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## Strengthening Virginia's Wetlands Management Programs

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# Final Report

## **Strengthening Virginia's Wetlands Management Programs**

Virginia Institute of Marine Science  
Center for Coastal Resources Management

Environmental Protection Agency  
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## Introduction

Wetland management in Virginia has made significant strides in the past decade in terms of the information available to the managers making permit decisions. There are, however, a number of continuing and emerging challenges to the Commonwealth's goal of achieving no net loss and a net resource gain. This project was designed to begin development of comprehensive wetland management plans for localities that address the threat of climate driven impacts to tidal wetlands.

The tidal wetlands management program was originally designed with an implicit assumption that the managed resource base was fixed in space. As a result all the program efforts are aimed at minimizing direct impacts within the technically defined bounds of existing tidal wetlands. While this program continues to struggle to achieve no net loss through effective compensation for permitted impacts, the resource base is undergoing increasingly significant natural changes. There is clear and expanding evidence that climate change will generate losses that are not addressed by the current regulatory program. A study recently concluded by CCRM mapped tidal wetlands in the Lynnhaven River watershed and showed that one third of current wetland habitat will be lost over the next century as a result of sea level rise (CCRM, 2009). The model used a conservative rate of sea level rise (4.25 mm/yr), lower than many more recent predictions; suggesting losses may be even greater.

In this report, we address this issue by demonstrating that the threat to the resource is real and already occurring and by developing proposed enhancements to the management program. Public awareness of issues is developing, but translation of concern to policy action requires more concrete evidence of immediate risk than model predictions for many of our legislators and managers. Fortunately, we have the opportunity to generate that evidence thanks to the original inventory of tidal wetlands developed by VIMS in the early 1970s. That effort mapped the size and location of all the Commonwealth's tidal wetlands, and included a quantitative description of the plant community composition for each mapped wetland unit. It represents a quantified baseline for areal and biotic change over a 30+ year period.

We chose to focus on the York River system because this system has the potential to provide evidence of the three major consequences we anticipate from climate change:

- loss of marsh area due to erosion and inundation;
- wetland transgression into low-lying uplands; and
- plant community shifts driven by sea level rise and salinity shifts.

We also have established the capacity for more highly resolved future monitoring of changes by establishing strategically located permanent monitoring transects.

The information on observed changes will be combined with modeled future changes to develop a resource risk assessment; producing a rationale for expanded management for delivery to policy-makers and managers critical for authorizing new management

options. The goal for the entire line of work is to produce comprehensive wetland management plans for every coastal locality. In Virginia this is the level of government at which existing tidal wetlands management occurs, as well as land use planning and management.



# Chapter 1: Geo-Spatial Assessment of Tidal Wetlands Change in the York River Watershed

## Introduction

Tidal wetlands were originally mapped and catalogued for the York River Watershed over a period of time beginning in 1973 through 1979 depending on location and locality. The process produced 1:24,000 scale maps delineating tidal wetlands, which corresponded to field observations that classified each marsh using a Community Type index (Virginia Institute of Marine Science *et al.*, 1993) and estimated distribution of plant species by percent of total plant composition and marsh area. This effort was largely to promote and support the Commonwealth's tidal wetlands regulatory program which started in 1972. This program offered protection to tidal wetlands through regulation of human use activities such as coastal construction.

Today the interest in delineating marsh boundaries extends beyond regulatory and protection programs overseeing impacts to tidal wetlands associated with human use. Concerns over the long-term sustainability of the habitat are mounting due to sea level rise predictions, coupled with modification of the natural coastal landscape, and competition of space brought on by development. An analysis of change in baseline parameters over the last 33-39 years assesses how the resource has responded over time to natural and anthropogenic stress and the vulnerability of the resource with respect to sustainability into the future.

To focus on the geo-spatial change in marsh boundaries a new delineation of tidal wetlands was undertaken using recent base-mapping products combined with field surveys. A remotely sensed interpretation of recent aerial imagery was conducted to develop the current delineation of wetlands. This delineation was compared to the 1970's baseline delineation to look at spatial change in distribution. The sections below discuss the methods for developing this change analysis, the findings, and the assumptions and challenges that result when trying to predict change from time series studies. In Chapter 2, the changes in plant community are discussed.

For the change analysis we selected high resolution, geo-referenced natural color imagery collected in 2009 by the Virginia Base Mapping Program. No other information source that could be used to estimate change over time was available at scales more comparable to the original survey. Comparable field reviews were conducted to collect data necessary to report and compare on species and plant community changes. These surveys also provided quality control and assurance (QA/QC) for the remotely sensed interpretive boundary data.

## Methodologies for Wetland Delineations

The analysis to compute spatial change in the distribution of tidal wetlands is based on the areal difference between an early tidal marsh boundary delineation published in 1979 (TMI79) and a recent boundary delineation generated for current conditions from 2009

imagery (TMI09). Technology, baseline data, and resources differ greatly between the two survey periods. Some of these differences will affect the accuracy of our change calculations. Therefore some discussion of each approach is warranted. In both study cases, the best available technology and resources available within the scope of the project budget were used.

### **1979 Tidal Marsh Delineation**

The areal extent and geospatial position of the 1979 marshes were derived from tracings of marsh symbology illustrated on 7.5' (1:24,000) USGS topographic maps. Highly comprehensive field surveys focused on developing the plant community structure for each marsh, but also provided a system to QA/QC the topographic marsh boundaries. A number of marshes, particularly narrow fringe marshes were not shown on the 1:24,000 topographic basemaps. These were drawn as a single line. The length of these features, were estimated using maps features, aerial photography and best professional judgment. Aerial imagery, where available, was also used as a reconnaissance tool to correct marsh boundaries that had changed significantly since the topographic survey.

Beginning in the early 1990s original topographic maps printed on stable based mylar were placed on Numonics 2200 series digitizing tablets and marsh boundaries were hand digitized using precision cursors. Tablets were interfaced with SUN Unix workstations running the ESRI software ArcInfo®. Mylar maps were geo-registered on the tablet using a quality assurance digitizing standard of RMS = 0.002 inches or better. Other program and computer based standards were put in place to insure accuracy of the digital product. These included the node snap tolerance (<0.05 inches), and fuzzy tolerance (0.001 inches = 1.0 meters in UTM); procedural standards that control digitizing accuracy and final product quality (Berman *et al.*, 1993). Despite these measures, the accuracy of the final product in geographic space can be no better than the accuracy of the base map. USGS 7.5 minute topographic maps have reported horizontal accuracy of +/-12.2 meters.

Interpretive protocols called for operators to follow the depicted marsh symbology or adjusted lines on the mylar maps to delineate the marshes as polygons. Single line fringe marshes noted from the field surveys were also digitized as a polygon without regard to scale using a standard marsh width of approximately 5 meters. During the digitizing process each marsh was coded with a number marsh, community type, and the Federal Information Processing Standard (fips) code for locality. QA/QC measures were in place to reduce factual errors associated with the coding process. The geomorphic type (fringe, extensive, embayed, marsh island) and species compositions were added to the database later.

### **2009 Tidal Marsh Delineation**

Heads-up digitizing was performed to develop the boundary delineation for current wetland distribution. Heads-up digitizing is a common method for capturing vector objects directly from the computer screen using a mouse or cursor. The method is

considered more accurate than traditional tablet digitizing since the user can resolve more features using zoom functions. Digital high resolution (6”) color infrared aerial photography collected in 2009 was used to develop the current wetland boundaries. Photo interpretation techniques were used to identify wetland objects on the screen and ArcMap versions 9.3 and 10.0 were used for heads-up digitizing.

Digitizing scale was set at 1:1,000 to maximize the benefit of the high resolution source data while still adhering to time constraints established by the project schedule. Ancillary data sets including the VA Shoreline Inventory, TMI79 and National Wetlands Inventory (NWI) clipped for the study area was used to help identify narrow fringe marshes masked by tree canopy or visual scale.

When digitizing was complete the file was smoothed to improve the cartographic quality. The smoothing algorithm used was PAEK (Polynomial Approximation with Exponential Kernel) using a smoothing tolerance of 5 meters. Digitized marsh polygons were attributed with a marsh number and morphologic type (embayed, extensive, fringe or island) for a direct comparison with the earlier TMI79.

Quality control and assurance (Q&A) was by independent staff scientist review. Additional Q&A was conducted during field work for the plant community composition. During field observations, marsh boundaries were added or visually adjusted on rectified image base maps. Digital corrections were made in the lab when community composition data were added to the attribute files.

Consistency in identifying and digitizing the marsh boundary was tested using repetitive sampling techniques. Six marshes of varying size and complexity were selected and each digitized three times. Each digitized area was compared to the mean (Table 1).

**Table 1. Digitizing Accuracy Test using Repetitive**

<b>Sample</b>	<b>Mean area (acres)</b>	<b>Diff. 1</b>	<b>Diff. 2</b>	<b>Diff. 3</b>
1	39.024	+0.044	+0.055	-0.099
2	22.684	-0.077	+0.008	+0.070
3	9.335	+0.052	-0.080	+0.027
4	6.893	+0.030	-0.003	-0.028
5	3.874	-0.075	+0.016	+0.058
6	3.695	+0.113	+0.023	-0.135

The average difference in calculation of area for each sample was +/- 0.0003 acres. It is impossible to resolve or explain these small differences with any certainty. They are largely operator error and must be accepted as inherent in the data product in order to move on with the change analysis.

## Methodology – Change Analysis

The analysis of change in tidal wetland area is based on the areal difference between the 1979 TMI and the 2009 TMI. Both coverages were clipped to the same study boundaries to insure a direct comparison between surveys. Using superposition techniques in ArcMap the analysis of change reveals marsh area accretion, marsh loss, and areas of no change. An overall calculation of change is also made. Calculations based on marsh morphology type were also computed to determine if there is any relationship between measured change and marsh type.

## Change Analysis Results

Combined areas of wetland loss throughout the study equate to 4,875 acres of marsh land lost. The change analysis reveals that roughly 3,081 acres of new wetlands were mapped in 2009 for an overall change of -1,794 acres. Approximately 16,320 acres remain unchanged (Figure 1).

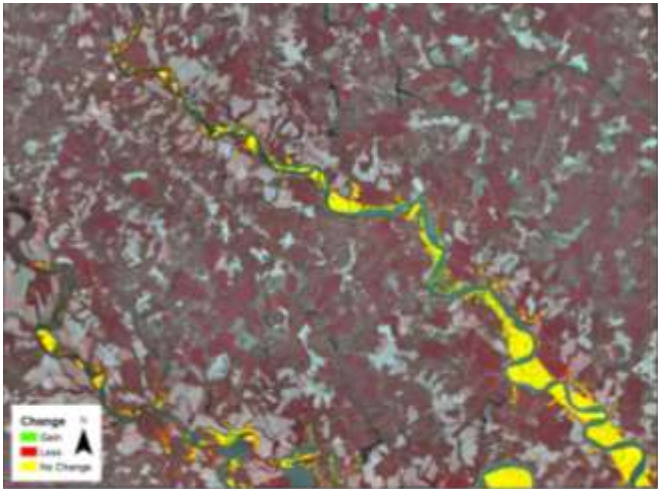
Plausible explanations for the new wetland areas might be conversion of upland to marsh, vertical accretion of tidal flats to an elevation that could support vegetative growth, or restoration and creation. Marsh losses can be attributed to marsh edge erosion, sea level rise, and development stressors. Unfortunately no distinct patterns or systematic evidence explaining these changes are obvious. It leads us to consider that some of the change computed could be due to inherent problems of comparing two datasets developed using different technologies, and basemaps.

Erosion and accretion of marshes throughout the study area exhibited no geo-spatial pattern, but appeared relatively randomly dispersed. We re-examined the data by marsh class to see if any relationships emerged (Table 2).

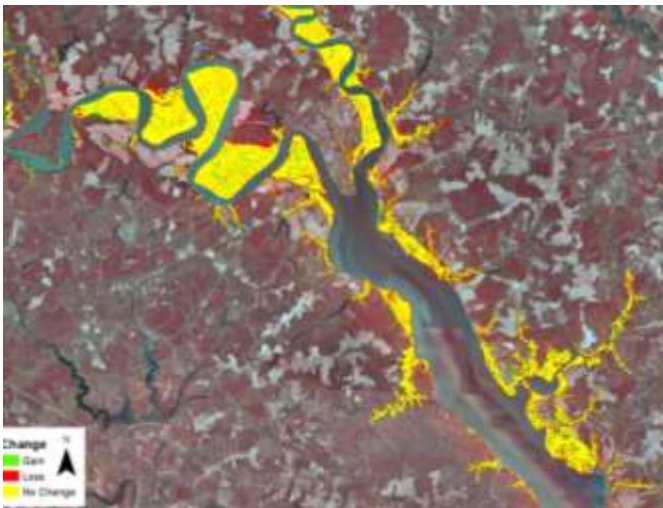
**Table 2. Calculated change of marsh area by marsh class**

Marsh Class	1973 TMI	2009 TMI	Change
Embayed	5,462.279	4,872.708	-589.571
Extensive	13,934.873	13,077.216	-857.657
Fringe	999.927	714.400	-285.527
Marsh Island	798.140	736.492	-61.648
Total	21,195.219	19,400.816	-1,794.403

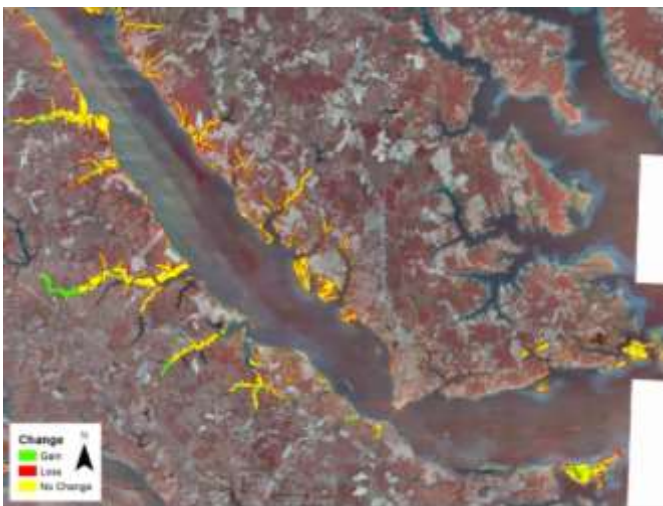
Marsh Class	Unchanged (ac)	Loss (ac)	Gain (ac)
Embayed	3,570.313	1,892.015	1,312.850
Extensive	11,872.249	2,062.629	1,222.298
Fringe	310.659	689.270	384.962
Marsh Island	566.777	231.363	160.708
Total	16,319.998	4,875.277	3,080.718



**Upper York River**



**Middle York River**



**Lower York River**

**Figure 1. Tidal wetlands change in the York River Watershed**

With only one exception the majority of wetlands remained unchanged in each class. Fringe marshes were the exception where 69% of the original marshes were lost over time compared with embayed (34%); extensive (15%), and marsh islands (29%).

A close examination of the data and original delineations suggest that the ability to quantify change accurately may differ between the different marsh types. The following discussion presents examples from each of these classes that demonstrate important issues that must be factored into an analysis of change.

### **Fringe Marshes**

Quantifying change in fringe marshes may be difficult. Since the original basemaps used in the 1970s didn't depict most fringe marshes present, these areas were estimated based on visual observations made in the field and translated manually to basemaps or aerial imagery. Recall from the discussion above that while the alongshore length of the features may closely approximate their true locations, their width was standardized as per the protocol to be 5 meters. Since many of these features are quite narrow this would result in a large overestimation of the original acreage of fringe marshes. Therefore, when compared to a high precision delineation developed in 2009 the loss is more extreme.

Figure 2 illustrates what happened in the 1979 TMI when the field crew located a fringe marsh not shown on the topographic map. It was recorded as a line feature on the base map. Later, it was digitized to the correct length, but the width was grossly exaggerated since protocols used a standard average width of 5 meters (shown in pink). The area was re-digitized using 2009 imagery (shown in yellow). In this example, a quantitative comparison between the two surveys shows the marsh area went from about 9 acres to less than half an acre.



**Figure 2. Fringe marsh along the York River with 1979 delineation in pink and 2009 delineation in yellow.**

In cases of larger fringe marshes which are actually depicted on the basemaps (Figure 3) the errors may not be so great. A fringe marsh which was wide enough to be captured on the 7.5" topographic map is shown in Figure 3a. The delineation is compared to the 2009 survey (in yellow) in Figure 3b. The obvious smoothing at the upland edge due to the coarse scale of the basemap created some detection limitations.



**Figure 3a. Original delineation of fringe marsh from topographic map (1979)**



**Figure 3b. Comparison of 1979 fringe marsh (pink) and 2009 fringe marsh (yellow)**

Topographic maps have a reported horizontal accuracy of approximately 12.2 meters. This means that the true position of the line can vary by that amount in any direction. The variability shown in Figure 4b could be attributed, in part, to the accuracy of the map medium from the 1979 delineation. It is likely that some elements of change illustrated are “true change” while others are artifacts of the source products. The alongshore length of the marsh is very close between the two years. The seaward edge of the marsh suggests a slight retreat of the marsh edge over time. This is certainly plausible with sea level rise, as well as marsh edge erosion brought on by boat wakes and tidal currents. The inland boundary has also shifted seaward. Here the change could be due to incorrect positioning of the landward edge in the original survey; which is often more difficult to define. The upland edge appears to include a narrow forest fringe which could mask the delineation in either survey.

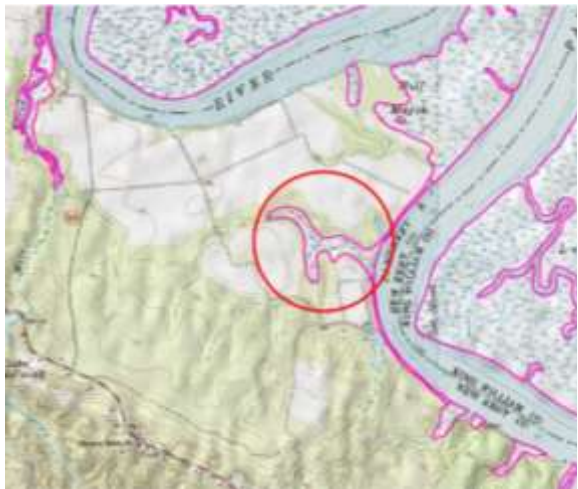
We expect fringe marshes to spatially respond quicker to natural and anthropogenic processes. These narrow features are conceivably less stable and impacts associated with processes such as wave and tidal current erosion may have a greater impact on their sustainability versus a larger more established marsh community. Fringe marshes are also subject to die backs associated with shading that occurs as riparian forest canopy expands with time. Human impacts at the upland edge occur more frequently as well. Installation of erosion control structures behind the fringe marsh will prevent inland transgression of the marsh over time. Structures can also exacerbate wave action, which can deepen the critical depth of the intertidal flat to a depth which can no longer support

the marsh vegetation. More studies looking at variations in these features over time compared with more extensive or embayed marshes would be necessary for these findings to be conclusive.

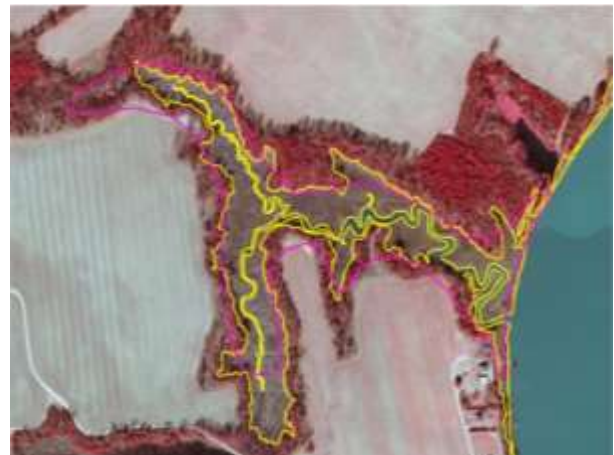
### **Embayed and Extensive Marshes**

Looking at change spatially over time in embayed marshes or extensive marshes also presents levels of uncertainty that should be mentioned. Since topographic scales are significantly coarser than scales of the high resolution image products, the boundaries of tidal channels that form or dissect these features may not be well defined or apparent on the base maps. Figure 4a and 4b is an embayed marsh on the Pamunkey River. Precision digitizing of the basemap in 1979 represents the marsh boundary accurately (Figure 4a). However the actual creek bed was not even apparent on the basemap and therefore not digitized. This results in a slightly larger marsh surface area in the earlier delineation than in 2009 delineation.

The coarser scale also means there is an absence of detail available to digitize. Lines are smoother and often times straighter. In Figure 4b the difference in detail between the two surveys is evident. Overall, however, the boundary differentiations are relatively close.



**Figure 4a. An embayed marsh on the Pamunkey river is delineated in 1979**



**Figure 4b. The 1979 marsh in pink is compared with the new 2009 marsh boundary in yellow.**

In extensive marshes like the one shown in figure 5a, this situation is even more pronounced. The 1979 delineation (in pink) excludes nearly all tidal channels. This results in an apparent larger marsh surface area. When the tidal creeks are digitized in the 2009 the surface area of the marsh is reduced due to surface water area within the channel margins. A comparison between the two could yield significant loss overtime. This has been noted in several extensive marshes. In this particular marsh, however; another mapping problem presents itself that is noteworthy. The expanded view of the focal area shown in Figure 5b illustrates the width of the tidal channel in 1979 to be nearly four times the width of the same channel in 2009. This discrepancy is not due to



tidal infilling or any natural or anthropogenic process. Rather the difference is the result of a map scale problem originating with the basemap in 1979.



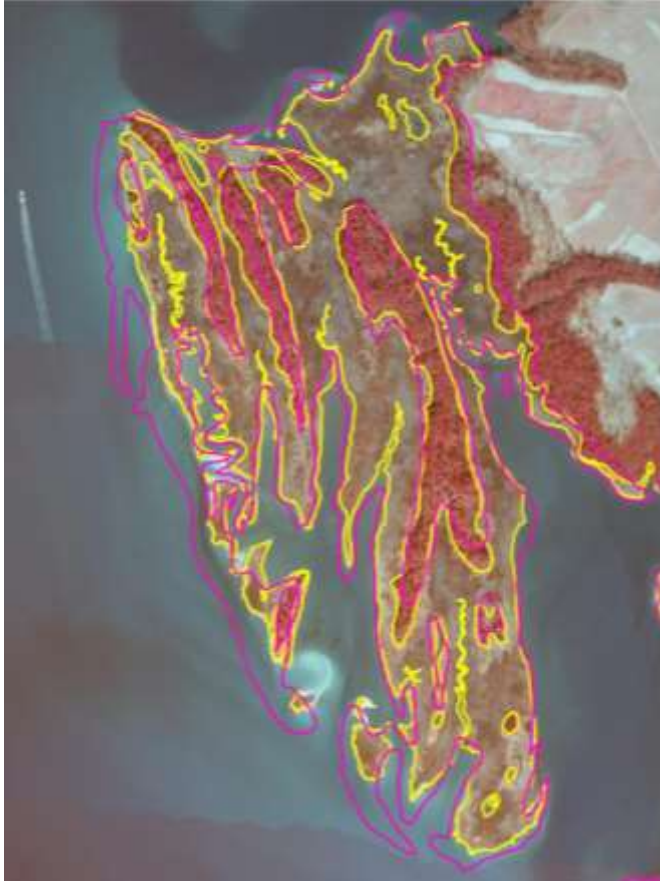
**Figure 5a. Sweethall Marsh showing 1979 (pink) delineation and 2009 (yellow) delineation.**



**Figure 5b. Tidal channel in Sweethall Marsh. Pink is 1979. Yellow is 2009**

No emerging patterns were evident when we examined the marsh types independently of each other. Great variation in geo-spatial change occurred and the explanations for those variations are numerous. The marsh examples presented above are all located in the mesohaline portion of the York River where wave energy is typically lower. We also reviewed similar marsh types in the lower, polyhaline, portion of the river where wave energy is typically higher and physical forces to the marsh edge would be greater under storm conditions.

The first example is an extensive marsh located on the northeast exposed shore of the York River (Figure 6). This is an emergent salt marsh with a limited and narrow attachment to the upland. The marsh/upland complex has a ridge and swale type morphology. Aerially, the marsh has lost approximately 44 acres between 1979 and 2009. A significant portion of that loss has occurred along the main river edge. This loss could easily be explained by marsh edge retreat associated with shoreline erosion. Studies mapping shoreline change between 1937 and 2007 in this same area found rates varied between 0 to -1.5 meters per year (Milligan *et al.*, 2010). These rates would be consistent with the change we are seeing here.



**Figure 6. Catlett Islands marsh change on the York River**

Marsh edge erosion illustrated in Figure 7a is also corroborated by a shoreline change analysis conducted by Milligan *et al.*, 2010 (Figure 7b). That study mapped shoreline change from aerial imagery collected over time. Their results show the continuous recessive nature of the marsh edge. The site, located on the southwest shore of the York River is exposed to northeast winds. The erosion can be attributed to high wave energy and sea level rise. Some inland migration of the marsh is evident along portions of the upland boundary. Our comparison shown in Figure 7a between the 1979 delineation (pink) and 2009 (yellow) computes 134 acres of loss.



Figure 7a. Extensive marsh on southwest shore of York River

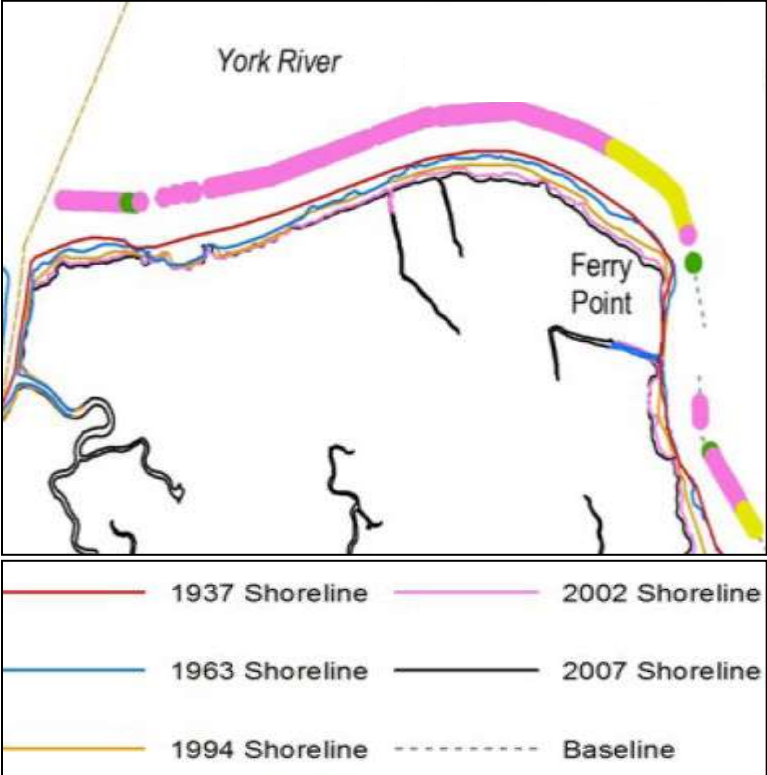


Figure 7b. Shoreline change analysis for Ferry Point (after Milligan *et al.*, 2007)

## Marsh Islands

True marsh islands in the Chesapeake Bay have been reported to exhibit a more uniform rate of edge erosion than marshes attached to the mainland. Land loss on the islands due to sea level rise results in marsh edge erosion and loss of interior land associated with ponding (Wray *et al.*, 1995). Historical change along the Mumfort Islands off the northeast shore of the York River, shows that shoreline recession of the marsh island edge persists throughout time, but is not uniform around the island (Figure 8a and 8b). Erosion is considerably less on the upland-facing side of the island versus the marsh edge facing the open water. We suspect a combination of wave erosion and sea level rise is responsible for the increased rates of recession on the open water edges. The upland clearly offers protection to the marsh edge.

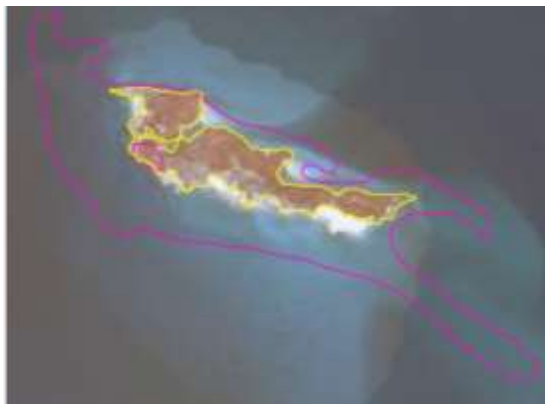


Figure 8a. Mumfort Island mapped in 1979 (pink) and 2009 (yellow).

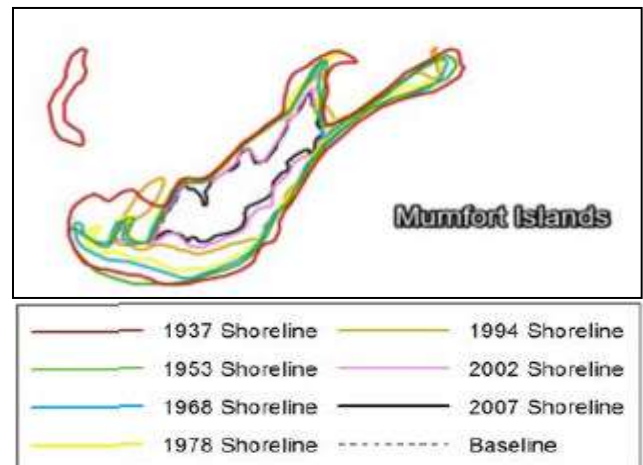


Figure 8b. Shoreline retreat around Mumfort Islands (after Milligan *et al.*, 2010)

The 1979 delineation from topographic maps shown in pink in Figure 8a follows the same trend and geometry that is illustrated in Figure 8b from Milligan *et al.*, 2010. The 1963 survey from that study is very close to the delineation from the topographic baseline.

The small marsh island located off the mouth of Purtan Creek on the northeast shore of the York River has receded significantly since the 1979 survey (Figure 9a). An examination of the historical image archive shows this was not a true marsh island but rather an extension of a large spit that traversed the creek mouth at least until 1937. The tip was breached and the island was formed sometime between 1968 and 1978. This could be the result of high waves or simply sea level rise. By 1979 the island remained and the comparison with the 2009 survey shows ongoing erosion. Like the Mumforts downriver of this site, erosion is greatest on the exposed river edge of the island.



**Figure 9a. Purtan Creek Marsh Island 1979 (pink) vs. 2009 (yellow)**



**Figure 9b. 1937 (red) vs. 2009 (black) (after Milligan et al., 2010)**

Within the York system not all marsh islands are eroding. The example in Figure 10a and 10b is from the tidal freshwater portion of the York where some islands appear to be accreting. These marsh islands are located in a section of the Pamunkey River where the river widens slightly and sediment deposition could be occurring. The depositional pattern is similar for both islands suggesting accretion occurs under the same process. Between 1979 and 2009 an increase of 4 acres was measured for this site.



**Figure 10a. Marsh island in Pamunkey River**



**Figure 10b. Accretion of Pamunkey marsh islands (pink: 1979; yellow: 2009)**

## Discussion

The findings above demonstrate great variability across the York River with respect to geospatial change in tidal marshes over time. The greatest uncertainty is calculations of change most likely occur in tidal fringe marshes which are the most difficult to detect from any basemap product. These marshes which typically have long marsh-upland edges may also be extremely sensitive to conditions along the upland edge for long-term sustainability over time. One hypothesis is that marshes that have a longer marsh-water edge than a marsh-upland edge are less affected by anthropogenic pressures on the upland and more impacted by events associated with storms and tidal conditions.

Despite the numerous technical issues associated with developing certainty in change analysis, external studies conducted in the same locations measuring shoreline change over historical records were in agreement with the findings here. More importantly, there was agreement between the period of record closest to our record of greatest uncertainty (1979). This suggests that our speculation of error for this dataset may be over stated.

The absence of trends along the river or among the different marsh types may also be due to the large number of sites. Table 3 compares the overall distribution of marsh classes for the 2009 dataset. The study looked at 3,230 marshes from this year class. The results of change reported in Table 2 suggest that fringe marshes are the most vulnerable marsh class. Further analysis would be necessary to determine vulnerability of marshes by classes. Establishing monitoring stations for long-term monitoring is one type of research initiative to explore. As part of this study, Chapter 4, monitoring stations have been established for long-term trends. These initial sites have been set up to look at not only geo-spatial change but also plant community shifts. Chapter 2 discusses the results of the baseline investigation in community change between 1979 and 2009 across the entire study area.

**Table 3. Distribution of Marshes in the York River in 2009**

<b>Marsh Class</b>	<b>Total Acres</b>	<b>Number of Polygons</b>
Embayed	4,859	1,256
Extensive	13,092	542
Fringe	714	1,283
Marsh Island	737	149

## Chapter 2: Changes in community composition

### Methods

The original Tidal Marsh Inventories along the York River, VA, were conducted in the 1970s and published between 1974 and 1987. For convenience, they will be referred to as “historic data/communities” or “circa 1970” through the remainder of the report. Data from the York River was incorporated into several reports as seen in the following table:

**Table 4. Original TMIs**

County	Year (Pub)	Authors
York	1974	Gene Silberhorn
Gloucester	1976	Kenneth A. Moore
King and Queen	1987	Walter I. Priest III, Gene M. Silberhorn and Andrew W. Zacherle
James City	1980	Kenneth A. Moore
New Kent	1979	Damon G. Doumlele
King William	1987	Gene M. Silberhorn and Andrew W. Zacherle

Field sampling procedures for each of the surveys can be found in the appropriate report (reports are found at: [http://ccrm.vims.edu/publications/tidal\\_marsh\\_inventories.html](http://ccrm.vims.edu/publications/tidal_marsh_inventories.html)). However, the sampling methods are very similar to the procedure used for the current survey, described below.

For the current survey (referred to as “current data/communities” or “2010 data” in this report), every marsh visible from boat-accessible waters on the York River and its associated creeks and rivers was surveyed in June, July, and August 2010. Two teams were used, with each team responsible for one side of the river. An additional team was used for QA/QC purposes.

At each site:

1. The survey team would assess marshes designated by the tidal marsh inventory. If the original marsh was absent, each team made appropriate notes regarding surrounding conditions (bank height, anthropogenic alterations, etc.).
2. The marsh plant species were identified and percent cover for each species was recorded using a preset handheld computer. Video and digital photos were taken as needed to enhance the data collection effort.
3. Detailed notes were taken about any aspects of the site assessment that may be helpful in analyzing the data or a future return trip to the same marsh site.

In order to prevent team “drift” in scoring, the two field teams trained together at the beginning of the field season. A field test, designed to ensure inter-team consistency, was performed immediately after training. Two more such events took place during the field season as teams shift from mesohaline to oligohaline to tidal freshwater at approximately three weeks and five weeks after the initial field test. The field test consisted of both field teams visiting a site with predominately tidal brackish vegetation and a site with predominately tidal freshwater vegetation with a review meeting to discuss any discrepancies between teams. A team of experts (derived from the Principle Investigator Team) performed a re-assessment of approximately 1% of the points to further insure QA/QC.

Data were downloaded to georeferenced datasheets for analysis. The creation, loss and fragmentation of marshes over the past 30 years lead to the identification of more marshes in the current survey than were identified in the historic survey. Much of the increase in the number of marshes was due to a more precise definition of marsh edges than in the historic survey (e.g. an entire creek system was historically considered a single marsh, while in the current survey it might consist of 8 smaller marshes) and does not reflect a significant change in the geological extent of the marshes. For comparison purposes, historic marshes which have been lost were excluded from the historic data, new marshes were excluded from the current data, and current marshes were combined to match the historic data. These manipulations resulted in 264 marshes which can be directly compared between the two datasets.

Species lists from both the historic and current surveys were also reduced to help elucidate patterns of change. Trace species (<0.5% of a community) were removed entirely. Most of the analysis focused on 10 species (9 of which were the most common species, together making up over 90% of the majority of marsh communities, and *Phragmites australis*, an invasive reed species). Marsh communities were also assigned a “type” based on the community types in the historic surveys. However, for consistency, all communities, current and historic, were assigned new types under a standard protocol.

**Table 5. Marsh community types**

Marsh Type	Criteria for assignment
Type 1	<i>Spartina alterniflora</i> > 50% (Saltmarsh Cordgrass)
Type 2	<i>Spartina patens</i> + <i>Distichlis spicata</i> > 50% (Saltmeadow Hay & Saltgrass)
Type 3	<i>Juncus roemerianus</i> > 50% (Black Needlerush)
Type 4	<i>Baccharis halimifolia</i> + <i>Iva frutescens</i> > 50% (Saltbushes)
Type 5	<i>Spartina cynosuroides</i> > 50% (Big Cordgrass)
Type 6	<i>Typha angustifolia</i> + <i>Typha latifolia</i> > 50% (Cattails)
Type 7	<i>Peltandra virginica</i> + <i>Pontederia cordata</i> > 50% (Arrow Arum & Pickerel Weed)
Type 8	<i>Phragmites australis</i> > 50% (Reedgrass)
Type 9	<i>Nuphar luteum</i> > 50% (Yellow Pond Lily)
Type 10	<i>Salicornia spp.</i> > 50% (Saltwort)



Type 11	<i>Peltandra virginica</i> + <i>Pontederia cordata</i> + other freshwater species > 50%
Type 12	<i>Spartina alterniflora</i> + <i>Spartina patens</i> + <i>Distichlis spicata</i> + other brackish species > 50%
Type 16	Approximately equal mix of freshwater and brackish water species

Changes in community type were noted and the type of shift was categorized.

## Results

All of the dominant species, as well as most other species were found in both years (see Figure 11) although their distribution may have changed. Some species (such as Sweet Flag) which were only found in one year, have short periods during which they are easily visible (frequently flowering) and their appearance/lack of appearance is likely due more to shifts in sampling timing rather than ecological processes. Some of the historic surveys done in the Mattaponi and Pamunkey Rivers occurred during the early fall, while the current surveys were done in the mid- to late summer. The flowering component of freshwater marshes can vary dramatically from month to month, making certain species appear more prevalent at certain times. For that reason, we have chosen to focus our analysis on the more dominant species which tend to be observable throughout the growing season.

Categorizing the marshes into community types circumvents the problem of surveys occurring during different months. All the species found in the freshwater marsh are typical of a freshwater community, even though the exact community composition may have changed. Exploring the question of whether upstream salt wedge movement is affecting the marsh communities can be answered by looking for a shift in community type from freshwater communities to transitional/brackish communities. However, it occasionally can lead to an apparent detection of change where no ecologically significant change has actually occurred. Because dominance is defined as “> 50 coverage by a given species or collection of species”, a marsh which was 45% Saltmarsh Cordgrass and 55% Arrow Arum in the historic survey but is now 51% Saltmarsh Cordgrass and 49% Arrow Arum will have changed community type (from a Type 7 to a Type 1). However, this shift is more likely attributed to annual variability or differences in observer bias between the two surveys than true community change.

Shifts in marsh community composition between historic and current surveys were apparent although the type of shifts seen differed along the length of the river and between the north and south shores. Comparisons between the two surveys of the dominant plant species can be seen in Figure 12. One significant change in marsh community has been the introduction of Reedgrass (*Phragmites australis*) along the length of the York River. This species was not found in the York River at all during the historic survey. Reedgrass is currently found in discrete clusters in the York and Mattaponi Rivers and is fairly widespread in the Pamunkey River (see Figure 15). Given the aggressive nature of Reedgrass, it is actually less well-spread than would be expected.

This argues for either a fairly recent introduction of this species into the York River system or that it is not well adapted to this system for unknown reasons.

Indications of marsh flooding (possibly due to sea level rise) can be seen in the York River system where areas which historically had significant high marsh communities appear to have converted almost entirely to low marsh. This type of shift appears to be more prevalent along the north shore of the river in the brackish region. The north shore may be more susceptible to flooding due to lower shoreline topography than is found on the southern shore. The shift from high to low marsh community can also be seen in the Pamunkey River, where Big Cordgrass is being replaced by Saltmarsh Cordgrass, although the mechanism driving this is unclear since Saltmarsh Cordgrass is more tolerant of both salinity and flooding than Big Cordgrass.

Indications of salinity shifts can be seen in the lower reaches of both the Pamunkey and Mattaponi Rivers where currently Saltmarsh Cordgrass appears to have a more significant presence upriver. On the Mattaponi River, the historic data shows little Saltmarsh Cordgrass north of the bridge at West Point, while in the current survey; it can be commonly found almost the entire surveyed length. On the Pamunkey River, marshes which were previously a mix of communities (including freshwater species, Wild Rice and Arrow Arum) are now almost entirely brackish marshes (Saltmarsh Cordgrass and Big Cordgrass). In the mid-York River, marshes in which the high marsh community was historically Big Cordgrass, now consist of a diverse group of high marsh plants including Saltmeadow Hay, Saltgrass and saltbushes. These plants are somewhat more tolerant of salinity than Big Cordgrass and may also indicate salinity shifts.

Results from the change in community type analysis are consistent with the species specific analysis, although the changes in community type give heavy weight to the most common species and therefore are potentially more conservative. Community types from both surveys can be seen in Figure 13. There is not a 1:1 correspondence of points between the two years because marshes were lost, gained or split in the intervening time period. Similar to the previous analysis, there appear to be different processes acting on the north and south shores of the mainstem York River (See Figure 14), although many sites show no change in community type. The north shore marshes (in particular the north shore creek marshes) frequently show increased low marsh. In the mainstem this generally indicates a shift from a Mixed Brackish community to a Saltmarsh Cordgrass community. In the Mattaponi and Pamunkey Rivers this could indicate a shift from a variety of high marsh community types to either a Saltmarsh Cordgrass or Arrow Arum/Pickerel Weed community.

The mainstem York River south shore frequently shows changes in community type that are indicative of increased diversity and are typically a shift from a Big Cordgrass community to a more mixed community type. The mouths of the Pamunkey and Mattaponi Rivers are areas where a great deal of change is occurring, although not in a particularly consistent pattern according to this method of data analysis.

Figure 11. Historic and Current Species.

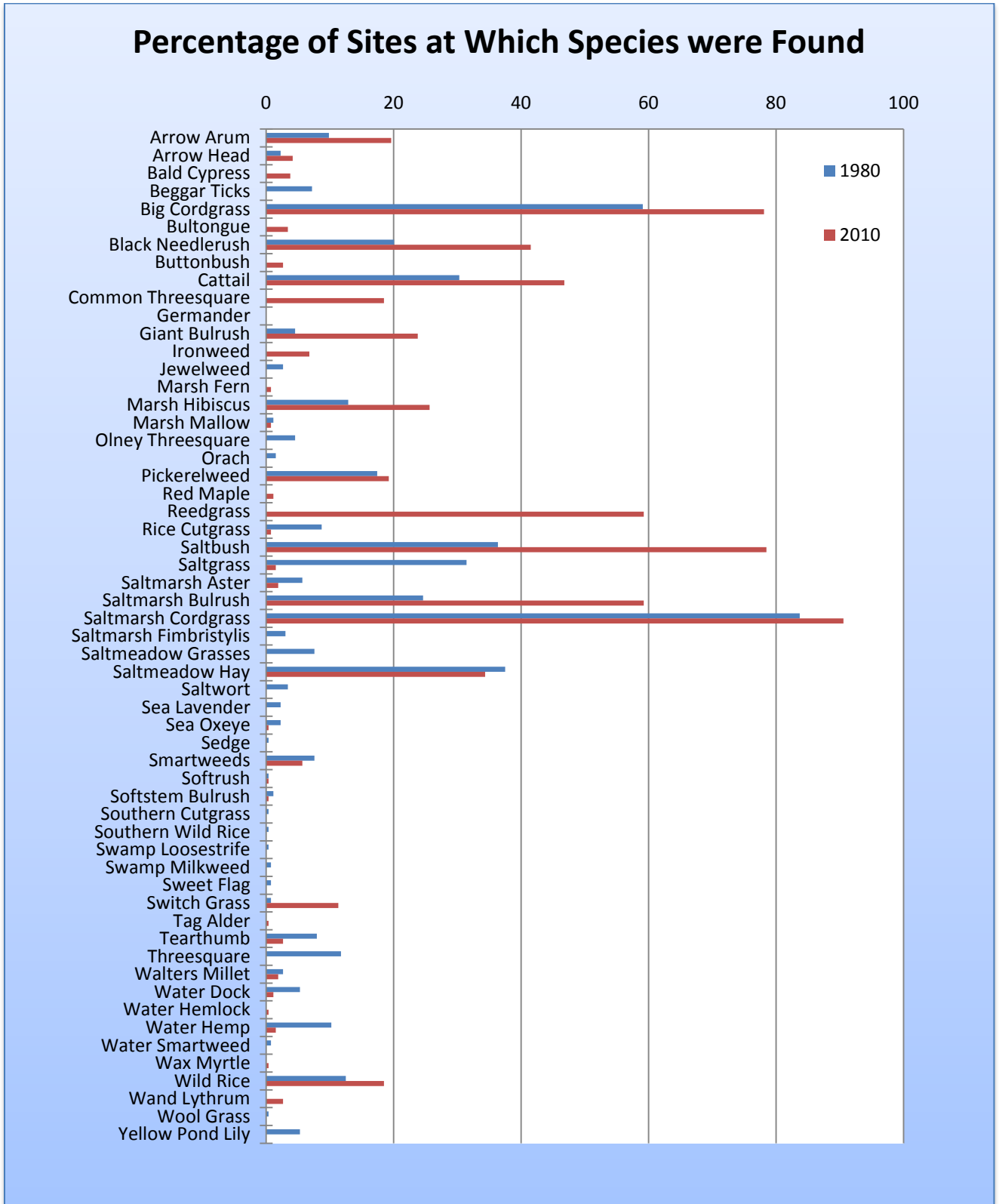
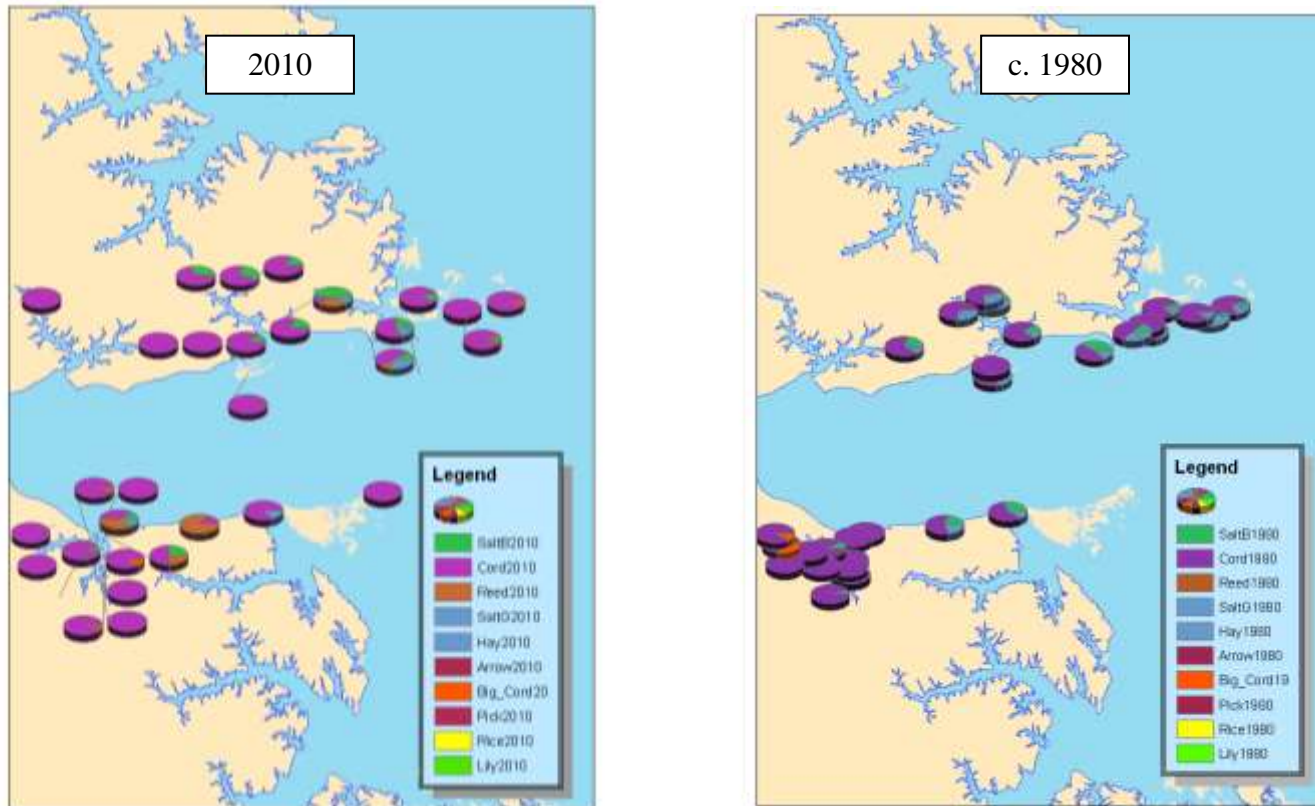
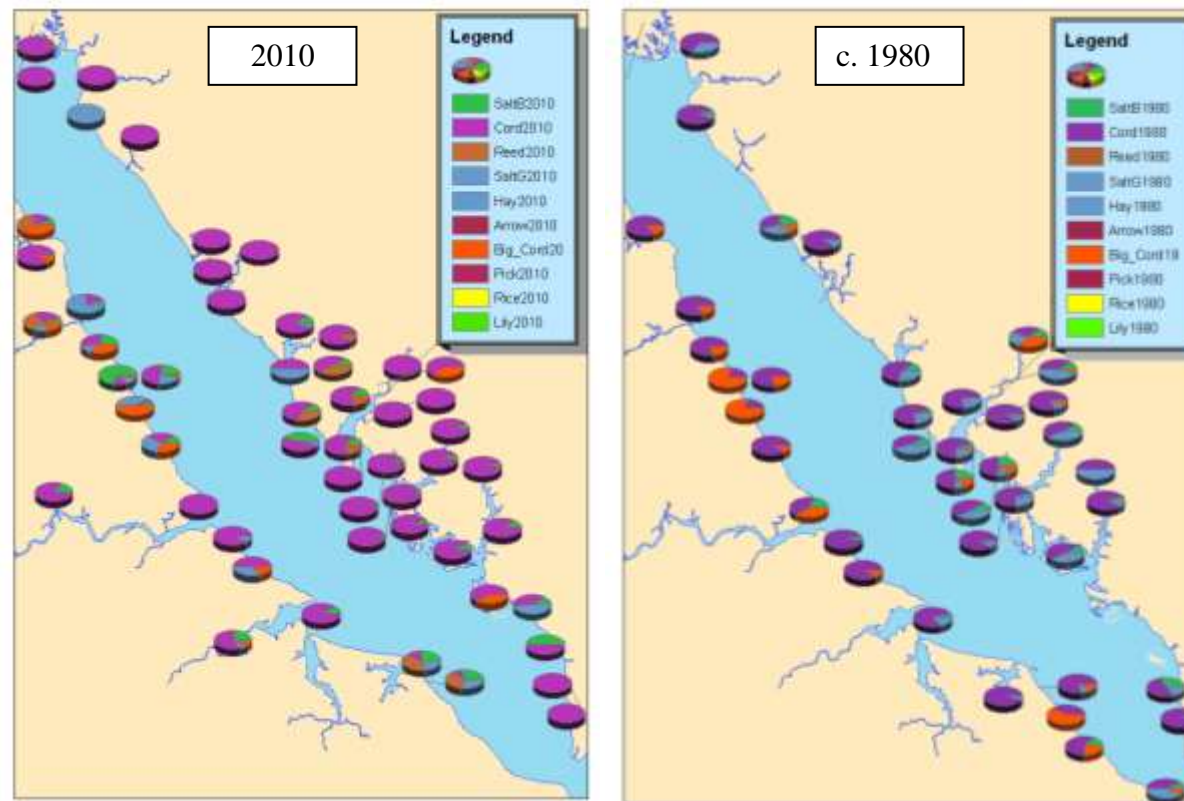


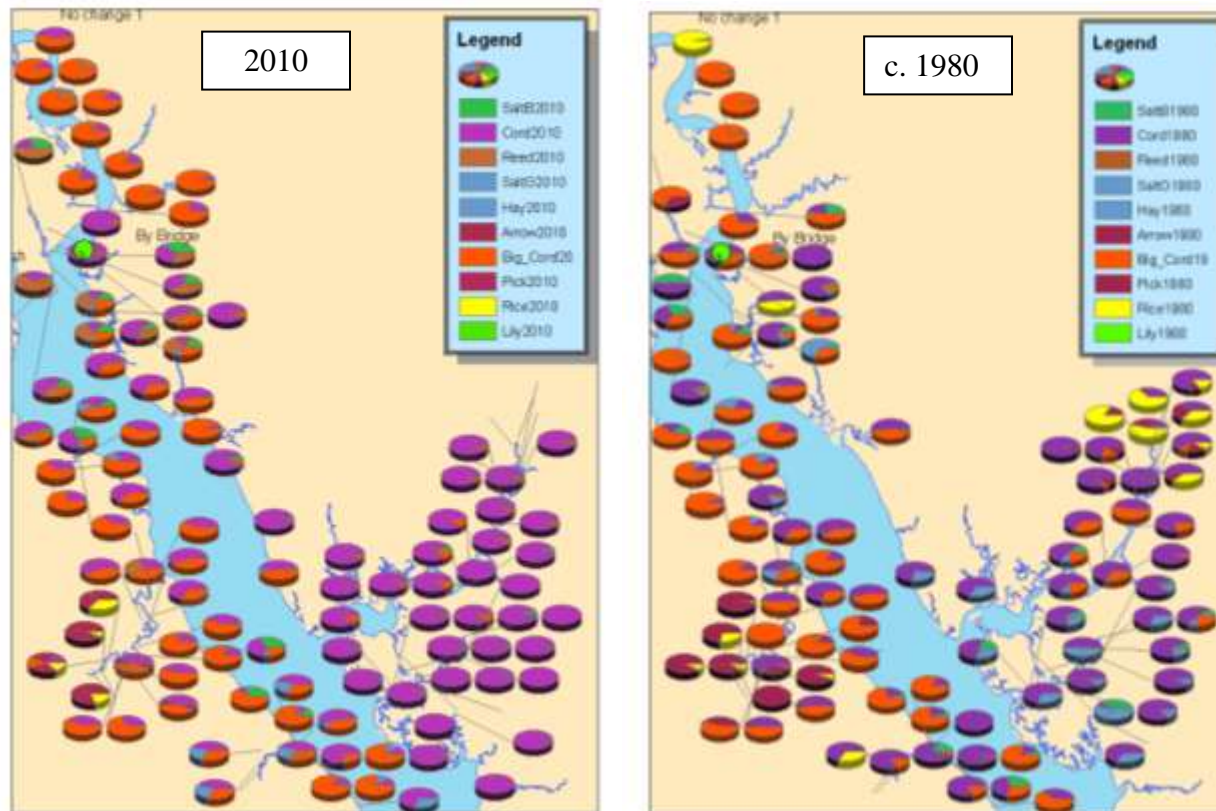
Figure 12. Plates of the York River Showing Dominant Species Composition in the Historic and Current Surveys



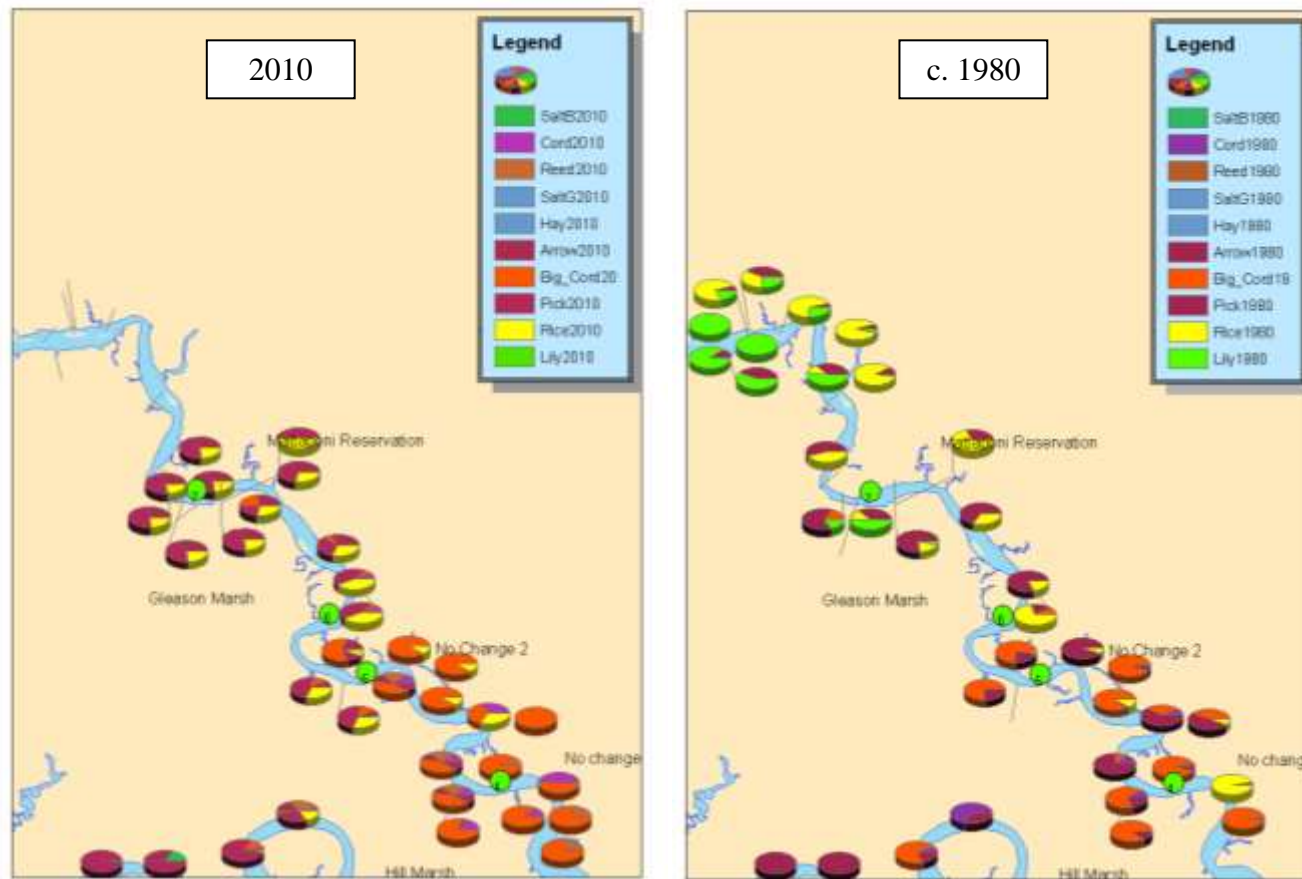
Lower York River. The overall trend in this section of the York River between the original surveys and the survey done in 2010 is a reduction in marsh diversity concurrent with the appearance of *Phragmites australis*. During the original survey, the marshes on the mainstem and the in the northern creeks were characterized as Type 1 (*Spartina alterniflora* marshes), but typically had significant low and high marsh components to their community, as well as a salt bush community. By 2010, the high marsh community has been lost from most of the sites, leaving only the low marsh and saltbush communities. The southern creek communities were dominated by *Spartina alterniflora* in both the original and current survey. However, there has been some loss of high marsh community and a few of the marshes now have significant *Phragmites australis* communities.



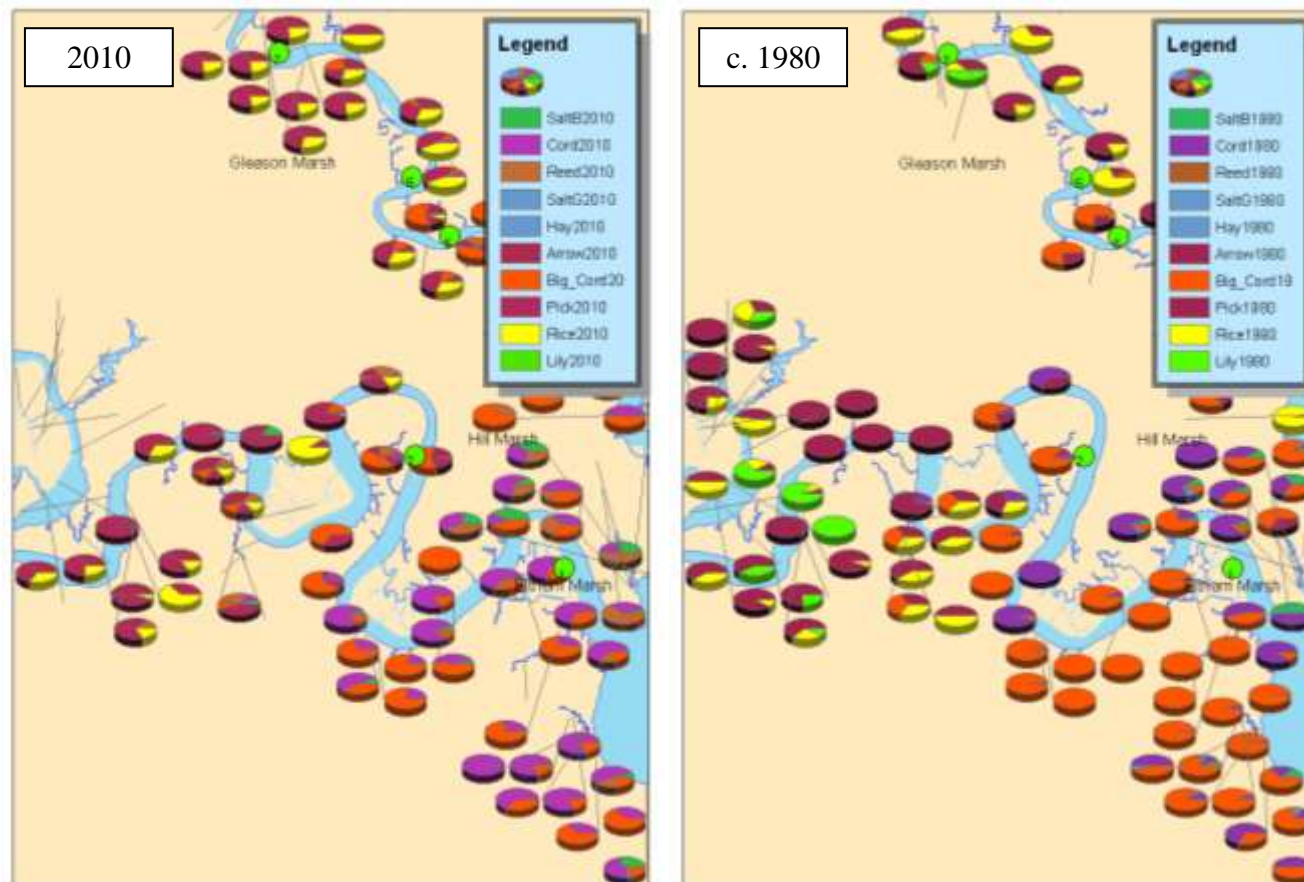
Mid-York River. Similar to the lower section of the York River, on the north shore, the overall trend has been a reduction in diversity, with marshes which used to have high and low marsh and occasionally a saltbush line, converting to predominately low marsh systems. Some of the marshes have shifted type (from a Type 12 to a Type 1), but most of them were historically Type 1, *Spartina alterniflora* marshes. One section of the river has several marshes where *Phragmites australis* has begun to grow; however, it is not overall prevalent on the northern shore of the river. Conversely, although the trend is less developed, diversity appears to have increased on the southern shore of the river, with marshes originally composed of *Spartina alterniflora* and *Spartina cynosuroides* now supporting diverse high marsh and saltbush communities. Several of these marshes have changed type (from either Type 5 or Type 1 to Type 12). *Phragmites australis* growing at 2 locations on this stretch of the southern shore, interestingly not at the same point as on the northern shore, but again is not prevalent.



Upper York River. There was little change in community on the southern shore of the York River during this time frame. The downstream marshes along the southern shoreline show some changes from a *Spartina cynosuroides* high marsh, to one that includes *Spartina patens* and *Distichlis spicata*. This may reflect a change in the salinity regime in this section of the river. The primary change in this area was the appearance of Phragmites in some of the marshes. The creek on the north shore has shifted from fairly diverse marshes with ¼ to ½ high marsh communities to ones that are almost entirely *Spartina alterniflora*. This pattern is true along the entire creek length, even though the historic community types suggest that the upper end of the creek was fresh water and the lower end of the creek was brackish.



Mattaponi River. The lower end of the Mattaponi River seems to be showing an increase in *S. alterniflora* presence (also *P. australis* presence). Although still not a dominant species in these marshes, increased *S. alterniflora* presence may indicate an upstream shift in the transitional boundary between fresh and brackish water. The upper marsh community shows little change and is typically a mix of *Zizania aquatica* and *S. cinusoroides*. *P. australis* presence is restricted to the lower end of the river where it can be found in approximately  $\frac{1}{4}$  of the marshes. In the upper portion of the Mattaponi River, the community shows little change. The apparent loss of Yellow Pond Lily from these marshes is likely attributed to differences in survey season. The historic survey went further upriver than the 2010 survey.



Pamunkey River. Similar to the Mattaponi River, the lower reaches of the Pamunkey River seem to have an increased *S. alterniflora* presence. Marshes which were once dominated by *S. cynosuroides* are now dominated by *S. alterniflora*. This may be indicative of a shift in salinity, and increase in sea level or both. *P. australis* can be found throughout most of the survey river, in approximately 1/3 of the marshes. The apparent loss of Yellow Pond Lily from these marshes is likely attributed to differences in survey season. The historic survey went further upriver than the 2010 survey.



Figure 13. Community Types Along the York River from both Surveys.

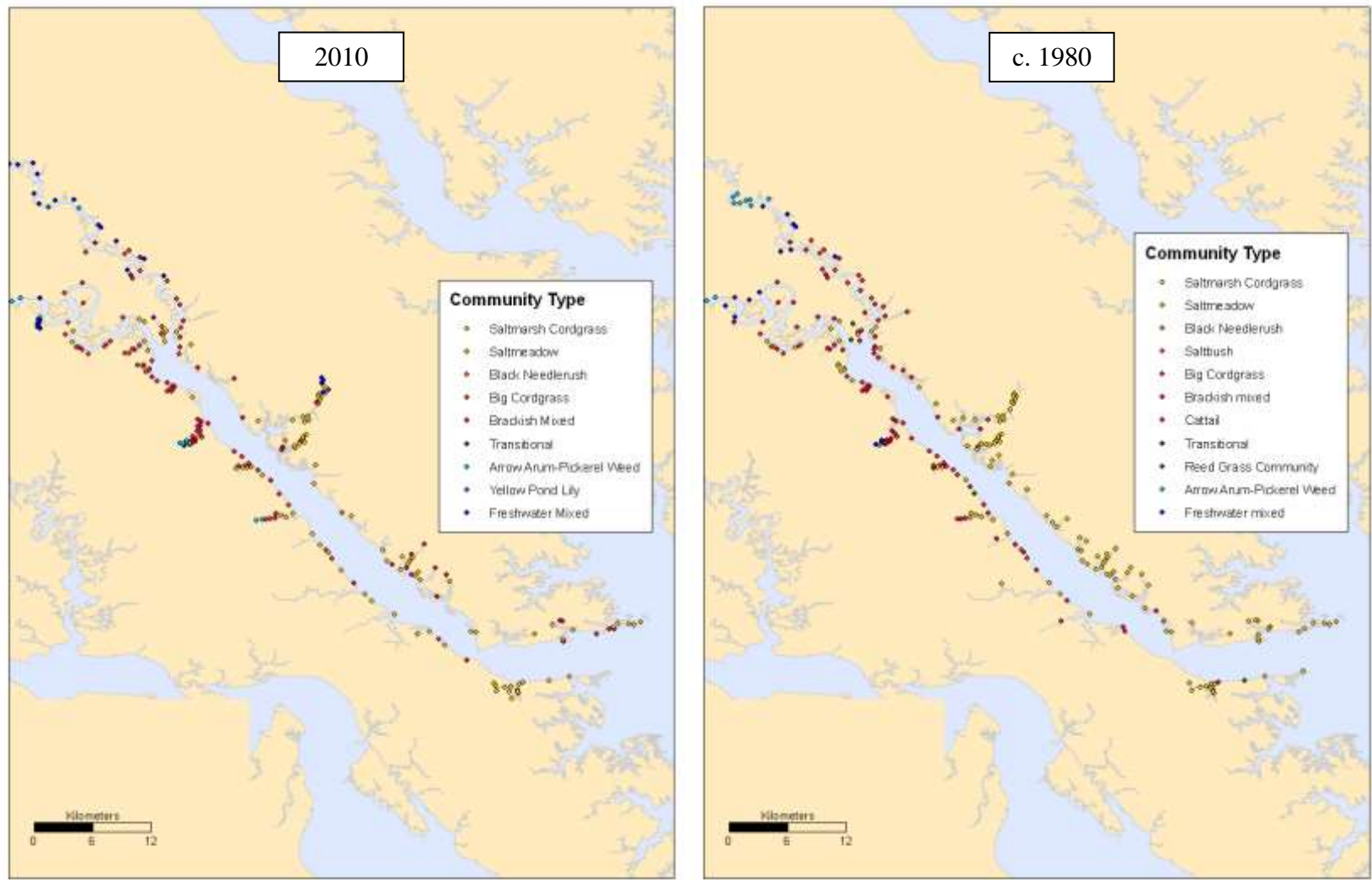


Figure 14. Characterization of Community Change Between Surveys.

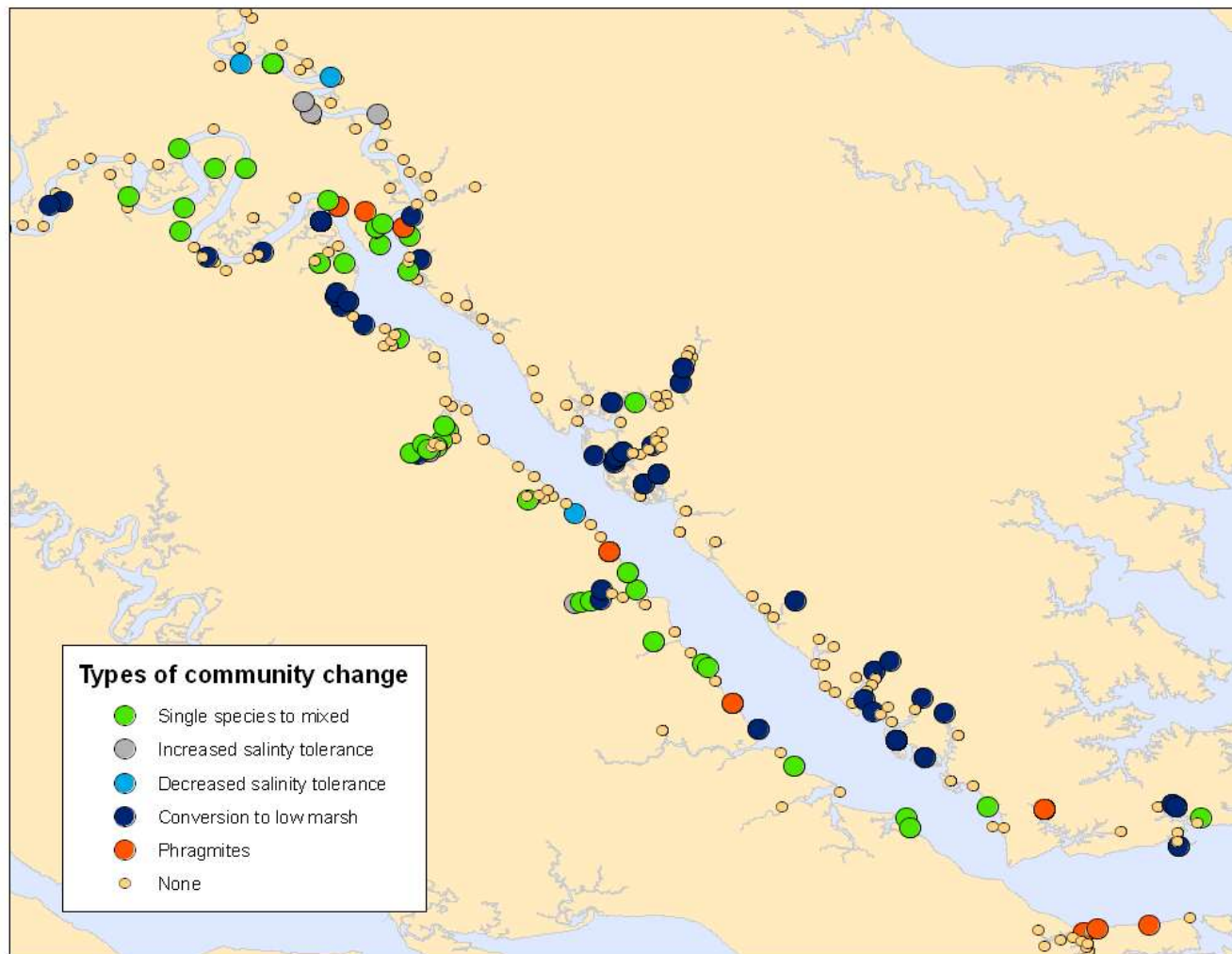
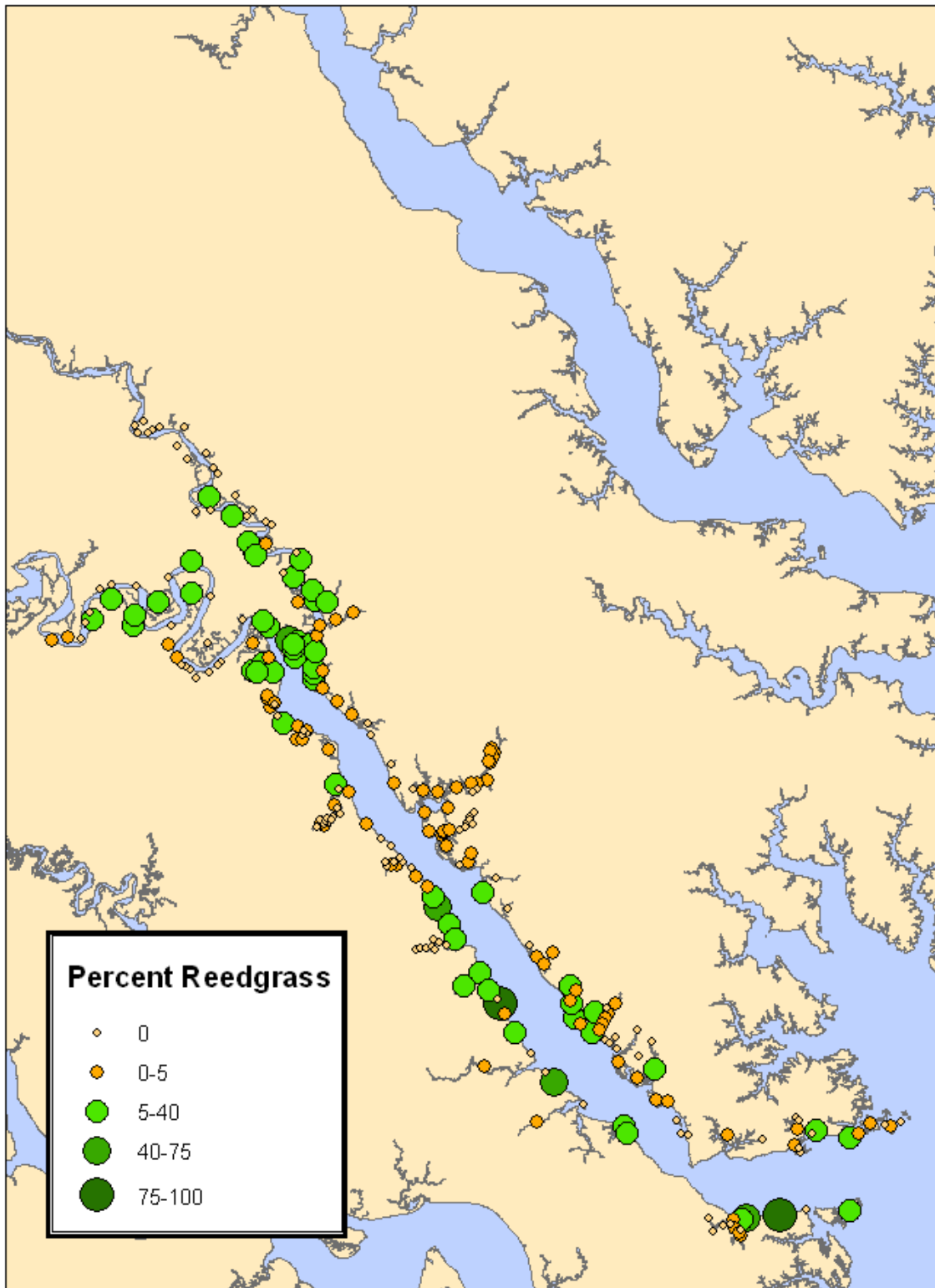


Figure 15. *Phragmites australis* distributions in 2010.



## Chapter 3: Long-term transect establishment and sampling

### Methods

Sites were selected using different criteria on the mainstem of the York and the two tributaries (Mattaponi and Pamunkey rivers) because the geological settings differ, leading to different dominant processes affecting community change in the two settings. Since the York River is brackish, we expect most long term change to be driven by sea level rise. On the Mattaponi and Pamunkey rivers, we expect both sea level rise and changing salinity to be important drivers of community composition. For the purpose of this study only fringe marshes were considered. This information was extracted from the tidal marsh inventory dataset developed using the high resolution basemap imagery (2009).

Based on the variables discussed above, 12 combinations are possible. ESRI's ArcGIS® 10 was employed to randomly select sampling points within each set of conditions (when possible, 4 sites per group of conditions were selected). The Subset Features tool located in the Geospatial Analysis Toolbox in ArcGIS® 10 was used for this purpose. Basically, this tool selects the number of features predetermined by the user and generates random values from a uniform [0,1] distribution. The final output is a file with random data points, which correspond to the marshes that need to be assessed. This random selection of sites was employed to assess the marshes when possible. If some of the random sites were not feasible to be reached (e.g. marshes that cannot be accessed by boat due to shallow waters), other marshes with the same conditions in the surrounded area were selected by the field crew.

On the York River 34 long-term monitoring sites were selected based on the following criteria:

1. The marsh was identified as a fringe marsh. (This category of marsh comprises the majority of the York River marshes and is arguably at the highest risk from sea level rise.)
2. There were no shoreline structures in place at the time of sampling in 2010. (Shoreline structures impede the natural landward retreat of wetlands. We expect marshes in front of structures to disappear as sea level rise continues.) These data come from the Shoreline Inventory developed at the Center for Coastal Resources Management, VIMS.
3. Sites were split between the north and south sides of the river and the creek systems. (Different processes may be affecting the marshes in these three systems.)
4. Sites were split between high and low bank sites. (High banks block landward retreat of marshes, but may help feed them while low banks allow for landward retreat.) These data come from the Shoreline Inventory developed at the Center for Coastal Resources Management, VIMS.

Sampling sites were selected from the population of sites matching these conditions using ESRI's ArcGIS® 10 to randomly select sampling points within each set of conditions (when possible, 4 sites per group of conditions were selected). The Subset Features tool located in the Geospatial Analysis Toolbox in ArcGIS® 10 was used for this purpose. Basically, this tool selects the number of features predetermined by the user and generates random values from a uniform [0,1] distribution. The final output is a file with random data points, which correspond to the marshes that need to be assessed. This random selection of sites was employed to assess the marshes when possible. If some of the random sites were not feasible to be reached (e.g. marshes that cannot be accessed by boat due to shallow waters), other marshes with the same conditions in the surrounding area were selected by the field crew.

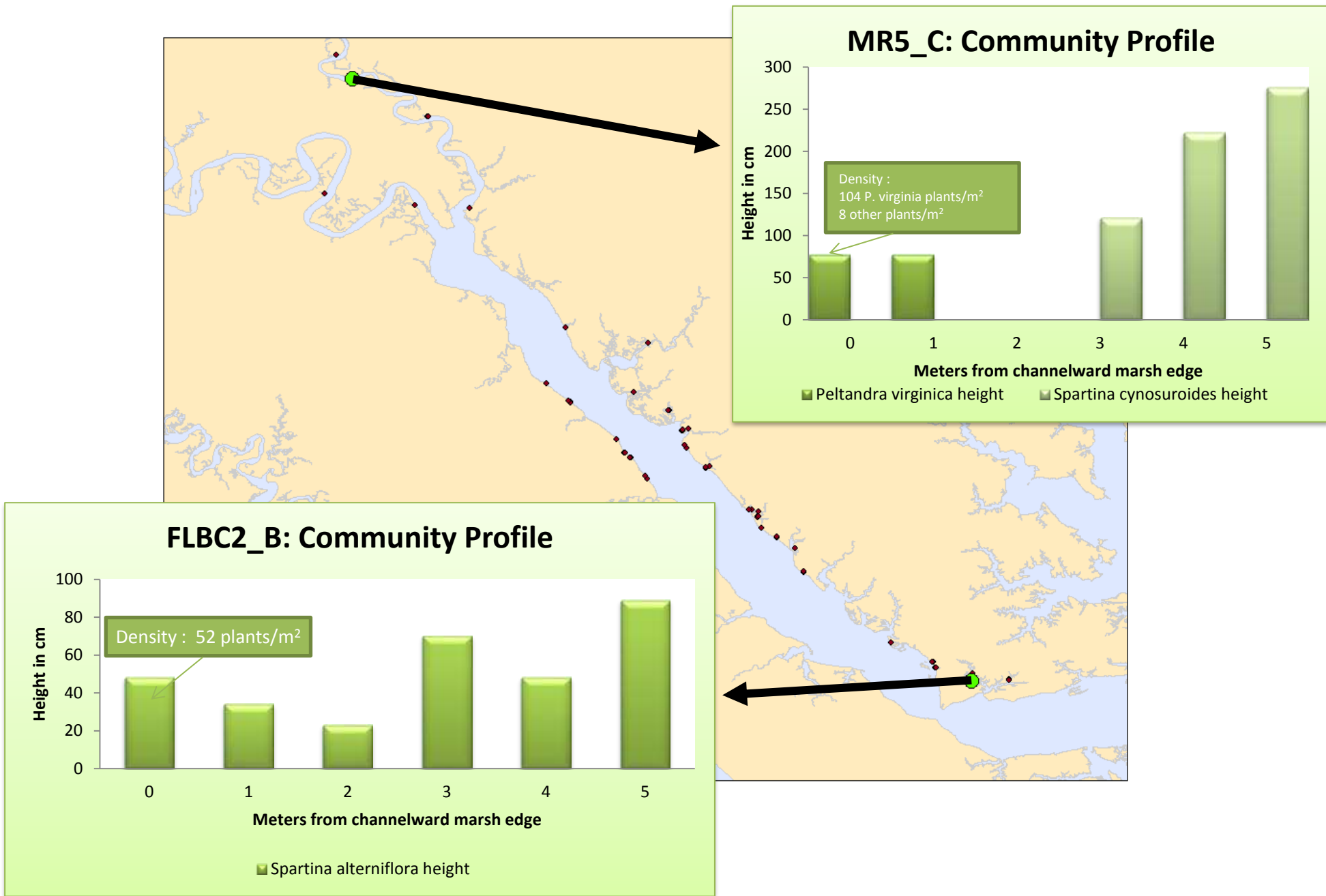
On the Mattaponi and Pamunkey rivers, six additional sites were selected based on community shifts from historic data. The sites were selected to cover a range of community shifts, with two sites showing an increase in low marsh species cover, two sites showing increase diversity, two sites with no change in community and one site where the percent of freshwater species has increased. Most of the sites selected for long-term monitoring transects were on the Mattaponi River because there are existing long-term monitoring efforts already in place on the Pamunkey River.

At each monitoring site, four transects were established approximately 10m apart. Transects extended from the channelward edge of the marsh landward to the saltbush line, or to 10m, whichever came first. Every meter, starting at the channelward edge of the marsh, plants were identified and measured. At 1m landward of the channelward end of the transect (low marsh, 1m into the marsh) and 1m channelward of the landward end of the transect (high marsh, typically 9m into the marsh) quarter-meter quadrats were established where plants were counted and identified. Only the channelward quadrat was established at sites where the low marsh extended greater than 10m into the marsh. Location of each transect and quadrat was marked via GPS to ensure a return to the exact location for future sampling.

## Results

Community data was collected from all of the transect sites. The community profiles will be converted into a georeferenced file for data examination and to make it easier to revisit the sites. Figure 16 shows the location of sites along the York River and an example of the type of community information collected at each site.

Figure 16. Map of the York River, VA, showing all transect sites (red and green dots). Community profiles from two transects (located at the green dots) are shown in the graphs.



## Chapter 4: Non-tidal wetlands

### Wetlands Water Quality and Impaired Waters

Decades of research on wetlands performance of water quality functions (uptake, sequestration, or modification of nutrients and sediments) has produced a vast array of findings that serve as testament to the natural variability of these systems. While the geochemistry of saturated soils and the presence of growing plants create a potential for wetlands to be a sink for some compounds and a source of others, precise net fluxes per unit area evade prediction.

Because absolute performance is so variable, the model we have used does not attempt to rate wetlands on the basis of the rates at which any of these processes may be occurring. We presume this is essentially unknowable if the goal is to assess a very large number of wetlands. Perhaps more importantly, our review of the extant literature leads us to conclude that the biogeochemical characteristics in a wetland that affect the rates at which these processes occur are largely beyond the purview of most management programs. Generally management programs are not focused on altering the redox potential in the soil profile of wetlands, or increasing the soil temperatures.

The water quality function model we have used makes a basic assumption that all wetlands have some capacity to perform water quality related functions. We further assume that this capacity is greatest when the system is not subject to any stresses that might degrade that performance. We further reason that the wetland processes that impact water quality are concentrated in upper layers of the soil column where plant roots and rhizomes are found. The model therefore considers stressors that would alter the natural hydrology of a wetland, reducing the potential interaction between compounds transported through the wetland and the root zone of the vegetation to be deleterious to these functions. Such stressors would decrease in the flow path or transmission time of waters entering the wetland from the surrounding landscape, and/or increase the thickness of the vadose zone in a wetland. The model assumes that these changes will effectively reduce the rate at which a wetland might otherwise perform water quality functions.

To develop a prediction of stressor occurrence, we examined the relationship between landuse/land cover patterns in surrounding landscapes and occurrence of channelized surface flows, incised streams, and “drier” wetland plant communities. The results indicated that the percent of a 200 meter buffer around a wetland that was developed land, the percentage that was natural lands, and the size of a wetland were the best predictors of the stressors that would alter a wetland’s hydrology.

Each of the three factors determined to have predictive power for water quality function stressors was found to have a threshold value for increased probability of stressors being found. In the case of developed lands, the probability of finding altered hydrology in a wetland increased when they constituted more than 1% of the area in a 200 meter buffer around the wetland. Similarly the probability of altered hydrology increased when natural lands were less than 71% of that area. Finally, wetlands larger than 2.4 acres were less likely to have altered hydrology than others.

These three observations were compiled for each mapped wetland. Each parameter was assigned a score with reference to the threshold value and the scores were simply averaged to generate a final characterization of the relative capacity of the wetland to perform wetland functions. The scoring system assigned a value of 1 to unimpaired systems and a 0.1 to highly stressed systems.

An analysis of average wetland water quality scores in relationship to the percent of impaired waters in 52 HUCs across ecoregions (Figure 17) in Virginia was conducted. The preliminary analysis found a modest inverse relationship between wetland water quality scores and amount of impaired waters. In HUCs with impaired waters greater than 2% of the total water surface area, HUCs with higher average wetland water quality scores had lower percentages of impaired waters (Pearson -0.30,  $p = 0.029$ ). An example is presented in Figure 18 and the complete data is available at [http://139.70.26.78/JSViewer\\_nontidal\\_wtlnDs/wetlandviewer.html](http://139.70.26.78/JSViewer_nontidal_wtlnDs/wetlandviewer.html)

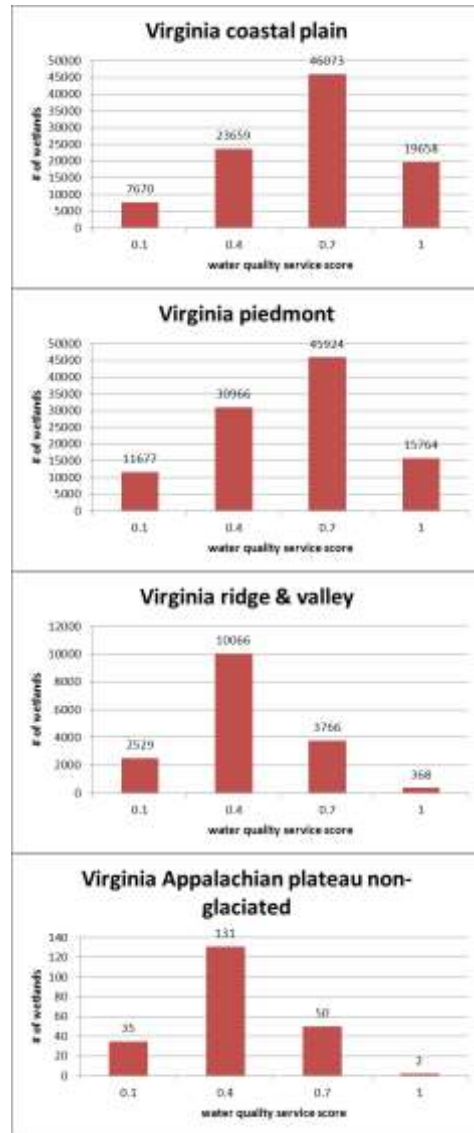


Figure 17. Water quality service score frequency distribution for Virginia physiographic provinces.



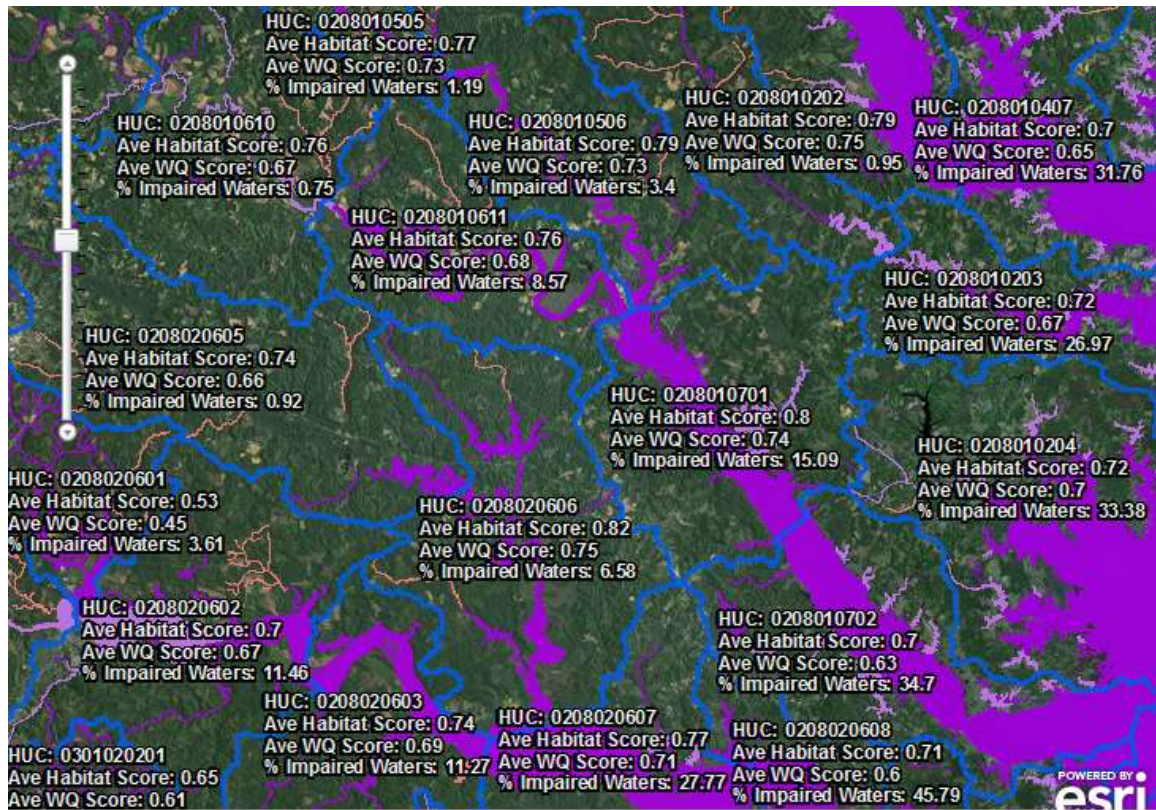


Figure 18. Example of average wetland water quality scores and percent impaired waters within 10 digit HUCs in Virginia.

## **Chapter 5: Management implications**

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