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The Herring River Estuary: Understanding Salt Marsh Restoration and How to Predict the Response of Phragmites australis to Increased Tidal Flow

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An aerial view of a portion of the Herring River estuary in Cape Cod, MA. On the left side of the photo the dike that restricts tidal flow to the system is clearly visible.

INTRODUCTION

This paper was prepared for the Herring River Restoration Committee, a multi-agency group (National Park Service, US Fish and Wildlife Service, NOAA Restoration Center, Natural Resources Conservation Service, Massachusetts Division of Ecological Restoration, Town of Wellfleet, Town of Truro, Association for the Preservation of Cape Cod) providing technical guidance during the planning stages for restoration of the 1,100 acre Herring River estuary, located mostly within Cape Cod National Seashore. Restoration of this highly degraded estuary is complex with numerous ecological and societal issues to address. To be successful the restoration must be carefully planned and monitored to assure that the project is environmentally beneficial and that societal concerns are accommodated. The restoration managers will adopt an adaptive management approach, whereby changes to the system will be carefully monitored. If unexpected changes occur then managers will address the issue or adapt the restoration process. This paper focuses on one adaptive issue – predicting the restoration response of *Phragmites australis*, an invasive plant species.

STUDY AREA AND BACKGROUND

The Herring River estuary is a disturbed salt marsh ecosystem at Cape Cod National Seashore which has been diked since 1909 (Fig. 1). The estuary was diked at the mouth to help control mosquitoes and create arable land. Further ditching and stream channelization followed to drain the estuary. The effects of these dramatic changes have been felt throughout the estuary. Extensive amounts of salt marsh habitat have been replaced with freshwater wetlands or dry deciduous forest (Roman et al. 1995). The large reduction in tidal exchange has greatly degraded water quality in the system. High acidity, low dissolve oxygen levels, and high counts of fecal coliform have all become issues within the estuary (Portnoy 1991; Portnoy 1999). Degraded water quality has led to fish kills and the closing of the shellfish beds near the dike (Portnoy and Allen 2006).

As we have begun to realize the important benefits of salt marsh ecosystems and the negative effects that diking has had on these systems, efforts have been ongoing to restore salt marshes along the coast (Roman and Burdick 2012). Since the 1980s the National Park Service, in cooperation with many partners, has been engaged in efforts to restore the now impacted 1,100 acre estuary by re-introducing tidal flow. The restoration is designed to restore tidal flow and salinity throughout the floodplain. The restoration would have a number of benefits including restoration of natural sedimentation to help counter wetland subsidence and sea level rise, improvement of water quality which would be beneficial to the estuarine fauna, and reestablishment of native salt marsh plants and animals.

Despite these strong incentives there are also many concerns involved with restoring such an intricate system. Ecosystems are extremely complex and restoration efforts cannot be done successfully without proper planning and management. Since the Herring River estuary has been diked for over a century any restoration would cause significant changes to the now established system. A few rare species of concern for the Commonwealth of Massachusetts, such as the American and least bitterns, the eastern box turtle, and the water willow stem-borer, occur in the freshwater wetland portions of the Herring River and the restoration could cause potential habitat reductions. Upstream reaches are expected to remain as freshwater and may provide continued habitat for these and other freshwater species, but it is important to consider the potential impact of the restoration project on these species.

Additionally the public has many concerns about the proposed restoration as reviewed by Portnoy (2012). Since the dike was built there has been some development within the floodplain -- private residences, domestic wells and septic systems, low-lying roads, and a golf course. Also much of the once marsh-dominated Herring River estuary now supports woody and shrubby vegetation. When flooded with high salinity waters these salt-intolerant plants will die and may be viewed as aesthetically unpleasing. Further, there is concern that productive shellfish areas located downstream of the dike may be impacted by fine sediments transported after the restoration occurs. Given these concerns, managers planning the restoration must strive to understand as best as possible the responses to the restoration with sound science. They use this information to communicate and work with the public to gain their approval by helping to resolve and mitigate their concerns.

PROPOSED RESTORATION DESIGN

Flow between Wellfleet Harbor and the Herring River estuary is currently restricted by two tide gates (each 1.8m (6ft) wide) and one sluice gate (1.5m (5ft) wide x 0.6m (2ft) high; Fig. 2). The tide gates do not allow any input of seawater into the estuary during flood tides, but allow for drainage of water during the ebb. The sluice gate allows a minimum of seawater inflow and outflow during the ebb. Mean tidal range immediately downstream of the dike is about 3.5m (11.5ft) and is restricted to just over 0.5m (1.5ft) upstream of the dike and gates. Extensive hydrologic modeling has been conducted to predict the exchange of tidal waters under several wider and less restrictive water control structure openings. The analysis presented in this report has focused on the proposed modeled alternative of replacing the existing gates with an opening that is 47.5m (156ft) wide with an adjustable height to a maximum of 3m (10ft). However, it is noted that the actual restoration process would implement small incremental openings over time. It would take many years before the structure was actually opened to the maximum 3m (10ft) opening. The modeled restoration would also involve removing additional infrastructure barriers throughout the estuary including roads and an old railroad grade, which also impede water flow.

ADAPTIVE MANAGEMENT AND PURPOSE OF THE STUDY

The overall goal of this restoration is to change the estuary back to a salt marsh ecosystem. However, with a restoration this extensive and complicated there is much uncertainty, especially because the system has been degraded for so many years. There are concerns about high sulfide levels, which are toxic to plant nutrient uptake and growth, if standing water occupies much of the restored system and creates anoxic conditions (Roman et al. 1995, Portnoy 1999). These areas could become unvegetated or barren. There are also concerns that extensive areas could become occupied with invasive species. These concerns facilitate the need for adaptive management and proper planning. There are multiple aspects to adaptive management but the one focused on for this analysis has been the adaptive management of vegetation in the estuary.

Predicting the change in species dynamics of the estuary can help managers determine if restoration is proceeding accordingly or if they need to make changes. If the system is not responding as expected, then pre-determined adaptive management actions can be taken. To assist with this process this project focuses on predicting change in the distribution of a particularly problematic invasive species- *Phragmites australis*. In trying to predict the possible outcomes of the invasive reed grass, *Phragmites*, this project will evaluate how this species might react to restoration and what can be done to limit and diminish the spread of this invasive as the restoration proceeds.

FOCUS PLANT: *Phragmites australis*

The analysis focuses on the invasive reed grass *Phragmites australis*. *Phragmites* has native and non-native haplotypes (Saltsonstall, 2003). The invasive non-native haplotype was introduced in the 19th century and has been expanding rapidly across the United States, especially invading human-disturbed salt marshes (Burdick and Konisky 2003, Chambers et al. 2012). The non-native haplotype often forms large monocultures which can greatly reduce plant biodiversity, and alter animal communities, soils, and hydrologic features (Chambers et al., 2012). The vigorous clonal growth of *Phragmites* via rhizome extension allows this plant to displace other species and inhibit the growth of its competitors. *Phragmites* can also avoid physical and biological stress by spreading clonally while still accessing resources from more favorable locations (Burdick and Konisky, 2003). It is the non-native type that has invaded the Herring River estuary under the tide-restricted regime. Because of the significant structural changes and the loss of biodiversity to the system that *Phragmites* causes, there has been significant interest in preventing its spread and reducing current stands. This is why this paper focuses on predicting its responses to the proposed restoration. In contrast while many studies point to the low avian support function of *Phragmites* marshes (e.g. Benoit and Askins 1991), some note that *Phragmites* stands can provide favorable nesting habitat for some species, including rare species (e.g., Parson 2003). Given these studies the loss of bird habitat remains a valid concern that should be considered in the planning the adaptive management of *Phragmites.*

Phragmites australis does not tolerate high saline conditions, thus it is expected to be limited in distribution as restoration proceeds. However, since there will still be low salinity areas throughout the estuary there is a risk of *Phragmites* spreading into and invading the low salinity areas of the marsh. Additionally because the restoration will proceed incrementally there are concerns that *Phragmites* will have the time to slowly spread upstream as salinity levels slowly increase in the system. The data analysis and mapping of this project serve to identify the areas most optimal for *Phragmites* spread and the areas least suitable for the spread of *Phragmites*. This information will hopefully serve to support the development of adaptive management strategies for *Phragmites* control.

SUITABILITY MODELING

To predict the changes in *Phragmites* distribution in the Herring River estuary a suitability analysis was conducted to find the most suitable regions for mature *Phragmites* growth and *Phragmites* seed dispersal and germination. A suitability analysis considers the appropriateness of an area for a specific activity and can also be used to determine the most optimal areas for that activity. In this case a model was designed to assess the suitability of the estuary for *Phragmites* growth and to determine the most likely areas where *Phragmites* would occur at the 3m (10ft) restoration alternative.

ArcGIS software was used for the suitability analyses, which allowed for the interpretation and presentation of geographical data about the system. The first step was to decide on the main factors that control the spread of *Phragmites* in the Herring River estuary. Literature review helped to establish salinity tolerance, flooding tolerance, sulfide toxicity, access to nutrients, and opportunity to spread or establish as the most important factors governing control (Burdick et al. 2003). However, it soon became apparent that the available data from the National Park Service was not adequate for inclusion of nutrients and sulfide in the analysis. While all important factors are discussed in this essay, only salinity tolerance, flooding tolerance, and opportunity to spread or establish, were included in the analysis. While these factors can’t fully represent the suitability of an area for *Phragmites*, they are some of the most important factors in determining the growth of *Phragmites*. In fact research has shown that salinity tolerance and flooding are the main physical stressors for *Phragmites* (Burdick et al. 2003). As a result the final analysis should be a useful tool for managers interested in adaptive management of this invasive species.

To conduct the suitability analysis the relevant factors for *Phragmites* growth had to be organized and weighed to find the suitability of all areas in the estuary to *Phragmites* growth. Below is a description of the controlling factors for Phragmites growth and a detailed guide into how the GIS analysis of the suitability modeling was conducted. Also below is a summarized table of the weights used for each of those factors.

***PHRAGMITES* SUITABILITY CONSIDERATIONS**

An extensive literature review on *Phragmites* tolerances was used to derive values for the important controlling factors for *Phragmites* spread. The literature review indicated that mature *Phragmites* will show the most vigorous growth at salinity levels below 20 ppt (Chambers et al. 2003, Burdick 2003). Additionally, mature plants are generally not competitive with other salt marsh species above 20 ppt (Buchsbaum et al. 2006). However it is not until 25 ppt that *Phragmites* has greatly diminished growth (Smith et al., 2009). *Phragmites* seeds and seedlings are more sensitive to salinity than mature plants. Studies show that seeds will establish best at 0-10 ppt, poorly between 10-20 ppt, and will not germinate above 20 ppt (Bart and Hartman 2003, Chambers et al. 2003).

In terms of flooding tolerance mature *Phragmites* is capable of spreading into a range of flooding levels as well, although it does have preferences. It shows generally poor growth in low marsh areas (Amsberry et al. 2000). *Phragmites* is not adapted to grow in the anoxic conditions that often develop in these low marsh, regularly flooded areas (Amsberry et al. 2000). However, *Phragmites* may establish in the low marsh by clonally spreading via rhizomes from plants established in the high marsh (Amsberry et al. 2000). *Phragmites* will also generally not spread into permanently flooded areas (Chambers et al. 2003).

For opportunity to spread and establish, threshold values were somewhat harder to quantify. The values of yearly *Phragmites* rhizome growth ranged from 1 m to 2 m based on the literature (Amsberry et al. 2000, Michigan Department of Environmental Quality [Michigan DEQ] 2008). *Phragmites* seedling dispersal distances were difficult to quantify based on the literature so an arbitrary distance of 25 meters was used in the analysis below. Since the diking of the Herring

River estuary, much of the system has converted into woody areas (Roman et al. 1995). Woody vegetation can impede the delivery of wracked-filled seeds and also impede seedling growth because this dense vegetation limits light (Smith 2007).

Sulfide tolerance also affects *Phragmites* growth and the plants will not thrive at sulfide levels above 375 µM (Chambers et al. 2003). However, sulfide was not included in this modeling exercise because only limited sulfide data were available for the Herring River system.

Phragmites development depends on high levels of nitrogen being incorporated into its biomass as it grows (Meyerson et al. 2000). Human development of estuaries often inputs high amounts of nitrogen into the system which has in many cases helped to promote development of *Phragmites* (Silliman and Bertness, 2004; King et al. 2007). A quantifiable analysis of the different nitrogen inputs into the system has not been done, but it is expected that the adjacent golf course as well as suburban development within the watershed could contribute extra nitrogen to the system. Given this information it is possible that these elevated nitrogen levels may give *Phragmites* more of an edge than is to be expected from the model.

GIS MODELING OF *PHRAGMITES* SUITABILITY

The initial step in the GIS modeling analysis involved obtaining estimates of the restored salinity and flooding levels after the restoration occurred. This was accomplished by using an existing hydrodynamic model (Woods Hole Group, unpublished) that predicted salinity and water levels under the various dike opening alternatives, including the 47.5m (156ft) wide x 3m (10ft) high opening. After some discussion it was concluded that it would be most useful to consider the most extreme circumstance, the 3m (10ft) opening. Polygons that estimated the tidal water levels under the 3m (10ft) modeled scenario were provided by the hydrodynamic model. The shapefiles were converted to rasters for the suitability analysis. Areas that remained wet at mean low water would have standing water on the surface even at low tide, meaning they would essentially be permanently flooded. Because the literature review found that *Phragmites* does not tolerate standing water, the mean low water polygon zone was used as an area where *Phragmites* would not prefer to spread into. Areas of the marsh that were not regularly flooded were clipped out and defined as high marsh-- the areas between mean high water and mean high water spring (defined by Niering and Warren 1980). Based on the literature review high marsh zones were given higher preference as areas where *Phragmites* would show preference for growth.

The next step was to convert the jpg files of modeled salinity from the hydrodynamic model into a workable salinity raster format (Fig 3). Jpg files for different extents of the tidal cycle from mean low water to annual high water were available. After advisement I chose to use the salinity values for mean high water spring because that would be the highest salinity stress the estuary would face on a semi-regular basis, twice a month. It was believed that these bi-monthly high salinity values would have control over the spread of *Phragmites*. To convert the jpg image of salinity into a useable format, the files were first geo-referenced to the NAD 1983 UTM Zone 19 N projection that was used for all data in the analysis. Then the jpg file was clipped to the outline of the estuary. An unsupervised classification function converted the clipped jpg into a workable raster with 20 distinct classes based on color. Then the 20 classes in the raster were reclassified into salinity levels as estimated from the salinity scale on the original jpg files of modeled salinity (Fig 3). The raster calculator was used to find the most suitable zone for *Phragmites* growth (defined as areas under 20ppt), semi-suitable (defined as areas 20 to 25 ppt), and zones unlikely to see *Phragmites* growth (defined as areas greater than 25ppt). The best zones for *Phragmites* seedling suitability were also calculated (very suitable 0-10 ppt, suitable 11-20 ppt, and unsuitable above 20 ppt).

To consider the fact that *Phragmites* could potentially spread as the incremental restoration occurred, preference was given to a buffered zone around current *Phragmites* stands. This was intended to give some temporal accountability to the analysis. To accomplish this a shapefile had to be made of current *Phragmites* stands in the estuary. A map from the NPS provided a very broad estimate of current *Phragmites* stands within the NPS land. Unfortunately there are some parts of the estuary that are owned by private landowners and data on the *Phragmites* areas in those regions was lacking, another limitation to the model. I was, however, able to digitize one *Phragmites* stand outside of the National Seashore, an area which was observed during my visit to the Herring River estuary in March 2012. The stand was near Moby Dick's restaurant, along Rt. 6, and was digitized using an aerial photo. Then the spatial analysis Euclidian distance tool was used to make a raster of a 40 meter buffered distance around each existing *Phragmites* stand. The zones were chosen as follows. The first 20 meters around the area was given a higher weight based on this zone accounting for the maximum rhizome growth expected within 10 years, the next 20 meters was given a more moderate weight based on spreading being likely within 20 years. All areas beyond 40 meters were not given any weighted preference for the possibility of rhizome growth from current stands. These breaks were chosen because the research showed that *Phragmites* rhizomes can extend at maximum up to 2 meters per year (Michigan DEQ 2008). The 2 meter value was used as an overly cautious estimate of *Phragmites* growth. The 10 to 20 years timeline was chosen because it is a reasonable estimate of how long it will take managers to open the tidal gate to the full 3m (10ft) opening. This factor was designed to help incorporate spatial and temporal scales into the modeling.

Preference for proximity to *Phragmites* was done differently for the seedling suitability map. The highly suitable zones the mature *Phragmites* model, defined as those scoring at or above a 17, were used as predicted *Phragmites* stands in the seedling model. The zone was clipped out of the original model and used in the suitability analysis of the second model. Again, the Euclidian distance tool was used to make a raster this time of a 100 meter buffered distance around each predicted *Phragmites* stand. The zones were arbitrarily broken into 25 meter regions and decreasing preference was given for each zone.

Finally, to assess possible physical and light-limiting obstructions to seedling establishment and seedling dispersal the aggregate vegetation map of the Herring River estuary was used. Classifications in the vegetation map that would be composed of woody vegetation were chosen as areas likely to impede seedling dispersal and growth (Smith 2007). All non-woody areas were converted into a raster to show areas where seed germination would more likely because they would not be impeded by dead woody vegetation.

MODEL SPECIFICS

Each of the raster factors was assigned a weighed value of 0 to 10 based on how suitable or unsuitable the factor was for *Phragmites* growth. Table 1 summarizes the weights used in the suitability maps for each factor.

Table 1: A summary of the suitability factors and values for two suitability analyses conducted for a proposed tidal-restoration of the diked Herring River estuary in Cape Cod, MA. The two analyses were for the suitability of mature *Phragmites australis* growth and the suitability of *Phragmites australis* seed germination within the estuary.

MATURE *PHRAGMITES*		
Suitability Factor	**Factor divisions**	**Weighted points**
Salinity	0 to 20 ppt	10
	21 to 25 ppt	8
	26 to 28 ppt	2
Flooding tolerance	Marsh above MHWS	2
	'High marsh' zone (MHW-MHWS)	8
	'Low marsh' zone- (MLW-MHW)	2
	Marsh below MLW	0
Proximity to *Phragmites*	Current *Phragmites* stands with 20 meters buffer	5
	20 to 40 meter buffer	2
***PHRAGMITES* SEEDLING SUITABILITY**		
Suitability Factor	**Factor divisions**	**Weighted points**
Salinity	0 to 10 ppt	10
	11 to 20 ppt	5
	21 ppt to 28 ppt	2
Flooding tolerance	Marsh above MLW	5
	Marsh below MLW	0
Proximity to predicted *Phragmites* stands	predict *Phragmites* stands with 25 meters buffer	10
	>25 to 50 meter buffer	8
	>50 to75 meter buffer	5
	>75 to 100 meter buffer	2

RESULTS AND DISCUSSION OF THE SUITABILITY MODEL

Figures 4 and 5 show the result of the suitability analyses for mature *Phragmites* growth and *Phragmites* seed germination and growth, respectively. Both figures show the unique values of the suitability results from the high values that are most suitable to the lowest values that are unsuitable for *Phragmites* growth. Unique values were used instead of the standard deviation or quantile method of dividing data because the values were not evenly distributed. Unique values with less than 50 pixels were not included in the legend because there areas were too small to be significant. Figure 4 illustrates that the most *Phragmites* prone areas are in the far reaches of the basin where salinity levels will not be as high. The edges of the main basin will likely see a persistence of *Phragmites* because salinity levels there will be lower. However, most of the main basin shows very low suitability for *Phragmites* and it is expected that the restoration will eliminate much of the *Phragmites* in this region. Areas of moderate suitable to high suitability should be of concern to managers. Upper Pole Dike Creek, Pole Dike Creek East and Upper Bound Brook should be of particular concern because they already have current *Phragmites* stands in the area, which are likely to spread. Figure 6 shows the different sub-basins and the current stands of *Phragmites* for reference. Figure 5 shows where seeds are likely to germinate and seedlings survive. Again it is the far reaches of the basin that are most suitable to invasion by *Phragmites*.

LIMITATIONS TO THE MODELS

There are several limitations based on the data used and my personal bias in creating the weighted factors for the suitability model. These limitations should be understood when considering and interpreting the final maps. However, despite the limitations these maps should provide a useful estimate of areas of concern for *Phragmites* spread.

The mean low water zones, or areas of permanent standing water used in this analysis, were from the 3m (10ft) opening scenario of the hydrodynamic model. However since the restoration process will be incremental over several years, and perhaps decades, there is expected to be accretion of the salt marsh that may result in the mean low water zones being less extensive than modeled. Thus the model may underestimate the likelihood of spread in those areas.

Additionally, as mentioned before there may be some inaccuracy in the model because of the lack of data to consider some controlling factors such as sulfide toxicity and nitrogen enrichment. Lack of consideration of sulfide toxicity may mean that the models overestimate the suitability of *Phragmites*, while lack of consideration of nitrogen enrichment may mean the models underestimate the suitability of *Phragmites*.

The modeled salinities used were from the hydrodynamic model for the Mean High Water Spring-- a tidal extent, that occurs twice monthly, in which the marsh would experience maximum salinity levels. However, the estuary will actually see great fluctuations in salinity levels based on daily tides, storm tides, ground water flows, varying amounts of precipitation, and seasonal changes. Thus, the actual salinity that the *Phragmites* has to contend with will be somewhat different from the MHWS values used in the suitability model.

MANAGEMENT APPLICATIONS BASED ON THE MODELS

Figure 4 illustrates the high to moderately suitable areas of *Phragmites.* These zones should be of particular concern for managers and they may want to take measures to prevent or limit the spread of *Phragmites* into these areas. For the small existing *Phragmites* plots in the upper reaches of the estuary that have a high potential to spread, such as the one at the head of the Pole Dike Creek basin, it may be appropriate to apply herbicide to the area before restoration occurs to manage the small patch of *Phragmites* before it has a chance to spread. Of course herbicide use has potential negatives that would need to be weighed against the positives of arresting *Phragmites* development.

Seedling prone areas as shown in Figure 5 can also be monitored to help assist managers in charge of adaptive vegetation management. One possibility for *Phragmites* seedling prone areas that become open mudflats after restoration is to plant these areas with native salt marsh vegetation to assist the native species and give them a restorative advantage over *Phragmites*.

THE FUTURE

If the National Park Service is pleased with the results they may want to consider applying this model to other scenarios including the smaller 0.9m (3ft) opening, to obtain a better sense of the effect of the incremental restoration process on *Phragmites*. Also, a similar analysis can be conducted for other plant species of interest, such as *Typha* spp. (Cattail) or salt marsh species (e.g., *Spartina patens* and *Spartina alterniflora*). Like for *Phragmites*, flood tolerance and salinity are fundamental controlling factors for these species.

This suitability model should be evaluated after the restoration occurs to see how effective it was in predicting the eventual distribution of *Phragmites.* If found to be effective the model can be adapted to other salt marsh restoration projects.

Figure 1: An Aerial view of the Herring River estuary in Cape Cod, MA showing the full extent of its basins in yellow. The tidal-restricting dike is highlighted in red.

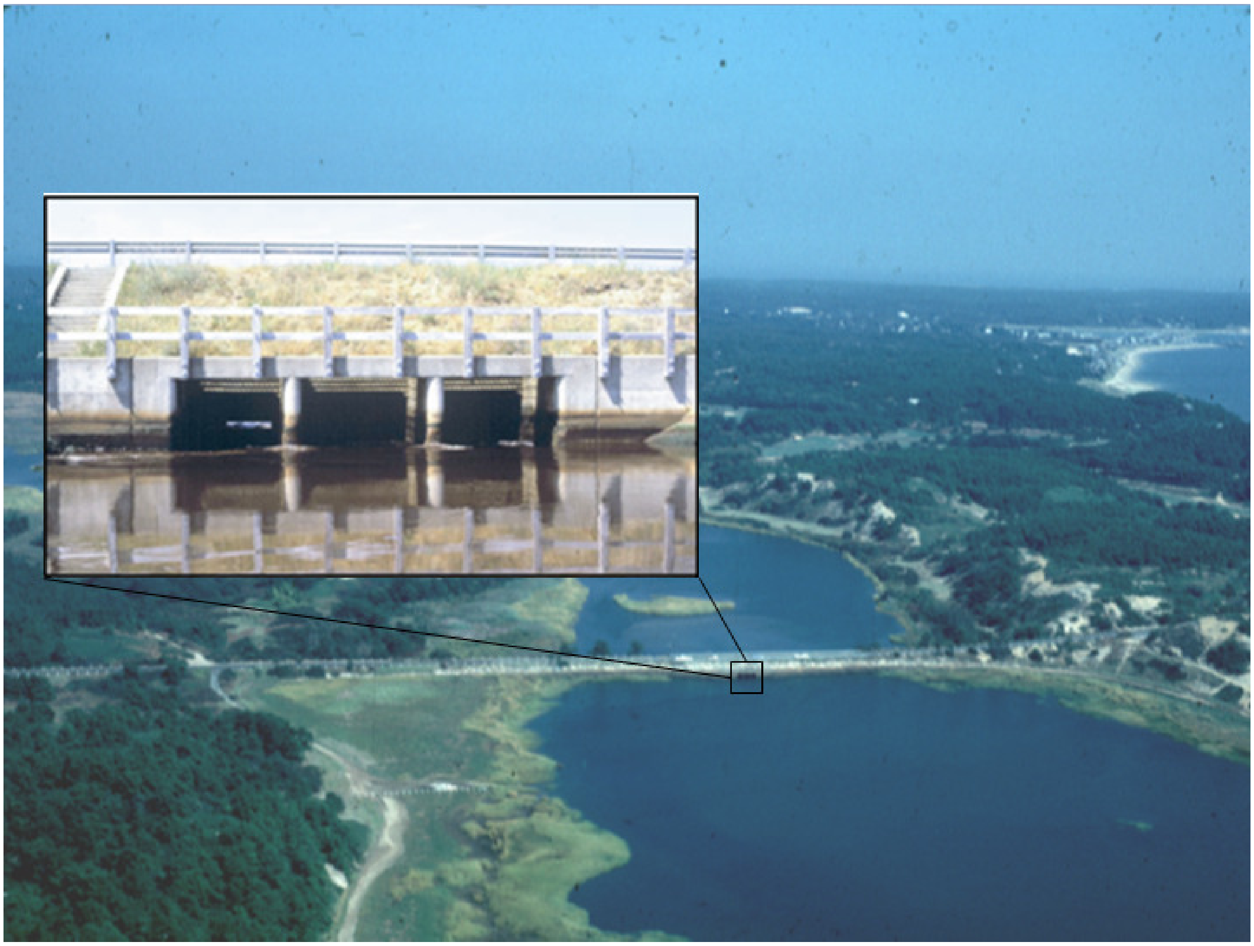

Figure 2: An aerial photo of Herring River estuary in Cape Cod, MA with a zoomed-in view of the tidal gates and sluice gate.

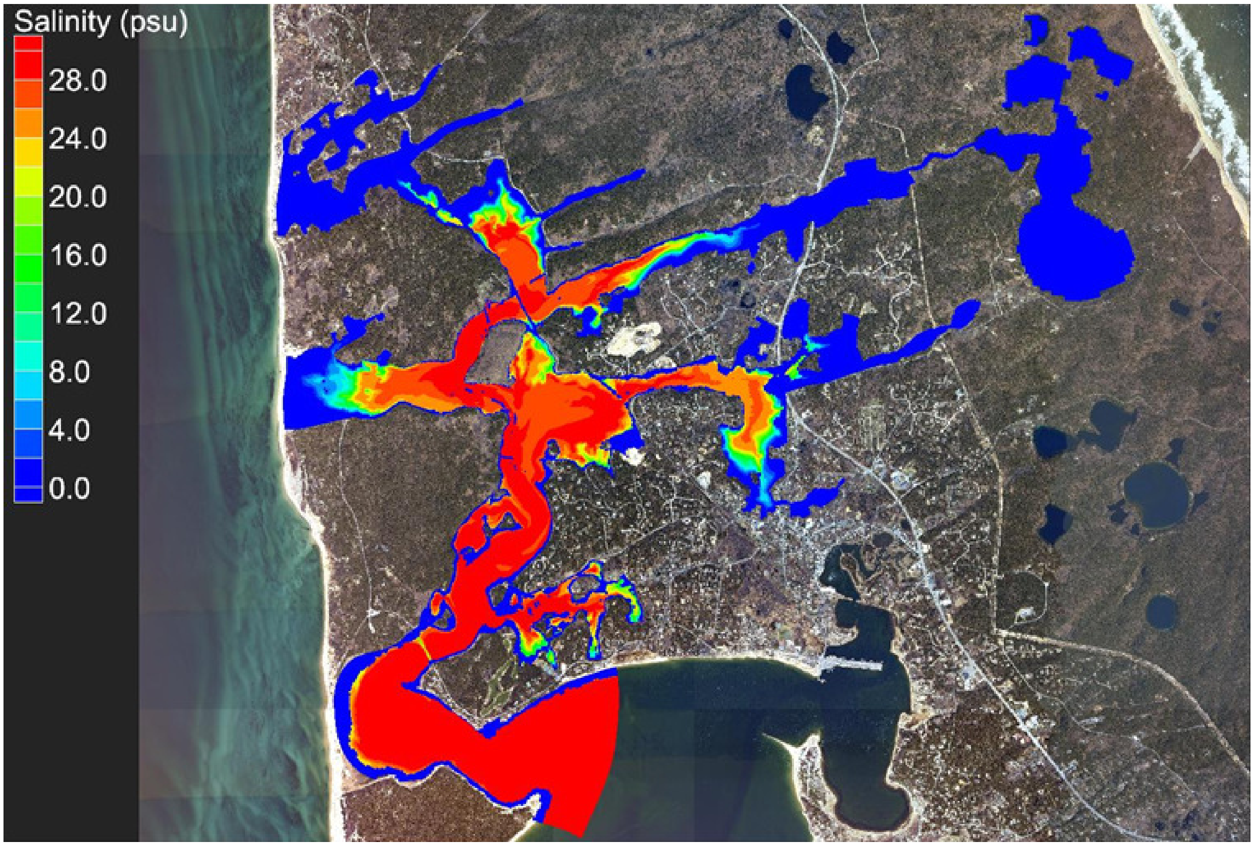

Figure 3: An aerial view of the Herring River estuary in Cape Cod, MA. This jpg shows the modeled salinity values for the Mean High Water Spring tide of the proposed 47.5m (156ft) gate control structure at the 3m (10ft) opening. Salinity estimates in the analysis were based on the scale shown on the left. Data source: Woods Hole Group.

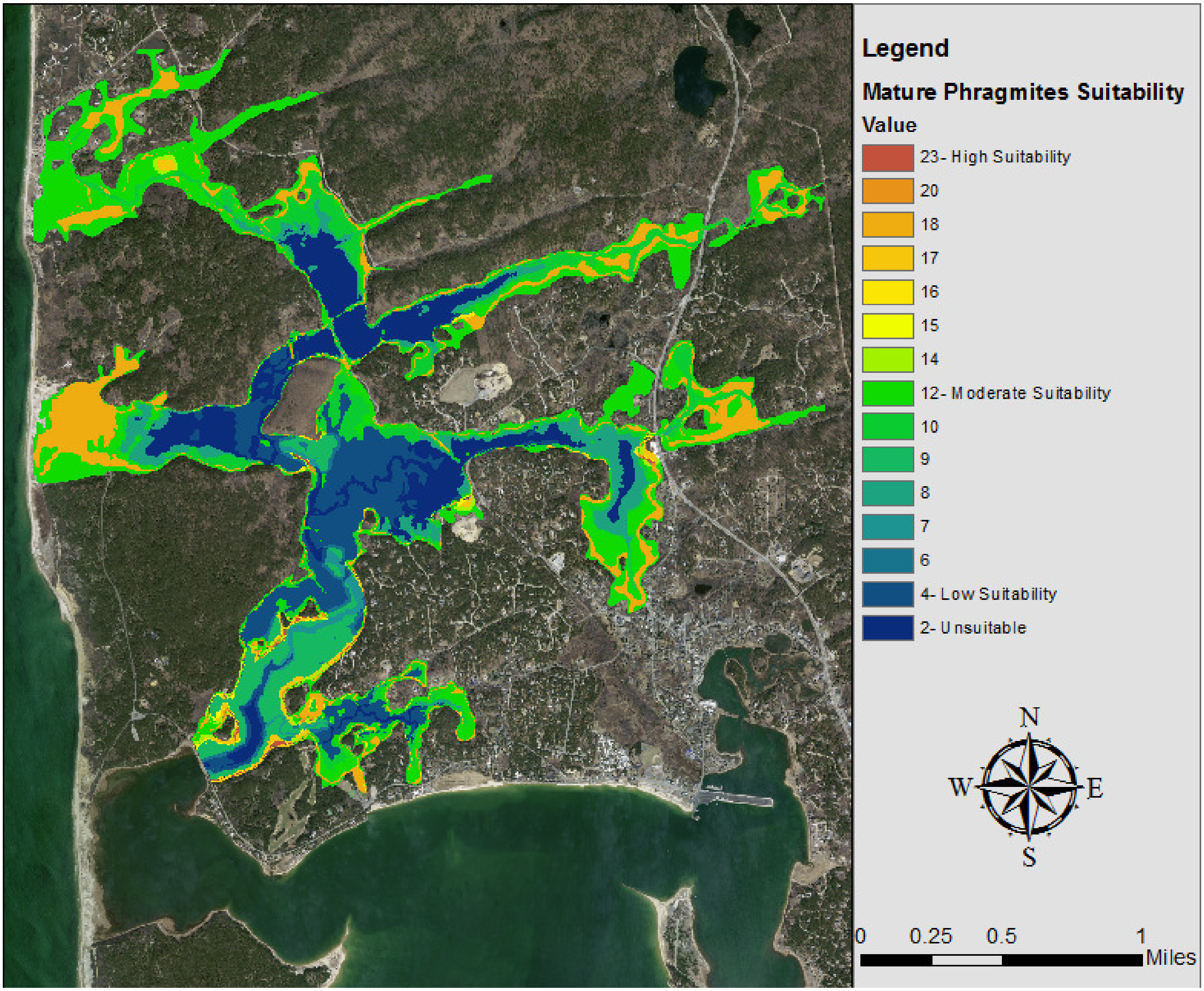

Figure 4: Modeled suitability analysis of *Phragmites australis* growth in the Herring River estuary in Cape Cod, MA, in the event that a tidal-flow restoration occurs. The map shows how suitable or unsuitable mature *Phragmites* will find different areas of the basin. Suitability was based on modeled salinity and flooding levels, as well as proximity to current stands of *Phragmites*.

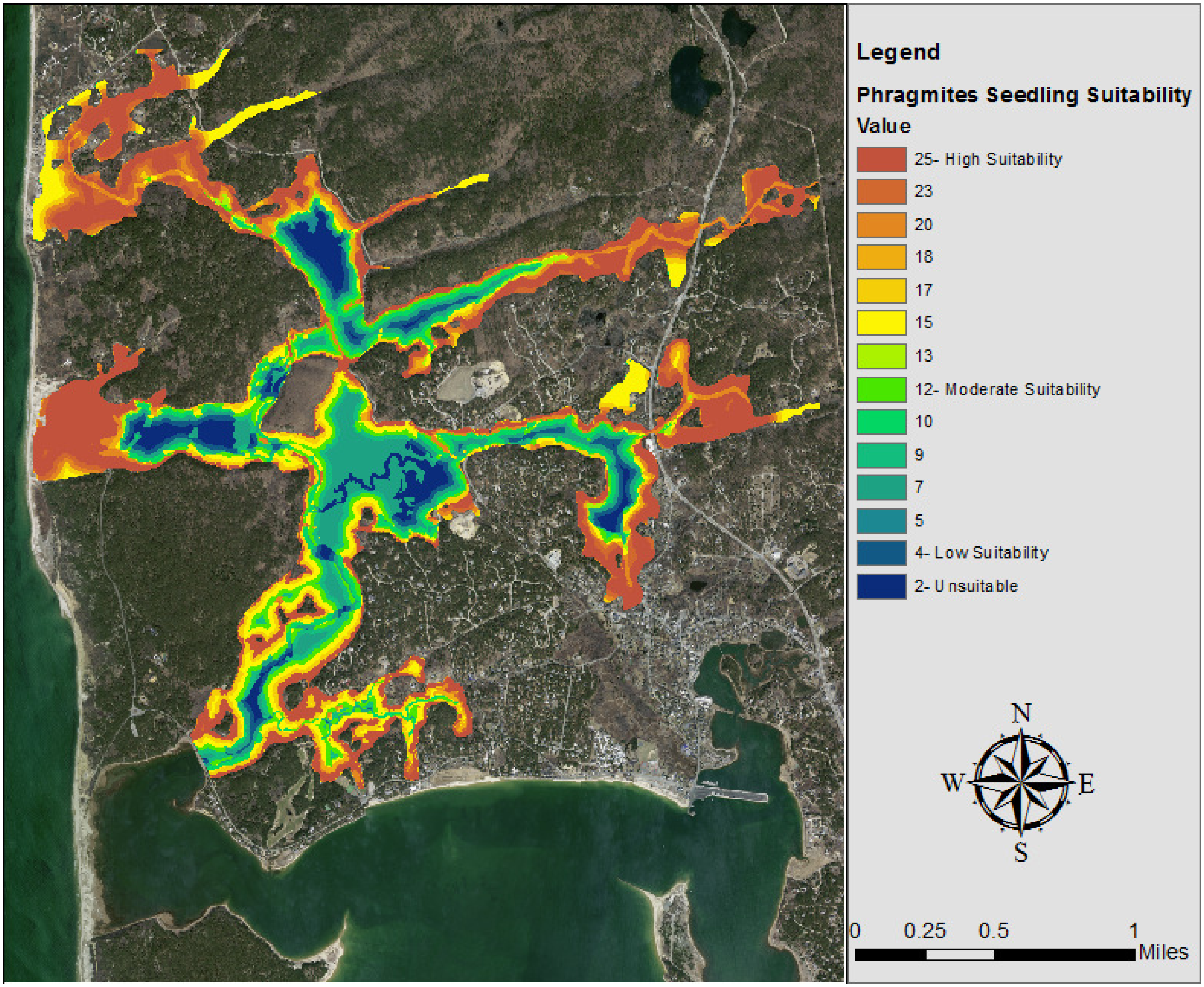

Figure 5: Modeled suitability analysis of *Phragmites australis* seedling growth in the Herring River estuary in Cape Cod, MA, in the event that a tidal-flow restoration occurs. The map shows how suitable or unsuitable *Phragmites* seeds will find different areas of the basin. Suitability was based on modeled salinity and flooding levels, proximity to predicted stands of *Phragmites*, and the impediment of seedling development by woody vegetation.

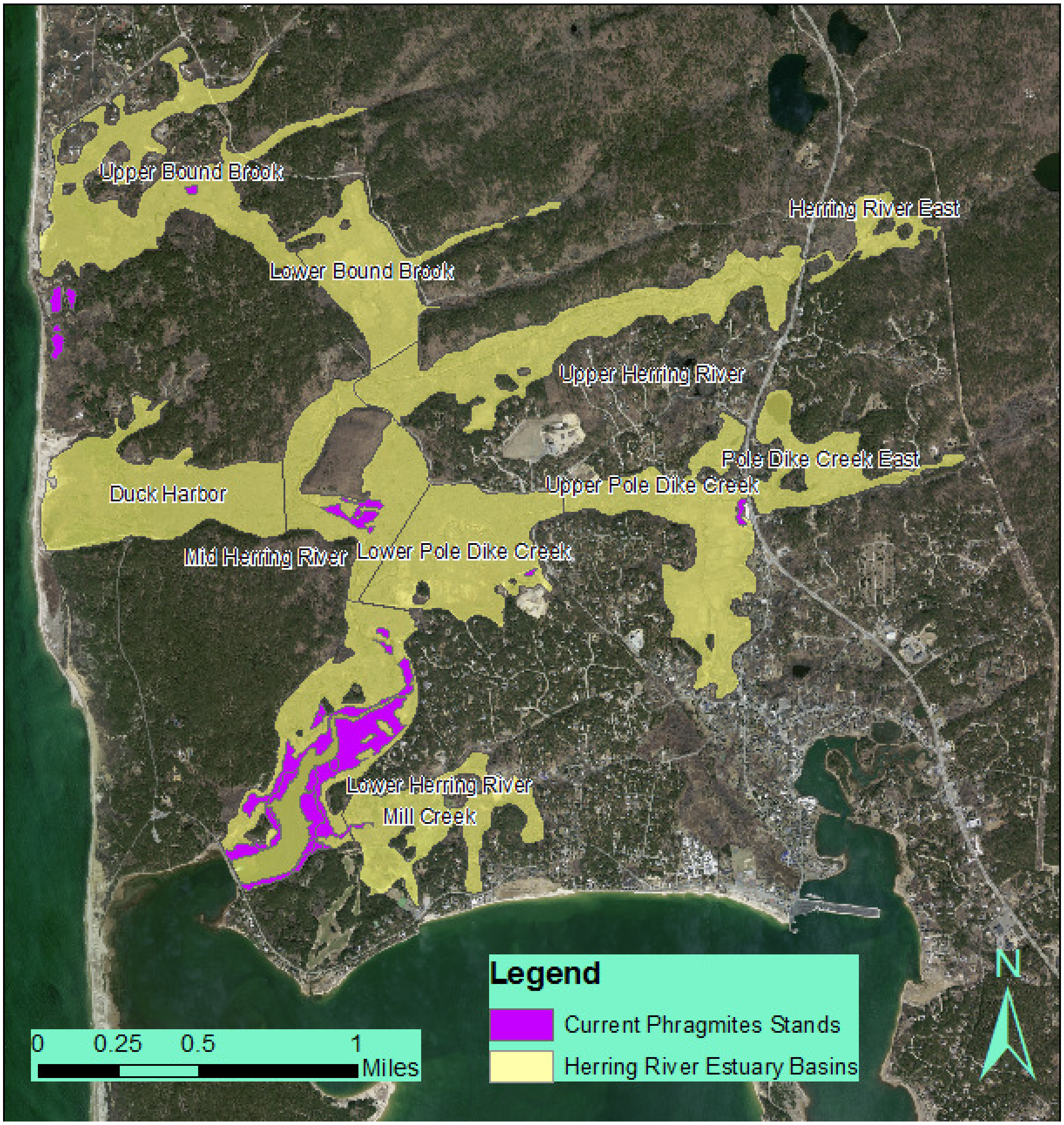

Figure 6: An aerial view of the Herring River estuary in Cape Cod, MA, showing the different basins of the estuary in yellow and the current stands of *Phragmites australis* in purple.

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REFERENCES:

Amsberry L, M.A. Baker, P.J. Ewanchuk, and M.D. Bertness. 2000. Clonal integration and the expansion of *Phragmites australis*. Ecological Applications **10 :**1110-1118.

Bart D., and J. M. Hartman. 2003. The role of large rhizome dispersal and low salinity windows in the establishment of common reed, *Phragmites* australis, in salt marshes: New links to human activities. Estuaries **26:** 436-443.

Benoit, L. K., and R. A. Askins. 1991. Impact of the spread of *Phragmites* on the distribution of birds in Connecticut marshes. Wetlands **19:** 194-208

Buchsbaum R. N., J. Catena, E. Hutchins, and M. James-Pirri. 2006. Changes in salt marsh vegetation, *Phragmites australis*, and nekton in response to increased tidal flushing in a New England salt marsh. Wetlands **26:**5 44-557.

Burdick D. M., and R.A. Konisky. 2003. Determinants of expansion for *Phragmites australis*, common reed, in natural and impacted coastal marshes. Estuaries **26:** 407-416.

Chambers R.M., D.T. Osgood, D.J. Bart, and F. Montalto. 2003. *Phragmites australis* invasion and expansion in tidal wetlands: interactions among salinity, sulfide, and hydrology. Estuaries **26:** 398-406

Chambers, R.M., L.A. Meyerson, and K. Lellis-Dibble. 2012. Ecology of *Phragmites australis* and responses to tidal restoration. *in* C. Roman and D. Burdick, editors. Restoring tidal flow to salt marshes: A synthesis of science and management. Island Press, Washington, DC.

King, R.S., W.V. DeLuca, D.F. Whigham, and P.P. Marra. 2007. Threshold effects of coastal urbanization on Phragmites australis (common reed) abundance and foliar nitrogen in Chesapeake Bay. Estuaries and Coasts **30:** 469‐481.

Meyerson L.A., K. Saltonstall, L. Windham, E. Kiviat & S. Findlay. 2000. A comparison of *Phragmites australis* in freshwater and brackish marsh environments in North America. Wetlands Ecology and Management **8:** 89-103.

Michigan Department of Environmental Quality [Michigan DEQ]. 2008. A guide to the control and management of invasive *Phragmites.* Michigan Department of Environmental Quality. Michigan, USA.

Niering W.A. and R.S. Warren. 1980. Vegetation patterns and processes in New England salt marshes. BioScience **30:** 301–307.

Parsons, K. C. 2003. Reproductive success of wading birds using *Phragmites* marsh and upland nesting habitats. Estuaries **26:** 596-601.

Portnoy, J. W. 1991. Summer oxygen depletion in a diked New England estuary. Estuaries **14:** 122-129.

Portnoy, J. W. 1999. Salt marsh diking and restoration: Biogeochemical implications of altered wetland hydrology. Environmental Management **24:** 111-120.

Portnoy, J. W., and J. R. Allen. 2006. Effects of tidal restrictions and potential benefits of tidal restoration on fecal coliform and shellfish-water quality. Journal of Shellfish Research **25:** 609-617.

Roman, C. T., R. W. Garvine, and J. W. Portnoy. 1995. Hydrologic modeling as a predictive basis for ecological restoration of salt marshes. Environmental Management **19:** 559-566.

Roman C. T., and D. M. Burdick, editors. 2012. Tidal marsh restoration: A synthesis of science and management. Island Press, Washington DC, USA.

Silliman R.B. and M.D. Bertness. 2004. Shoreline development drives invasion of *Phragmites australis* and the loss of plant diversity on New England salt marshes. 2004. Conservation Biology. **18:** 1425-1434

Smith, S. M. 2007. Removal of salt-killed vegetation during tidal restoration of a New England salt marsh: effects on wrack movement and the establishment of native halophytes. Ecological Restoration **25:** 268-273.

Smith, S. M., C. T. Roman, M. J. James-Pirri, K. Chapman, J. Portnoy, and E. Gwilliam. 2009. Responses of plant communities to incremental hydrologic restoration of a tide-restricted salt marsh in southern New England (Massachusetts, U.S.A.). Restoration Ecology **17:** 606-618.

Staltonstall, K. 2003. Genetic variation among North American populations of *Phragmites australis:* Implications for management. Estuaries **26:** 444-451.