

ABSTRACT

Title of Document: REMOTE SENSING OF TIDAL
FRESHWATER MARSH ELEVATION,
CHANNELS, AND VEGETATION
STRUCTURE

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I mapped three elevation classes in a Mid-Atlantic tidal freshwater marsh using QuickBird multi-spectral imagery and field measurements of elevation and channel networks. The elevation map reveals heterogeneous surfaces at a 2.4 m pixel scale. Field collected vegetation cover data differed among the three elevation classes. Species richness and the number of positive associations among species were higher in the mid- and high-marsh than the low marsh. The number of negative associations among species increased with rising elevation. Random forest classification of elevation class using species cover vectors selected only *Impatiens capensis* and *Nuphar luteum* and had an out of box predictive error of 26%. My research shows that the vegetation of freshwater tidal marshes is diverse with ill-defined boundaries between distinct communities. Yet vegetation shifts beyond the traditional low and high marsh communities could be detected, which should provide a useful tool for restoration and management.

REMOTE SENSING OF TIDAL FRESHWATER MARSH
ELEVATION, CHANNELS, AND VEGETATION STRUCTURE

By

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Chapter 1: Introduction

Motivation for the study

Tidal freshwater marshes are highly productive transitional zones that provide ecosystem services vital to the health of our coastal regions and oceans. They improve downstream water quality by trapping sediments, toxins, and nutrients (Bridgham et al., 2006; Johnston et al., 1984; Mitsch et al., 1995; Orson et al., 1990), and also by removing nitrogen to the atmosphere via microbial denitrification in the sediments (Hume et al., 2002; Hunter and Faulkner, 2001). They stabilize coastal shorelines in storm events (Odum and McIvor, 1990; Sather and Smith, 1984), maintaining land area and further reducing sediment loads in adjacent waterways. Each year, tidal freshwater marshes produce seasonal biomass similar to the primary production seen in tropical rainforests (Mitsch and Gosselink, 2000). This primary productivity, along with environmental and vegetative diversity, makes these marshes home to many terrestrial and aquatic animal species. At least 280 bird species and 102 amphibians and reptiles are associated with tidal freshwater marshes (Odum et al., 1984). And many commercial fisheries species use tidal freshwater marshes as nurseries (Brundage and Meadows, 1982; Odum et al., 1984), while other fish populations persist there for the entire span of their lives (Mitsch and Gosselink, 2000). Given the important ecosystem and habitat functions they offer, it is important to maintain and enhance tidal freshwater marshes in coastal estuaries like the Chesapeake Bay region, the location of the current study.

Tidal freshwater marshes, along with coastal salt marshes, salt and freshwater tidal swamps, and the many varieties of terrestrial wetlands, are disappearing in the United States. Approximately one third to one half of all wetlands in the coterminous United States have been lost to anthropogenic activities over the last 200 years (Tiner, 1984). In an attempt to halt and, if possible, reverse losses, the recreational, environmentalist, scientific, and policy communities joined together in the 1970s to draft policies encouraging preservation of our remaining wetlands. They also sought to compensate for future losses through restorations and new wetland construction projects.

However, such restorations and new constructions, frequently referred to as “mitigations,” have met with mixed results. A 2001 review of compensatory wetland mitigations published by the National Research Council (NRC) found that, despite 20 years of progress, wetlands were still undergoing net losses. Most of the projects reviewed failed to compensate for losses either because the actions required in their permits were never attempted, were only partially attempted, or, even when fully implemented, suffered from poor designs or workmanship (Gladwin and Roelle, 1992; NRC, 2001; Race and Fonseca, 1996; Sudol and Ambrose, 2002; Zedler, 2000). The NRC committee found many reasons for this, including inconsistent policy implementation; a focus of human and material resources on permitting and acreage allotments instead of long-term tracking of ecosystem services in projects already implemented; a lack of watershed perspectives when choosing sites; and a scientific knowledge-base that was too small to ensure consistent mitigation outcomes (NRC, 2001).

My thesis work focused on increasing the knowledge-base informing the expansion of one particular marsh: Dyke Marsh Preserve, a tidal freshwater marsh in Fairfax County, VA, that was extensively mined for sand and gravel prior to 1972. The National Park Service, which took over direct management of the marsh in 1976, is currently considering whether degraded areas can be successfully restored. The goal of my study was to characterize and map the existing marsh vegetation as it relates to topography. It is hoped that mimicking current marsh topographic structure in newly restored areas, in combination with some targeted planted of desired species, will assist in promoting desired vegetation assemblages. I used analytical techniques that are broadly applicable to tidal freshwater marshes in the eastern United States. Thus, my study should be useful to restoration and monitoring projects throughout tidal freshwater portions of the Chesapeake Bay estuary.

Emergent macrophytes in tidal freshwater marshes

The spatial and temporal patterning of emergent vegetation in Eastern U.S.A. tidal marshes has been widely studied (Baldwin et al., 2001; Bertness, 1991; Capers, 2003; Havens et al., 2002; Leck and Simpson, 1994; Odum, 1988; Odum et al., 1984; Perry and Hershner, 1999; Peterson and Baldwin, 2004). However, while clear zonation patterns dependent on hydrology and salinity gradients have been repeatedly documented in coastal saltwater marshes (Bertness, 1991; Havens et al., 2002), vegetation patterning in tidal freshwater marshes is less well understood.

It is likely that zonation patterns are not widely reported for tidal freshwater marshes because, unlike their coastal cousins, which are strongly limited by salinity gradients and dominated by small numbers of perennial species, freshwater marshes

sustain large numbers of both perennial and annual species that persist in a variety of habitats subject to different hydroperiod regimes (Leck and Simpson, 1994; Odum et al., 1984; Simpson et al., 1983). For example, in their work in a New Jersey tidal freshwater marsh on the Delaware River, Leck and Simpson (1994) found that *Nuphar advena* occurred only in tidal channels and on adjacent banks, *Acnida cannabina* and *Ambrosia trifida* concentrated along stream banks and levees, *Acorus calamus* occurred in high marsh locations, and *Polygonum arifolium* and *Peltandra virginica* occurred everywhere in the marsh. Because many freshwater macrophyte ranges overlap in this way, tidal freshwater marshes are likely to support mixed, or “fuzzy,” vegetation assemblages with little discernable zonation,

Field studies of the fuzzy species distributions in tidal freshwater marshes can be further confused by temporal variation. Annual and perennial species tend to mature at different rates, which leads to seasonal shifts in the dominant species observed (Baldwin et al., 2001). Interannual variation in annual species also occurs, driven by such factors as the availability of seeds (Leck and Simpson, 1994) and propagules (Capers, 2003), inter-specific competition (Gaudet and Keddy, 1995b; Wilson and Keddy, 1986), and environmental factors that affect seedling establishment, such as hydroperiod (Baldwin et al., 2001; Leck, 1996), sedimentation or burial (Leck, 1996; Peterson and Baldwin, 2004), temperature and light conditions (Leck, 1996), and animal disturbance (Baldwin and Pendleton, 2003). Experimental findings for emergent macrophytes in tidal freshwater settings can therefore not always be generalized to other seasons or years.

Most of the literature restricts itself to categorizing low marsh species assemblages dominated by the floating-leaved hydrophyte *Nuphar advena* and broad-leaved monocots, like *Peltandra virginica* and *Pontedaria cordata*, which are then contrasted with high marsh areas of mixed annuals and perennials (Odum et al., 1984). General marsh locations and elevation ranges have been documented for individual species in several tidal freshwater marshes (Leck and Simpson, 1994; Odum et al., 1984; Simpson et al., 1983), but studies that map vegetation with respect to elevation or other environmental gradients remain elusive. However distributions of many species are strongly affected by hydrology, and elevation is correlated with hydrologic drivers. This provides an opportunity to map the highly heterogeneous vegetation found in tidal freshwater marshes with respect to the marsh's elevation gradient. My thesis developed a methodology for this kind of mapping, using a combination of remotely sensed data and field data.

Remote sensing of vegetation

Aerial photography, remote sensing satellite platforms, and sophisticated computer processors and software have ushered in a new era of data-intensive spatial analyses. In essence, remote sensing analysis involves evaluating the light reflected by the objects and captured in an image. In the case of vegetation, reflectance in the visible part of the spectrum is controlled by the absorption of blue and red light by the chlorophyll present in plant tissues (Curran, 1980), while reflectance in the near infrared (NIR) portion of the spectrum is controlled by changes in the refractive index within the mesophyll tissues of plant leaves (Verdebout et al., 1994). Size and orientation of leaves, vertical arrangement of the canopy, the degree of senescence,

underlying soil reflectance, and the degree to which this soil is obscured by the vegetation canopy will combine to produce a canopy reflectance value for any given digital point, or pixel, in a remotely sensed image of vegetation (Curran, 1980).

At the site of the current study, vegetation reflectance should be one important contributor to the canopy reflectance captured by the satellite. The image was taken late in the growing season and is therefore likely to contain some species-specific variability due to seasonal senescence patterns. Additionally, organic content of the soils at the site is generally high, and it is possible that soil moisture will vary with length of exposure in the tidal cycle, due to the slow draining associated with highly organic soils. This soil moisture gradient would affect canopy reflectance anywhere that the vegetation canopy is incomplete. Also important at this site is surface water beneath or immediately adjacent to the vegetation, as light reflected by the vegetation will be mixed with adjacent reflectance before it is received and recorded by the satellite.

Satellite image classification involves the use of multivariate techniques to separate canopy reflectance values and ancillary data into similar and dissimilar classes. The resulting classes may then be evaluated for physical or vegetation differences. The objective of the classification is to minimize within-class reflectance variance while maximizing between-class variance, thereby producing distinct classes that contain similar pixels. The data collected by sensors is often analyzed directly or may be used to create band ratios or spectral indices that combine specific wavelength data in various ways to enhance information that can be used to identify classes of interest (Armitage et al., 2000).

The spectral information in the image is tied to vegetation composition or physical variables by collected field, or “ground-truth” data that details the nature of the features making up each spectral class. Ground truth data can either be collected prior to image classification and used to inform the choice of classes in the multivariate analysis, a process called “supervised classification,” or it can be gathered after the fact in “unsupervised classification” by going into the field to determine which plants make up classes delineated in the image beforehand. In either case, the identified vegetation classes may then be color coded or outlined in the image to produce a map detailing site features.

This method of image analysis, while suitable for some vegetation studies, is somewhat limited at small spatial scales or heterogeneous sites, as it was originally developed to produce spatially explicit land-use maps that, for example, might identify open water, urban, agricultural, and forested areas. Phytosociologists and ecologists, on the other hand, have traditionally classified vegetation according to species composition, rather than by spectral reflectance. The two classification approaches are often at odds, as the classes developed from an analysis of species composition may or may not correspond to spectral classes. For example, two different ecologically derived vegetation classes might reflect similarly in an image, or a single such class might produce different reflectance values if, for example, the canopy were structured differently or the underlying soil reflectance differed. Much of the recent literature in involving remote sensing for vegetation analysis has revolved around developing ways to classify spectral information in ecologically informative ways (Lu and Weng, 2007; Richards, 2005).

One such method involves masking or excluding certain portions of the spectral data from the analysis to improve classification interpretability in the remaining data. Urban or open-water areas that might be ecologically uninformative can be removed from the dataset prior to classification to prevent their influence on the multivariate analysis. Similarly, the dataset can be iteratively broken into subsets, which are analyzed independently of their parent datasets. This method was employed by Kandus et al. (1999) in their development of a regional land cover classification scheme and by Townsend and Walsh (2001) in their development of a flood-plain forest classification. The latter case also illustrates how spatially explicit and ecologically relevant information, independent of the spectral data in the image, may be included in the masking or partitioning process. Relying on soil maps of their study area, Townsend and Walsh divided their forest land cover class into sites occurring on mineral soils and sites occurring on organic soils before continuing with further spectral classifications of these two forested groups.

Another suite of remote sensing techniques addresses the fact that naturally occurring vegetation species rarely exist as discrete groups in the landscape, but rather grade into one another or overlap extensively. Thus, a given pixel may, in fact, be appropriately assigned to more than one class. Dennison and Roberts (2003) addressed this difficulty by spectral mixture analysis, in which individual pixel reflectance values are treated as blended values, or combinations of the predetermined “pure” reflectances of which they are composed. Townsend (2000) developed a fuzzy set similarity index that can be used when validating a classification to quantitatively assess the degree to which any given element in an image is similar to

the class to which it has been assigned. Schmidtlein and Sasson (2004), in their analysis of grassland species distributions, avoided spectral and ecological classification altogether by mapping the ways in which individual species graded into one another along spatially correlated environmental gradients.

My original intent in this study was to map vegetation classes directly and then evaluate their relationships to elevation and tidal channel networks. However this proved problematic owing to heterogeneity and spectral confusion among the final vegetation classes, and also because the data were unevenly distributed among the vegetation classes. Seasonal change prohibited the collection of additional data that might have overcome such difficulties, so I turned to the possibility of mapping elevation classes instead, which could then be evaluated for vegetation structure. This approach proved viable, probably because vegetation assemblages, soil moisture, and surface water reflectance are likely to be closely tied to elevation and channel networks in this setting.

I used classification trees to develop spectrally distinct elevation classes. Classification trees use recursive binary splitting to create “trees,” in which each branching divides the data into two groups of observations such that individuals within a group are more like each other than they are like members of the other group (McCune and Grace, 2002). The branch-points, called “splitting nodes,” will detail a succession of splitting rules that continue to divide the data until an acceptable threshold, such as minimum sample number or homogeneity of classes, is reached in the terminal nodes. Recursive binary trees can be created with ground truth data, and splitting node information can then be used in decision tree mapping to allocate all

pixels in an image to their respective terminal node memberships (Research Systems Inc., 2005). A couple of factors make this methodology appealing for this study. First, it is a non-parametric procedure that makes no assumptions about underlying distributions in the data. This is an important consideration because the vegetation community data captured by the image will rarely be normally distributed (McCune and Grace 2002). Secondly, decision trees can easily incorporate non-spectral data, nominal, ranked, or continuous data, and data measured on different scales. This allowed me to incorporate channel size and proximity information in the image classification, so that channel influence on soil moisture could be accounted for when analyzing the imagery for elevation differences.

I used the random forest algorithm in R statistical computing software for the classification trees. This method, originally developed by Leo Breiman and Adelle Cutler (Breiman, 2001) in Fortran, has been implemented in R statistical software by Andy Liaw and Matthew Wiener (Liaw and Wiener, 2002). The random forest algorithm grows a series of bootstrapped trees that each employ randomly selected predictor variables and leave out different randomly selected “out-of-bag” subsets of the data. The “forest” of these accumulated trees is then evaluated to assess how well each splitting variable reduces final within-class distance in multivariate space, and a final averaged tree employing the best predictor variables is created (Breiman, 2001; Liaw and Wiener, 2002; Prasad et al., 2006). I chose this algorithm, which has been shown to perform well with ecological data (Prasad et al., 2006), for several reasons. First, it allowed me to review the ranking of each predictor variable’s ability to effectively divide the data. Secondly, the final unbiased variable selection and

multiple out-of-bag trees contributing to the final tree eliminated the need for trimming because the final tree is unbiased by nature (Prasad et al., 2006). This allowed for the creation of detailed trees that may illuminate complex non-linear relationships in the data. Finally, because so many individual trees are grown in the random forest algorithm (generally 500 to 2000), and because the out-of-bag samples from all of these trees are used to create an unbiased error rate for the final tree, the predictive error for the final tree is likely to be closer to the true predictive error of the population than the error associated with any single tree or validation sample would be (Prasad et al., 2006).

Overall, techniques for answering ecological questions with remotely sensed data are still developing rapidly. While I chose to use random forest decision tree mapping, many other methods are currently available. For an extensive account and comparison of 64 classification techniques currently employed in spectral classification and mapping, see Lu and Weng (2007). Richards (2005) also provides a recent summary of developments. It is likely that new approaches will continue to appear in the literature as researchers grapple with ways to answer specific ecological questions with emerging technologies and increasingly sophisticated multivariate techniques.

Study site

Dyke Marsh Preserve is a freshwater tidal marsh in Fairfax County, Virginia. It lies just south of Washington, DC, on the west bank of the Potomac River, with a centroid of 321951 E and 4292947 N in UTM zone 18N in the WGS84 datum. Its

formation began in the Pleistocene Era, 5,000 to 7,000 years ago, and has largely been due to the deposition of sediments carried in Hunting Creek, which drop out of the water column when they reach the relatively slower waters of the Potomac River (Palermo and Zeigler, 1976). Early European explorers recorded that local indigenous tribes favored Dyke Marsh Preserve for hunting and fishing (Parsons et al., 1976). In the early 1800's settlers built dikes around the marsh to create grazing land. When the resulting pastures did not prove suitable for grazing, the maintenance of the dikes ceased, and tidal channels eventually breached the dikes and reestablished intertidal marshland (NPS, 1977; Palermo and Zeigler, 1976).

In more recent times, Smoot Sand and Gravel Corporation began mining open water areas immediately adjacent to the marsh for sand and gravel around 1950. The National Park Service took ownership of the marsh in 1959 and allowed dredging at the marsh to continue until concerns over wildlife habitat preservation and increased environmental degradation prompted a dredging moratorium in 1972. At that point, marsh acreage had shrunk from about 263 hectares of wetland, recorded in the 1930s, to about 196 hectares, 81 of which had been dredged (NPS, 1977). The Park Service, which took over direct management responsibilities at the site in 1976, is currently considering whether Dyke Marsh Preserve can be successfully restored.

Dyke Marsh Preserve is located in the Atlantic Coastal Plain in Virginia, where oceanic deposits overlay older crystalline rocks (Johnston, 2000; NPS, 1977). The marsh sits on approximately 15 m of interlayered gravel, sand, mud, and clay. The layering represents changes in hydrological conditions over time, where slow moving waters allowed fine sediment deposition, and swift waters allowed only

heavy particle deposition (NPS, 1977). In 2004, soil samples taken from the top 10 cm of Dyke Marsh Preserve ranged from silty clay, to silty clay loam, to silty loam, all with organic matter content between 60 and 90 percent (Hopfensperger et al., 2004).

Small hills border the western edge of the marsh, and the Potomac River borders the eastern edge (NPS, 1977). Elevation in the emergent marsh varies from below mean-low-tide to greater than 1 m above mean-low-tide. Highest marsh areas are therefore less regularly inundated by the location's approximate 1m tidal range than are lower areas (NPS, 1977). Topography is marked by tidal channels of varying size and by elevation changes, with relatively high elevations at the northeastern border of the central marsh, lower areas in the middle of the marsh, and gradually rising elevations towards the upland areas to the west. Water movement is still somewhat restricted by the remaining dikes, and the marsh generally fills and drains via tidal creeks and interior tidal channels (Harper and Heliotis, 1992). Harper and Heliotis found that overall flushing rate, based on mean high and low tide volumes, was $0.92 \text{ m}^3/\text{sec}$, and that dye tests confirmed the existence of high flushing rates per tidal cycle throughout the marsh. In extreme weather events, particularly storms from the northeast, even the highest elevations may remain flooded for several days.

Water quality in Dyke Marsh Preserve is strongly influenced by anthropogenic pressures and is marked by high turbidity, high nutrient loads, and some heavy metals and toxins (Johnston, 2000; NPS, 1977). In the past, high turbidity and poor water quality contributed to the loss of submersed aquatic vegetation (SAV) and dependent herbivorous populations (Johnston, 2000; NPS,

1977; Rybicki and Landwehr, 2007). However, intermittent data collected in the Potomac River near Dyke Marsh Preserve between 1962 and 2004 shows gradual water quality improvement (CBP, 2004; Johnston, 2000). And Rybicki and Landwehr (2007), in their 1985 to 2001 study of water quality, SAV populations, and dependent waterfowl populations in the tidal Potomac River and Estuary, including the portion bordering Dyke Marsh Preserve, found a reduction in total nitrogen in the water, and large, interannually variable increases in both SAV biomass and diversity.

Vegetation in Dyke Marsh Preserve is characterized by three major communities: tidal freshwater marsh, swamp forest, and floodplain forest. In a survey conducted for the National Park Service in 1991, Xu inventoried 373 species, 60 of which were obligate wetland species. He estimated that his team inventoried 90% of all specimens in the marsh, including flood plain forest, swamp forest, and emergent macrophyte species. Vegetation surveys in 2003 and 2004 (Hopfensperger et al., 2004) of the emergent marsh, the focus of this study, documented 30 species in 2003, and 31 species in 2004, with *Peltandra virginica*, *Impatiens capensis*, and *Typha* spp. occurring in over 50 percent of plots sampled in both years. Other frequently observed species included *Nuphar luteum*, *Acorus calamus*, *Bidens laevis*, *Schoenoplectus fluviatilis*, *Polygonum arifolium*, *Polygonum punctatum*, and *Leersia oryzoides*. Seed bank studies from the same years (Hopfensperger et al., 2004) found that 12 out of 25 species in 2003, and 11 out of 32 species in 2004, were annuals. Hopfensperger (2007), examining seed bank and standing vegetation relationships at Dyke Marsh Preserve over three years, found interannual shifts in species composition at a small spatial scale, but not in the marsh overall. Further, she found

that standing vegetation from the previous year did not strongly influence the following year's seed bank, but that seed bank and standing vegetation from the same year were similar and dependent on landscape structure. Specifically, higher elevations and locations further removed from channels showed higher similarities between the seed bank and standing vegetation than lower elevations close to channels. These findings suggest that vegetation assemblage composition in Dyke Marsh Preserve is largely dependent on tidal mixing and the attendant removal and delivery of seeds by tidal scouring and deposition.

Objectives and hypotheses

Dyke Marsh Preserve provides a unique management opportunity because a natural marsh remains at the proposed expansion site and can be used both as a template in restoration design and as a mirror in which to evaluate progress in restored areas over time. Because field sampling to discern vegetation structure is labor intensive, especially at broad spatial scales, my first objective was determine whether spectral and ancillary information could be used to locate classes of interest in the field data and extrapolate this information to unsampled areas, thereby creating a map.

Preliminary evaluation of phytosocial vegetation classes based on field samples showed that such classes were both unevenly sampled and spectrally confused. Because additional sampling of undersampled classes was not possible, due to end of season assemblage shifts, I chose to map elevation classes, which could be reliably mapped with reliable user accuracies.

Because tidal channels focus the delivery of water, nutrients, and water-dispersed seeds and propagules throughout the marsh, variations in channel size and proximity are likely to affect both soil moisture and vegetation composition at any given location. I therefore included a channel size and proximity metric in the classification to help quantify vegetation and soil moisture differences not attributable to elevation alone.

My second objective was to look for vegetation differences among elevation classes, testing the hypothesis that the plant species in tidal freshwater marshes are distributed, at least in part, across an elevational gradient that reflects the varying abilities of individual species to establish themselves and persist in locations with different hydroperiods. While Dyke Marsh Preserve is alternately flooded and drained twice daily, some areas are saturated more frequently and for longer periods than others. My study therefore tested whether the small elevation gradient present at Dyke Marsh Preserve acts as a forcing function for vegetation selection.

Beyond the question of environmental forcing gradients lies the more basic question of whether or not tidal freshwater marsh vegetation does, in fact, grow in discernable patterns that can be spectrally identified and mapped. These marshes are bathed with fresh nutrients from incoming tides twice daily. Wastes are likewise flushed twice daily. Because wetland plants are structurally and physiologically adapted to withstand flooded conditions and because it is unlikely that species need to compete for abundant soil resources, making light availability the most obvious limiting resource, it is possible that tidal freshwater vegetation is not structured and distinct vegetation communities can not be discerned. My alternative null hypothesis

was, therefore, that vegetation at Dyke Marsh Preserve is randomly distributed with respect to elevation. If the null hypothesis is not falsifiable, then vegetation classes cannot be mapped, at least not in an ecologically meaningful way.

In summary, my study sought to identify spectrally distinct elevation classes at Dyke Marsh Preserve and test for expected relationships between vegetation species and elevation. My goal was to both catalog current emergent macrophyte distributions and find meaningful topographical relationships that would assist ecological engineers in their designs for the marsh expansion.

Brief summary of methods

A high spatial resolution QuickBird image was collected in late September, 2005. I ran an unsupervised ISODATA classification on the image to develop a stratified random sampling scheme that would help insure collection of enough spectrally distinct samples to stabilize subsequent modeling. Following the sampling scheme, I took field measurements of GPS location, elevation, and species percent cover estimates within 1 m quadrats at 102 sampling sites in early October.

In addition to elevation and vegetation sampling, I digitized a channel map, referencing both the tasked satellite image and a leaf-off aerial image from the previous year. I followed this task with cross-sectional areal field measurements of channels informed by the channel size classes that could be visually discriminated in the images. These channel classes were used as weights in calculating the Cumulative Inverse Squared Index (CISD) values (Sanderson et al., 2001) that I used to quantify channel influence on the sampled vegetation. The CISD index measures the distance from the sample site to a series of 5-meter increments along the nearest

channels within a 25 m radius. The index then computes an aggregated inverse squared distance score, weighted by average channel cross-sectional area for each site.

To determine spectrally distinct elevation classes of interest, I used PC-Ord software (McCune and Mefford, 1999) to run a series of 4-, 5-, and 7-class flexible beta classifications in of the QuickBird multi-spectral data, NDVI, a similar index employing green and red data $((\text{green}-\text{red})/(\text{green} + \text{red}))$, and a Principle Components Analysis reduction of the multi-spectral image and indices. I then plotted class frequencies on elevation and evaluated them to select appropriate boundaries for the elevation classes. I followed with a classification tree to identify spectrally confused classes, which I then combined to yield the final elevation classes: 1) elevation under 0.2 m, 2) elevations between 0.2 and 0.4 meters, and 3) elevation over 0.4 meters.

To create the elevation class map, I ran a random forest classification tree of elevation classes using the randomForest package (Liaw and Wiener, 2002) in R statistical computing software (R Development Core Team, 2007). CISD and the first three of four PCA axes contributed to the final random forest tree. I used the node information from the tree to run a spectral decision tree classification in ENVI image processing software (2005).

I tested for differences in vegetation cover-class values among elevation classes with Multi-response Permutation Procedures (MRPP) in PC-Ord Software (McCune and Mefford, 1999). I evaluated species frequency and cover data with respect to elevation graphically, and with Spearman rank correlations, and then

implemented a random forest tree to evaluate species vectors for their ability to predict elevation class.

What follows

The second chapter of this thesis summarizes the thesis project in manuscript format to facilitate submission of the chapter for publication. The third and final chapter contains a short discussion the challenges I encountered using the initial vegetation classification as an approach to mapping and some of the lessons learned.

Chapter 2: Remote Sensing of Tidal Freshwater Marsh Elevation, Channels, and Vegetation Structure

Introduction

Tidal freshwater marshes provide important ecosystem services for human society (Ramsar convention, 2007). As ecotones, these marshes buffer potential negative effects between terrestrial and aquatic ecosystems, protecting shorelines and reducing sediment loads in downstream waters (Orson et al., 1990; Sather and Smith, 1984). They trap sediment-sorbed pollutants (Bridgham et al., 2006; Johnston et al., 1984; Mitsch et al., 1995; Orson et al., 1990), and remove nitrogen to the atmosphere via microbial processing in the sediments (Hume et al., 2002; Hunter and Faulkner, 2001). Tidal freshwater marshes are sites of high primary productivity (Mitsch and Gosselink, 2000) and provide much needed habitats for both terrestrial and aquatic species, supporting many fisheries (Brundage and Meadows, 1982; Mitsch and Gosselink, 2000; Odum et al., 1984). Because anthropogenic pressures are threatening the health and persistence of many tidal freshwater marsh systems (NRC, 2001; Ramsar convention, 2007; Tiner, 1984), we must expand our efforts to conserve existing sites and restore lost areas if we are to continue to benefit from the goods and services they provide (NRC, 2001; Ramsar convention, 2007).

Maintenance and construction of healthy marsh ecosystems require detailed knowledge of the structure and functioning of these systems in space and time (NRC, 2001). However, gathering such information can be labor intensive and therefore hard to obtain. As a result, management and restoration decisions have often been

made without the knowledge needed to promote successful outcomes (NRC, 2001). Our study suggests a combined field sampling and remote sensing approach that can be used to facilitate the gathering of data at a whole-system scale to produce maps of the vegetation and physical structure of tidal freshwater marshes. Such maps can be used to assess changes in marsh vegetation over time and inform restoration designs and monitoring efforts.

The structure of tidal freshwater marshes can be roughly characterized as the interaction of wetland vegetation, marsh surface elevation, and tidal channel networks. Understanding the spatial distribution of these three elements is therefore critical when planning or monitoring a wetland reconstruction. Elevation and tidal networks are likely to be good spatial indicators of length and frequency of inundation, as tidal freshwater marshes tend to be relatively flat. Channels, at least to some degree, direct the flow of water in and out of the marsh, and lower elevations tend to flood before higher areas. Also, when channel levies are over-topped, adjacent locations may flood more quickly than remote locations. Thus, small elevation changes and channel placement are likely to determine the spatial arrangement of the hydroperiod gradient in tidal freshwater marshes.

The plants in tidal freshwater marshes, unlike their halophytic counterparts along the coasts, are often observed in highly heterogeneous mixtures of both perennial and annual species. The literature generally refers to “high” and “low” marsh areas, the low marsh being dominated by a few species that are tolerant of frequent lengthy inundations, such as *Nuphar luteum*, *Pontedaria cordata*, and *Zizania aquatica* (Mitsch and Gosselink, 2000), and the high marsh, which generally

contains a diversity of annual and perennial species that tolerate a broad range of less extreme flooding conditions (Leck and Simpson, 1994; Odum et al., 1984; Simpson et al., 1983).

Given that wetland plants are adapted to the saturated environment of wetland substrates to varying degrees, and given that nutrients, sediments, seeds, and vegetative propagules are delivered and removed with tidal flux, the hydroperiod gradient formed by elevation and channel networks is likely to drive the distribution of vegetation communities in tidal freshwater marshes. Distributional responses of emergent macrophytes to elevation gradients and channel networks are well documented in the literature (Baldwin and Pendleton, 2003; Parker and Leck, 1985; Peterson and Baldwin, 2004; Seabloom et al., 2001; van der Valk, 2005). In controlled field experiments, Baldwin et al. (2001) demonstrated a rise in species richness in response to 10 cm incremental increases in tidal freshwater marsh elevation. Parker and Leck (1985) noted density variations in seed banks, seedlings, and mature plants along transects perpendicular to stream channels. However, because many freshwater macrophyte species have broad, overlapping elevation ranges (Leck and Simpson, 1994; Odum et al., 1984; Simpson et al., 1983), identifying and mapping distinct communities is difficult.

This study explores whether remote sensing technology can be used to identify and map tidal freshwater marsh vegetation structure with respect to elevation. Our specific objectives were to a) explore whether elevation classes could be identified in high spatial resolution imagery and extrapolated to map the entire marsh; b) test the identified elevation classes for differences in vegetation cover of dominant

marsh species; c) examine species frequency distributions and canopy cover across the sampled elevational gradient to gain further insights into the vegetation structure of elevation classes; d) examine possible influences of channel size and location on vegetation distributions within elevation classes; and e) determine whether species cover could be used to predict mapped elevation classes. Specifically, we tested the hypothesis that elevation and channel size and distance influence the distribution and cover of tidal freshwater marsh species, as well as associations among species. This hypothesis was tested against the null hypothesis that species cover is randomly distributed with respect to elevation and channel networks. To address the objectives and hypothesis, we studied a tidal freshwater marsh in the Potomac estuary of the Chesapeake Bay watershed using a QuickBird multi-spectral image and field surveys of elevation, global position, vegetation cover, and tidal channel cross-sectional area.

Methods

Study site

Dyke Marsh Preserve is a freshwater tidal marsh on the Potomac River near Alexandria, VA, with an approximate centroid of 321951 E and 4292947 N in UTM zone 18N in the WGS84 datum. Extensive sand and gravel mining in the early part of the last century reduced this marsh to its current size of approximately 196 hectares before the marsh came under direct management by the National Park Service in 1976. The Park Service is currently considering whether Dyke Marsh Preserve can be successfully restored to a larger size. An apparently thriving historic marsh at the proposed expansion site offers the opportunity to examine, in situ, the landscape

structure (elevations, tidal channels) and vegetation distributions to be targeted in a successful restoration.

Image acquisition

Because the U.S. Secret Service has issued a “no fly rule” for small aircraft in the airspace over the study site, collection of aerial imagery with both high spectral and spatial resolution was not possible at this location. We therefore contracted with DigitalGlobe to task the QuickBird satellite for high spatial resolution imagery. The satellite collected a multi-spectral image consisting of blue, green, red, and near infrared bands with 2.4 m pixel resolution on 30 September, 2005.

Image registration and preliminary processing

We used channel centers located in our field collected elevation data as ground-truth points for a 1st degree polynomial warp of the QuickBird image in ENVI image processing software (Research Systems Inc., 2005), re-registering the image to provide higher spatial agreement between the image and surveyed locations. To test the accuracy of the final registration, we measured the distance between 36 identifiable locations throughout the image and their ground-truth GPS locations using ArcMap GIS software (ESRI, 2001-2005). The average deviation in the repositioned image for the 36 measurements was 0.8 m (st. dev. = 1.1 m), and the maximum deviation was 3.6 m.

In addition to the original QuickBird multi-spectral data, we developed 5 additional bands in ENVI for use in later processing: an NDVI band, to capture red-edge vegetation variation, a similar index that referenced green and red data ((green-

red)/(green+red)), and three Principle Components Analysis bands that accounted for virtually all of the variation in the QuickBird bands and derived indices.

Additionally, we created a mask that screened out upland areas, forested wetland, anthropogenic features, such as roads and boardwalks, and open-water features, to enable subsequent spectral analyses to focus exclusively on emergent marsh areas.

Channel mapping

To evaluate channel influences on species cover, and to account for them in decision tree mapping of elevation classes, we digitized the marsh's channel network in ArcMap (ESRI, 2001-2005) referencing both the 2005 QuickBird image and a National Park Service leaf-off 2004 aerial photo that was re-registered to the repositioned QuickBird image in ENVI (Research Systems Inc., 2005).

We then categorized all of the digitized channels according to their apparent size in the images and performed a stratified ground survey of 16 randomly selected channels. During this survey, we measured cross-sectional areas using a transect tape, stretched across the channel, and a meter stick for taking vertical measurements at regular intervals along the tape. These vertical and horizontal measurements were compiled in annotated diagrams of adjacent polygons, and the polygonal areas were calculated and summed to determine the cross-sectional area at each sample location (Sanderson et al., 2001).

Sampled cross-sectional areas revealed which channel size categories could and could not be reliably discriminated from visual assessment of the imagery. Final channel size classes were allocated accordingly, and we returned to the field for a

final stratified random survey of 27 new channel cross-sections. A final confusion matrix reports the accuracy of the channel map predictions in light of the final survey.

Cumulative inverse squared distances (CISD) values

Channel influences were quantified using the transformed cumulative inverse squared distance function (CISD) developed by Sanderson, Foin, and Ustin (2001), which models exponential decay of channel influences with increased distance, weighted by channel size. The model employs a GIS algorithm that identifies all channels within 25 m of each pixel's center, divides the channels into a series of 0.5 m segments, and calculates the distance from the center pixel to each of the nearest channel segments within the 25 m radius. The CISD value for the pixel is then calculated with the following equation:

$$\text{CISD} = \sum w_{\text{order}} * (1/d)^2$$

where d is the distance from the pixel center to each of the channel segments, and w_{order} is a weighting factor based on channel order. Weights in the Sanderson et al. study generally increased one order of magnitude for each change in channel order and were based on estimated channel discharge and cross-sectional area. Channel order does not exist, per se, at Dyke Marsh Preserve, owing to cross-connections with multiple water sources. We therefore used average cross-sectional areas for each channel class to determine channel weights. The CISD algorithm was developed in Avenue programming software and run in ArcView 3.2 (ESRI, 1992-1999).

Vegetation and elevation field surveys

We ran an unsupervised IsoData classification of the QuickBird image in ENVI software (Research Systems Inc., 2005) and developed a stratified sampling design for each of 10 image classes. A minimum of ten 1 m x 1 m quadrats were sampled for each image class at locations spread throughout the marsh for a total of 102 quadrats. Our goal was to maximize the chance of collecting vegetation samples from locations that were spatially and spectrally distinct. Owing to lack of spatial agreement with the initial registration of the image (sampling was performed prior to re-positioning the image), we placed quadrats well inside the boundaries of spectral classes at each sampling location to minimize the chance of accidentally sampling adjacent classes. We recorded species presence and cover in early October 2005. Understanding that the error in estimating cover in the field is high, we estimated actual cover in the field and transformed the cover values into cover classes. Because the use of traditional Braun-Blanquet cover classes in numeric analyses has lately been called into question (Podani, 2006), we characterized vegetation cover using ordinal transform values (OTV) as proposed by van der Maarel (2007). Visual percent cover assessments were converted to their OTV class values using the formula $OTV = a \ln C + 2$, where C is the estimated cover percentage, and a is a factor that weights percent cover values to better approximate species biomass (table 1). Vouchers for species unidentified in the field were taken back to the lab for later identification with plant keys (Brown and Brown, 1984; Crow and Hellquist, 2000a, b; Godfrey and Wooten, 1979; Tiner, 1993).

BrBl	OTV	OTV ~%
r	1	0.6
+	2	1.2
1	3	2.5
	4	5
2	5	10
	6	20
3	7	40
4	8	80
5	9	160

Table 1: Comparison of Braun-Blanquet cover-abundance scale, Ordinal Transform Value (OTV), and OTV approximate percent cover, as per van der Maarel, 2007

In addition to vegetation species cover, five hundred eighteen elevation and global position readings were also recorded throughout the marsh using high precision (± 2 cm) surveying equipment (Topcon Hiper Plus / GB-500, GD, 1Hz, RTK). Elevation and position readings were taken at each vegetation plot, along hiking routes, and at channel centers and levies encountered during the survey.

Elevation class selection

To maximize spectral differences among elevation classes and promote even distribution of samples among classes, we performed flexible beta hierarchical clustering on surveyed elevations and spectral data in PC-Ord software (McCune and Mefford, 1999) using Sørensen distances and a beta value of -0.25. This algorithm provides results similar to a Ward's classification, but allows for use of Sorensen Distance, which is incompatible with Ward's clustering but better suited to community data (McCune and Grace, 2002). We evaluated graphs of the flexible

beta class plot frequencies against elevation to select the elevation boundaries most likely to represent apparent frequency distributions in the spectral data. Plots were then assigned to discrete elevation classes, divided by these estimated boundaries, and tested for spectral confusion with conditional inference recursive partitioning trees implemented with the “party” add-on package to the R system for statistical computing (Hothorn et al., 2007). Classes that were spectrally confused were aggregated to create three final elevation classes that could be reliably separated with the spectral data.

Mapping

We used the randomForest package (Liaw and Wiener, 2002) in R statistical computing software (R Development Core Team, 2007), to classify CIRD values and 3 bands of a Principle Components Analysis reduction of the original QuickBird spectral bands and derived index values. We chose the random forest algorithm for several reasons: it performs well with community data compared to other decision tree classifiers (Prasad et al., 2006); it eliminates the problem of overfitting associated with many decision tree algorithms (Prasad et al., 2006); it assesses how well each splitting variable (in this case, CIRD or spectral reflectance band) reduces final distance between classes in multivariate space (Liaw and Wiener, 2002); and finally, the error represented in the final tree is likely to be close to the true predictive error of the population, rather than representing error associated with any particular tree’s subset or “sample” of the data (Prasad et al., 2006).

We included CIRD values in the random forest classification to improve its ability to predict elevation, irrespective of channel size and proximity. The decision

rules developed in the random forest algorithm were then entered into a decision tree classifier in ENVI software (Research Systems Inc., 2005) to produce a final map of elevation classes. One third of elevation observations were removed from the collected elevations prior to random forest analysis for use as an independent dataset in final map validation. They were selected by random assignment, stratified by elevation class. Final map producer and user accuracies from the random forest classification were compared with these independent validation accuracies for consistency.

Analyzing vegetation data

We evaluated the vegetation data in several ways. First, we estimated species richness for each elevation class with species accumulation curves in PC-Ord software (McCune and Mefford, 1999). We then removed species occurring in fewer than 5% of study plots to reduce the number of zeros in the dataset before proceeding with subsequent analyses.

We evaluated species for differences in OTV among elevation classes with non-parametric Multi-Response Permutation Procedures (MRPP) testing (Mielke and Berry, 2001). The MRPP test is a good alternative to Analysis of Variance for analyzing cover data, which are unlikely to be normally distributed with uniform variances (McCune and Grace, 2002). In general, MRPP testing involves calculating a mean within-group distance, or δ , weighted by the number of items in the group, in this case, the class N-value. Then, δ values are calculated for all possible partitions of the data into groups of the same size. It is then assumed that all of these partitions could have occurred with equal chance. The proportion of partitions having a δ value

less than that actually observed is then calculated. This is the p-value for the test, measuring the likelihood of the observed δ value being due to chance. In addition to the p-value, group homogeneity, or A, is also reported. The value of A will equal $1 - \delta/m_\delta$, where m_δ is the δ value expected by random chance. If all items within groups are identical, $\delta = 0$ and A will equal 1. If heterogeneity within groups equals that expected by chance, then A will equal 0. We ran pair-wise comparisons of species OTV values for all pairings of the three elevation classes using PC-Ord software (McCune and Mefford, 1999) and a Bonferonni corrected experimental error of 0.05.

To examine the distribution of individual species along the elevation gradient, we ran a Non-Metric Multidimensional Scaling ordination. To more closely detail species responses within elevation classes, we calculated the proportion of plots occupied by each species within 5-cm elevation increments. For example, if eight plots were located within the 10-15 cm elevation range and four of the plots supported species x, then the proportion of species x within that elevation range was 0.5. For the same elevation ranges, we also calculated each species' mean OTV cover. Species proportions and mean covers were then plotted against elevation to highlight optimal elevations for each species and assess overlaps in species ranges.

Spearman rank correlation was used to describe patterns of species associations, and to examine associations between species and environmental gradients (i.e., elevation and CISD).

Finally, to assess the effectiveness of using species OTV values to predict elevation class, we created a random forest classification tree of species OTV values

using the randomForest package (Liaw and Wiener, 2002) in R statistical computing software (R Development Core Team, 2007).

Results

Preliminary channel mapping and CISD values

Areal cross-sections (table 2) for the final channel classes (figure 1) show interior tidal channel sizes ranging across approximately 2.5 orders of magnitude. The confusion matrix for the final accuracy assessment of the channel map (table 3) shows some confusion between class 2, the most prevalent channel class (6.675 km cumulative length), and class 1 (1.177 km cumulative length). Therefore, channels predicted to be in class 1 were actually class 2 channels in approximately 29% of cases (table 3). Relatively few class 1 channels exist; however those channels that are incorrectly assigned would be likely to have cross-sectional areas that average 10-fold higher than reported, and their CISD values would likewise be undervalued.

Channel Class	Sample N	Mean csa (m ²)	W _(order)
1	7	0.05	0.1
2	21	0.40	1
3	9	2.02	5
4	3	7.47	15
5	5	17.60	40

Table 2: Mean cross-sectional areas (csa) and samples size (N) for 5 channel classes. W_(order) is the weight assigned for Cumulative Inverse Squared Distance (CISD) index calculations.

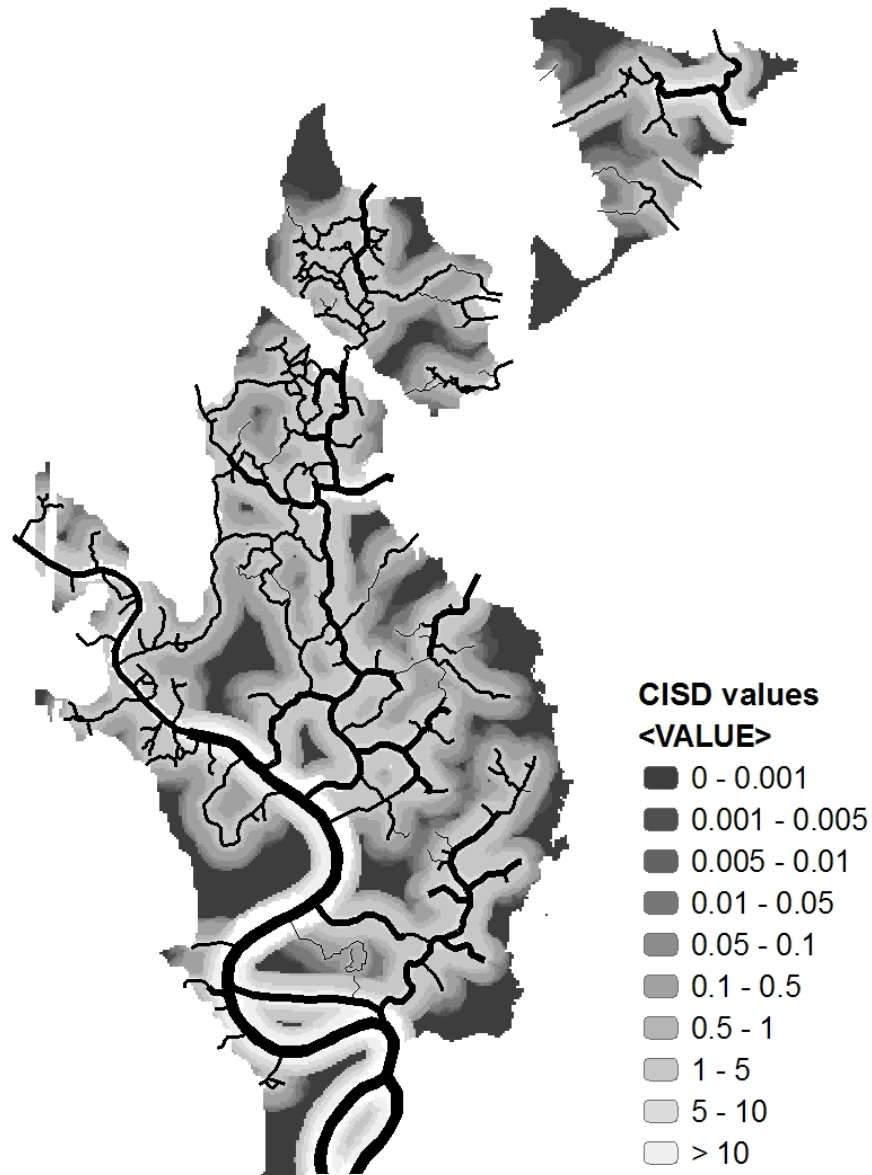


Figure 1: Five channel classes and CISD values for Dyke Marsh Preserve. Increases in mean cross-sectional area for the channel classes are depicted by increasing width of channel.

Channel classes 3 and 4, having respective cumulative lengths of 2.732 km and 0.353 km, were correctly identified in the ground-truth survey (table 3). Channel class 5 encompassed the lower portion (1.422 km) of Hog Island Gut and was confirmed to

have over 10 m² cross-sectional areas based on GPS cross-sectional elevation measurements taken during the elevation survey.

Predicted	Actual				User Accuracy
	Class 1	Class 2	Class 3	Class 4	
Class 1	5	2	0	0	71%
Class 2	0	13	0	0	100%
Class 3	0	0	5	0	100%
Class 4	0	0	0	2	100%
Producer Accuracy	100%	87%	100%	100%	

Table 3: Confusion matrix for channel class predictions, based on visual examination of Fall 2005 and early Spring 2004 imagery, versus field assessments for 27 channels. User accuracies predict how frequently mapped classes accurately predicted channel class in the field, and producer accuracies predict how frequently a field observation was accurately assigned to the predicted class.

Elevation class selection

Sampled elevations for the emergent marsh ranged from -0.52 m – 0.58 m with respect to sea level. The frequency distribution for 7 flexible beta classes of elevation and spectral signatures suggested four preliminary elevation class boundaries (-0.15, 0.20, 0.30, and 0.40; figure 2). However, spectral confusion among elevation classes delineated by these boundaries (table 4) prompted us to merge strongly confused classes to produce three final classes: class 1) elevations below 0.2 m; class 2) elevations between 0.2 m and 0.4 m; and class 3) elevations above 0.4 m.

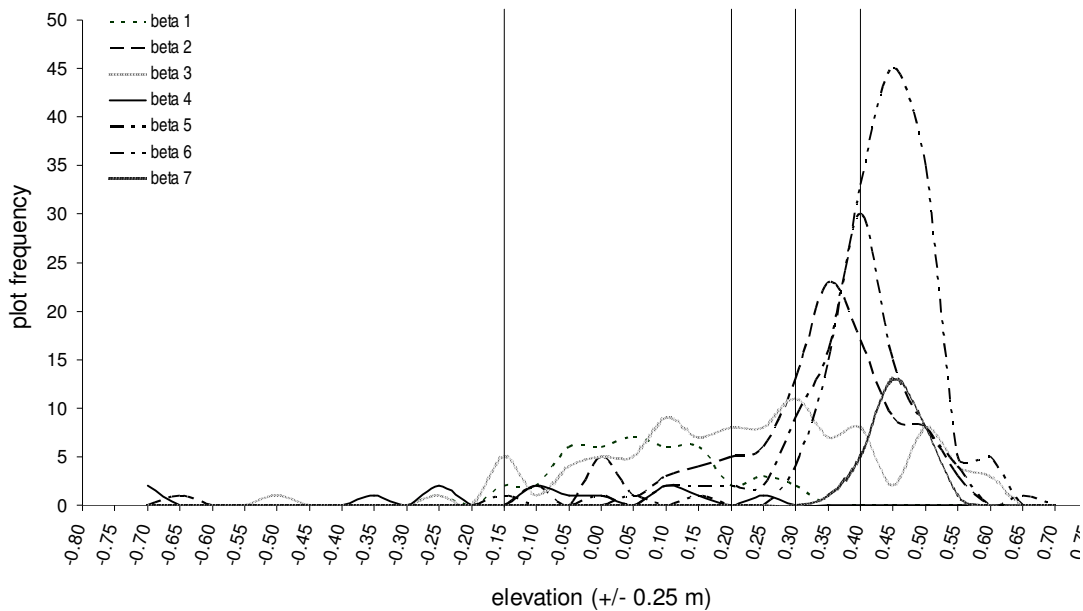


Figure 2: Frequency distribution for a 7-class flexible beta classification of elevation and spectral signatures from 518 plots in Dyke Marsh Preserve

Predicted	Actual					User Accuracy
	< -0.15 m	-0.15 m – 0.2 m	0.2 m – 0.3 m	0.3 m – 0.4 m	> 0.4 m	
< -0.15 m	4	2	1	4	2	31%
-0.15 m – 0.2 m	26	32	16	5	0	41%
0.2 m – 0.3 m	0	0	0	0	0	na
0.3 m – 0.4 m	1	12	17	40	23	43%
> 0.4 m	0	7	5	54	93	59%
Producer Accuracy	13%	60%	0%	39%	79%	

Table 4: Confusion matrix for conditional inference classification tree of spectral signatures for 5 preliminary elevation classes. Spectral confusion prompted the combination of classes circled.

Map Creation

Random forest classification indicated that the splitting variables best able to divide the elevation data into the three elevation classes were CISD value, and the first three axes of a Principle Components Analysis reduction of the original QuickBird data and derived indices. A random forest of 5000 trees using CISD and PCA variables produced the final decision rules used to create the elevation map (figure 3) with an out-of-box error rate of 31.1%. The data predicted elevation classes 1 and 3 better than class 2 with the most confusion arising between classes 2 and 3 (table 5). Independent validation using data withheld from the random forest classification yielded an overall error rate of 36.8%. This adds credibility to the accuracy of the random forest predictive error rates, as it is similar to the error rates of some of the individual boot-strapped trees within the forest.

Predicted	Actual			User Accuracy
	Class 1	Class 2	Class 3	
Class 1	69	20	5	73%
Class 2	24	120	37	66%
Class 3	2	41	100	70%
Producer Accuracy	73%	66%	70%	

Table 5: Confusion matrix for a random forest classification of CISD and the first three bands of a principle components analysis reduction of QuickBird multi-spectral data and derived indices.

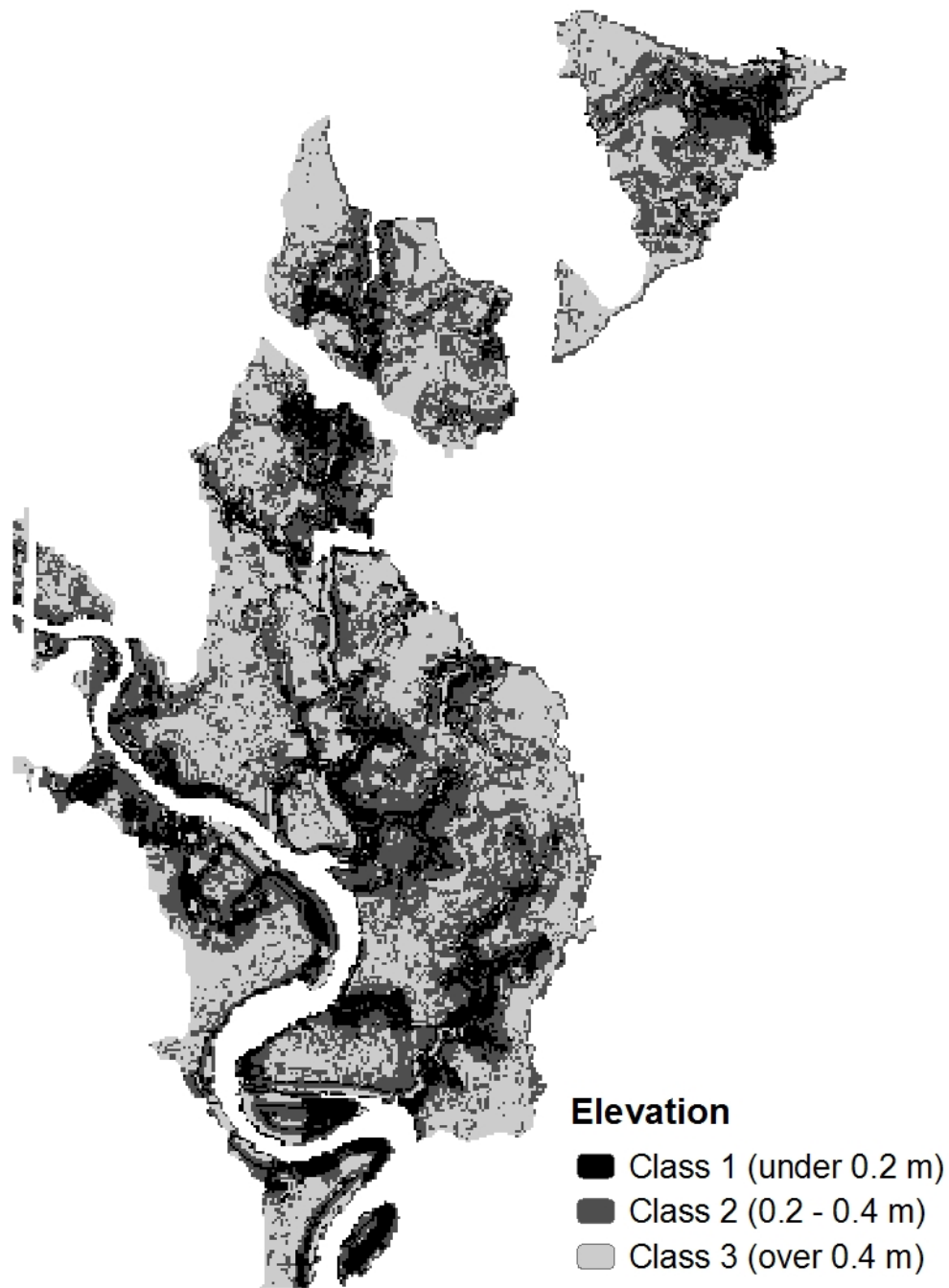


Figure 3: Decision tree classification of elevations at Dyke Marsh Preserve based on a random forest classification of CISD and the first three bands of a principle components analysis reduction of QuickBird multi-spectral reflectance bands and derived indices.

Vegetation among elevation classes

The three elevation classes were fairly evenly represented in the vegetation data, with 35 observations in class 1, 40 observations in class 2, and 43 observations in class 3. Randomized species accumulation curves (figure 4) suggest that species richness in the middle and highest elevation classes were similar but that fewer species were present in the lowest elevation class.

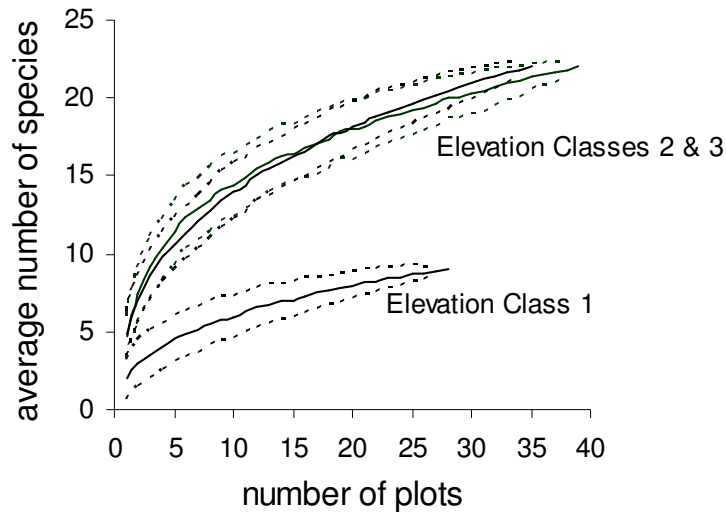


Figure 4: Randomized species accumulation curves and 95% confidence intervals for each of three elevation classes. Curves were developed with PC-Ord software (McCune and Mefford, 1999).

Pairwise MRPP tests confirmed differences in community structure among all pairings of the elevation classes (table 6). A histogram of species frequency within each elevation class (figure 5) reveals that *Impatiens capensis*, *Nuphar luteum*, *Peltandra virginica*, and *Typha* spp. were most frequently sampled, with *N. luteum* concentrated in the lowest and middle elevation classes, *I. capensis* and *Typha* concentrated in the middle and upper classes, and *P. virginica* well represented in all classes.

Elevation Classes	Species OTV	
	p-Value	A
All Classes	0.0000	0.2428
1 vs. 2	0.0000	0.2405
1 vs. 3	0.0000	0.3324
2 vs. 3	0.0056	0.0197

Table 6: MRPP test results for differences in OTV species abundance among elevation classes

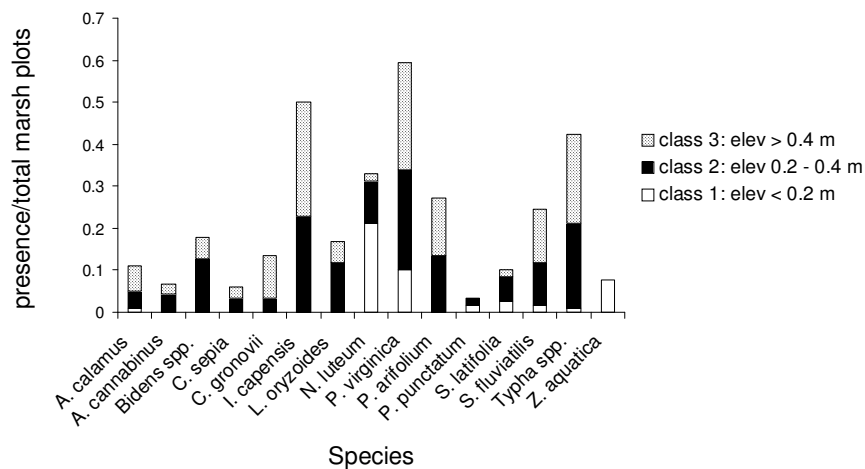


Figure 5: Species frequencies within elevation classes

Species ranges

Differences between high and low marsh vegetation distributions are evident in the NMS ordination (figure 6). Middle and high marsh could not be distinguished through the ordination. Attempts to run ordinations within elevation classes did not converge, presumably owing to the minimal within-class elevation changes.

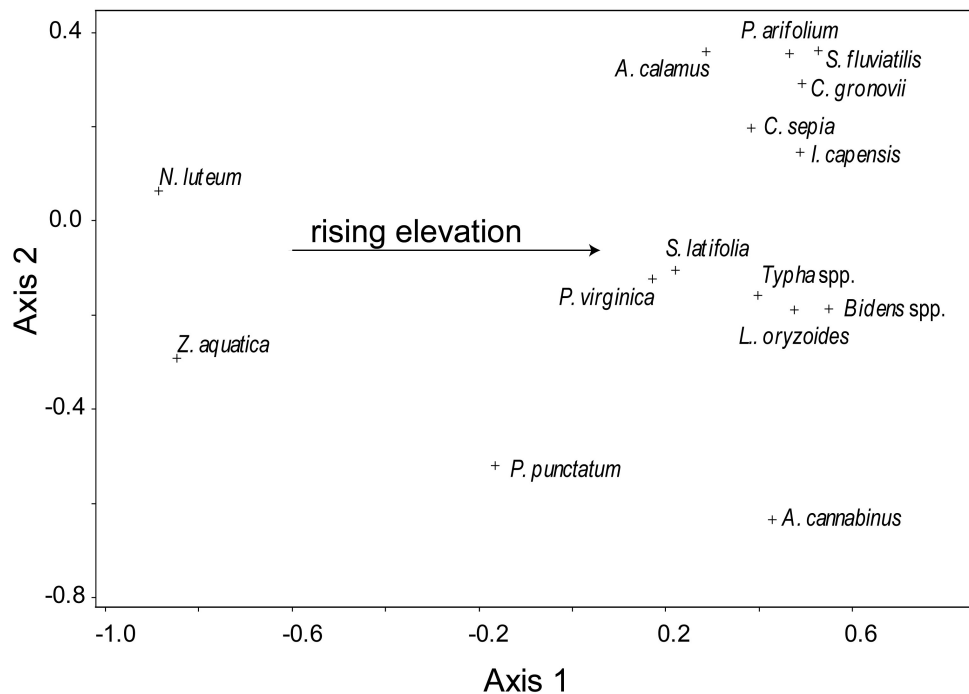


Figure 6: NMS ordination of species OTV across all sampled elevations. Final stress for the 3-dimensional solution was 14.85, explaining 88% of variation. Axis 1 explains 57% and axis 2 explains 17% of variation.

To explore finer scale vegetation trends with respect to elevation, we calculated proportional frequencies (# plots in which a species is observed / total number of plots within an elevation range) for 5 cm elevation increments across the elevation gradient. Much like the ordination (figure 6), the plot of proportional frequency against elevation (figure 7) shows that in low marsh areas *Nuphar luteum* was the dominant species at elevations below sea level. *Zizania aquatica* and *P.*

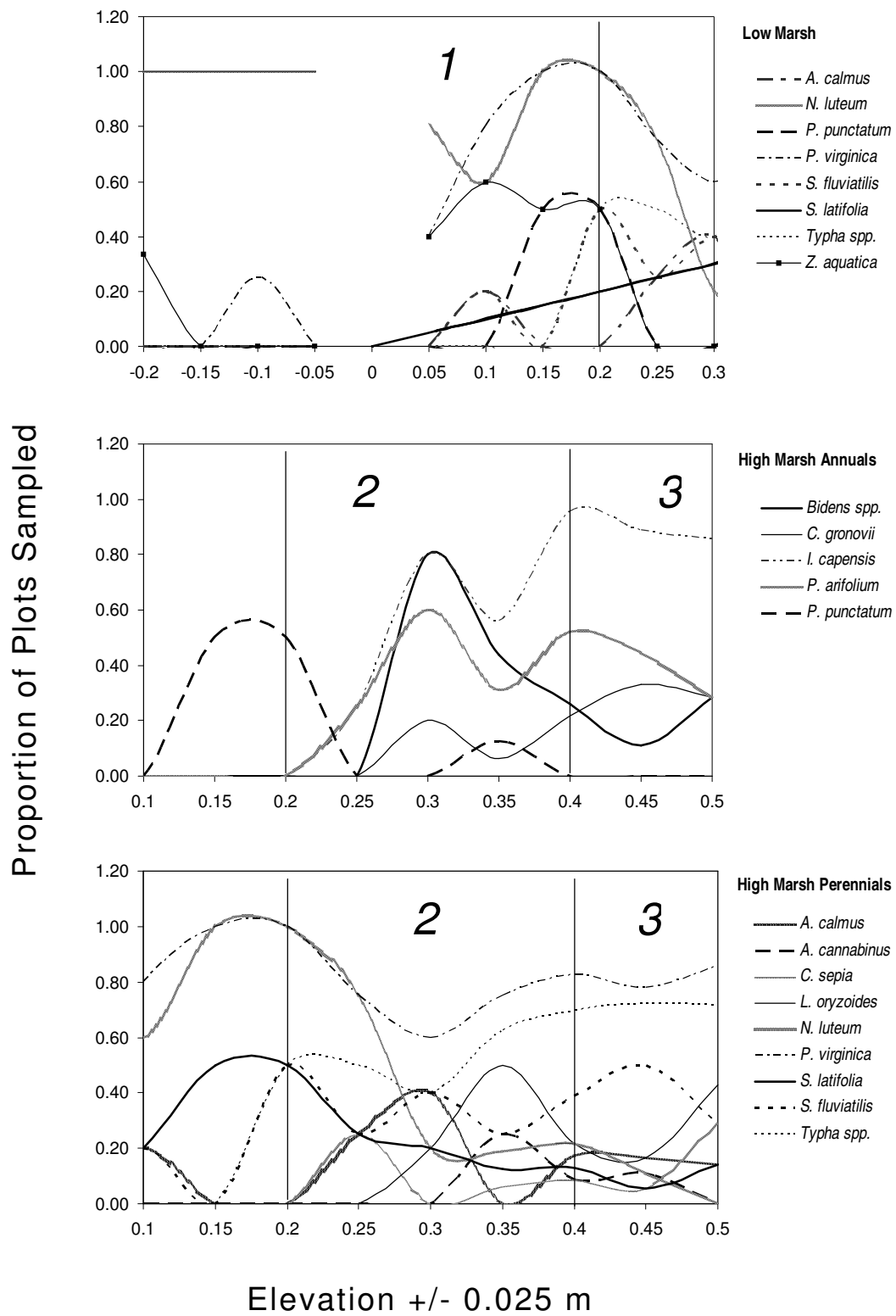


Figure 7: Species frequencies with respect to elevation; large italicized numerals designate elevation classes bounded by vertical lines. No plots were sampled between elevations -0.05 m and 0 m.

virginica also featured prominently in elevation class 1, being sampled in over 40% of plots between 0 - 0.2 m. Species that were infrequently sampled in class 1 included *Sagittaria latifolia*, *Acorus calamus*, *Polygonum punctatum* and *Schoenoplectus fluviatilis*.

The plot of high marsh annuals (figure 7) shows that *Bidens* spp. and *I. capensis* were present in similar frequencies near the center of elevation class 2, but that *Bidens* spp. proportional frequency tailed off at highest elevations in class 2 while *I. capensis* frequency continued to rise. *Cuscuta gronovii* proportional frequency increased across elevation classes 2 and 3, while *Polygonum arifolium*, with moderately high frequencies (20% – 60%), remained steady or declined slightly. *Polygonum punctatum*, while present in class 2, had highest proportional frequencies at lower elevations.

Perennial proportional frequencies (figure 7) suggest that *N. luteum* presence declined sharply in class 2 between 0.2 to 0.3 m elevations and then continued a more gradual decline into elevation class 3. *Sagittaria latifolia* showed steady declines across both classes 2 and 3. *Typha* spp. proportional frequencies rose across class 2 to closely follow frequencies of the other perennial dominant, *P. virginica*, at elevations over 0.3 m. *Acorus calamus* was most frequently sampled in class 2, at 0.3 m elevations. *Schoenoplectus fluviatilis* was fairly prominent (20%-50%), and had similar frequencies across classes 2 and 3. *Leersia oryzoides* showed proportional frequency peaks in both classes 2 and 3, but was present only above 0.3 m elevation. *Amaranthus cannabinus*, also present only above 0.3 m, showed higher presence in

class 2 than in class 3. *Calystegia sepium* was present in highest proportions at lower elevations in class 2 and higher elevations in class 3.

Cover patterns across species ranges

As is often the case with community data, vegetation cover trends for several species could not be easily inferred due to low numbers of observations. These species included *A. calamus*, *A. cannabinus*, *C. sepium*, *P. punctatum*, and *S. latifolia*. Anecdotally, *Acorus calamus* and *A. cannabinus* appeared to be less frequently observed than in a preliminary study the previous year, whereas *Bidens* spp. were more prevalent. *Polygonum punctatum* was observed to grow on levies but was found infrequently in the data, most likely because our sampling design targeted assemblages that could be captured in imagery with 2.4 m pixel spatial resolution. Because the marsh levies were often very narrow, vegetation specializing in this habitat was typically overlooked.

Additionally, several frequently sampled species showed cover patterns (figure 8 a-d) unlikely to represent peak-season cover, due to the onset of senescence. *Zizania aquatica*, generally reaching peak cover in mid to late August, was typically leafless at the time of sampling. *Cuscuta gronovii* and *P. arifolium* were green in some locations and withered in others. *Peltandra virginica*, a species with broad leaves and generally high cover in peak season, frequently consisted solely of seed pods at sampling.

Leersia oryzoides and *Bidens* spp., the latter being almost uniformly *Bidens laevis*, showed relatively stable cover values across their observed ranges (figure 8 e-f). *Impatiens capensis* and *S. fluviatilis* (figure 8 g-h) showed generally rising cover

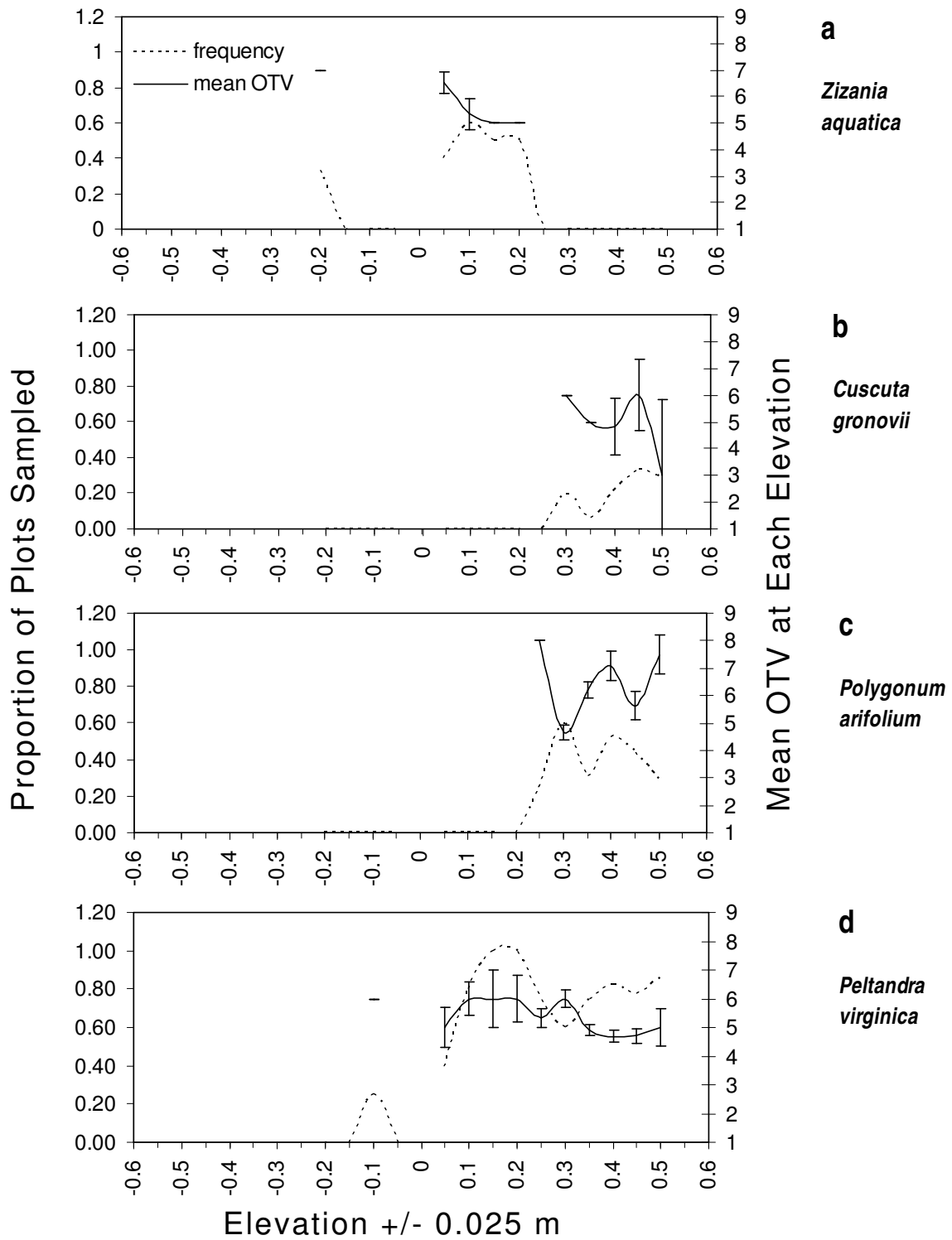


Figure 8: Mean OTV abundance across species range on the elevation gradient at Dyke Marsh Preserve. Proportional frequencies are plotted for reference. Error bars are standard error of the mean.

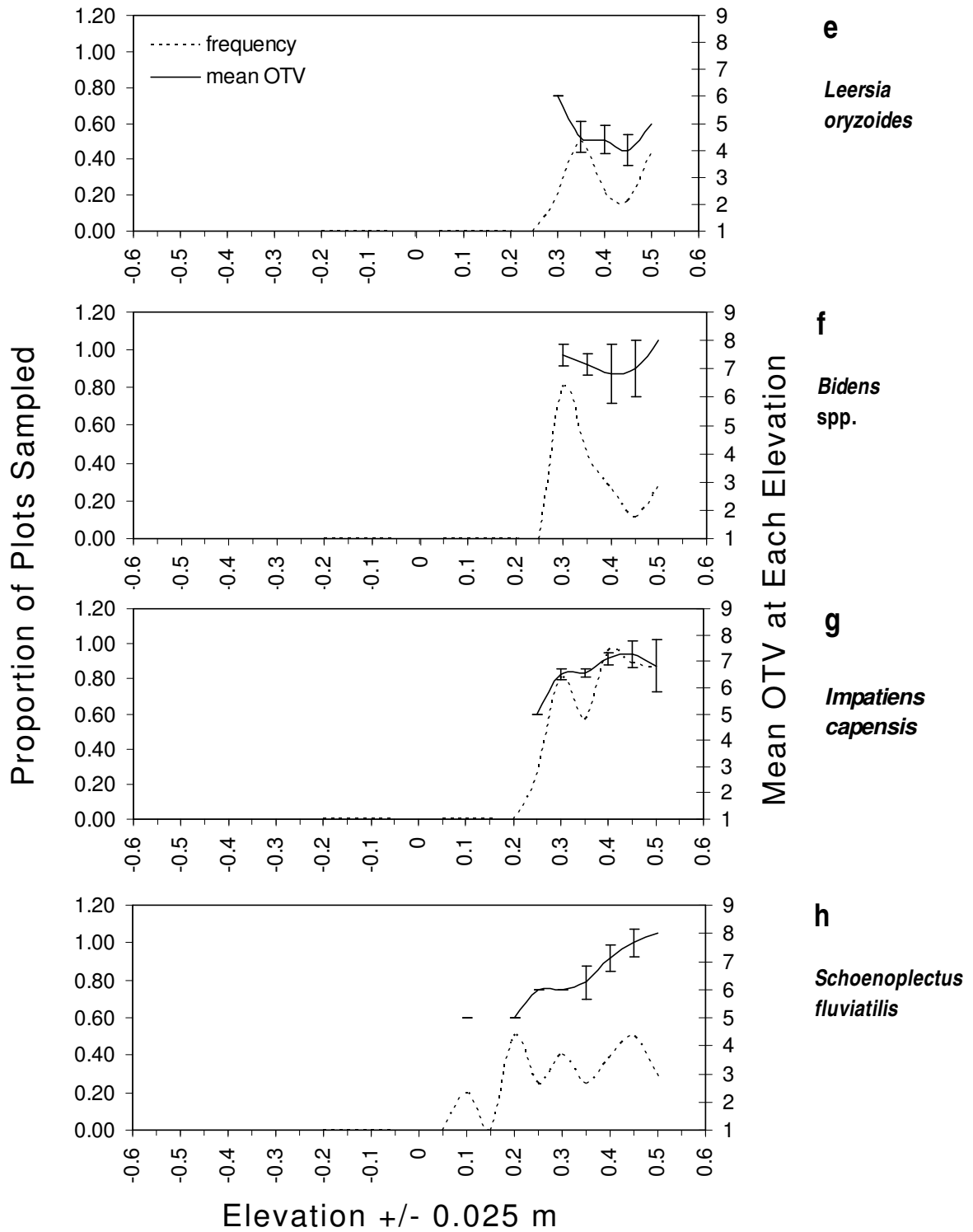


Figure 8 (cont.): Mean OTV abundance across species range on the elevation gradient at Dyke Marsh Preserve. Proportional frequencies are plotted for reference. Error bars are standard error of the mean.

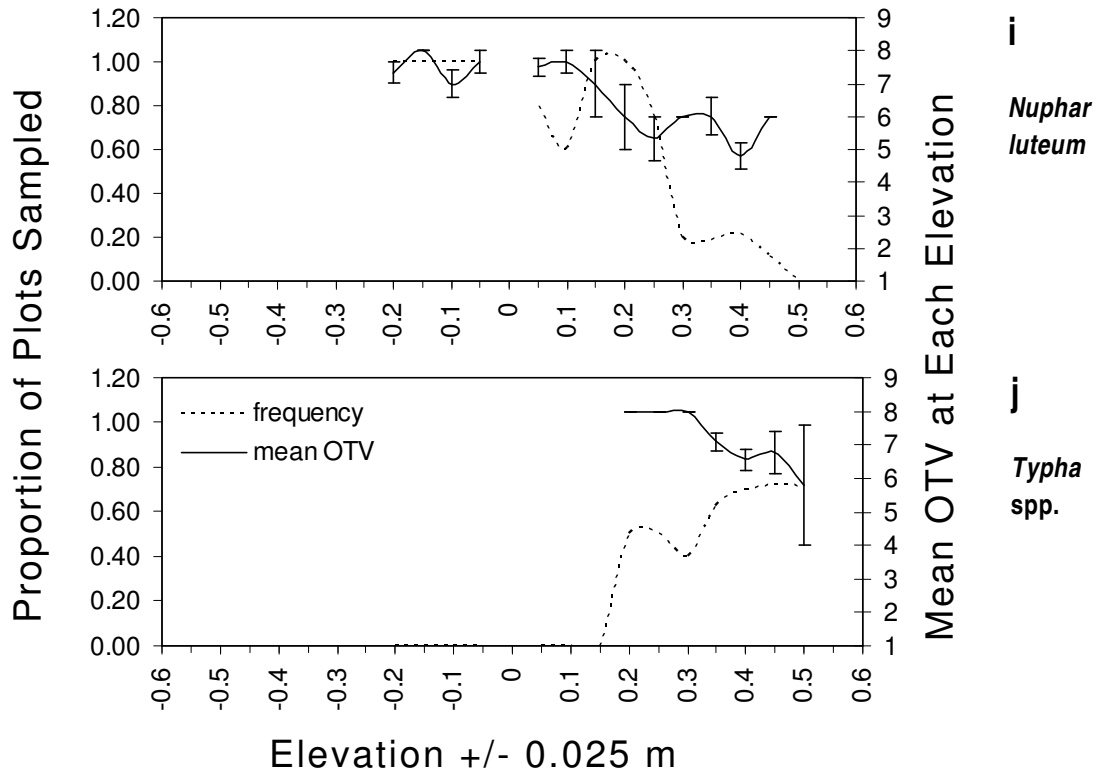


Figure 8 (cont.): Mean OTV abundance across species range on the elevation gradient at Dyke Marsh Preserve. Proportional frequencies are plotted for reference. Error bars are standard error of the mean.

across their ranges, with significant cover differences in MRPP testing between elevation classes 2 and 3 ($p = 0.007$, $A = 0.066$; and $p = 0.009$, $A = 0.149$ respectively). *Nuphar luteum* cover (figure 8 g) declined between elevation classes 1 and 2 ($p = 0.000$, $A = 0.333$), and *Typha* spp. (figure 8 h) declined between elevation classes 2 and 3 ($p = 0.017$, $A = 0.062$).

Vegetation associations

Spearman rank correlations across the entire elevation range (figure 9) revealed a high correlation between elevation and *I. capensis* OTV ($r = 0.69$, $p < 0.001$) and a high negative correlation between elevation and *N. luteum* ($r = -0.70$, $p <$

0.001). Positive correlations were also found between elevation and the following species: *Typha* spp. ($r = 0.44$, $p < 0.001$), *C. gronovii* ($r = 0.38$, $p < 0.001$), *P. arifolium* ($r = 0.35$, $p < 0.001$), and *S. fluviatilis* ($r = 0.33$, $p < 0.001$). CISD and elevation were negatively correlated ($r = -0.63$, $p < 0.001$) and, although associations between CISD and vegetation species were also found (figure 9), *Polygonum arifolium* was the only species that was more strongly correlated with CISD ($r = -0.47$, $p < 0.001$) than it was with elevation.

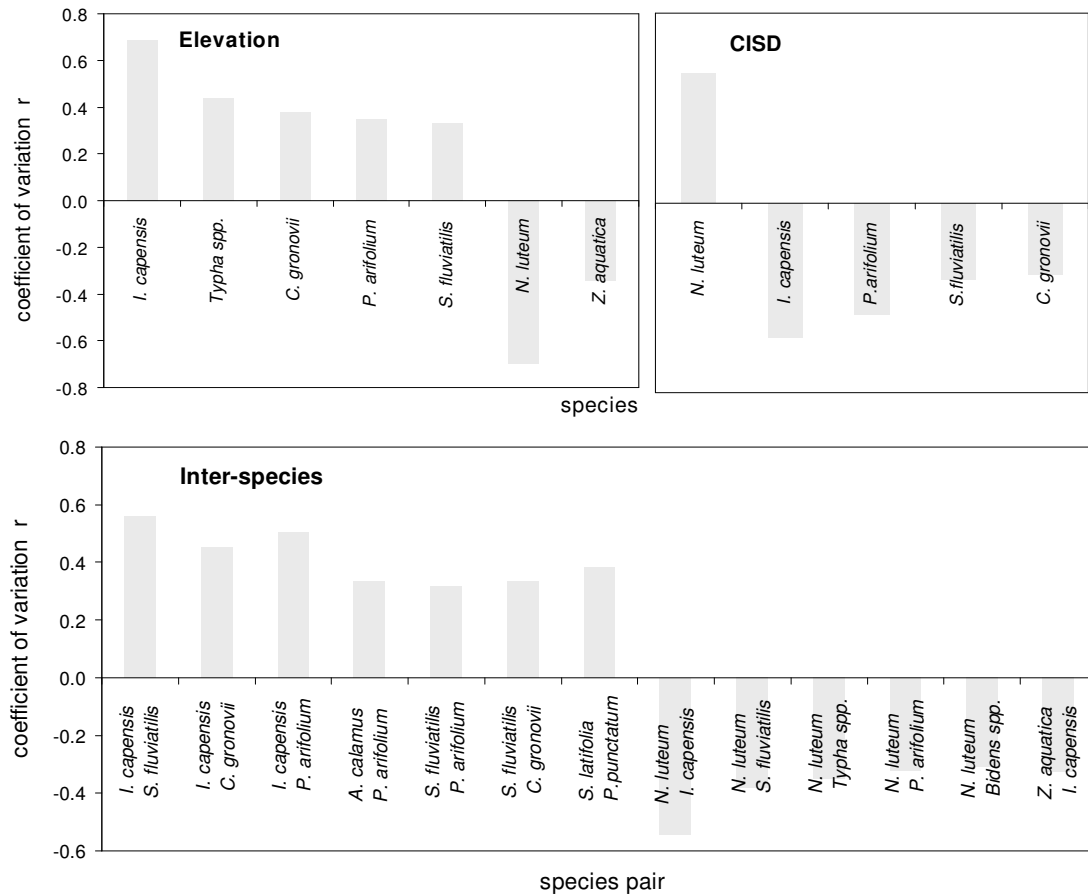


Figure 9: Spearman's rank correlations ($|r| > 3$, $p < 0.002$) for species-environment and species-species associations for all surveyed elevations at Dyke Marsh Preserve

When correlations are examined within individual elevation classes, additional patterns are revealed. Direct comparisons between the lowest elevation class, class 1, and the upper elevation classes are prohibited owing to lower species counts in elevation class 1. However, a summary of within-class correlations (figure

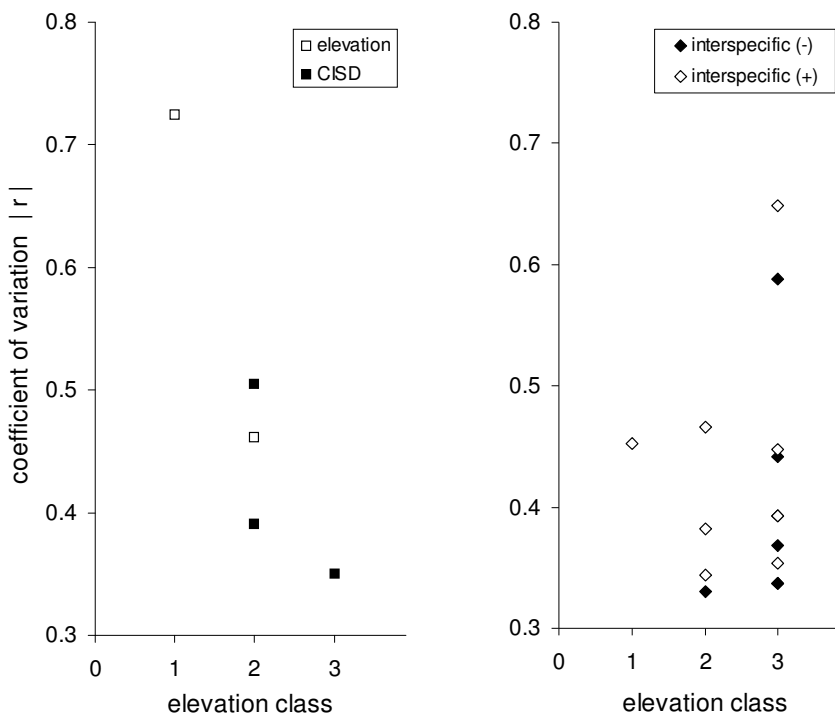


Figure 10: Absolute value of Spearman rank coefficients ($|r| > 3$, $p < 0.05$) for species-environment and species-species associations within elevation classes at Dyke Marsh Preserve. Class 1 data were sampled at elevations under 0.2 m, Class 2 data were from 0.2 to 0.4 m, and class 3 data were from elevations over 0.4 m.

10) reveals that the number and strength of associations between vegetation and environmental gradients decrease, while the number and strength of interspecific associations increase (particularly negative associations) at higher elevations.

Peltandra virginica was correlated with elevation within the lowest elevation class ($r = 0.72$, $p < 0.001$). *Zizania aquatica* was correlated with *P. virginica* ($r = 0.45$, $p = 0.02$). Other inter-specific and environmental correlations were found among species at this elevation, but are not reported due to their extremely low frequencies (i.e., under 4 observations).

At intermediate elevations (i.e., elevation class 2), *P. arifolium* showed the strongest association with the environment, being negatively correlated with CISD value ($r = -0.50$, $p = 0.001$), and therefore exhibiting decreased cover with increasing channel size and proximity. *Amaranthus cannabinus* was positively correlated with CISD value ($r = 0.39$, $p = 0.014$), and *I. capensis* was positively correlated with elevation ($r = 0.46$, $p = 0.003$). Interspecific correlations included *P. arifolium* and *A. calamus* ($r = 0.47$, $p = 0.003$), *L. oryzoides* and *A. cannabinus* ($r = 0.38$, $p = 0.16$), and *I. capensis* and *S. fluviatilis* ($r = 0.34$, $p = 0.032$). The only negative interspecific relationship in class 2 was between *S. fluviatilis* and *Typha* spp. ($r = -0.33$, $p = 0.040$).

Elevation class 3, representing highest elevations, showed no elevation correlations for species present in more than 3 plots. *Cuscuta gronovii* showed a negative correlation with CISD ($r = -0.35$, $p = 0.039$). *Impatiens capensis* was strongly correlated with *S. fluviatilis* ($r = 0.65$, $p < 0.001$), and also showed positive associations with *P. arifolium* ($r = 0.45$, $p = 0.007$) and *C. gronovii* ($r = 0.35$, $p = 0.037$). *Cuscuta gronovii* was also positively correlated with *S. fluviatilis* ($r = 0.39$, $p = 0.020$). The many negative interspecific associations found in class 3 included *S. fluviatilis* with *Typha* spp. ($r = -0.59$, $p < 0.001$); *P. virginica* with *C. gronovii* ($r = -0.44$, $p = 0.008$), with *Bidens* spp. ($r = 0.39$, $p = 0.020$), and with *S. fluviatilis* ($r = -0.34$, p

= 0.048); and *I. capensis* with *Typha* spp. ($r = -0.37$, $p = 0.030$) and with *Bidens* spp. ($r = -0.34$, $p = 0.048$).

Random forest classification by species

The random forest classification tree performed with all species OTV data selected only two species, *N. luteum* and *I. capensis*, as the splitting variables able to most accurately predict elevation class. The overall out-of-box estimate of error for the final tree was 26.5%, and elevation class 1 was most accurately predicted (table 7).

Predicted	Actual			User Accuracy
	Class 1	Class 2	Class 3	
Class 1	22	6	0	79%
Class 2	3	28	8	72%
Class 3	1	9	25	71%
Producer Accuracy	85%	65%	76%	

Table 7: Confusion matrix for random forest classification tree based on 500 trees using 83 observations of OTV abundance for *I. capensis* and *N. luteum*

Discussion

The map of elevation classes (figure 3) suggests that the microtopography of Dyke Marsh Preserve is heterogeneous at the 2.4 m resolution scale of the multi-spectral image. The spatial heterogeneity of vegetation evident in the field survey appears to be an imprint of the spatial heterogeneity of elevation, as we show that very fine elevation changes, measurable in centimeters, were associated with vegetation assemblage changes. The link between microtopographic heterogeneity and increased species richness has been documented in several wetland studies

(Bledsoe and Shear, 2000; Brose, 2001; Bruland and Richardson, 2005). Variable marsh surface elevations can increase both surface area (Peach and Zedler, 2006) and the number of available microhabitats (Peach and Zedler, 2006; Vivian-Smith, 1997) and, in tidal settings, may alter hydrologic flows in ways that reduce scouring and encourage seed and propagule deposition.

Species frequency and cover differed among the three spectrally identified elevation classes mapped in this study, refuting the null hypothesis of random species distributions. Even at 5-cm elevation increments, we observed strong directional changes in species frequency and cover, with some species preferring lower elevations (e.g., *N. luteum*, *Z. aquatica*), some preferring intermediate elevations (e.g., *Bidens* spp., *A. calamus*), and some preferring highest elevations (e.g., *I. capensis*, *S. fluviatilis*). Some species, such as *P. virginica*, had broad elevation ranges, while others, such as *Z. aquatica* and *Bidens* spp., had more restricted ranges.

Individual species abundance was generally more strongly correlated with elevation than with channel size and proximity, as measured by CISD. However, the CISD correlations observed within middle and upper elevation classes suggest that channels may play a role in determining spatial distributions of certain species within limited elevation ranges. The positive correlation between CISD and *Amaranthus cannabinus* at intermediate elevations suggests that the distribution of this species is in part influenced by channel networks that deposit seeds of *A. cannabinus* close to tidal channels. *Amaranthus cannabinus* was the third most abundant species sampled in water surface trawls along the shoreline of Dyke Marsh (Hopfensperger, 2007) after *I. capensis* and *Pilea pumila*. *Impatiens capensis* is abundant throughout the

marsh such that its distribution is not limited by seed influx, and, hence, the positioning of tidal channels. Indeed, elevation was more strongly correlated with *I. capensis* abundance than was CISD. *Pilea pumila* was sampled infrequently and therefore included in only the species richness analysis for this study. However observations from preliminary work in 2004 suggest that this species is only abundant at the margins of the marsh, where elevations are higher, and that *P. pumila* distribution is therefore probably limited by habitat more than seed influx.

In some cases a decrease in abundance with increasing CISD values may be related to differences in species senescence. Specifically, *P. arifolium* and *C. gronovii* were in varying stages of senescence across the marsh at the time of sampling, and their cover diminished with increased senescence. Negative CISD correlations with species abundance may represent spatial patterns of accelerated senescence due to hydrologic stress. On the other hand, the smallest channels in the marsh are less subject to tidal scouring and could selectively deposit more propagules and seeds of *P. arifolium* and *C. gronovii* than larger channels would. As mid-season abundance, seedling establishment, and senescence were not measured in this study, clear conclusions about how channels may be impacting species not at peak biomass cannot be drawn.

When examining species associations across all elevations combined (Figure 9), we observed that *S. fluviatilis*, *I. capensis*, *P. arifolium*, and *C. gronovii* were strongly associated with each other and with elevation, but negatively associated with *N. luteum* and CISD. These observations corroborate earlier evidence that species associations do exist within the marsh and that species are therefore not randomly

distributed. However, when looking through this lens, it is unclear whether the observed patterns are simply an imprint of environmental gradients or whether species interactions are at work as well. Shifting the focus to examine species associations within elevation classes controls for variations in elevation to illuminate negative and positive patterns of species interactions. While some correlations with environment were still observed within elevation classes, they were not as dominant as species-environment correlations across the entire environmental gradient. We observed an increase in interspecific associations at higher elevations. This observation is congruent with patterns predicted by plant strategy theorists and has been documented in several empirical studies of freshwater wetlands. Grime's competitor – stress tolerator – ruderal (CSR) theory predicts that more stressful habitats will select for stress tolerant species and less stressful undisturbed habitats will favor competitive species (Campbell and Grime, 1992; Grime, 1977). Tilman's resource-ratio competitive models recognize that variations in resources and habitat productivity will favor shifting dominance by different species (Fargione and Tilman, 2006; Tilman, 1985). In Dyke Marsh Preserve, elevation class 1 habitats, which feature high hydrologic stress, clearly favor a few flood tolerant species. As elevation class 1 grades into elevation class 2, increases in species richness and positive interspecific correlations suggest that the resources available in these locations can be successfully utilized by a greater number of species and that, perhaps, facilitative interactions may be at work (Ervin, 2005; Freestone, 2006). At highest elevations (class 3), the number of negative interspecific correlations increased, despite an equal number of species sampled in elevation classes 2 and 3, suggesting that distributions

may be increasingly governed by competitive interactions (Brose and Tielborger, 2005; Ervin and Wetzel, 2003; Farnsworth and Meyerson, 2003; Gaudet and Keddy, 1995b). The strength of competitive dynamics may be a result of the increased vigor of more competitive species under decreased hydrologic stress. For example, *Impatiens capensis*, which emerges early in the season, showed cover increases with rising elevation, which may have reduced available light for species that germinated later. On the other hand, it is also possible that highest elevations are inundated infrequently enough to reduce nutrient availability and allow species that are better able to capture resources or withstand nutrient draw-down to flourish. However, Hopfensperger (2007) showed that sediment total nitrogen (57.5 ± 4.6 mg N/L), and nitrate (7.1 ± 1.0 mg N/L) increased with elevation at Dyke Marsh Preserve, suggesting that more nutrients are available for plant uptake at higher elevations. This points to light as the most limiting factor in these highly productive areas of the marsh, which is a hypothesis that will need to be tested with carefully controlled experiments to further explore the mechanisms at work at different marsh surface elevations.

Perhaps the most intriguing and interpretable negative correlation among species was observed between *Schoenoplectus fluviatilis* and *Typha* spp., which were negatively correlated at both intermediate (class 2) and high (class 3) elevations. Both species occupied similar ranges, with *Typha* spp. generally observed more frequently. Their cover patterns, however, showed opposite trends. *Typha* spp. cover decreased with increasing elevation, while *S. fluviatilis* cover increased. This matches ground observations, particularly in the western central portion of the marsh, of areas where

S. fluviatilis appears to replace *Typha* spp. as the dominant cover of similar growth habit. Because *Typha* spp. is typically a dominant genus in tidal freshwater marshes (Farnsworth and Meyerson, 2003; Mitsch and Gosselink, 2000), a prevalent vegetation class with reduced *Typha* spp. presence is interesting.

While we are unaware of any studies of competitive dynamics between *Typha* spp. and *S. fluviatilis*, both appear to be very strong competitors. Gaudet and Keddy (1995a) ranked *Typha x glauca* and *S. fluviatilis* 5th and 6th, respectively, when comparing 40 species' abilities to suppress growth in a reference phytometer (*Lythrum salicaria*). In a similar study, Keddy, et al. (Keddy et al., 2000) ranked *Typha x glauca* and *S. fluviatilis* 1st and 5th, respectively, where the *Typha* hybrid suppressed 95% of above-ground biomass and 97% of below-ground biomass in the phytometer, and *S. fluviatilis* suppressed 92% and 93% of above- and below-ground biomass, respectively, in high nutrient conditions.

Several studies link competitive ability to productivity (Farnsworth and Meyerson, 2003; Gaudet and Keddy, 1995b; Keddy et al., 1994). In a comparison of *Typha angustifolia* and *Schoenoplectus tabernaemontani*, which grows taller than *S. fluviatilis*, Anderson and Mitsch (2005) observed that *T. angustifolia* produced larger above- and below- ground biomass than *S. tabernaemontani*. While it is uncertain whether the *Typha* spp. at Dyke Marsh Preserve are *T. angustifolia*, the *x glauca* hybrid of *T. angustifolia* and *Typha latifolia*, or a mixture, it seems probable that the *Typha* spp. present in this marsh are more productive than *S. fluviatilis* and would therefore have any competitive advantage that size conferred.

Size notwithstanding, high *S. fluviatilis* cover and reduced *Typha* spp. cover are evident at highest surveyed elevations in Dyke Marsh. Nutrient limitation is unlikely at this elevation, and since neither species is shade tolerant (NRCS, 2007), competitive advantage may lie in which species is able to colonize an area first, establishing and maintaining dominance through local dispersal and aggregation (Gardner and Engelhardt 2007).

Management Implications

Our results show that high spatial resolution multi-spectral imagery can be used to map and monitor relationships between topography and vegetation in tidal freshwater marshes. Topography in the currently existing portion of Dyke Marsh Preserve was found to be quite complex, consisting of interconnected tidal channel networks and highly heterogeneous elevation shifts. Further, the data reveal that extremely fine-scale changes in elevation and channel size and proximity resulted in vegetation changes. We suggest, therefore, that robust native vegetation assemblages will be more likely to develop in future restored areas if constructed areas mimic these complex surfaces.

Chapter 3: Reflections

Earlier Approach

When I first set out to explore the utility of using remote sensing to describe vegetation assemblages and environmental gradients in tidal freshwater marshes, I approached the task from a different direction. My first inclination was to look for traditional phytosocial vegetation assemblages, using flexible beta clustering of vegetation data, and then determine whether a) the vegetation classes could be located with remotely sensed spectral data, b) elevation would differ among vegetation classes, and c) whether CISA values would differ among vegetation classes. This approach quickly became quite complex, when it became apparent that spectral signatures did not correlate well with the vegetation classes. And because my original sampling scheme relied on spectral classes delineated with an ISODATA unsupervised image classification, many of the vegetation classes detected with flexible beta clustering were under-represented in the data.

Because the QuickBird image did not arrive until early October, sampling occurred quite late in the season. This was helpful on the one hand, because it meant that spectral distinctions due to species-specific senescence patterns would be captured. But on the other hand, it meant that field sampling occurred right at the end of the vegetation season. By the time I processed the data and grasped the difficulties presented by uneven sample numbers in the vegetation classes, it was too late to return to the field to gather additional data.

While the subsequent analysis of the data from a vegetation class basis was unsatisfactory in many respects, it was still informative. The maps that resulted from this approach convey slightly different information than the elevation map that was reported in the final work. Also, the difficulties and successes encountered seem worth reporting as they may be useful information for others.

Data structure

The earlier analysis relied on species presence/absence rather than species cover data. This was done because it allowed me to add additional data to the analysis that were collected earlier in the season in July. (Hopfensperger, 2007). While seasonal differences prohibited using species cover information, the marsh was at the peak of the growing season when it was sampled in July, and presence/absence data were therefore not likely to have differed from the time when I sampled.

Secondly, I examined the influence of infrequent species on the analysis by analyzing parallel datasets. The 1% data included all species occurring in more than 1% of study plots, and the 5% data included all species occurring in more than 5% of study plots.

Vegetation classes

Indicator Species Analyses were performed on a series of flexible beta clustering classifications to pinpoint the optimal number of classes to use when characterizing vegetation species associations (figure 3-1). I chose to use the beta classifications with the largest number of classes that still retained sufficient

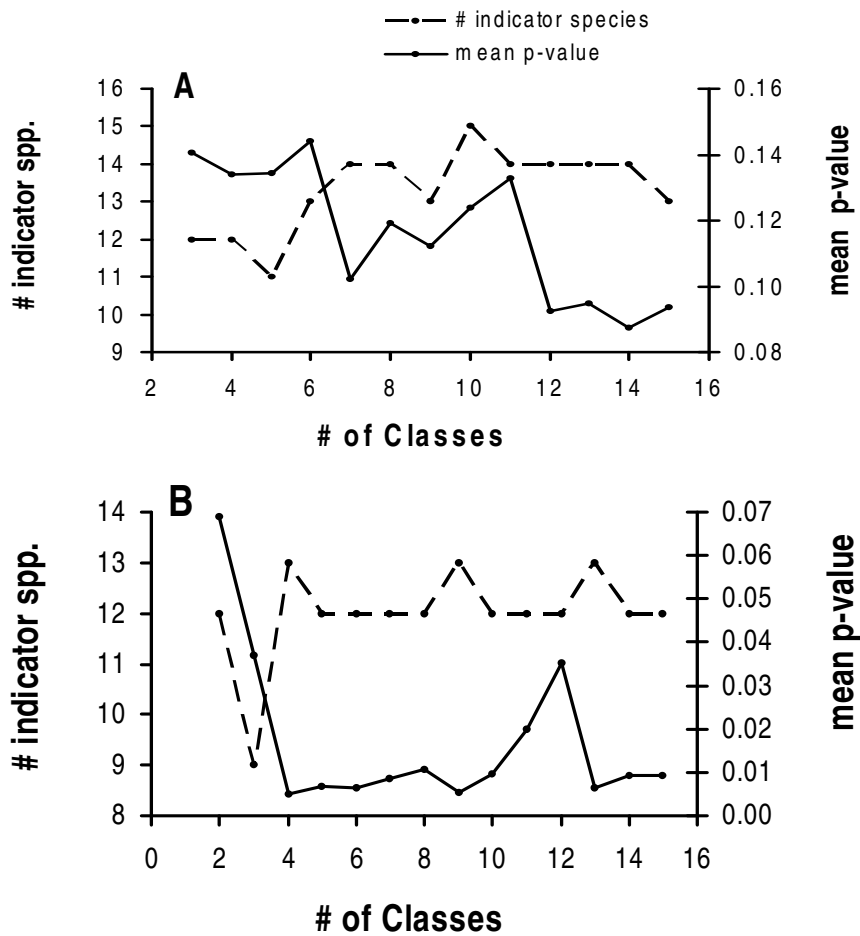


Figure 3-1: Indicator Species Analysis results for a series of flexible beta cluster analyses with differing numbers of classes. Higher numbers of indicator species and lower mean p-values are desirable when selecting the optimal class number. A) The 1% dataset removed species occurring in 1% or fewer of all study plots prior to analysis, and B) the 5% dataset removed species occurring in 5% or fewer plots.

observations for subsequent analyses and also maximized the number of indicator species and minimized mean p-values in the data. This resulted in the selection of a 7-cluster solution for the 1% data and a 9-cluster solution for the 5% data.

To look for spectral separability, I ran Multi-Response Permutation Procedure pairwise comparisons of spectral data for all vegetation classes as well as unbiased recursive partitioning trees, reviewing predictive accuracies for each of the classes.

These examinations prompted me to combine vegetation classes that could not be reliably located spectrally and resulted in four final vegetation classes for each dataset.

An examination of the species present in the final vegetation classes, ordered by mean elevation of each class, shows that the 1% vegetation classes (table 3-1) generally characterized the vegetation present in the following way:

LM = low marsh plots, dominated by *Nuphar luteum*;

HM1 = high marsh plots dominated by the perennials *Peltandra virginica*, *Typha* spp. and *Leersia oryzoides*;

HM2 = high marsh plots with low *Typha* spp. presence and dominated the annuals *Impatiens capensis* and *Polygonum arifolium* as well as the graminoid *Schoenoplectus fluviatilis*; and

HM3 = high marsh plots dominated by *I. capensis*, *Typha* spp., and *P. virginica*.

The 5% vegetation classes (table 3-2) characterized the marsh vegetation in a slightly different way at middle elevations:

lm = low marsh plots, dominated by *N. luteum*;

hm1 = high marsh plots without annual species, dominated by *Typha* spp.;

hm2 = high marsh plots dominated by annuals; and

hm3 = high marsh plots dominated by *I. capensis*, *Typha* spp., and *P. virginica*.

Class	Indicator Species for 1% Data													
	<i>Data Type</i>	<i>A. calamus</i>	<i>A. canna-binus</i>	<i>Bidens spp.</i>	<i>C. gro-novii</i>	<i>I. ca-pensis</i>	<i>L. ory-zoides</i>	<i>N. lutea</i>	<i>P. vir-ginica</i>	<i>P. aus-tralis</i>	<i>P. ari-folium</i>	<i>P. punc-tatum</i>	<i>S. fluvi-atilis</i>	<i>Typha spp.</i>
Low Marsh (N = 43)														
flexible beta class 28 (N=12)	0	0	0	0	0	0	10	9	0	0	2	3	1	10
flexible beta class 41 (N = 31)	4	0	0	2	4	1	31	6	0	5	0	0	6	0
<i># Low Marsh plots w/species</i>	4	0	0	2	4	1	41	15	0	5	2	3	7	10
<i>species % per Low Marsh plots</i>	9%	0%	0%	5%	9%	2%	95%	35%	0%	12%	5%	7%	16%	23%
<i>species % per total species plots</i>	12%	0%	0%	7%	4%	4%	67%	13%	0%	10%	29%	6%	8%	100%
High Marsh 1 (N = 11)														
flexible beta class 11 plots (N = 11)	0	3	3	0	1	6	0	10	2	0	4	0	7	0
<i># High Marsh 1 plots w/species</i>	0	3	3	0	1	6	0	10	2	0	4	0	7	0
<i>species % per High Marsh 1 plots</i>	0%	27%	27%	0%	9%	55%	0%	91%	18%	0%	36%	0%	64%	0%
<i>species % per total species plots</i>	0%	27%	8%	0%	1%	26%	0%	9%	100%	0%	57%	0%	8%	0%
High Marsh 2 (N = 33)														
flexible beta class 1 (N = 17)	17	0	10	0	16	2	2	16	0	12	0	4	6	0
flexible beta class 16 (N = 16)	0	0	4	2	15	3	0	10	0	12	0	16	6	0
<i># High Marsh 2 plots w/species</i>	17	0	14	2	31	5	2	26	0	24	0	20	12	0
<i>species % per High Marsh 2 plots</i>	52%	0%	42%	6%	94%	15%	6%	79%	0%	73%	0%	61%	36%	0%
<i>species % per total species plots</i>	50%	0%	37%	7%	30%	22%	3%	22%	0%	50%	0%	42%	13%	0%
High Marsh 3 (N = 69)														
flexible beta class 7 (N = 69)	13	8	21	23	68	11	18	65	0	19	1	25	63	0
<i># High Marsh 3 plots w/species</i>	13	8	21	23	68	11	18	65	0	19	1	25	63	0
<i>species % per High Marsh 3 plots</i>	19%	12%	30%	33%	99%	16%	26%	94%	0%	28%	1%	36%	91%	0%
<i>species % per total species plots</i>	38%	73%	55%	85%	65%	48%	30%	56%	0%	40%	14%	52%	71%	0%
Total study plots where present	34	11	38	27	104	23	61	116	2	48	7	48	89	10

Table 3-1: 1% Dataset Summary – indicator species presence for emergent macrophytes within 1 m² quadrats. Species present in < 1% of all plots have been removed from the dataset.

Class	Indicator Species for 5% Data												
	Data Type	<i>A. calamus</i>	<i>A. canna- binus</i>	<i>Bidens spp.</i>	<i>C. gro- novii</i>	<i>I. ca- pensis</i>	<i>L. ory- zoides</i>	<i>N.- lutea</i>	<i>P.vir- ginica</i>	<i>P. ari- folium</i>	<i>S.lati- folia</i>	<i>S. fluvi- atilis</i>	<i>Typha spp.</i>
low marsh (N = 34)													
flexible beta class 1 (N=12)	0	0	0	0	0	0	10	9	0	4	3	1	10
flexible beta class 3 (N = 22)	3	0	0	0	0	0	22	1	1	0	0	0	0
total low marsh plots w/species	3	0	0	0	0	0	32	10	1	4	3	1	10
species per low marsh plots	9%	0%	0%	0%	0%	0%	94%	29%	3%	12%	9%	3%	29%
species per species total plots	9%	0%	0%	0%	0%	0%	52%	9%	2%	25%	7%	1%	100%
high marsh 1 (N = 11)													
flexible beta class 7 (N = 11)	0	0	0	0	0	2	5	11	0	0	0	8	0
total high marsh 1 plots w/species	0	0	0	0	0	2	5	11	0	0	0	8	0
species per high marsh 1 plots	0%	0%	0%	0%	0%	18%	45%	100%	0%	0%	0%	73%	0%
species per species total plots	0%	0%	0%	0%	0%	9%	8%	10%	0%	0%	0%	9%	0%
high marsh 2 (N = 10)													
flexible beta class 35 (N = 10)	10	0	10	0	9	2	2	9	6	2	1	1	0
total high marsh 2 plots w/species	10	0	10	0	9	2	2	9	6	2	1	1	0
species per high marsh 2 plots	100%	0%	100%	0%	90%	20%	20%	90%	60%	20%	10%	10%	0%
species per species total plots	30%	0%	26%	0%	9%	3%	3%	8%	13%	13%	2%	1%	0%
high marsh 3 (N = 99)													
flexible beta class 2 (N = 39)	4	5	17	5	38	4	16	35	1	5	4	36	0
flexible beta class 13 (N = 28)	7	1	5	15	28	3	2	27	12	2	28	15	0
flexible beta class 23 (N = 9)	1	0	1	4	8	3	4	0	9	0	5	8	0
flexible beta class 32 (N = 17)	8	1	0	2	16	3	0	17	15	0	4	15	0
flexible beta class 40 (N = 6)	0	4	5	0	3	6	0	5	2	3	1	3	0
total high marsh 3 plots w/species	20	11	28	26	93	19	22	84	39	10	42	77	0
species per high marsh 3 plots	20%	11%	28%	26%	94%	19%	22%	85%	39%	10%	42%	78%	0%
species per species total plots	61%	100%	74%	100%	91%	83%	36%	74%	85%	63%	91%	89%	0%
Total study plots where present	33	11	38	26	102	23	61	114	46	16	46	87	10

Table 3-2: 5% Dataset Summary – indicator species presence for emergent macrophytes within 1 m² quadrats. Species present in < 5% of all plots have been removed from the dataset.

Elevation among vegetation classes

Multi-Response Permutation Procedure tests for elevation differences among vegetation classes revealed that, in the case of the 1% data, the LM class differed from each of the high marsh classes in pairwise comparisons ($p = 0.000$, $A = 0.22 - 0.44$), but that the high marsh classes did not differ from one another when Bonferonni corrections were taken into account. In the case of the 5% data, all classes differed in pairwise comparisons ($p = 0.000$, $A = 0.07 - 0.52$), with the exception of hm1 and hm2. This meant that the class devoid of annuals did not significantly differ in elevation from the class dominated by annuals.

CISD values among vegetation classes

Pairwise MRPP tests for CISD differences among vegetation classes revealed differences between the low marsh class and each of the high marsh classes in the 1% data ($p = 0.000 - 0.004$, $A = 0.04 - 0.12$). The low marsh class only differed from the two highest elevation classes in the 5% data ($p = 0.000$, $A = 0.09 - 0.13$) when Bonferonni corrections were taken into account. Plot CISD values did not differ among any of the high marsh vegetation classes in either dataset.

Initial conclusions

These results suggest that, while several vegetation classes could be reliably delineated and mapped with the aid of remotely sensed reflectance data (figure 3-2),

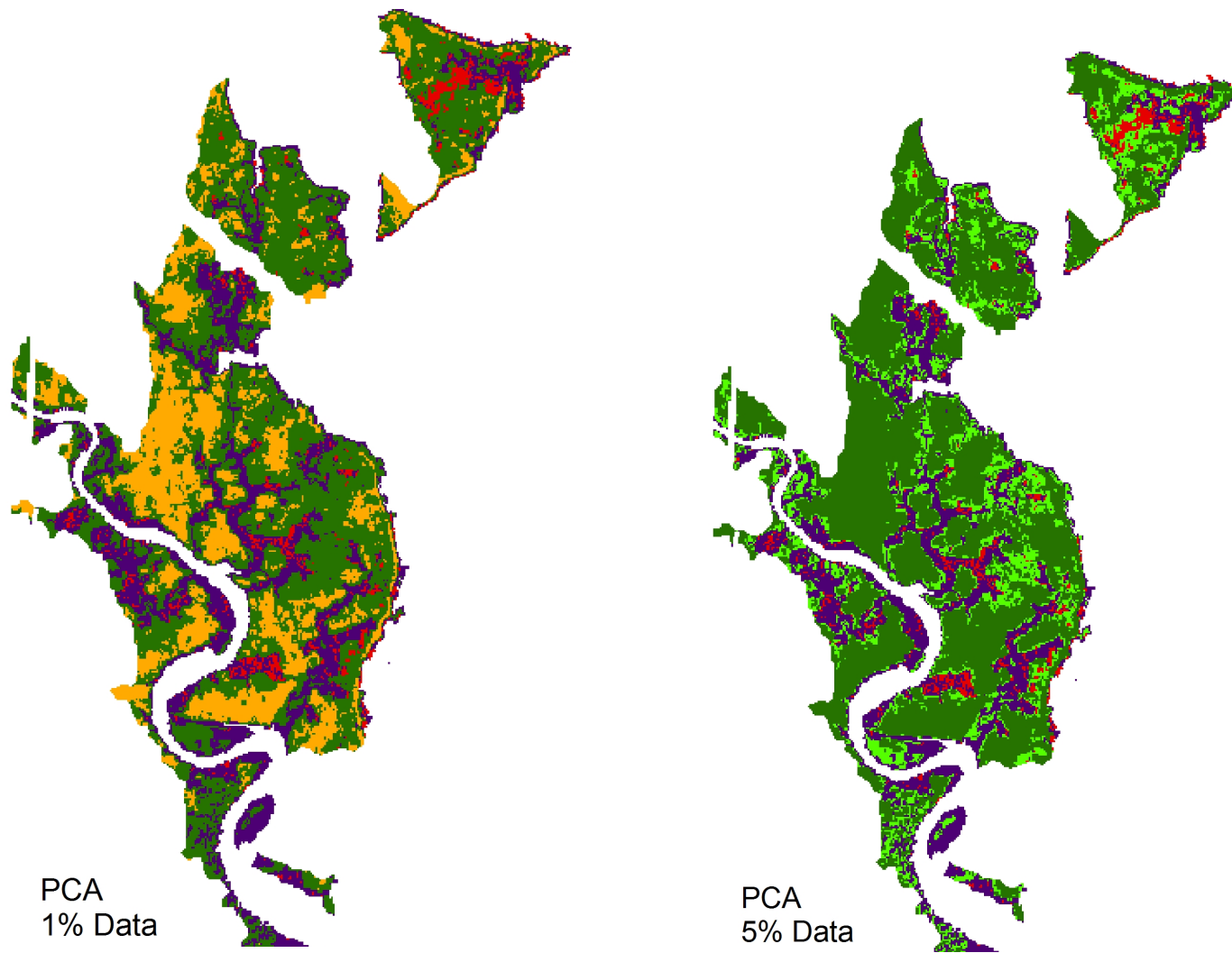


Figure 3-2: Decision tree image classifications of marsh vegetation for the 1% data, where species occurring in 1% or fewer of plots were removed from the analysis, and for the 5% data, where species occurring in 5% or fewer plots were removed. Colors represent vegetation classes ordered by decreasing mean elevation as follows: orange, dark green, light green, red, and purple.

the resulting high marsh vegetation classes did not differ for either elevation or CIRD. This seemed a reasonable outcome; however, canopy reflectances may have a strong substrate component, and might be correlated with elevation or soil moisture differences just as much or more than they could be attributed to vegetation differences. I therefore created a map that was derived from elevation and spectral data alone, without reference to vegetation data.

Elevation map and revised conclusions

I ran a 5-cluster flexible beta analysis of the remotely sensed spectral data and the 537 elevation readings taken in the field (table 3-3). Multi-Response Permutation Procedures tests confirmed that the resulting classes differed for elevation ($p = 0.000$,

Class	Mean Elev. (m)	St. Err.
a	Hog Island Gut - unvegetated	
b	0.033	0.022
c	0.275	0.014
d	0.362	0.011
e	0.423	0.006

Figure 3-3: Mean elevations for 5 elevation classes derived from QuickBird spectral reflectance and field measurements of elevation at Dyke Marsh Preserve.

$A = 0.306 - 0.676$), so I followed up with a spectral classification tree of the composite elevation/spectral classes. The resulting map (figure 3-3) was remarkable in that it clearly mimicked features of both the 1% and 5% vegetation maps previously produced. In particular, low marsh areal coverage was virtually identical in the three maps; and highest marsh elevations correlated with the spatial arrangement of the HM3 class in the 1% data, which featured low *Typha* spp.

Elevation Classes

Predictive Accuracy and Coverage (ha)

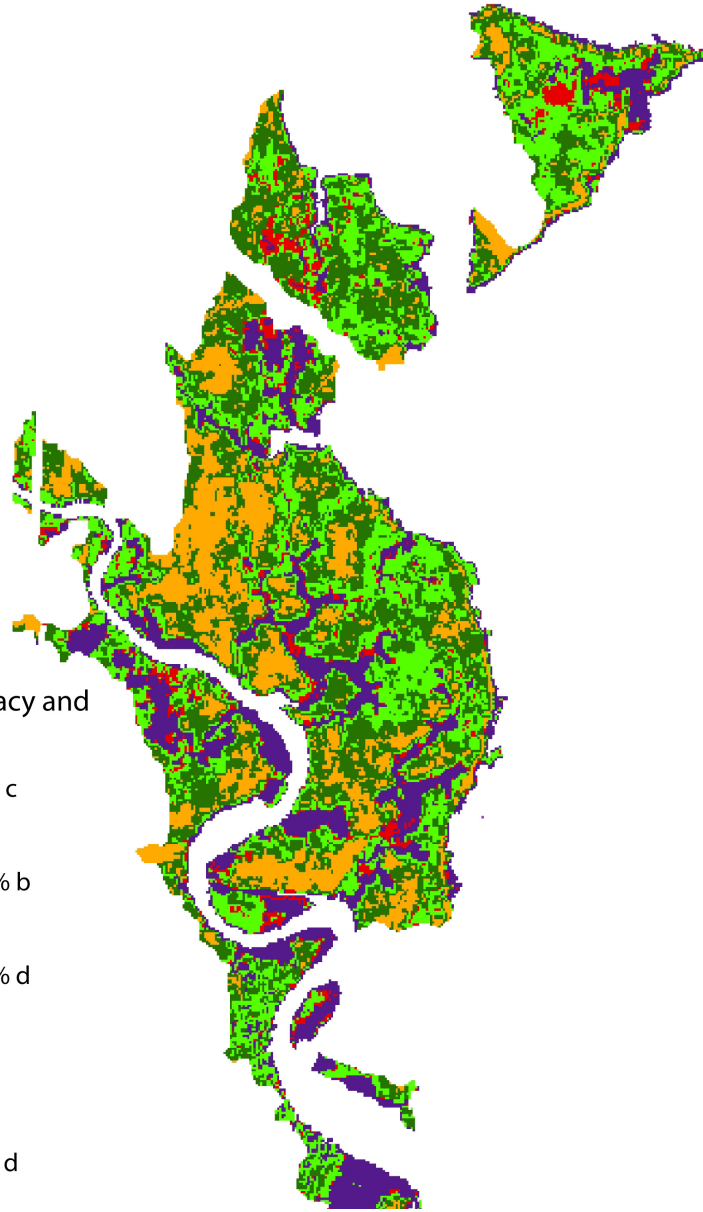
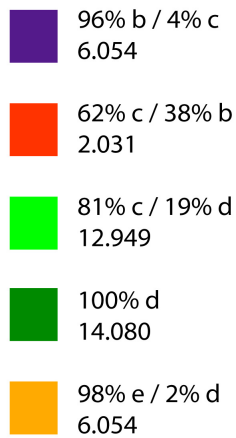


Figure 3-3: Decision tree image classification for composite spectral and elevation classes. Percentages in map legend reflect terminal node impurities in the classification tree.

presence and high *S. fluviatilis* presence. This finding seemed to directly refute the conclusion that elevation did not differ among vegetation classes. The mapped elevation classes clearly differed for elevation, and they were spatially correlated with the vegetation classes. It seemed likely, therefore, that the vegetation data was too heterogeneous and unevenly limited in observations to detect elevation differences

that were likely to actually exist. This prompted my revisiting the problem from a different direction.

The initial study showed that retaining more infrequently observed species in the presence/absence data (the 1% dataset) allowed finer discrimination of vegetation assemblages higher elevations. The data removing more infrequent species discriminated more finely at moderate elevations. This suggests that the least frequently sampled species may have been intolerant of prolonged hydroperiods.

Note on elevation class maps

While the elevation map in Figure 3-3 was a useful tool for discovering connections between elevation and vegetation in the data, there was no way to independently validate the patterns. When creating this elevation map, observations were assigned to their respective elevation classes using both spectral and elevation data. As a result, all the data were weighted, by the nature of how they were created, to spectrally locate the classes of interest. This is why I approached the new analysis using elevations alone as the basis for class creation. While the map resulting from the latter approach was not as finely detailed, it could be independently validated, referencing elevations from withheld plot data.

Note on channel influences

I am uncertain whether multi-spectral mapping can fully illuminate channel impacts on vegetation distributions in tidal freshwater marshes. Channels occur at almost all elevations in Dyke Marsh Preserve. Additionally, they have many

morphologies, combining all permutations of narrow, broad, shallow, and deep. In some places, the banks almost imperceptibly grade into the surrounding marsh, while at others the banks steeply drop. Levies vary in similar fashion, sometimes appearing non-existent, and at other times being clearly defined. While the cross-sectional area embedded in the CISD metric captures a portion of channel influence, it does not account for such morphological differences. It seems likely that the impacts of tidal flushing will vary substantially in channels of the same volumetric size but differing morphologies. If this difference could be quantified, perhaps with fine-scale LIDAR or other topographic mapping, channel impacts on vegetation distributions might be more fully explored.

Appendix I, Quadrat Locations, Elevations, CISD values

ID	Plot	NORTHING WGS84, UTM 18N	EASTING WGS84, UTM 18N	ELEVATION (m)	CISD value
1	07-01	4292918.6334	321685.3290	-0.1952	1.580
2	07-06	4293101.2729	321969.0377	0.4415	0.188
3	08-01	4293550.6212	322286.2467	-0.1343	75.444
4	08-02	4293239.1247	321893.5040	0.0304	8.813
5	08-03	4292927.3211	321756.3884	-0.0841	2.041
6	08-04	4292912.5648	321915.0728	-0.1694	23.859
7	08-05	4292887.0865	321972.3121	0.0514	13.017
8	08-06	4292794.9516	321899.2279	-0.0650	13.577
9	08-07	4292670.9454	321838.2665	0.0919	3.123
10	08-08a	4292718.1929	321927.9465	0.0022	6.026
11	08-08b	4292708.1859	321971.7042	0.1392	300.982
12	08-09	4292617.2548	321893.4644	-0.0895	2.189
13	08-10	4293241.2899	321921.2695	-0.1856	15.751
14	09-01	4293549.9774	322303.8563	-0.1073	53.241
15	09-02	4293317.9583	322002.2087	0.4481	0.002
16	09-03	4293246.1809	321885.3319	0.1265	3.622
17	09-04	4293182.8627	321906.5295	-0.2846	8.252
18	09-05	4293041.5985	321659.9883	0.1104	0.651
19	09-06	4292860.2812	321767.5967	0.1767	0.224
20	09-07	4292911.7256	321789.5460	0.0307	11.792
21	09-08	4292667.0060	321830.9411	0.0846	6.990
22	09-09	4292741.6258	322016.5466	0.0957	12.192
23	10-01	4293537.9420	322222.7240	0.4911	0.537
24	10-02	4293521.0484	322261.7654	0.4601	1.115
25	10-03	4293421.3461	321976.6774	0.5123	0.080
26	10-04	4293208.6281	321898.2556	0.3699	3.922
27	10-05	4292853.6448	321784.4347	0.2845	0.422
28	10-06	4292905.9534	321994.3158	0.1781	3.226
29	10-07	4292892.2721	322024.3921	0.2654	2.669
30	10-08	4292597.3897	321863.0569	-0.0304	12.956
31	10-09	4292702.1901	322021.4490	-0.1885	4.760
32	10-10	4293039.4633	321641.7681	0.0653	0.112
33	11-01	4293575.1331	322203.3352	0.4634	0.401
34	11-02	4292862.5872	322031.9283	0.3491	0.066
35	11-03	4292883.8847	322040.5048	0.3630	0.099
36	11-04	4292599.3359	321847.2880	-0.5165	22.891
37	11-06	4293378.4723	321976.5042	0.3888	0.159
38	11-07	4292895.3173	322055.7217	0.3790	0.023
39	11-08	4293347.3681	322033.7269	0.4162	0.349
40	11-09	4293021.5481	321998.5718	0.3402	0.486
41	11-10	4293147.9123	321617.2473	0.2504	0.235
42	12-01	4293342.8804	321961.0161	0.3686	3.537
43	12-02	4293394.3567	321978.0320	0.4590	0.280
44	12-03	4293182.0619	321827.8558	0.2548	22.012
45	12-04	4293178.8076	321878.3085	0.3279	1.075

ID	Plot	NORTHING WGS84, UTM 18N	EASTING WGS84, UTM 18N	ELEVATION (m)	CISD value
46	12-05	4293041.6845	321935.0947	0.3036	0.191
47	12-06	4292854.8498	322041.8812	0.3741	0.041
48	12-08	4293201.4572	321802.2331	0.4027	0.755
49	12-09	4292618.3947	321878.7663	-0.0442	2.867
50	12-10	4292904.9814	322028.3564	0.3971	1.523
51	13-03	4293183.6805	321844.4043	0.3887	0.460
52	13-04	4293051.6908	321985.6297	0.4589	0.000
53	13-05	4292813.0444	321988.0265	0.3692	0.196
54	13-06	4293291.5091	322005.1186	0.4985	0.034
55	13-08a	4292729.3410	321815.2532	0.3941	0.000
56	13-08b	4292915.1511	322103.8296	0.4535	0.000
57	13-09	4292938.7084	322012.3444	0.3902	0.849
58	13-10	4292793.8879	322092.4086	0.4441	0.227
59	14-02	4293059.5747	322003.3065	0.4649	0.000
60	14-03	4293030.4994	321981.1220	0.4243	0.000
61	14-04	4292959.0327	321817.7185	0.4403	0.023
62	14-05	4292886.0742	321866.0590	0.4270	2.320
63	14-06	4292945.5822	322005.5811	0.3965	0.235
64	14-07	4292869.6014	321927.8668	0.3963	2.915
65	14-08	4292775.8865	322055.3157	0.3805	1.578
66	14-09	4292749.6147	321992.3850	0.3946	0.105
67	14-10	4292923.1611	322090.0822	0.4135	0.121
68	15-01	4293584.6739	322137.3939	0.4913	0.011
69	15-02	4293405.2452	321893.6119	0.2936	1.807
70	15-03	4293029.8610	321960.3165	0.3931	0.080
71	15-04	4293064.1258	321864.4938	0.4447	0.461
72	15-05	4292963.7838	321798.9428	0.4180	0.000
73	15-06	4292874.6961	321877.5875	0.4983	0.398
74	15-07	4292792.7671	321959.1944	0.3104	0.236
75	15-08	4292649.2359	322045.2599	0.4424	0.024
76	15-09	4292724.2928	322006.7676	0.3739	0.247
77	15-10	4292766.7913	322046.9994	0.4017	2.915
78	16-01	4293279.5768	321824.0277	0.4864	0.064
79	16-02	4293057.5323	321879.4831	0.3967	0.041
80	16-03	4293041.8724	321815.8036	0.4404	0.163
81	16-04	4292969.5671	321775.2707	0.3925	0.096
82	16-05	4292806.6878	321957.3335	0.3466	0.338
83	16-06	4292784.4888	321864.9508	0.4056	0.000
84	16-07	4292748.9222	321979.0473	0.3833	0.000
85	16-08	4292713.3193	321787.1346	0.4871	0.101
86	16-09	4292658.7941	321880.7546	0.4403	0.704
103	A-01	4293559.7125	322275.9861	0.4666	5.082
104	A-02	4293474.3624	321901.8653	0.5499	0.000
105	A-03	4293361.2006	321939.2346	0.3438	8.340
106	A-04	4292951.6442	321836.7523	0.4545	0.252
107	A-05	4293021.9671	321966.1241	0.4075	0.004
108	A-06	4293039.7275	321929.8867	0.3454	0.729
109	A-07	4293035.3616	321803.2759	0.3316	1.112
110	A-08	4292944.1638	322014.3773	0.3477	0.417

ID	Plot	NORTHING WGS84, UTM 18N	EASTING WGS84, UTM 18N	ELEVATION (m)	CISD value
111	A-09	4292718.6148	321925.3996	-0.0943	3.714
112	A-10	4292274.7833	322037.1800	0.0908	0.000
113	A-11	4292615.9807	322025.6555	0.4299	0.000
114	A-12	4292661.6454	321892.2368	0.4206	0.006
115	A-13	4292665.7518	321837.9316	0.2403	0.000
116	A-14	4292947.8381	322082.4908	0.3182	0.617
117	A-15	4293260.7406	322053.8365	0.3560	0.546
118	A-16	4293248.7331	322037.2593	0.3667	3.290

Appendix II, Quadrat Vegetation OTV Values

Species Abbreviations

ACOCAL	<i>Acorus calamus</i>
AMACAN	<i>Amaranthus cannabinus</i>
APIAME	<i>Apios americana</i>
BIDsp	<i>Bidens</i> species
CALSEP	<i>Calystegia sepium</i>
CEPOCC	<i>Cephalanthus occidentalis</i>
CORAMA	<i>Cornus amomum</i>
CUSGRU	<i>Cuscuta gronovii</i>
ECHCRU	<i>Echinochloa crusgalli</i>
ECHMUR	<i>Echinochloa muricata</i>
HIBPAL	<i>Hibiscus palustris</i>
IMPCAP	<i>Imaptiens capensis</i>
LEEORI	<i>Leersia oryzoides</i>
MIKSCA	<i>Mikania scandens</i>
NUPLUT	<i>Nuphar luteum</i>
PELVIR	<i>Peltandra virginica</i>
PHRAUS	<i>Phragmites australis</i>
PILPUM	<i>Pilea pumila</i>
POLARI	<i>Polygonum arifolium</i>
POLCOC	<i>Polygonum coccineum</i>
POLPUN	<i>Polygonum punctatum</i>
PONCOR	<i>Pontederia cordata</i>
ROSPAL	<i>Rosa palustris</i>
SAGLAT	<i>Sagittaria latifolia</i>
SCHFLU	<i>Schoenoplectus fluviatilis</i>
TYPsp	<i>Typha</i> species
VITsp	<i>Vitis</i> species
ZIZAQU	<i>Zizania aquatica</i>

ID	Plot	ACOCAL	AMACAN	APIAME	BIDsp	CALSEP	CEPOCC	CORAMA	CUSGRO	ECHCRU	ECHMUR
63	14-06	0	0	0	0	0	0	0	0	0	0
64	14-07	0	0	0	0	0	0	0	0	0	0
65	14-08	0	0	0	0	5	0	0	8	0	0
66	14-09	0	5	0	5	0	0	0	0	0	0
67	14-10	5	0	0	0	0	0	0	0	0	0
68	15-01	0	0	0	0	0	6	0	0	0	0
69	15-02	6	0	0	8	0	0	0	0	0	0
70	15-03	0	0	0	0	0	0	0	0	0	0
71	15-04	0	0	0	0	0	0	0	0	0	0
72	15-05	0	0	0	0	0	0	0	0	0	0
73	15-06	0	0	0	0	0	0	0	1	0	0
74	15-07	0	0	0	8	0	0	0	0	0	0
75	15-08	0	0	0	0	0	0	0	4	0	0
76	15-09	0	0	0	8	0	0	0	0	0	0
77	15-10	0	8	0	7	0	0	0	0	0	0
78	16-01	4	0	0	8	0	0	0	0	0	0
79	16-02	0	0	0	0	0	0	0	0	0	0
80	16-03	0	0	0	0	0	0	0	0	0	0
81	16-04	4	0	0	0	0	0	0	0	0	0
82	16-05	0	0	0	7	0	0	0	0	0	0
83	16-06	0	0	0	0	0	0	0	6	0	0
84	16-07	0	0	0	0	5	0	0	0	0	0
85	16-08	0	0	0	0	6	0	0	0	0	0
86	16-09	0	0	0	0	0	0	0	0	0	0
103	A-01	0	7	0	0	0	0	0	0	0	0
104	A-02	4	0	0	0	0	0	0	4	0	0
105	A-03	0	5	0	0	0	0	0	0	0	8
106	A-04	0	0	0	0	0	0	8	0	0	0
107	A-05	0	0	0	8	0	0	0	0	0	0
108	A-06	0	0	0	8	0	0	0	0	0	0
109	A-07	0	0	0	0	0	0	8	0	0	0

ID	Plot	ACOCAL	AMACAN	APIAME	BIDsp	CALSEP	CEPOCC	CORAMA	CUSGRO	ECHCRU	ECHMUR
110	A-08	0	0	0	0	0	0	0	5	0	0
111	A-09	0	0	0	0	0	0	0	0	0	0
112	A-10	0	0	0	0	0	0	0	0	0	0
113	A-11	0	0	0	8	0	0	0	0	0	0
114	A-12	0	0	0	0	0	0	0	0	0	0
115	A-13	7	0	0	0	4	0	0	0	0	0
116	A-14	4	0	0	0	0	0	0	0	0	0
117	A-15	0	0	0	6	0	0	0	0	0	0
118	A-16	0	7	0	6	0	0	0	0	4	0

ID	Plot	HIBPAL	IMPCAP	LEEORI	MIKSCA	NUPLUT	PELVIR	PHRAUS	PILPUM	POLARI	POLCOC
1	07-01	0	0	0	0	7	0	0	0	0	0
2	07-06	0	7	0	0	0	5	0	0	0	0
3	08-01	0	0	0	0	8	0	0	0	0	0
4	08-02	0	0	0	0	8	0	0	0	0	0
5	08-03	0	0	0	0	7	0	0	0	0	0
6	08-04	0	0	0	0	8	0	0	0	0	0
7	08-05	0	0	0	0	7	4	0	0	0	0
8	08-06	0	0	0	0	7	0	0	0	0	0
9	08-07	0	0	0	0	8	6	0	0	0	0
10	08-08a	0	0	0	0	0	6	0	0	0	0
11	08-08b	0	0	0	0	6	7	0	0	0	0
12	08-09	0	0	0	0	7	0	0	0	0	0
13	08-10	0	0	0	0	7	0	0	0	0	0
14	09-01	0	0	0	0	8	0	0	0	0	0
15	09-02	0	7	0	0	0	4	0	0	0	0
16	09-03	0	0	0	0	8	5	0	0	0	0
17	09-04	0	0	0	0	7	0	0	0	0	0

ID	Plot	HIBPAL	IMPCAP	LEEORI	MIKSCA	NUPLUT	PELVIR	PHRAUS	PILPUM	POLARI	POLCOC
18	09-05	0	0	0	0	7	6	0	0	0	0
19	09-06	0	0	0	0	5	5	0	0	0	0
20	09-07	0	0	0	0	7	0	0	0	0	0
21	09-08	0	0	0	0	8	5	0	0	0	0
22	09-09	0	0	0	0	0	7	0	0	0	0
23	10-01	0	0	0	0	0	5	0	0	0	0
24	10-02	0	0	0	0	0	6	0	0	0	0
25	10-03	0	8	0	0	0	0	0	0	7	0
26	10-04	0	0	0	0	0	5	0	0	0	0
27	10-05	0	6	0	0	0	5	0	0	0	0
28	10-06	0	0	0	0	7	7	0	0	0	0
29	10-07	0	0	0	0	6	5	0	0	0	0
30	10-08	0	0	0	0	8	0	0	0	0	0
31	10-09	0	0	0	0	8	0	0	0	0	0
32	10-10	0	0	0	0	8	6	0	0	0	0
33	11-01	0	6	0	0	0	5	0	0	0	0
34	11-02	0	0	0	0	0	6	0	0	6	0
35	11-03	0	0	5	0	0	6	0	0	0	0
36	11-04	0	0	0	0	8	0	0	0	0	0
37	11-06	0	6	0	0	0	0	0	0	7	0
38	11-07	0	0	5	0	0	6	0	0	0	0
39	11-08	0	8	0	0	0	4	0	0	0	0
40	11-09	0	7	0	0	0	5	0	0	0	0
41	11-10	0	0	0	0	6	6	0	0	0	0
42	12-01	0	7	3	0	0	6	0	0	5	0
43	12-02	0	8	3	0	0	4	0	0	0	0
44	12-03	0	5	0	0	0	5	0	0	0	0
45	12-04	0	6	0	0	6	5	0	0	0	0
46	12-05	0	6	6	0	0	0	0	0	0	0
47	12-06	0	7	5	0	0	5	0	0	6	0
48	12-08	0	8	0	0	0	5	0	0	8	0

ID	Plot	HIBPAL	IMPCAP	LEEORI	MIKSCA	NUPLUT	PELVIR	PHRAUS	PILPUM	POLARI	POLCOC
49	12-09	0	0	0	0	8	0	0	0	0	0
50	12-10	0	7	3	0	4	4	0	0	0	0
51	13-03	0	7	0	0	0	0	0	0	0	0
52	13-04	0	8	0	0	6	0	0	0	5	0
53	13-05	0	7	0	0	0	5	0	0	0	0
54	13-06	0	6	4	0	0	5	0	0	0	0
55	13-08a	0	5	0	0	5	4	0	0	0	0
56	13-08b	0	8	0	0	0	5	0	0	6	0
57	13-09	0	7	0	0	0	6	0	0	0	0
58	13-10	0	8	4	0	0	5	0	0	5	0
59	14-02	0	8	0	0	0	4	0	0	0	0
60	14-03	0	8	0	0	0	5	0	0	0	0
61	14-04	7	8	0	0	0	5	0	0	5	0
62	14-05	0	0	0	0	6	5	0	0	0	0
63	14-06	0	7	0	0	0	4	0	0	4	0
64	14-07	0	6	0	0	4	5	0	0	0	0
65	14-08	0	8	5	0	5	4	0	0	0	0
66	14-09	0	7	5	0	0	6	0	0	4	0
67	14-10	0	8	0	0	0	4	0	0	8	0
68	15-01	6	7	0	0	0	6	0	0	0	0
69	15-02	0	0	0	0	0	6	0	0	5	0
70	15-03	0	7	0	0	6	0	0	0	8	0
71	15-04	0	7	0	0	0	0	0	0	5	0
72	15-05	0	8	0	0	0	5	0	0	8	0
73	15-06	0	8	0	0	0	5	0	0	0	0
74	15-07	0	7	0	0	0	7	0	0	5	0
75	15-08	0	8	5	0	0	0	0	0	6	0
76	15-09	0	0	0	0	0	0	0	0	6	0
77	15-10	0	5	0	0	0	5	0	0	0	0
78	16-01	0	6	6	0	0	5	0	4	0	0
79	16-02	0	8	0	0	0	4	0	0	8	0

ID	Plot	POLPUN	PONCOR	ROSPAL	SAGLAT	SCHFLU	SCHTAB	TYPsp	VITsp	ZIZAQU
4	08-02	0	0	0	0	0	0	0	0	0
5	08-03	0	0	0	0	0	0	0	0	0
6	08-04	0	0	0	0	0	0	0	0	0
7	08-05	0	0	0	0	0	0	0	0	0
8	08-06	0	0	0	0	0	0	0	0	0
9	08-07	0	0	0	0	0	0	0	0	0
10	08-08a	0	0	0	0	0	0	0	0	7
11	08-08b	0	0	0	0	0	0	0	0	5
12	08-09	0	0	0	0	0	0	0	0	0
13	08-10	0	0	0	0	0	0	0	0	0
14	09-01	0	0	0	0	0	0	0	0	0
15	09-02	0	0	0	0	8	0	0	0	0
16	09-03	5	0	0	4	0	0	0	0	0
17	09-04	0	0	0	0	0	0	0	0	0
18	09-05	0	0	0	0	0	0	0	0	6
19	09-06	5	0	0	4	5	0	0	0	5
20	09-07	0	0	0	0	0	0	0	0	6
21	09-08	0	0	0	0	0	0	0	0	5
22	09-09	0	0	0	4	5	0	0	0	5
23	10-01	0	0	0	0	0	0	8	0	0
24	10-02	0	0	0	0	0	0	8	0	0
25	10-03	0	0	0	0	8	0	4	0	0
26	10-04	0	0	0	0	0	0	8	0	0
27	10-05	0	0	0	0	6	6	0	0	0
28	10-06	0	0	0	0	0	0	8	0	0
29	10-07	0	0	0	0	0	0	8	0	0
30	10-08	0	0	0	0	0	0	0	0	0
31	10-09	0	0	0	0	0	0	0	0	0
32	10-10	0	0	0	0	0	0	0	0	7
33	11-01	0	0	0	0	0	0	8	0	0
34	11-02	0	0	0	0	0	0	8	0	0

ID	Plot	POLPUN	PONCOR	ROSPAL	SAGLAT	SCHFLU	SCHTAB	TYPsp	VITsp	ZIZAQU
35	11-03	0	0	0	0	0	0	8	0	0
36	11-04	0	0	0	0	0	0	0	0	0
37	11-06	0	0	0	5	4	0	0	0	0
38	11-07	0	0	0	0	0	0	8	0	0
39	11-08	0	0	0	0	8	0	0	0	0
40	11-09	0	0	0	0	0	0	8	0	0
41	11-10	0	0	0	4	6	0	0	0	0
42	12-01	0	0	0	0	0	0	7	0	0
43	12-02	0	0	0	4	8	0	4	0	0
44	12-03	0	0	0	0	0	0	8	0	0
45	12-04	0	0	0	0	0	0	8	0	0
46	12-05	0	0	0	0	0	0	8	0	0
47	12-06	0	0	0	0	7	0	7	0	0
48	12-08	0	0	0	0	6	0	0	0	0
49	12-09	0	0	0	0	0	0	0	0	0
50	12-10	0	0	0	0	0	0	8	0	0
51	13-03	0	0	0	0	0	0	7	0	0
52	13-04	0	0	0	0	0	0	8	0	0
53	13-05	0	0	0	4	0	0	6	0	0
54	13-06	0	0	0	4	0	0	0	0	0
55	13-08a	0	0	0	0	0	0	0	0	0
56	13-08b	0	0	0	0	0	0	8	0	0
57	13-09	0	0	0	6	0	0	6	0	0
58	13-10	0	0	0	0	0	0	7	0	0
59	14-02	0	0	0	0	7	0	0	0	0
60	14-03	0	0	0	0	0	0	7	0	0
61	14-04	0	0	0	0	8	0	6	0	0
62	14-05	0	0	0	0	0	0	7	0	0
63	14-06	0	0	0	0	0	0	8	0	0
64	14-07	0	0	0	0	0	0	8	0	0
65	14-08	0	0	0	0	8	0	7	0	0

Appendix III, Elevation & CISD Data

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
1	4292918.6334	321685.3290	-0.1952	1.580
2	4293101.2729	321969.0377	0.4415	0.188
3	4293550.6212	322286.2467	-0.1343	75.444
4	4293239.1247	321893.5040	0.0304	8.813
5	4292927.3211	321756.3884	-0.0841	2.041
6	4292912.5648	321915.0728	-0.1694	23.859
7	4292887.0865	321972.3121	0.0514	13.017
8	4292794.9516	321899.2279	-0.0650	13.577
9	4292670.9454	321838.2665	0.0919	3.123
10	4292718.1929	321927.9465	0.0022	6.026
11	4292708.1859	321971.7042	0.1392	300.982
12	4292617.2548	321893.4644	-0.0895	2.189
13	4293241.2899	321921.2695	-0.1856	15.751
14	4293549.9774	322303.8563	-0.1073	53.241
15	4293317.9583	322002.2087	0.4481	0.002
16	4293246.1809	321885.3319	0.1265	3.622
17	4293182.8627	321906.5295	-0.2846	8.252
18	4293041.5985	321659.9883	0.1104	0.651
19	4292860.2812	321767.5967	0.1767	0.224
20	4292911.7256	321789.5460	0.0307	11.792
21	4292667.0060	321830.9411	0.0846	6.990
22	4292741.6258	322016.5466	0.0957	12.192
23	4293537.9420	322222.7240	0.4911	0.537
24	4293521.0484	322261.7654	0.4601	1.115
25	4293421.3461	321976.6774	0.5123	0.080
26	4293208.6281	321898.2556	0.3699	3.922
27	4292853.6448	321784.4347	0.2845	0.422
28	4292905.9534	321994.3158	0.1781	3.226
29	4292892.2721	322024.3921	0.2654	2.669
30	4292597.3897	321863.0569	-0.0304	12.956
31	4292702.1901	322021.4490	-0.1885	4.760
32	4293039.4633	321641.7681	0.0653	0.112
33	4293575.1331	322203.3352	0.4634	0.401
34	4292862.5872	322031.9283	0.3491	0.066
35	4292883.8847	322040.5048	0.3630	0.099
36	4292599.3359	321847.2880	-0.5165	22.891
37	4293378.4723	321976.5042	0.3888	0.159
38	4292895.3173	322055.7217	0.3790	0.023
39	4293347.3681	322033.7269	0.4162	0.349
40	4293021.5481	321998.5718	0.3402	0.486
41	4293147.9123	321617.2473	0.2504	0.235
42	4293342.8804	321961.0161	0.3686	3.537
43	4293394.3567	321978.0320	0.4590	0.280
44	4293182.0619	321827.8558	0.2548	22.012
45	4293178.8076	321878.3085	0.3279	1.075

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
46	4293041.6845	321935.0947	0.3036	0.191
47	4292854.8498	322041.8812	0.3741	0.041
48	4293201.4572	321802.2331	0.4027	0.755
49	4292618.3947	321878.7663	-0.0442	2.867
50	4292904.9814	322028.3564	0.3971	1.523
51	4293183.6805	321844.4043	0.3887	0.460
52	4293051.6908	321985.6297	0.4589	0.000
53	4292813.0444	321988.0265	0.3692	0.196
54	4293291.5091	322005.1186	0.4985	0.034
55	4292729.3410	321815.2532	0.3941	0.000
56	4292915.1511	322103.8296	0.4535	0.000
57	4292938.7084	322012.3444	0.3902	0.849
58	4292793.8879	322092.4086	0.4441	0.227
59	4293059.5747	322003.3065	0.4649	0.000
60	4293030.4994	321981.1220	0.4243	0.000
61	4292959.0327	321817.7185	0.4403	0.023
62	4292886.0742	321866.0590	0.4270	2.320
63	4292945.5822	322005.5811	0.3965	0.235
64	4292869.6014	321927.8668	0.3963	2.915
65	4292775.8865	322055.3157	0.3805	1.578
66	4292749.6147	321992.3850	0.3946	0.105
67	4292923.1611	322090.0822	0.4135	0.121
68	4293584.6739	322137.3939	0.4913	0.011
69	4293405.2452	321893.6119	0.2936	1.807
70	4293029.8610	321960.3165	0.3931	0.080
71	4293064.1258	321864.4938	0.4447	0.461
72	4292963.7838	321798.9428	0.4180	0.000
73	4292874.6961	321877.5875	0.4983	0.398
74	4292792.7671	321959.1944	0.3104	0.236
75	4292649.2359	322045.2599	0.4424	0.024
76	4292724.2928	322006.7676	0.3739	0.247
77	4292766.7913	322046.9994	0.4017	2.915
78	4293279.5768	321824.0277	0.4864	0.064
79	4293057.5323	321879.4831	0.3967	0.041
80	4293041.8724	321815.8036	0.4404	0.163
81	4292969.5671	321775.2707	0.3925	0.096
82	4292806.6878	321957.3335	0.3466	0.338
83	4292784.4888	321864.9508	0.4056	0.000
84	4292748.9222	321979.0473	0.3833	0.000
85	4292713.3193	321787.1346	0.4871	0.101
86	4292658.7941	321880.7546	0.4403	0.704
87	4293158.7450	321942.5472	-0.2697	7.011
88	4293160.9003	321943.1689	-0.2527	5.494
89	4293174.3534	321934.0398	0.2721	0.797
90	4293561.7796	322334.6610	0.5133	0.447
91	4293551.9831	322268.2067	-0.0378	11.625
92	4293555.0867	322269.1201	0.0326	8.227
93	4293482.1422	322267.7462	0.4905	0.181
94	4293479.3693	322267.5876	0.4677	0.209

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
95	4293496.9614	322252.9689	0.5058	0.403
96	4293340.5582	322135.7839	0.4765	0.000
97	4293369.9140	321905.5902	0.0959	145.843
98	4293442.8976	321938.2979	-0.1258	3.262
99	4293442.4733	321940.6658	-0.1520	5.460
100	4293418.3219	321991.0053	0.5078	0.003
101	4293419.7645	321989.6722	0.5121	0.005
102	4293417.5879	321988.8024	0.4957	0.010
103	4293559.7125	322275.9861	0.4666	5.082
104	4293474.3624	321901.8653	0.5499	0.000
105	4293361.2006	321939.2346	0.3438	8.340
106	4292951.6442	321836.7523	0.4545	0.252
107	4293021.9671	321966.1241	0.4075	0.004
108	4293039.7275	321929.8867	0.3454	0.729
109	4293035.3616	321803.2759	0.3316	1.112
110	4292944.1638	322014.3773	0.3477	0.417
111	4292718.6148	321925.3996	-0.0943	3.714
112	4292274.7833	322037.1800	0.0908	0.000
113	4292615.9807	322025.6555	0.4299	0.000
114	4292661.6454	321892.2368	0.4206	0.006
115	4292665.7518	321837.9316	0.2403	0.000
116	4292947.8381	322082.4908	0.3182	0.617
117	4293260.7406	322053.8365	0.3560	0.546
118	4293248.7331	322037.2593	0.3667	3.290
119	4293571.1105	322203.0486	0.2014	1.219
120	4293568.1193	322204.3927	0.4596	2.320
121	4293562.6263	322209.1109	-0.0169	13.040
122	4293558.8040	322213.9404	0.4574	5.616
123	4293545.1915	322218.8262	0.4431	0.773
124	4293536.6972	322228.3615	0.5051	1.336
125	4293531.1920	322238.1601	0.2902	2.562
126	4293528.6409	322243.1154	0.0664	20.821
127	4293525.1858	322256.8408	0.4636	2.407
128	4293533.0828	322265.6342	0.4797	4.941
129	4293538.0274	322269.5424	0.1941	9.518
130	4293542.6854	322276.2937	-0.0484	21.132
131	4293550.6531	322294.9077	-0.0371	166.306
132	4293554.7564	322314.5785	0.0280	4.593
133	4293557.8607	322319.8048	0.3558	2.772
134	4293567.1922	322306.5262	0.1515	5.987
135	4293569.3858	322303.7266	0.3179	3.706
136	4293568.4834	322299.6804	0.4625	3.261
137	4293564.8958	322296.0386	0.1350	4.106
138	4293546.9828	322296.2806	0.0565	43.000
139	4293544.6742	322306.0310	-0.0149	21.159
140	4293539.2627	322317.9871	0.3841	4.917
141	4293564.2205	322271.8850	0.3054	2.852
142	4293564.5903	322267.7483	0.2381	2.529
143	4293561.8087	322260.2718	0.3844	3.839

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
144	4293573.7470	322174.6890	0.5333	0.000
145	4293575.4167	322155.7170	0.5748	0.003
146	4293575.6155	322155.8094	0.5691	0.003
147	4293453.3267	321887.8810	0.5522	0.000
148	4293414.0015	321888.3768	0.4295	0.578
149	4293405.2496	321898.6561	0.1539	10.864
150	4293405.2492	321898.6167	0.1889	10.864
151	4293382.1590	321904.8487	0.3223	1.083
152	4293371.7210	321913.8872	0.3246	1.468
153	4293368.0398	321912.5194	0.0640	4.173
154	4293364.4406	321939.6952	-0.1088	53.115
155	4293347.2772	321946.1215	0.0837	279.853
156	4293345.7690	321959.6766	0.3194	42.153
157	4293359.8556	321949.6708	0.0569	25.841
158	4293368.3326	321954.8934	0.3783	2.069
159	4293372.5482	321968.4053	0.3887	0.790
160	4293406.8824	321982.2185	0.4693	0.108
161	4293416.8464	321983.8715	0.5540	0.029
162	4293435.8436	321979.1371	0.5508	0.000
163	4293090.7303	321617.2670	-0.0098	3.324
164	4293138.6937	321621.0520	0.2517	3.334
165	4293043.2534	321654.5147	0.2380	1.773
166	4293051.9812	321655.8157	0.3612	3.660
167	4292941.5656	321705.9883	0.3041	3.270
168	4292939.4159	321699.4226	0.2926	1.680
169	4292939.4419	321691.7372	0.3219	0.849
170	4292934.3272	321693.6163	-0.0340	1.092
171	4292872.7860	321870.8621	0.3414	1.006
172	4292873.7714	321881.6753	0.4914	0.199
173	4292880.8616	321879.4185	0.4795	0.440
174	4292881.1227	321884.3168	0.4720	0.108
175	4292883.6925	321875.1767	0.4274	0.579
176	4292886.5910	321871.2464	0.4235	1.168
177	4292906.8005	321874.6685	0.4604	1.159
178	4292912.0536	321877.3220	0.4229	0.699
179	4292917.8919	321873.2179	0.2902	2.057
180	4292920.6680	321870.7611	0.0917	3.060
181	4292917.0481	321862.9866	0.0533	5.603
182	4292932.6924	321858.0133	0.2516	4.759
183	4292938.3215	321858.5615	0.4184	2.142
184	4292940.5634	321852.5509	0.3802	2.986
185	4292944.8260	321848.0116	0.4264	1.126
186	4292970.2682	321822.8889	0.4671	0.000
187	4292970.6771	321817.7933	0.5443	0.000
191	4292970.4881	321812.2816	0.5759	0.000
192	4292973.1532	321806.7378	0.4854	0.000
193	4292971.0824	321801.7955	0.4880	0.000
194	4292965.1584	321792.0333	0.5130	0.000
195	4292968.0265	321785.9490	0.4795	0.003

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
196	4292971.7049	321780.9362	0.4625	0.085
188	4292962.6517	321815.5172	0.5664	0.000
189	4292958.0447	321810.8114	0.5823	0.000
190	4292951.7439	321813.7530	0.4804	0.031
197	4292967.6413	321770.8898	0.4378	0.158
198	4292964.4615	321765.6621	0.4071	0.240
199	4292957.7574	321761.4779	0.3850	0.876
200	4292951.7508	321761.1213	0.1046	4.678
201	4292948.3233	321758.2197	-0.0311	3.120
202	4292940.0509	321753.3578	0.1292	2.104
203	4292908.1447	321807.8294	0.0306	8.734
204	4292904.2927	321818.7905	0.0439	9.675
205	4292897.4107	321828.9352	0.0677	10.542
206	4292896.4011	321832.7550	0.3177	7.632
207	4292896.9729	321839.4822	0.4596	4.992
208	4292891.5010	321843.6777	0.4590	6.447
209	4292883.3483	321851.8691	0.4249	9.007
210	4292780.9709	322094.8052	0.3767	1.321
211	4292786.7233	322091.4061	0.4726	0.630
212	4292791.4222	322092.0047	0.5129	0.332
213	4292791.5970	322078.6254	0.3758	0.688
214	4292786.9918	322066.8852	0.4265	0.968
215	4292781.6316	322061.6812	0.4270	1.259
216	4292770.5928	322052.2982	0.4829	2.345
217	4293102.4375	321965.9538	0.4396	0.076
218	4293112.7422	321964.1674	0.4656	0.000
219	4293072.6380	322001.2738	0.5129	0.071
220	4293065.3031	321998.1766	0.5139	0.000
221	4293052.4239	321999.2977	0.4281	0.000
222	4293053.0117	321991.7880	0.4800	0.000
223	4293057.1667	321986.8997	0.4632	0.000
224	4293050.9678	321977.4845	0.4567	0.031
225	4293046.5286	321972.8584	0.4485	0.016
226	4293039.4799	321971.5782	0.4856	0.011
227	4293035.5105	321974.8768	0.4431	0.001
228	4293027.5802	321982.4098	0.4198	0.004
229	4293023.6084	321979.6410	0.4435	0.004
230	4293024.9648	321974.8564	0.4403	0.003
231	4293027.6776	321971.3655	0.4423	0.003
232	4293028.6615	321966.5186	0.4323	0.015
233	4293024.0563	321960.9615	0.4305	0.069
234	4293019.9158	321970.6598	0.3888	0.003
235	4293018.6945	321977.0354	0.4468	0.013
236	4293016.6321	321984.2312	0.3972	0.028
237	4293012.7275	321990.4416	0.3453	0.147
238	4293008.4002	321996.9373	0.3378	0.856
239	4293021.0036	321990.5714	0.3454	0.044
240	4293020.9934	321992.6798	0.1885	0.227
241	4293030.8666	321958.1689	0.3914	0.107

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
242	4293032.6885	321951.8767	0.3298	0.236
243	4293036.4159	321945.8424	0.0993	0.413
244	4293039.0650	321940.0274	0.1478	0.325
245	4293039.3208	321937.3867	0.1790	0.298
246	4293036.2602	321925.6304	0.3585	1.136
247	4293037.9185	321921.2544	0.3792	1.653
248	4293044.5024	321917.9416	0.3710	3.173
249	4293048.1035	321915.6815	0.3104	4.165
250	4293049.1192	321910.1457	-0.6786	27.817
251	4293053.2036	321905.1271	0.3076	4.290
252	4293055.3951	321898.6294	0.3382	1.677
253	4293058.8427	321892.3397	0.3583	1.094
254	4293060.5958	321875.3972	0.4239	0.089
255	4293067.7819	321854.2690	0.1346	6.125
256	4293066.2978	321859.4223	0.4276	1.685
257	4293066.4332	321848.5995	0.2242	4.450
258	4293065.6872	321840.3669	0.4066	1.634
259	4293057.5755	321834.6024	0.3949	0.962
260	4293054.6415	321828.1775	0.4146	0.491
261	4293046.3021	321820.6402	0.4415	0.141
262	4293176.9485	321821.7311	-0.0824	9.470
263	4293181.4931	321820.1885	0.1755	5.189
264	4293192.2930	321815.7082	0.2536	3.238
265	4293198.1341	321811.1111	0.4120	0.784
266	4293207.8070	321801.6416	0.1439	0.729
267	4293213.7918	321803.0844	0.3912	0.632
268	4293219.9696	321808.1501	0.4360	0.556
269	4293226.2719	321807.6410	0.4013	1.498
270	4293233.5571	321807.2167	0.4077	2.139
271	4293261.9813	321818.4482	0.4385	0.517
272	4293274.5392	321827.4898	0.4588	0.134
273	4293277.5851	321841.6121	0.4508	0.229
274	4293271.4693	321849.9300	0.4443	0.511
275	4293264.4004	321858.5271	0.3001	1.848
276	4293263.2058	321866.2059	0.1061	9.256
277	4293255.3198	321875.5502	0.1078	16.765
278	4293247.4546	321883.7451	0.2838	4.762
279	4293237.6108	321913.9615	0.3678	3.735
280	4293238.2568	321918.2059	0.0233	6.742
281	4293231.6018	321930.8202	-0.0146	4.516
282	4293227.0346	321930.7285	0.2202	4.313
283	4293217.7878	321928.6891	0.3502	3.101
284	4293212.2650	321922.3116	0.1423	3.155
285	4293207.3293	321911.7634	-0.2895	12.750
286	4293205.8794	321900.9880	-0.0257	5.276
287	4293202.7362	321899.7035	-0.0838	12.817
288	4293195.6364	321901.1476	-0.0189	7.749
289	4293188.2879	321900.9242	0.3198	3.895
290	4293177.8282	321905.7798	0.0277	4.348

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
291	4293173.5422	321903.4557	0.1830	58.319
292	4293173.5821	321897.0114	-0.0081	8.117
293	4293176.1784	321889.2202	0.2955	1.813
294	4293177.0147	321883.9379	0.2536	1.359
295	4293178.2459	321866.8003	0.3604	0.411
296	4293174.2486	321857.1374	0.3773	0.527
297	4293177.9835	321848.4722	0.3651	0.529
298	4293187.5746	321837.6083	0.2685	0.576
299	4293340.0147	322052.0328	0.1449	3.978
300	4293339.1116	322044.3923	-0.0749	4.240
301	4293342.5481	322042.9470	0.2714	0.926
302	4293342.5067	322031.0299	0.4326	0.423
303	4293338.8499	322026.0746	0.3825	0.436
304	4293333.7998	322020.2030	0.4353	0.250
305	4293325.0711	322018.9212	0.4673	0.090
306	4293317.9684	322015.0656	0.4448	0.012
307	4293313.3616	322011.8528	0.4677	0.000
308	4293317.1178	322006.5375	0.5226	0.004
309	4293320.2165	321997.0660	0.4788	0.069
310	4293311.1682	321995.2689	0.5025	0.000
311	4293304.8174	321995.7593	0.4881	0.000
312	4293299.6397	321997.7795	0.4836	0.000
313	4293294.9282	321998.1347	0.5049	0.000
314	4293286.9867	321995.1827	0.5037	0.015
315	4293326.4957	322061.4504	-0.0816	1.063
316	4293311.6102	322067.8402	-0.3816	0.698
317	4292871.5298	321809.0819	0.3086	7.649
318	4292863.9965	321808.3807	0.0602	6.735
319	4292860.8630	321802.2840	0.1844	1.740
320	4292857.2347	321794.1500	0.2534	0.954
321	4292858.0142	321782.2079	0.2227	0.417
322	4292856.4133	321772.1239	0.2269	0.098
323	4292854.4384	321763.3009	0.2169	0.061
324	4292863.9840	321775.8303	0.2166	0.308
325	4292853.1139	321795.1464	0.2403	0.878
326	4292808.0139	321903.9756	-0.1248	0.000
327	4292782.9272	321893.2457	0.1331	2.812
328	4292780.8942	321885.6226	0.3655	0.000
329	4292777.0669	321880.4263	0.3957	0.000
330	4292778.2575	321873.2481	0.3693	0.000
331	4292777.7989	321865.9320	0.4187	0.000
332	4292784.7867	321861.4150	0.3807	0.000
333	4292856.8213	321915.0139	-0.0211	4.887
334	4292859.6564	321924.9649	0.1816	3.282
335	4292863.7586	321930.0993	0.3112	3.919
336	4292883.9430	321925.4324	0.3015	4.133
337	4292900.7965	321915.8201	0.2311	13.768
338	4292915.9916	321921.0473	0.1169	3.411
339	4292916.8734	321926.9178	0.3915	2.477

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
340	4292917.7149	321932.3024	0.3799	1.901
341	4292913.2400	321940.8314	0.2129	1.639
342	4292907.7672	321942.1279	0.3385	2.787
343	4292909.1638	321950.6915	0.3104	1.621
344	4292903.4936	321953.9652	0.3943	1.410
345	4292901.5673	321963.0488	0.3079	1.677
346	4292890.8591	321982.8323	0.0347	9.430
347	4292902.3717	321993.9742	0.1619	4.554
348	4292910.7061	321997.9467	0.2407	13.252
349	4292912.8690	321998.4916	0.0931	4.101
350	4292918.3621	322001.8294	0.2909	2.484
351	4292925.4760	322006.8543	0.2618	2.917
352	4292927.3720	322008.9521	0.1909	2.098
353	4292934.8525	322009.8124	0.3803	1.011
354	4292943.1891	322016.4899	0.2708	0.637
355	4292948.9739	322011.7771	0.3924	0.147
356	4292935.8940	322019.2572	0.3117	2.421
357	4292931.5546	322024.8913	0.2109	4.935
358	4292927.1205	322026.1307	0.1441	7.332
359	4292920.6487	322028.6939	0.2570	9.987
360	4292915.2770	322028.3085	0.3601	2.853
361	4292902.5899	322024.0673	0.2732	2.159
362	4292900.9644	322022.1855	0.0807	3.495
363	4292892.2771	322018.5888	0.3485	1.185
364	4292890.6772	322029.4791	0.2922	1.181
365	4292891.6271	322036.1948	0.2838	0.689
366	4292891.4939	322039.8271	0.2799	0.333
367	4292885.1396	322043.2626	0.4034	0.071
368	4292879.2448	322036.8399	0.4035	0.097
369	4292875.6418	322041.2298	0.4012	0.043
370	4292877.5528	322047.4719	0.3988	0.035
371	4292889.0598	322052.8912	0.3891	0.030
372	4292874.8289	322034.2287	0.3601	0.067
373	4292867.2533	322032.3002	0.3134	0.026
374	4292856.8440	322033.4222	0.3461	0.121
375	4292852.9058	322034.2712	0.3793	0.163
376	4292851.5602	322038.4007	0.3132	0.076
377	4292858.7813	322042.6415	0.4384	0.035
378	4292850.4637	322030.2561	0.3145	0.628
379	4292846.8677	322028.9773	0.3474	2.320
380	4292842.2993	322023.6293	0.3432	2.429
381	4292834.0668	322019.5150	0.3488	0.559
382	4292829.4946	322012.0602	0.3689	0.379
383	4292825.8661	322004.9199	0.3220	0.111
384	4292820.2018	321999.3065	0.3573	0.020
385	4292820.1687	321999.3475	0.3547	0.020
386	4292815.9611	321992.0303	0.3795	0.151
387	4292816.6879	321980.5266	0.2632	0.542
388	4292815.2595	321975.1654	-0.0541	18.248

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
389	4292811.2005	321969.4828	0.2749	1.717
390	4292806.5720	321966.2856	0.2613	1.252
391	4292806.1685	321960.6808	0.3386	0.418
392	4292799.6864	321956.2798	0.3758	0.159
393	4292795.9379	321954.3841	0.3746	0.138
394	4292791.0557	321953.4401	0.4030	0.070
395	4292792.7960	321948.5062	0.3732	0.041
396	4292794.0528	321940.2628	0.3581	0.000
397	4292799.7213	321937.1402	0.2521	2.901
398	4292806.6152	321934.7203	0.3358	2.987
399	4292817.6080	321933.2201	0.3246	0.820
400	4292825.0414	321930.6938	-0.1874	4.683
401	4292826.9571	321928.4219	-0.1246	1.447
402	4292829.2012	321924.9377	0.3472	1.025
403	4292833.6350	321917.2466	0.3868	4.021
404	4292839.4782	321910.6600	0.1996	6.252
405	4292844.6966	321911.7018	0.2464	3.978
406	4292748.4719	321875.2775	0.2218	3.268
407	4292753.1653	321871.9154	0.3936	0.000
408	4292750.2738	321864.6784	0.3747	0.000
409	4292748.1721	321855.3049	0.4355	0.000
410	4292744.6583	321847.7656	0.4609	0.000
411	4292739.9722	321841.7301	0.4568	0.000
412	4292738.5877	321834.5341	0.4638	0.000
413	4292731.5442	321825.6608	0.4778	0.000
414	4292733.4035	321816.7893	0.4209	0.000
415	4292728.8652	321807.9134	0.4460	0.000
416	4292722.5597	321802.0997	0.4823	0.000
417	4292717.2456	321798.5252	0.4572	0.013
418	4292714.4348	321792.6271	0.4817	0.052
419	4292700.3167	321786.8535	0.4566	0.396
420	4292695.0226	321781.2571	0.3294	0.883
421	4292691.1296	321781.7168	-0.1612	9.056
422	4292688.0657	321778.5890	0.3871	3.679
423	4292675.4112	321773.9169	0.4335	0.252
424	4292681.8292	321768.4533	0.4132	0.228
425	4292680.7110	321784.0175	0.4416	0.501
426	4292677.8769	321789.6491	0.4835	3.550
427	4292681.2341	321792.6045	0.4522	4.553
428	4292735.5061	322039.7277	0.2852	30.592
429	4292677.3557	321798.4720	0.1805	12.776
430	4292743.6199	322030.4358	0.0443	256.387
431	4292666.2902	321792.4018	0.3417	7.169
432	4292744.9612	322020.3174	0.0068	84.211
433	4292663.7816	321797.1439	0.2040	13.169
434	4292747.2192	322004.4849	0.2605	1.206
435	4292747.2816	321999.5486	0.3835	0.465
436	4292750.4558	321986.2131	0.4433	0.000
437	4292742.3811	321999.1403	0.3060	0.475

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
438	4292736.1774	322003.7852	0.3558	0.271
439	4292729.0577	322003.2554	0.3983	0.320
440	4292713.8624	322008.5462	0.4302	0.440
441	4292710.5603	322011.2865	0.3655	0.903
442	4292706.8500	322014.5860	0.0862	1.541
443	4292728.6955	321997.7865	0.4279	0.116
444	4292724.2583	321991.7079	0.4188	0.042
445	4292720.4104	321987.4588	0.4005	0.831
446	4292714.0445	321983.0148	0.2845	1.379
447	4292712.7051	321978.4316	0.1219	2.519
448	4292705.4506	321982.6277	0.2918	4.094
449	4292701.1460	321985.9997	0.1103	13.529
450	4292696.5436	321989.1391	0.4017	3.374
451	4292692.3167	321996.2358	0.4570	2.255
452	4292699.8203	322000.2055	0.0812	2.941
453	4292704.4378	322003.8288	0.3658	1.068
454	4292704.1311	322009.3687	0.2631	1.500
455	4292710.1487	322018.5188	0.0376	1.433
456	4292717.5188	322023.6941	0.3888	1.446
457	4292723.1917	322028.5600	0.3942	2.264
458	4292727.5186	322032.4567	0.3779	2.833
459	4292701.3593	322038.1294	-0.0193	23.714
460	4292700.5092	322039.3433	0.2163	23.714
461	4292700.0596	322042.6015	0.2862	10.544
462	4292696.2630	322043.3234	-0.0146	5.528
463	4292692.2154	322044.2601	-0.0750	7.377
464	4292687.6740	322044.2323	-0.0604	75.521
465	4292683.4386	322042.6036	0.1444	6.644
466	4292677.3100	322039.9410	0.3679	2.303
467	4292670.4764	322044.0215	0.3770	1.687
468	4292664.8627	322046.7303	0.3895	0.887
469	4292657.2746	322045.6278	0.4304	0.245
470	4292645.0309	322042.9947	0.4147	0.041
471	4292638.6734	322039.2145	0.4139	0.051
472	4292630.1645	322034.8512	0.4143	0.026
473	4292626.5917	322033.0104	0.3841	0.010
474	4292620.5344	322028.3106	0.3870	0.004
475	4292645.0024	321899.4447	0.4488	7.011
476	4292649.1231	321894.0215	0.4641	3.881
477	4292655.9373	321889.6804	0.4460	1.479
478	4292611.6274	321899.6051	0.3197	6.629
479	4292615.4107	321856.8605	0.3028	1.748
480	4292623.8371	321863.8753	-0.0115	4.611
481	4292628.2047	321883.5115	-0.1779	10.231
482	4292633.9721	321893.3068	-0.7395	40.845
483	4292634.9974	321893.8838	-0.7257	266.644
484	4293048.7281	321724.7972	0.3021	1.946
485	4293041.1886	321731.6541	0.3349	1.030
486	4293037.7101	321735.5004	0.3184	0.846

ID	NORTHING WGS84, EASTING WGS84,		ELEVATION m	CISD
	UTM 18N	UTM 18N		
487	4293032.2232	321746.1993	0.2681	0.451
488	4293022.5703	321750.2975	0.2422	0.513
489	4293017.9359	321756.2670	0.3438	0.294
490	4293006.0882	321772.9262	0.3825	0.645
491	4293346.9309	321941.5521	0.3826	4.118
492	4293007.3841	321782.9046	0.3670	1.255
493	4293005.8757	321786.9485	0.0483	5.743
494	4293006.5204	321792.1814	0.3240	1.949
495	4293013.6789	321793.6999	0.3727	2.398
496	4293017.9494	321796.8486	0.3915	3.625
497	4293016.9261	321802.6889	0.3805	0.919
498	4293021.1074	321805.9927	0.4275	0.557
499	4293027.6856	321811.8655	0.4824	0.285
500	4293032.5797	321812.1863	0.4617	0.284
501	4293028.6695	321800.8231	0.3466	3.350
502	4293028.0750	321798.0732	0.3352	8.709
503	4292661.6185	321830.3745	0.1863	6.900
504	4292662.0232	321822.4513	0.0996	13.025
505	4292920.6864	322128.3562	0.6298	0.000
506	4292919.3918	322122.2601	0.4759	0.000
507	4292917.5778	322113.4734	0.4539	0.000
508	4292921.1566	322102.0435	0.3877	0.024
509	4292929.1183	322090.5126	0.4057	0.216
510	4292935.6876	322085.5302	0.3366	1.122
511	4292941.7415	322084.2346	0.1091	1.805
512	4293277.4523	322076.1928	0.1001	0.438
513	4293256.0651	322044.8252	0.2591	1.418
514	4293240.0797	322034.9501	0.4301	0.496
515	4293235.3811	322033.7597	0.4535	0.284
516	4293250.1974	322040.7929	-0.1299	7.620
517	4293259.7421	322061.0266	0.2753	0.719
518	4293016.0908	321801.6531	0.4256	0.919

Appendix IV, Hog Island Gut Elevations

ID	NORTHING WGS84, UTM 18N	EASTING WGS84, UTM 18N	ELEVATION (m)	DESCR
519	4293039.5280	321719.1679	-1.1419	ch1
520	4293013.0098	321716.8349	-0.4310	ch2a
521	4293013.5556	321716.0716	-0.8361	ch2b
522	4293014.1073	321715.2817	-0.8751	ch2c
523	4293014.7306	321714.4203	-0.9451	ch2d
524	4293015.3718	321713.4946	-1.1703	ch2de
525	4293016.1870	321712.1932	-1.3065	ch2df
526	4293016.9613	321711.1517	-1.0320	ch2g
527	4293017.6350	321710.2049	-0.5619	ch2h
528	4292981.7081	321702.8482	-0.2471	ch3a
529	4292982.4595	321701.0571	-0.5412	ch3b
530	4292983.2917	321698.7418	-0.5835	ch3c
531	4292983.6248	321695.9623	-0.6519	ch3d
532	4292983.9343	321693.5883	-0.8304	ch3e
533	4292983.6427	321691.8409	-0.9154	ch3f
534	4292984.2384	321690.3034	-0.7268	ch3g
535	4292984.5151	321688.7203	-0.3092	ch3h
536	4292934.9513	321734.1961	-0.3678	ch4a
537	4292933.3314	321733.2067	-0.6476	ch4b
538	4292931.7230	321732.2198	-0.9398	ch4c
539	4292930.3936	321730.9311	-1.0105	ch4d
540	4292928.9608	321729.3271	-0.9001	ch4e
541	4292927.3983	321727.7968	-0.7089	ch4f
542	4292926.4942	321726.6908	-0.3131	ch4g
543	4292911.4746	321786.7650	0.0436	ch5a
544	4292909.1496	321786.8441	-0.1688	ch5b
545	4292906.5527	321786.7903	-0.4219	ch5c
546	4292904.2071	321786.7871	-0.3962	ch5d
547	4292901.9557	321786.5859	-0.4035	ch5e
548	4292899.3592	321786.4360	-0.4890	ch5f
549	4292897.1099	321786.0563	-0.6422	ch5g
550	4292895.4457	321785.9106	-1.2474	ch5h
551	4292893.9585	321785.5506	-1.5046	ch5i
552	4292892.2560	321785.3398	-1.0849	ch5j
553	4292891.3711	321785.0633	-0.6837	ch5k
554	4292889.8540	321784.6892	-0.3208	ch5l
555	4292888.2322	321784.2189	0.0359	ch5m
556	4292878.8659	321849.6699	-0.0002	ch6a
557	4292876.8990	321847.9320	-0.4331	ch6b
558	4292874.9536	321846.0788	-0.6619	ch6c
559	4292873.5221	321844.7354	-0.7189	ch6d
560	4292872.2414	321843.5244	-0.8531	ch6e
561	4292871.0598	321842.6794	-1.0949	ch6f

ID	NORTHING WGS84, UTM 18N	EASTING WGS84, UTM 18N	ELEVATION (m)	DESCR
562	4292869.6617	321841.2814	-0.9755	ch6g
563	4292868.1209	321840.1794	-0.8058	ch6h
564	4292866.8266	321838.7572	-0.4662	ch6i
565	4292865.3424	321837.5143	-0.1845	ch6j
566	4292863.1614	321836.1759	-0.0477	ch6k
567	4292861.0832	321835.0566	0.0181	ch6l
568	4292858.8089	321833.8568	0.0200	ch6m
569	4292856.2595	321832.1688	0.2225	ch6n
570	4292832.2595	321914.7054	0.0796	ch7a
571	4292830.6082	321912.8262	-0.5808	ch7b
572	4292829.1981	321911.3286	-0.7361	ch7c
573	4292827.8349	321909.8848	-0.7493	ch7d
574	4292826.4504	321908.6286	-0.7970	ch7e
575	4292824.7424	321906.8833	-0.8800	ch7f
576	4292823.5261	321905.8095	-1.1502	ch7g
577	4292822.3135	321904.6015	-1.2841	ch7h
578	4292820.5869	321902.8636	-1.0741	ch7i
579	4292819.1405	321901.3074	-0.6277	ch7j
580	4292817.3819	321899.9862	-0.3308	ch7k
581	4292816.0487	321898.9302	-0.2664	ch7l
582	4292813.7375	321897.2851	-0.1415	ch7m
583	4292811.4311	321895.0171	-0.1638	ch7n
584	4292809.0414	321892.9792	-0.1348	ch7o
585	4292806.7381	321890.6620	-0.0581	ch7p
586	4292804.5121	321888.9428	0.1607	ch7q
587	4292712.7567	321892.6713	-0.0776	ch8a
588	4292714.7775	321891.0531	-0.4564	ch8b
589	4292716.4014	321890.1441	-0.7308	ch8c
590	4292717.7924	321889.0532	-0.8585	ch8d
591	4292718.6684	321888.1166	-1.2063	ch8e
592	4292719.6817	321887.5660	-1.7109	ch8f
593	4292720.5350	321887.1056	-1.8062	ch8g
594	4292721.4583	321886.4081	-1.7764	ch8h
595	4292722.4174	321885.6251	-1.5833	ch8i
596	4292723.2045	321884.9676	-1.0211	ch8j
597	4292724.4098	321883.3064	-0.5233	ch8k
598	4292725.7456	321882.6629	-0.2776	ch8l
599	4292727.2139	321882.2007	-0.0943	ch8m
600	4292729.2117	321880.6282	-0.1628	ch8n
601	4292731.7692	321879.2307	-0.3326	ch8o
602	4292733.9361	321877.8015	-0.4465	ch8p
603	4292735.7175	321876.8934	-0.5690	ch8q
604	4292738.3234	321875.4771	-0.5549	ch8r
605	4292740.4633	321874.1553	-0.4555	ch8s
606	4292743.2350	321872.4291	0.0427	ch8t
607	4292665.0828	321820.5915	-0.0758	ch9a
608	4292665.4337	321819.3108	-0.4954	ch9b

ID	NORTHING WGS84, UTM 18N	EASTING WGS84, UTM 18N	ELEVATION (m)	DESCR
609	4292665.4144	321818.6568	-1.0489	ch9c
610	4292665.5397	321817.9616	-1.3672	ch9d
611	4292665.6376	321816.9408	-1.3099	ch9e
612	4292665.8500	321815.3032	-1.3396	ch9f
613	4292666.3326	321813.7250	-1.1761	ch9g
614	4292667.0237	321811.6668	-1.1050	ch9h
615	4292667.3399	321809.5557	-1.0752	ch9i
616	4292667.8576	321807.2757	-0.9541	ch9j
617	4292668.4646	321805.2970	-0.9002	ch9k
618	4292669.0219	321803.1234	-0.7323	ch9l
619	4292669.1258	321801.2413	-0.7035	ch9m
620	4292669.3191	321799.0361	-0.4382	ch9n
621	4292669.8466	321797.4137	0.0923	ch9o
622	4292612.2003	321903.6883	0.2340	ch10a
623	4292609.6054	321904.4976	0.0282	ch10b
624	4292606.6478	321906.0294	-0.1720	ch10c
625	4292604.3803	321907.0877	-0.3570	ch10d
626	4292602.4754	321908.0278	-0.5021	ch10e
627	4292600.5194	321909.1843	-0.8077	ch10f
628	4292598.9413	321909.7308	-1.2304	ch10g
629	4292597.1643	321910.5836	-1.7346	ch10h

Appendix V, Channel Cross-Sections

Sampling Station	X-Sectional area	Predicted Class*	Actual Class	Easting ** WGS84 UTM Zone 18	Northing** WGS84 UTM Zone 18
Acs1	0.221	na	2	322227	4293554
cG	0.090	na	1	321978	4293341
csA	7.263	na	4	322282	4293541
csB	0.193	na	2	322178	4293540
csC	2.914	na	3	321952	4293406
csD	1.720	na	3	321939	4293361
csE	0.725	na	2	321959	4293347
csF	0.163	na	2	321964	4293347
csH	3.115	na	3	322038	4292696
csJ	0.593	na	2	322097	4292835
csK	0.056	na	1	322043	4292989
csL	1.843	na	3	322031	4293032
csM	0.558	na	2	321917	4293390
s01	0.046	1	2	321819	4293066
s02	6.857	4	4	321825	4292636
s03	0.012	1	1	321942	4292648
s04	3.025	3	4	321938	4292842
s05	0.225	2	2	321957	4292848
s06	1.092	3	3	321991	4292870
s07	0.301	2	2	321972	4292815
s08	0.612	2	2	321950	4292827
s09	0.103	1	2	321847	4292936
s10	0.760	3	2	321853	4292901
s11	0.428	2	2	321773	4292884
s12	0.821	2	2	321692	4292931
s13	0.400	2	2	321619	4293122
s14	0.216	2	2	321623	4293137
s15	0.669	2	2	321871	4293103
s16	0.212	2	2	321801	4293099
s17	0.046	1	1	321902	4292678
s18	0.037	1	1	321854	4293050
s19	1.131	3	2	321909	4293057
s20	0.150	2	2	321888	4293087
s21	8.287	4	4	321911	4293161
s22	0.773	2	4	321863	4293164
s23	0.034	1	1	missing	missing
s24	0.600	2	2	321877	4293229
s25	2.602	3	3	321836	4293153
s26	0.245	2	2	321836	4293258
s27	0.170	1	2	321847	4293293

*na sites were used for calibrating visual interpretation of channel sizes in imagery

** limited accuracy – taken with Garmin Etrex Venture, not survey equipment will generally be within 3 to 5 m of the surveyed location

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