birds previously observed in 1982. Most marshes where there were four or fewer sparrows were detected in 1982, had no detections in 2007 or 2008. Only two salt marshes had an increase in sparrow index values (Berry et al. 2009). One marsh, Galilee Wildlife Refuge in Narragansett, had undergone major restoration efforts to simulate the natural hydrology of the salt marsh (Tefft 1998).

Photo interpretation and analytical efforts by the EPA showed that the general size and shape of the marshes apparently remained the same over the 25 year study, yet there was considerable difference in residential development adjacent to many of the marshes (Berry et al. 2009). A qualitative analysis suggested that increased development adjacent to the salt marshes had more influence over decreased numbers as opposed to marshes that were bordered by agricultural or protected lands (Berry et al. 2009). For example, East Beach in Charlestown, where adjacent habitats remained relatively unchanged due to protection, exhibited no changes in sparrow indices. However, because of the number of marshes that lost entire populations of birds over time, finding significant relationships between the decline of Seaside Sparrows and changes in land use patterns was difficult (Berry et al. 2009).

In 2009, the EPA decided to implement a multifunctional assessment model to determine habitat quality of salt marshes. There are habitat models that have been developed to associate a value based on a numerical score that is calculated by multiplying weighted attributes associated with salt marsh habitat and productivity (McKinney et al. 2008). In general, high quality habitat includes an even distribution and high diversity of marsh habitat types, including low marsh, high marsh, shallow open water, tidal flats, wooded islands, etc (McKinney et al. 2008). In addition, diversity of plant material with emphasis on emergent aquatic vegetation and high heterogeneity of edge habitat is considered high quality salt marsh. Other factors such as surrounding land use and anthropogenic modifications to tidal hydrology are important in determining salt marsh habitat quality (McKinney et al. 2008).

The particular model used by the EPA was specifically designed for Rhode Island and provided a sociological element (Johnston et al. 2002). The model inputs were gathered through a series of expert surveys. The survey was designed to determine values of various ecological relationships within the salt marshes. This study investigated how the physical attributes of salt marshes and their surrounding landscape contributed to associated habitat functions for birds, fish and shellfish (Johnston et al. 2002). The attributes are similar in nature to the previously mentioned study. The added sociological element integrated the values RI residents have for the marshes their local coastal landscapes. By incorporating this parameter, future restoration efforts can be seen as the potential for the greatest improvement with a fixed budget (Johnston et al. 2002).

Study sites

Salt marshes are considered low-energy systems, despite their high ecological productivity (Bertness 1999, Mitch and Gosselink 2007). Barrier islands and spits (beaches), along with the mouths of coastal rivers, provide salt marshes with protection from strong winds and currents. The slow moving waters allow for sediments to accumulate vertically and eventually horizontally along the coast, making conditions conducive for halophytic plant production and salt marsh development (Bertness 1999, Niering and Warren 1980, Redfield 1972).

I had 25 study sites, of which about 50% were located in the Narragansett Bay. Narragansett Bay encompasses approximately 342 km^2, with salinity ranging from 18% to 33% and little fresh water input (Bricker-Urso et al. 1989). The watershed rests on carboniferous bedrock; however sedimentation rates are stable with sea-level rise, due to the fact that there has been little to no marsh loss in the past 80 years (Bricker-Urso et al. 1989).

The south shore of RI is located on outwash plains and is composed of a series of shallow coastal lagoons that have historically been breached, although at least one remains intact. Tide levels and salinity percentages are comparable to those in the Bay (Bricker-Urso et al. 1989).

Methods

The goal of this project was to supply the EPA with accurate data that corresponded to the attributes, as inputs for the model. The intention was to use readily available data to gather the pertinent information for each salt marsh site. For most of the inputs, analyzing the appropriate State of Rhode Island Geographic Information System (RIGIS) datasets along with RI's latest georeferenced aerial photography worked well. However, two of the parameters required specific information that was not available as a concise dataset. The first limiting input asked for percentage of *Spartina spp*. coverage verses coverage of *Phragmites australis*. In order to accurately input this information to the model, each study site was mapped according to its vegetation composition (Table 2).

This extensive mapping project was the focus of this study. It became clear in my preliminary research of salt marsh ecology and obligate salt marsh passerines (Seaside and Saltmarsh sparrows), that these two grass species supported different habitat functions. Therefore, although the model required pooling both species into the category *Spartina spp*., I chose to separate *S. patens* and *S. alterniflora* on the maps in order to present a more versatile data set. I also mapped the shrub layer that is included in the dataset. For aesthetic values as well as for the purpose of assessing ecological functions, each study site represents a comprehensive vegetation map of *S. alterniflora, S. patens*, shrub, (general estuarine scrub shrub species) and *Phragmites australis*. As a result, the maps are an essential baseline tool for the model and provide other management implications.

The second parameter that proved difficult to determine was the percentage of low marsh consisting of tall-form *S. alterniflora*. The categories, which were determined by the authors of the model based on the expert solicitations (Johnston et al. 2002), included 1) < 4% low marsh and 2) <10% low marsh. In recent discussions with URI faculty, I learned that most all of RI's salt marshes do not contain more than 4% low marsh. The task of accurately mapping that specific vegetation type would prove to be tedious and beyond the scope and time frame of this project. Therefore, I opted to do a general field survey at each site and estimate the approximate percentage of low marsh based on ocular estimates.

Preliminary Work

In order to accurately map vegetation at each site, I had to conduct fieldwork to groundtruth aerial photographs. Prior to fieldwork, 57 site maps were created to cover the 25 study site. Multiple maps were needed for each site due to the size and shape of the marshes. The maps were created using the 2008 E911 Digital color orthophotography available from the Rhode Island Digital Atlas Map Services (http://maps.edc.uri.edu/ArcGIS/rest/services/Atlas_imageryBaseMapsEarthCover/2008_RIE911/MapServer). This readily available imagery has the highest resolution of any orthophotography for RI at the moment and has a 4 inch pixel size. Most of the site maps were photointerpreted at scales from 1:2,000 - 1:4,000. There are a few maps that have extreme scale measures due to the nature of the marsh and imagery. For example, if a salt marsh was relatively small, the scale would be much larger than 1:2000. The imagery also has different tones and shadows based on the angle in which the photo was taken and the topography of the ground surface. To utilize the imagery for optimum texture contrast, a wide range of scales were necessary.

Fieldwork

The fieldwork consisted of visiting a portion of each marsh at least once. I used preliminary vegetation maps to delineate the marshes in the field until I was confident that all tones and textures were accounted for. Each combination of tone and texture represented different vegetation species. I was able to visit all 25 marshes between September and November of 2010.

I recorded date, time, tide, temperature, wind speed and cloud cover for each marsh in my field notes for each visit. As mentioned previously, because I was not specifically mapping tall form of *S. alterniflora*, I noted the approximate percentage cover in my field notes.

Digital Mapping and Model Inputs

I used heads-up digitizing to map salt marsh vegetation classes. This is a common way to geocode information into Geographic Information Systems (GIS, Bolstad 2005). It is an interactive process in which a vector GIS dataset is created by tracing features off a source base map in the area of interest. I used the 2008 color georeferenced imagery as the primary source of information. Arc Map software (version 10.0, Environmental Systems Research Institute, Redlands, CA) was used to heads-up digitize the collected information. Each vegetation species was digitized as feature classes in a file geodatabase. The feature classes include the following vegetation types: *Phragmites australis*, *Spartina alterniflora*, *Spartina patens*, and shrub (i.e., estuarine scrub-shrub species as well as *Iva fructescens*). Once the vegetation types were complete for each marsh system, the individual feature classes were merged into a new single feature class in order to check for polygon overlaps, gaps and slivers within the dataset. I used the topology tool in ArcGIS for this assessment of topological irregularities. Areas were calculated to acres in order to compute the percentage of vegetation cover for each species in a marsh.

FGDC-compliant metadata were developed for the final dataset. The final vector feature dataset and corresponding metadata is available on the URI MESM web site (http://edc.uri.edu/mesm/docs/Majorpapers/mn/).

Information for the other model inputs was gathered through RIGIS datasets (Table 2). Distances to corresponding wetland and land use and land cover features were measured using the 'measure tool' in Arc Map 10.0.

Results

Most of the salt marsh sites occur as fringing landscapes that run parallel to land (Table 3). The average size of the salt marshes studied is 51.3 acres (45 SD), ranging from Barrington River Island (about 10 acres) to Seapowet in Tiverton, approximating 177 acres (Table 1).

Most (70%) of the study sites had a high percentage of *Spartina spp*. coverage and low *Phragmites* coverage (10-27% *Spartina spp*., 69-83% *Phragmites australis*), 17% fell into the medium category (50-62% *Spartina spp*., 32-50% *Phragmites australis*), and 13% were marshes that had low *Spartina spp*. coverage and high *Phragmites sp*. coverage (80-95% *Spartina spp*., 4-9% *Phragmites australis*). All marsh sites had less than 4% tall-form *S. alterniflora* low marsh. Additionally, only 4% of the marshes had severe tidal restrictions, which could indicate the high percentage of *Spartina spp*. over the amount of *Phragmites*.

Most marsh sites (80%) do not have eelgrass in adjacent waters, which could be the result of eutrophication and over development of coastal areas (Bertness 1999, Roman et al. 2000). Consequently, 68 % of the marsh sites showed medium to high (30->65%) density development. In addition, 68% of salt marshes had low to medium forest cover within 500 ft buffer of the marsh boundary, with an additional 16% in the buffer with no forested land (Table 3).

Alternatively, 76% of the marsh sites had a significant amount of tidal flats and about 50% of the marshes had a substantial number of pools and pannes (15-20% of total marsh area) suggesting that these marshes could have high habitat value to wading birds and shorebirds (Table 3).

Discussion

It is clear that anthropogenic modifications have played a significant role in determining the vegetation composition and overall habitat quality of Rhode Island's salt marshes. Not only has coastal development and hydrological alterations caused a huge net loss of coastal wetlands, the disturbance has initiated other ecological factors that have compromised the integrity and quality of the remaining salt marshes and their resident wildlife.

Phragmites

The spread of *Phragmites* has the potential to alter species diversity and abundance on the marsh. Although it has been reported that vital food sources to terrestrial and avian fauna, such as fish and aquatic invertebrates, are relatively unaffected by the presence of *Phragmites*, there is still an issue with food availability within dense monocultures of this invasive reed (Benoit and Askins 1999). Overall occupancy rates and abundance of birds was significantly reduced in salt marshes dominated by dense stands of *Phragmites* (Benoit and Askins 1999). None of the avian species studied used the *Phragmites* for breeding. This presents a major issue for salt marsh-dependent birds if *Phragmites* continues to invade and dominate the preferred shorter, native grasses. This would decrease the availability of preferred foraging and nesting habitats for birds in salt marshes, thereby decreasing species abundance. Seaside Sparrows are highly territorial (Marshall and Reinert 1990), and typically use marshes with extensive patches of the tall-form *S. alterniflora*. Thus, declines in the availability of this preferred nesting habitat will lead to declines in this species (Benoit and Askins 1999, Marshall and Reinert 1990). Therefore, the invasion of *Phragmites* could be a potential reason in the decline of Seaside sparrows, despite the fact that RI still has a high percentage of *Spartina spp*. relative to *Phragmites*.

When *Phragmites* was intermixed with pools and native high marsh grasses avian use improved (Benoit and Askins 1999). This type of habitat created more *Phragmites* edge habitat, which can be foraging habitat for some avian species. However, most birds avoid foraging within the dense thickets of *Phragmites* (Benoit and Askins 1999). This suggests that thin strips of stable *Phragmites* populations along the upland edge of the marsh or dispersed within the high marsh can provide some beneficial aspects to the ecological function of salt marshes. *Phragmites* in this setting can add to the plant mosaic of the high marsh, creating habitat heterogeneity and bird species richness (Benoit and Askins 1999). Regardless, *Phragmites* spreads at very rapid rates once established, and monitoring the spread of invasion is necessary to maintaining quality salt marsh habitat and sustaining valuable avian fauna on the marsh.

Sea Level Rise

With current sea level rising at an accelerated rate (Donnelly and Bertness 2001) it is important to reflect on the information of development and vegetation patterns that were noted in past salt marshes in order to plan future management strategies. Sea level rise attributes to the development of pools and pannes as well as horizontal accretion or landward migration of the salt marsh (Bertness et al. 2002, Gedan et al. 2009). There has been a rapid increase in horizontal accretion over the last 200 years that has coincided with the observed increase in sea level rise (Bertness et al. 2002). During this time it was noted through peat sampling that there was a decrease in high marsh plant species such as *S. patens* and *Juncus* and an increase in *S. alterniflora* (Neiring and Warren 1980). Although productivity of *S. alterniflora* is stimulated by a rise in sea level, the high marsh species cannot tolerate waterlogged conditions. This leads to very low productivity levels and eventually drowning of the high marsh (Gedan et al. 2009). This is especially true if the high marsh has no room to migrate landward. Landward accretion was feasible prior to anthropogenic restrictions that prevented the marsh from moving horizontally across the landscape. The ability of salt marshes to shift according to the sea level is permanently disrupted by the hardened shoreline that society has created. Realizing these irreversible effects can enlighten future salt marsh conservation efforts. Areas that can be revitalized to accommodate landward migration should be considered priorities in spatial coastal management.

Climate Change

Understanding the climate variation of ecological processes will help to assess the consequences of future climate change. For example, as temperature rises in the summer months, the high marsh is more susceptible to elevated soil salinities due to evapotranspiration (Bertness et al. 2002). The facultative properties of *Salicornia* and *Disticlis* ameliorate soil salinities for the succession of *S. patens* and *Juncus*. However, there is no definitive answer as to how these interactions will continue under elevated stress levels (Bertness and Ewanchuk 2002).

Although much is known about the physical and biological factors affecting plant zonation on salt marshes, it is not that clear how these patterns will respond to projected climate changes (Bertness et al. 2002). In order to accurately predict future ecological alterations, it is important to be able to generalize the existing mechanisms for spatial distribution in salt marsh vegetation. Theoretically, salt marshes that occupy similar latitudes in opposite hemispheres should share related mechanisms for vegetation patterns and species distribution relative to low and high marsh vegetation (Farina et al. 2009). Recently, studies to predict salt marsh dynamics along a latitudinal gradient have attempted to resolve these unanswered questions (Farina et al. 2009). Chilean marshes were used because of the lack of ongoing research applied to these areas in order to compare marshes along a latitudinal gradient in California and New England, where mechanisms for species distribution are well known (Farina et al. 2009). It was found that interspecific competition was a component in plant zonation for both locations, but comparing salt marsh ecosystems in opposite hemispheres did not render enough generalizations to suite a predictive model (Farina et al. 2009). Because the initial predictions for the Chilean marshes were based on only one climate type (California), the outcome was not accurate (Farina et al. 2009). The Chilean marshes proved to be a hybrid of the two climate zones that represented mechanisms for plant distribution in climates represented by California and New England. Therefore, generalizations for vegetation composition cannot be made across a latitudinal gradient (Farina et al. 2009). This point stresses the need for regional experimental ecology in order to understand community structure on a regional scale (Farina et al. 2009).

Management Implications

The vegetation maps created for this study are part of a larger project being conducted by the EPA to develop an ecological assessment model of habitat quality. However, these maps have potential to facilitate many other research efforts. Because the vegetation composition of the marsh can indicate certain hydrologic functions, these maps provide the general baseline information for understanding the volume and chemistry of freshwater runoff in to the marsh. In this example, vegetation could be compared with land use and land cover datasets to quantify relationships with different levels of population density and water quality.

The maps of the study sites are distributed fairly evenly across the state giving the researcher a good idea of RI's overall estuarine composition. Although salt marshes have relatively similar qualities with respect to supporting a diverse ecosystem, I have noticed through fieldwork and GIS methods that the vegetation composition is very different on each of the marshes. This could be useful in studying the degree of habitat fragmentation along the coast. These ideas could also lead to localized and comprehensive coastal land cover assessments that would determine new conservation management practices.

Predators function as top-down regulators to shape the estuarine ecosystem by keeping trophic cascades in balance (Bertness 1999). The level of primary production is a good indicator of species diversity and abundance at the top of the food web. Instead of managing landscapes with a top-down approach, which can often lead to issues that are well advanced, the vegetation maps could be part of a series of tools used to diagnose disruptions in the food chain, indicating health issues within the marsh before the ecosystem reaches it's resilience threshold.

Finally, although these maps are available for further expansion, they are also permanent datasets that can be used for years to show vegetation composition in the marshes of RI in 2010. This could be helpful in monitoring *Phragmites* invasion rates as well as distribution of marsh plants as the earth's climate continues to change. Hopefully this would lead to answering unresolved issues in predicting the future pathways of salt marsh ecology.

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Appendix I.

Table 1. Vegetation coverage in acres. Total represents total area of marsh site.

	Site	Phragmites	Shrub	*S.alterniflora*	*S.patens*	Total
1	100_ac_cove	10.09	0.42	52.83	21.54	84.88
2	Barr_river	0.33	0.77	5.91	2.72	9.73
3	Bluff_hill	2.17	6.33	20.31	1.88	30.69
4	Briggs_marsh	5.68	0	2.06	0	7.74
5	Ch_beach	7.47	20.99	5.78	0.75	34.99
6	Ch_Breach	17.94	42.53	27.87	3.28	91.62
7	Chafee_south	2.81	0.29	18.34	3.41	24.85
8	Chfee_north	0	1.3	13.21	1	15.51
9	East_beach	15.13	67.61	23.9	4.1	110.74
10	Fox_hill	0.62	1.55	13.92	8.9	24.99
11	Galilee	18.14	42.32	42.06	15.77	118.29
12	Kickimuit	0.38	1.8	4.34	8.58	15.1
13	Marsh_meadows	2.58	0.86	30.93	21.27	55.64
14	Palmer_East	2.58	4.04	13.92	10.13	30.67
15	Palmer_West	3.99	1.2	25.33	16.11	46.63
16	Potters_pond	7.38	8.86	10	3.99	30.23
17	Potowomut	3.33	8.73	18.55	5.44	36.05
18	Quicksand	19.76	0	0	0	19.76
19	Quonoch	2.04	4.47	57.4	5.84	69.75
20	Rumstick_Pt	2.96	0.87	10.85	13.31	27.99
21	Sachuest	8.63	12.04	8.14	2.9	31.71
22	Seapowet	28.52	16.18	91.81	40.86	177.37
23	Succotash	4.98	11.03	41.91	18.82	76.74
24	Winn	10.96	34.31	86.54	18.12	149.93
25	Barr_river_Island	0	0	9.76	0	9.76
26	100_tounge			14	6.3	20.3
	Average	7.14	11.54	24.99	9.04	51.99
	SD	7.47	17.36	24.14	9.52	45.35
	Min	0	0	0	0	7.74
	Max	28.52	67.61	91.81	40.86	177.37

Table 2. Model categories with relative tactics and resources for collecting data.

1. Fringe: Shape of the contiguous wetland area is a long, fringing marsh. Alternative is "meadow-like".	**E911 imagery**
2. Size (round to closest area): Mapped study site. Actual size, closest to 1 acre, closest to 5 acres, closest to 17 acres, closest to 65 acres	**Vegetation Maps**
3. Spartina Coverage: Percentage of contiguous wetland area covered in Spartina spp., Low (10-27% Spartina spp., 69-83% Phragmites australis) Medium (50-62% Spartina spp., 32-50% Phragmites australis) High (80-95% Spartina spp., 4-9% Phragmites australis)	**Vegetation Maps**
4. Percent low marsh: Indicates a high percentage of low marsh (Spartina alterniflora) in salt marsh area, 10% low marsh versus 4% (average in region).	**Field notes**
5. Presence of brackish marsh, which refers to all brackish areas excluding those dominated by Phragmites australis. Yes/No	**Narragansett Bay Estuarine Bay Habitat/ South Coast Estuarine Habitat**
6. Presence of estuarine scrub-shrub wetland area. Yes/No	**Narragansett Bay Estuarine Bay Habitat/ South Coast Estuarine Habitat**
7. Presence of significant tidal flats (around 25% or more of the size of the coastal wetland area). Yes/no	**E911 Imagery**
8. Presence of eelgrass in the adjacent waters. Yes/No	**Eelgrass beds in RI (poly) Narragansett Bay Estuarine Bay Habitat/ South Coast Estuarine Habitat**
9. Type of adjacent water body, including coastal river, coastal embayment and salt pond.	**E911 Imagery**
10. The degree of tidal restriction, including no tidal restrictions - which means the tidal inlet has no impacts to tidal geometry, moderately restricted - where the majority of tidal prism remains despite restrictions, or severely restricted - where the tidal prism has been largely destroyed.	**E911 Imagery**
11. Percent of marsh covered by pools, which are not likely to dry out. Levels include 1% of marsh area, 10% of marsh area and 20% of marsh area.	**E911 Imagery Narragansett Bay Estuarine Bay Habitat/ South Coast Estuarine Habitat**
12. Percent of marsh covered by pannes, which are likely to dry out. Levels include 1% of marsh area, 5% of marsh area or 15% of marsh area.	**E911 Imagery Narragansett Bay Estuarine Bay Habitat/ South Coast Estuarine Habitat**
13. The presence of sub-tidal channels, which are inundated during all tidal stages.	**E911 Imagery**
14. The presence of intertidal creeks, which usually drain completely at low tide.	**E911 Imagery**
15. Presence and type of vegetated buffer in 100-foot area around wetland. May be no buffer, a shrub buffer or a forested buffer.	**Land Cover/Land Use for RI 2003/2004**
16. Percentage of developed land use in the 500-foot area around wetland. Developed land including residential, commercial and industrial land uses. Levels of development include low with 19-27% developed, medium with 35-56% developed and high with 72-83% developed.	**Land Cover/Land Use for RI 2003/2004**
17. Percentage of agricultural and managed grassland in the 500-foot area around the wetland.	**Land**

17. Percentage of agricultural and managed grassland in the 500-foot area around the wetland. Agricultural and managed grassed areas including crops and grazing land, soccer and other playing fields, golf courses and extensive lawns. Levels include low with 8-18% in managed area and medium with 29-62% managed.	**Land Cover/Land Use for RI 2003/2004**
18. Percentage of forested land in the 500-foot area around the wetland. Forested land including brushland, forested upland and forested freshwater wetlands. Levels include low with 8-16% in forested land and medium with 29-65% forested.	**Land Cover/Land Use for RI 2003/2004**
19. Distance to the nearest freshwater wetland, which may be contiguous to the coastal wetland, not contiguous but within ¼ mile, or farther than ¼ mile from the coastal wetland.	**Wetlands of RI**
20. Presence of another salt marsh within ½ mile of the described marsh.	**Wetlands of RI**
21. Access to the coastal wetland is not limited, as compared to restricted access.	**Public Shoreline Access: shore**

Table 3. Overall Percentages (%) for values of RI salt marshes

1. Type	Meadow	24
	Fringe	76
2. Size	closest to 1 acre	0
	closest to 5 acres	12
	closest to 17 acres	48
	closest to 65 acres	40
3. Spartina Coverage	Low	13
	Medium	17
	High	70
4. % Low marsh	around 4% Low marsh or less	100
	10% Low marsh or more	0
5. Brackish wetland present	brackish yes	36
	brackish no	64
6. ESS wetland present	ESS wetland yes	60
	ESS wetland no	40
7. Significant Tidal Flats	significant tidal flats yes	76
	significant tidal flats no	24
8. Eelgrass adjacent	Eelgrass – adjacent yes	20
	Eelgrass – adjacent no	80
9. Adjacent water body	Coastal river	32
	Coastal embayment	16
	Salt pond	52

10. Tidal restrictions	No tidal restrictions	76
	medium tidal restrictions	20
	Severe tidal restrictions	4
11. Pools	Pools 1% cover	20
	Pools 10% cover	32
	Pools 20% cover	48
12. Pannes	Pannes 1 % cover	20
	Pannes 5% cover	32
	Pannes 15% cover	48
13. Sub-tidal channels	Channels present	92
	Channels absent	8
14. Creeks	Creeks present	36
	Creeks absent	64
15. Buffer	No Buffer	68
	Shrub buffer	4
	Forest buffer	28
16. Developed land	developed land - none	16
	Low % developed land (<30%)	16
	Medium developed land (30-65%)	40
	High developed land (>65%)	28
17. Agricultural land	Land agricultural – none	52
	Land agricultural – low (<25%)	20
	Land agricultural – Medium (25-65%)	20
	Land agricultural – high (>65%)	8
18. Forested land	no forested land	16
	Land forested – low (<25%)	40
	Land forested – Medium (25-65%)	28
	Land forested – high (>65%)	16
19. Freshwater wetland nearby	Freshwater wetlands adjacent	92
	Freshwater wetlands Within ¼ mile	100
	Freshwater wetlands Over ¼ mile	100
20. Other salt marsh nearby	Other salt marsh – within ½ mi. yes	100
	Other salt marsh – within ½ mi. no	0
21. Public access	Public access restricted	40
	Public access available	60

Appendix II. Vegetation Composition Maps

Barrington
- 100 Acre Cove
- 100 Acre Cove "the tongue" and Barrington River Island
- Palmer River-West
- Rumstick Point

Charlestown
- East Beach
- Breachway
- Town Beach
- Quonochontaug
- Winnapaug

East Matunuck
- Potter Pond
- Succotash Marsh

Jamestown
- Fox Hill Marsh
- Marsh Meadows

Little Compton
- Quicksand Pond
- Briggs Marsh

Middletown
- Third Beach

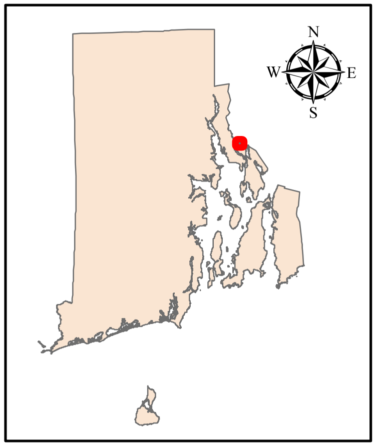
Narragansett
- Chafee Wildlife Refuge-North
- Chafee Wildlife Refuge-South
- Bluff Hill Cove
- Galilee Wildlife Refuge

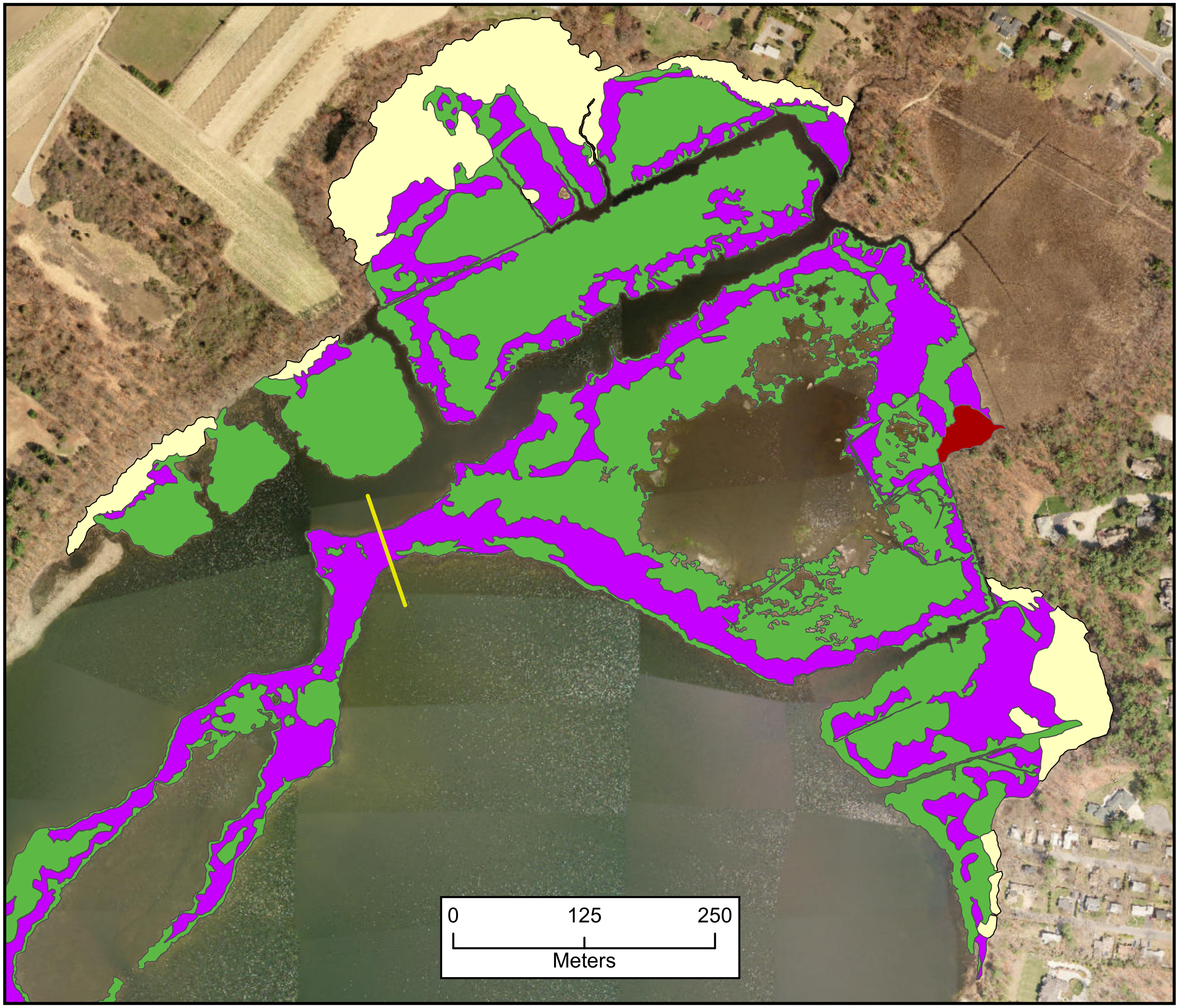
Tiverton
- Seapowet Marsh

Warren
- Kickemuit
- Palmer River-East

Warwick
- Potowomut Marsh

Hundred Acre Cove, Barrington, RI

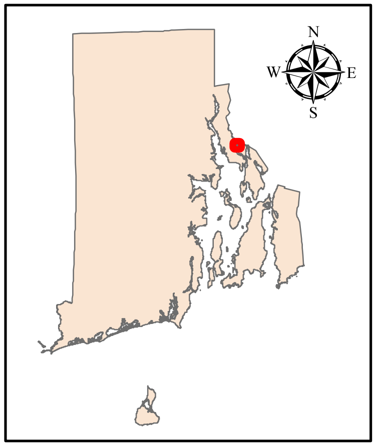




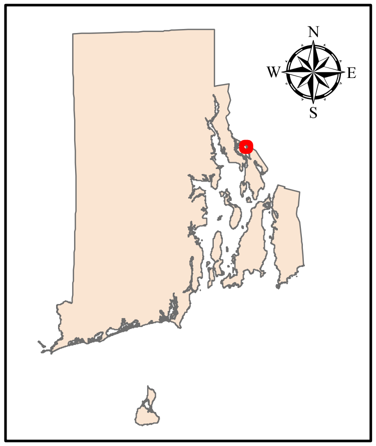
Vegetation Classes

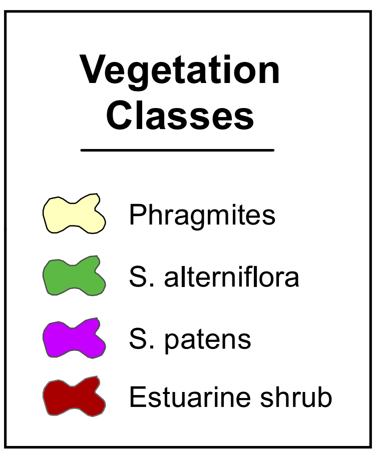
- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub

Hundred Acre Cove-Tongue & Barrington River Island, Barrington, RI

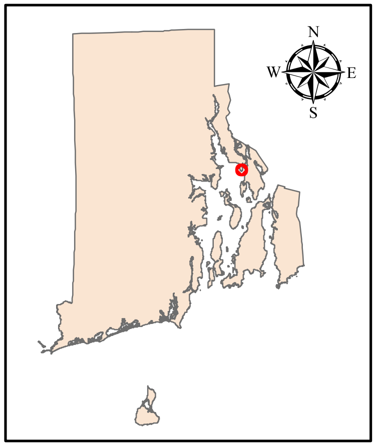


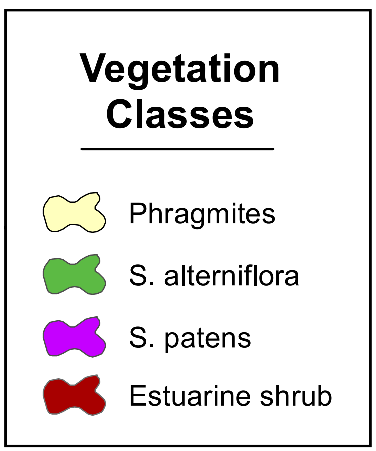
Palmer River-West, Barrington, RI

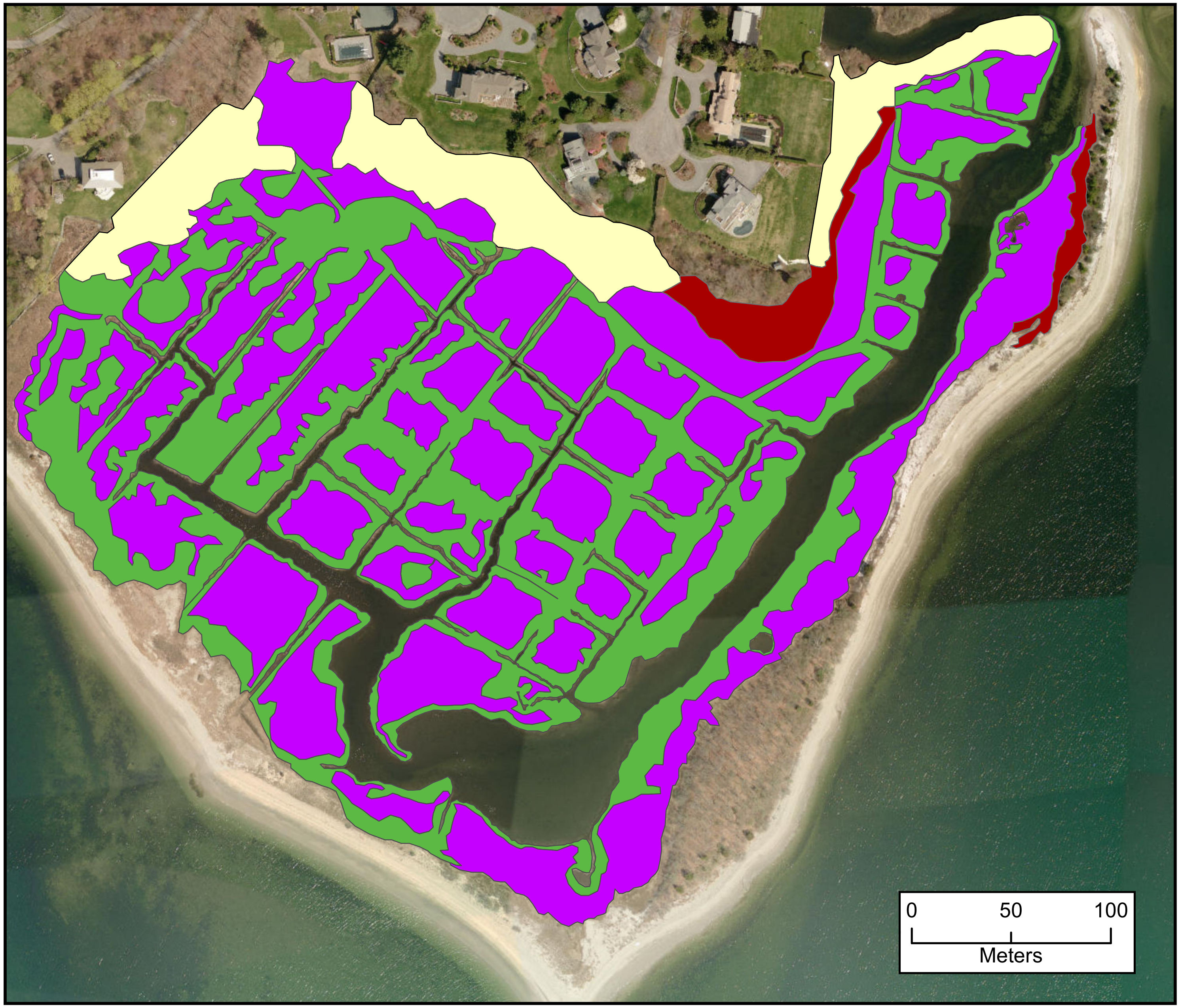




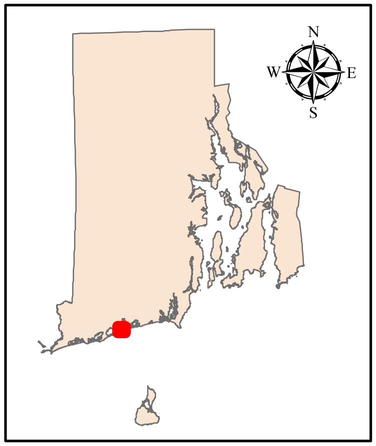
Rumstick Point, Barrington, RI





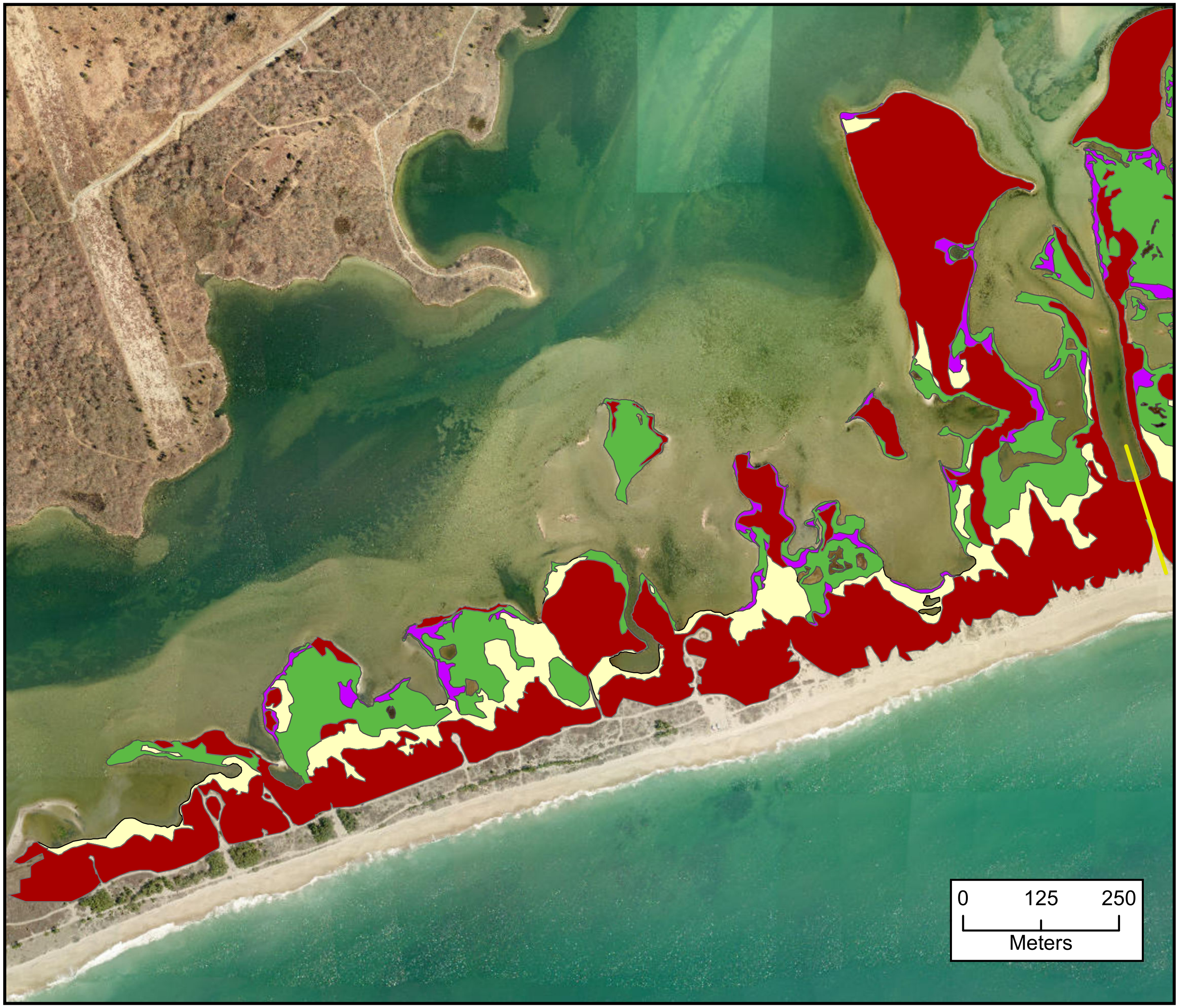


East Beach, Charlestown, RI

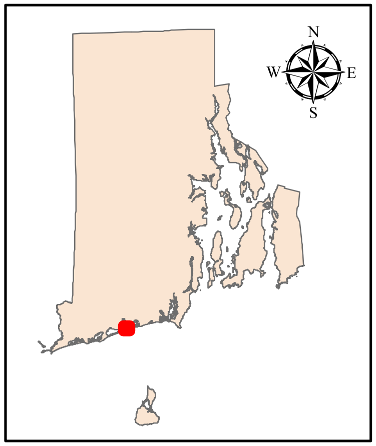


Vegetation Classes

- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub



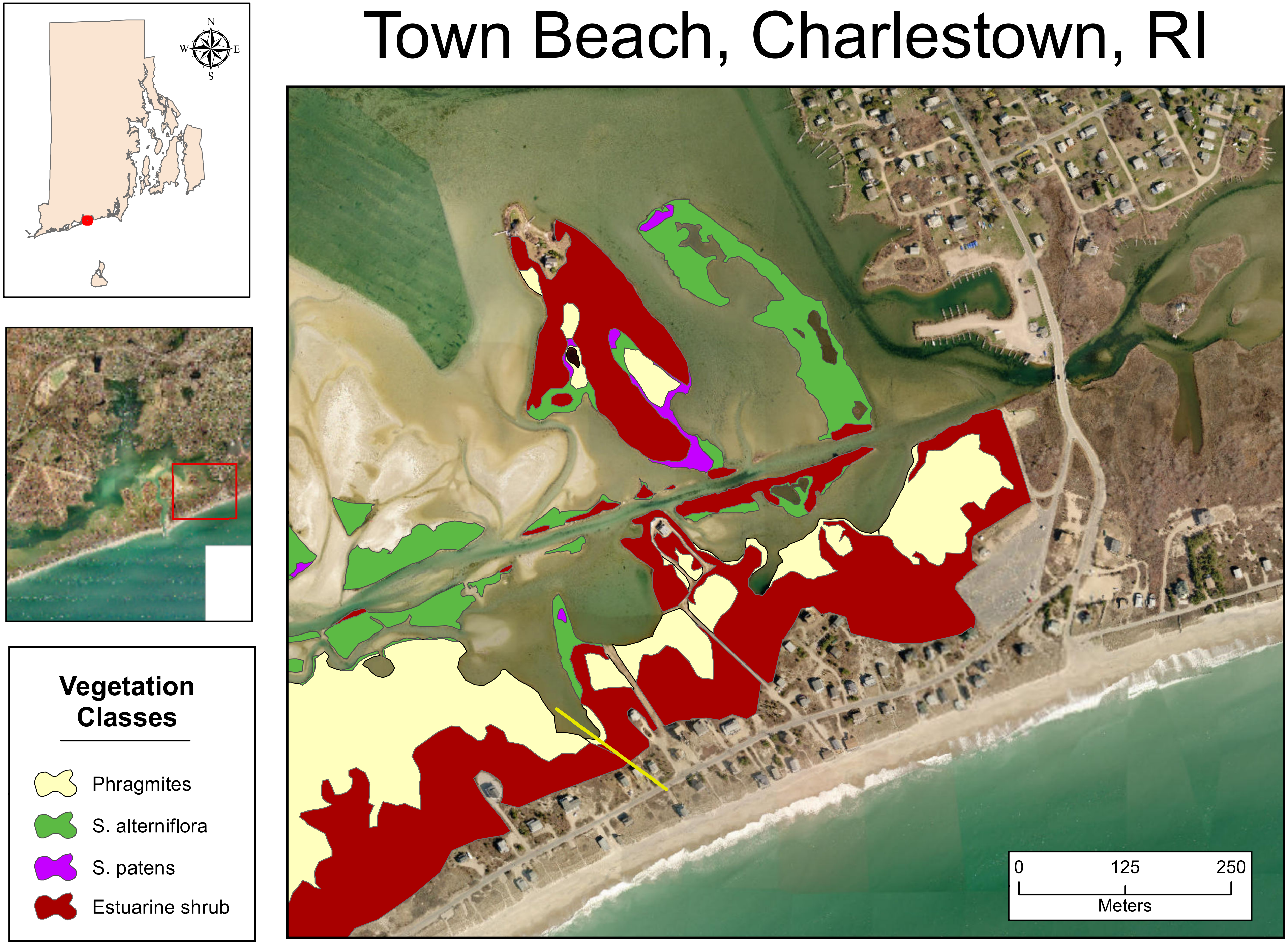
Breachway, Charlestown, RI

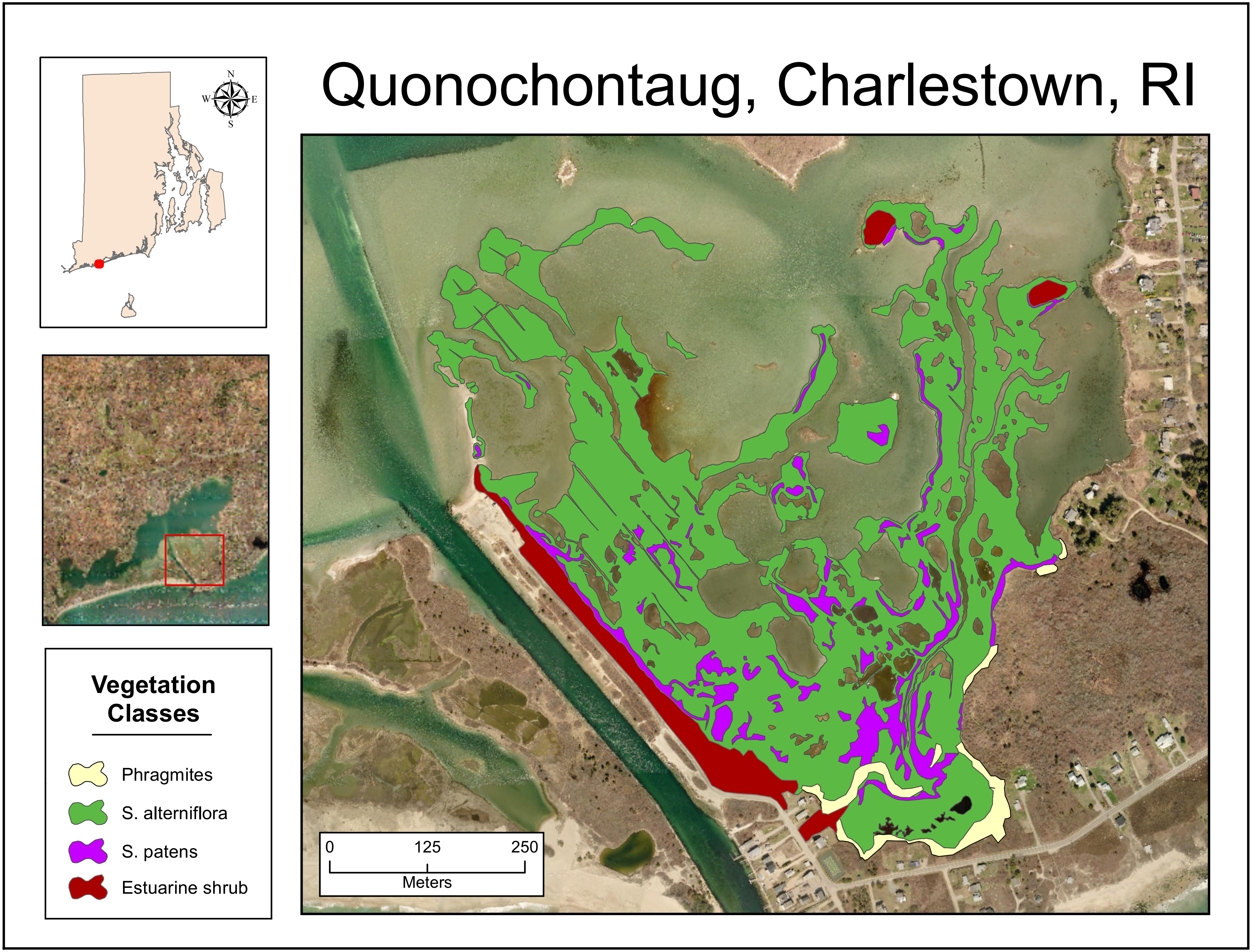


Vegetation Classes

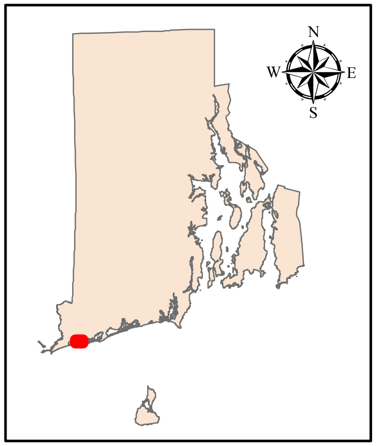
- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub

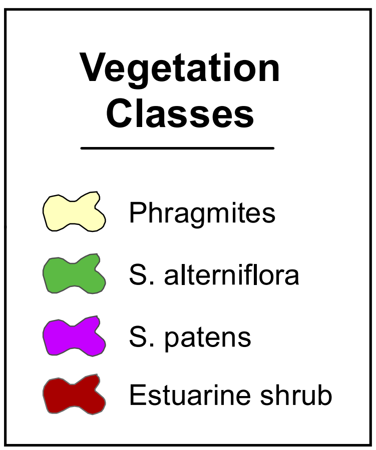
Town Beach, Charlestown, RI

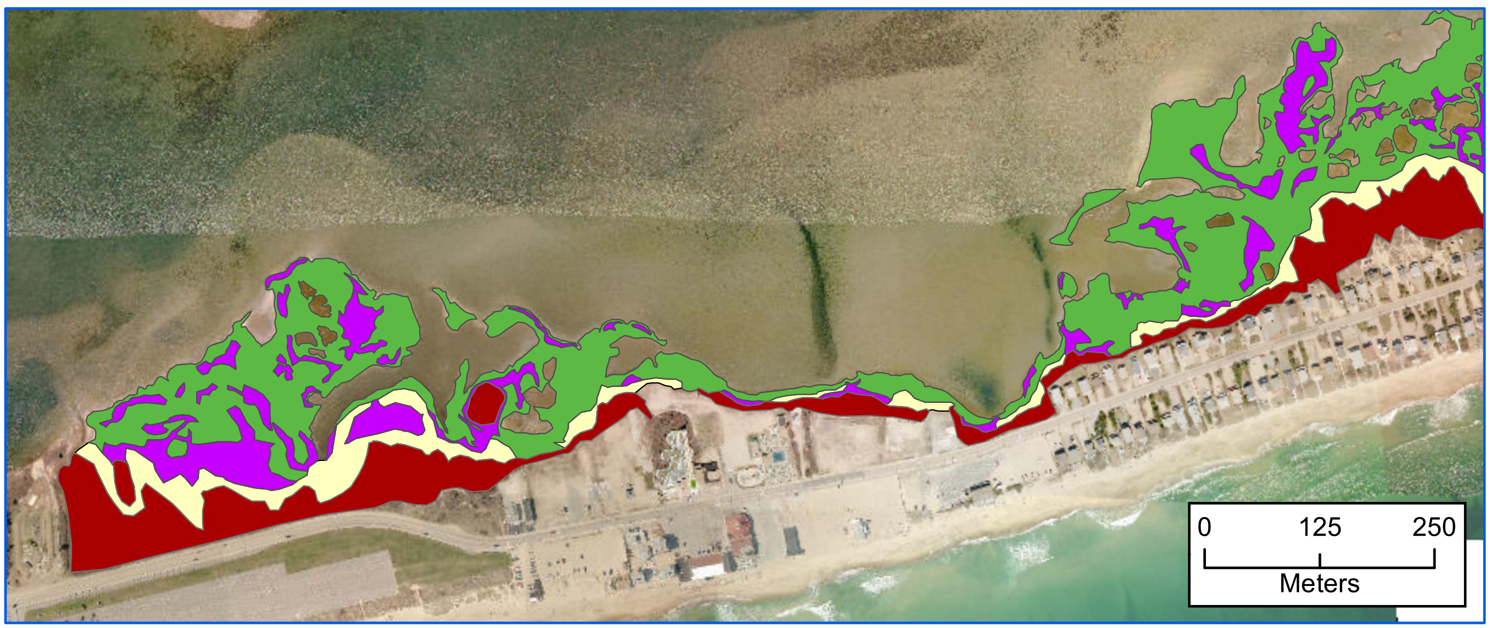



Quonochontaug, Charlestown, RI
N
W
E
S
Vegetation Classes
Phragmites
S. alterniflora
S. patens
Estuarine shrub
0
125
250
Meters

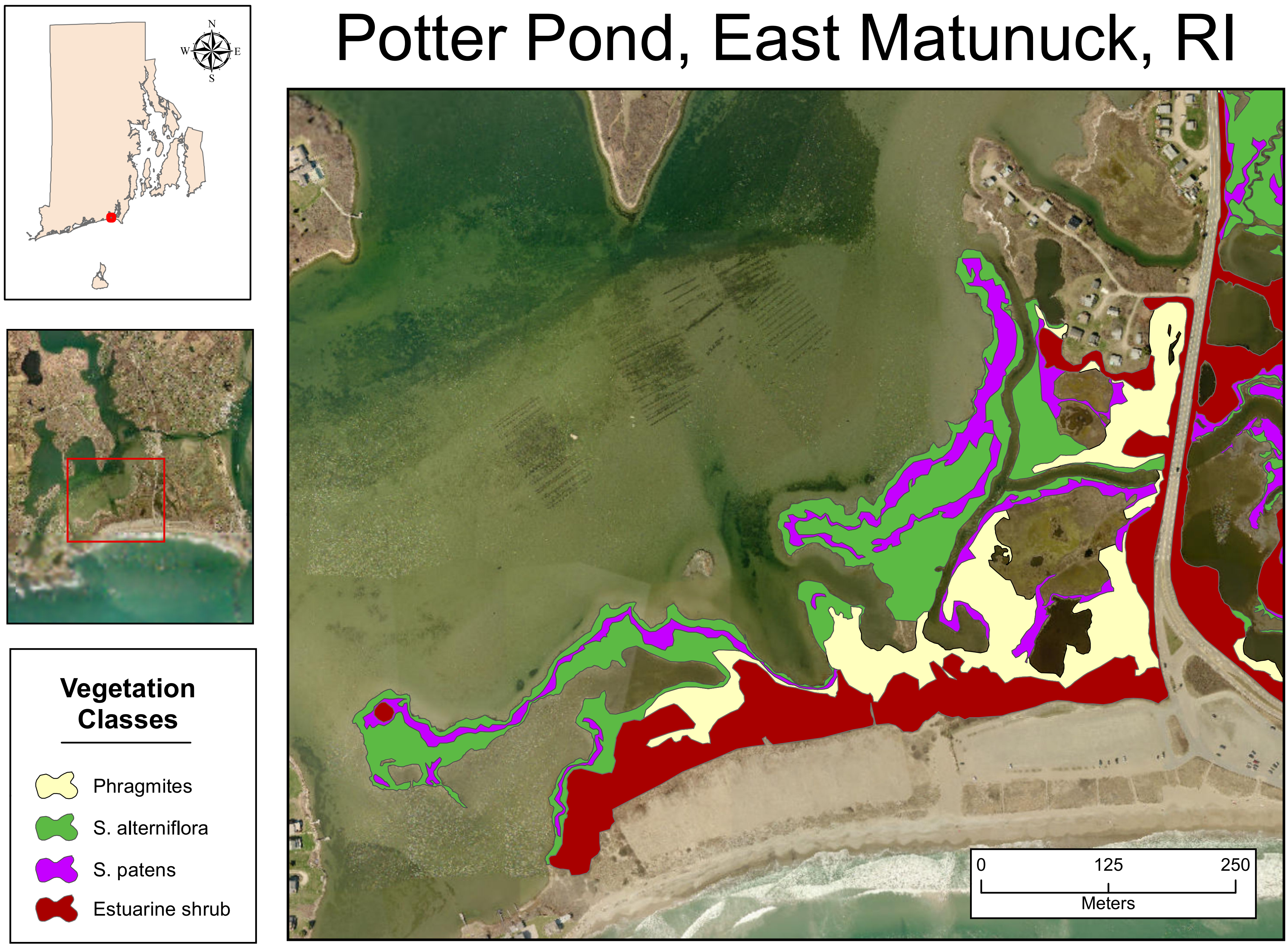
Winnapaug, Charlestown, RI



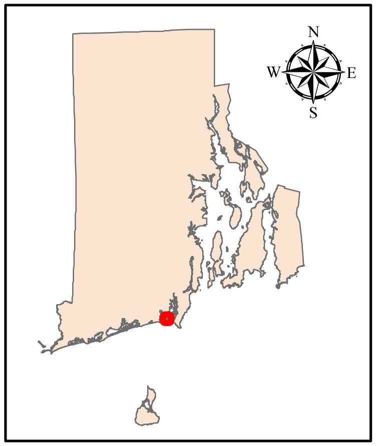


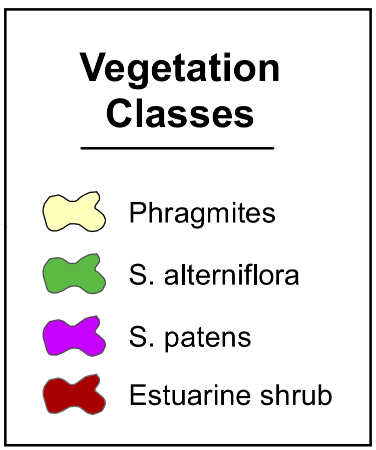


Potter Pond, East Matunuck, RI

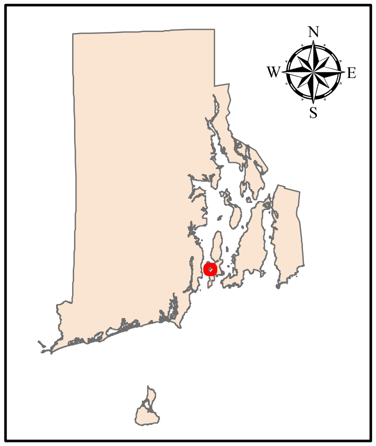


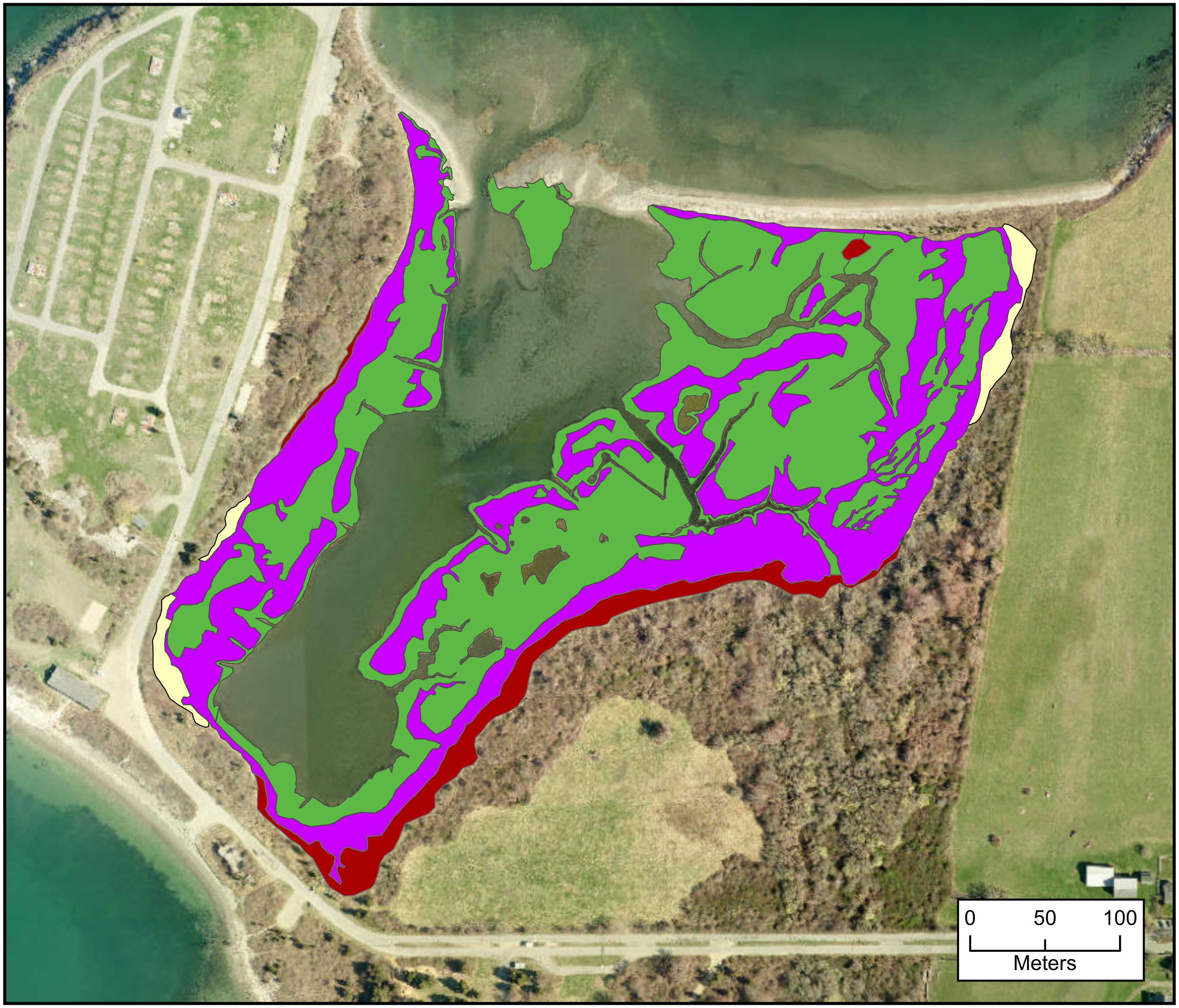
Succotash Marsh, East Matunuck, RI





Fox Hill Marsh, Jamestown, RI

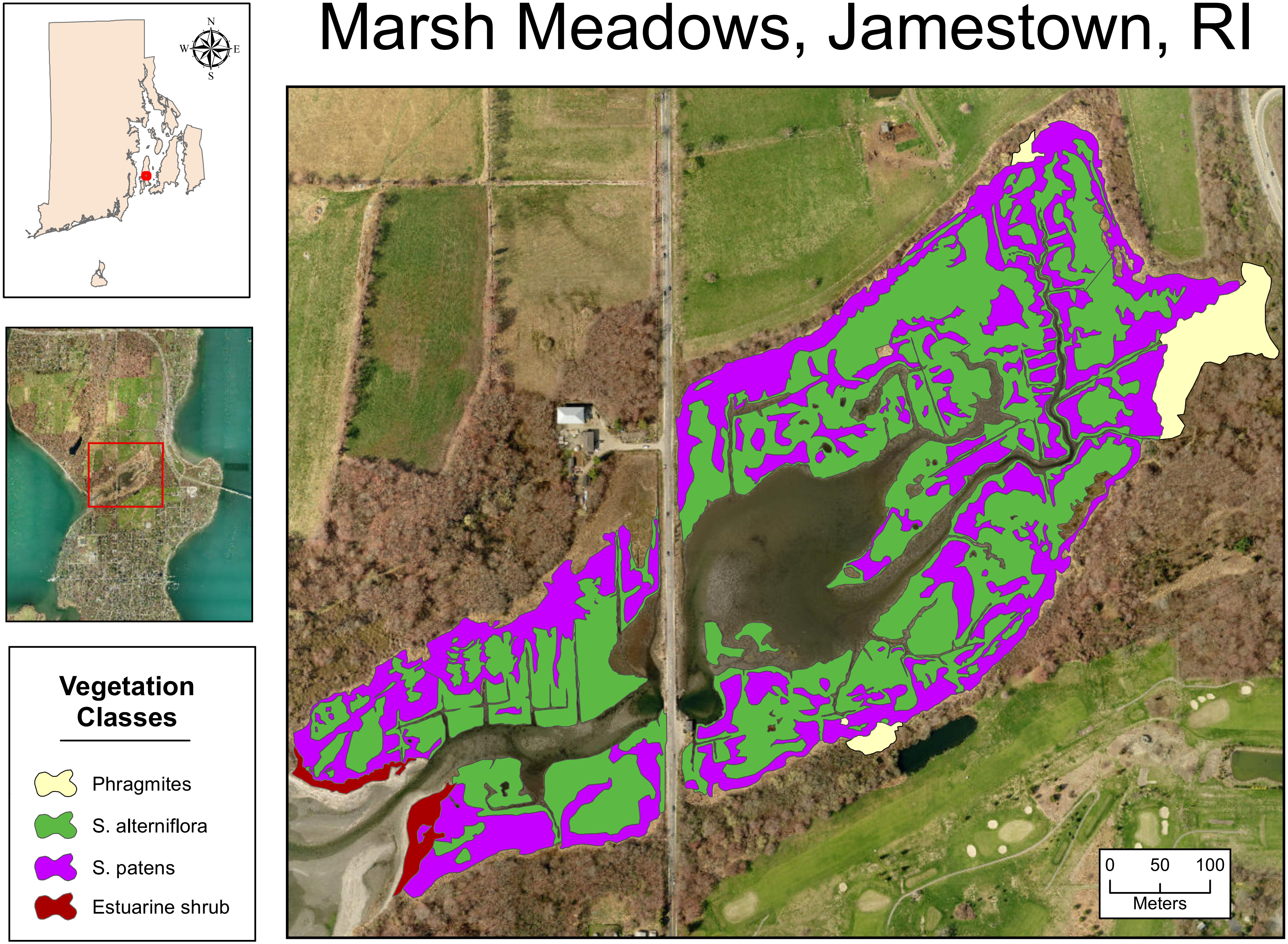




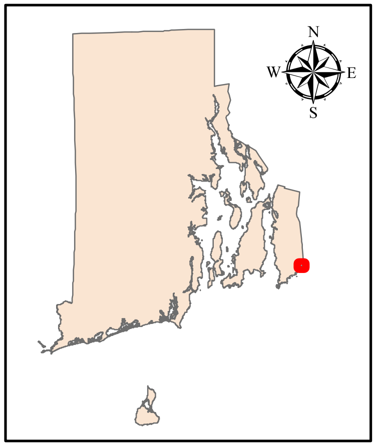
Vegetation Classes

- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub

Marsh Meadows, Jamestown, RI



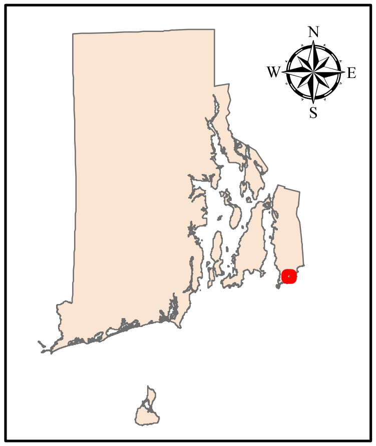
Quicksand Pond, Little Compton, RI



Vegetation Classes

- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub

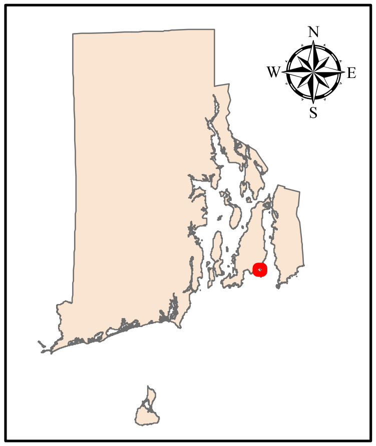
Briggs Marsh, Little Compton, RI

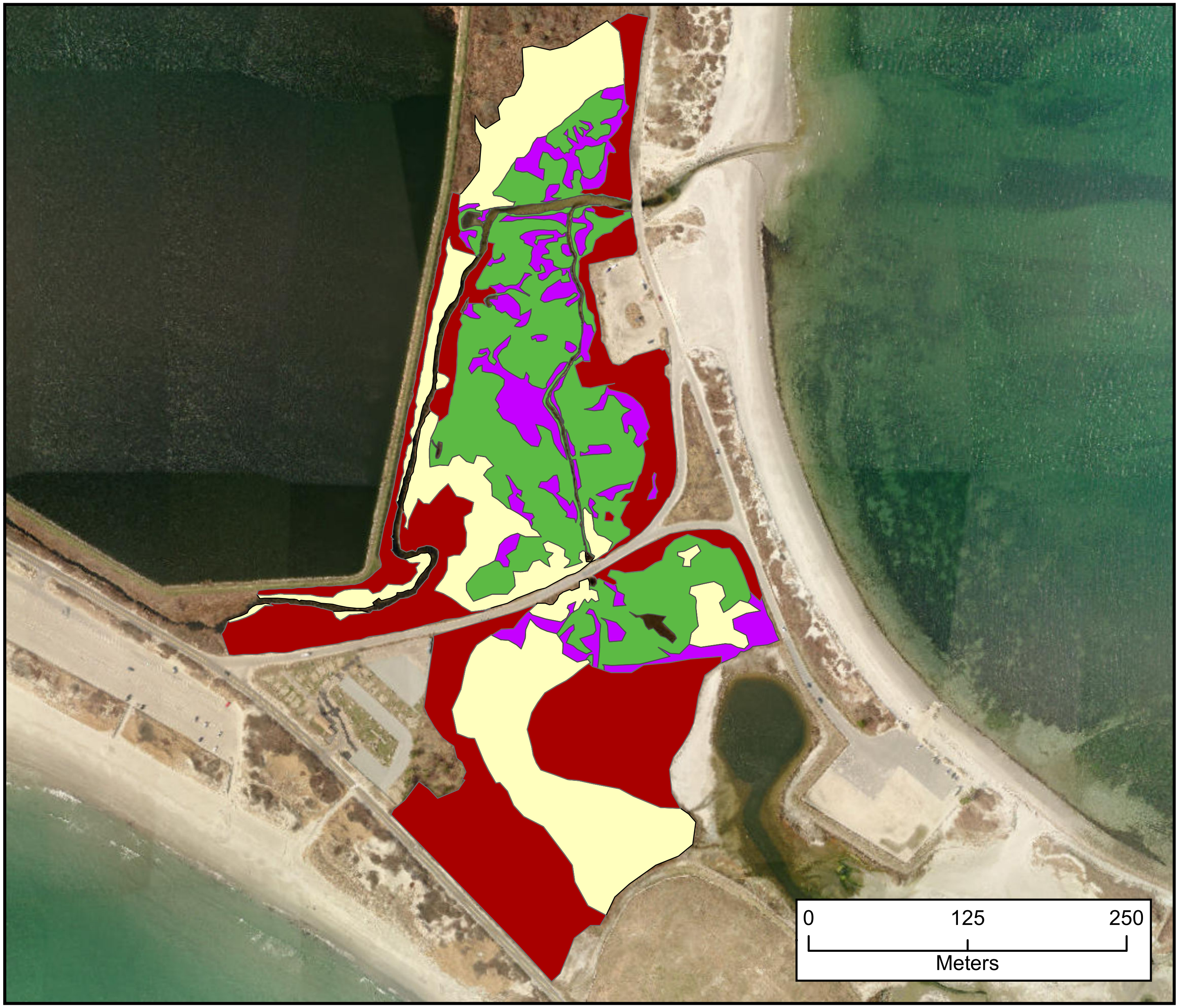


Vegetation Classes

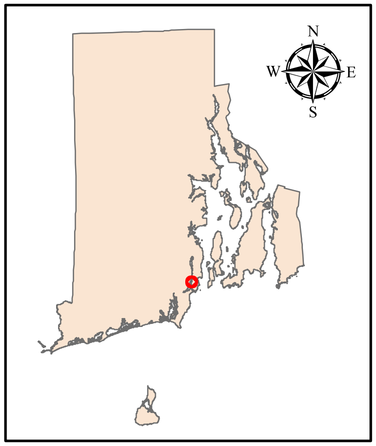
Phragmites
S. alterniflora
S. patens
Estuarine shrub

Third Beach, Middletown, RI



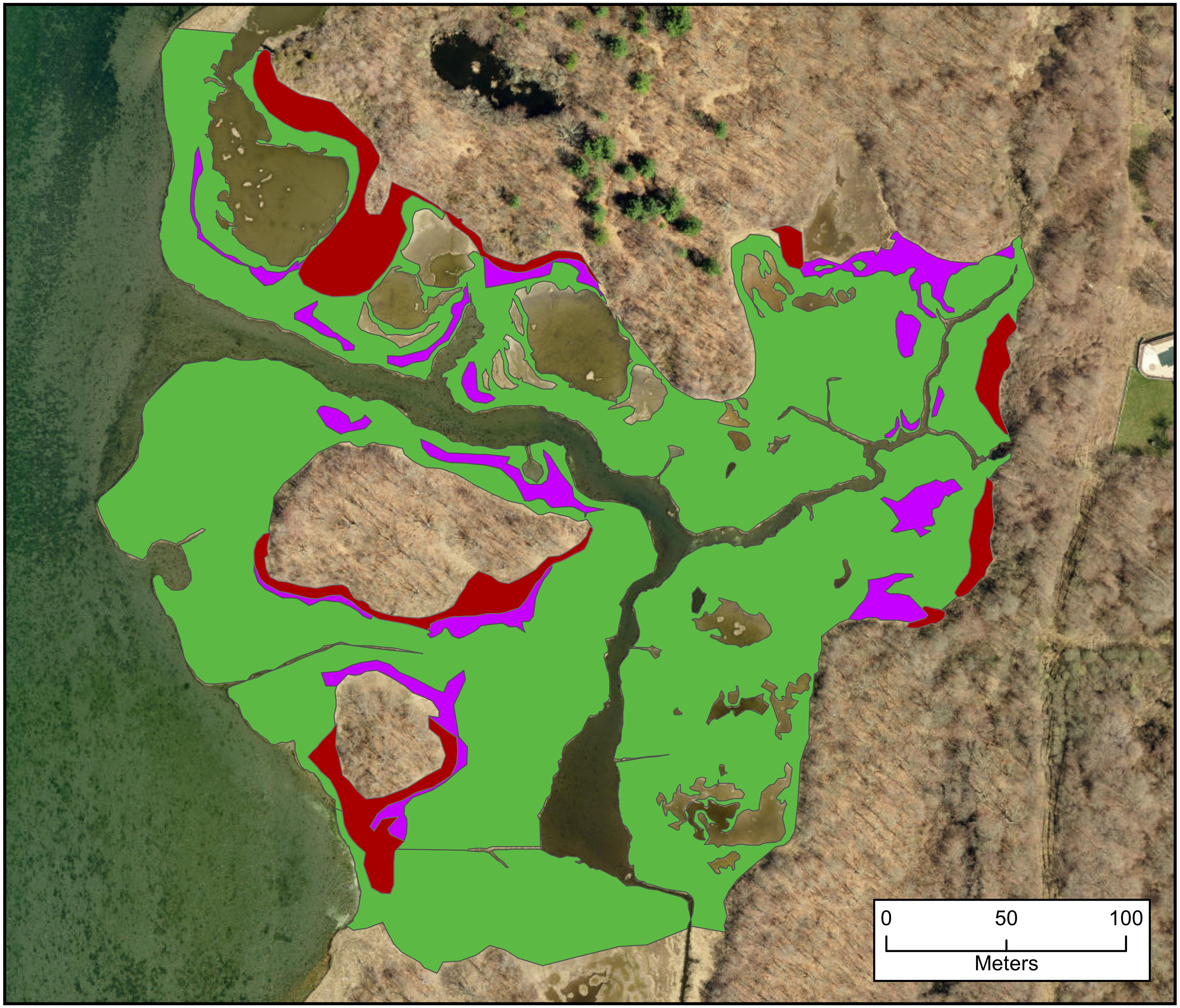


Chafee Wildlife Refuge-North, Narragansett, RI

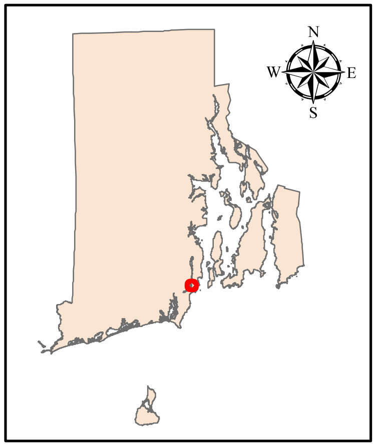


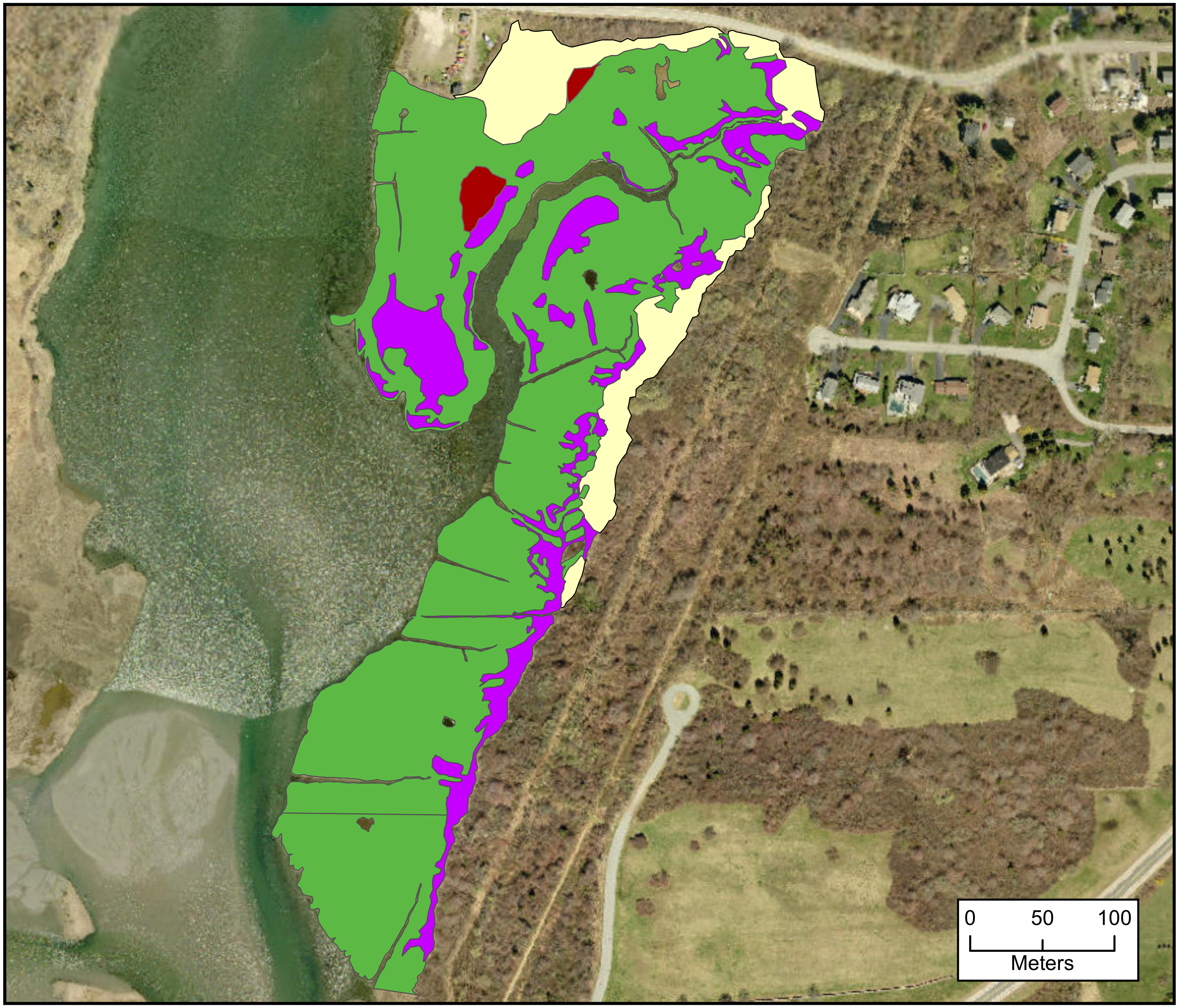
Vegetation Classes

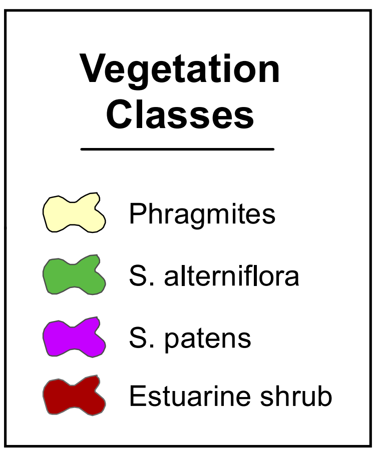
- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub



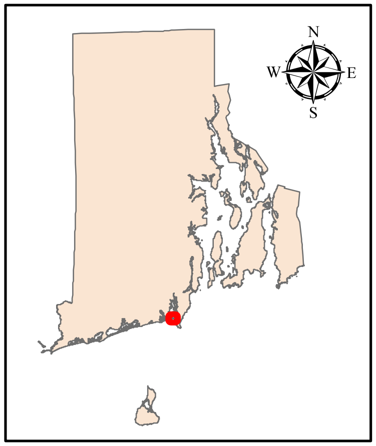
Chafee Wildlife Refuge-South, Narragansett, RI

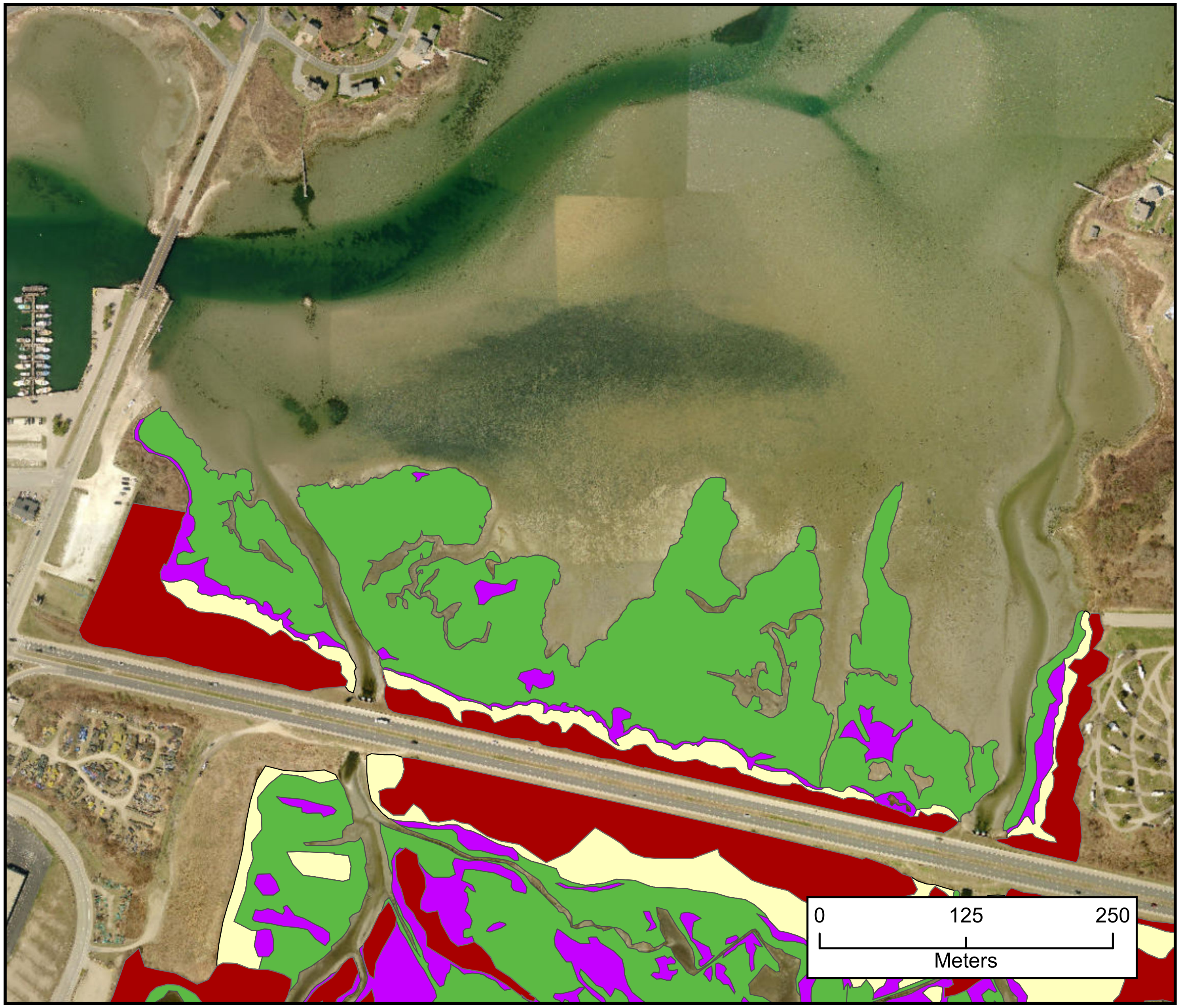


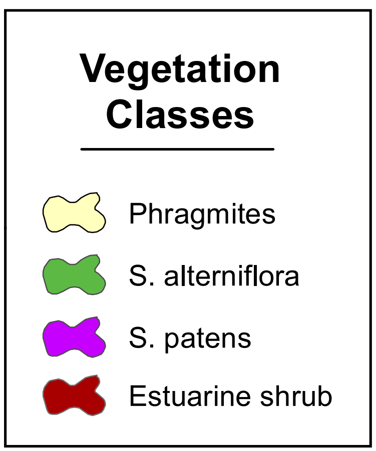




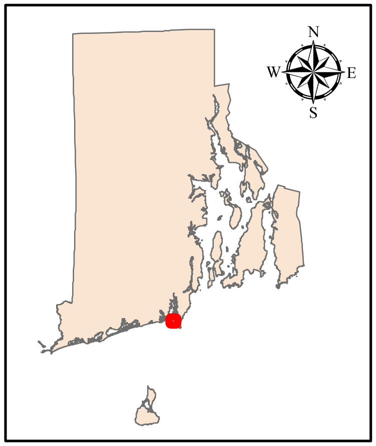
Bluff Hill Cove, Narragansett, RI





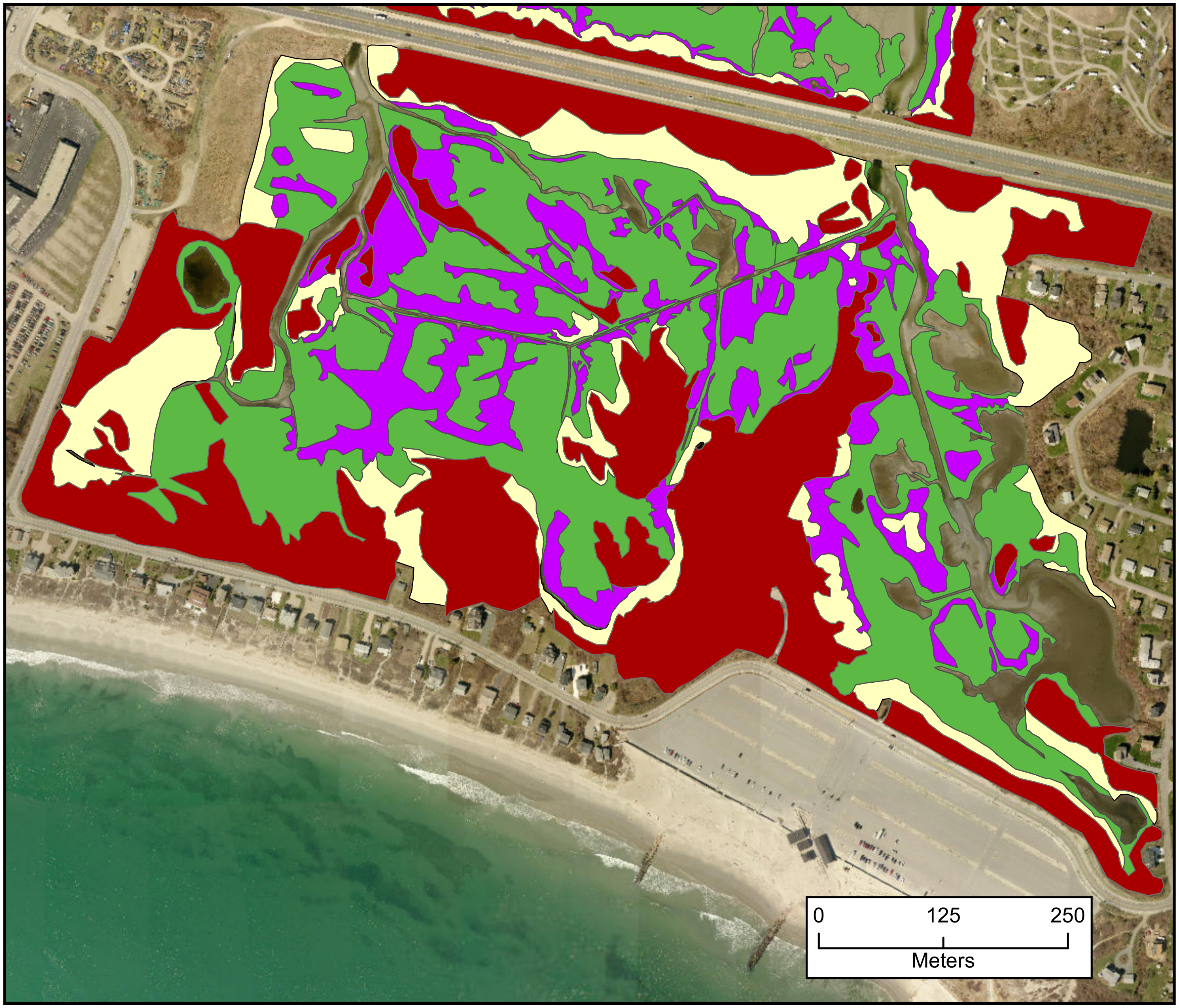


Galilee Wildlife Refuge, Narragansett, RI

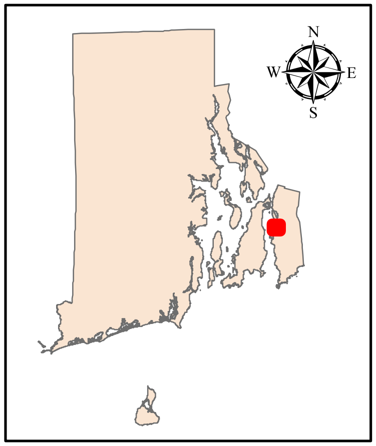


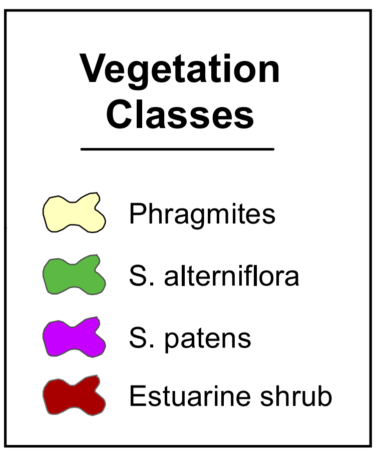
Vegetation Classes

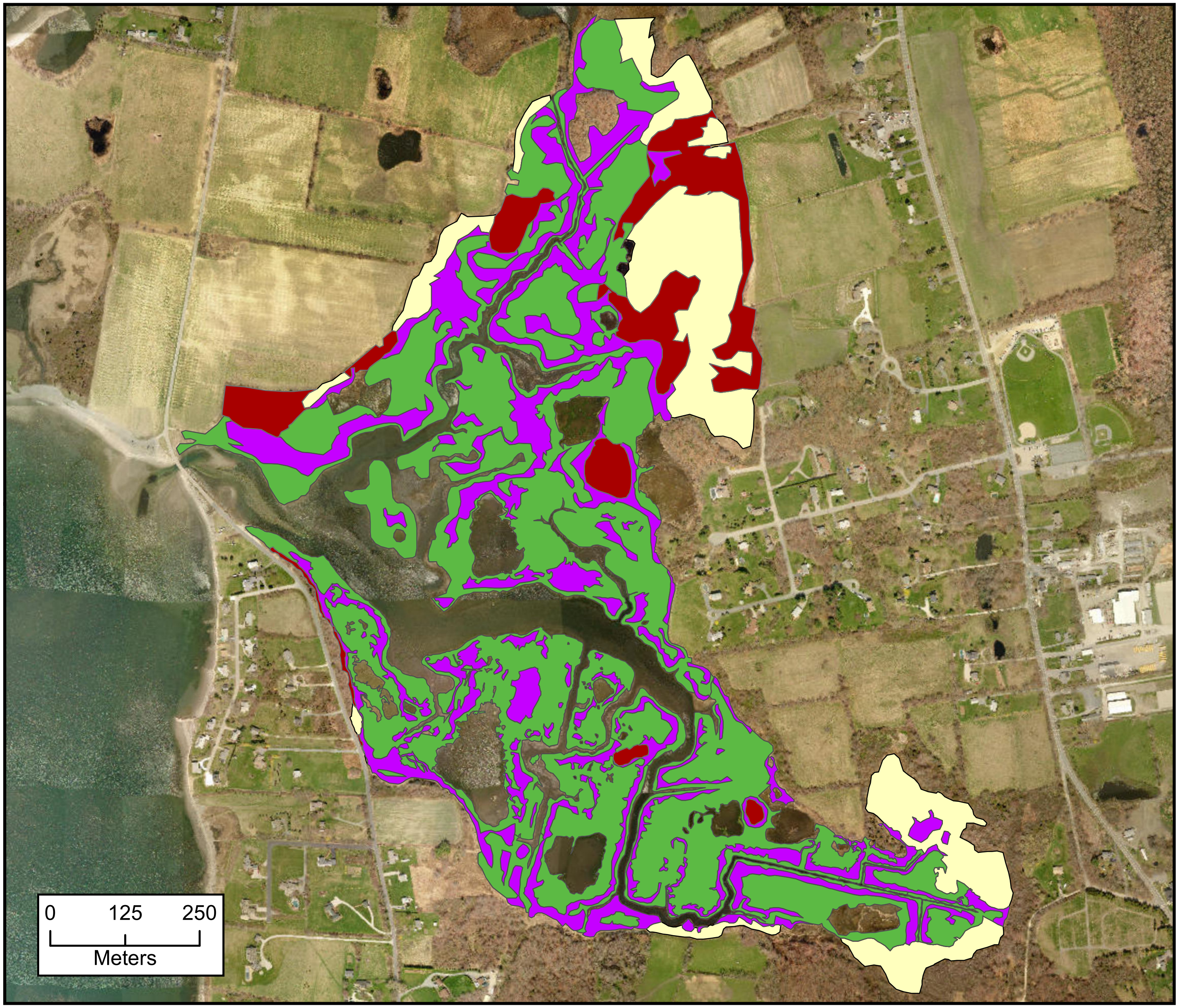
- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub



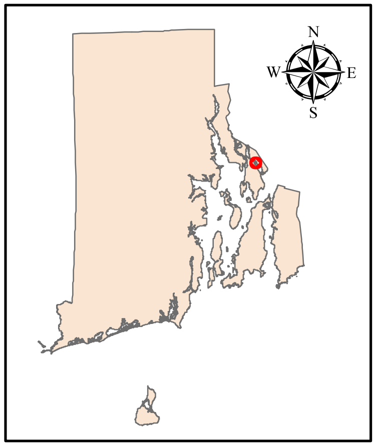
Seapowet, Tiverton, RI

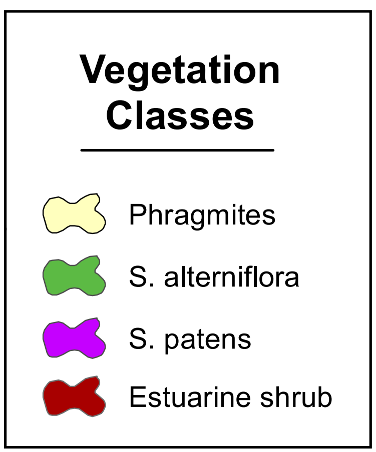




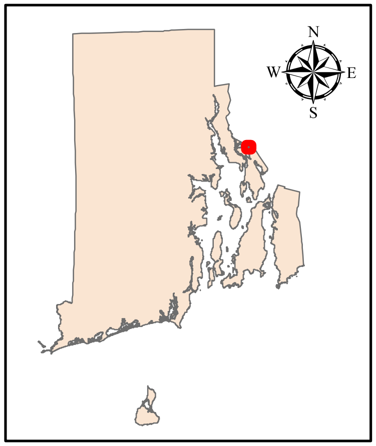


Kickemuit, Warren, RI





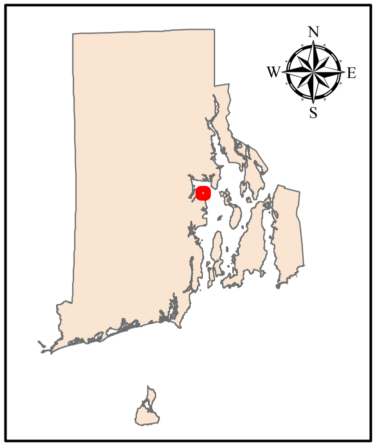
Palmer River-East, Barrington, RI



Vegetation Classes

- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub

Potowomut Marsh, Warwick, RI



Vegetation Classes

- Phragmites
- S. alterniflora
- S. patens
- Estuarine shrub