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Spatial Patterns of Herbaceous and Woody Recruitment in a Recently Restored Mixed Tidal Regime Freshwater Wetland

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Spatial Patterns of Herbaceous and Woody Vegetative Recruitment in a Recently Restored Mixed Tidal Regime Freshwater Wetland

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science at Virginia Commonwealth University.

by

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SPATIAL PATTERNS OF HERBACEOUS AND WOODY RECRUITMENT IN A RECENTLY RESTORED MIXED TIDAL REGIME FRESHWATER WETLAND

By James B. Deemy, B.S.

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science at Virginia Commonwealth University.

Virginia Commonwealth University, 2012

Director: Dr. Edward R. Crawford, Assistant Professor, Department of Biology & Center for Environmental Studies

Abstract

Ecological restoration of a converted wetland was characterized within a recently drained impoundment along the James River in Charles City County, Virginia. Colonizing vegetation was assessed over three growing seasons in both tidal and non-tidal environments. Study objectives were to (1) examine geospatial relations of recruitment patterns among colonizing species over three growing seasons, (2) quantify species composition and potential differences between extant species cover and soil seed banks across restored and natural wetland habitats and (3) assess geospatial patterns to develop a GIS model of bald cypress (*Taxodium distichum* L.) recruitment. The two most common native colonizing species during 2009, 2010 and 2011 growing seasons were narrow-leaf cattail (*Typha angustifolia* L.) and rice cutgrass (*Leersia oryzoides* L.). Vegetative communities dominated by these two species covered 72% of the basin in each growing season. Differences were observed between extant species cover in the field and seed bank species across habitats. Two hundred and eighty *T. distichum* individuals have been

located in wetland habitats at the VCU Rice Center. Using a GIS weighted suitability model we identified potential areas within the restored wetland for natural and facilitated bald cypress recruitment. At the VCU Rice Center ~9.7 ha have potential for natural regeneration and ~48.5 ha have potential for facilitated restoration of *T. distichum*.

Introduction

The United States has lost more than fifty percent of its wetland coverage; rapidly increasing urban and suburban development continues to threaten remaining wetland areas (Mitsch et al. 2009). Bottomland hardwood forests were once abundant wetland ecosystems in the southeastern United States but now cover a small fraction of the area that they once inhabited (Mitsch and Gosselink 2007). These ecosystems are one of the more complex wetland ecosystem types and contain a large portion of the biodiversity of a given region (Mitsch et al. 2009).

Ecological restoration is defined as restoring anthropogenically impacted ecosystems to a more natural condition (NRC 1992). Wetland restoration may be a viable method for recovering wetland structure and function lost from anthropogenic degradation and destruction (Zedler 2000, Mitsch and Gosselink 2007). Most wetland restoration efforts have focused on marshes because of shorter establishment times and lower complexity (Zedler 2000, Mitsch and Gosselink 2007, Mitsch et al. 2009). However expected restoration outcomes are difficult or impossible to predict (Zelder and Callaway 1999). Less is known about restoration of forested wetlands because of longer establishment time and the complexity of these ecosystems compared to marshes (Crawford et al. 2007, Mitsch and Gosselink 2007, Mitsch et al. 2009). Most forested wetland restoration studies have focused on the Mississippi Valley bottomland hardwood forests and less is known about swamp restoration in the Mid-Atlantic and Southeast regions (Mitsch and Gosselink 2007). Forested wetland restoration is important in the Southeastern United States because these systems have been lost in large proportions compared to historical distributions (Mitsch and Gosselink 2007, Faulkner et al. 2009). These systems can regenerate naturally through wetland forest succession where they have been degraded and/or destroyed but succession is sometimes limited because of changes in hydrology or soils associated with loss of the forest (Mitsch and Gosselink 2007).

Background

Ecological succession can be generally described as the change or replacement in biological communities after a disturbance event through time toward a climax or self-promoting community (Molles 2005). Two models of succession are often used to explain successional patterns: autogenic succession and allogeneic succession (Glenn-Lewis and van der Maarel 1992). Autogenic succession is used to describe succession under biotically dominated conditions (Glenn-Lewis and van der Maarel 1992). The autogenic theory involves three basal concepts: vegetation occurs in distinct or recognizable communities, biota drives community change through time, and these changes are linear and move towards a stable climax ecosystem (Mitsch and Gosselink 2007). Allogenic succession describes succession under environmentally or abiotically dominated conditions (Glenn-Lewis and van der Maarel 1992). This theory of succession does not involve vegetative communities but a process of continual invasion and replacement of species with no particular direction or stable climax. Under this theory of succession varying responses of species assemblages to environmental cues drive succession (Mitsch and Gosselink 2007).

Community patterns are generally indicative of both abiotic and biotic influences and at times may follow one model or the other but are unlikely to follow one through succession completely (Glenn-Lewis and van der Maarel 1992). Both models of succession have been applied to wetland plant communities (Mitsch and Gosselink 2007). Anthropogenic disturbance events and even natural disturbance events can be significant enough to cause the re-initiation of secondary or even primary succession (Schrift et al. 2008). Plant community succession can be difficult to predict due to the stochasitic nature of plant recruitment to denuded or bare landscapes (Del Moral and Wood 1993). Natural and unnatural ecosystem development is the product of biological and physical conditions acting upon an area (Sklar et al. 1985).

Ecological succession beginss on bare substrates and can be classified into two categories based upon the nature of the substrate. Primary succession occurs on newly formed or raw substrate (Molles 2005). Such substrates have no history of biological modification (Glenn-Lewis and van der Maarel 1992). Examples of substrates in primary succession are newly formed dunes, elevating seashores, glacial forefields, granite outcrops and volcanic deposits (Glenn-Lewis and van der Maarel 1992). Secondary succession occurs when biologically impacted soils remain after a disturbance event destroys the above ground biotic components (Glenn-Lewis and van der Maarel 1992). Grazing, fire, storm damage, and flood damage are a few examples of processes that can all instigate secondary succession (Molles 2005).

Primary succession in wetlands occurs when wetland ecosystems develop where there has previously been no macrophyte coverage; examples of wetland primary succession would be exposed deltatic sediment deposits, river depositional sand bars, and sandbars in lagoons formed from sediments deposited during hurricanes (Batzer and Sharitz 2006). Secondary succession in wetlands occurs where there has previously been plant coverage and a disturbance of some kind has removed it; examples of such disturbances are fires, hurricanes and major sediment deposit during storm events (Batzer and Sharitz 2006).

Succession of created wetlands has been divided into two temporally different categories or phases: the “Arrival and Establishment Phase” and the “Autogenic Dominance” phase (Noon 1996). Arrival and Establishment is based on stochastic or chance events that bring aquatic macrophytes to an area and the physicochemical conditions that either keep them from establishing or allow them to become established. Successional stage may be more of a determinant in vegetation establishment than proximity to potential colonization sources (Deberry and Perry 2004).

Development of wetland plant communities is influenced by the abiotic and biotic conditions under which they become established and subsequent events of colonization (Batzer and Sharitz 2006). These communities change through time based upon further abiotic and biotic conditions acting upon them (Batzer and Sharitz 2006). Biotic conditions that influence succession can include both intra-species and interspecies competition. Biotic interactions can also include species' influence directly through competition/predation or indirectly by altering shared physical environment (Hastings et al. 2007). External factors such as meteorological disturbances and climate shifts can also serve to facilitate plant community successional dynamics (Batzer and Sharitz 2006). Tree species, particularly bald cypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*), two model wetland tree species of the Mid-Atlantic and Southeastern wetlands, often need a severe disturbance to allow them to be competitive in establishment when faced with heavy herbaceous cover, as their seedlings are inferior competitors with herbaceous plants (Dunn and Sharitz 1987). The early colonizing species of a wetland can often have very minor effects on later species or resist succession altogether (Connell and Slatyer 1977).

External abiotic drivers of succession seem to have the most effect on wetland plant community changes, with the greatest of these factors being those that affect hydrology (van der Valk 1981, Batzer and Sharitz 2006). Wetlands by their very nature and definition are dominated by hydrologic factors, thus the plant communities are going to reflect spatial and temporal differences in surficial and soil/sediment hydrologic conditions (Verhoeven and Sorrell 2010). Edaphic characteristics often shape community structure (Tilman 1988) and hydrologic regime can have a great effect on these characteristics. Water table fluctuations served as a driving component of wetland plant communities when considered as part of a complex system of hydrologic factors (Yu and Ehrenfeld 2010). The spatial and temporal variations in hydrologic conditions of a wetland control both where and when species become established (van der Valk 1992), thus affecting succession and community dynamics through time. Factors that specifically shape marsh plant communities are salinity, time of inundation, sulfide concentrations and substrate composition (Odum 1988). When considering tidal freshwater marshes that are part of an estuarine gradient, salinity is the most important factor controlling species composition and consequently species richness (Odum 1988).

Wetland Restoration

Baseline vegetative assessments can be vital to efficient and successful wetland restoration (Zedler 2000). Forested wetlands need approximately one hundred years to develop naturally (Mitsch and Gosselink 2007). Forested wetland restoration efforts seek to accelerate the natural process of wetland succession by planting woody species. Data on the standing cover and potentially colonizing species can increase the efficiency and success of these efforts by locating areas where seedlings and saplings will have the least competition.

The restoration of an anthropogenically impacted wetland to a forested mixed tidal regime freshwater wetland at the VCU Rice Center offers a unique opportunity to study the natural and anthropogenic influence on the restoration of a highly complex ecosystem. Before this restoration can take place it is necessary to gather information about the current state of plant community succession in this newly formed wetland ecosystem. This is also an opportunity to study the role of plant community succession before and during anthropogenic restoration of a complex wetland ecosystem. This study will use interdisciplinary methods to address environmentally and ecologically important questions of succession and vegetative community dynamics in a recently restored mixed tidal regime freshwater wetland.

Wetland Monitoring

The literature shows that geographic information systems (GIS) and remote sensing techniques have been used to study wetland ecosystems across multiple spatial scales (e.g. Best et al. 1981, Hardisky et al. 1986, Kauffman-Axelrod and Steinberg 2010, Klemas 2001). Multiple invasive species in the Everglades have been geospatially assessed utilizing aerial photography (McCormick 1999). Satellite Landsat Thematic Mapper and satellite based radar were used to track the impact of, and subsequent recovery from Hurricane Katrina on forested wetlands at the Louisiana-Mississippi border (Ramsey et al. 2009). Light Detection and Ranging or LiDAR is commonly used for elevation mapping and has been used for mapping of inundated areas at the landscape level (Lang and McCarty 2009). LIDAR is also commonly used for the creation of digital elevation models, a tool often used in geologic studies. When combined with GIS tools DEMs have great potential for use in studies of an ecological nature. GIS can be used to analyze the wetland elevation changes and microtopography that are crucial to wetland hydrology which drives the community structure (Verhoeven and Sorrell 2010).

Despite the wealth of literature on both primary and secondary ecological succession, there are still a great number of questions to be addressed (Tilman 1988). The questions in this study address spatial aspects of succession across a short temporal scale (two to three growing seasons) and as such could provide insight about the early stages of colonization and succession in reservoirs that were products of wetland flooding. This study will integrate field intensive methods and laboratory research with emerging environmental technologies. Study objectives of this project address geospatial patterns of plant community development and succession and invasive species dynamics in a restoration setting for a developing forested wetland. Results from this project may be used to build upon the foundation of wetlands ecology and restoration in the Mid-Atlantic region.

Site Description

This study was conducted in a mixed tidal regime freshwater riparian wetland associated with Kimages Creek and neighboring reference swamp (mixed tidal regime freshwater forested wetland associated with an unnamed tributary we refer to as Harris Creek) (Figure 1). The wetlands examined in this study were located on the Virginia Commonwealth University Walter and Inger Rice Center for Environmental Life Sciences' property. The VCU Rice Center is located in Charles City County, Virginia along the James River. Both wetlands have tidal communication with the James River. Kimages Creek wetland is currently in a marsh dominated early stage of wetland succession but woody recruitment is occurring along the wetland/upland ecotone as well as encroaching within the marsh interior. There is woody recruitment in non-tidal and tidal areas of this wetland. Reestablishment of Kimages Creek's historical stream channel was completed in December of 2010 with the removal of approximately ~100 m of the impoundment (dam) and a spillway.

Kimages Creek was logged during the Civil War (Egghart 2009) and again in 1927. The 1927 logging and subsequent impoundment were for the purpose of creating a recreational lake.

Between the time of impoundment and 2006 the reservoir was known as Lake Charles. During the fall of 2006 an un-named tropical storm caused a breach in the impoundment and draw-down of about 1 m occurred. After 16 months of further dam erosion a channel reconnected to the James River. Vegetative studies began in the draw down portion of the upper basin in spring 2007. Community delineations began in August of 2009 as a follow up study on the initial baseline surveys. Harris Creek was used as a "reference site" and "benchmark" for the seed bank and the *T. distichum* restoration model studies. Community delineations and transect cover studies were not performed within Harris Creek. This forested wetland also serves as a reference site for Kimages Creek forested wetland restoration goals. Harris Creek is comparable to Kimages Creek in many aspects, except the watershed area for Harris Creek is smaller than that of Kimages Creek. The watersheds for the two wetlands share a boundary on Rice Center property.

Study Objectives

There were three primary objectives stemming from the overarching goal of restoring a newly created mixed tidal regime freshwater wetland on the VCU Rice Center property to its historical condition as a forested wetland. The first objective of this study was to map and spatially assess colonizing herbaceous vegetative communities in the newly formed freshwater wetland over several growing seasons. The second objective of this study was to compare the extant vegetative communities with potential colonizing species found in the soil seed banks. The third objective of this study was to locate and develop a model to identify areas with a high potential for *T. distichum* restoration by analyzing the age structure and spatial distribution of the current local population. Elucidation of these objectives involved the use of integrative field and laboratory methods so currently and potentially colonizing vegetation could be assessed at the greatest detail in the time available. The following sections represented as a series of chapters describe three studies conducted toward the goals of baseline data collection prior to restoration of a freshwater mixed tidal regime forested wetland in Mid-Atlantic region.

Chapter 1

Vegetative Community Dynamics with Special Regard to Invasive Species in a Newly Restored Freshwater Wetland

Abstract

The ecological restoration of a prior converted wetland was characterized within a recently drained impoundment along the James River in Charles City County, Virginia. We quantified the recruitment and colonization of native and non-native wetland vegetation within a former impoundment using global positioning system and geographic information system technology. Colonizing vegetation was assessed over three growing seasons in both tidal and non-tidal environments. Standing herbaceous cover was assessed with GPS community delineations and line intercept transects. Fifty nine species were identified in Kimages Creek wetland. The two most common native colonizing species during the study were narrow-leaf cattail (*Typha angustifolia* L.) which covered ~9 ha in each growing season and rice cutgrass (*Leersia oryzoides* L.) which covered ~5 ha in each growing season. The two most common exotic invasive species were Asian spiderwort (*Murdannia keisak* Hassk.) which increased from 1.9 ha in 2009 to 2.8 ha in 2010 and to 3.6 ha in 2011 and Japanese stilt grass (*Microstegium vimineum* Trin.) which accounted for <0.5 ha in each growing season. We determined that narrow-leaf cattail and Asian spiderwort were the most dominant species in tidal portions of the basin. In non-tidal portions of the basin rice cutgrass tended to dominate vegetative communities and there were fewer invasive species present.

Introduction

Freshwater marshes in the Mid-Atlantic region would be typically dominated by emergent herbaceous vegetation such as graminoids, sedges, broad-leaved monocotyledons, and herbaceous dicotyledons (Mitsch et al. 2009). Typical graminoids to be expected would be *Spartina cynosuroides* (big cordgrass), *Zizania* spp. (wild rice), *Leersia oryzoides* (rice cutgrass) and gramminoid-like sedges and rushes (Silberhorn 1999, Mitsch et al. 2009). Examples of sedges often observed in freshwater marshes of the Mid-Atlantic region would be *Carex* spp., *Shoenoplectus tabernaemontani* (bulrush), *Scripus fluviatilis* (river bulrush), and *Eleocharis* spp. (spike rush) (Mitsch et al. 2009). Broad-leaved monocotyledons likely to be encountered would be *Sagittaria* spp. (arrowhead, bull tounge), *Peltandra virginica* (arrow arum),and *Pontederia cordata* (pickerel weed) (Mitsch et al. 2009). Plants from the herbaceous dicotyledons that would normally be expected in freshwater marsh are *Ambrosia* spp. (ragweed), *Nuphar luteum* (cow lilly) and *Polygonum* spp. (smartweeds) (Mitsch et al., 2009).

Typha spp. (cattails) are considered effective invaders (while they are not exotic invaders they could be said to fall under a category of native invaders) and colonizers in freshwater wetlands (Svengsouk and Mitsch 2000); this may be related to their resilience to extended hydroperiods (Anderson and Mitsch 2005), and rhizomal growth pattern (Mitsch et al., 2009). Ecophysiological characteristics of *Phragmites australis* and *Typha* spp. also play a role in their invasive nature (Farnsworth and Meyerson 2003). An important ecophysiological characteristic that aids *Typha domingensis* (an effective invader of the Florida Everglades) is a very high capacity for phosphate uptake and subsequent utilization; this trait may also aid in compensation for intense redox conditions (Li et al. 2010). Other *Typha* spp. are likely to share similar traits and tendencies leading to strong competitive abilities and a potentially invasive nature. Wetlands on the Mid-Atlantic coastal plain (especially those in watersheds with high agricultural uses and urbanization along rivers) have high nutrient deposition rates (Noe and Hupp 2005) and consequently have availability of phosphorus that may lend vulnerability to *Typha* invasion (Urban et al. 1993).

Aquatic ecosystems also, even when healthy, have niche space available to colonizers which makes them susceptible to invasive species (Capers et al. 2007). Thus the restoration of native plant species in disturbed wetlands can have variable success, due to compounding effects: persistent established species, extirpated original inhabitants, and opportunistic invasive species colonization (Mitsch and Gosselink 2007). When hydrology is restored to disturbed broadleaf marshes response can be variable. Some marshes show twice their previous broadleaf cover and others show no or very little change in broadleaf cover; species establishment during the disturbance period and invasive species colonization may play a part in this variability of restoration success (Toth 2009).

Microcosm experiments have shown that nutrient availability and hydrology may have an effect on wetland plant species richness and community assemblages (Nygaard and Ejrnaes 2009). Lower nutrients and more waterlogged soils increase species richness by limiting seedling establishment of competitively superior species (Nygaard and Ejrnaes 2009). Higher nutrient levels lead to asymmetrical competition or competitive exclusion (Nygaard and Ejrnaes 2009). Stress resulting from anaerobic soil conditions and nutrient scarcity may prevent the seedlings of competitively superiors species from becoming established in early stages of succession (Nygaard and Ejrnaes 2009). Elevation gradients often dictate hydrology in marshes, thus influencing marsh plant species composition and/or species location within the marsh (Suchrow and Jensen 2010).

Keystone species, which are species that have inordinate influences on community structure (Paine 1966, Paine 1969), can dictate community structure (Molles 2005). These species are not always dominant competitors so may need disturbances to stay locally extant (Mallik 2003). Disturbances that could increase species diversity of plants with inferior competitive abilities may include herbivory (Lubchenco 1978), as well as physical disturbances such as meteorological events or fire (Engelhardt and Ritchie 2002, Mallik 2003).

Current wetland restoration practices seek to create a wetland with structure and function as similar as possible to the original wetland (Mitsch and Gosselink 2007). Most restoration efforts use a reference wetland in combination with baseline information from the original wetland prior to wetland disturbance or destruction (Mitsch and Gosselink 2007). Active and passive wetland restoration efforts in depressional wetlands and other wetland types often have unpredictable results and successional trajectories (Zedler 2000, De Steven et al. 2006). Similar results have been found in forested wetland mitigation sites (Matthews et al. 2009). These systems were often less species rich than reference conditions and had unpredictable successional pathways (Matthews et al. 2009). Successional position will be important for efficient guidance of restoration efforts (Mitsch and Gosselink 2007).

Objectives

The primary objective was to determine colonizing species within the restored wetland and then assess changes in community coverage over three growing seasons. A secondary objective was to determine if invasive species were present within the existing vegetative communities colonizing the newly formed wetland and to determine if these species were expanding over time. A tertiary objective was to quantify and compare species coverage along line intercept transects in tidal and non-tidal areas within the restored wetland both before historical channel reestablishment and after historical channel reestablishment.

Methods

During the growing seasons of 2009, 2010, and 2011 vegetative community delineations were performed in the field over the entire former Lake Charles basin at the VCU Rice Center using a GPS receiving unit and various dichotomous keys to identify macrophyte species. Nomenclature followed: Godfrey and Wooten 1981, Duncan and Duncan 1987, Uva et al. 1997, and Silberhorn 1999. In the field, plant communities were identified by top three dominant species. A GPS receiving unit (Garmin MAP60) was used to collect waypoints and form polygons demarcating each vegetative community. Data obtained from the field were then used in a GIS environment to form polygons of vegetative communities. GPS waypoints formed the vertices for polygon features representing plant communities. Plant community boundaries were then overlaid on a map of former Lake Charles using geographic information systems (GIS) software. All GIS methods were completed with ERSI's ArcGIS software suite. GIS derived aggregate area of communities were compared by dominant species to examine change across the three growing seasons.

Line intercept transects were used to quantify species' coverage across the site (Crawford and Young 1998a). Transects were established prior to channel reestablishment in September and October of 2010 and repeated after channel reestablishment in September and October of 2011 (Figure 1). A Sorensen dissimilarity index was used to examine the difference between pre-channel reestablishment and post-channel reestablishment species coverage (Judd and Lonard 2002). Species coverage was separated by year (2010, 2011) and the difference was calculated in the software package PC-ORD. Transect cover was also separated by tidal and non-tidal areas to be compared across years using the Sorensen dissimilarity index.

For this study species coverage refers to the area along a line intercept transect that is covered by a certain species and community coverage describes the area of a community as determined by the top three dominant species within that community. Each method was used to achieve different objectives and are not used interchangeably. The combination of these methods (community delineations and line intercept transects) was used to more completely describe the standing cover than either method can achieve alone. This combination also provided two levels of detail in baseline vegetative assessment, both of which will be important for restoration and management purposes.

Results

Thirteen woody species and forty-six herbaceous species were observed in the standing cover of the restored wetland (Table 1). Species were identified during community delineations over three growing seasons and line intercept transect methods over two growing seasons. Nine species dominated vegetative communities in the 2009 growing season (Figure 2). Eleven species dominated vegetative communities in the 2010 growing season (Figure 3). Nine species dominated vegetative communities in the 2011 growing season (Figure 4). The vegetative communities were variable in spatial arrangement but the dominant species have remained stable from year to year with occasional minor alterations (Figures 2, 3, and 4, respectively). The total area (~20 ha) covered by vegetative communities was nearly the same across three growing seasons. Fourteen different species were a primary dominant within one or more vegetative communities from 2009-2011 (Table 2).

When comparing community coverage by species communities are referred to by their domiant species (i.e. *Leersia oryzoides* dominated communities are referred to as *L. oryzoides* communities). *Typha angustifolia* communities covered the greatest area in the basin across all growing seasons (2009: 9 ha, 2010: 9.4 ha, 2011: 8.7 ha) while the second greatest portion of the basin was vegetated by *Leersia oryzoides* communities (2009: 5.3 ha, 2010: 5.5 ha, 2011: 5.6 ha) (Figure 5). Area covered by *L. oryzoides* communities increased gradually throughout the study. *Typha angustifolia* communities (2009: 46%, 2010: 46%, 2011: 43%) and *Leersia oryzoides* communities (2009: 27%, 2010: 27%, 2011: 28%) covered the greatest portion of area throughout the study (Figure 5). These two species' communities accounted for approximately 72% of the coverage in the basin during all growing seasons.

Several other native species communities that covered a much lower extent (<10%) of the basin than those of *L. oryzoides* or *T. angustifolia* may play a role in future development of this ecosystem (Table 2). *Agrostis stolonifera* communities increased during each growing season this study was conducted (2009: 0.2 ha, 2010: 0.6 ha, 2011: 0.9 ha). These communities were largely limited to non-tidal portions and wetland/upland ecotones of the tidal portions in the wetland. *Polygonum sagitatum* communities had a small (2-3 ha) but stable presence during each growing season (Table 2, Figure 5). While *P. sagitatum* communities were limited to non-tidal areas the species was observed in the tidal portions of the basin and may play a future role there. *Hibiscus mosheutos* emerged as a community dominant by 2011 in the non-tidal portion of the basin. *Heteranthia reniformes* communities covered several hectares of the tidal wetland areas during 2009 but were replaced by *Murdania keisak* in 2010 and did not reemerge as a dominant plant in 2011 (Figure 5).

Non-native invasive species dominated several communities in each growing season (Figures 2, 3, and 4 respectively). Community coverage by *M. keisak* increased during each growing season within the tidal portions of the wetland. *Microstegium vimineum* communities were most common in non-tidal portions and in the arm on the east side of the basin (Figures 2, 3, and 4 respectively). *Phragmites australis* was present in several tidal communities and along the eastern wetland ecotone during 2010 and 2011 growing season but had not yet become dominant in any community.

Between the 2010 and 2011 growing seasons the standing cover quantified by the line intercept method was 15% dissimilar according to the Sorensen index of dissimilarity (Table 3). There was 32% dissimilarity between non-tidal portions of the basin between growing seasons (Table 3). In the tidal portion of the basin there was 14% dissimilarity between the 2010 and 2011 growing seasons (Table 3). The non-tidal portion of the basin was 38% dissimilar from the tidal portion in both growing seasons (Table 3). Species richness over the composite transect was 26 in 2010 and 39 in 2011 (Table 4 and Figure 6). Along transects in non-tidal areas of the wetland species richness was 19 during 2010 (Table 4 and Figure 6). Species richness was also 19 along tidal transects in 2010 (7 species were different between non-tidal and tidal transects) (Table 4 and Figure 6). In 2011 species richness was 26 along non-tidal transects. Species richness was 31 along tidal transects (eight species were different between non-tidal and tidal transects) (Table 4 and Figure 6).

Leersia oryzoides (2010: 33%, 2011: 35%) and *T. angustifolia* (2010: 30%, 2011: 26%) covered the greatest percentages of the composite line intercept transect during both growing seasons (Table 4). *Typha angustifolia* covered the most area in the tidal portions of the basin and was largely limited to tidal habitats, covering less than 1% of the non-tidal transect area in both years (Table 4). Substantial areas, in tidal and non-tidal portions of the basin, were covered by *L. oryzoides* in a given transect per year (between 14% and 21%) (Table 4). *Juncus effusus* increased percent cover in the non-tidal portions of the basin (Table 4). In 2010 *E. hieracifolia* accounted for almost 5% of the non-tidal transect cover and nearly disappeared from standing cover in 2011 (Table 3).

Non-native invasive species observed along the line intercept transect were *M. keisak* and *M. vimineum* (Table 3). *Murdania keisak* was most prominent in tidal areas of the basin covering 17% in 2010 and 13% in 2011 (Table 3). In non-tidal areas *M. keisak* covered ~2% transect area in each year (Table 3). *Microstegium vimineum* was limited to non-tidal portions of the basin in 2010 (1%) and covered less than 1% of the tidal transect area in 2011 (Table 3). In 2011 *M. vimineum* covered about 4% of the transect area in non-tidal portions of the basin (Table 3). Despite being present in several communities *P. australis* was not present along transects.

Over three growing seasons fourteen different species have dominated communities in this wetland (nine in 2009, eleven in 2010, and nine in 2011) (Table 2). Three herbaceous exotic invasive species have been observed in the restored wetland and two currently play a dominant role this ecosystem. Tree and shrub species are colonizing both non-tidal and tidal portions of the basin and are encroaching from the wetland/upland ecotone towards Kimages Creek's main channel. No communities have become primarily dominated by woody species as of the 2011 growing season. However, based on qualitative observation in the field *Acer rubrum* (red maple), *Betula nigra* (river birch), *Liquidambar styraciflua* (sweet gum), *Nyssa sylvatica* var. *biflora* (black gum), *Pinus taeda* (loblolly pine), *Platanus occidentalis* (sycamore), *Salix nigra* (black willow) and *T. distichum* are among the dominant woody species colonizing the wetland.

Discussion

The vegetative communities within the basin were variable in spatial arrangement but the species dominating these vegetative communities remained stable from 2009-2011 with occasional minor alterations (Figures 2, 3, and 4, respectively). This was expected because wetland ecosystems have increased colonization rates during early succession (Mitsch et al. 1998, Mitsch et al. 2012). While *T. angustifolia* and *L. oryzoides* accounted for the greatest community coverage over the past three years, fourteen different species have dominated at least one community over the past three growing seasons in this wetland. Species richness of primary community dominants peaked in 2010 and eleven, which was up two species from 2009. In 2011 community dominant species richness returned to nine.

Typha angustifolia and *L. oryzoides* combine for an average of 72% of community coverage in the wetland for each year over the three growing seasons sampled. These two species combine to dominate the greater parts of both non-tidal (*L. oryzoides*) and tidal parts (*T. angustifolia*) of the basin. Despite high community coverage by *T. angustifolia* it was not the dominant species in the wetland by transect coverage. *Leersia oryzoides* covered more area along the total transect in both 2010 and 2011. Transect cover for *L. oryzoides* is similar in both the non-tidal and tidal portions of the basin. *Typha angustifolia* seems to be largely limited to tidal portions of the basin based on transect data and community delineations, although this species is present in the non-tidal areas.

Community development in non-tidal and tidal areas of Kimages Creek wetland appears to be a product of propagule dispersal, abiotic factors and biotic competition, which is expected under the environmental sieve hypothesis (Lambers et al. 2006). Vegetative tidal and non-tidal areas appear to potentially have differing propagule sources. These areas also appear to have differing biotic and environmental factors influencing vegetative community development. Patterns in community development in the Kimages Creek wetland also seem to be a product of evolving hydrologic conditions, patterns of elevation, and low level disturbances (particularly beaver activity, herbivory and sediment deposits). In the non-tidal portions of the basin the stream channel is affected by several areas of beaver activity (dams), which may be limiting full tidal exchange in these areas (potentially making areas northern areas of Kimages Creek wetland non-tidal). Beaver activity also seems to be limiting recruitment of *Salix nigra* in northern parts of the tidal basin (anecdotal evidence). No quantification of beaver activity has been conducted to this point. In fall 2010 a large portion of the above ground biomass for *T. angustifolia* was affected by herbivory from the larval stage of the cattail caterpillar moth, (*Simyra insularis*). According to the U.S. Army Corps of Engineer's Engineer Research and Development Center Aquatic Plant Management Information Center, *S. insularis* is listed as a Biocontrol agent for cattails. Effects of this disturbance event were not quantified; although it may have contributed to the decrease in *T. angustifolia* transect coverage during 2011. It has also been noted during each growing season that sediment deposits are a common localized disturbance to the non-tidal vegetative communities. These deposits have observationally similar patterns to those seen during barrier island overwash events. Specific effects of these disturbances have not yet been quantified.

Community spatial extent may fluctuate between growing seasons while total species cover remains semi-stable. The Sorensen index of dissimilarity showed a 15% difference in species composition along transects between the 2010 and 2011 growing seasons. This difference shows that there was a slight change in the wetland vegetative community at the site between growing seasons. It is likely that vegetative cover at the site will continue to fluctuate between growing seasons in a potentially similar to upland early successional forest ecosystems (Weiher et al. 1996, Swanson et al. 2011). Based on the Sorensen index tidal areas of the basin changed about 14% between growing seasons which followed site trends. Non-tidal areas of the basin changed more than twice that of tidal areas (32%). The plant communities in the non-tidal area have appeared to be in a greater state of flux throughout the study (anecdotal evidence). Perhaps tidal flux is a strong abiotic factor limiting competition in tidal areas and plants in the non-tidal areas, which may experience reduced stress, are able to compete more adequately with each other. The almost complete disappearance of *E. hieracifolia* (4.4% in 2010 and less than 1% in 2011 of non-tidal transect area) and increases in *J. effusus* (5% in 2010 and 8% in 2011) as well as increases in *M. vimineum* (1% in 2010 to 5% in 2011) probably account for much of the change in non-tidal areas. The 14% change in transect coverage between growing seasons for tidal areas of the basin may largely be due to a 7% increase in *L. oryzoides* (14% in 2010 and 21% in 2011) and decreases in both *M. keisak* (16% in 2010 and 13% in 2011) and *T. angustifolia* (29% in 2010 and 25% in 2011). This may indicate that channel reestablishment is causing decreases in both native and non-native invasive species allowing desired cover to increase.

In restored ecosystems, invasive species presence is important to assess because of the ability these species possess to inhibit or subvert restoration goals (Ehrenfeld and Toth 1997). Exotic species were shown in this study to have varying trends across growing seasons and tidal regime. Invasive species cover increased in non-tidal portions of the basin and was primarily due to expansion of *M. vimineum*. It should be noted that *M. keisak* also increased in the non-tidal portion of the basin although the percentage of increase was much lower that *M. vimineum*. Increase in *M. vimineum* could be detrimental to restoration efforts because this plant is shade tolerant (Barden 1987) and will most likely not be affected by natural woody succession or anthropogenic tree plantings. This species may need to be controlled by herbicides or hand removal (Flory 2010). Herbicide and hand removal treatments were shown to be effective at a variety of forested sites and across a variety of environmental conditions in Indiana (Flory 2010). Following removal of *M. vimineum* floristic communities were also shown to recover in these ecosystems (Flory 2010). These methods may be applicable to the non-tidal portions of Kimages Creek marsh.

Tidal portions of the basin showed a decrease in coverage of both non-native species and native invasive species. Decreases in transect coverage by these species are counter to expectation as invasive species tend to persist and expand once established (Zedler 2000). *Murdania keisak* decreased along the transect approximately 4% (2010: 17% to 2011: 13%) and *T. angustifolia* decreased approximated 4% (2010: 29% to 2011: 25%) in tidal portions of the basin. *Microstegium vimineum* did increase marginally but not enough to alter the overall trend of non-native invasive species in tidal portions of the basin. In addition to herbivory, the decrease in *T. angustifolia* may also be due to increased tidal flux resulting from channel reestablishment. Reduced seedling emergence has been shown for other *Typha* spp. in response to increased inundation (Baldwin et al. 2001). This may result in less competition for *L. oryzoides* as increased inundation does not affect germination for this species (Baldwin et al. 2001). Tidal portions of the basin are also being colonized by *Zizania aquatica* (northern wild rice) and *Zizaniopsis miliacea* (southern wild rice). These two native species may be responding favorably to increased tidal flux, as they appeared along the transect in 2011, and may compete with invasive species in the tidal portion of the basin. Invasive species that are non-native should be monitored through ecosystem succession because of their ability to persist and potentially hinder succession once they have become established (Zedler 2000, Gutrich et al. 2009,).

Overall species richness along the transects increased between the 2010 and 2011 growing seasons from 26 to 39, increases were detected in both non-tidal and tidal portions of the basin between the two growing seasons. Decreases in both *M. keisak* and *T. angustifolia* may contribute to or be caused by increased species richness in tidal portions of the wetland. Channel reestablishment may also have played a role in species richness increase and decrease of invasive species cover along transects in the tidal portion of the basin. Increases in species richness were unexpected in the presence of high invasive species cover based on a study of created depressional marshes (Gutrich et al. 2009). Species richness may also be increasing with age of the site which is an expected trend during ecological succession (Odum 1969). Species richness will probably continue to increase as the ecosystem develops due to interactions of extant vegetation canopy, seed availability, elevation, and litter accumulation (Xiong et al. 2003).

The species observed colonizing this wetland are typical and expected in the Mid-Atlantic and Southeast regions of the United States (Odum 1988, Batzer and Sharitz 2006, Mitsch and Gosselink 2007, Mitsch et al. 2009). Trends in species richness were also expected under the theory of succession (Odum 1969) but contrary to expectations under conditions of high invasive species coverage at the site (Gutrich et al. 2009). Decreases in coverage of invasive species were also unexpected but may be a product of restoration efforts, other abiotic factors, or biotic factors. Annual fluctuations in spatial extent of plant communities of this ecosystem support trends seen in early successional ecosystems of forests and dunes (Cowles 1899, Swanson et al. 2011).

Future directions should include continuing to annually delineate plant communities and repeating transect based cover studies. It may also be prudent to set up permanent plots for assessing succession and vegetative community dynamics at a finer resolution in a restoration context. Continued studies of this nature could quantify temporal and spatial scale questions about how many of these dominant plants affect ecosystem succession. Data from community delineations and transects after woody species plantings have occurred will be good for investigating the response of the herbaceous community to woody encroachment. A full study on natural woody recruitment in the basin would also be a logical and necessary next step to this research. This would establish a quantitative baseline of woody recruitment with which to assess tree/shrub planting needs.

Quantification of beaver activity and it's affect on hydrologic conditions in the non-tidal portions of the basin and it's affects on woody species recruitment in the basin would also be an important next step for assessing the vegetative cover at this site. Other environmental data such as soil composition or nutrient conditions along transects may be important for understanding why the basin has developed its current vegetative community. Quantifying the effects of sediment deposits and "overwash" events in the non-tidal portions of the basin may also be important for understanding vegetative community development in the northern end of the basin. Understanding the role of herbivory would also be a good follow up study to address the mechanisms of community development in Kimages Creek wetland. Performing community delineations and cover transect studies on Harris Creek may also provide useful information to help guide restoration efforts of Kimages Creek.

Continuing to collect these data through various steps of the restoration process will be important to documenting and elucidating vegetative community dynamics within this restoration setting. Understanding the interaction of management, anthropogenic restoration, natural regeneration, and succession of both natural and restored vegetation at this site will improve wetland science throughout the Mid-Atlantic region. Over time, ecosystem structure changes which can alter abiotic environmental conditions through succession, making long term monitoring of restored sites vital to the success of restoration efforts (Ballantine and Schneider 2009). There are also many long term questions about forested wetland succession that can be answered by continuing monitoring studies of this nature. The baseline framework for studying restoration that this study has set up could be used to answer many long term questions about succession in a restoration context for forested freshwater wetlands of the Mid-Atlantic region.

Chapter 2

Spatial and Temporal Seed Bank Dynamics within Natural and Restored Wetland Settings

Abstract

Extant soil seed banks in four wetland habitats were assessed and compared to their respective standing cover for potential similarity and differences based on habitat type. Habitats sampled were Harris Creek forested wetland, Kimages Creek non-tidal marsh, Kimages Creek tidal marsh, and Kimages creek unvegetated mudflats. Soil seed banks were also assessed for the presence of invasive species. Differences in extant species at the site and seed bank composition were observed across all habitats. Ten woody and forty-three herbaceous species emerged from the soil seed bank. Woody species only emerged from Harris Creek samples. Eight species differed in seedling density (#/m^2) among habitats. Non-metric multidimensional scaling separated seed bank species composition by habitats into three groups (Kimages non-tidal, Kimages mudflats, and Harris Creek/Kimages Creek tidal marsh).

Introduction

Soil seed banks can be defined as the ungerminated seeds in the soil that can and may potentially replace adult plants after natural or anthropogenic removal (Baker 1989). Seed banks are composed of all seeds resting on the soil, buried in the soil and contained in associated litter (DeBerry and Perry 2000a). Soil seed banks are critical to the establishment of vegetative wetland communities during draw down and flooding events, where some species may become newly established and others will become extirpated (van der Valk 1981). The soil seed bank is one of the most important components of wetland ecosystems (DeBerry and Perry 2000a).

For a variety of ecosystems the soil seed banks consist of successional annuals, rather than the standing perennial community, being well represented in the soil seed banks (Roach 1983, Leck and Simpson 1995, Kalin et al. 1999, Capon and Brock 2006, Caballero et al. 2008). Standing cover in wetlands may not affect composition of the seed bank, but will affect germination and seedling success, in turn affecting the community structure of the extant vegetation (Baldwin et al. 1996). Extant wetland seed banks allow species to become established under varying hydrologic conditions, which at times promotes annual species and at other times promotes persistence of perennials (van der Valk 1992). Sedimentation events also play a critical role in the dynamics of soil seed banks by altering which species will germinate and go to seed after such an event (Jurik et al. 1994). Disturbances such as heavy sedimentation events can prevent some invasive species with small seed sizes, such as *Typha* spp. from germinating (Jurik et al. 1994). Such disturbance events may aid species with larger seeds and inferior competitive abilities (Jurik et al. 1994).

Invasive species can be well represented in seed banks (Welling and Becker 1990), and can prevent reestablishment of native species during restoration or post-disturbance successional recovery (Zedler 2000). Such species may dominate and can potentially arrest succession (Zedler 2000). Seeds from *Typha* spp. cannot persist in the seed bank under prolonged drought or inundated conditions and must rely on colonizing wetlands during short periods of drought or other disturbance (Batzer and Sharitz 2006). This makes it possible in *Typha* spp. dominated marshes that the seed bank will be representative of communities that were succeeded or replaced by *Typha* spp. Seeds can enter wetland seed banks through a variety of dispersal vectors, including anemochory, hydrochory, and zoochory (Figuerola and Green 2002, Soons 2006, and Mitsch et al. 2009). Species represented in the seed bank are largely limited by the first physiological filter under the environmental sieve hypothesis which is physical access of propagules to an area (Lambers et al 2006). In comparrison the seed bank can also house seeds of rare plants that are not well represented or even represented at all in each and every growing season (Bailey et al. 2006). It is possible that a rare plant's population in an ideal year contribute to occurrence of a population in a subsequent growing season and that this may be a function of the seed bank (Bailey et al. 2006). Seed banks can also aid rare plants in maintaining genic diversity (McCue and Holtsford 1998).

Objectives

This study was designed to assess potentially colonizing species found within the soil seed banks of four freshwater wetland habitats: Kimages Creek non-tidal marsh, Kimages Creek tidal marsh, Kimages Creek tidal mudflats, and within a tidal forested wetland (Harris Creek). The primary objective of this study was to determine if species present within the soil seed banks were representative of the standing cover. The second objective was to determine if there were differences in species emerging from the soil seed banks among the four wetland habitats. The third objective was to assess the seed bank for the presence of both native and exotic invasive species.

Methods

Assessment of soil seed bank species composition was done using the seedling emergence technique (DeBerry and Perry 2000a). Ten 500 cm^3 soil samples were collected in each of the four freshwater wetland habitats sampled at the VCU Rice Center. Seed bank samples were collected from Harris Creek swamp (a reference forested tidal freshwater wetland), from tidal portions of Kimages Creek marsh, from non-tidal portions of Kimages Creek marsh, and from newly exposed mudflats of Kimages Creek during mid-March, 2011 (Figure 1). The VCU Trani Life Sciences greenhouse was used to grow seedlings from samples placed in trays 1 m x 0.5 m containing MiracleGro® potting soils. Four control trays were potted with random allotments of sterile potting soil from each bag used to pot other treatments and watered the same as other treatments. Combined potting soil and substrate depth was ~7.5 cm, sterile potting soil depth in control trays was ~7.5 cm. Obvious rhizomes and roots were removed before potting. As soon as germinating seedlings could be positively identified they were removed from the sample tray (seedlings removed from trays were kept as voucher specimens for each species) (Crawford and Young 1998a, DeBerry and Perry 2000a, Deberry and Perry 2000b, Peterson and Baldwin 2004). Including controls a total of 44 trays were used in this study. Treatments were terminated after ~9 months of emergence in December of 2011. Voucher samples have been archived in herbarium collections of the VCU Department of Biology and the VCU Rice Center. Seedlings were identified using various dichotomous keys and nomenclature follows: Godfrey and Wooten 1981, Duncan and Duncan 1987, Uva et al. 1997, and Silberhorn 1999.

Species represented in the soil seed bank of each community were compared with standing cover species observed in the field using Sorenson's (Bray-Curtis) index of dissimilarity (DeBerry and Perry 2004, Neill et al. 2009). This index was originally created to use binary data, although the index works well with abundance data (McCune and Grace 2002). Species richness of the seed banks across all treatments of the basin was compared to species richness of standing cover in each habitat. Density (#/m^2) of seedlings in soil seed bank was compared for species emerging in all treatments using an ANOVA and Tukey HSD *post hoc* test, α=0.05. Species diversity was assessed across seed bank treatments using the Shannon-Weiner diversity index.

Non-metric multidimensional scaling was used to ordinate communities according to habitat (Nicol et al. 2003). Multiple response permutation procedures were used to assess within group separation of samples based on habitat and tidal regime (Nicol et al. 2003). Multiple pairwise comparisons were corrected using the Bonferroni method (Brosofske et al. 2001).

Results

Ten woody and forty-three herbaceous species emerged from the soil seed banks (Table 5). The woody species were only found in the Harris Creek seed bank treatment (Table 5). Notably *T. distichum* was observed in both the seed bank and standing cover of Harris Creek, and was the most abundant woody species in the seed bank followed by *Platanus occidentalis* (Table 6). Eight species differed in seedling density (#/m^2) among habitats (Table 6). The highest herbaceous species richness among seed banks was also observed in Harris Creek (Figure 7). The lowest herbaceous species richness was observed in the Kimages Creek wetland non-vegetated mud flat treatment (Figure 7). Based on the Shannon-Weiner Index highest soil seed bank diversity was observed in the Harris Creek treatment and lowest in the Kimages Creek tidal vegetated treatment (Figure 8). Species, such as *L. oryszoides* and *M. keisak,* that have substantial coverage in the field seem to reflect this in the seed bank with high seedling abundances and may skew diversity results.

Sorensen (Bray-Curtis) index of dissimilarity results showed considerable differences in standing cover and seed bank composition across all treatments (Table 7). Differences within the treatment for standing cover and seed bank ranged from 48% (Kimages Creek wetland non-tidal habitat) to 76% (Harris Creek) (Table 7). Differences among habitats for seed bank and standing cover ranged from 30% (Kimages Creek non-vegetated mudflat seed bank and Kimages Creek tidal vegetated seed bank) to 92% (Kmiages Creek tidal vegetated seed bank and Harris Creek standing cover) (Table 7). Non-metric multi-dimensional scaling revealed separation in seed bank composition based on habitat type (Figure 9). Seed bank composition in Kimages non-tidal marsh had distinct grouping from seed bank composition in other habitats (Figure 9). Seed bank composition of Kimages Creek mudflats was also distinctly grouped from other habitats (Figure 9). Harris Creek seed bank and Kimages tidal marsh seed bank composed the third group on the ordination. Multi-response permutation process showed less within group separation than would be expected by chance in species composition for habitat (Table 8). After the Bonferroni correction, significant differences ($p<0.01$) existed in seed bank composition between all habitats except Harris Creek and Kimages Creek vegetated tidal areas ($p=0.75$) (Table 8).

The invasive species *Murdania keisak* was present in all seed bank treatments. *Microstegium vimineum* was present in all seed bank treatments except the non-vegetated mudflats. *Albizia julibrissia* (mimosa tree) and *Ailianthus altissima* (tree of heaven) were observed in the Harris Creek seed bank treatment. *Phragmites australis* has been observed in the standing cover of the Kimages Creek tidal wetland but was not observed in any seed bank treatments. *Typha angustifolia* was observed in all seed bank treatments except those from tidal vegetated areas of Kimages Creek.

Discussion

The community composition in the seed bank treatments did not reflect the standing cover but may be indicative of pioneering communities in these habitats. The seed bank composition was different than the standing cover for all habitats sampled, which was similar to findings of other seed bank studies from a variety of terrestrial and aquatic ecosystems (Roach

1983, Leck and Simpson 1995, Crawford and Young 1998a, Kalin et al. 1999, Capon and Brock 2006, Caballero et al. 2008). Results from this study may support a similar trend for seed banks of tidal freshwater wetlands. Seed bank treatments also exhibited differences among habitats except between Kimages Creek tidal vegetated and Harris Creek which were similar according to the NMS ordination and subsequent MRPP. Other habitats were distinctly separated from each other and the Harris Creek/Kimages Creek Tidal marsh group.

The large difference between Harris Creek standing cover and Kimages Creek tidal vegetated seed bank (92%) in combination with 50% dissimilar seed bank, as determined by the Sorensen index, may indicate that recruitment in Harris Creek may be, at least, partially limited by shade from standing cover (forested canopy). Although this suggestion needs further study because the standing cover in each habitat was different than their respective seed banks. The multiple response permutation procedure performed on the NMS results indicated that the there was no statistical separation between the species compositions of Harris Creek and Kimages tidal marsh seed bank treatments. The lack of separation in these two treatments means that they likely have a similar seed source, which is probably a result of tidal exchange with the James River. This information is beneficial for restoration purposes because shading may be a potential method to remove invasive species within the restored site. Differences in the standing cover of the two wetlands may also indicate that reforestation of this wetland will limit shade intolerant invasive species spread. Differences in seed bank composition may also be related to varying hydrologic regimes across the four habitats sampled (Schneider and Sharitz 1986).

Interestingly, no woody species were observed in the soil seed bank outside of the Harris Creek treatment. Harris Creek is a forested wetland so woody species were expected in that treatment. Despite the fact that the other treatments were in herbaceous dominated wetlands or non-vegetated mudflat habitats it was expected that some woody species would be observed. This expectation was based on current observations of woody encroachment from the wetland/upland ecotone and colonization of interior habitats by woody species in both non-tidal and tidal vegetated areas of Kimages Creek. In both vegetated non-tidal and tidal areas of the Kimages Creek wetland woody species recruitment is occurring, hence the expectation of woody species in the soil seed bank. Even though woody species did not emerge from Kimages Creek soil seed banks (non-tidal marsh, tidal marsh, and mudflat), it does not mean that they were absent from the soil seed bank. Seeds from woody species may not have been collected with the samples initially or they may not have germinated under high light, warm temperatures, and adequate moisture conditions of the greenhouse in these three treatments. This result is not entirely surprising, because woody species may not persist as long in seed banks of both herbaceous and woody dominated wetlands (Middleton 2003).

Patterns of seed bank species richness and diversity could be related to site age and structural development (Leck 2003). Habitats with substrate that have been exposed the longest showed the highest species richness in the study. Harris Creek had the most species rich seed bank, followed by Kimages Non-tidal marsh, Kimages tidal marsh and Kimages mudflats respectively. The mean seed bank abundance of *M. keisak* in tidal marsh samples was more than twice that of any other habitat, which likely affected the diversity results.

Non-native and native invasive species were present in the seed bank, which was expected due to their presence in the standing cover. *Murdania keisak* had the greatest abundence of any invasive species in the seed bank which mirrors its role in the standing cover. *Microstegium vimineum* was also present in the seed bank study in a capacity similar to its current standing cover. While *T. angustifolia* was present in the seed bank it was not frequent enough to mirror its role in the standing cover. The decreased presence of *T. angustifolia* was expected because the inability of its seeds to persist for long in the seed bank (Batzer and Sharitz 2006), yet unexpected as it is the dominant species in Kimages Creek tidal marsh. The absence of *Phragmites australis* in the seed bank was partially expected because it tends to have low seed viability (Kettering and Whigham 2009) and was only sparsely represented in Kimages Creek wetland.

Seed banks in other habitats often do not reflect the current community structure but show species of a pioneering or early successional nature (Roach 1983, Leck and Simpson 1995, Crawford and Young 1998a, Kalin et al. 1999, Capon and Brock 2006, Caballero et al. 2008). This same trend may be occurring at the VCU Rice Center wetland. Invasive species were present in seed banks of each sampled habitat. Seed banks were most diverse in habitats with standing cover that had been developing the longest (Harris Creek and Kimages Creek non-tidal marsh). The seed bank in each habitat of Kimages probably reflects species from initial colonization that have been succeeded by species in the current standing cover. Differences in the non-tidal and tidal areas of Kimages Creek were probably influenced by the time of draw-down, which was approximately 16 months, before the basin became initially reconnected with the James River. The current non-tidal area was initially the only portion of the basin exposed and the current tidal areas were slowly exposed after about 16 months of draw-down from the initial breach. Species in Harris Creek and Kimages Creek tidal areas samples were probably from propagules deposited from James River tides. The non-tidal area of Kimages Creek likely receives propagules that disperse via hydrochory from upstream portions of Kimages Creek as well as anemochorous and zoochorous dispersal vectors. The tidal areas of Kimages creek probably receive propagules from combined sources of Kimages Creek, anemochory and zoochory as well as tidal input from the James River.

There are more than 2.5 million dams in the United States that impact current of former riparian habitats (NRC 1992). Restoration some of these impoundments to their historical condition as wetland ecosystems is an important aspect of restoring natural hydrologic conditions to rivers and streams (Zedler 2000). This study may be relevant to restoration of wetlands in former impoundments throughout the Mid-Atlantic region, and may be of particular importance to Chesapeake Bay tributary restoration projects where dam removal projects are among restoration priorities (Hassett et al. 2005). Assessment of the vegetative cover and seed bank has been suggested as a useful monitoring technique in wetland restoration projects (Baldwin and Derico 1999).

Tidal freshwater rivers can provide large influxes of seeds to newly restored wetlands and may make plantings or soil additions unnecessary (Leck 2003). However, restoration of formerly farmed forested flood plain wetlands in the Mississippi Valley may rely on plantings, rather than natural recruitment, because critical vegetative community components are not well represented in the seed bank (Middleton 2003). Results of this study showed that herbaceous vegetation was well represented in seed banks of the newly restored ecosystem and the reference site. Woody species were present in seed banks of the reference wetland and not in seed banks of the newly restored wetland. Based on this study and a study by Middleton (2003) trends of woody species representation in the seed bank during early stages of restoration may be similar in the Mid-Atlantic region and the Mississippi Valley. This may mean that plantings are necessary for restoring forested wetlands throughout out much of their former range on the eastern seaboard and Mississippi corridor.

Chapter 3

Taxodium distichum Age Structure Analysis and Habitat Suitability for Recruitment: Assessment of Restoration Potential for an Obligate Model Wetland Canopy Tree

Abstract

Taxodium distichum is a model wetland canopy species in the VCU Rice Center wetland restoration. This study collected baseline information on current geospatial relations of *T. distichum* and population age structure to aid restoration efforts. Ocular reconnaissance was used in combination with GPS/GIS methods to locate and map *T. distichum*. Two hundred and eighty bald *T. disticum* individuals were located and mapped using GPS methods within the Kimages Creek and Harris Creek wetlands. Within the restored Kimages Creek wetland over 75% of the *T. disticum* individuals found were seedlings or saplings. The population in Kimages Creek appears to be growing while the population in Harris Creek does not appear to be replacing itself. Using a GIS weighted suitability model we have identified potential areas within the restored wetland for natural and facilitated recruitment. Approximately 9.7 ha of Rice Center property (including both Kimages and Harris Creek wetlands) were identified by the model to have natural regeneration and 48.5 ha were modeled to have facilitated regeneration potential.

Introduction

Taxodium distichum (L.) Richard (Cupressaceae), is a common canopy tree of swamps in the Mid-Atlantic and Southeastern United States (Silberhon 1999, Batzer and Sharitz 2006, Mitsch and Gosselink 2009). Logging of southern swamps has greatly reduced many populations of *T. distichum* (Mitsch and Gosselink 2007). Existing populations are under threats from anthropogenic influences such as altered hydrologic regimes, altered nutrient regimes, altered sediment loadings and activities such as harvesting for timber and other forestry products (Faulkner et al. 2009). Harvesting these trees is particularly devastating because it alters the hydrologic regime sufficiently to turn them into open marsh (Faulkner et al. 2009). This generally results from the inability *T. distichum* to regenerate under continuously flooded conditions.

The ability of a *Taxodium* swamp to regenerate after disturbance by sapling/seedling recruitment is critical to the future community structure and composition of the swamp (Middleton 2009a, Middleton 2009b). Drawdown, with sufficient time for seedlings to achieve heights where at least fifty percent of their crowns are above the high water line, is necessary for *T. distichum* to recruit naturally (Faulkner et al. 2009). *Taxodium distichum* cones shed winged seeds in October, are dispersed by water (hydrochory) and have a short germination window (Middleton 2000). Seeds are dispersed during flooding in winter and then during a drawdown period in the subsequent growing season they germinate (Middleton 2000). Less than five percent of *T. distichum* seeds remained viable after one year on the soil surface (Middleton 2000).

Hydrochory is the main dispersal method for *T. distichum* propagules (Schnieder and Sharitz 1988). *Taxodium distichum* propagules are buoyant and can float for several months (Schnieder and Sharitz 1988). Propagule buoyancy increases the chances that seeds from this species will reach elevated substrate suitable for germination (Schnieder and Sharitz 1988, Howe and Smallwood 1982). Regeneration of forested wetlands is aided by increased flood pulsing, while impounded conditions are negatively associated with forested wetland regeneration (Middleton 2000). Slow moving riverine inputs are beneficial to seed dispersal for *T. distichum*

(Souther and Shaffer 2000). *Taxodium distichum* can tolerate a wide range of light conditions but grows fastest in high light environments (Neufeld 1983).

Forested wetland ecosystems are very complex and can be difficult to restore (Mitsch and Gosselink 2007). After disturbance or clearing these systems often do not have critical vegetative components stored in the seed banks and successful restoration often depends on reestablishing natural flood pulse conditions (Middleton 2003). Maintenance of diverse floodplain wetlands may at least partially depend on reestablishing the hydrologic seed inputs to these systems (Middleton 2003). This may also mean that the most important component for restoring forested wetlands is restoring the hydrology of these systems (Middleton 2003). Other challenges that have been cited in forested wetland restoration, particularly those dominated by *N. aquatica* and *T. distichum*, include herbivory, nutrient limitations and shading from a willow (*S. nigra*) canopy cover (Conner et al. 2000, Dulohery et al. 2000, Effler et al. 2006).

Objectives

Taxodium distichum is a model wetland canopy tree and a target species for wetland restoration at the VCU Rice Center. A small remnant adult population of *T. distichum* at the VCU Rice Center coupled with a recently initiated wetland restoration provides an opportunity to study recruitment and population dynamics of this model wetland species. The primary objective was to create a site specific restoration suitability model to guide *T. distichum* restoration at the VCU Rice Center and potentially similar areas along this reach of the James River. The secondary objective was to map all *T. distichum* individuals located on the VCU Rice Center property. A tertiary objective was to assess age structure of the extant *T. distichum* populations and determine if they are growing, stable or declining.

Methods

Taxodium distichum individuals were located by ocular reconnaissance along or in various wetland areas within the VCU Rice Center. Diameter at breast height or DBH (centimeters) was recorded for trees, saplings and seedlings (when possible) using vernier calipers. Seedlings heights were measured using meter sticks or meter tapes as needed. GPS locations and waypoints for the individual locations of *T. distichum* seedlings, saplings and trees at the VCU Rice Center were recorded by using a Trimble 5000XL GPS unit. Spatial trends in *T. distichum* occurrence and recruitment were investigated using GIS software. The GIS software used was ERSI's ArcGIS software suite. Regeneration areas of *T. distichum* can be divided into the following three categories that have been modified from restoration classes from Faulkner et al. (2009). The first of these categories is "natural regeneration", areas with potential for natural regeneration of *T. distichium* (RCC-I in Faulkner et al. 2009). The second is "managed regeneration", areas with potential for artificial restoration (RCC-II in Faulkner et al. 2009). The third is "no restoration", areas with no potential for either natural or artificial regeneration (RCC-III in Faulkner et al. 2009). We used these categories to classify the various areas of Kimages Creek marsh targeted for woody restoration.

Data for the development of this model were acquired from a variety of sources ranging from government run websites, GIS data creation methods and field studies using GPS methods. Land use/land cover data (2006, 30 m resolution) were downloaded from the Coastal Change Analysis Program, on the NOAA Coastal Services Center website (http://www.csc.noaa.gov/crs/lca/faq_data.html). Elevation data (10 m) were downloaded from the National Elevation Dataset through the USDA Geospatial Data Gateway website (http://datagateway.nrcs.usda.gov/GDGOrder.aspx). Wetlands data were downloaded from the

United States Fish and Wildlife Service National Wetlands Inventory website (http://www.fws.gov/wetlands/data/index.html). The area of interest for the landscape level analysis was created through attribute selection methods in the GIS. *Taxodium distichum* GPS points and DBH class data were acquired as the result of a pilot study on population age structure of this species at the VCU Rice Center. Diameter at breast height was used as a rudimentry proxy for age in this study. Seedlings and saplings were identified as individuals with DBH measurements <10 cm. Adult trees were classified as those individuals ≥10 cm in DBH. The area of interest for the VCU Rice Center was created through heads up digitizing (HUD). GIS methods and (methodology) are outlined in this section, refer to Appendix A: Graphic Work Flow and Appendix B: Detailed Work Flow for specifics.

Creation of distance classes was based seedling frequency of seedling proximity to adults. A histogram of Euclidian distances of seedlings from adults was created to determine the three distance classes. Fifty meters was used as the first ideal class because 75% of seedlings occurred within 50 m of an adult *T. distichum*. The majority of the remaing seedlings ~24% occurred within 100 m of an adult so 50-100 m was chosen as an intermediate disitance, and anything greater than 100 m was classified as non-ideal.

Creation of elevation classes was based on probability of hydrochorous seed exposure during high water events and potential for hydric soils. Elevations of 5 m and lower were chosen to be ideal, because they would likely experience inundation during high water events and have potential for hydric soils. Elevations between 5-10 m were given an intermediate class because they may become inundated during extreme high water events and have potential for hydric soils. Above 10 m it was deemed these evelations would probably not experience exposure to high waters or have hydric soils so were classified as non-ideal.

Land use/land cover data was classified based on wetland status or potentially suitable habitat for canopy trees. Wetland land cover classes were reclassified to ideal because they would likely have soil conditions and hydrology conducive to *T. distichum* recruitment. Forest land cover classes were given an intermediate designation because they have the potential to support canopy trees and at ideal elevations likely have hydric soils. Other land use/land cover classes were classified to be non-ideal.

A model for identifying areas of three different restoration classes was created. This model involved extraction of raster data based on a spatial mask, reclassification of data based on its suitability for *Taxodium distichum* restoration, and use in a weighted overlay. The data inputs and weights for this analysis were as follows: Euclidian distance from mature cypress (70%), elevation (20%), and land use/land cover classification (10%). The raster created by this model spatially classifies areas of the Rice Center by restoration suitability for *T. distichum*. This raster was then converted into a vector polygon file for the ancillary purpose of classifying the seedlings and mature trees by their restoration class. Distance from adults, as a proxy for seed rain falling from adults, was considered to be the most important factor for natural regeneration; so Euclidian distance from adults was expected to receive the highest weight in the model. After examining a frequency histogram and creating distances for reclassification this anticipation fueled iterations of the model experimenting with various weights (ranging from 40% to 80%). In combination with the other factors it was decided that the weight of 70% best described the spatial patterns in recruitment trends. Elevation which limits potential seed exposure via hydrochory was decided to be the next highest (also with multiple iterations of the model with various weights ranging from 5% to 40%) so 20% was decided upon. Land use/land cover was also decided on after multiple iterations of the model to describe the data best at 10%.

Resulting datasets from this study were exported from the GIS as database files. These database files were subsequently imported to a Microsoft Access database. Key files for explaining numeric codes in the databasefiles were created in Microsoft Excel and imported to the database. These key files were then related to the appropriate GIS exports. This database was also designed with the purpose of organizing the data from future research on this topic.

Results

Two hundred and eighty *T. distichum* individuals were mapped in the Kimages Creek and Harris Creek wetlands and along the James River on VCU Rice Center property (Figure 10). The *T. distichum* population within the Kimages Creek wetland is increasing (Figure 12). The population in Harris Creek seems to be stable or decreasing (i.e. the adult trees are not replacing themselves) (Figures 11). These trends are inferred when using DBH as a rough proxy for age. The majority of *T. distichum* individuals in Kimages Creek have DBH measurements of less than 10 cm (Figure 12). In Harris Creek the population of recruits is much smaller and only 3 individuals have DBH measurements under 10 cm (Figure 11).

All individuals found in Harris Creek were adults with the exception of two saplings and one seedling. In the Kimages Creek wetland more than 75% of individuals located were seedlings or saplings (Figure 10). The greatest concentrations of individuals were in the arm extending off the east side of the wetland (Figure 10). The non-tidal portions of the basin had the fewest individuals (Figure 10). A general trend of more individuals occupying in the southern and eastern portion of the basin was observed (Figure 10). The arm area on the eastern side of the basin has the highest concentration of adult *T. distichum* (6) and consequently the highest concentration of seedlings and saplings within the Kimages Creek wetland. Fewer adults and saplings/seedlings were observed on the western side of the basin (Figure 11).

The weighted suitability analysis yielded restoration classes for the various parameters of the model as well as a concluding suitability model for the restoration of *Taxodium distichum* at the VCU Rice Center (Figure 11). The area of the Rice Center property covered by each restoration class generated by these analyses was also quantified (Figure 13). Facilitated restoration covered most of Kimages Creek and Harris Creek wetlands (Figure 10). Natural regeneration was identified on about 9.7 ha of the property. The majority of both natural regeneration and artificial restoration areas are in the Kimages Creek wetland. It should be noted that if an area is classified as managed restoration it may still naturally experience *T. distichum* recruitment. However, the level of recruitment is likely to be low enough that managed restoration is required to achieve the target number of trees for those areas.

Discussion

The highest concentration of seedlings and saplings as well as overall number of *Taxodium distichum* individuals occurred in the arm region on the east side of the restored Kimages Creek wetland. Natural regeneration is occurring in this area and is likely to continue. Current levels of natural recruitment may make facilitated restoration unnecessary in the arm and along the northeast corner of the remaining dam.

Despite the low recruitment of *Taxodium distichum* in the non-tidal portion of Kimages Creek wetland, it appears to be an ideal area for managed restoration of this species based on physicochemical recruitment factors such as draw-down and light availability. Natural regeneration is likely to be limited or very slow in this area because of low proximity to adults and physical barriers to seed access. Physical access of propagules to an area is the first physiological filter to recruitment (Lambers et al. 2006). This principle in combination with our observed results may mean that facilitated restoration could be necessary in non-tidal areas for other woody species that disperse via hydrochory.

Tidal areas of the marsh seem to have natural *T. distichum* regeneration occurring but at a slower rate relative to the arm region. The larger number of adults in the arm may be a potential reason for increased regeneration, as close proximity to adults increases the number of seedlings observed. Seeds have dispersed via hydrochory throughout the tidal portions of the basin, so it seems that there is potential for much of the basin to regenerate naturally. However in much of the basin the density of seedlings currently recruiting is much lower than areas that have been identified as "natural regeneration" areas.

The reference swamp (Harris Creek) had 11 adults (DBH>10 cm) adults and *T. distichum* was the most abundant woody species present in the seed bank for this site. Despite seed rain, ideal elevation and wetland conditions, biotic competition via shading (Lambers et al. 2006), could be limiting recruitment in Harris Creek. Only two saplings and one seedling were observed within this wetland. Canopy gaps are important for enhancing environmental variability in shade limited shrub-dominated barrier island ecosystems and may play a role in establishment of species from different seral stages (Crawford and Young 1998b). Consequently, canopy gaps may be important for the recruitment of *T. distichum* in forested wetland ecosystems. Inundation depth or heavy canopy cover (Jones et al. 1989, Faulkner et al. 2009) may be limiting the recruitment of *T.* distichum in Harris Creek. Antithetically, low canopy cover and potentially more favorable hydrologic conditions in the Kimages Creek tidal restoration site may be contributing to more successful germination and recruitment.

According to this model, areas with the greatest potential for natural regeneration of this species occur where there is a combination of wetland land use classification, suitable elevation (0-5 meters), and proximity to adult *T. distichum*. Light limitation on the west shoreline of Kimages tidal marsh may be limiting recruitment as recruitment is higher on the east shore (west facing bank) of Kimages tidal marsh. Afternoon sun angle and subsequent light availability could be a limiting factor in seedling recruitment along the western shoreline (east facing aspect) because these sights are similar in tidal inundation and proximity to the James River. It should be noted here that the highest number of seedlings/saplings occurring outside of natural regeneration areas fall on the eastern side of the basin. This area also receives the most sunlight exposure throughout the course of a day. Further study would be needed to quantify light limitation and recruitment for *T. distichum* at the VCU Rice Center.

One anomalous area of seedling recruitment along the James River side of the dam was excluded from the model because it was likely affected by a silt fence installed during dam restoration efforts and falls outside of Kimages Creek riparian habitat. Over 100 seedlings were found "upstream" of the silt fence and would likely not have recruited there had the fence been absent. It is worth noting that more than one hundred seedlings located here were excluded from the model which could potentially strengthen seed rain proximity to adults. This density of recruitment at this site may indicate an ideal place to collect seed rain or seedlings to be transplanted to aid in restoration. This area was located adjacent to the relic population of adults on the southeast end of the dam near the original 2006 breach.

The current model was useful in explaining the data and yielded a raster data set for guiding restoration efforts. This map confirms that the entire basin of former Lake Charles (excluding the main channel of Kimages Creek) has the potential to be reforested with *Taxodium* *distichum*. Roughly one third of the seedlings were found in the managed restoration class. This is an encouraging sign the basin has conditions that are necessary for germination of seedlings and subsequent growth. It may also imply that tides and seasonal high water events do not reach a depth sufficient to kill the seedlings or keep seeds from germinating but are efficient at disseminating the propagules to favorable germination sites. At the current time it appears that the largest limiting factor in seedling recruitment in the basin is seed dispersal. Despite the wetland obligate classification of *T. distichum* it can grow in well drained conditions and so may survive even when it germinates under upland conditions (Havens 2004).

Currently at the VCU Rice Center natural restoration of *T. disticum* is occurring. Areas with potential for managed restoration were also identified by this study. Within these areas, seedling recruitment, germination, and growth requirements are being met. This has important management implications for the planned restoration at this site as these areas should be able to support planted saplings. These areas may also be suitable for planting other woody species that disperse via hydrochory, such as *Nyssa* spp., and species that may potentially disperse via hydrochory *L. styraciflua*, and *P. taeda* (Schneider and Sharitz 1988).

Models should be used within the scope of their original objectives and intent. In the case of the model created in this study it is important to note that the model was used to find areas for natural and artificial regeneration by explaining data collected from this site. This explanatory model has elucidated patterns of *T. distichum* recruitment at the site based upon Euclidian distance from adults, elevation and habitat type. The model should be used in accordance with uncertainties and error associated with the data sets used to make it.

A genetic study to investigate relationships between adults and seedlings may be a logical next step in researching the *T. distichum* population at the VCU Rice Center. Genetic studies may also help in determining planting methods to maintain local genetic diversity and distinct local populations (if it is discovered that any locally and genetically distinct populations exist). In future iterations of this study it would be useful to collect environmental data when mapping seedlings, saplings and adults. There seems to be a gap in the literature when addressing field conditions necessary for recruitment of *T. distichum*. General information on light and inundation are available (Mitsch and Gosselink 2007, Mitsch et al. 2009, Batzer and Sharitz 2006, Faulkner et al. 2009). Conducting studies on combined effects of shading, root competition and flooding have also been useful in understanding recruitment of this species (Jones et al. 1989). Incorporating these environmental variables into future modeling studies may be useful to increase restoration efficiency. Collecting such data on interactions between abiotic and biotic forested wetland components has been suggested as a way to improve restoration efforts (Bledsoe and Shear 2000). Mapping patterns of recruitment is important for the future of restoration efforts because two factors affecting forest dynamics are germination patterns and seedling pools (Battaglia et al. 2000). Using high resolution elevation data, such as that collected with LIDAR, will be important for improving this model.

The population of *T. distichum* appears to be replacing itself and increasing in tidal areas of the Kimages Creek wetland. In Harris Creek recruitment is low and the population does not appear to replacing itself. Recruitment in Harris Creek may be limited to gap areas but further study is required to say this definitively. All areas of the former lake basin, except in the stream channel, appear to be good candidates for either natural recruitment or facilitated regeneration of *T. distichum*. The weighted suitability model created in this study is appears to be useful for guiding *T. distichum* restoration at the VCU Rice Center.

Literature Cited

Literature Cited

Anderson C., and W. Mitsch. 2005. Effect of Pulsing on Macrophyte Productivity and Nutrient Uptake: A Wetland Mesocosm Experiment. The American Midland Naturalist. 154(2): 305-319

Bailey, D., J. Perry, and D. DeBerry. 2006. *Aeschynomene virginica* (Fabaceae) Habitat in a Tidal Marsh, James City County, Virginia. Banisteria. 27: 3-9

Baker, H. 1989. Some Aspects of the Natural History of Seed Banks. In: Leck, A., V. Parker, and R. Simpson, (eds.). 1989. Ecology of Soil Seed Banks. Pp. 9-21. Academic Press, San Diego, CA.

Ballantine, K., and R. Schneider. 2009. Fifty-Five Years of Soil Development in Restored Freshwater Depressional Wetlands. Ecological Applications. 19(6): 1467-1480

Baldwin, A., K. McKee, and I. Mendelssohn. 1996. The Influence of Vegetation, Salinity, and Inundation on Seed Banks of Oligohaline Coastal Marshes. American Journal of Botany. 83(4): 470-479

Baldwin, A., and E. Derico. 1999. The Seed Bank of a Restored Tidal Freshwater Marsh in Washington, DC. Urban Ecosystems. 3: 5-20

Baldwin, A., M. Egnotovich, and E. Clarke. 2001. Hydrologic Change and Vegetation of Tidal Freshwater Marshes: Field, Greenhouse, and Seed-Bank Experiments. Wetlands. 21(4): 519-531

Barden, L. 1987. Invasion of *Microstegium vimineum* (Poaceae), an Exotic, Annual, Shade-Tolerant, C4 Grass, into a North Carolina Floodplain. American Midland Naturalist. 118(1): 40-45

Baskin C., and J. Baskin. 1998. Seeds: Ecology, Biogeography and Evolution of Dormancy and Germination. Academic Press, San Diego, CA. 666 pp.

Battaglia, L., S. Fore, and R. Sharitz. 2000. Seedling Emergence, Survival and Size in Relation to Light and Water Availability in Two Bottomland Hardwood Species. Journal of Ecology. 88: 1041-1050

Batzer, D., and R. Sharitz. 2006. Ecology of Freshwater and Estuarine Wetlands. 1st University of California Press, Berkeley. 566 pp.

Best R., M. Wehde, and R. Linder. 1981. Spectral Reflectance of Hydrophytes. Remote Sensing of the Environment. 11: 27-35

Bledsoe, B. and T. Shear. 2000. Vegetation Along Hydrologic and Edaphic Gradients in a North Carolina Coastal Plain Creek Bottom and Implications for Restoration. Wetlands. 20(1): 126-147

Brosofske, K., J. Chen, and T. Crow. 2001. Understory vegetation and site factors: implications for a managed Wisconsin landscape.

Caballero, I., J. Olano, J. Loidi, and A. Escudero. 2008. A Model for Small-Scale Seed Bank and Standing Vegetation Connection Along Time. Oikos. 117(12): 1788-1795

Capers, R., R. Selsky, G. Bugbee, and J. White. 2007. Aquatic Plant Community Invasibility and Scale-Dependent Patterns in Native and Invasive Species Richness. Ecology. 88(12): 3135-3143

Capon, S., and M. Brock. 2006. Flooding, Soil Seed Bank Dynamics and Vegetation Resilience of a Hydrologically Variable Desert Floodplain. Freshwater Biology. 51: 206-223.

Crawford E., and D. Young. 1998a. Comparison of Gaps and Intact Shrub Thickets on an Atlantic Coast Barrier Island. American Midland Naturalist. 140: 68-77

Crawford E., and D. Young. 1998b. Spatial/Temporal Variations in Shrub Thicket Soil Seed Banks on an Atlantic Coast Barrier Island. American Journal of Botany. 85(2): 1739-1744

Crawford, E., F. Day, and R. Atkinson. 2007. Influences of Environment and Substrate Quality on Root Decomposition in Naturally Regenerating and Restored Atlantic White Cedar Wetlands. Wetlands. 27(1): 1-11

Connell, J., and R. Slatyer. 1977. Mechanisms of Succession in Natural Communities and Their Role in Community Stability and Organization. The American Naturalist. 111: 1119-1144

Connor, W., L. Inabinette, and E. Brantley. 2000. The Use of Tree Shelters in Restoring Forest Species to a Floodplain Delta: 5-Year Results. Ecological Engineering. 15: S47-S56

Cowles, H. 1899. The Ecological Relations of Vegetation on the Sand Dunes of Lake Michigan. In: eds. Real, L., and J. Brown. 1991. Foundations of Ecology. The University of Chicago Press, Chicago, Il. 28-57

Deberry, D., and J. Perry. 2000a. An Introduction to Wetland Seed Banks. Wetlands Tech Report 2

Deberry, D., and J. Perry. 2000b. An Introduction to Wetland Seed Banks. Wetlands Tech Report 4

DeBerry, D., and J. Perry. 2004. Primary Succession in a Created Freshwater Wetland. Castanea. 69(3): 185-193

Del Moral, R., and D. Wood. 1993. Early primary succession on a barren volcanic plain at Mount St. Helens, Washington. American Journal of Botany. 80(9): 981-991

De Steven, D., R. Sharitz, J. Singer, and C. Barton. 2006 Testing a Passive Revegetation Approach for Restoring Coastal Plain Depressional Wetlands. Restoration Ecology. 14(3): 452-460

Dulohery, C., R. Kolka, and M. McKevlin. 2000. Effects of a Willow Overstory on Planted Seedlings in a Bottomland Restoration. Ecological Engineering. 15: S57-S66

Duncan, W., and M. Duncan. 1987. Seaside plants of the Gulf and Atlantic Coasts. Smithsonian Institution Press, Washington D.C. 409 pp.

Dunn, C., and R. Sharitz. 1987. Revegetation of a *Taxodium-Nyssa* Forested Wetland Following Complete Vegetation Destruction. Vegetaio. 72: 151-157

Effler, R., R. Goyer, and G. Lenhard. 2006. Baldcypress and Water Tupelo Responses to Insect Defoliation and Nutrient Augmentation in Maurepas Swamp, Louisiana, USA. Forest Ecology and Management. 236: 295-304

Egghart, C. 2009. The Walter and Inger Rice Center for Environmental Life Science Through Time (Masters Thesis). Retrieved from: https://digarchive.library.vcu.edu/bitstream/handle/10156/2488/Egghart%20Rice%20Center%20Thesis%20Final%20May%2009.pdf?sequence=1

Ehrenfeld, J., and L. Toth. 1997. Restoration Ecology and the Ecosystem Perspective. Restoration Ecology. 5(4): 307-317

Fransworth, E., and L. Meyerson. 2003. Comparative Ecophysiology of Four Wetland Plant Species Along a Continuum of Invasiveness. Wetlands. 23(4): 750-762

Faulkner, S., P. Bhattarai, Y. Allen, J. Barras and G. Constant. 2009. Identifying Bald Cypress- Water Tupelo Regeneration Classes in Forested Wetlands of the Atchafalaya Basin, Louisiana. Wetlands. 29(3): 809-817

Figuerola, J., and A. Green. 2002. Dispersal of Aquatic Oranisms by Waterbirds: A Review of Past Research and Priorities for Future Studies. Freshwater Biology. 47: 483-494

Flory, S. 2010. Management of *Microstegium vimineum*Invasions and Recovery of Resident Plant Communities. Restoration Ecology. 18(1): 103-112

Glenn-Lewin, D., and E. van der Maarel. 1992. Establishment, colonization and persistence. In: Glenn-Lewin, D., R. Peet, T. Veblen (eds.). 1992. Plant Succession Theory and Prediction. Chapman and Hall. London. UK. Pp. 11-44.

Godfrey, R., and J. Wooten. 1981. Aquatic and Wetland Plants of the Southeastern United States. University of Georgia Press, Athens, GA.

Gutrich, K., Taylor, and M. Fennessy. 2009. Restoration of Vegetation Communities of Created Depressional Marshes in Ohio ad Colorado (USA): The Importance of Initial Effort for Mitigation Success. Ecological Engineering. 351-368

Hastings, A., J. Byers, J. Crooks, K. Cuddington, C. Jones, J. Lambrinos, T. Talley and W. Wilson. 2007. Ecosystem Engineering in Space and Time. Ecology Letters. 10: 153-164

Hardisky, M., M. Gross, and V. Klemas. 1986. Remote Sensing of Coastal Wetlands. BioScience. 36(7): 453-460

Hassett, B., M. Palmer, E. Bernhardt, S. Smith, J. Carr, and D. Hart. 2005. Restoring Watersheds Project by Project: Trends in Chesapeake Bay Tributary Restoration. Frontiers of Ecology and the Environment. 3(5): 259-267

Howe, H., and J. Smallwood. 1982. Ecology of Seed Dispersal. Annual Review of Ecology and Systematics. 13: 201-228

Jones, R., R. Sharitz, and K. McLeod. 1989. Effects of Flooding and Root Competition on Growth of Shaded Bottomland Hardwood Seedlings. American Midland Naturalist. 121(1): 165-175

Judd, F. and R. Lonard. 2002. Species Richness and Diversity of Brackish and Salt Marshes in the Rio Grande Delta. Journal of Coastal Research. 18(4): 751-759

Jurik, T., S. Wang, and A. van der Valk. 1994. Effects of Sediment Load on Seedling Emergence From Wetland Seed Banks. Wetlands. 14(3): 159-165

Kauffman-Axelrod, and J., S. Steinberg. 2010. Development and Application of an Automated GIS Based Evaluation to Prioritize Wetland Restoration Opportunities. Wetlands. 30: 437-448.

Kettenring, K., and D. Whigham. 2009. Seed Viability and Seed Dormancy of Non-native *Phragmites australis* in Suburbanized and forested Watersheds of the Chesapeake Bay, USA. Aquatic Botany. 91: 199-204

Klemas, V. 2001. Remote Sensing of Landscape-Level Coastal Environmental Indicators. Environmental Management. 27(1): 47-67

Lambers, H., F. Chaplin, and T. Pons. 2006 Plant Physiological Ecology. 1st. Springer Science and Business Media, LLC, New York New York. 3-6

Lang, M., and G. McCarty. 2009. LIDAR Intesity for Improved Detection of Inundation Below the Forest Canopy. Wetlands. 29(4): 1166-1178

Leck, M., and R. Simpson. 1995. Ten-Year Seed Bank and Vegetation Dynamics of a Tidal Freshwater Marsh. American Journal of Botany. 82(12): 1547-1557

Leck, M. 2003. Seed-Bank and Vegetation Development in a Created Tidal Freshwater Wetland on the Deleware River, Trenton, New Jersy, USA. Wetlands. 23(2): 310-343

Li, S., J. Lissner, I. Mendelssohn, H. Brix, B. Lorenzen, K. McKee, and S. Miao. 2010. Nutrient and growth responses of cattail (*Typha domingensis*) to redox intensity and phosphate availability. Annals of Botany. 105: 174-184

Lubchenco, J. 1978. Plant Species Diversity in a Marine Intertidal Community: Importance of Herbivore Food Preference and Algal Competitive Abilities. The American Naturalist. 112(983): 23-39

Matthews, J., C. Spyreas and A. Endress. 2009. Trajectories of Vegetation-based indicators Used to Assess Wetland Vegetation Progress. Ecological Applications. 19(8): 2093-2107

Mallik, A., 2003. Conifer Regeneration Problems in Boreal and Temperate Forests with Ericaceous Understory: Role of Disturbance, Seedbed Limitation, and Keystone Species Change. Critical Reviews in Plant Sciences. 22(3&4):341-366

McCormick, C. 1999. Mapping Exotic Vegetation in the Everglades from Large-Scale Aerial Photographs. Photogrammetric Engineering & Remote Sensing. 65(2): 179-184

McCune, B., and J. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, Oregon. 300 pp.

Middleton, B. 2000. Hydrochory, Seed Banks, and Regeneration Dynamics Along the Landscape Boundaries of a Forested Wetland. Plant Ecology. 146: 169-184

Middleton, B. 2003. Soil Seed Banks and the Potential Restoration of Forested Wetlands After Farming. Journal of Applied Ecology. 40: 1025-1034

Middleton, B. 2009a. Effects of Hurricane Katrina on the Forest Structure of *Taxodium distichum* Swamps of the Gulf Coast, USA. Wetlands 29(1): 80-87

Middleton, B. 2009b. Note: Effects of Hurricane Katrina on Tree Regeneration in *Taxodium distichum* Swamps of the Gulf Coast. Wetlands 29(1):135-141

Mitsch, W., L. Zhang, K. Stefanik, A. Nahlink, C. Anderson, B. Bernal, M. Hernandez, and K. Song. 2012. Creating Wetlands: Primary Succession, Water Quality Changes and Self-Design over 15 Years. BioScience. 62(3):237-250

Mitsch W., J. Gosselink, C. Anderson and L. Zhang. 2009. Wetland Ecosystems. 1st. John Wiley and Sons, Inc, Hoboken New Jersey. 123-129.

Mitsch W. and J. Gosselink. 2007. Wetlands. 4th John Wiley and Sons, Inc, Hoboken New Jersey. 528 pp.

Mitsch,W., X. Wu, R. Nairn, P. Weihe, N. Wang, R. Deal, and C. Boucher. 1998. Creating and Restoring Wetlands. Bioscience. 48(12): 1019-1027+1029-1030

Molles, M. 2005. Ecology: Concepts and Applications. 3rd McGraw-Hill New York, NY. 485-507

National Research Council 1992. *Restoration of aquatic ecosystems*. National Academy of Science, Washington, D.C

Noe, G., and C. Hupp. 2005. Carbon, Nitrogen and Phosphorus Accumulation in Floodplains of Atlantic Coastal Plain Rivers, USA. Ecological Applications. 15(4): 1178-1190

Noon, K. 1995. A model of wetland primary succession. Landscape and Urban Planning. 34: 97-123

Neufled, H. 1983. Effects of Light on Growth, Morphology, and Photosynthesis in Bald Cypress (*Taxodium distichum* (L.) Rich.) and Pondcypress (*T. ascendens* Brongn.) Seedlings. Bulletin of the Torrey Botanical Club. 110(1): 43-54

Nicol, J., G. Ganf, and G. Pelton. 2003. Seed Banks of a Southern Australian Wetland: the Influence of Water Regime on the Final Floristic Composition. Plant Ecology. 168: 191-205

Neill, C., M. Bezerra, R. McHorney, and C. O'Dea. 2009. Distribution, Species Composition and Management Implications of Seed Banks in New England Coastal Plain Ponds. Biological Conservation. 142: 1350-1361

Nygaard, B., and R. Ejrnaes. 2009. The Impact of Hydrology and Nutrients on Species Composition and Richness: Evidence from a Microcosm Experiment. Wetlands. 29(1): 187-195

Odum, E. 1969.The Strategy of Ecosystem Development. Science. 164: 262-270

Odum, W. 1988. Comparative Ecology of Tidal Freshwater and Salt Marshes. Annual Review of Ecology and Systematics. 19:147-176

Paine, R. 1966. Food Web Complexity and Species Diversity. The American Naturalist. 100(910): 65-75

Paine, R. 1969. A Note on Trophic Complexity and Community Stability. The American Naturalist. 103(929): 91-93

Peterson, J., and A. Baldwin. 2004. Seedling Emergence from Seed Banks of Tidal Freshwater Wetlands: Response to Inundation and Sedimentation. Aquatic Botany. 78: 243-254.

Ramsey, E., A. Rangoonwa, B. Middleton, and Z. Lu. 2009. Satellite Optical and Radar Data Used to Track Wetland Forest Impact and Short-Term Recovery from Hurricane Katrina. Wetlands. 29(1): 66-79

Roach, D. 1983. Buried Seed and Standing Vegetation in Two Adjacent Tundra Habitat, Northern Alaska. Oecologia. 60:359-364

Schneider, R., and R. Sharitz. 1986. Seed Bank Dynamics in a Southeastern Riverine Swamp. American Journal of Botany. 73(7):1022-1030

Schneider, R., and R. Sharitz. 1988. Hydrochory and Regeneration in a Bald Cypress-Water Tupelo Swamp Forest. Ecology. 69(4): 1055-1063

Schrift, A., I. Medelsshon, and M. Materne. 2008. Salt marsh restoration with sediment-slurry amendments following a drought-induced large-scale disturbance. Wetlands. 20(4): 1071-1085

Silberhorn, G. 1999. Common Plants of the Mid-Atlantic Coast. Revised Edition. The Johns Hopkins University Press, Baltimore, Maryland. 294 pp.

Sklar, F., R. Costanza, and J. Day. 1985. Dynamic spatial simulation modeling of coastal wetland habitat succession. Ecological Modelling. 29: 261-281

Soons, M. 2006. Wind Dispersal in Freshwater Wetlands: Knowledge for Conservation and Restoration. Applied Vegetation Science. 9: 271-278

Souther, R.and G. Shaffer. 2000. The Effects of Submergence and Light on Two Age Classes of BaldCypress (*Taxodium distichum*(L.) Richard) Seedlings. Wetlands. 20(4): 697-706

Suchrow S., and K. Jensen. 2010. Plant Species Responses to an Elevational Gradient in German North Sea Salt Marshes. Wetlands. 30(4):735-746

Svengsouk, L., and W. Mitsch. 2000. Dynamics of Mixtures of *Typha latifolia* and *Schoenoplectus tabernaemontani* in Nutrient-enrichment Wetland Experiments. The American Midland Naturalist. 145(2): 309-324

Swanson, M., J. Franklin, R. Beschta, C. Crisafulli, D. DellaSala, R. Hutto, D. Lindenmayer, and F. Swanson. 2011. The Forgotten Stage of Forest Succession: Early-successional Ecosystems on Forest Sites. Frontiers in Ecology and the Environment. 9(2): 117-125

Tilman, D. 1988. Plant Strategies and the Dynamics and Structure of Plant Communities. Princeton University Press. Princeton, New Jersey.

Toth, L. 2010. Restoration Response of Relict Broadleaf Marshes to Increased Water Depths. Wetlands. 30: 263-274

Urban, N., S. Davis, and N. Aumen. 1993. Fluctuations in Sawgrass and Cattail Densities in Everglades Water Conservation Area 2A Under Varying Nutrient, Hydrologic and Fire Regimes. Aquatic Botany. 46: 203-223

Uva, R., J. Neal, and J. DiTomaso. 1997. Weeds of the Northeast. Cornell University Press, Ithaca New York. 396 pp.

van der Valk, A. 1981. Succession in Wetlands: A Gleasonian Approach. Ecology. 62(3): 688-696

van der Valk, A. 1992. Establishment, colonization and persistence. In: Glenn-Lewin, D., R. Peet, and T. Veblen (eds.). 1992. Plant succession theory and prediction. Chapman and Hall. London, UK. pp. 61-92

Verhoeven, J., and B. Sorrell. 2010. Plant adaptations and microbial processes in wetlands. Annals of Botany. 105: 127

Weiher, E., I. Wisheu, P. Keddy, and D. Moore. 1996. Establishment, Persistence, and Management Implications of Experimental Wetland Plant Communities. Wetlands. 16(2):208-218

Welling, C. and R. Becker. 1990. Seed Bank Dynamics of *Lythrum salicaria* L.: Implications for Control of this Species in North America. Aquatic Botany. 38(2-3): 303-309

Xiong, S., M. Johansson, F. Hughes, A. Hayes, K. Richards, and C. Nilsson. 2003. Interactive Effects of Soil Moisture, Vegetation Canopy, Plant Litter and Seed Addition on Plant Diversity in a Wetland Community. Journal of Ecology. 91: 976-986

Yu, S., and J. Ehrenfeld. 2010. Relationships among plants, soils and microbial communities along a hydrological gradient in the New Jersey Pinelands, USA. Annals of Botany. 105: 185-196

Zedler, J. 2000. Progress in Wetland Restoration Ecology. Trends in Ecology and Evolution. 15(10):402-407

Zedler, J., and J. Callaway. 1999. Tracking Wetland Resotration: Do Mitigation Sites Follow Desired Trajectories? Restoration Ecology. 7(1):69-73

Table 1. Species presemt in standing cover for tidal and non-tidal areas of Kimages Creek wetland and Harris Creek reference swamp.The first two letters in headers represent habitat (KN = Kimages Non-tidal Marsh, KT = Kimages Tidal Marsh). The second two letters in headers represent standing cover (SC = Standing Cover).

Species	Family	KNSC	KTSC
Woody			
Acer rubrum	Aceraceae	+	+
Alnus serrulata	Betulaceae	-	+
Baccharis halimifolia	Asteraceae	-	+
Ilex opaca	Aquifoliaceae	-	+
Liquidambar styraciflua	Hamamelidaceae	+	+
Liriodendron tulipifera	Magnoliaceae	+	+
Morella cerifera	Myricaceae	+	+
Nyssa sylvatica var. *biflora*	Cornaceae	+	-
Pinus taeda	Pinaceae	+	+
Platanus occidentalis	Platanaceae	+	+
Salix nigra	Salicaceae	+	+
Taxodium distichum	Cupressaceae	-	+
Toxicodendron radicans	Anacardiaceae	+	-
Herbaceous			
Agrostis stolonifera	Poaceae	+	+
Bidens frondosa	Asteraceae	-	+
Boehmeria cylindrica	Urticaceae	+	+
Carphephorus odoratissimus	Asteraceae	+	-
Cyperus strigosus	Cyperaceae	+	+
Echinochloa crus-galli	Poaceae	+	+
Eleocharis obtusa	Cyperaceae	-	+
Eleocharis parvula	Cyperaceae	-	+
Erechtites hieraciifolia	Asteraceae	+	+
Eupatorium capillifolium	Asteraceae	+	+
Eupatorium serotinum	Asteraceae	+	-
Heteranthera reniformis	Asteraceae	-	+
Hibiscus mosheutos	Asteraceae	+	+
Hydrocotyle umbelatta	Asteraceae	-	+
Hypericum mutilum	Asteraceae	-	+
Impatiens capensis	Asteraceae	+	+
Juncus effusus	Juncaceae	+	+
Leersia orozoides	Poaceae	+	+
Lobelalia cardinalis	Campanulaceae	+	-
Ludwigia alterniflora	Onagraceae	+	+

Ludwigia palustris	Onagraceae	-	+
Ludwigia peruviana	Onagraceae	+	+
Microstegium vineum	Poaceae	+	+
Mikania scandens	Asteraceae	-	+
Murdannia keisak	Commelinaceae	+	+
Nuphar luteum	Nymphaeaceae	+	+
Peltandra virginica	Araceae	-	+
Phragmites australis	Poaceae	-	+
Phytolacca americana	Phytolaccaceae	+	-
Pilea pumila	Urticaceae	-	+
Polygonum arifolium	Polygonaceae	+	+
Polygonum hydropiperoides	Polygonaceae	+	+
Polygonum pensylvanicum	Polygonaceae	-	+
Polygonum persicaria	Polygonaceae	+	+
Polygonum punctatum	Polygonaceae	-	+
Polygonum sagitatum	Polygonaceae	+	+
Pontederia chordata	Pontederiaceae	-	+
Rhexia virginica	Melastomataceae	+	+
Saccharum giganteum	Poaceae	+	-
Sagittaria latifolia	Alismataceae	+	+
Schoenoplectus tabernaemontani	Cyperaceae	-	+
Scirpus americanus	Cyperaceae	-	+
Scirpus cyperinus	Cyperaceae	+	+
Typha angustifolia	Typhaceae	+	+
Zizania aquatica	Poaceae	-	+
Zizaniopsis miliacea	Poaceae	-	+
Total		**37**	**52**

Table 2. Aggregate community coverage by dominant species derived from area calculated using a GIS. The values here represent the combined area of communities covered with a given dominant species. Percentages are based on the ha covered in a given growing season relative to total ha mapped in that particular growing season.

Year	2009		2010		2011	
Species	ha	%	ha	%	ha	%
Agrostis stolonifera	0.2	1.20	0.6	2.72	0.9	4.55
Erechitites hieracifolia	0.4	2.18	1.1	5.40	0.0	0.00
Bohemia cylindrica	0.0	0.00	<0.1	0.17	0.0	0.00
Heteranthea reniformes	1.1	5.62	0.0	0.00	0.0	0.00
Juncus effusus	0.0	0.00	0.0	0.00	0.2	0.84
Leersia oryzoides	5.3	27.14	5.5	26.97	5.6	28.17
Microstegium vimineum	0.3	1.59	0.1	0.69	0.4	1.86
Hibiscus mosheutos	0.0	0.00	0.0	0.00	0.2	0.80
Murdannia keisak	1.9	9.83	2.8	13.69	3.6	18.11
Polygonum punctatum	0.8	4.25	0.1	0.55	0.0	0.00
Polygonum sagittatum	0.3	1.73	0.6	2.83	0.5	2.26
Polygonum hydropiperoides	0.0	0.00	<0.1	0.18	<0.1	0.00
Sagitaria latifolia	0.0	0.00	0.1	0.59	0.0	0.00
Typha angustifolia	9.0	46.46	9.4	46.21	8.7	43.41
Total area mapped	19.4		20.3		20.0	

Table 3. Sorensen dissimiarity scores for transect cover in tidal and non-tidal areas between 2010 and 2011. Habitat designations are by year and tidal or non-tidal status. Years precede tidal or non-tidal status. Tidal or non-tidal status is denoted as follows: KT = Tidal, KN= non-tidal. Habitat based transect comparisson is based on aggregate transects from tidal or nontidal areas in a given year. Complete year transects are the aggregate coverage of all transects for a given year.

Transect	Transect			
Habitat	2010 KN	2011KN	2010 KT	2011 KT
2010 KN	-	0.3152	0.6167	0.5688
2011KN	0.3152	-	0.6224	0.6236
2010 KT	0.6167	0.6224	-	0.1436
2011 KT	0.5688	0.6245	0.1436	-
Year	2011			
2010	0.149			

Table 4. Aggregate transect cover of species by year in tidal and non-tidal areas. Values represented here are the results of combined transects based on a given year and tidal or non-tidal status. Tidal or non-tidal status: KN = Non-tidal, KT = Tidal. Year precedes tidal or non-tidal designation.

Species	2010 KN		2011 KN		2010 KT		2011 KT	
	Meters	%	Meters	%	Meters	%	Meters	%
Acer rubrum	0.475	0.064	9.676	1.219	0.000	0.000	1.208	0.152
Agrostis stolinifera	21.923	2.943	4.210	0.530	13.219	1.775	4.713	0.594
Algal spp.	0.425	0.057	0.000	0.000	0.000	0.000	0.000	0.000
Alnus serrulata	0.000	0.000	0.000	0.000	0.000	0.000	0.220	0.028
Erechitites hieracifolia	32.142	4.315	0.030	0.004	0.670	0.090	0.000	0.000
Bidens frodrosa	0.000	0.000	0.000	0.000	0.102	0.014	5.688	0.716
Bohemia cylindrica	2.250	0.302	2.335	0.294	0.635	0.085	0.008	0.001
Pilea pumila	0.000	0.000	0.000	0.000	0.000	0.000	0.080	0.010
Cyperus strigosus	0.100	0.013	0.100	0.013	0.000	0.000	0.000	0.000
Carphrphorus odoratissimus	0.495	0.066	1.220	0.154	0.250	0.034	0.038	0.005
Eleocharis parvula	0.000	0.000	0.000	0.000	0.000	0.000	2.670	0.336
Echinochloa crusgalli	0.020	0.003	0.000	0.000	0.000	0.000	0.000	0.000
Eupatorium capillifolium	0.000	0.000	0.320	0.040	0.000	0.000	0.000	0.000
Hydrocotle umbelatta	0.000	0.000	0.220	0.028	0.000	0.000	0.000	0.000
Juncus effusus	38.726	5.199	65.836	8.292	3.655	0.491	1.093	0.138
Impatiens capensis	0.000	0.000	0.245	0.031	0.000	0.000	0.000	0.000
Leersia orozoides	141.508	18.997	110.664	13.937	106.252	14.264	170.515	21.475
Liquidambar styraciflua	1.615	0.217	5.786	0.729	0.325	0.044	1.300	0.164
Ludwigia palustris	0.000	0.000	0.000	0.000	2.000	0.268	2.670	0.336
Ludwigia peruviana	0.000	0.000	0.000	0.000	2.679	0.360	1.830	0.230
Mikania scandens	0.200	0.027	0.000	0.000	0.000	0.000	0.000	0.000
Microstegium vineum	6.680	0.897	37.072	4.669	0.000	0.000	2.323	0.293
Morella cerifera	0.000	0.000	0.850	0.107	0.200	0.027	0.113	0.014
Hibiscus mosheutos	0.475	0.064	0.420	0.053	0.000	0.000	0.000	0.000
Murdania keisak	12.266	1.647	17.413	2.193	124.510	16.715	106.132	13.367
Open Water	0.000	0.000	1.360	0.171	0.000	0.000	7.060	0.889
Peltandra virginica	0.000	0.000	0.000	0.000	0.000	0.000	0.200	0.025
Pinus taeda	0.000	0.000	0.075	0.009	0.000	0.000	0.000	0.000
Polygonum arifolium	0.000	0.000	0.000	0.000	0.000	0.000	0.125	0.016
Pontederia chordata	0.000	0.000	0.000	0.000	0.000	0.000	0.160	0.020
Polygonum hydropiperoides	2.383	0.320	4.569	0.575	0.636	0.085	0.249	0.031
Polygonum Sagitatum	4.879	0.655	10.835	1.365	0.400	0.054	0.260	0.033
Sagitaria latifolia	0.000	0.000	0.100	0.013	0.050	0.007	0.558	0.070
Salix nigra	0.000	0.000	0.960	0.121	0.552	0.074	0.480	0.060
Scirpus cyperinus	0.767	0.103	0.723	0.091	1.530	0.205	0.848	0.107
Schoenoplectus tabernaemontani	0.000	0.000	0.000	0.000	0.000	0.000	0.400	0.050
Hypericum mutilum	0.000	0.000	0.115	0.014	0.000	0.000	0.000	0.000
Platanus occidentalis	0.000	0.000	0.000	0.000	0.100	0.013	0.500	0.063

Typha angustifolia	3.678	0.494	7.500	0.945	216.136	29.015	197.284	24.846
Un-identified Aster	0.000	0.000	0.068	0.009	0.000	0.000	0.040	0.005
Zizania aquatica	0.000	0.000	0.000	0.000	0.000	0.000	1.355	0.171
Zizaniopsis miliacea	0.000	0.000	0.000	0.000	0.000	0.000	1.200	0.151

Table 5. Species present (+) in soil seed banks and standing cover for the four wetland habitats sampled. The first two letters in headers represent habitat (KM = Kimages Mudflat, KN = Kimages Non-tidal Marsh, KT = Kimages Tidal Marsh, HC = Harris Creek). The second two letters in headers represent seed bank or standing cover (SB = Seed Bank, SC = Standing Cover).

Species	Family	KM SB	KT SB	KN SB	HC SB	HC SC	KN SC	KT SC	KM SC
Woody									
Acer rubrum	Aceraceae	-	-	-	-	+	+	+	-
Ailanthus altissima	Simaroubaceae	-	-	-	+	-	-	-	-
Albizia julibrissin	Fabaceae	-	-	-	+	-	-	-	-
Alnus serrulata	Betulaceae	-	-	-	-	+	-	+	-
Baccharis halimifolia	Asteraceae	-	-	-	-	-	-	+	-
Celtis occidentalis	Ulmaceae	-	-	-	+	-	-	-	-
Fraxinus pennsylvanica	Oleaceae	-	-	-	+	+	-	-	-
Ilex opaca	Aquifoliaceae	-	-	-	-	-	-	+	-
Ligustrum sinense	Oleaceae	-	-	-	+	+	-	-	-
Lindera benzoin	Lauraceae	-	-	-	-	+	-	-	-
Liquidambar styraciflua	Hamamelidaceae	-	-	-	+	+	+	+	-
Liriodendron tulipifera	Magnoliaceae	-	-	-	+	+	+	+	-
Morella cerifera	Myricaceae	-	-	-	-	-	+	+	-
Nyssa sylvatica var. *biflora*	Cornaceae	-	-	-	+	+	+	-	-
Pinus taeda	Pinaceae	-	-	-	-	+	+	+	-
Platanus occidentalis	Platanaceae	-	-	-	+	+	+	+	-
Salix nigra	Salicaceae	-	-	-	-	-	+	+	-
Smilax rotundifolia	Smilacaceae	-	-	-	-	+	-	-	-
Taxodium distichum	Cupressaceae	-	-	-	+	+	-	+	-
Toxicodendron radicans	Anacardiaceae	-	-	-	-	+	+	-	-
Herbaceous									
Acorus calamus	Acoraceae	-	-	-	-	+	-	-	-
Agrostis stolonifera	Poaceae	+	+	+	-	-	+	+	-
Andropogon virginicus	Poaceae	-	-	+	+	-	-	-	-
Anthemis cotula	Asteraceae	-	-	+	-	-	-	-	-
Aster pilosus	Asteraceae	-	+	-	+	-	-	-	-

Bidens frondrosa	Asteraceae	+	+	-	+	-	-	+	-
Bignonia capreolata	Bignoniaceae	-	-	-	-	+	-	-	-
Bohemia cylindrica	Urticaceae	+	+	+	+	-	+	+	-
Carphephorus odoratissimus	Asteraceae	+	-	-	+	-	+	-	-
Chasmanthium latifolium	Poaceae	-	-	-	+	-	-	-	-
Commelina communis	Commelinaceae	-	-	-	+	-	-	-	-
Coronopus didymus	Brassicaceae	-	-	-	+	-	-	-	-
Cyperus strigosus	Cyperaceae	+	+	+	+	-	+	+	-
Diodia virginiana	Rubiaceae	-	-	+	-	-	-	-	-
Echinochloa crusgalli	Poaceae	-	-	+	+	-	+	+	-
Eclipta prostrata	Asteraceae	-	-	-	+	-	-	-	-
Eleocharis obtusa	Cyperaceae	-	+	+	-	-	-	+	-
Eleocharis parvula	Cyperaceae	-	+	+	-	-	-	+	-
Erechitites hieracifolia	Asteraceae	+	+	+	+	-	+	+	-
Eupatorium capillifolium	Asteraceae	-	+	+	+	-	+	+	-
Eupatorium serotinum	Asteraceae	-	-	+	-	-	+	-	-
Heteranthia reniformes	Asteraceae	-	-	-	-	-	-	+	-
Hibiscus mosheutos	Asteraceae	-	-	-	-	-	+	+	-
Hydrocotle umbelatta	Asteraceae	-	-	-	-	-	-	+	-
Hypericum mutilum	Asteraceae	+	-	-	+	-	-	+	-
Impatiens capensis	Asteraceae	-	-	-	-	+	+	+	-
Ipomoea purpurea	Convolvulaceae	-	-	-	+	-	-	-	-
Iris pseudacorus	Iridaceae	-	-	-	-	+	-	-	-
Juncus effusus	Juncaceae	+	+	+	+	-	+	+	-
Leersia orozoides	Poaceae	+	+	+	+	-	+	+	-
Lobelalia cardinalis	Campanulaceae	-	-	-	-	-	+	-	-
Ludwigia alterniflora	Onagraceae	+	+	+	+	-	+	+	-
Ludwigia palustris	Onagraceae	+	+	+	+	-	-	+	-
Ludwigia peruviana	Onagraceae	+	+	-	+	-	+	+	-
Microstegium vineum	Poaceae	-	+	+	+	-	+	+	-
Mikania scandens	Asteraceae	-	-	-	-	-	-	+	-
Murdania keisak	Commelinaceae	+	+	+	+	+	+	+	-
Nuphar luteum	Nymphaeaceae	-	-	-	-	-	+	+	-

Panicum virgatum	Poaceae	-	-	-	-	+	-	-	-
Peltandra virginica	Araceae	-	-	-	-	+	-	+	-
Phragmites australis	Poaceae	-	-	-	-	-	-	+	-
Phytolacca americana	Phytolaccaceae	-	-	-	+	-	+	-	-
Pilea pumila	Urticaceae	-	-	-	+	-	-	+	-
Polygonum arifolium	Polygonaceae	-	-	-	+	+	+	+	-
Polygonum hydropiperoides	Polygonaceae	+	+	+	+	+	+	+	-
Polygonum pensylvanicum	Polygonaceae	+	-	-	-	+	-	+	-
Polygonum persicaria	Polygonaceae	-	-	-	-	+	+	+	-
Polygonum punctatum	Polygonaceae	-	-	-	-	-	-	+	-
Polygonum sagitatum	Polygonaceae	-	-	+	+	+	+	+	-
Pontederia chordata	Pontederiaceae	-	-	-	-	+	-	+	-
Portulaca oleracea	Portulacaceae	-	-	+	-	-	-	-	-
Rhexia virginica	Melastomataceae	+	-	+	-	-	+	+	-
Saccharum giganteum	Poaceae	-	-	-	-	-	+	-	-
Sagitaria latifolia	Alismataceae	-	-	-	-	+	+	+	-
Saururus cernuus	Saururaceae	-	-	-	-	+	-	-	-
Schoenoplectus tabernaemontani	Cyperaceae	-	-	-	-	-	-	+	-
Scirpus americanus	Cyperaceae	-	-	-	-	-	-	+	-
Scirpus cyperinus	Cyperaceae	-	+	-	+	-	+	+	-
Setaria geniculata	Poaceae	-	-	+	+	-	-	-	-
Typha angustifolia	Typhaceae	+	-	+	+	-	+	+	-
Zizania aquatica	Poaceae	-	-	-	-	-	-	+	-
Zizaniopsis miliacea	Poaceae	-	-	-	-	+	-	+	-
Unidentified aster		-	-	-	+	-	-	-	-
Unidentified herbaceous		+	+	+	+	-	-	-	-
Unidentified graminoid		-	-	-	+	-	-	-	-
Unidentified sedge		-	-	-	+	-	-	-	-

Table 6. Mean seedling density (#/m^2) ± one standard error in seed banks by species across habitats. Habitats are designated as follows: KM = Kimages Mudflat, KN = Kimages Non-tidal Marsh, KT = Kimages Tidal Marsh, HC = Harris Creek. Different letters denote statistically significant differences (ANOVA and Tukey HSD *post hoc* test α=0.05).

Species	HC	KM	KT	KN
Woody				
Ailanthus altissima	0.4 ± 0.4	0 ± 0	0 ± 0	0 ± 0
Albizia julibrissin	0.4 ± 0.4	0 ± 0	0 ± 0	0 ± 0
Celtis occidentalis	0.2 ± 0.2	0 ± 0	0 ± 0	0 ± 0
Fraxinus pennsylvanica	1 ± 0.45	0 ± 0	0 ± 0	0 ± 0
Ligustrum sinense	0.2 ± 0.2	0 ± 0	0 ± 0	0 ± 0
Liquidambar styraciflua	1 ± 0.81	0 ± 0	0 ± 0	0 ± 0
Liriodendron tulipifera	0.2 ± 0.2	0 ± 0	0 ± 0	0 ± 0
Nyssa sylvatica var. *biflora*	1 ± 0.34	0 ± 0	0 ± 0	0 ± 0
Platanus occidentalis	2.4 ± 1.03	0 ± 0	0 ± 0	0 ± 0
Taxodium distichum	3.4 ± 1.84	0 ± 0	0 ± 0	0 ± 0
Herbaceous				
Agrostis stolonifera	0 ± 0	24.2 ± 10.11	5.2 ± 5.2	51 ± 27.34
Andropogon virginicus	0.2 ± 0.2	0 ± 0	0 ± 0	4.4 ± 4.4
Anthemis cotular	0 ± 0	0 ± 0	0 ± 0	0.2 ± 0.2
Aster pilosus	0.4 ± 0.27	0 ± 0	6 ± 6	0 ± 0
Bidens frondrosa	1.4 ± 0.67	0.2 ± 0.2	1 ± 1	0 ± 0
Boehemia cylindrica	15 ± 4.42[a]	0.4 ± 0.27[b]	2.4 ± 0.84[b]	5 ± 2.16[b]
Carphrphorus odoratissimus	0.2 ± 0.2	1.4 ± 1.4	0 ± 0	0 ± 0
Chasmanthium latifolium	2.4 ± 1.03	0 ± 0	0 ± 0	0 ± 0
Commelina communis	1.4 ± 1.2	0 ± 0	0 ± 0	0 ± 0
Coronopus didymus	1.4 ± 1.2	0 ± 0	0 ± 0	0 ± 0
Cyperus strigosus	11.4 ± 5.2[b]	291.2 ± 81.8[a]	6.2 ± 2.98[b]	9.2 ± 3.66[b]
Diodia virginiana	0 ± 0	0 ± 0	0 ± 0	1.2 ± 1
Echinochloa crusgalli	7.4 ± 3.96	0 ± 0	0 ± 0	10.4 ± 8.26
Eclipta prostrata	0.4 ± 0.4	0 ± 0	0 ± 0	0 ± 0
Eleocharis obtusa	0 ± 0	0 ± 0	213.6 ± 180.66	98.6 ± 58.56
Eleocharis parvula	0 ± 0	0 ± 0	0.2 ± 0.2	0.4 ± 0.4
Erechitites hieracifolia	1.8 ± 0.56[b]	1.4 ± 1.4[b]	0.4 ± 0.27[b]	33 ± 17.23[a]
Eupatorium capillifolium	0.6 ± 0.43	0 ± 0	2.4 ± 2.4	1 ± 0.62
Eupatorium serotinum	0 ± 0	0 ± 0	0 ± 0	0.4 ± 0.4
Hypericum mutilum	1.4 ± 0.85	0.2 ± 0.2	0 ± 0	0 ± 0
Ipomea purporea	0.8 ± 0.8	0 ± 0	0 ± 0	0 ± 0
Juncus effusus	3.2 ± 2.22[b]	18.8 ± 8.09[b]	1.4 ± 1.4[b]	250.6 ± 111.74[a]
Leersia orozoides	178.2 ± 58.14[b]	76.6 ± 32.85[b]	189.8 ± 55.35[b]	544.8 ± 141.48[a]

Ludwigia alterniflora	0.2 ± 0.2	2 ± 0.9	1.2 ± 0.8	0.4 ± 0.27
Ludwigia palustris	76.8 ± 30.75[b]	349.8 ± 131.48[ab]	637 ± 258.91[a]	53.2 ± 18.1[b]
Ludwigia peruviana	0.4 ± 0.27	2 ± 0.9	1.4 ± 0.53	0 ± 0
Microstegium vineum	1.6 ± 1.23	0 ± 0	8.2 ± 5.49	6.2 ± 4.97
Murdania keisak	237.6 ± 101.61[ab]	9.4 ± 6.01[b]	660 ± 244.22[a]	10.8 ± 4.95[b]
Phytolacca americanna	0.2 ± 0.2	0 ± 0	0 ± 0	0 ± 0
Pilea pumila	20.2 ± 13.16	0 ± 0	0 ± 0	0 ± 0
Polygonum arifolium	7.2 ± 2.07	0 ± 0	0 ± 0	0 ± 0
Polygonum hydropiperoides	1.4 ± 1[b]	10.4 ± 3.71[b]	7.2 ± 2.93[b]	45.4 ± 16.01[a]
Polygonum pensylvanicum	0 ± 0	0.2 ± 0.2	0 ± 0	0 ± 0
Polygonum sagitatum	0.4 ± 0.27	0 ± 0	0 ± 0	19.8 ± 9.57
Portulaca oleracea	0 ± 0	0 ± 0	0 ± 0	0.2 ± 0.2
Rhexia virginica	0 ± 0	0.4 ± 0.27	0 ± 0	2.2 ± 2.2
Scirpus cyperinus	15.2 ± 15.2	0 ± 0	1.4 ± 1.04	0 ± 0
Seteria geniculata	1 ± 1	0 ± 0	0 ± 0	23 ± 23
Typha angustifolia	0.4 ± 0.4	32.4 ± 9.96	0 ± 0	0.6 ± 0.6
Unidentified Aster	2 ± 1.08	0 ± 0	0 ± 0	0 ± 0
Unidentified Species	0.8 ± 0.62	29 ± 16.49	4.8 ± 2.62	14.4 ± 6.63
Unidentified Grammanoid	12 ± 12	0 ± 0	0 ± 0	0 ± 0
Unidentified Sedge	6 ± 4.27	0 ± 0	0 ± 0	0 ± 0

Table 7: Sorensens dissimilarity index scores between seed banks and standing cover for all habitats sampled. The first two letters in headers represent habitat (KM = Kimages Mudflat, KN = Kimages Non-tidal Marsh, KT = Kimages Tidal Marsh, HC = Harris Creek). The second two letters in headers represent seed bank or standing cover (SB = Seed Bank, SC = Standing Cover).

Habitat and Community (Seed Bank or Standing Cover)	Habitat and Community (Seed Bank or Standing Cover)							
	KMSB	KTSB	KNSB	HCSB	HCSC	KNSC	KTSC	KMSC
KMSB	-	0.2973	0.3953	0.5238	0.8723	0.5273	0.5362	-
KTSB	0.2973	-	0.3182	0.5	0.9167	0.5357	0.5143	-
KNSB	0.3953	0.3182	-	0.5143	0.8889	0.4839	0.5263	-
HCSB	0.5238	0.5	0.5143	-	0.7568	0.4634	0.5	-
HCSC	0.8723	0.9167	0.8889	0.7568	-	0.5758	0.55	-
KNSC	0.5273	0.5357	0.4839	0.4634	0.5758	-	0.3182	-
KTSC	0.5362	0.5143	0.5263	0.5	0.55	0.3182	-	-
KMSC	-	-	-	-	-	-	-	-

Table 8. Multiple response permutation procedure seed bank community compostion results. Habitat designations are as follows: Harris Creek = HC, Kimages Creek tidal mudflat = KM Kimages Creek tidal marsh = KT, Kimages Creek non-tidal marsh = KN.

Pairwise Comparrison	A	p	Corrected p
HC vs. KM	0.15	$p<0.01$	$p<0.01$
HC vs. KT	0.02	0.12	0.75
HC vs. KN	0.08	$p<0.01$	$p<0.01$
KM vs. KT	0.12	$p<0.01$	$p<0.01$
KM vs. KN	0.15	$p<0.01$	$p<0.01$
KT vs. KN	0.09	$p<0.01$	$p<0.01$

Figure Captions

Figure 1. Locations of transects and approximate locations of seed bank sample collection. Transects are classified by tidal or non-tidal. Seed bank collection points are classified by habitat.

Figure 2. Vegetative community cover from 2009 within the restored basin. Communities are represented on this map based upon the top dominant aquatic macrophyte species.

Figure 3. Vegetative community cover from 2010 within the restored basin. Communities are represented on this map based upon the top dominant aquatic macrophyte species.

Figure 4. Vegetative community cover from 2011 within the restored basin. Communities are represented on this map based upon the top dominant aquatic macrophyte species.

Figure 5. Aggregate community coverage by dominant macrophyte species over the three growing seasons sampled (2009-2011).

Figure 6. Line intercept transect species coverage for the 2010 and 2011 growing seasons.

Figure 7. Species richness of soil seed banks and standing cover for the four habitats sampled. The first two letters in headers represent habitat (KM = Kimages Mudflat, KN = Kimages Non-tidal Marsh, KT = Kimages Tidal Marsh, HC = Harris Creek). The second two letters in headers represent seed bank or standing cover (SB = Seed Bank, SC = Standing Cover).

Figure 8. Seed bank diversity, calculated using a Shannon-Wiener Index, across the four wetland habitats. The first two letters in headers represent habitat (KM = Kimages Mudflat, KN = Kimages Non-tidal Marsh, KT = Kimages Tidal Marsh, HC = Harris Creek).

Figure 9. Separation of seed bank competition into three habitat based groups in species space by the NMS ordination of seed bank samples across habitats..

Figure 10. *Taxodium distichum* restoration areas within Kimages Creek, Harris Creek and along the James River shoreline at the VCU Rice Center. The Kimages Creek wetland has areas where *T. distichum* may potentially regenerate naturally and where artificial regeneration is likely to be successful.

Figure 11. Histogram shows the number of individuals in each DBH class for *T. distichum* individuals found in Harris Creek. Individuals with DBH $\geq$ 10cm are considered adults.

Figure 12. Histogram shows the number of individuals in each DBH class for *T. distichum* individuals found in Kimages Creek wetland. Individuals with DBH $\geq$ 10cm are considered adults.

Figure 13. Area cover by each restoration class on Rice Center property. This includes Kimages Creek, Harris Creek and James River shoreline.

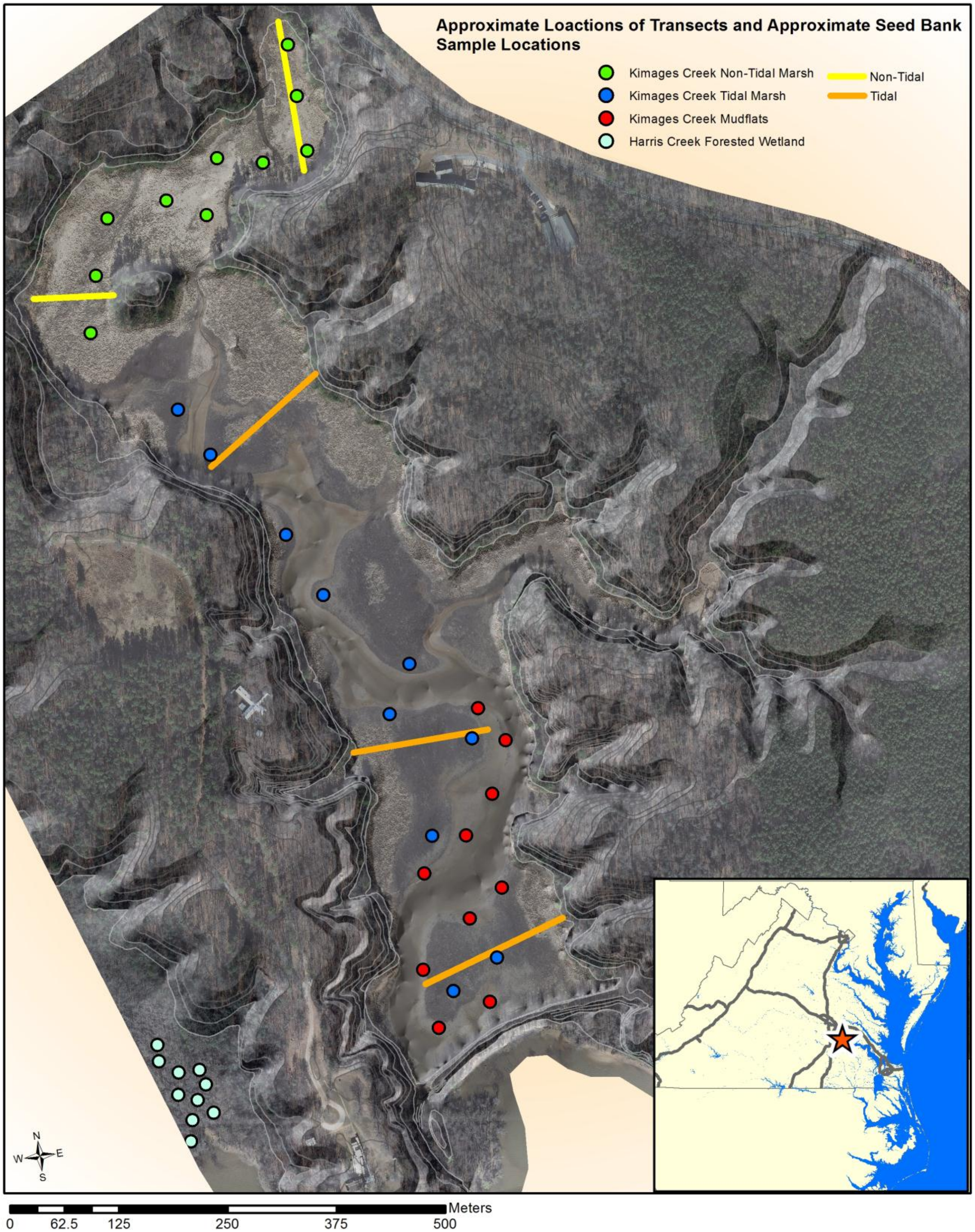

Approximate Loactions of Transects and Approximate Seed Bank Sample Locations
Kimages Creek Non-Tidal Marsh
Kimages Creek Tidal Marsh
Kimages Creek Mudflats
Harris Creek Forested Wetland
Non-Tidal
Tidal
N
W
E
S
Meters
0
62.5
125
250
375
500

Figure 1

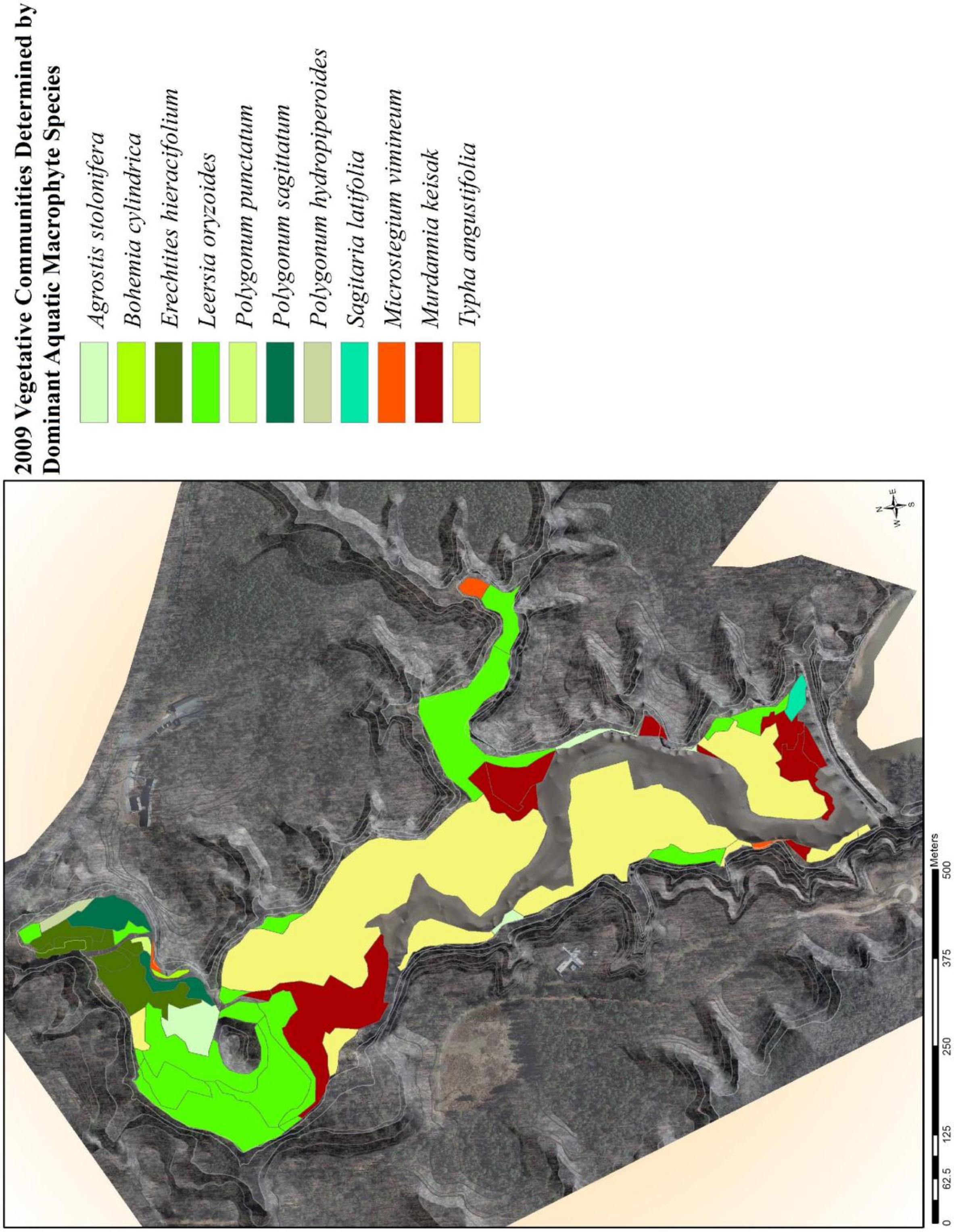

2009 Vegetative Communities Determined by
Dominant Aquatic Macrophyte Species
Agrostis stolonifera
Bohemia cylindrica
Erechtites hieracifolium
Leersia oryzoides
Polygonum punctatum
Polygonum sagittatum
Polygonum hydropiperoides
Sagitaria latifolia
Microstegium vimineum
Murdannia keisak
Typha angustifolia
Meters
0
62.5
125
250
375
500

Figure 2

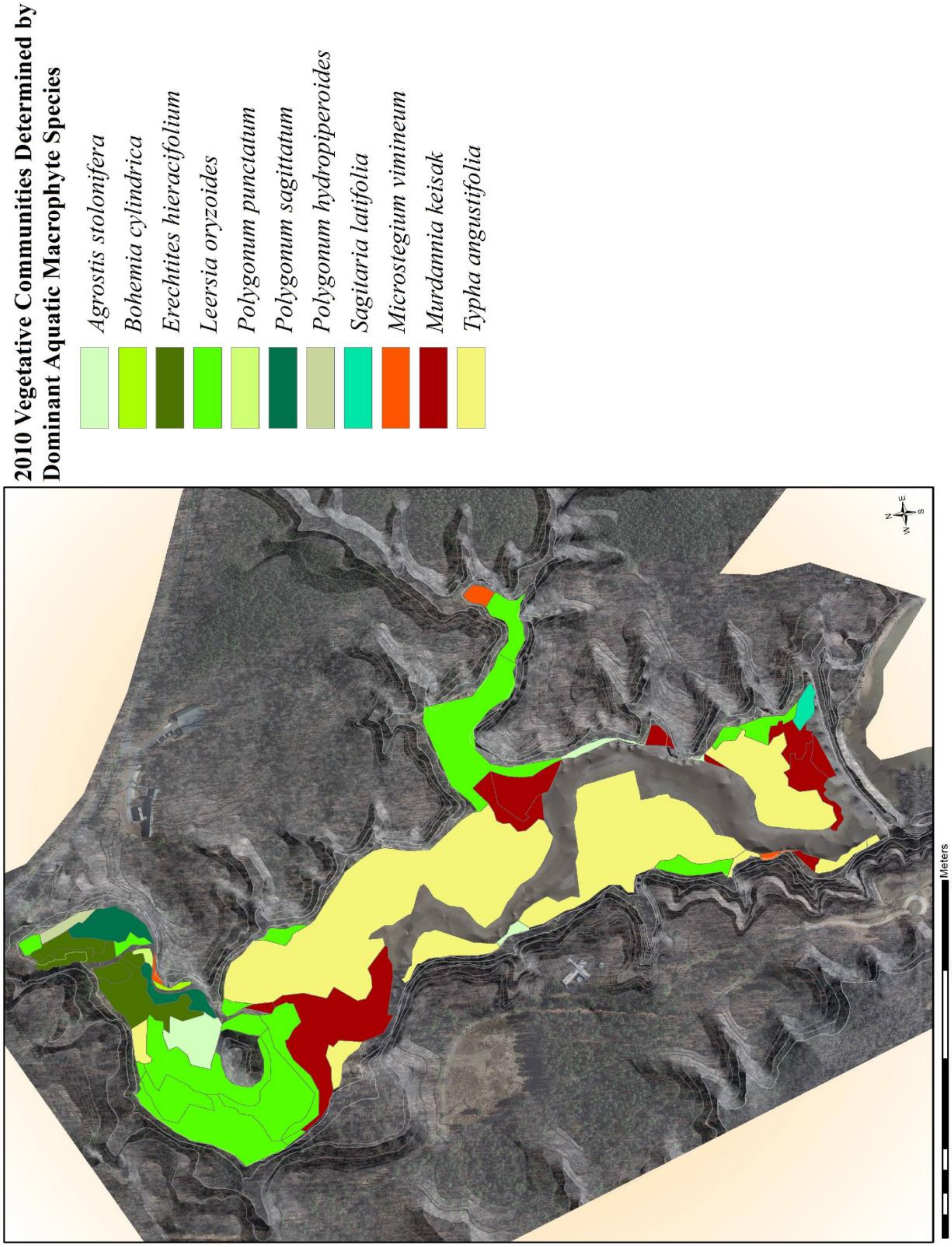

2010 Vegetative Communities Determined by Dominant Aquatic Macrophyte Species
Agrostis stolonifera
Bohemia cylindrica
Erechtites hieracifolium
Leersia oryzoides
Polygonum punctatum
Polygonum sagittatum
Polygonum hydropiperoides
Sagitaria latifolia
Microstegium vimineum
Murdannia keisak
Typha angustifolia
N
E
S
W
Meters
0
62.5
125
250
375
500

Figure 3

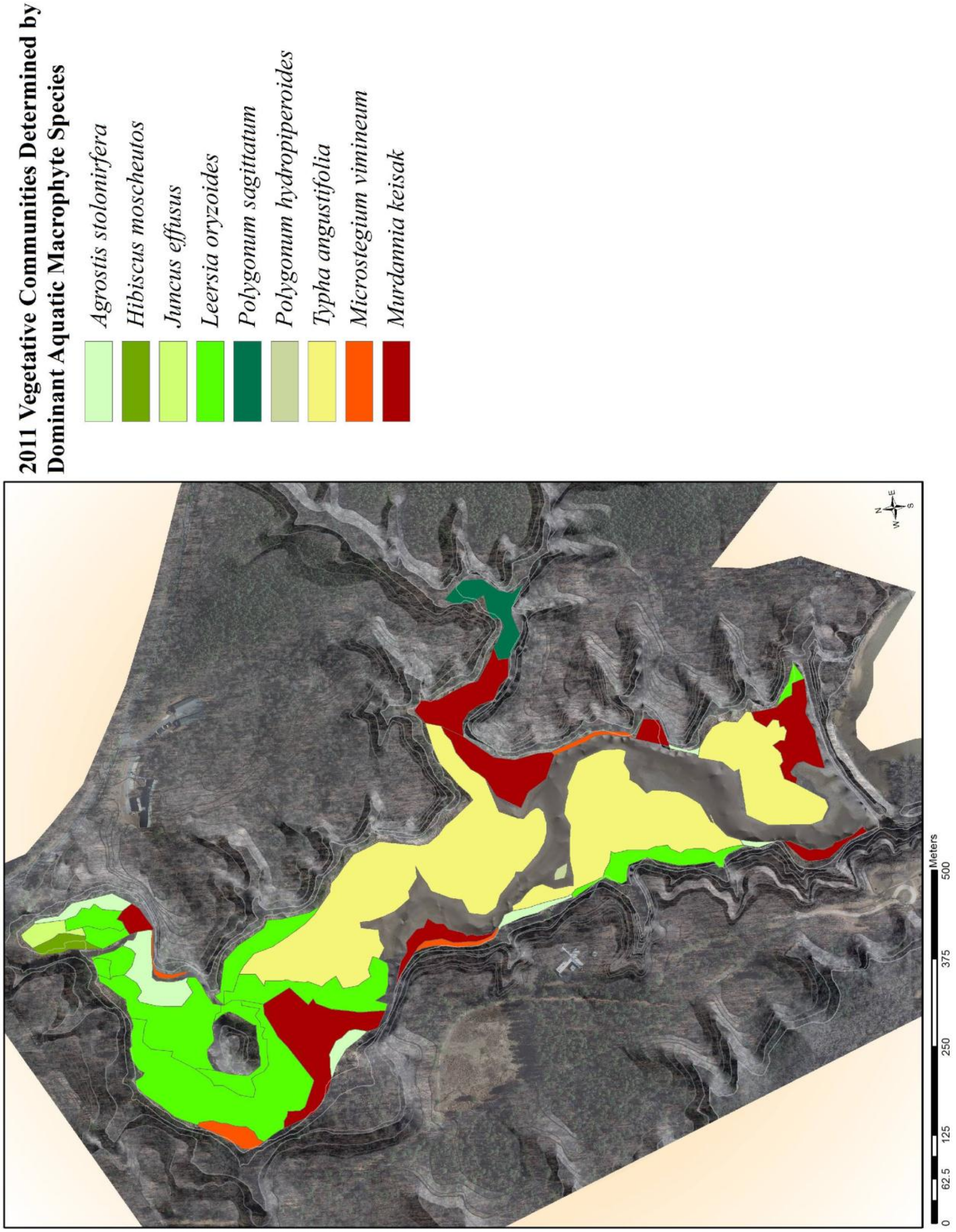
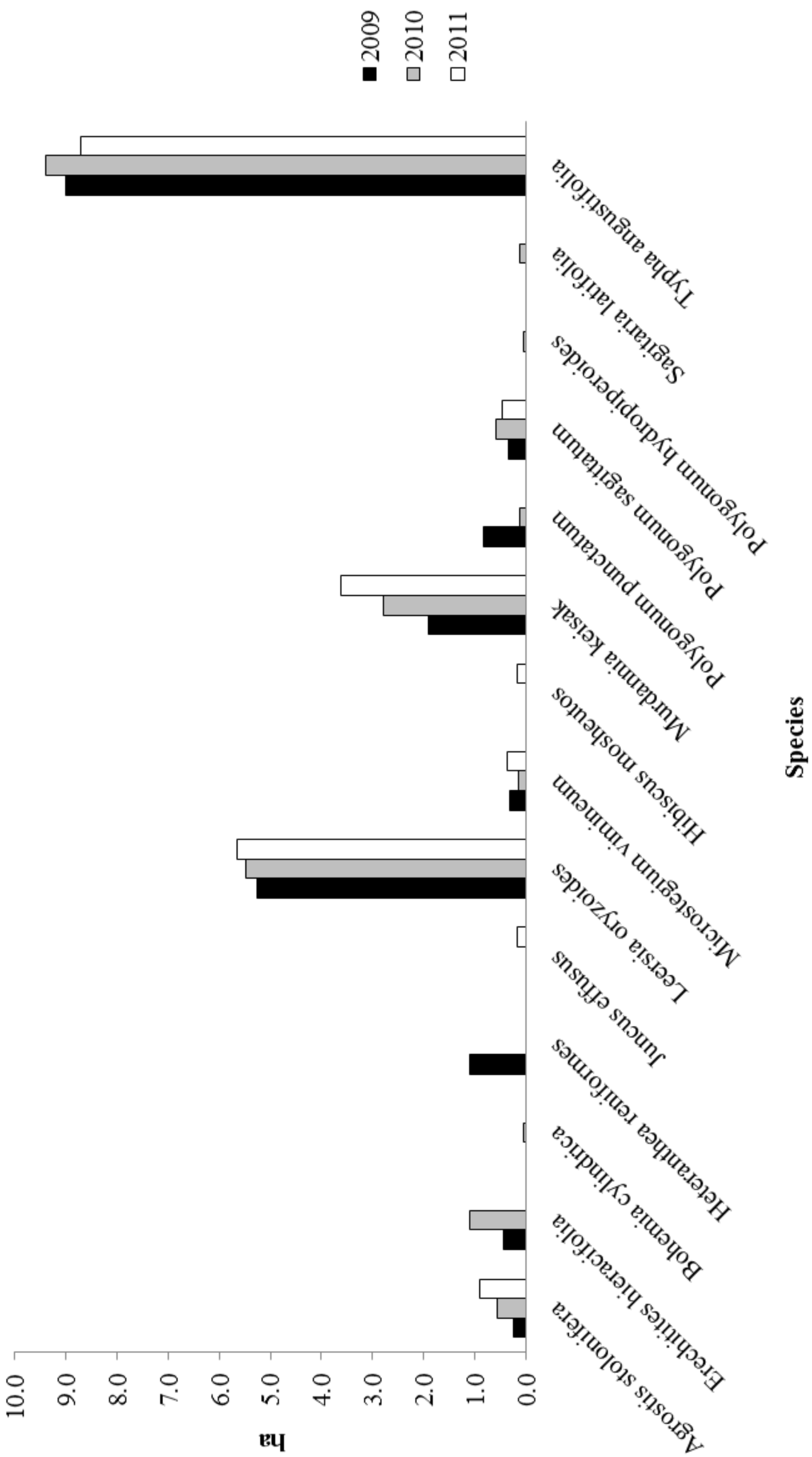
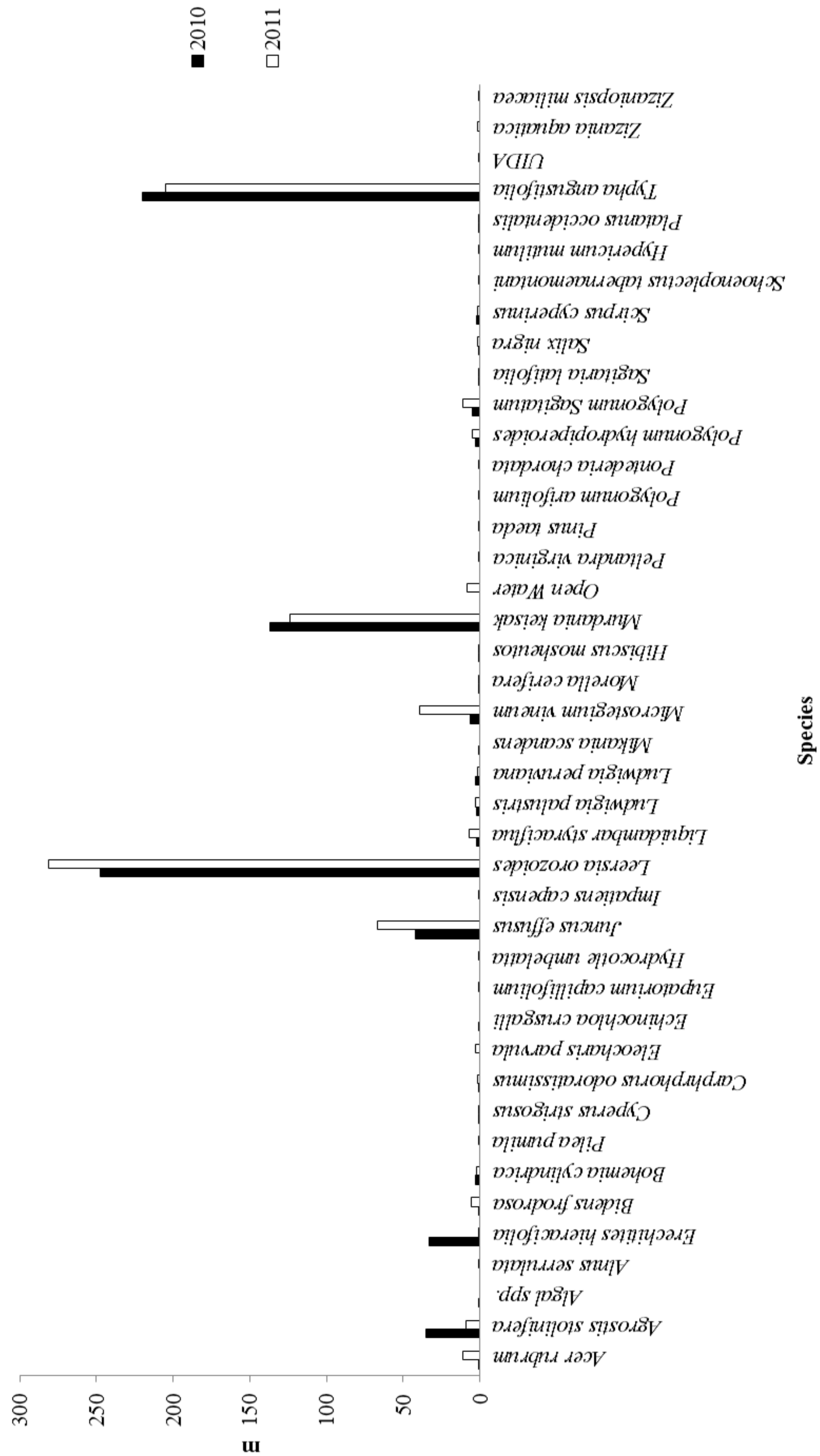

2011 Vegetative Communities Determined by
Dominant Aquatic Macrophyte Species
Agrostis stolonirfera
Hibiscus moscheutos
Juncus effusus
Leersia oryzoides
Polygonum sagittatum
Polygonum hydropiperoides
Typha angustifolia
Microstegium vimineum
Murdannia keisak
Meters
0
62.5
125
250
375
500
N
E
S
W

Figure 4



2009
2010
2011
ha
0.0
1.0
2.0
3.0
4.0
5.0
6.0
7.0
8.0
9.0
10.0
Agrostis stolonifera
Erechtites hieraciifolia
Bohemia cylindrica
Heteranthea reniformes
Juncus effusus
Leersia oryzoides
Microstegium vimineum
Hibiscus mosheutos
Murdannia keisak
Polygonum punctatum
Polygonum sagitatum
Polygonum hydropiperoides
Sagitaria latifolia
Typha angustifolia
Species

Figure 5



2010
2011
Acer rubrum
Agrostis stolinifera
Algal spp.
Alnus serrulata
Erechtites hieracifolia
Bidens frodrosa
Bohemia cylindrica
Pilea pumila
Cyperus strigosus
Carphephorus odoratissimus
Eleocharis parvula
Echinochloa crusgalli
Eupatorium capillifolium
Hydrocotle umbelata
Juncus effusus
Impatiens capensis
Leersia orozoides
Liquidambar styraciflua
Ludwigia palustris
Ludwigia peruviana
Mikania scandens
Microstegium vineum
Morella cerifera
Hibiscus mosheutos
Murdania keisak
Open Water
Peltandra virginica
Pinus taeda
Polygonum arifolium
Pontederia chordata
Polygonum hydropiperoides
Polygonum Sagitatum
Sagitaria latifolia
Salix nigra
Scirpus cyperinus
Schoenoplectus tabernaemontani
Hypericum mutilum
Platanus occidentalis
Typha angustifolia
UIDA
Zizania aquatica
Zizaniopsis miliacea
Species
m
0
50
100
150
200
250
300

Figure 6

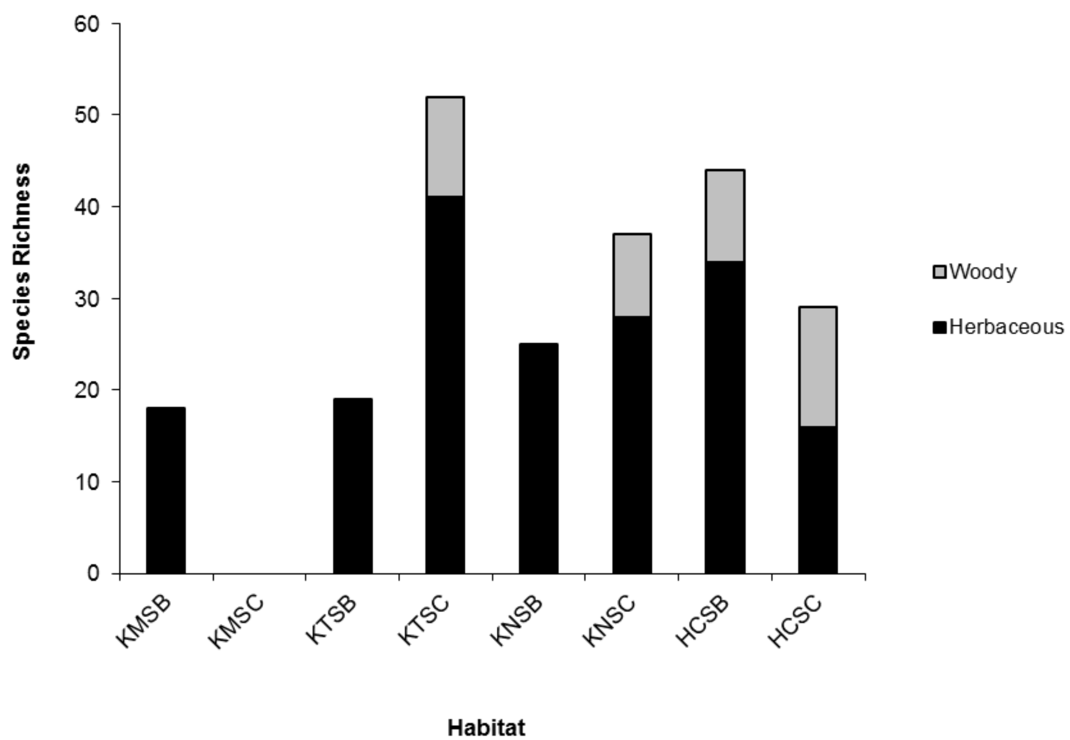


Figure 7

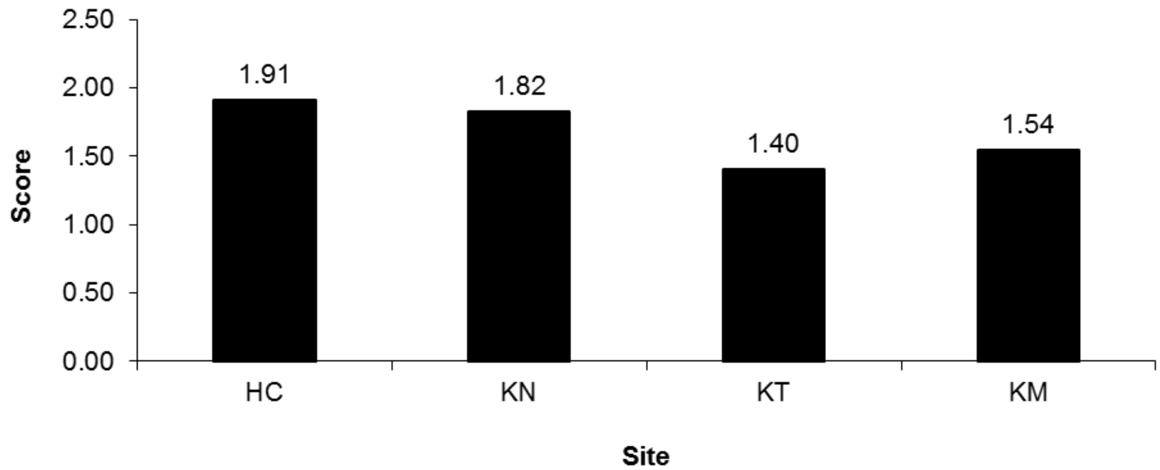
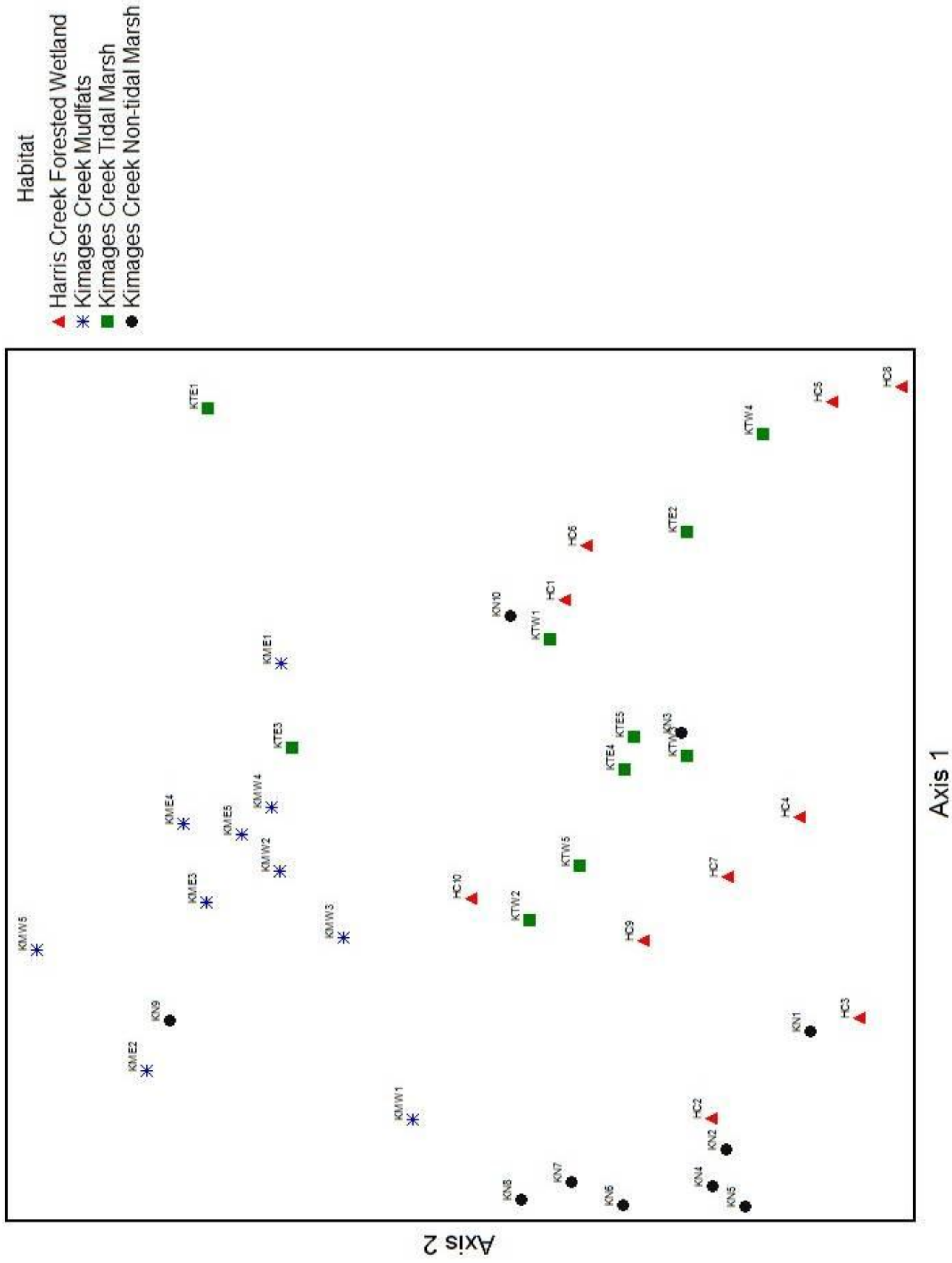


Figure 8



Habitat
Harris Creek Forested Wetland
Kimages Creek Mudflats
Kimages Creek Tidal Marsh
Kimages Creek Non-tidal Marsh
Axis 1
Axis 2

Figure 9

VCU Rice Center Bald Cypress Restoration

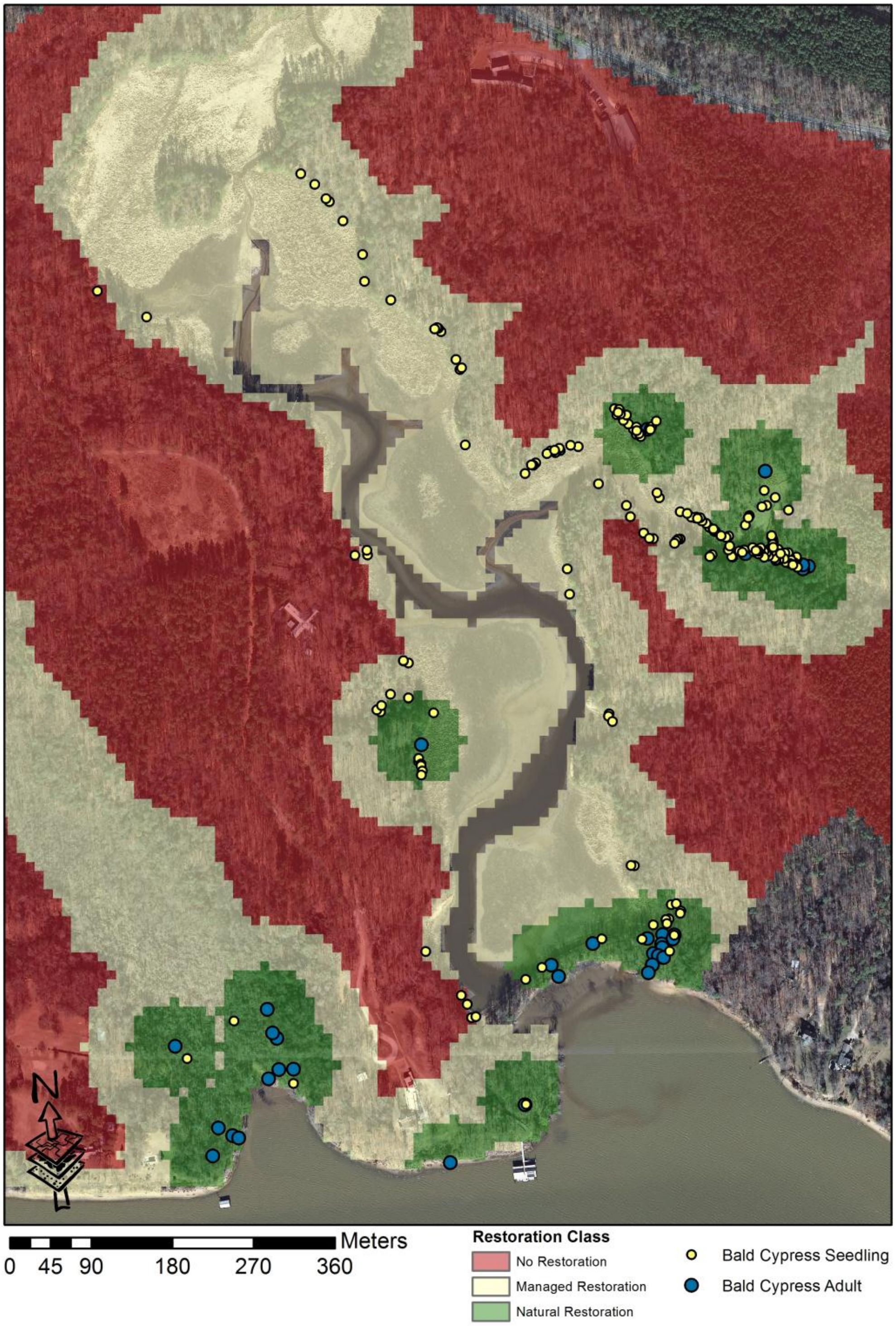


Figure 10

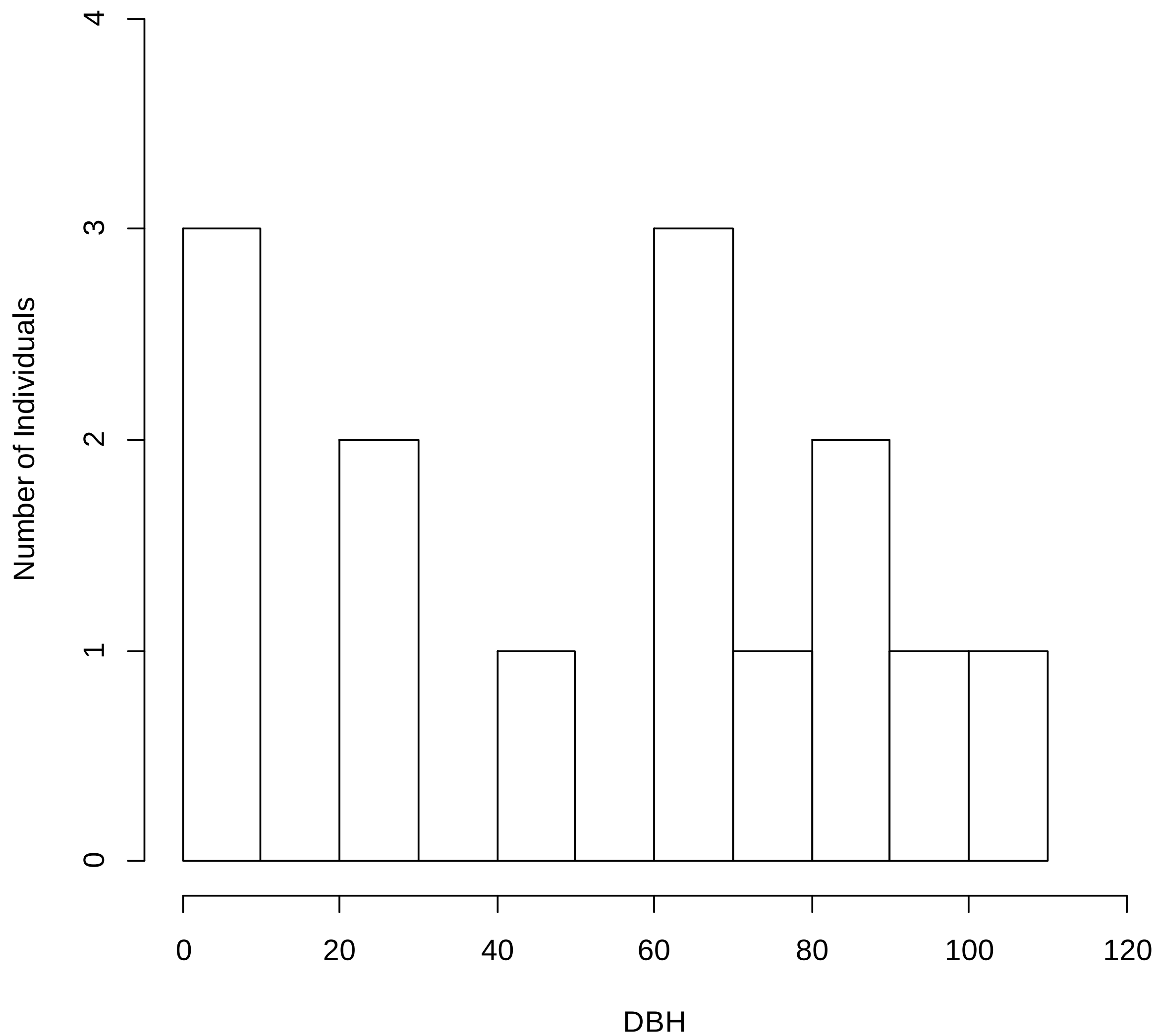
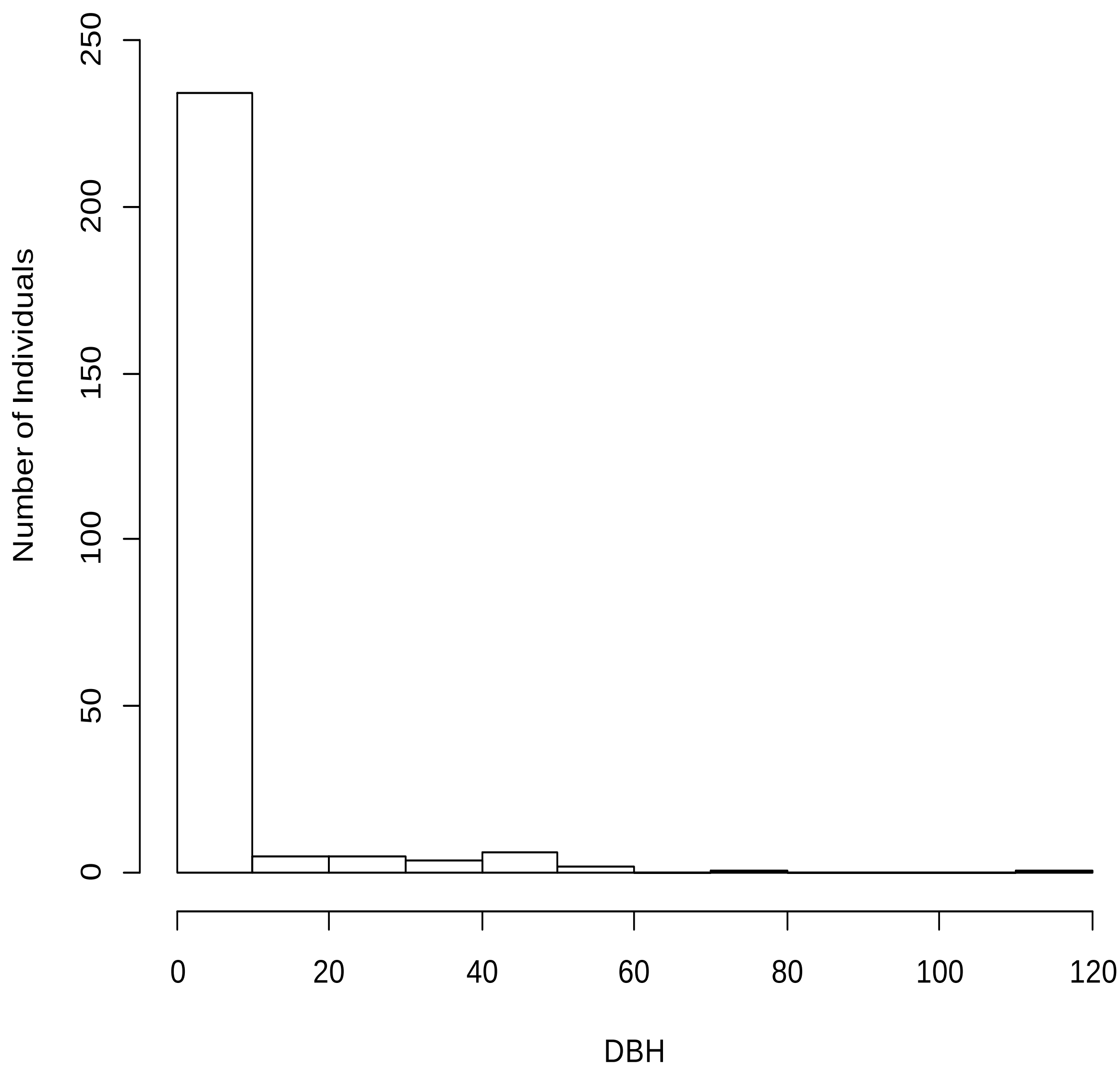


Figure 11



Number of Individuals
0
50
100
150
200
250
0
20
40
60
80
100
120
DBH

Figure 12

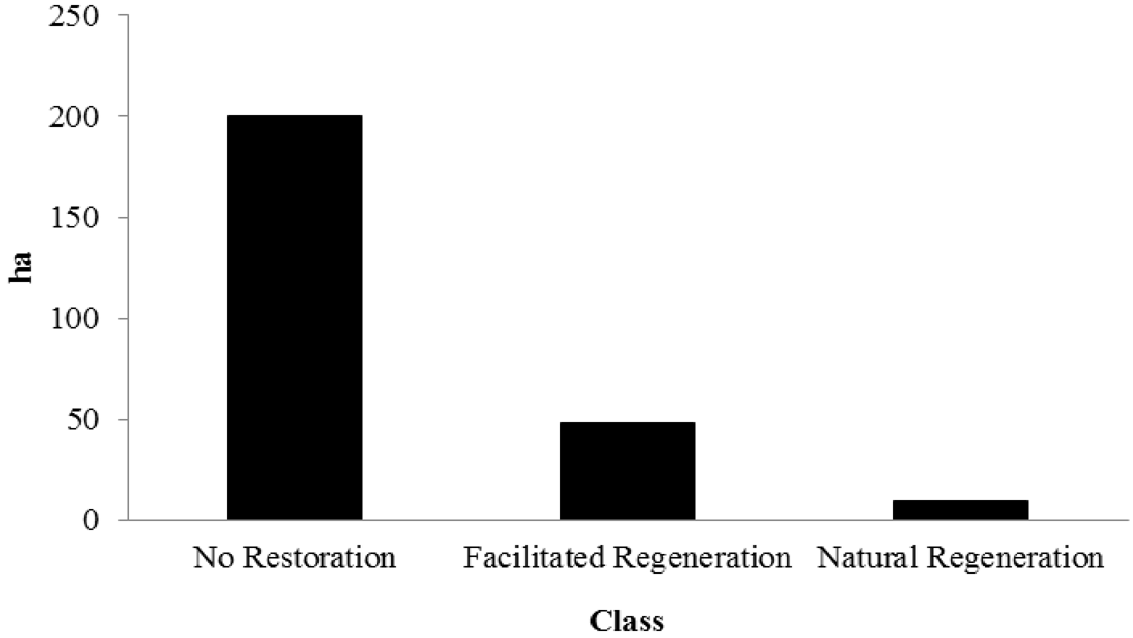
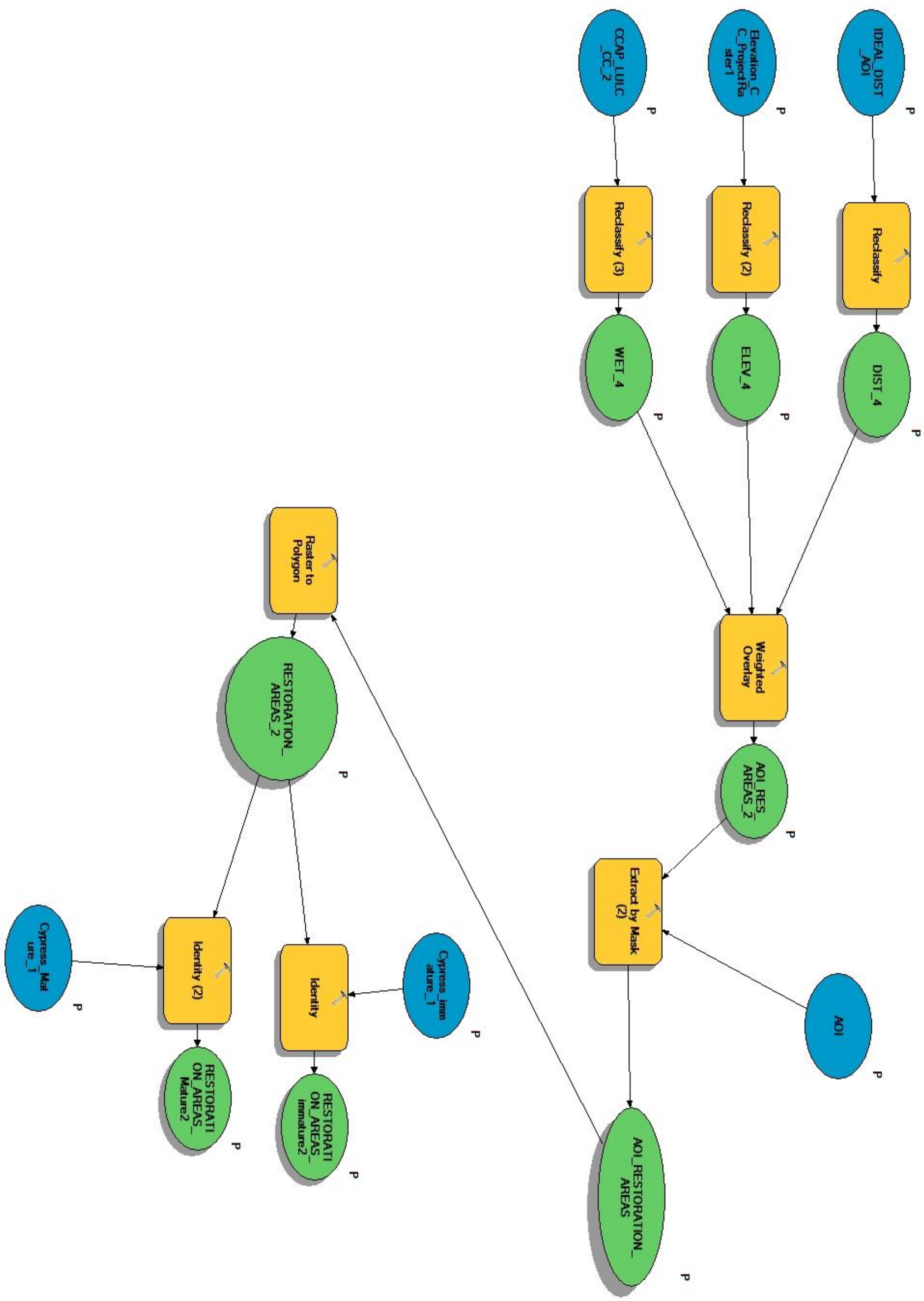

250
200
150
100
50
0
ha
No Restoration
Facilitated Regeneration
Natural Regeneration
Class

Figure 13

Appendix A: *Taxodium distichum* GIS Restoration Model Graphic Workflow



Appendix B: Detailed GIS *Taxodium distichum* Restoration Model Work Flow

NOTE: Names here past the names of original data are for the purposes of this workflow only, it is good practice to use proper nomenclature that is intuitive to the user so exact names are not necessary here.

Local Scale at the VCU Rice Center

Import all necessary data. Projection to Lambert NAD 1983 was completed in ArcCatalog 10, new rasters were of 10m resolution since data storage was not an issue and redundancy in the 30m data was not a problem.

Charles City County Boundary (copied from a selection into a new feature class during a previous project) Named CC for this workflow.

VA CCAP LULC 2006 data

NED 10m Elevation data for Charles City County

Used heads up digitizing (HUD) to create an area of interest (AOI) for the VCU Rice Center utilizing aerial photography

Imported field collected GPS points of Mature *Taxodium distichum* individuals at the Rice Center: named here as MatureCypress.

Imported field collected GPS points of Seedling *Taxodium distichum* individuals at the Rice Center: named here as ImmatureCypress.

Used Near Function to get distances of immature cypress from mature ones, statistically manipulated these results in Software package R (histogram of seedling distances from adults) to get basis for reclassifying Euclidian distances from Mature Cypress.

Ran Euclidian Distance tool to create a distance raster from the mature cypress named EUDISTcypress

Extract by mask on Charles City County elevation using AOI file as mask.

New file named here as Elev_AOI

Extract by mask on LULC using AOI file as mask.

New file named here as LULC_AOI

Extract by mask on Euclidian distance using AOI file as mask.

New file named here as EUDIST_AOI

Reclassified LULC_AOI based on suitability for restoration

Wetland Classes and open water* (13-18, 21) = 3

Upland forest types = 2

All Others = 1

Named LULC_AOI_IDEAL

* open water at site is now marsh ecosystem

Reclassified Elev_AOI based on suitability for restoration

0-5 feet = 3

5-10 feet = 2

10+ feet = 1

Named Elev_AOI_IDEAL

Reclassified EUDIST_AOI based on suitability for restoration based on statistical results

0-50 meters = 3

50-100 meters degrees = 2

100+ meters degrees = 1

Named EUDIST_AOI_IDEAL

Weighted overlay was run using these layers and the weights listed after them

EUDIST_AOI_IDEAL: 70%

Elev_AOI_IDEAL: 20%

LULC_AOI_IDEAL: 10%

Named: Restoration_Classes_AOI

Converted Restorarion_Classes_AOI to vector polygon file (DID NOT GENERALIZE).
Performed Idenity with ImmatureCypress, and MatureCypress named: ImmatureCypressRes and MatureCypressRes respectively.
Preformed Frequency statistics on ImmatureCypressRes. Not applicable to MatureCypressRes as they were all the same class, being part of the data used to create this model.
Built maps and exported
Exported tables

Deliver

Data management
Import all appropriate tables to Access
Create key tables in excel
Import keys to Access
Create appropraite relationships in Access based on Code field in keys and Gridcode of Value fields in Access.
Save
Deliver

Vita

James Burton Deemy was born in Richmond, Virginia on July 15th 1988. He grew up in King William County, Virginia and graduated high school there (at King William High School and Chesapeake Bay Governors School). James began undergraduate study at Virginia Commonwealth University in the fall of 2006. There he simultaneously completed his B.S. degrees in Environmental Studies and Biology in May 2010. He started his M.S. in Environmental Studies at Virginia Commonwealth University in the Summer of 2010. James plans on pursuing a career in ecological research.