Effects and Effectiveness of USDA Wetland Conservation Practices in the Mid-Atlantic Region: A Report on the Conservation Effects Assessment Project Mid-Atlantic Regional Wetland Assessment 2008 - 2015

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Investigators:
University of Maryland Department of Geographical Sciences
Megan Lang

U.S. Department of Agriculture, Agricultural Research Service
Hydrology and Remote Sensing Laboratory
Greg McCarty

U.S. Department of Agriculture, Agricultural Research Service
Coastal Plains Soil, Water, and Plant Research Center
Thomas Ducey, Patrick Hunt, and Jarrod Miller

University of Maryland, Department of Environmental Science and Technology
Martin Rabenhorst, Andrew Baldwin, Daniel Fenstermacher, Metthea Yepsen, Eliza McFarland, and Amir Sharifi

U.S. Department of Agriculture, Agricultural Research Service
Pasture Systems and Watershed Management Research Unit
Clinton Church

U.S. Geological Survey
Judy Denver and Scott Ator

Mitchell Ecological Research Service
Joseph Mitchell

Smithsonian Environmental Research Center
Dennis Whigham

U.S. Department of Agriculture, Agricultural Research Service
Office of National Programs
Mark Walbridge

Cover Photo: Photo taken by Metthea Yepsen at a MIAR CEAP-Wetland study site.

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Executive Summary

Although successful wetland restoration is generally considered to provide net benefits to society, the large investment that the US Department of Agriculture (USDA) has made in wetland restoration and increasing societal need for wetland ecosystem services highlights the importance of environmental research and monitoring. These efforts are needed to better understand the effects and effectiveness of wetland conservation practices, and to develop wetland restoration, implementation, and management practices that result in greater environmental outcomes and societal benefits. The Mid-Atlantic Regional (MIAR) Wetland Conservation Effects Assessment Project (CEAP-Wetland) is one of several USDA CEAP-Wetland investigations undertaken to collect and interpret data on ecosystem functions provided by wetlands established through USDA conservation programs. The MIAR CEAP-Wetland study employed a multiscale approach to meet project goals in the Mid-Atlantic portion (Maryland, Delaware, New Jersey, Virginia, and North Carolina) of the Gulf Atlantic Coastal Flat Physiographic Province, focusing on the effects and effectiveness of depressional non-tidal wetland restorations. All study activities were carried-out along a wetland alteration gradient, including hydrologically restored and relatively undisturbed natural wetlands, as well as prior converted croplands. A total of forty-eight primary study sites were selected (18 restored, 16 prior converted cropland, and 14 natural) to support MIAR CEAP-Wetland goals, with additional data being collected at ancillary study sites to support individual project components. Overall study results indicate a trend of wetland functional recovery subsequent to restoration, but this trend was not shared amongst all wetland functions, and intra-regional differences amongst certain criteria were significant.

Climate Regulation

Carbon related functions primarily support the provision of climate regulation services, as well as help drive nutrient regulation services. Some wetland restoration practices (i.e., excavation) were found to cause an initial, significant decrease in soil carbon stocks, which is in addition to the very significant decrease in soil carbon stocks associated with cultivation. In contrast, less invasive restoration approaches (e.g., ditch plugs and berms) did not have this effect. Above-ground plant biomass increased with time since restoration, indicating that carbon inputs, in addition to reduced oxidation through suspension of tillage and increased anaerobic conditions, should eventually compensate for soil carbon losses. The presence of a key functional gene in the soil microbial community that supports climate regulation services through reduction of nitrous oxide (a potent greenhouse gas) emissions was found to be significantly different and intermediate to natural wetlands and prior converted croplands within restored wetlands on mineral soil, but significantly lower than the other wetland types on histosols. This indicates that the microbial community is in transition, with likely implications for nitrous oxide emissions from restored wetlands.

Mitigation of Pollutants (Nutrients)

Wetland soil physicochemical characteristics and biogeochemical functions that mitigate pollutants (e.g., nitrogen and phosphorus) appear to be recovering post-restoration, but have not achieved levels identical to natural wetlands within a decade post-restoration. The effects of fertilization and liming are still evident within restored wetlands, but impacts of these management practices have decreased compared to prior converted croplands. Excavation redistributed nutrients and potentially soil microbial communities with potential impacts on nitrate removal and other biogeochemical processes. Wetland restoration practices appear to have enhanced phosphorus regulation capacity, but there is still potential for the provision of this service to be substantially increased through the natural weathering process. Although excavation reduces the provision of many ecosystem services, it enhances P mitigation services through the removal of P rich topsoil, assuming the topsoil is placed in a landscape position less prone to anaerobic conditions, leaching, and erosion (e.g., a vegetated upland). In areas dominated by mineral soils, potential nitrate removal in restored wetlands was more similar to natural wetlands, but in areas dominated by histosols potential nitrate removal exhibited a more incremental recovery. Nitrate removal services are also substantial within prior converted croplands. Therefore nitrate regulation services provided by both restored wetlands and prior converted croplands, which are likely to receive nutrient rich waters, are vital for maintaining the health of adjacent waters, including the Chesapeake

Bay. However, the level of pollutant regulation services ultimately provided by these areas will largely be determined by local surface and groundwater flow pathways.

Regulation of Hydrologic Flows and Mitigation of Natural Hazards (Flooding)

Wetland restoration in the MIAR helps to support the regulation of hydrologic flows and natural hazards (e.g., flooding). Natural wetlands exhibited relatively continuous flow into adjacent streams in contrast to prior converted croplands, which provided flashier flows directly after precipitation events. Restored wetlands exhibited surface water flows intermediate to natural wetlands and prior converted croplands. Wetland area was found to be significantly correlated with the periodicity of surface water flows. Even when depressional wetlands are not directly connected to streams via surface water flow, their size and arrangement has been found to be critical for supporting flow in adjacent streams (McLaughlin et al. 2014). Although wetland restoration has been found to exert a positive effect on the regulation of hydrologic flows and likely natural hazards, the extremely large volume of surface water storage that has been lost at a landscape scale relative to the modest gains in water storage made possible by restoration highlights the need for increased, sustained restoration.

Support for Biodiversity

Wetland restoration was found to have a strong, positive effect on plant and amphibian biodiversity and community quality, but restored communities were significantly different than those found in natural wetlands. Restored wetlands were hotspots of plant biodiversity, surpassing the diversity of natural wetlands. However restored wetlands are early successional ecosystems dominated by native herbaceous vegetation, whereas natural wetlands are dominated by native woody plants. We predict that restored wetlands will develop similarly to natural sites if succession is allowed to progress for decades. Natural and restored wetlands supported a similar number of amphibian species, while prior converted croplands harbored significantly fewer species. Restored and natural wetlands had approximately equal proportions of amphibian habitat generalists and specialists, but community similarity was low. These findings imply that overall landscape scale biodiversity is enhanced through the presence of a combination of natural and restored ecosystems.

Geospatial Analysis

Landscape-scale analysis, including the use of remotely sensed imagery, was found to: 1) identify segments of the landscape that experience unique, wetland-related ecosystem processes; 2) help develop guidelines pertaining to naturally occurring wetland morphologies; and 3) extrapolate field-based findings across the landscape. The next several years will bring dramatic changes to the availability of not only remotely sensed images that are well suited for the monitoring of wetlands and wetland conservation practices, but also the availability of wetland related products (i.e., maps) that can be quickly incorporated into decision support systems. The strong potential of remotely sensed data and products for improving conservation practice assessment and implementation paired with the rapid increase in the availability of such datasets highlights the critical role that this information source will play in future conservation efforts.

Implications for Wetland Restoration Implementation and Management in the MIAR

Actions can be taken by managers to encourage the provision of wetland ecosystem services subsequent to restoration. A list of these potential actions can be found below, divided between recommendations that should generally support the provision of all or most ecosystem services, those that generally support ecosystem service provision but have notable trade-offs, and actions that should be investigated to determine merit before implementation.

<u>General Recommendations</u>

1) Longer easement/contract periods should be promoted to allow time for slower environmental processes to proceed.

2) Soil compaction should be avoided to encourage root growth and the movement of nitrate rich groundwater into wetland soils capable of nitrate removal.
3) Either a greater number of restored wetland cells and/or larger wetland cells should better support the regulation of hydrologic flows and groundwater levels, and the mitigation of natural hazards, such as flooding.
4) Natural wetlands should be conserved, not only due to the high level of ecosystem services that they provide, but also because they directly enhance provision of ecosystem services from restored wetlands and prior converted croplands.
5) Because local topographic relief does not predict groundwater flow pathways in flat landscapes, an effort should be made to restore wetlands in locations that are low relative to broader-scale topographic gradients and are more likely to intercept upgradient groundwater containing agricultural contaminants, such as nitrate.
6) Wetland basins should be relatively shallow with gently sloping topographies, such that they support hydroperiods and water depths characteristic of natural wetlands to encourage colonization and growth of species that are representative of more natural conditions. Water depths and hydroperiods should be deep/long enough to discourage colonization by upland plants, reduce loss of carbon to the atmosphere, and support development of amphibian larvae but shallow/short enough to encourage plant growth and discourage establishment of predatory fish populations.
7) Intra-regional variations in physical and biological parameters should be considered when targeting, implementing, and managing wetland conservation practices.
8) Greater incorporation of geospatial datasets and techniques into precision conservation practice implementation and management strategies would serve to enhance not only ecosystem service provision but also the determination of derived benefits at a landscape scale, thus enhancing accountability.

Service Specific Recommendations

1) Overall our findings suggest that excavation should be minimized through enhanced targeting. Although excavation can reduce the provision of many ecosystem services, it can enhance P mitigation services. Therefore the merits of excavation should be considered relative to desired outcomes.
2) The practice of mowing should undergo benefit analysis, since it prevents the establishment of woody species, which are more characteristic of natural wetlands in the MIAR. However, increased forest canopy cover has been found to reduce the abundance and diversity of amphibian larvae or may impact suitability for migratory bird habitat.

Recommendations Requiring Additional Research

1) It is possible that wetland restoration practitioners could hasten development of more natural soil conditions through the active lowering of pH.
2) Due to the significant level of ecosystem services found to be provided by prior converted croplands, and the large area that they occupy at the landscape scale, conservation practices should be considered that directly apply to prior converted croplands, without taking those lands out of production (e.g., controlled drainage).

Conclusions

The MIAR CEAP-Wetlands study developed a broad collaborative base, which facilitated the collection and dissemination of novel integrative findings regarding wetlands in agricultural watersheds. This study provides critical information that will advance our ability to restore a diverse array of wetlands, thus enhancing the provision of ecosystem services. These findings not only help scientists, managers, and policy-makers better understand the impacts of wetland restoration relative to the existing wetland resource, but also support the improved allocation of resources and refinement of conservation implementation and management practices to optimize environmental outcomes.

A. Introduction

1. The Mid-Atlantic Region: Gulf Atlantic Coastal Flats

The U.S. Department of Agriculture (USDA) Mid-Atlantic Regional (MIAR) Wetland Conservation Effects Assessment Project (CEAP-Wetland) study area covers approximately ~58,000 km^2 in the eastern United States, including areas of the Gulf Atlantic Coastal Flats Physiographic Province in five states (North Carolina, Virginia, Maryland, Delaware, and New Jersey) and the District of Columbia (Figure 1). The study area abuts the Atlantic Ocean to the east and includes coastal bays of ecologic and commercial importance, including the Delaware and Chesapeake Bays to the north and the Pamlico Sound to the south. The region is generally flat with slopes decreasing towards the east and stream incision increasing towards the west. Climate is humid and temperate (north) to subtropical (south), with precipitation averaging approximately 120 cm per year and about half returning to the atmosphere via evapotranspiration (ET; Ator et al. 2013). About two-thirds of the precipitation that does not rapidly return to the atmosphere via ET recharges shallow groundwater (Leahy and Martin 1993), which is typically within a few meters of the land surface. Soils are generally permeable, but vary in texture. While mineral soils are dominant in the northern portion of the study area, Histosols are common in North Carolina. The region is predominantly in natural land cover (e.g., forested 18 %; scrub-shrub 8 %; grassland-herbaceous 3 %; palustrine wetland 31 %; estuarine wetland 4 %; total 64 %; National Oceanic and Atmospheric Administration Coastal Change Analysis Program 2010 map; http://www.coast.noaa.gov/dataregistry/search/collection/info/ccapregional), but includes areas with a high density of agriculture (i.e., the Delmarva and eastern central North Carolina) accounting for 28 % of total regional land cover. Small to medium sized urban centers are dispersed throughout the region, and include Hamilton, New Jersey, Dover, Delaware, Cambridge and Salisbury, Maryland, the Virginia Beach-Norfolk-Newport News-Hampton metropolitan area within Virginia, and Greenville and Wilmington, North Carolina (8 % developed land).

Wetlands are abundant within the MIAR CEAP-Wetland study area, in large part due to the region's relatively flat topography, close proximity to groundwater and the coast, and relatively high precipitation to ET ratio. Most of the area's wetlands are found inland, and the vast majority of these are forested or scrub-shrub wetlands located in floodplains, between drainage systems in broad flats, and in upland depressions. Although riparian, depressional, and flats wetlands are commonly found throughout the region, they tend to be concentrated in certain portions of the region. For example, within the Delmarva Peninsula, depressional wetlands (e.g., Delmarva bays) are concentrated to the north whereas riparian wetlands and wetland flats are more abundant to the south. Estuarine wetlands are found throughout the coastal margins of the Peninsula. They are concentrated at and south of Dorchester County, Maryland adjacent to the Chesapeake Bay, and along Delaware Bay and Virginia coastal bays along the Atlantic Ocean. Pocosins, bogs dominated by evergreen shrubs, are common in North Carolina (Richardson 1983). It is common to find wetlands interspersed with cropland because these are the areas that were more difficult to convert to crop production. Fluxes between these two land covers include surface and groundwater, biotic, and atmospheric pathways.

Mid-Atlantic wetlands provide critical ecosystem services, including the provision of freshwater, regulation of pollutants (e.g., nutrients), climate, hydrological flows, and natural hazards, as well as support for biotic communities, which in turn enhance the provision of multiple ecosystem services. The study area's wetlands are especially important as they help to maintain water quality and aquatic habitat in multiple inland Bays, comprising some of the largest and most productive estuarine ecosystems in the United States, and provide ecosystem services to a large and rapidly increasing human population. Wetlands are critical areas for nutrient transformation and flux. An overabundance of nitrogen and phosphorus has led to the eutrophication of many inland water bodies, including Mid-Atlantic coastal bays (Chapter B8). Biogeochemical processes which disproportionately occur in wetlands and other aquatic habitats (e.g., denitrification) can reduce the flux of nitrate to water bodies, thus improving water quality. However, incomplete denitrification can lead to the emission of N_2O (nitrous oxide), a powerful greenhouse gas. Phosphorus can be mobilized and thus enter the water column through the anaerobic conditions that are often present in wetland ecosystems, but aquatic processes can also lead to the burial or uptake of phosphorus by plants.

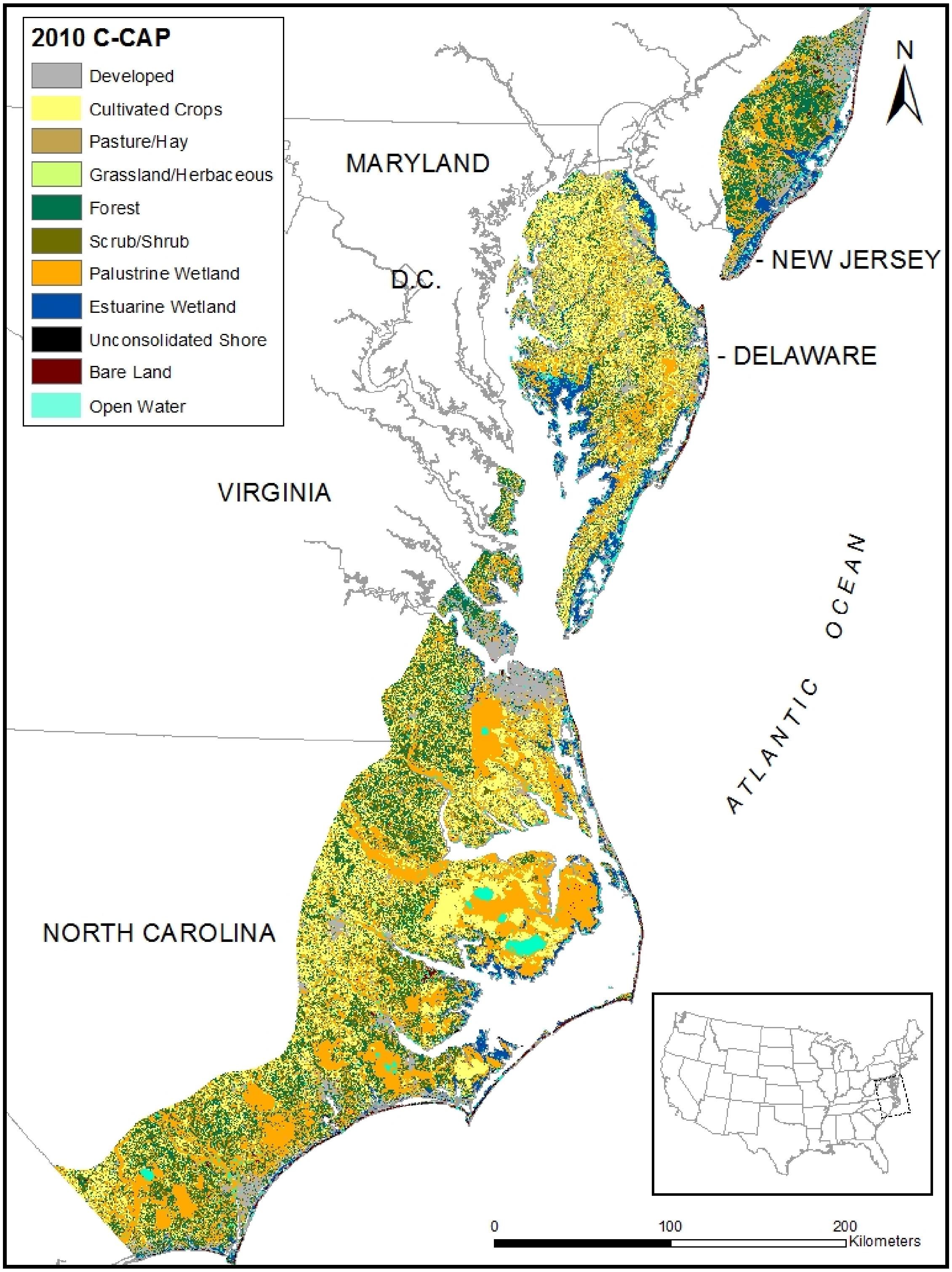

Figure 1: Land cover in the Mid-Atlantic Regional CEAP-Wetland study area. Land cover map provided by the National Oceanic and Atmospheric Administration Coastal Change Analysis Program (C-CAP). Note that some C-CAP land cover classes have been summarized (i.e., all categories of developed land were combined into a general developed class; all forest types were combined into a general forest class; and all types of palustrine and estuarine wetlands were classified as palustrine or estuarine wetlands, respectively).

At a global scale, the threat of climate change has received a great deal of attention recently due to its potentially devastating consequences, including drought, flood, and sea level rise. Wetlands are regulators of climate through their influence on carbon, nitrogen, and other biogeochemical cycles. For example, wetlands are the world's greatest source of methane to the atmosphere (20-40 % annually) with methane's contribution to radiative forcing being about half that of CO_2 (carbon dioxide; Prigent et al. 2007). At a global scale, annual emission of carbon dioxide from inland waters is similar to the uptake of this greenhouse gas by all of the world's oceans, and burial of organic carbon within inland aquatic systems is higher than carbon sequestration on the ocean floor (Verpoorter et al. 2012). The effect of wetlands and wetland management on the carbon cycle is particularly relevant in coastal North Carolina, which contains significant deposits of peat (Chapter B2), but systematic human alterations to carbon storage have the potential to significantly affect greenhouse gas regulation throughout the Mid-Atlantic region.

Unfortunately many of the region's wetlands have already been lost to agriculture, silviculture, urbanization/suburbanization, sea level rise, and other causes, with agriculture being the dominant historic driver of wetland loss (Dahl 1990 and 2011). Between European colonization and 1980 wetland area in New Jersey, Maryland, Delaware, Virginia, and North Carolina declined by 39 %, 73 %, 54 %, 42 %, and 49 %, respectively (Dahl 1990). The importance of remaining wetlands is enhanced by the fact that surrounding upland areas are densely populated and these populations are rapidly expanding. Therefore, the need for wetland functions, such as nutrient reduction, is increasing while wetland area is simultaneously being reduced through development. Mid-Atlantic Coastal Plain wetlands have been identified as having a high probability for future loss (U.S. Fish and Wildlife Service 2002). The Mid-Atlantic/New England region has a higher population density than any other area in North America and its population is expected to increase at the same time that scientists are forecasting changes in climate. These changes in climate will undoubtedly alter wetland hydroperiod through shifting water balance and will likely alter wetland extent and function (Moore et al. 1997). These alterations will, in turn, influence the provision of a host of wetland mediated ecosystem services, including the provision of freshwater, regulation of pollutants and surface water flows, and mitigation of natural hazards.

2. Wetland Restoration

Recognition of the large amount of historic wetland loss and the importance of ecosystem services provided by wetlands (e.g., Costanza et al. 1997) has led to efforts within the United States and elsewhere to reduce wetland loss and encourage the addition of wetlands to the landscape, in part through wetland restoration. The United States has dedicated a substantive amount of effort and resources towards the conservation, enhancement and restoration of wetlands. In the past, an average of $500 million of the Wetland Restoration Program budget, and part of the $1.8 billion Conservation Reserve Program budget were spent on wetland restorations annually (American Planning Association 2010). These expenditures have had a measureable effect at the national scale. Wetland restoration practices, supported by United States Farm Bill conservation programs and other federal, state, and non-governmental organizations efforts (Council on Environmental Quality 2008), are currently being implemented across the United States and have contributed to a measureable increase in wetland area (Dahl 2011). However, the types and locations of wetlands being restored or created are often different than those being lost. These alterations have implications for wetland functioning and the provision of ecosystem services (e.g., Bedford 1996, 1999), but the effects of these changes are largely unknown. Although wetland restoration occurs in many landscape positions and morphologies, wetland restorations supported by USDA Farm Bill programs are often depressional in nature, or include depressional wetland cells. Within the Mid-Atlantic region restored wetlands with a depressional morphology may have once been depressional wetlands or they may have been a different wetland hydrogeomorphic type, such as a wetland flat. Furthermore, the effects of restoring wetlands that have not only been drained, but also altered chemically and mechanically under active cropland management are also unclear, as are the effects of specific implementation and management practices on the provision of different ecosystem services. What is known is that wetland restoration efforts, in part, are contributing to a national shift in wetland type, and likely wetland location, and that a sizeable proportion of restoration is occurring within agricultural landscapes, which are physically, chemically, and biologically unique relative to more natural ecosystems. A study of wetland change within the United States between 2004 and 2009 found that freshwater ponds on agricultural land increased by an estimated 5.4 % or 61,700 ha, likely a significant portion of which was due to wetland restoration efforts (Dahl 2011). The importance of wetland restoration on agricultural lands will increase in the future. Crop production is predicted to grow due in large part to the needs of an increasing population for sustenance and biofuel energy. Conservation practices, including wetland restoration, will become even more critical for sustaining enhanced crop productivity while reducing impacts to natural resources and enhancing the provision of ecosystem services (Eckles 2011). In order for the USDA to best allocate wetland restoration funds, a better understanding of the effects and effectiveness of wetland restoration, in terms of the delivery of ecosystem services, is necessary.

3. The Mid-Atlantic Regional CEAP-Wetland Study

The Mid-Atlantic Regional (MIAR) Wetland Conservation Effects Assessment Project (CEAP-Wetland) is one of several USDA Conservation Effects Assessment Project Wetland Component investigations undertaken to collect and interpret data on ecosystem services provided by wetlands established through USDA conservation practices. Planning for the MIAR CEAP-Wetland study was initiated in federal fiscal year 2008. A multi-disciplinary group of scientists was selected by Diane Eckles, Natural Resources Conservation Service (retired), in December 2008 to collect information regarding three primary research areas, including pollutant (i.e., nutrient) mitigation, regulation of greenhouse gas emissions including carbon sequestration, and support of amphibian and plant communities. A comprehensive project proposal was accepted by the Natural Resources Conservation Service (NRCS) in May of 2009. The study team was composed of regional experts from multiple federal agencies (i.e., Agricultural Research Service, U.S. Geological Survey, and Smithsonian Environmental Research Center), as well as the University of Maryland and private industry partners. This broad, collaborative science team was assembled to enhance understanding of the effects and effectiveness of wetland hydrologic restoration in terms of wetland functions resulting in ecosystem service provision, and to develop the methods necessary to classify wetlands according to primary functional drivers to support extrapolation of field-based findings. The ultimate goal of this study is to increase the benefits of wetland conservation practices and environmental outcomes directly through enhanced targeting, implementation, and management of conservation practices, and indirectly through enhanced estimation of landscape scale effects of wetland conservation practices via process-based and statistical modeling.

The MIAR CEAP-Wetland study employed a multi-scale approach to meet project goals in the Mid-Atlantic portion (Maryland, Delaware, New Jersey, Virginia, and North Carolina) of the Gulf Atlantic Coastal Flat Physiographic Province, focusing on the effects and effectiveness of depressional non-tidal wetland restorations. All study activities were carried-out along a wetland alteration gradient, including hydrologically restored and relatively undisturbed natural wetlands, as well as prior converted croplands (Figure 2). Natural wetlands with native vegetation were selected to represent the average condition of local, minimally disturbed wetlands. They provide context necessary to judge the effectiveness of wetland restoration relative to the stated goal of the USDA NRCS Wetland Restoration (657) Practice Standard, which is to "restore wetland function, value, habitat, diversity and capacity to a close approximation of the pre-disturbance condition." The importance of this historic perspective was emphasized by Bedford (1999): "By definition [wetland restoration] seeks to replace what has been lost. By definition then, it should be undertaken with knowledge of what has been lost." prior converted croplands provide a baseline from which to judge the effects of wetland restoration. Although prior converted croplands are no longer considered to be wetlands within a wetland regulatory, or typically wetland definition framework, they often continue to function as a wetland to some degree. Therefore a baseline of zero or a baseline commensurate with upland portions of an agricultural field is not adequate when judging the effects of wetland restoration. Hydrologically restored wetlands were selected by USDA NRCS staff from participating states to represent typical USDA wetland restoration practices resulting in a depressional morphology.

Two general geographic focus areas were identified within the Mid-Atlantic region based on conservation practice distribution patterns. Wetland restorations were found to be primarily located within the Delmarva Peninsula and the eastern Virginia/North Carolina border area. Sites were selected within these two broad areas to include non-tidal, locally representative current or former wetlands located at least1 km from one another but not more than 4 km from one another, if possible. State/district NRCS representatives were asked to select depressional, hydrologically restored wetlands that were formerly agricultural fields and were representative of NRCS wetland restoration practices within the state. Study collaborators subsequently visited each selected site to confirm that they met study criteria and could be safely accessed. After restored sites were selected, multiple dates of aerial photography and digital elevation models were used to identify prior converted croplands near selected wetland restorations in areas with similar soil, climate, and landscape position. These areas were ranked based on the general criteria listed above and this information was sent to state and/or district NRCS staff as shapefiles and/or maps. State and/or district NRCS staff used this information to identify sites and contact land owners to request study participation. After restored and prior converted cropland sites were identified, natural sites were selected based on the location and character of the existing sites, with preference given to sites with the lowest amount of anthropogenic impact. Natural sites were located on lands owned by individuals, as well as lands owned by the state, federal government, and non-governmental organizations (e.g., The Nature Conservancy). If more sites than necessary were identified within a state, a random selection approach was used to select study sites. Sites were chosen to minimize natural differences and maximize anthropogenic differences.

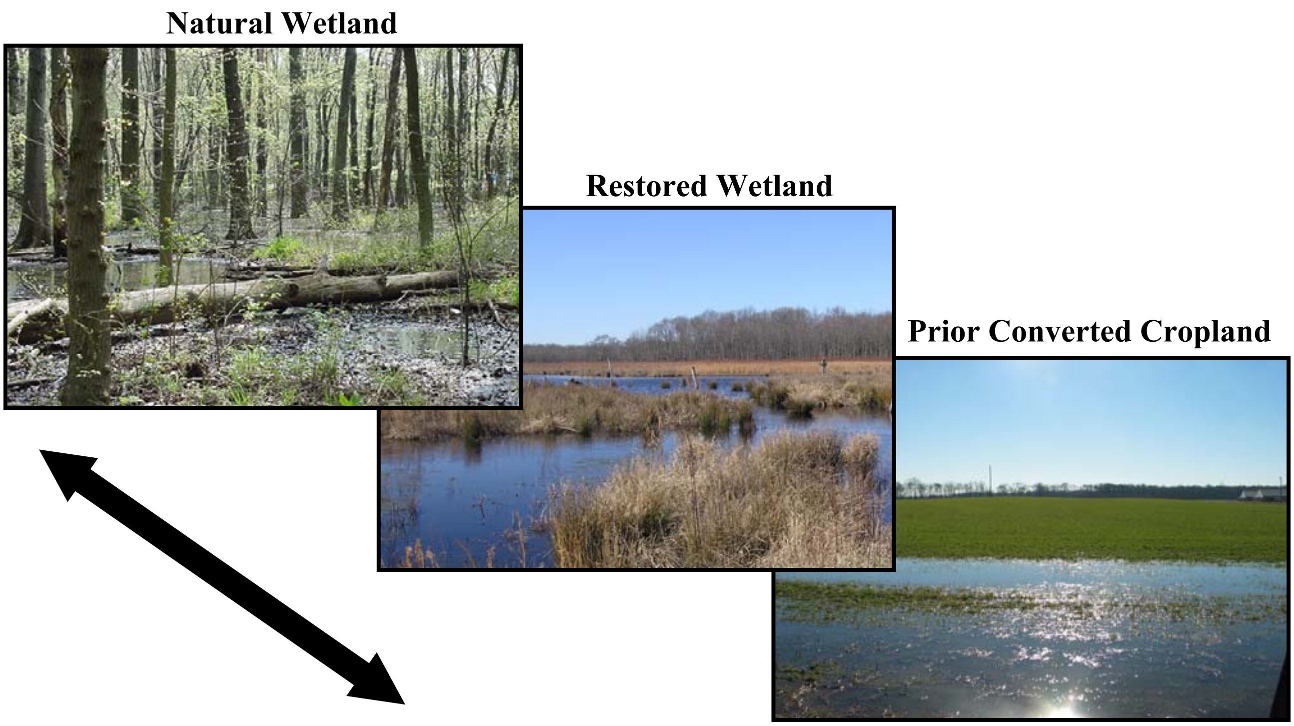

Figure 2: Photographs illustrating the wetland alteration gradient, including (from left to right) a relatively unaltered wetland with native vegetation, a hydrologically restored wetland, and a prior converted cropland.

A total of forty-eight primary study sites were selected along a wetland alteration gradient (Figure 3), to support MIAR CEAP-Wetland goals with additional data being collected at ancillary study sites to support individual project components. The sites consisted of 18 restored and 14 natural wetlands, as well as 16 prior converted croplands. Restored wetlands were hydrologically restored according to USDA NRCS Conservation Practice Standard 657 (Wetland Restoration) between 2001 and 2008 via multiple USDA conservation programs, including the Wetland Reserve Program and the Conservation Reserve Program. Before restoration these areas were used to grow row-crops, typically either corn (*Zea mays*) or soybeans (*Glycine max*). Restored sites were established through a combination of techniques including the plugging of ditches, creation of berms, excavation (removal of soil to form a depression), and soil compaction and ranged in size from 0.12 to 1.13 hectares. Microtopography or larger hummocks (i.e., islands) were found at some restorations. Little to no planting of wetland species was done as part of the restorations although trees were planted within some upland buffers. Plant community composition varied across the different restored sites, but common species found across most of the sites were *Ludwigia palustris, Echinochloa cru-galli, Xanthium strumarium, Scirpus purshianus,* and *Boehmeria cylindrica* (Yepsen et al. 2014). Prior converted croplands are wetlands converted to upland cropland before 1985 and continuously used for agriculture through the present time. All prior converted croplands were used to produce row crops, mainly corn (*Zea mays*) or soybeans (*Glycine max*) at the time of this study. All prior converted croplands were either ditched or had ditches nearby to facilitate drainage. Natural wetlands were defined as those that are dominated by native wetland plants with no or minimal direct alteration of hydrology. Truly pristine sites are rare within the Mid-Atlantic region. All natural forested sites had been logged at some point in the past. Natural sites were characterized as either depressions or flats under the Hydrogeomorphic Classification (HGM) (Brinson 1993). All but one natural site was forested. Natural sites were dominated by native plant species, including as *Liquidambar styraciflua, Acer rubrum, Clethra alnifolia,* and *Smilax rotundifolia* (Yepsen et al. 2014). For detailed descriptions of study site soils and vegetation please see Fenstermacher et al. (2012) or Yepsen et al. (2014), respectively.

4. Report Format

Previously, the methods and findings of MIAR CEAP-Wetland study components were published as peer-reviewed journal articles, graduate theses, book chapters, and USDA NRCS CEAP Science Notes (Lang and McCarty 2009; Fenstermacher 2012; Lang et al. 2012; Yepsen 2012; Lang et al. 2013; Ator et al. 2013; Lang et al. 2013; Fenstermacher et al. 2014; Hunt et al. 2014; Lang and McCarty 2014; Yepsen et al. 2014; Ducey et al. 2015; Lang et al. 2015a; Lang et al. 2015b; Lang et al. 2015c; McFarland et al. 2015; Fenstermacher et al. 2015; Church et al. [in review]; Mitchell [in review];Sekellick et al. [in review]), as well as other publications and unpublished reports. The following report summarizes the methods and findings of several of these publications in detail, and provides implications and recommendations for implementation and management of wetland restorations. Each primary MIAR CEAP-Wetland component chapter (see section B - below) begins with a summary of key findings, conservation practice implications, and identification of the peer-reviewed journal article that was the primary source of chapter content. Readers should refer to these source documents for additional information regarding the research discussed in each study component chapter. All MIAR CEAP-Wetland research and resultant publications were funded by the USDA NRCS. In addition to the MIAR CEAP-Wetland publications listed above the report also relies on findings from the earlier Choptank CEAP-Wetland study (Denver et al. 2014, Fox et al. 2014, McDonough et al. 2015, and McDonough [in review]), which focused on a much smaller geographic area. The limited geographic scope of the Choptank CEAP-Wetland study allowed more intensive data collection and analysis that was not possibly across the entire Mid-Atlantic region. Please note that references for study components one through nine can be found directly after each chapter, whereas citations used in the introduction and later portions of the report can be found at the end of the document.

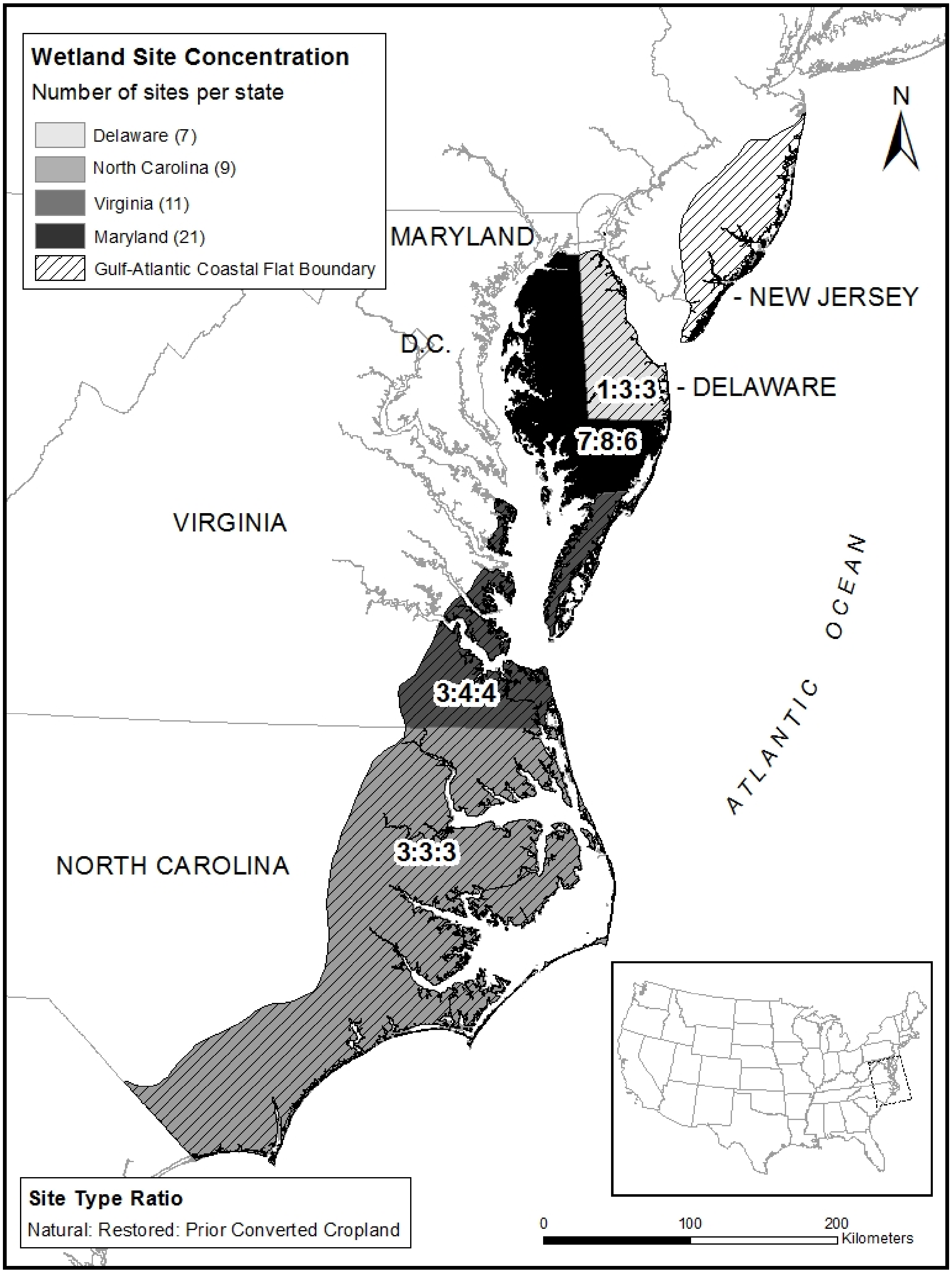

Figure 3: MIAR CEAP-Wetland study site locations, identified by state and wetland type. Sites were located based on the distribution of USDA wetland conservation practices and are concentrated in the Delmarva Peninsula and the eastern border between Virginia and North Carolina.

B. MIAR CEAP-Wetland Study Components

1. Above-Ground Plant Biomass and Nutrient Content

Key Findings

-At MIAR sites, herbaceous biomass was higher in restored wetlands relative to natural wetlands, while standing woody biomass was highest in natural sites. Nutrient concentrations in herbaceous biomass were significantly higher in prior converted cropland sites relative to natural sites, and restored sites demonstrated values intermediate to the other site types.

-At an additional 9 restored sites that were assessed to gain supplementary information on the trajectory of restored wetlands, both herbaceous and woody biomass were found to increase with time since restoration.

- Nutrient data indicate that the restored wetlands will become less nitrogen limited as the impacts of previous agricultural activities decline.

-Although all restored wetlands are in early stages of plant succession, they are following a trajectory of recovery and we predict that they will develop similarly to natural sites if succession is allowed to progress for decades.

Recommended Practices: Plant succession is often a slow process, taking decades if not longer. Assuming that the goal of wetland restoration is to restore wetland habitat to a close approximation of the pre-disturbance condition and acknowledging that the vast majority of MIAR depressional wetlands are forested, longer easement periods should better allow restored wetlands to approximate their natural counterparts. This time could be reduced by planting late-successional species characteristic of undisturbed local natural sites.

***Primary Chapter Source*:** McFarland, E., LaForgia, M., Yepsen, M., Whigham, D., Baldwin, D., and Lang, M. 2015. Plant biomass and nutrients (C, N, and P) in natural, restored, and prior-converted depressional wetlands in the Mid-Atlantic Coastal Plain, U.S. *Folia Geobotanica.* (In Press)

Introduction

Assessments of restoration success have focused on a range of outcomes (e.g., hydrologic, wildlife habitat, and nutrient cycling), but many efforts compare vegetation in restored wetlands to vegetation in natural (i.e., undisturbed) sites (Gutrich et al. 2008; De Steven et al. 2010). Plant nutrient concentrations, especially nitrogen to phosphorus ratios, can provide added insights into the developmental status of restored wetlands, as N:P ratios are related to nutrient availability and they reflect the species composition of plant communities (Güsewell and Koerselman 2002; Güsewell et al. 2003). This MIAR study component addressed two issues regarding restoration outcomes: 1) similarities and differences in plant biomass, nutrient content and community composition between restored, natural, and prior converted cropland sites and 2) changes in these parameters through time. The second portion of this study component was made possible by a study of nine restored wetlands on the Delmarva Peninsula conducted in 1996. These nine older restorations were sampled in addition to the MIAR sites, providing not only a comparison between wetland types but also wetland restoration ages. Since the older sites were originally visited when they were similar in age (years since restoration) to the MIAR restoration, these older sites provided valuable context and allowed further conjecture regarding the trajectory of the MIAR sites.

Methods

Study Sites

This study component included nine older wetland restoration sites, in addition to the 48 sites included in most other study components. The nine older sites were originally sampled in 1996 as part of a Smithsonian Environmental Research Center study (Pepin and Whigham 1999; Whigham et al. 2002; Jordan et al. 2003). These study sites will be hereafter referred to as SERC sites (Figure 1). The restoration process used at the SERC sites is detailed in Whigham et al. (2002) and, in each instance, involved removal of soil to expand the size of the original depression, using the excavated soil to create a berm around the downstream side of the site and the inclusion of water control structures to regulate water levels. Similar to the original MIAR restored sites, the SERC restorations were not planted and management within the wetland area was rare.

Data Collection

In October-November of 2011, the biomass of herbaceous species at the MIAR sites was harvested from 0.25 m^2 (50 x 50 cm) quadrats located 3 meters to the northwest and southeast of each survey plot established by Yepsen (2014). At the SERC sites, biomass was harvested in October-November of 2011 from the same plant hydrologic zones (i.e., emergent and transitional to upland) that had been harvested in the earlier study (Whigham et al. 2002), except in areas that lacked emergent vegetation due to water depth. Vegetation was cut at the soil surface and biomass was weighed in the field. Samples were collected at the prior converted cropland sites for use in the nutrient analysis only. A composited subsample of each harvested plot was weighed in the field and returned to the laboratory where the samples were dried for a minimum of 48 hours at 60 ^oC. The moisture content of the subsamples was used to calculate biomass of the entire samples harvested and weighed in the field. The dried subsamples of herbaceous vegetation were ground to a powder and analyzed for carbon (C) and nitrogen (N) with a CHN analyzer (CE-440 Elemental Analyzer). For phosphorus (P), 2 mg of dried, ground plant material was placed in a muffle furnace at 550 ^oC for two hours (Miller 1998), followed by colorimetric analysis using the ammonium molybdate method (Clesceri et al. 1998). Biomass and nutrient concentration data were used to calculate total C, N, P, and N:P for the harvested herbaceous vegetation.

Trees, defined as single stems taller than 1 m and a diameter at breast height (DBH) > 2.5 cm, were sampled at all sites to estimate standing biomass in 100 m^2 circular plots (diameter of 5.64 m). We measured trees in 3-6 randomly selected plots per MIAR site, with the number of plots sampled based on Yepsen et al.'s (2014) plot selection, and in 3 randomly selected plots in each of the SERC wetlands. Within each plot, trees were identified and diameter at breast height (DBH) measured. Tree biomass in each plot was calculated using DBH measurements and the equation presented in Jenkins et al. (2004). Neither leaves nor leaf litter was collected as part of the study; thus biomass estimates represent existing standing aboveground woody biomass and not annual production. The biomass of trees was calculated as g/m^2 to enable comparisons with herbaceous vegetation; we also provided, however, the tree biomass equivalents in kg/ha for comparison to other literature [given in brackets].When present, trees were counted and identified to species. Frequency of each woody species in the three wetland types in which they occurred (MIAR natural, MIAR restored, SERC 2011) was calculated by dividing the number of plots where the species occurred by the total number of plots sampled in the wetland. Sample size (n) represents the number of wetlands where trees were documented; only wetlands with trees were used in this analysis.

Statistical Analysis:

Comparison of Wetland Type: For MIAR sites, biomass and nutrient stock data were first square root transformed to normalize data prior to statistical analysis. Biomass comparisons between restored and natural sites were made using one-way Analysis of Variance (ANOVA). Percent nutrient composition and N:P ratios were arcsine-square root transformed to normalize data, and then calculated for all three wetland types using one-way ANOVA. Replicates were wetlands.

Temporal Comparison: All metrics measured in 2011 (biomass, nutrient standing stocks, and percent nutrient composition) were compared to 1996 data from the earlier study (Whigham et al. 2002) using one-way ANOVAs. Biomass and nutrient stock data were first square root transformed prior to statistical analysis, and percent nutrient content and N:P ratios were arcsine-square root transformed to normalize data. All statistical analyses were performed using JMP Pro 11 (SAS software, JMP Pro 11, SAS Institute Inc., Cary, NC).

Results

Comparison of Wetland Type:

Average aboveground biomass ($\pm$ 1 SE) of herbaceous vegetation at restored sites (423.0 ± 44.6 g/m^2) was four times higher (Figure 1) than natural sites (99.1 ± 41.7 g/m^2) and the difference was highly significant (Table 1A). In contrast, aboveground biomass of trees (Figure 2) was almost fifteen times higher at natural ($2.94*10^4 \pm 4.1*10^3$ g/m^2; [$2.94*10^5 \pm 4.1*10^4$ kg/ha]) compared to restored sites ($1.57*10^3 \pm 8.2*10^2$ g/m^2; [$1.57*10^4 \pm 8.2*10^3$ kg/ha]). Standing stocks of P, N, and C were also significantly higher (Table 1A) at restored sites with means ($\pm$ 1 SE) of 0.83 ± 0.09 g/m^2 P, 5.3 ± 0.6 g/m^2 N, and 190.8 ± 18.4 g/m^2 C. Standing stocks of P, N, and C at natural sites were 0.15 ± 0.07, 2.06 ± 0.89, and 85.6 ± 33.9 g/m^2, respectively.

There was an overall significant difference for wetland type in P and N concentrations in herbaceous vegetation and the three site types were significantly different from each other with the highest means in the prior converted cropland sites, intermediate in the restored sites, and lowest in the natural sites (Table 1B and Figure 3a and 3b). The carbon content of herbaceous vegetation also differed significantly (Table 1B). Both the prior converted cropland and restored sites, which did not differ from each other, were significantly higher in carbon than the natural sites (Figure 3c). The mean N:P ratios of herbaceous vegetation at all sites were below 16, a possible indication of N limitations (Güsewell et al. 2003). Significantly higher N:P ratios at the natural sites compared to the prior converted cropland and restored sites (Table 1B) indicated that N may become less limiting over time following restoration. The N:P ratios for the prior converted cropland and restored sites did not differ from each other (Figure 3d).

Temporal Comparison:

The biomass of herbaceous vegetation was significantly higher (Table 2A) in 2011 compared to 1996 (Figure 4). There was no tree biomass at the SERC sites when they were sampled in 1996, but 15 years later trees had colonized the outer zones of the wetlands and woody biomass averaged $1.38*10^3 + 4.9*10^2$ g/m^2 [$1.38*10^4 + 4.9*10^3$ kg/ha]. Standing stocks of P, N, and C were also significantly higher (Table 2A) in 2011, with means ($\pm$ 1 SE) of 1.36 ± 0.12 g/m^2 P, 12.89 ± 1.50 g/m^2 N, and 253.09 ± 29.96 g/m^2 C. Standing stocks of P, N, and C in 1996 were 0.53 ± 0.04, 2.89 ± 0.23, and 110.34 ± 9.1 g/m^2, respectively.

Concentrations of P, N, and C in herbaceous biomass differed significantly between 1996 and 2011 (Table 2B; Figure 5). All nutrient concentrations were significantly higher in 2011 (Phosphorus: Figure 5a; Nitrogen:

Figure 5b; Carbon: Figure 5c). The N:P ratios indicated N limitations in both 1996 and 2011, and the means were significantly higher in 2011 (Figure 5d).

<u>Tree species:</u>
The frequency of tree occurrence was significantly higher in natural wetlands than in either MIAR restored or SERC wetlands in 2011 (Table 3), and the natural sites had the highest diversity of trees. Tree species that occurred at all three wetland types were *Acer rubrum* (red maple), *Liquidambar styraciflua* (sweet gum), *Pinus taeda (*loblolly pine*)*, and *Quercus phellos* (willow oak). The CEAP and SERC restored wetlands, in addition to having the four species just listed, both also had *Salix nigra* (Table 3).

Discussion

Study component results clearly demonstrate that both sets of restored sites are on a trajectory to become more similar to natural sites, both in terms of plant community composition and function. Sipple and Klockner (1984) found that natural depressional wetlands were typically forested on the Delmarva, but that other vegetation types could occur; when this situation exists, the interior areas of wetlands are often dominated by herbaceous species and outer margins are dominated by trees. Sampling the SERC sites 15 years after the original study clearly demonstrates that if undisturbed restored depressional wetlands within the study region should eventually be partially or completely dominated by trees with species compositions similar to natural wetlands (Sipple and Klockner 1984; Tyndall et al. 1990; Tyndall 2000, 2001; McAvoy and Bowman 2002). As with natural sites, the plant composition of restored sites will be largely determined by hydroperiod. Nutrient concentrations in herbaceous biomass were lower in the restored wetlands relative to the prior converted cropland sites, an indication that the agriculture legacy was declining. Unlike N and P concentrations, the concentration of carbon did not decrease. This fact, as well as the increased presence of trees with time at the restored sites, indicates increased potential for carbon storage in wetland restorations with time. However, herbaceous biomass data indicate that the restored sites are still at an early stage of succession, a state that occurs naturally, but less commonly, in depressional wetlands.

Phosphorus and nitrogen concentrations in aboveground plant tissues were found to be highest in prior converted cropland sites and lowest in natural sites, with intermediate values at restored sites (Figure 3). This is not surprising based on the current management of the prior converted croplands and agricultural legacy of the restored sites, as well as their adjacency to currently cultivated croplands. These differences suggest that nutrient concentrations in restored wetland vegetation are likely to decrease over time, if not influenced too greatly by adjacent croplands. Results of the comparison of N and P concentrations in samples collected at the SERC sites in 1996 and 2011 also provide evidence that nutrient concentrations will continue to change over time (Figure 5).

Differences in N:P ratios provide insight into the direction of nutrient limitations (Koerselman and Meulmann 1996) with values less than 14 indicative of nitrogen limitations, the condition found at all three types of sites (Figure 3). The N:P ratios of the SERC sites indicate that ratios in restored sites are likely to continue to increase (Figure 5). The suggestion that our sites were all nitrogen limited is the opposite of what has been suggested by Kirkman et al. (2012) for depressional wetlands in the southeastern U.S. Kirkman et al. concluded that phosphorus availability limited primary production in depressional wetlands in the southeast. However, high soil P saturation has been found at MIAR sites (Chapter B4), likely due, at least in part, to the application of organic manures (i.e., chicken manure). Furthermore, in agricultural settings, we would expect a shift in nutrient limitations from phosphorus to nitrogen. Phosphorus cycling in wetlands is closely bound by biological processes and movement of organic phosphorus into wetland substrates could result in a long-lived pool of available phosphorus (Cheesan et al. 2014**)** and a shift toward nitrogen limitation promoted by denitrification. The long-term consequence of the trends in N:P ratios are unknown, but as carbon storage increases in biomass, especially in trees and shrubs, we predict that nutrient utilization would likely increase and restored wetlands will provide more and more nutrient retention services over time.

Conclusions

Data on species composition of restored and natural sites reported in Chapter B6 and data reported here demonstrate that restored depressional wetlands provide important ecological services through increased primary production, as well as increased plant biodiversity. Biomass production provides important pathways for nutrient, including carbon, storage. Resampling the SERC restored wetlands 15 years after they were initially studied supports the viewpoint that successional processes continue, albeit slowly, toward the establishment of trees, representing the conditions of most undisturbed natural Delmarva bays. We hypothesize that restored Delmarva bays will ultimately develop into alternate steady state ecosystems with many of the characteristics of pre-existing natural Delmarva bays, provided anthropogenic disturbances like mowing and excavation are limited, and that the goods and services provided by restored depressional wetlands should be maintained or continue to increase over time.

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Tables

Table 1A: ANOVA comparisons of biomass and nutrient standing stocks between MIAR sites (natural [NAT] and restored [RST]). Nutrient values were calculated using herbaceous biomass only (adapted from McFarland et al. 2015).

Metric	*Wetland Types*	*F ratio*	*df*	*Significance*
Herbaceous Biomass	NAT, RST	F = 32.39	1,28	p < 0.0001
Woody Biomass	NAT, RST	F = 63.02	1,28	p < 0.0001
Total Phosphorus	NAT, RST	F = 50.28	1,28	p < 0.0001
Total Nitrogen	NAT, RST	F = 31.61	1,28	p < 0.0001
Total Carbon	NAT, RST	F = 31.16	1,28	p < 0.0001

Table 1B: ANOVA comparisons of nutrient concentrations between MIAR sites (natural [NAT], restored [RST], and prior converted cropland [PCC]). Nutrient values were calculated using herbaceous biomass only (adapted from McFarland et al. 2015).

Metric	*Wetland Types*	*F ratio*	*df*	*Significance*
% Phosphorus	NAT, RST, PCC	F = 19.99	2,41	p < 0.0001
% Nitrogen	NAT, RST, PCC	F = 12.83	2,41	p < 0.0001
% Carbon	NAT, RST, PCC	F = 8.44	2,41	p = 0.0009
N:P	NAT, RST, PCC	F = 9.70	2,41	p = 0.0003

Table 2A: ANOVA comparisons of biomass and nutrient standing stocks in SERC sites between the first (1996) and second (2011) studies. Total nutrient values were calculated using herbaceous biomass only. Sample size was n=9 for both years (adapted from McFarland et al. 2015).

Metric	*F-ratio*	*df*	*Significance*
Herbaceous Biomass	5.91	1,16	p = 0.0272
Total Phosphorus	7.56	1,16	p = 0.0143
Total Nitrogen	11.78	1,16	p = 0.0034
Total Carbon	5.78	1,16	p = 0.0286

Table 2B: ANOVA comparisons of nutrient concentrations in SERC sites between the first (1996) and second (2011) studies. Nutrient concentration values were calculated using herbaceous biomass only. Sample size was n=9 for both years (adapted from McFarland et al. 2015).

Metric	*F-ratio*	*df*	*Significance*
% Phosphorus	1.66	1,237	p = 0.2155
% Nitrogen	8.22	1,237	p = 0.0112
% Carbon	1.40	1,237	p = 0.2542
N:P	7.09	1,237	p = 0.0170

Table 3: Tree species presence in MIAR restored wetlands, SERC wetlands in 2011, and MIAR natural wetlands. Frequency of tree species occurrence data values represent mean frequency of species occurrence throughout wetlands ± 1 S.E.; average tree diameters (cm) represent average diameter at breast height (DBH) of individual trees measured of each species within each wetland type ± 1 S.E.; Diameter of largest tree selected within each species for each wetland type, and is a single value without deviation. Sample size (n) represents the number of wetlands where trees were documented; only wetlands with trees were used in this analysis (adapted from McFarland et al. 2015).

Species	***Common name***	***MIAR Restored (n=5)***			***SERC 2011 (n=9)***			***Natural (n=13)***		
		Frequency of occurrence	*Average tree diameter*	*Diameter of largest tree*	*Frequency of occurrence*	*Average tree diameter*	*Diameter of largest tree*	*Frequency of occurrence*	*Average tree diameter*	*Diameter of largest tree*
Acer rubrum	Red Maple	0.20 ± 0.20	8.5 ± 0.8	21.1	0.22 ± 0.12	11.7 ± 3.5	37.2	0.71 ± 0.097	19.3 ± 0.66	112.8
Carya tormentosa	Mockernut Hickory	0.20 ± 0.20	7.1 ± 1.0	15.2	0	0	0	0	0	0
Diospyros virginiana	Common Persimmon	0	0	0	0.41 ± 0.16	11.0 ± 0.8	58.1	0.026 ± 0.026	4.3 ± 7.8	5.3
Fagus grandifolia	American Beech	0	0	0	0	0	0	0.026 ± 0.026	4.5 ± 1.7	6.1
Fraxinus pennsylvanica	Green Ash	0	0	0	0	0	0	0.013 ± 0.013	4.5 ± 4.5	4.5
Ilex opaca	American Holly	0	0	0	0	0	0	0.14 ± 0.07	10.2 ± 1.3	22.2
Liquidambar styraciflua	Sweet Gum	0.83 ± 0.11	7.4 ± 0.8	28.4	0.67 ± 0.15	10.2 ± 0.5	20.6	0.66 ± 0.098	17.6 ± 0.7	53.2
Magnolia virginiana	Sweetbay Magnolia	0	0	0	0	0	0	0.25 ± 0.084	9.5 ± 0.3	22.1
Nyssa biflora	Swamp Tupelo	0	0	0	0	0	0	0.15 ± 0.10	19.4 ± 0.2	62.3
Nyssa sylvatica	Blackgum	0	0	0	0	0	0	0.19 ± 0.092	15.5 ± 0.2	38.8
Pinus taeda	Loblolly Pine	0.20 ± 0.20	5.1 ± 0.1	5.5	0.30 ± 0.13	14.7 ± 1.9	31.7	0.16 ± 0.090	26.2 ± 1.3	47.0
Prunus serotina	Black Cherry	0	0	0	0.11 ± 0.11	15.6 ± 0.9	16.5	0.013 ± 0.013	7.8 ± 7.8	7.8
Quercus alba	White Oak	0	0	0	0	0	0	0.051 ± 0.051	63.2 ± 8.5	78.0
Quercus coccinea	Scarlet Oak	0	0	0	0	0	0	0.026 ± 0.026	19.4 ± 6.6	29.5
Quercus palustris	Pin Oak	0.10 ± 0.10	17.7 ± 17.7	17.7	0	0	0	0.026 ± 0.026	35.7 ± 35.7	35.7
Quercus phellos	Willow Oak	0.13 ± 0.13	30.1 ± 4.2	42.9	0.037 ± 0.037	9.1 ± 9.1	9.1	0.095 ± 0.051	25.0 ± 4.7	71.6
Quercus rubra	Red Oak	0	0	0	0	0	0	0.026 ± 0.026	43.1 ± 43.1	43.1
Salix nigra	Black Willow	0.10 ± 0.10	13.9 ± 4.1	29.5	0.28 ± 0.12	14.9 ± 0.7	62.2	0	0	0

Figures

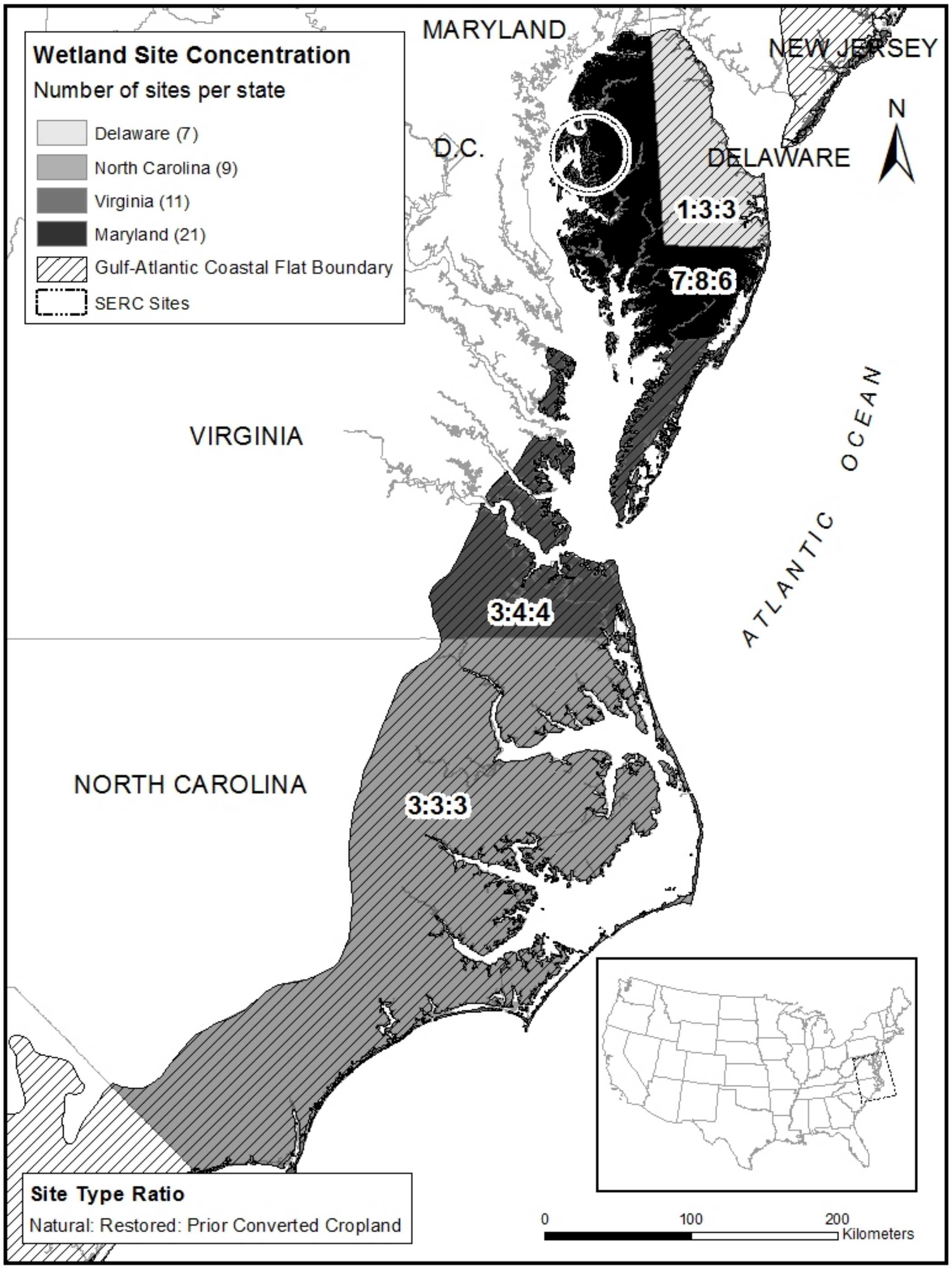

Figure 1: Map of study sites. MIAR sites in Delaware, Virginia, Maryland, and North Carolina. Numbers within counties represent ratios of Natural (14): Restored (17): Prior Converted Cropland (16) sites. SERC sites (9) were only in Queen Anne's, Talbot, and Kent counties of Maryland (adapted from McFarland et al. 2015).

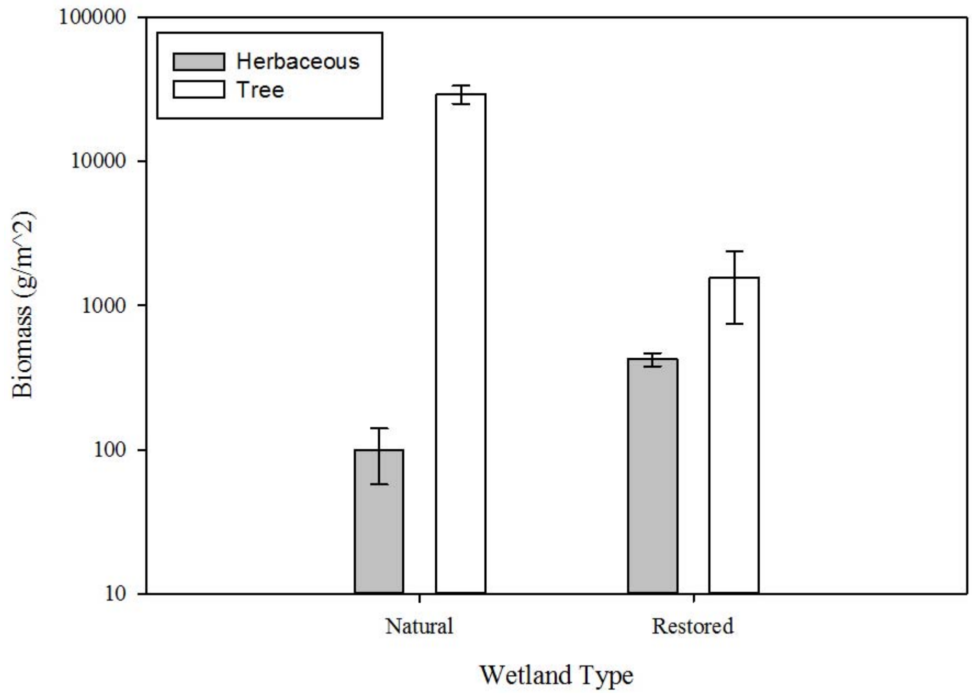

Figure 2: Comparison of biomass (herbaceous and tree) in natural and restored MIAR wetlands. Values standardized to grams biomass per meter squared, and plotted on a logarithmic scale. Plotted values are mean ±1SE. All pairs of means significantly different from each other (see Table 1A). Sample sizes were n=14 for natural sites and n=17 for restored sites. For the purposes of axis comparison between the two biomass types, the y-axes are log transformed (McFarland et al. 2015).

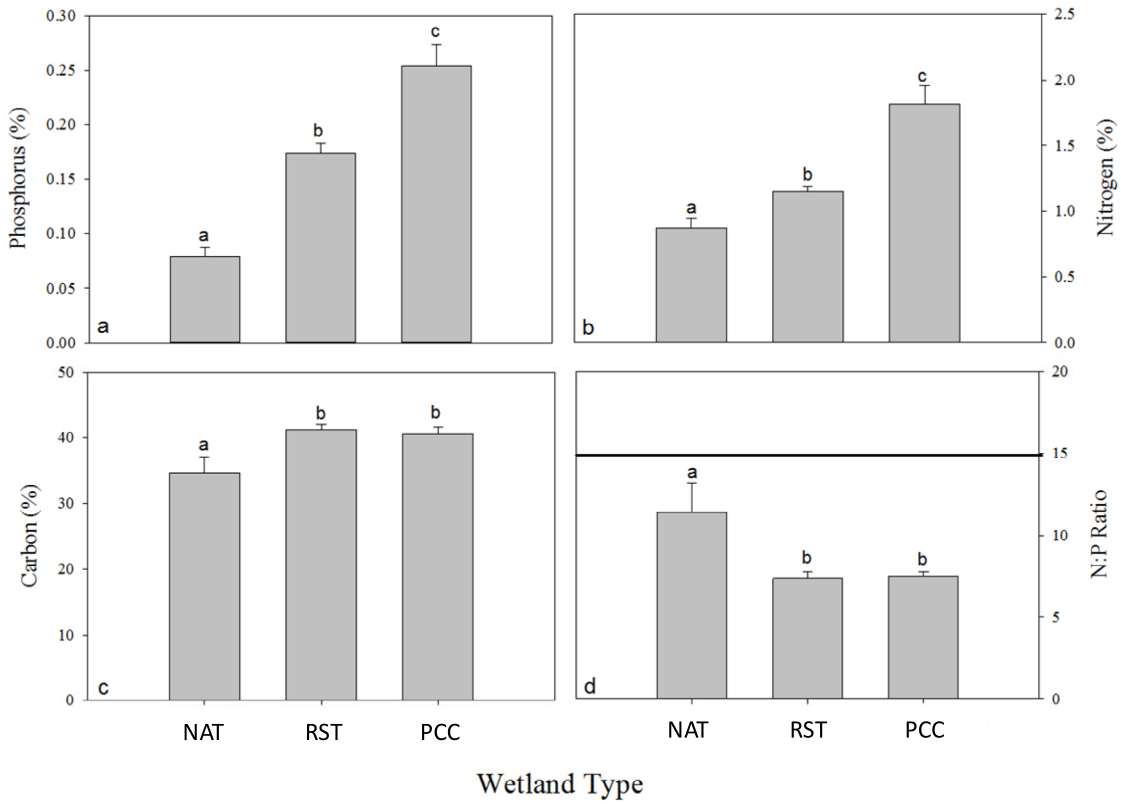

Figure 3: Mean concentrations (%) of A) Phosphorus, B) Nitrogen, C) Carbon, and D) ratio of N:P in aboveground herbaceous biomass at MIAR sites. Values are means + 1 standard error. Statistical comparisons were made between wetland type and the results are indicated with letters above the bars at $p \leq 0.05$ (See Table 1B). Only subplots that had vegetation were used in this analysis; sample sizes were n=13 for natural sites [NAT], n=16 for prior converted cropland sites [PCC], and n=17 for restored sites [RST]. The horizontal line in figure D is the line of non-limited N:P concentrations in freshwater wetlands (*Güsewell et al. 2003*), below which indicates N limitation (adapted from McFarland et al. 2015).

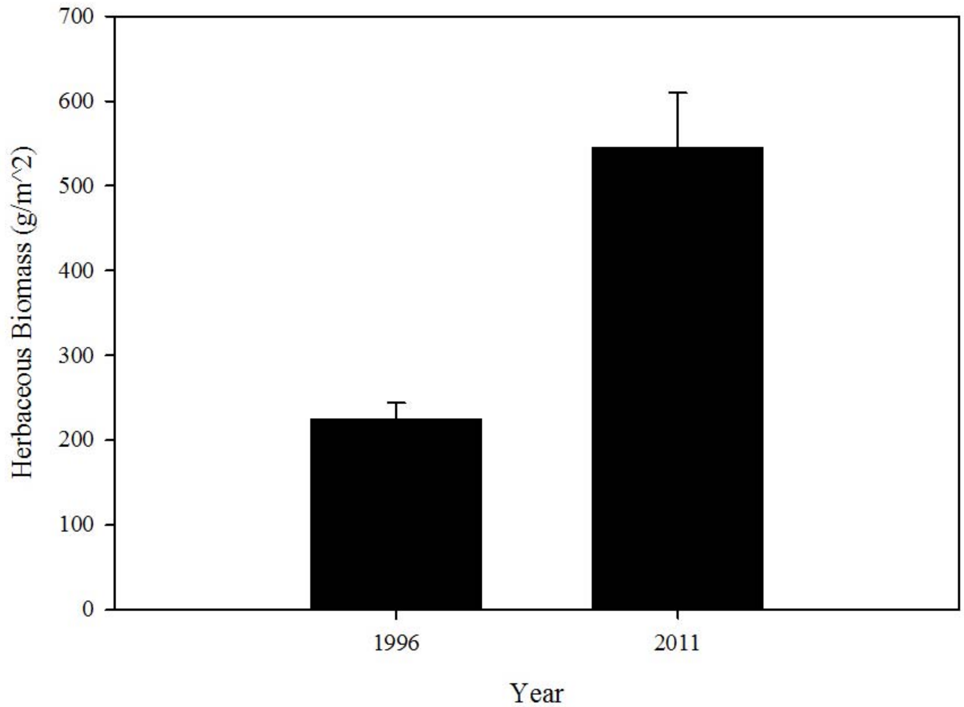

Figure 4: Comparison of herbaceous biomass in SERC wetlands in 1996 and 2011. Values standardized to grams biomass per meter squared. Plotted values are mean + 1 standard error. Means are significantly different from each other (see Table 2A). Sample size was n=9 wetlands from 1996, n=9 wetlands from 2011 (McFarland et al. 2015).

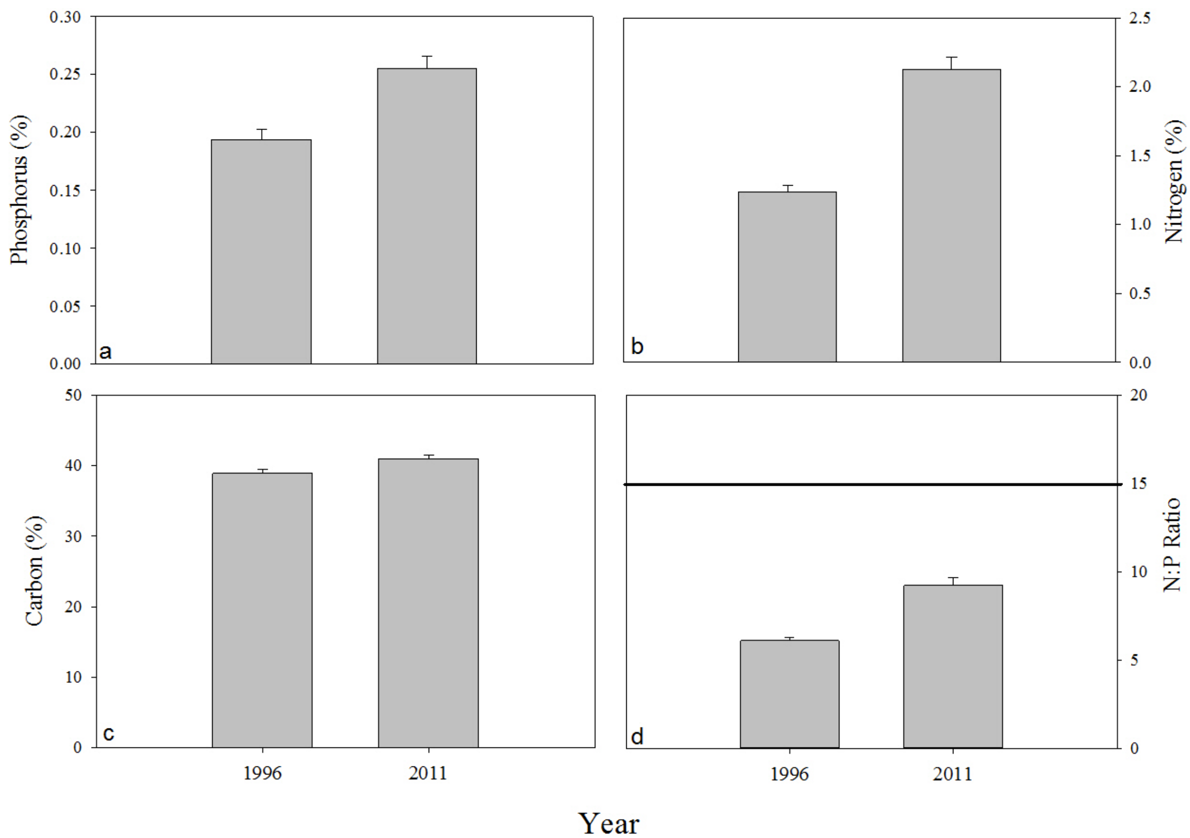

Figure 5: Mean concentrations (%) of A) Phosphorus, B) Nitrogen, C) Carbon, and D) the ratio of N:P in aboveground biomass of herbaceous vegetation in SERC wetlands in 1996 and 2011. Values are means + 1 standard error. Means are significantly different from each other in all figures (see Table 2B). Only subplots that had vegetation were used in this analysis; sample size was n=9 wetlands from 1996, n=9 wetlands from 2011. The horizontal line in D is the line of balanced N:P concentrations in freshwater wetlands from *Güsewell et al.(2003*) (McFarland et al. 2015).

2. Soil Organic Carbon Storage

Key Findings

-In Delaware, Maryland, and Virginia soil carbon stocks in prior converted croplands were 63 % lower than carbon stocks in natural wetlands. Although the difference was not statistically significant, carbon stocks were lower in restored wetlands relative to prior converted croplands. This may, in part, be due to the relatively young age (5-10 years) of the restorations considered as part of this study in addition to implementation practices – see below.

-In Delaware, Maryland, and Virginia wetland restoration implementation practices were found to be significantly related to soil carbon stocks. While wetlands that were restored through practices which require less disturbance of the soil profile (e.g., berms and ditch plugs) had soil carbon stocks that were similar to prior converted croplands, wetlands that were restored via excavation had significantly lower soil carbon stocks. Relatively carbon rich excavated soil was found in topographic highs (e.g., berms) where oxidation and loss of carbon to the atmosphere is more likely. Excavation can also cause soil compaction, which can impede root growth and therefore carbon production.

-In North Carolina, differences in bulk density and percent carbon in the upper horizons suggest that subsidence and carbon loss has occurred, probably due to conversion to agriculture. Elevated carbon contents with lower bulk densities at the surface suggest that carbon sequestration is occurring following restoration.

Recommended Practices: To promote carbon sequestration and support climate regulation services, excavation should be avoided through enhanced targeting, including the examination of soils. When excavation cannot be avoided excavated topsoil should be replaced. Soil compaction should be avoided and water depths should be shallow enough to encourage plant growth.

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Introduction

Wetlands are effective carbon sinks because their primary productivity often exceeds the rate of decomposition. Saturated soil conditions often lead to an anaerobic soil environment, which inhibits decomposition and allows carbon to accumulate (Collins and Kuehl 2001). Although wetlands comprise approximately 3.5 % of the global terrestrial surface (Bridgeham et. al. 2006; Batjes 1996; Matthews and Fung 1987), they contain about 23 % of global soil carbon. In the conterminous United States, wetlands comprise about 5.5 % of the terrestrial surface (Dahl 2000), but are estimated to store roughly 22 % of soil organic carbon (Bridgeham et. al. 2006; Gou et. al. 2006). Carbon sequestration and storage within wetlands is of particular interest, since rising concentration of atmospheric carbon dioxide (CO_2) is contributing to climate change. Atmospheric carbon dioxide is expected to rise at an accelerated rate over the next several decades (Raupach et al. 2007) further emphasizing the importance of wetlands and wetland conservation practices for regulation of CO_2, an important greenhouse gas. The objective of this MIAR CEAP-Wetland study component was to assess the effects and effectiveness of wetland restoration practices in regards to greenhouse gas (CO_2) regulation in the Mid-Atlantic Coastal Plain by comparing soil carbon stocks along a human alteration gradient, including natural and restored wetlands, as well as prior converted croplands.

Methods

At each of the 48 MIAR study sites a minimum of two soil profile descriptions were made using shallow excavated pits and a bucket auger for deeper observations. One profile was randomly selected to be sampled for further analysis. Duplicate bulk density samples were collected from each horizon to a depth of 100 cm using the core method (Blake and Hartage 1986). Where high water tables impeded the use of the core method, and the soil material was soft enough, a 10 cm long half core (5 cm diameter) was collected using a McCauley sampler. When field conditions precluded the collection of samples to a depth of 1 m, percent carbon and bulk density for the deepest horizon was assumed to continue to the depth of 1 m. Bulk density samples were dried at 60 °C until reaching a constant weight. After bulk density was determined, samples were crushed and homogenized. A subsample was finely ground using a roller mill by placing the sample in a glass vial with two steel rods for 24 to 48 hours. Carbon analysis was performed in duplicate using the dry combustion method (Nelson & Sommers 1996) on a LECO TruSpec CN Analyzer. Total carbon stocks in each horizon were calculated using bulk density, percent carbon, and thickness of the horizon, and reported on an area (m^2) basis. Duplicate analyses for each horizon were averaged. Calculated carbon in all horizons to a depth of 1 m was then summed to obtain total carbon stocks (kg C m^{-2}) for each profile. Total carbon stocks were analyzed using an ANOVA based on mean values, and comparisons were made by land use class, followed by Tukey's test to separate means.

Sites in North Carolina differed dramatically from the other study sites, and were either Histosols or soils that had histic epipedons. These soils contained nearly seven times more carbon ($p < 0.0001$) than those in other parts of the study area. Soils collected in Maryland, Delaware, and Virginia were predominantly mineral soils and did not differ significantly in carbon stocks by state or subregion ($p=0.40$). Therefore North Carolina sites were analyzed independently. The North Carolina region contained three sites in each land use, while the remainder of the region included 11 natural and 15 restored wetland sites, as well as 13 prior converted cropland sites. Wetland hydroperiod at restored sites was established by either plugging drainage ditches or scraping the soil surface (i.e., excavation). These techniques were compared using a T-Test to determine if there was a significant difference in carbon stocks between restoration techniques.

Results and Discussion

Maryland, Delaware, and Virginia Sites

In general, soils at sites in Maryland, Delaware, and Virginia had loamy surface textures that transitioned into coarser substrata. Natural sites commonly contained thin Oe and occasionally Oa horizons, over thick A horizons. A single pedon among the eleven natural sites was a Histosol, and one other pedon had a histic epipedon. Of the other nine natural sites, four were poorly drained and five were very poorly drained based on soil profile characteristics. Mean bulk density for the upper 30 cm of the profile was low (0.92 g cm^{-3}), probably caused by high organic matter content. All prior converted cropland sites were cultivated and therefore lacked organic horizons. Drainage classes are assigned based upon persistent morphological characteristics that form under natural, undrained conditions. Therefore in situations where soils have been drained for agriculture, the drainage class based on morphology may not accurately depict current hydrological conditions. However, it may provide clues regarding the hydrology that was present prior to drainage. Prior converted cropland sites exhibited a wide range of drainage classes. Of the thirteen prior converted cropland sites, two were very poorly drained, five were poorly drained, five were somewhat poorly drained and one was moderately well drained. Mean bulk density for the upper 30 cm of the prior converted cropland sites was 1.53 g cm^{-3}. Wetland hydroperiod was reestablished at the restored sites using two different techniques, plugging of ditches or excavation. At three of the excavated sites coarser textured human transported materials were found at the surface. These excavated sites had a mean bulk density for the upper 30 cm of 1.66 g cm^{-3}, and values ranged from 1.42 to 1.88 g cm^{-3} (median = 1.64). The five sites that were restored by plugging had thicker A horizons. These plugged sites had a mean bulk density for the upper 30 cm of 1.53 g cm^{-3} (median = 1.52). Mean bulk density for the upper 30 cm of all restored wetlands across both restoration techniques was 1.59 g cm^{-3}.

Natural sites were found to have significantly greater carbon stocks (21.5 ± 5.2 kg C m^{-2}) than both prior converted cropland (7.95 ± 1.93 kg C m^{-2}; $p < 0.01$) and restored sites (4.82 ± 1.13 kg C m^{-2}; $p < 0.001$). It is likely that soil carbon was lost following conversion of natural ecosystems to agricultural due to drainage, alteration of plant communities, and cultivation (Everett 1983). Loss of approximately 63 % of carbon following the conversion of wetlands to agriculture was more than the 20 to 40 % loss in carbon stocks others reported (Anderson 1995; Davidson and Ackerman,1993; Gleason et al. 2008; Mann 1986). However, most of these studies were not conducted on wetlands. In non-wetland situations the primary driver of change is the alteration of plant communities and cultivation alone (Six et al. 2002).

Carbon stocks at the restored sites were not statistically different from the prior converted cropland sites, and were actually slightly lower (Figure 1). Primary factors that may have contributed to relatively low carbon stocks at restored sites include implementation techniques and time since restoration. As mentioned earlier, 10 of the 15 sites were restored through excavation of the soil surface. Excavation could remove the carbon rich surface horizons and bring the subsoil (Bg) horizons closer to the surface, thus lowering carbon stocks in the top 1 m. This result has been observed in other studies where excavation was used to achieve wetland hydrology (Ballantine and Schneider 2009; Fennessy et al. 2008; Stolt et al. 2000). One study in particular reported 36 % less carbon in the upper 40 cm of restored wetlands compared to their agricultural counterparts, which they attributed to grading and scraping in order to fill ditches and create micro-topography (Bruland et al. 2003). Other similar studies reported carbon stocks that were significantly lower than similar agricultural sites or were no different from their agricultural counter parts, but do not provide details on the restoration technique (Gleason et al. 2008; Marton et al. 2014). The organic rich horizons that are removed are usually used to form dykes or berms to retain water or mounds to create micro-topography. Often these materials end up in an aerobic environment, which would enhance the oxidation of the soil carbon. The remaining five wetland restorations that were considered as part of this study were restored through practices that did not require the excavation of topsoil, such as the plugging of ditches. These wetland restoration implementation practices had no observable effect on carbon stocks. When restoration was implemented without excavation, carbon stocks (6.06 ± 1.50 kg C m^{-2}) were found to be greater than when excavation was used (2.70 ± 0.38 kg C m^{-2}; $p=0.09$) (Figure 2). In addition to removing soil carbon, excavation can increase bulk density, which has been found to inhibit root growth (Shierlaw & Alston 1984). Although there was not a statistically significant difference in bulk density between the two restoration techniques, a small increase in bulk density can profoundly limit root growth (Daddow and Warrington 1983).

The restored wetlands in this study were only 5-10 years old, which represents a relatively short time period in which to observe change. Therefore, the lower than anticipated carbon stocks could also be the result of the youthful age of the restorations. A study conducted in New York reported that soil organic matter did not begin to significantly accumulate until at least 14 years after restoration, and then it was only within the upper 5 cm (Ballantine and Schneider 2009). A significant increase was observed throughout the soil profile after 35 years, although even after 55 years soil organic matter levels were still less than half of their natural counter parts. The accumulation of soil carbon is a slow process. It may take decades for significant increases in carbon stocks, and may require a century or more before carbon stocks in restored wetlands are similar to those in natural wetlands.

North Carolina Sites

All North Carolina wetlands had very poorly drained soils with four of the nine sites containing Histosols, four having histic epipedons, and one having an umbric epipedon. The natural sites included a Histosol, and soils with a histic epipedon and an umbric epipedon. They all had bulk densities between 0.13 and 0.29 g cm^{-3} (0.22 g cm^{-3} average). While this may suggest that the natural sites have experienced some degree of subsidence following drainage from local ditching (Daniel 1980), they did not appear to be impacted to the same degree as those that had been cultivated. The prior converted cropland sites were also organic-rich, with one site qualifying as a Histosol and two containing histic epipedons. Bulk densities of Oa horizons at these cultivated sites were greater than at the natural sites, with values ranging from 0.46 to 0.86 g cm^{-3} and horizons that were actually cultivated (Oap and Ap horizons) having higher bulk densities (0.73, 0.86, and 0.86 g cm^{-3}). This demonstrates strong evidence of subsidence. Profiles at all three prior converted cropland sites had lower percent carbon in the surface plow layer (Oap) than in the immediately underlying horizon (Oa), and in one case the surface horizon appears to have lost enough carbon to be considered a mineral horizon (Ap). This evidence suggests that primary subsidence and compaction due to dewatering, as well as secondary subsidence caused by loss of carbon due to oxidation (Everett 1983; Ewing and Vepraskas 2006), have occurred in the prior converted cropland sites. The restored sites also had organic rich soils, with two sites being Histosols and one site having a histic epipedon. Bulk densities in the Oa horizons were between

0.29 and 0.73 g $cm^{-3}$with bulk densities of 0.29, 0.37, and 0.57 g cm^{-3} in the surface Oap horizons.

Soil carbon stocks at natural (73.3 ± 27.4 kg C m^{-2}), prior converted cropland (75.5 ± 4.5 kg C m^{-2}), and restored sites (114.6 ± 42.6 kg C m^{-2}) were not significantly different (Figure 3). The quantity of carbon stocks was generally representative of the depth to the mineral subsurface horizons, which may be partially influenced by land use. The effects of land use could be seen in the bulk density of the upper 10 cm, where the natural (0.226 g cm^{-3}) and restored (0.409 g cm^{-3}) sites had lower bulk densities compared to the prior converted cropland sites (0.818 g cm^{-3}; $p=0.001$ and $p=0.004$, respectively). Typical bulk densities of undisturbed organic horizons are about 0.1-0.2 g cm^{-3} (Bruland et al. 2003; Ewing and Vepraskas 2006; Caldwell et al. 2007).

Although soil carbon stocks at restored sites were not significantly different than those at prior converted cropland sites, two of the three restored sites had elevated percent carbon content in surface horizons relative to the immediately subjacent horizons, and bulk densities in these surface horizons (0.29, 0.57, and 0.73 g cm^{-3}) were lower than those of soils at the prior converted cropland sites. The relatively short time since restoration may not have allowed for enough time to lead to any significant differences in carbon stocks within the first meter between wetland types in North Carolina, but this does suggest that restoration may be facilitating carbon accumulation.

Conclusions and Implications

The drainage and conversion of depressional wetlands to agricultural land has greatly lowered soil carbon stocks at prior converted cropland sites in Delaware, Maryland and Virginia, likely resulting in the release of carbon dioxide, an important greenhouse gas, to the atmosphere. In North Carolina wetland conversion has probably led to accelerated subsidence, compaction, loss of carbon, and release of carbon dioxide to the atmosphere. It is likely that wetland restoration will encourage the capture and sequestration of carbon, but this is typically a slow process, likely taking decades if not centuries for restored wetlands to recover soil carbon levels similar to natural wetlands. Wetland restoration implementation and management practices can significantly influence the ability of restored wetlands to accumulate and retain soil carbon. Excavation, which not only leads to the removal of carbon rich topsoil but also soil compaction, should be avoided when possible through enhanced targeting and the promotion of less invasive practices (e.g., plugging of ditches). Enhanced targeting could include both the use of remotely sensed data, such as lidar based digital elevation models, to locate prior converted croplands with relatively large watersheds and high natural relief, as well as the investigation of soil properties to identify areas that sustained relatively long hydroperiods in the past. Examination of soils cannot only identify areas that are more likely to support current wetland hydrology, but also areas with relatively high existing levels of soil carbon that may be better preserved under wetland conditions. When excavation cannot be avoided excavated topsoil should be replaced. Furthermore, soil compaction should be avoided and water depths and hydroperiod should be optimized to promote plant growth

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Figures

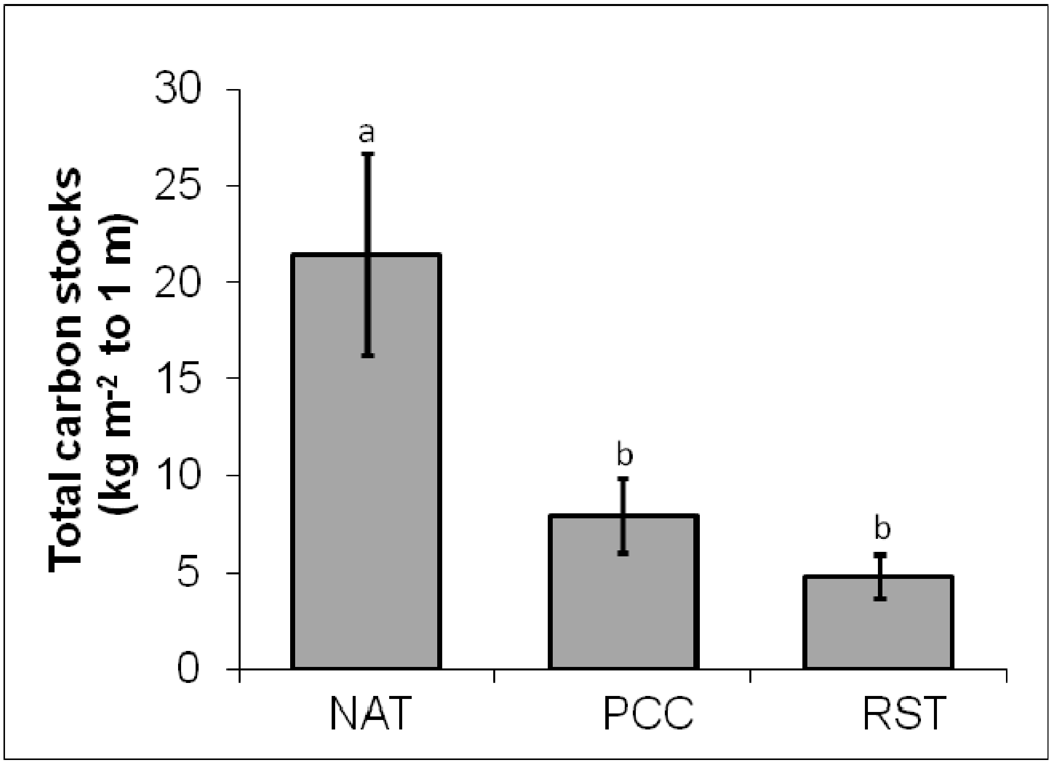

Figure 1: Mean total carbon stocks for natural (NAT, n=11), prior converted cropland (PCC, n=13), and restored (RST, n=15) depressional wetlands located in the Coastal Plain of Delaware, Maryland, and Virginia. Designations using the same lowercase letter indicate that there is no significant difference between the data (adapted from Fenstermacher et al. 2015).

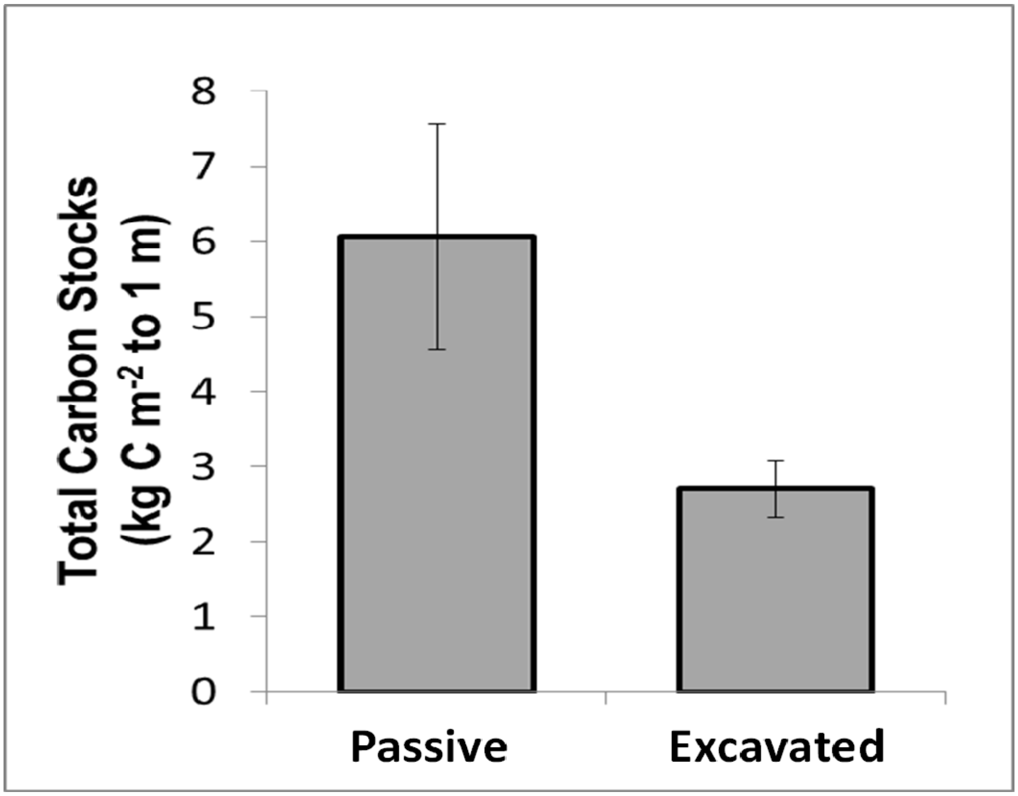

Figure 2: Mean total carbon stocks for the wetland restoration practices of passive techniques, including the use of ditch plugs (n=5) and scraping (n=9) utilized in the Coastal Plain of DE, MD, and VA (adapted from Fenstermacher et al. 2015).

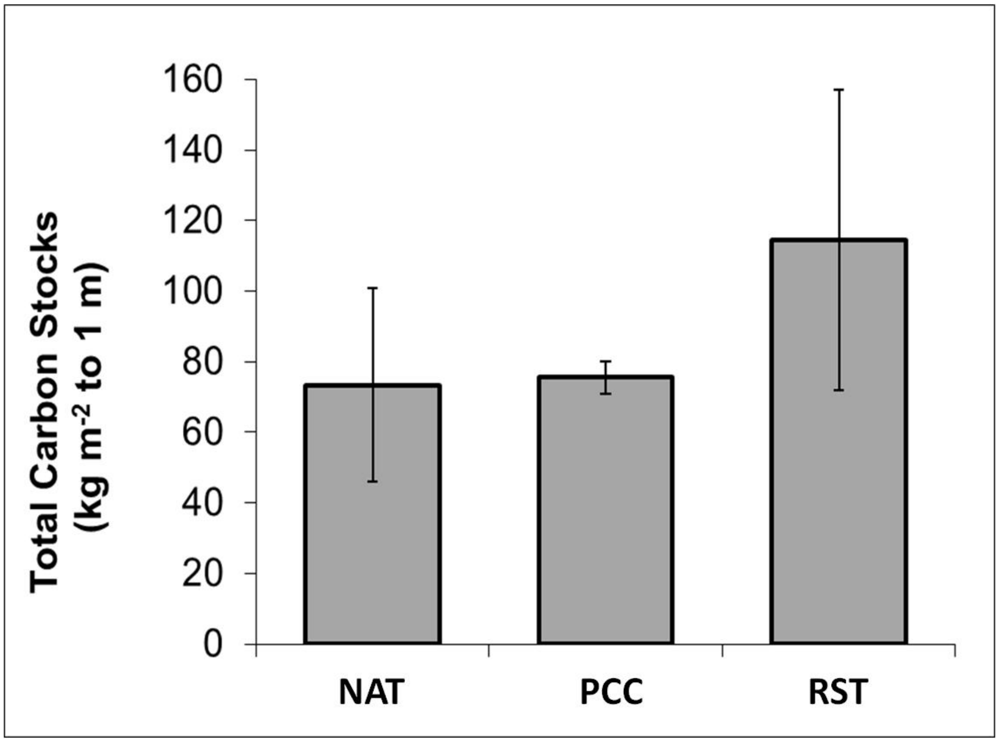

Figure 3: Mean total carbon stocks for natural (NAT, n=3), prior converted cropland (PCC, n=3), and restored (RST, n=3) sites located in the Coastal Plain of North Carolina. No significant difference was observed between land uses (adapted from Fenstermacher et al. 2015).

3. Soil Physicochemical Parameters, Potential Denitrification, and Abundance of Denitrifiers

Key Findings

-Soil physicochemical characteristics indicated a recovery trend post restoration. However, measured parameters exhibited markedly different patterns. Although certain parameters (e.g., P, Zn, pH, and soil moisture) exhibited relatively rapid recovery subsequent to restoration, others maintain an agricultural legacy and may take many years to mimic natural conditions (e.g., Mg); still others, such as Ca and EC, exhibited intermediate values. These trends indicate partial recovery of natural conditions subsequent to wetland restoration.

-The effects of liming were evident within restored wetlands, although reduced relative to prior converted croplands. pH may have a significant effect on not only soil biogeochemistry and nutrient availability (e.g., P chemistry), but also habitat for amphibians and other species.

-The effects of excavation were evident when examining differences in soil parameters across relative landscape positions (i.e., elevation) and between wetland types. Excavation appears to have redistributed not only C, but also nutrients and microbial communities, with impacts on denitrification potential (denitrification enzyme activity [DEA]).

-*Delaware, Maryland, and Virginia*: Denitrification potential (DEA) varied significantly with wetland type. Within wetland types there was a strong relationship between DEA and relative elevation (R^2 >0.90). DEA levels and observed relationships in restored wetlands were more similar to natural wetlands than prior converted croplands. Abundance of the gene *nosZ*, which encodes for nitrous oxide reductase, in restored sites was statistically similar to levels found in prior converted croplands, and higher than the levels found in natural wetlands.

-*North Carolina*: Unlike the primarily mineral soils of the northern portion of the MIAR, soils in North Carolina are relatively organic-rich and acidic. This significant difference in soil physical and chemical character likely led to differences in DEA levels and *nosZ* abundance between sites in North Carolina and other states. Similar to northern sites, DEA at North Carolina sites also generally increased with decreasing relative elevation. However, DEA tended to be lower in restored wetland sites relative to natural wetlands, but higher than prior converted croplands at lower relative elevations. Abundance of the gene *nosZ*, was statistically lower in the restored wetland sites.

Recommended Practices: Although hydrology has been restored, complete restoration of natural physical conditions necessitates not only restoration of hydrology but also wetland soil structure and chemistry. Full recovery of natural biogeochemical properties will likely take decades if not centuries, necessitating commensurate easement periods. The possibility that restored wetland sites will not approximate nearby adjacent natural wetland ecosystem functions in a reasonable timeframe should be taken into consideration, perhaps prompting an emphasis on a modified paradigm/expectations for restored wetlands. Excavation should be avoided through enhanced targeting based on field and geospatial observations. When this is not possible, topsoil should be replaced. Due to the large area that prior converted croplands occupy at the landscape scale (Chapters B5 and B9), and the continued provision of some degree of wetland services at these sites, the implementation of conservation practices that allow continued crop production within prior converted cropland sites (e.g., controlled drainage) should be emphasized in addition to wetland restoration. Significant differences in soil structure and chemistry within the MIAR region may require the adaptation of recommended practices based on these conditions.

***Chapter Sources*:**

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Introduction

In this region, as well as many other parts of the world, one of the important biogeochemical functions of wetland ecosystems is their cycling of nitrogen. Wetlands remove significant quantities of nitrogen via denitrification under anaerobic conditions. This is especially true when excess nitrogen enters wetlands in the form of nitrate, because nitrate is quite readily converted to dinitrogen gas via denitrification. The level of denitrification is frequently limited by available nitrogen rather than carbon (Hunt et al. 2004). Unfortunately, denitrification does not always go to completion with the formation of dinitrogen gas. Under some conditions, such as low Carbon/Nitrogen ratios, denitrification will produce nitrous oxide, a potent greenhouse gas which also degrades air quality (Dodla et al. 2008; Hunt et al. 2007; Ullah and Moore 2011). While this is of concern, natural wetlands do not appear to be a uniformly large contributor of nitrous oxide emissions (Audet et al. 2013; Jacinthe et al. 2012; Morse et al. 2012). Less information is available regarding the effect of wetland restoration on the nitrogen cycle, including denitrification and nitrous oxide emissions. Numerous methods have been employed to assess denitrification; two commonly used methods are denitrification enzyme activity and quantification of denitrifying gene abundances. The goal of this study component was to more fully understand the effects of wetland restoration on denitrification and nitrous oxide emissions in order to better assess restoration effectiveness for providing greenhouse gas and pollutant (nitrogen) regulation services in the MIAR. The specific objectives of this study component were to assess the following: 1) physiochemical conditions; 2) denitrification enzyme activity (DEA); and 3) *nosZ* gene abundances within soils of the natural and restored wetlands, as well as prior converted croplands.

Methods

A previous report demonstrated significant differences between physicochemical properties of soils in North Carolina and those in Maryland, Delaware, and Virginia (Kluber et al. 2014). While MIAR sites in Maryland, Delaware, and Virginia primarily had mineral soils, soils found at sites in North Carolina had higher levels of soil organic matter and were more acidic. Due to this fundamental difference in soil chemistry and structure, northern (Maryland, Delaware, and Virginia) and southern (North Carolina) sites were analyzed separately as part of this study component, although field and laboratory protocols were identical. There were a total of 39 northern sites (14 restored, 11 natural, 14 prior converted cropland). In the southern region a total of 9 sites were studied, including 3 restored, 3 natural, and 3 prior converted cropland sites.

All sites were stratified based on potential relative wetness using topography as a primary indicator. The gradient, along with sampling points, were determined prior to field sampling using LiDAR based digital elevation models (DEMs). The DEMs were used to define four evenly proportioned topographic classes within each site using ArcGIS (ESRI, Redlands, CA). Each topographic class served as one of four sampling locations within each site, and are referred to as "relative wetness" (RW) classes 0 (lowest elevation/wettest) through 3 (highest elevation/driest). Within each relative wetness class, sampling points were selected randomly. At each sampling point, three soil samples were collected from the upper 10 cm within a 0.5 m radius. Soil samples were combined into a composite sample, which was placed on ice and transported to the lab for analysis. Soil temperature, electrical conductivity and moisture were measured *in situ*. Sampling of each site was performed over a three year period from June 2009 until May 2011, during which each site was sampled at least three times including both spring and fall.

Soil carbon (C) and nitrogen (N) were measured using a TruSpec CN analyzer (Leco Corp, St Joseph, MI). Soil pH was measured using a 1:1 mixture of soil and water. Air dried soil samples were extracted using a Mehlich-1 solution and subsequently analyzed using inductively coupled plasma (ICP) atomic emission spectrometry (Vista Pro, Varian Inc., Walnut Creek, CA) for aluminum (Al), calcium (Ca), copper (Cu), iron (Fe), potassium (K), magnesium (Mg), sodium (Na), phosphorus (P), and zinc (Zn). Soil samples were also extracted using water, and anions (chlorine [Cl], nitrate nitrogen [NO_3-N], sulfate sulfur [SO_4-S], and phosphate phosphorus [PO_4-P]) were measured by Chemically Suppressed Ion Chromatography (IC) using a Dionex 2000 Ion Chromatograph (ASTM Standard D4327-11). All samples were analyzed for DEA using the acetylene inhibition method (Miller et al. 2012; Tiedje 1994); treatments were as follows:

(1) Complete denitrification: acetylene was added to the headspace at a final concentration of 5% (vol/vol) to block denitrification at the nitrous oxide (N_2O) reduction step, resulting in an increased accumulation of N_2O, a portion of which would typically be reduced to nitrogen gas (N_2).
(2) Incomplete denitrification: denitrification was allowed to occur unimpeded, with N_2O accumulating at natural rates.
(3) Potential complete denitrification: addition of nitrate (NO_3) in non-limited quantities and acetylene (see above) to measure maximal enzyme activity rates with blockage at the N_2O reduction step.
(4) Potential incomplete denitrification: addition of NO_3 in non-limited quantities to measure maximal enzyme activity rates.

Soil DNA extraction was performed using a PowerSoil DNA Extraction Kit (MO BIO Laboratories Inc., Carlsbad, CA) according to manufacturer specifications. All qPCR assays were performed using the LightCycler 480 Real-Time PCR Detection System (Roche Diagnostics, Indianapolis, IN); data were collected and processed using the LightCycler 480 software package. A *nosZ* DNA standard, derived from the linearized plasmid pCPDnosZ1 (Ducey et al. , 2011), was utilized to develop a standard curve from between 10^1 and 10^9 copies per reaction; this standard was also used to calculate an amplification efficiency of 1.90 according to the equation: $E = 1 + 10^{[-1/slope]}$ (Pfaffl, 2001).

All data were statistically analyzed using SAS v 9.3 (SAS Institute Inc., Cary, NC). DEA rates were analyzed using the GLIMMIX (General Linearized Mixed Models) procedure, with sites, sampling dates, and laboratory replicates pooled and considered random. Land use and the relative wetness class variable were considered fixed. DEA treatments were $\log^{10}$ transformed to meet normalization criteria and analyzed using the least squares mean (LSM) method; treatment differences of analyzed variables were compared using the pdiff option. The T value grouping for treatment LSM was $P \leq 0.05$. Soil physicochemical measures and *nosZ* gene abundances were analyzed using the GLM (General Linear Model) procedure, and Duncan's multiple range test ($P \leq 0.05$) was used to detect statistical differences. Relationships between DEA rates and *nosZ* gene abundances with environmental variables were performed using regression analysis. For visualization of physicochemical characteristic differences between land use, principle component analysis (PCA) plots, using a Sorensen distance measure, were produced in PCORD v6.0 (MjM Software Design, Geleden Beach, OR).

Results and Discussion

Soil physiochemical characteristics: Northern sites (Delaware, Maryland, and Virginia)

At northern sites natural wetlands were statistically different from prior converted croplands in all of the soil physiochemical characteristics (Tables 1, 2, and 3). Measured parameters in restored wetlands were typically between those of natural and prior converted cropland sites, although the affinity of restored site values varied greatly between measured parameters. For example, natural wetland soils had soil C contents three times higher than either the restored or prior converted croplands. In natural wetlands, soil carbon was greatest in the lowest landscape position (RW0) and lowest at the highest landscape position (RW3). Prior converted croplands exhibited a similar, but less pronounced pattern. In contrast, restored wetlands varied little with relative elevation. The lack of this trend is likely due to excavation of restored soils at the lowest landscape positions. Natural wetland C/N ratios could be suspect for incomplete denitrification. The likelihood of incomplete denitrification is even higher at restored wetlands and prior converted cropland sites, which had even lower C/N ratios (Hunt et al. 2007; Klemedtsson et al. 2005). In the case of soil pH, natural wetlands were quite acidic, while restored and prior converted cropland sites were more neutral. This was likely due to liming prior to cultivation (Hue and Licudine 1999). With the exception of the highest elevation in the restored sites, prior converted cropland sites were more neutral than the restored, indicating movement towards recovery of natural conditions. One of the greatest effects of restoration on physiochemical characteristics was found with electrical conductivity (EC) values, which were significantly lower in restored wetlands compared to prior converted croplands. Higher EC values are indicative of an agricultural legacy, as fertilizer use has been demonstrated to increase soil EC (Agassi et al. 1981). Soil moisture differences likely explain some of these changes. Natural wetlands were significantly wetter than prior converted croplands, and restored wetlands showed an intermediate pattern.

Significant differences also occurred in plant available nutrient content between wetland types. For plant available Al, Ca, Cu, K, and Mg, natural wetlands were statistically different from both restored and prior converted cropland sites (Table 2). For Fe, P and Zn, restored wetlands were more similar to the natural wetlands than to prior converted croplands. As for Na and Mg, restored wetlands were more similar to prior converted croplands. These values indicate partial recovery of soil nutrient content subsequent to restoration. At the individual landscape level, the highest nutrient contents were mostly in the lowest elevation for the natural wetlands and prior converted croplands. However, only Al, Fe, K, Na, P, and Zn were highest in the lowest elevation of the restored wetland. Again this pattern may have been influenced by excavation. In restored wetlands, reduced P, compared to prior converted cropland sites, without an increase in aluminosilicates (Al; compared to natural wetlands) indicates an incomplete transition of restored wetlands toward natural wetland conditions.

Soil physiochemical characteristics: Southern sites (North Carolina)

Similar to northern sites, physicochemical parameters (Tables 4, 5, and 6) indicate that restored wetland sites continue to exhibit an agricultural legacy, but effects of restoration are also evident. In restored wetlands, mean values for Al, Fe, and Mg are similar to levels found in prior converted cropland sites. Other properties appear to be in transition between their previous prior converted cropland status and natural wetlands, such as Ca, pH, EC, moisture, soil temperature, and bulk density. Still other soil properties, such as Na, P, Zn, TN and NO_3 (nitrate), have come to approximate natural wetland conditions

Similar to northern sites, immediate agricultural inputs, which can be measured by NO_3 and plant available P, were significantly lower in restored wetlands as compared to their prior converted cropland counterparts. This is a clear indication that these nutrients are removed from the soil quickly after restoration (Ardón et al. 2010). While plant available P (measured by Mehlich-I extraction) in restored wetlands had dropped to levels similar to those found in natural wetlands, soluble inorganic phosphate (PO_4) saw significant increases over both prior converted cropland and natural wetland sites. This is not unexpected, as hydrological restoration leads to anaerobic conditions with eventual Fe reduction and solubilization of Fe oxides, conditions conducive to soluble inorganic PO_4 release (Moorberg et al. 2015; Szogi et al. 2004). Ardón et al. (2010) predicted that restored wetlands would most likely release agricultural P for between 3 to 16 years post-initiation of restoration efforts, roughly the age of the MIAR sites. However, it should be noted that this trend only occurred within the southern sites. While wetlands are typically considered a sink for P, pocosin soils have been demonstrated to have low P retention capacity as compared to other wetland types; in part, due to the low levels of extractable Al from these soils (Richardson 1985). The restored wetland sites would have an even lower capacity for P retention given their significantly lower levels of soil Al (Table 2). However other nutrients, such as Ca and Mg, used as liming agents to increase soil pH, persist in the restored soils and are reflected in higher soil EC values. Similar to northern sites, Mg levels were essentially unchanged when compared to prior converted cropland sites. While Ca was approximately half of what could be found in prior converted cropland sites, it was still several times higher than the levels found in natural wetland sites.

Contrary to northern sites, one effect of restoration was a significant increase in total soil carbon (total carbon [TC]; Table 4) within the top 10 cm at restored wetlands. At the three lowest landscape positions (RW0, 1, and 2), restored wetland sites had significantly higher TC levels as compared to prior converted cropland sites. Their values were on par with their natural wetland counterparts. These results may indicate a sizeable increase in accumulated TC pools at restored wetland sites over the short period of time post-restoration or it may indicate that excavation has brought soils with higher C levels closer to the surface. Higher TC at restored sites has resulted in significantly greater C:N ratios relative to natural and prior converted cropland sites at RW0. High C:N ratios (> 25) are commonly associated with complete, rather than incomplete, denitrification (Hunt et al. 2007). Thus, the increase in C may be decreasing emission of N_2O relative to total denitrification at restored sites relative to prior converted cropland sites in lower landscape positions (Table 8).

Soil denitrification enzyme activity: Northern sites (Delaware, Maryland, and Virginia)

Denitrification enzyme activity (DEA) rates can be found in Table 7. For all four treatments, DEA rates generally decreased as relative elevation increased, and conversely soil moisture decreased. This trend was observed for all three wetland types. Addition of NO_3 increased DEA amongst all wetland types indicating NO_3 limitation, but this increase was less notable with prior converted croplands. They are likely to be less limited by NO_3 due to active fertilization. Addition of NO_3 generally resulted in higher DEA levels (see complete and potential incomplete DEA rates, Table 7). Although this increase is less relevant for natural depressional wetlands, which typically do not receive fertilizer inputs, this demonstrates that fertilization of prior converted croplands may lead to increased N_2O emissions. At lower relative elevations (RW0/1) prior converted croplands have higher DEA rates than the other wetland classes, but similar N_2O emissions. Relatively high rates of DEA within these wetter portions of prior converted croplands are quite notable due to the relatively large area that prior converted croplands occupy at the landscape scale (Chapters B5 and B9). At higher relative elevation (RW2/3) prior converted croplands have DEA rates similar to restored wetlands; both are higher than natural wetlands. However, restored wetlands generally have higher N_2O levels. This may be partially associated with the combination of relatively low C/N ratios (< 10) and greater soil moisture in these higher landscape positions at restored sites.

Soil denitrification enzyme activity: Southern sites (North Carolina)

Denitrification enzyme activity (DEA) rates, sorted by wetland type and relative elevation, are listed in Table 8. Similar to northern sites, DEA rates generally decreased as relative elevation increased and addition of NO_3 increased DEA amongst all wetland types indicating NO_3 limitation. Contrary to northern sites, natural wetlands generally had higher DEA and potential DEA rates as compared to restored and prior converted cropland sites, particularly at the lowest landscape position. Relatively low DEA at prior converted croplands is most likely impacted by a variety of additional environmental factors, including soil moisture and availability of soil C, but may also be affected by pH (Simek and Cooper, 2002).

Incomplete DEA rates were significantly higher for natural wetlands at both the highest and lowest landscape positions, while natural wetlands had significantly higher potential incomplete DEA at RW0 only. When compared to both prior converted cropland and natural wetland sites, restored sites generally exhibited a trend somewhere in between. However, in the driest areas of restored wetlands, complete DEA rates were the lowest amongst the wetland types, though none of the measured rates across the different relative wetness classes were statistically different (Table 8). These results indicate that although wetland restoration efforts have moved restored soils towards the natural condition, this transformation is currently incomplete. These results differed from DEA rates analyzed at the northern sites, in which restored sites demonstrated a trend similar to natural wetlands at lower elevation and higher rates at greater relative elevations.

Nitrous oxide reductase (nosZ) abundance: Northern Sites (Delaware, Maryland, and Virginia)

Nitrous oxide (N_2O) reductase *(nosZ)* gene abundances for northern sites can be found in Figure 1. Comparison of land management types reveals that prior converted croplands and restored wetlands had significantly ($p < 0.05$) greater abundances of *nosZ* than natural wetlands. Natural wetlands contained roughly half the number of *nosZ* gene copies per gram of soil (~2,750,000), as prior converted croplands (~6,000,000) and restored wetlands (~5,000,000). Abundances in prior converted croplands and restored wetlands were also statistically different from each other ($p < 0.05$). Levels of *nosZ* in these soils reflected levels previously reported in wetland and cropland soils (Ji et al. 2012; Miller et al. 2009). In a study by Miller et al. (2009), they hypothesized that high N fertilization rates in cropping system soils may have resulted in increased denitrifier populations.

Examination of *nosZ* gene abundances along the elevation gradient generally demonstrated increased *nosZ* gene abundances as relative elevation decreased (Figure 1B). These lower elevation soils are more likely to become saturated, resulting in an environment favorable to denitrification (Fellows et al. 2011; Hunter and Faulkner 2001). Closer inspection, however, revealed that while this trend held true for both prior converted croplands and natural wetlands, the inverse was true in restored wetlands. This is likely indicative of a microbial community currently in flux, likely compounded by the removal of topsoil during restoration (i.e., excavation). When looking at denitrification gene abundances in a series of agricultural and successional sites (i.e., transitioning to the natural condition), Morales et al. (2010) noted that impacts of agricultural management on soil microbial populations could last for decades after the practices have ceased.

When using regression analysis to evaluate the relationship between *nosZ* gene abundances and environmental variables according to wetland type, pH was a strong predictor ($R^2 = 0.51$, $P < 0.0001$). This strong predictive relationship between soil pH and *nosZ* abundances was also seen in southern sites. It is well documented that microbial community composition is strongly affected by soil pH, with diversity and activity being the greatest in neutral pH soils. This relationship is supported by the fact that the highest *nosZ* gene abundances are found at prior converted cropland sites. It should be noted however that the presence of a particular gene does not necessarily equate to activity, as denitrification is a complex biological process controlled by a number of environmental factors. Therefore, while it has been previously demonstrated that *nosZ* codes for the enzyme responsible for reducing N_2O to N_2, abundances of this gene are not directly correlated to DEA. This is not to be unexpected since microorganisms capable of performing denitrification only account, on average, for approximately 5 % of the total microbial soil community. However despite this lack of correlation, a relationship between *nosZ* gene abundance and $N_2O/(N_2+N_2O)$ has been previously reported, whereby higher abundances of the *nosZ* gene were correlated with lower N_2O production (Ducey et al. 2011; Philippot et al. 2009). Examining the relationship between mean *nosZ* gene abundances and mean $N_2O/(N_2+N_2O)$ (incomplete DEA/complete DEA) percentages revealed a similar relationship. A strong negative relationship ($y = -30.504x + 239.37$, $P = 0.01$, $R^2 = 0.44$) was demonstrated, indicating that as *nosZ* gene abundances increased, the amount of incomplete denitrification decreased. Of note is that these results follow along management type, with prior converted croplands having lower percentages of incomplete denitrification and natural wetlands having the highest percentages of incomplete denitrification.

Nitrous oxide reductase (nosZ) abundance: Southern Sites (North Carolina)

Abundances of the gene encoding for the enzyme N_2O reductase (*nosZ*), as determined by qPCR, are shown by land use (Figure 2A) and by relative wetness (Figure 2B). Based on land use, comparison of log transformed mean gene copy numbers (± S.E) per gram of soil show that prior converted cropland sites had the highest abundance of *nosZ* (6.64 ± 0.12), followed by natural (6.49 ± 0.12), and restored wetlands (6.09 ± 0.10); restored wetland sites had significantly ($p < 0.001$) lower *nosZ* abundances compared to prior converted cropland and natural wetland sites (Figure 2A). Measurement of *nosZ* copy numbers along the relative wetness gradient (Figure 2B) revealed several differences. Gene abundance patterns between the three wetland types varied, with prior converted cropland and restored wetland sites having lower mean gene abundance values at RW0, while mean gene abundance values were greatest in RW0 for natural wetlands.

Correlations between gene abundances and soil physicochemical properties revealed significant positive relationships of *nosZ* with pH ($r = 0.57$; $P < 0.05$) and NO_3 ($r = 0.57$; $p < 0.05$), while significant negative relationships were identified between *nosZ* and TC ($r = -0.73$; $p < 0.005$), C:N ratio ($r = -0.88$; $P < 0.001$), Na ($r = -0.71$; $P < 0.009$) and PO_4 ($r= -0.75$; $P < 0.005$). A negative relationship between *nosZ* and soil carbon has been previously documented in dairy-grazed pasture soils, and was associated with a concomitant positive relationship to NO_3 (Jha et al. 2012). These findings, similar to those reported in this study, potentially indicate that NO_3 availability is a stronger influence over denitrifier abundance than C availability

Unlike the northern sites, no relationship between *nosZ* gene abundances and DEA rates was confirmed for sites within North Carolina. A previous report demonstrated significant differences between the physicochemical properties of soil in North Carolina sites and the soils of sites in the Delmarva region (Kluber et al. 2014). The differences between soil properties found in these two distinct areas could potentially explain different responses in DEA and *nosZ* patterns with restoration.

At all sites throughout the MIAR, DEA and *nosZ* gene abundances were observed as responding to restoration efforts, but not approaching natural wetland site levels. This potentially indicates that restoration efforts have not fully restored microbial communities capable of functioning in these sites. These results are similar to Peralta et al. (2010), who demonstrated that wetland restoration practices did not successfully restore denitrifier communities. Additionally, a report by Bruland et al. (2006) reported two of three restored wetlands studied displayed lower DEA rates than adjacent natural wetland sites. These results led them to determine that the restored wetlands did not possess microbial communities capable of the increased denitrification rates demonstrated in natural wetlands. Another possibility, however, is that microbial populations have reached a different equilibrium given the new environmental conditions.

Conclusions

A number of physicochemical factors suggest that restoration efforts are resulting in conditions analogous to natural wetlands. Yet the continued agricultural legacy of these restored sites suggests that wetland reclamation is an on-going process that is contingent on more than hydrological restoration. This is illustrated in Figure 3, which shows: 1) sites within North Carolina and other states are unique, and 2) restored wetlands have physicochemical properties that are intermediate to prior converted croplands and natural wetlands. This study, and others (e.g., Morse et al. 2012), also demonstrate that current restoration efforts in the MIAR-CEAP region have not led to serious, unintended consequences, such as an increase in greenhouse gas emissions. Likewise, although not having returned to a natural state, many of the restored wetland functions are more similar to natural wetlands than their prior converted cropland counterparts. While this may not be considered an ideal outcome, this is an improvement relative to the provision of pollutant (N) regulating services. Therefore, this result is consistent with original conservation program goals. While restored wetlands continue to exhibit an agricultural legacy after almost a decade post-restoration, a number of physicochemical predictors (i.e., C sequestration, nutrient reduction, and plant community richness) indicate progress towards a state capable of increased provision of wetland ecosystem services (Gleason et al. 2008).

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Tables

Management	Relative Wetness	TC	TN	CN ratio	pH	EC	ORP	Moisture	Soil Temp
		-------%-------				µS/cm	mV	%	°C
NAT									
	0	7.5a†	0.48a	13.8d	4.43f	41.1e	621b	34.4a	21.8d
	1	7.1b	0.42b	15.2c	4.23g	37.7e	606b	32.5b	22.0d
	2	6.1c	0.35c	16.7b	4.17g	46.6e	624b	28.6c	21.4d
	3	5.6d	0.28d	19.7a	4.25g	44.0e	654a	22.1e	22.1d
RST									
	0	2.0e	0.18ef	9.3e	5.61e	69.2cd	500f	24.6d	25.0abc
	1	2.1e	0.19ef	9.1e	5.78d	53.0de	468g	19.5f	24.6bc
	2	2.1e	0.19ef	9.3e	5.72d	50.5e	538e	17.2gh	24.3c
	3	1.8ef	0.18ef	9.3e	6.12a	54.3de	549de	13.5i	24.7bc
PCC									
	0	2.0e	0.20e	9.3e	6.01b	141.9a	540de	18.6fg	25.3ab
	1	1.9ef	0.19ef	9.0e	5.92c	144.1a	543de	16.2h	25.0abc
	2	1.6fg	0.18ef	8.0f	6.02b	108.9b	574c	14.0i	24.9abc
	3	1.5g	0.17f	7.5g	5.90c	83.6c	558cd	11.2j	25.5a

† Based on Duncan's multiple range values ($P < 0.05$).

Table 1: Wetland soil physicochemical characteristics for northern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevations (adapted from Hunt et al. 2014).

Management	Relative Wetness	Al	Ca	Cu	Fe	K	Na	Mg	P	Zn
		-------------------------------------mg/kg-------------------------------------								
NAT										
	0	672a[†]	111f	0.31e	79ef	75f	24.7a	58f	25.1f	2.22bc
	1	692a	55g	0.29ef	75f	62g	20.0b	41g	21.6gh	1.74d
	2	609b	47g	0.20g	85e	54h	18.3c	35gh	19.0h	1.35fg
	3	443c	53g	0.24fg	137a	51h	14.0g	31h	13.0i	1.70d
RST										
	0	277e	500e	0.45d	113b	83e	16.9de	111d	25.3f	1.53e
	1	232fg	522e	0.44d	101c	72f	15.5f	110d	23.4fg	1.31g
	2	341d	507e	0.44d	83ef	70f	16.2ef	102e	25.7f	1.10h
	3	247f	613d	0.43d	51g	82e	12.4h	117c	31.7e	1.49ef
PCC										
	0	300e	1086a	1.05a	93d	149a	20.2b	178a	81.7a	2.56a
	1	282e	890b	0.98b	77f	132b	17.4cd	143b	69.9b	2.31b
	2	246f	778c	0.93b	49g	119c	15.3f	123c	66.4c	2.12c
	3	207g	624d	0.86c	29h	108d	12.4h	106de	51.9d	2.08c

[†] Based on Duncan's multiple range values ($P < 0.05$).

Table 2: Plant available nutrients (Mehlich I) for northern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes (adapted from Hunt et al. 2014).

Management	Relative Wetness	Cl	NO_3-N	SO_4-S	PO_4-P
		------------------------------mg/kg-------------------------------			
NAT					
	0	11.55cd†	4.91ef	49.93a	2.37d
	1	10.47de	5.32e	40.62b	2.65d
	2	11.05cd	5.29e	30.02c	2.41d
	3	11.73cd	4.51ef	17.21de	2.59d
RST					
	0	8.00e	6.13e	14.68ef	0.89f
	1	9.46de	2.81f	13.19fg	1.25e
	2	9.89de	6.92e	12.84fg	1.32e
	3	10.59d	11.13d	11.07g	2.35d
PCC					
	0	33.06a	21.36b	19.17d	4.73a
	1	21.97b	25.10a	14.76ef	4.12b
	2	21.76b	22.15b	11.11g	4.64a
	3	13.62c	17.81c	7.51h	3.74c

† Based on Duncan's multiple range values ($P < 0.05$).

Table 3: Water soluble anions for northern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes (adapted from Hunt et al. 2014).

Land Use	Relative Wetness	TC	TN	C/N ratio	pH	EC	Moisture	Soil Temp
		------%------				µS/cm	%	°C
NAT	0 (wettest)	29.1 (4.9†)[bc‡]	1.1 (0.1)[ab]	26.6 (3.2)[bcd]	4.6 (0.2)[de]	82.7 (7.1)[cd]	68.9 (1.6)[a]	19.3 (0.9)[e]
	1	32.0 (5.8)[bc]	1.0 (0.1)[ab]	30.7 (4.1)[bcd]	4.5 (0.2)[de]	84.3 (6.1)[cd]	63.8 (4.1)[ab]	19.3 (0.8)[e]
	2	47.0 (4.3)[a]	1.3 (0.1)[a]	37.0 (3.0)[abc]	3.9 (0.2)[e]	106.5 (16.0)[bcd]	57.5 (4.2)[b]	20.0 (0.9)[de]
	3 (driest)	15.5 (5.5)[de]	0.6 (0.2)[cd]	24.6 (2.7)[cd]	4.3 (0.2)[e]	51.0 (16.5)[d]	25.7 (4.9)[de]	20.4 (0.9)[de]
RST	0	40.7 (5.7)[ab]	1.0 (0.2)[ab]	45.7 (9.7)[a]	4.5 (0.2)[de]	140.4 (20.6)[abc]	57.0 (5.0)[b]	22.9 (1.1)[cde]
	1	41.5 (6.3)[ab]	1.0 (0.1)[ab]	40.6 (5.9)[ab]	4.4 (0.2)[de]	159.1 (17.0)[abc]	56.4 (3.5)[b]	22.7 (1.3)[cde]
	2	40.3 (6.6)[ab]	1.0 (0.1)[ab]	40.4 (5.7)[ab]	4.5 (0.3)[de]	176.3 (11.7)[ab]	56.8 (3.1)[b]	23.5 (1.0)[bcd]
	3	20.4 (5.5)[cd]	0.6 (0.1)[cd]	31.0 (3.6)[bcd]	5.2 (0.4)[cd]	87.5 (14.5)[bcd]	31.8 (4.7)[cd]	23.4 (1.5)[bcd]
PCC	0	15.1 (1.5)[de]	0.6 (0.1)[cd]	26.8 (2.1)[bcd]	6.0 (0.3)[ab]	210.6 (45.5)[a]	37.0 (3.3)[cd]	26.5 (1.2)[abc]
	1	19.2 (1.7)[cde]	0.7 (0.1)[bc]	28.8 (2.2)[bcd]	5.6 (0.2)[bc]	165.7 (34.5)[abc]	39.3 (3.4)[c]	26.1 (1.5)[abc]
	2	19.7 (2.5)[cde]	0.8 (0.1)[bc]	26.8 (2.2)[bcd]	6.1 (0.3)[ab]	215.7 (59.8)[a]	31.6 (3.3)[cd]	26.7 (1.4)[ab]
	3	5.2 (1.3)[e]	0.3 (0.1)[d]	17.7 (1.8)[e]	6.7 (0.1)[a]	101.5 (25.8)[bcd]	18.1 (2.3)[e]	28.1 (1.4)[a]

†Means and standard errors.

‡Columns statistically grouped according to Duncan's multiple range test based on a $P < 0.05$ level.

Table 4: Wetland soil physicochemical properties for southern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes (adapted from Ducey et al. 2015).

Land Use	Relative Wetness	Al	Ca	Cu	Fe	K	Na	Mg	P	Zn
		-------- mg/kg --------								
NAT	0	1304 (236[†])^{a}[‡]	507 (115)^{f}	0.15 (0.03)^{ab}	56 (11)^{ab}	66 (15)^{d}	45 (9)^{abc}	64 (14)^{c}	26 (5)^{c}	1.6 (0.3)^{b}
	1	969 (235)^{ab}	558 (193)^{f}	0.13 (0.04)^{ab}	54 (12)^{b}	63 (13)^{d}	44 (9)^{abc}	77 (25)^{c}	15 (4)^{c}	1.8 (0.6)^{b}
	2	755 (177)^{bc}	496 (119)^{f}	0.09 (0.02)^{ab}	57 (6)^{ab}	103 (21)^{bcd}	54 (9)^{a}	91 (45)^{c}	24 (4)^{c}	2.0 (0.3)^{b}
	3	514 (75)^{bcd}	394 (162)^{f}	0.21 (0.05)^{a}	80 (10)^{a}	58 (18)^{d}	30 (9)^{abc}	76 (24)^{c}	22 (4)^{c}	2.5 (0.9)^{ab}
RST	0	469 (165)^{cd}	2267 (592)^{de}	0.11 (0.03)^{ab}	19 (5)^{cd}	107 (27)^{bcd}	53 (10)^{a}	499 (132)^{b}	24 (7)^{c}	5.1 (1.5)^{ab}
	1	449 (148)^{cd}	2404 (548)^{de}	0.12 (0.04)^{ab}	20 (4)^{cd}	100 (19)^{bcd}	57 (8)^{a}	523 (113)^{b}	23 (6)^{c}	4.8 (1.4)^{ab}
	2	429 (159)^{cd}	2285 (533)^{de}	0.13 (0.05)^{ab}	18 (4)^{cd}	123 (33)^{bc}	49 (7)^{ab}	506 (112)^{b}	19 (5)^{c}	5.0 (1.6)^{ab}
	3	582 (104)^{bcd}	1840 (307)^{e}	0.14 (0.06)^{ab}	40 (16)^{bc}	57 (11)^{d}	41 (13)^{abc}	427 (79)^{b}	17 (4)^{c}	2.6 (0.4)^{ab}
PCC	0	439 (108)^{cd}	4618 (503)^{ab}	0.11 (0.04)^{ab}	14 (4)^{d}	154 (18)^{abc}	26 (3)^{bc}	486 (61)^{b}	78 (27)^{b}	5.2 (1.7)^{ab}
	1	593 (117)^{bcd}	3834 (318)^{bc}	0.09 (0.02)^{ab}	13 (4)^{d}	136 (25)^{abc}	32 (4)^{abc}	534 (87)^{b}	43 (14)^{bc}	6.0 (1.6)^{a}
	2	161 (36)^{d}	5273 (289)^{a}	0.05 (0.01)^{b}	5 (1)^{d}	222 (52)^{a}	35 (6)^{abc}	757 (58)^{a}	52 (15)^{bc}	4.6 (1.2)^{ab}
	3	308 (50)^{cd}	3269 (536)^{cd}	0.20 (0.05)^{a}	24 (5)^{cd}	167 (57)^{ab}	21 (3)^{c}	427 (74)^{b}	135 (40)^{a}	4.0 (1.2)^{ab}

[†]Means and standard errors.
[‡]Columns statistically grouped according to Duncan's multiple range test based on a $P < 0.05$ level.

Table 5: Plant available nutrients (Mehlich I) for southern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes (adapted from Ducey et al. 2015).

Land Use	Relative Wetness	Cl	SO_4	PO_4	NO_2+NO_3
		------- mg/kg -------			
NAT	0	28.6 (5.3†)^{b‡}	29.6 (6.8)^{abc}	1.6 (0.6)^{c}	4.4 (2.1)^{c}
	1	26.5 (2.5)^{b}	22.8 (4.0)^{abc}	1.5 (0.5)^{c}	2.6 (0.6)^{c}
	2	42.7 (5.5)^{ab}	42.6 (5.5)^{a}	7.9 (0.8)^{bc}	4.4 (1.3)^{c}
	3	20.6 (17.2)^{b}	22.2 (7.3)^{abc}	3.3 (1.2)^{c}	3.8 (1.3)^{c}
RST	0	30.4 (3.5)^{b}	30.4 (9.0)^{abc}	18.0 (7.8)^{a}	4.8 (2.0)^{c}
	1	31.4 (4.7)^{b}	34.1 (8.9)^{ab}	16.3 (6.4)^{ab}	4.9 (2.6)^{c}
	2	32.6 (6.3)^{b}	29.9 (6.6)^{abc}	9.1 (2.4)^{abc}	11.2 (5.0)^{c}
	3	26.7 (6.9)^{b}	15.7 (3.3)^{bc}	2.6 (0.9)^{c}	9.8 (4.7)^{c}
PCC	0	29.8 (10.0)^{b}	15.7 (3.0)^{bc}	2.5 (0.6)^{c}	87.1 (52.7)^{ab}
	1	35.3 (5.2)^{b}	20.5 (5.1)^{bc}	2.4 (0.6)^{c}	60.2 (34.0)^{b}
	2	63.0 (19.9)^{a}	25.5 (6.9)^{abc}	5.1 (1.7)^{c}	168.5 (89.6)^{a}
	3	16.8 (5.7)^{b}	10.9 (4.6)^{c}	5.7 (2.3)^{c}	50.9 (30.7)^{b}

†Means and standard errors.

‡Columns statistically grouped according to Duncan's multiple range test based on a $P < 0.05$ level.

Table 6: Water soluble anions for southern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes (adapted from Ducey et al. 2015).

Land Use	Relative Wetness	Incomplete	Complete	Potential Incomplete	Potential Complete
		no additions	+ acetylene	+ NO_3	+ NO_3 + acetylene
		μg N_2O-N kg^{-1} soil hr^{-1}			
PCC	0	35.0 (10.9†)^{a}	118.0 (22.3)^{a}	86.6 (19.5)^{ab}	159.8 (26.6)^{ab}
	1	10.0 (2.5)^{ab}	49.1 (7.9)^{bc}	28.6 (5.1)^{abc}	85.3 (13.8)^{abc}
	2	6.1 (1.7)^{b}	30.3 (5.7)^{bc}	22.0 (10.5)^{bc}	46.7 (13.4)^{bc}
	3	4.8 (0.9)^{b}	29.1 (8.9)^{bc}	15.0 (4.0)^{bc}	42.8 (13.2)^{bc}
RST	0	26.6 (8.7)^{ab}	83.2 (19.9)^{ab}	96.5 (30.1)^{a}	175.3 (54.2)^{a}
	1	20.7 (8.4)^{ab}	60.8 (16.5)^{bc}	81.0 (20.0)^{abc}	140.8 (28.8)^{ab}
	2	18.9 (5.2)^{ab}	43.7 (7.3)^{bc}	38.6 (8.7)^{abc}	75.7 (14.1)^{abc}
	3	13.7 (4.0)^{ab}	38.1 (7.3)^{bc}	23.2 (4.7)^{bc}	46.5 (7.1)^{bc}
NAT	0	14.0 (3.7)^{ab}	36.9 (10.2)^{bc}	83.0 (28.4)^{abc}	137.5 (40.3)^{abc}
	1	10.9 (2.2)^{ab}	27.1 (6.1)^{bc}	53.7 (14.2)^{abc}	95.8 (24.0)^{abc}
	2	10.6 (2.6)^{ab}	22.3 (5.1)^{c}	37.0 (14.3)^{abc}	56.3 (16.3)^{abc}
	3	4.0 (1.6)^{b}	9.3 (3.5)^{c}	7.6 (2.9)^{c}	11.0 (3.6)^{c}

†Mean and standard errors.
‡Columns statistically grouped based on pairwise Log^{10} transformed, LS-mean differences based on a $P < 0.05$

Table 7: Denitrification enzyme activity rates for northern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes

Land Use	Relative Wetness	Incomplete	Complete	Potential Incomplete	Potential Complete
		no additions	+ acetylene	+ NO_3	+ NO_3 + acetylene
		µg N_2O-N kg^{-1} soil hr^{-1}			
PCC	0	18.5 (7.8)^{bc}	49.7 (13.1)^{ab}	23.6 (9.3)^{bcd}	41.2 (10.5)^{b}
	1	35.9 (12.9)^{ab}	88.2 (28.8)^{a}	42.5 (15.4)^{bcd}	102.3 (27.5)^{ab}
	2	24.3 (9.3)^{ab}	58.1 (14.6)^{ab}	46.5 (15.9)^{abc}	75.3 (24.6)^{b}
	3	4.5 (1.8)^{c}	33.0 (15.5)^{c}	22.0 (12.8)^{de}	55.2 (21.8)^{bc}
RST	0	23.3 (12.9)^{bc}	89.7 (53.1)^{bc}	67.3 (30.6)^{abcd}	143.9 (58.3)^{ab}
	1	40.6 (18.5)^{ab}	77.0 (32.3)^{abc}	89.9 (39.4)^{ab}	132.2 (44.3)^{ab}
	2	19.2 (12.7)^{bc}	35.3 (22.4)^{cd}	34.1 (18.3)^{cde}	68.7 (32.4)^{bc}
	3	4.5 (2.0)^{c}	9.9 (3.3)^{c}	7.8 (3.4)^{e}	13.8 (5.4)^{c}
NAT	0	70.1 (32.4†)^{a‡}	145.3 (64.4)^{a}	231.1 (157.6)^{a}	330.5 (192.7)^{a}
	1	29.1 (11.7)^{a}	97.4 (49.6)^{ab}	139.8 (50.9)^{a}	210.2 (67.6)^{a}
	2	11.8 (4.1)^{ab}	21.3 (9.8)^{cd}	18.8 (5.9)^{bcd}	28.1 (9.4)^{bc}
	3	28.4 (15.4)^{b}	59.7 (34.6)^{cd}	48.2 (29.3)^{bcde}	81.6 (46.2)^{bc}

†Mean and standard errors.
‡Columns statistically grouped based on pairwise Log^{10} transformed, LS-mean differences based on a $P < 0.05$ level

Table 8: Denitrification enzyme activity rates at southern natural [NAT], restored [RST], and prior converted cropland [PCC] sites at different relative elevation classes (adapted from Ducey et al. 2015).

Figures

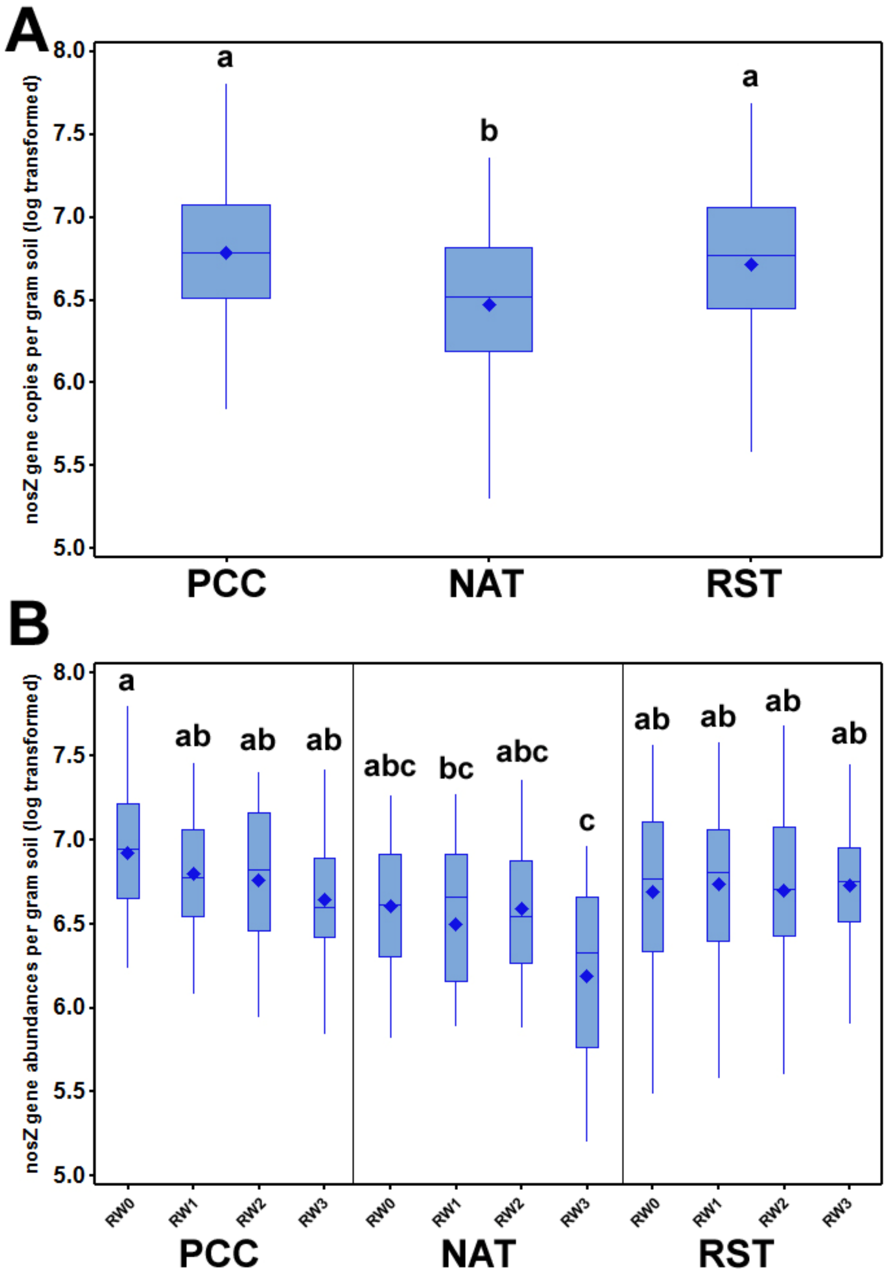

Figure 1: Box and whisker plots of *nosZ* gene abundances per gram of soil by land use (A; prior converted cropland [PCC], natural [NAT], and restored [RST]) and by relative wetness (B) of northern sites. All values have been log transformed. Statistical significance based on Duncan's multiple range test ($P < 0.05$). Those with the same letter are not significantly different. The ♦ represents the mean *nosZ* gene abundance for each treatment.

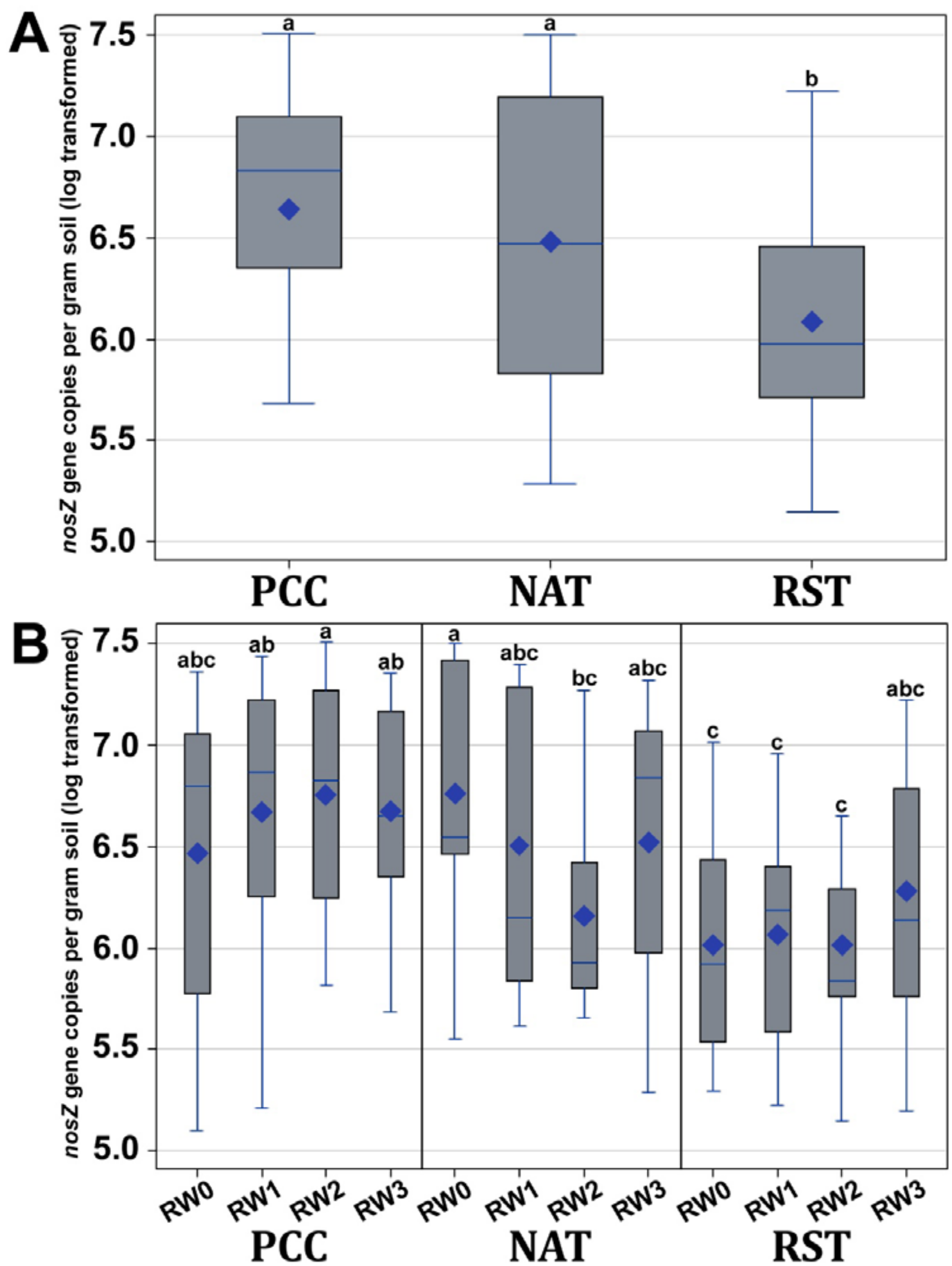

Figure 2: Box and whisker plots of *nosZ* gene abundances per gram of soil by land use (A; prior converted cropland [PCC], natural [NAT], and restored [RST]) and by relative wetness (B) of southern sites. All values have been log transformed. Statistical significance based on Duncan's multiple range test ($P < 0.05$). Those with the same letter are not significantly different. The ♦ represents the mean *nosZ* gene abundance for each treatment (adapted from Ducey et al. 2015).

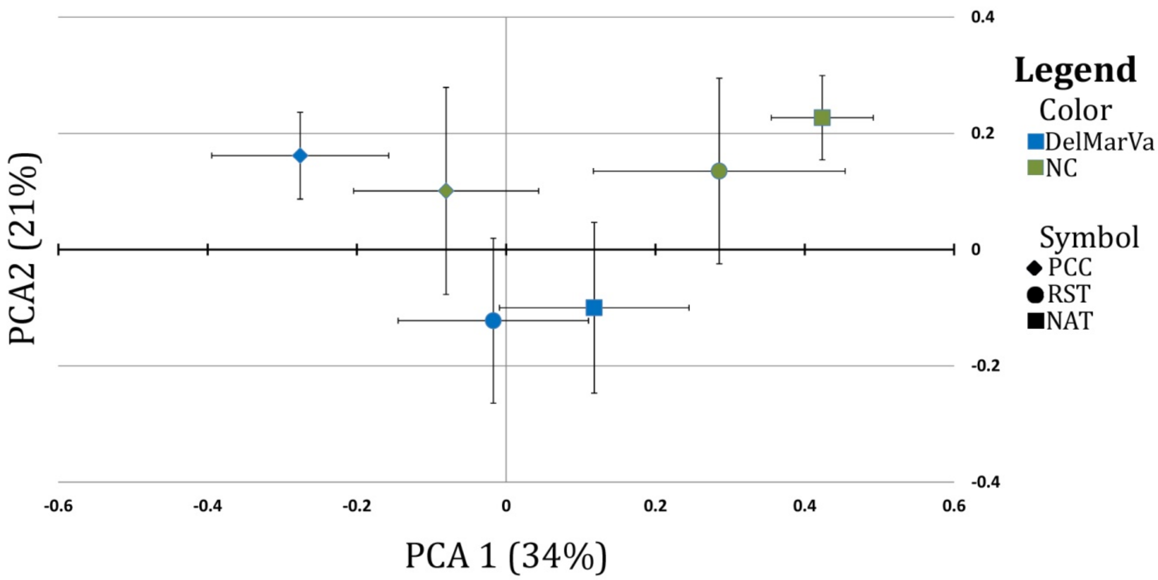

Figure 3: Principle components analysis (PCA) of all physicochemical variables for wetland types (prior converted cropland [PCC], natural [NAT], and restored [RST]) within North Carolina (NC) and Delaware, Maryland, and Virginia (DelMarVa).

4. Phosphorus Levels and Mobility

Key Findings
-Wetlands were converted to agricultural fields through not only drainage, but also liming to raise pH and thus prevent aluminum toxicity in crops.

-While total (environmentally relevant) P was highest in natural and prior converted cropland sites, P that is more likely to be transferred via hydrologic flows (i.e., P that is associated with amorphous forms of Fe and Al) was almost twice as high in the prior converted cropland sites, as compared to natural and restored sites.

-Results suggest that of the sites studied, natural wetlands have the greatest potential for P mitigation, while prior converted cropland sites have the lowest, and may even serve as a source of P to adjacent water bodies due to high P saturation.

-Wetland restoration practices appear to have enhanced P sorption capacity, but there is still potential for P saturation in restored soils to be substantially decreased through natural weathering processes, which can create new A horizon soils and lower pH.

Recommended Practices:
Although wetland hydrology has been restored, wetlands have not fully regained wetland chemistry, which was altered by liming. Since wetland conversion originally incorporated both drainage and chemical additions (i.e., liming), restoration of hydrology only addresses a portion of the changes necessary to fully restore wetlands on former croplands. Current restoration practices should allow soils to eventually revert to prior P mitigation capacity. However, this process could take a substantial amount of time. It is possible that wetland restoration practitioners could hasten development of more natural soil conditions through the active lowering of pH via the direct application of an acid to counter the effects of liming. However, additional research is needed to develop this implementation technique. It is notable that excavation, while reducing the provision of other ecosystem services, enhances P mitigation services through the removal of P rich topsoil.

Primary Chapter Source: Church, C., Kleinman, P., Miller, J., and Lang, M. Controls on soil phosphorus in native, disturbed and hydrologically restored agricultural wetlands. *Journal of Soil and Water Conservation*. (In Review)

Introduction

Phosphorus fate in wetlands and the role of wetlands as regulators of P pollution have received considerable attention. However, the interactions which control the transformation and movement of P within wetland environments deserve further study due to their complex nature (Craft and Chiang 2002). Phosphorus can enter wetlands in sediment-bound or dissolved form. Whether P is retained within the wetland, however, depends upon its form, the nature of soils, the amount and type of microbial and vegetative biomass, and hydrology, including retention time, of the ecosystem. Under some conditions, wetlands are a source of P, as in wetlands constructed on prior agricultural lands with significant soil P concentrations (Heiberg et al., 2009). Wetlands may be a source of P only during certain times of year (Connor and Martin 1989) or during large storms (Novak et al. 2007). Significant quantities of sediment associated P are deposited in wetlands (Walbridge and Struthers 1993; Axt and Walbridge 1999; Craft and Casey 2000). Dissolved P removal from water in wetlands occurs through P uptake by plants and soil microbes; adsorption by aluminum and iron oxides and hydroxides; precipitation of aluminum, iron, and calcium phosphates; and burial of P adsorbed to sediments or organic matter (Richardson 1985; Johnston 1991; Walbridge and Struthers 1993; Axt and Walbridge 1999; Craft and Casey 2000). Wetland soils can, however, reach a state of P saturation, after which P may be released from the system (Richardson 1985). The potential for long-term storage of P through adsorption to wetland soil is greater than the maximum rates of P accumulation possible in plant biomass (Walbridge and Struthers 1993; Johnston 1991). The capacity for P adsorption by a wetland, however, can be saturated in a few years if it has low amounts of aluminum and iron or calcium (Richardson 1985). Wetlands with large sediment inputs tend to have a high capacity for P adsorption, with their adsorption capacity replenished by the periodic deposition of clays rich in reactive cations (Gambrell 1994). The objective of the study was to evaluate wetland soils with respect to P mobility and sorption capacity across a wetland human alteration gradient, and to elucidate the processes controlling P sorption in different wetland soils and management systems.

Methods

A Geographic Information System (GIS) was used to select sampling locations within the 48 MIAR sites. A lidar-based digital elevation model (DEM) was used to classify each study site into four approximately equal area elevation classes (EC 3 – 0, highest to lowest elevation; Figure 1). Within each class, points were randomly selected for data collection. Near each point, three different soil samples were extracted from the upper six inches of the soil. A composite sample was mixed together and sub-sampled before placing the sample on ice. Soil samples were then returned to the lab for analysis, where they were dried, ground and sieved (2 mm) prior to analyses.

Samples were analyzed to identify soil pH, different pools of P, and to measure associated elements (e.g. Al and Fe). Calcium chloride extractions were performed to identify readily soluble P in soils following the method of Self-Davis (2009). For this extraction, 1 g of soil was added to 25 mL of a 0.01 M $CaCl_2$ solution and extracted, centrifuged and filtered prior to being analyzed by inductively coupled plasma-optical emission spectrometry (ICP-OES). Mehlich-3 extractions were performed as a measure of "agronomically available" P following the method of Mehlich (1984). Soils were extracted by mixing 2.5 g of soil with 25 ml of Mehlich-3 solution for 5 min and then filtered prior to being analyzed by ICP-OES. Acid ammonium oxalate extractions were performed to dissolve noncrystalline inorganic and organiccomplexed forms of Fe and Al from the soils, and to measure ammonium oxalate extractable P. These extractions were conducted using a modified McKeague and Day (1966) method. Briefly, 0.5 g of dried, sieved (2 mm) soil was added to 20 mL of a 0.2 M ammonium oxalate and 0.2 M oxalic acid before being extracted, centrifuged, filtered, and analyzed by ICP-OES. Aqua regia extractions were performed to measure “total” P using a modified Kimbrough and Wakakuwa (1989) method. Digestions were performed, and then extracts were filtered prior to being analyzed by ICP-OES.

Prior to analysis by conventional parametric statistics, data were tested for normality and homogeneity of variance to ensure compliance with Gaussian distribution requirements. Pearson’s correlation and differences among treatments were evaluated using Igor Pro V. 5.0 (Wavemetrics Inc., Lake Oswego, OR), analysis of variance (ANOVA), and Welch’s t-test (Welch 1947). When required due to skewing of data, the Brown and Forsythe test (Brown and Forsythe 1974) was used to calculate the F statistic resulting from an ordinary one-way analysis of variance on the absolute deviations from the median. Statistical results discussed in text were considered significant at $\alpha = 0.05$. Error bars in figures correspond to the standard error of all data from each category analyzed with no removal of outliers.

Results and Discussion

Aqua regia extractable phosphorus (P_{AR}) is often referred to as “total” P, though in reality, aqua regia extracts P that is likely to be environmentally relevant, leaving behind recalcitrant mineral forms of P (Kimbrough and Wakakuwa 1989; US EPA 1994). Surface soils in natural and prior converted cropland sites had about the same concentration of P_{AR}, and there were no significant differences between them (Table 1). Surface soils of restored sites, however, had significantly lower P_{AR} concentrations. Trends in P_{AR} of surface soils between prior converted cropland and restored sites were consistent with the removal of surface soils (i.e., excavation) during the restoration process, which was found to be a common practice at the MIAR sites (Fenstermacher et al. 2015). Long-term fertilization of soil typically causes vertical stratification of P in agricultural soils, with most P being located in the upper six inches of the soil profile (Vaughan et al. 2007). It is likely that much of the P_{AR} is not gone from the overall wetland site, but rather has been relocated to form berms, spoil piles, islands, and other mounded areas. Further evidence of excavation at many of the sampling sites can be seen both in P_{AR} concentrations across the topographic gradient within sites (Figure 2, discussed in more detail below), and in detailed soil descriptions created for study sites, which reveal that most of the A horizon is often removed from restored sites (Fenstermacher et al. 2012).

Trends in $CaCl_2$-P and Mehlich-3 P fractions stand in contrast with those observed with P_{AR}. Readily soluble P (available to water runoff), as represented by $CaCl_2$ extractable P, was relatively low in all soils sampled (Table 1). Given the low concentration of this readily extractable P, considerable variability was not observed across sites, and there were no significant differences between natural, prior converted cropland, and restored sites. This fraction accounted for less than 1 % of the P_{AR} for all site types, but is considered highly bioavailable and mobile. Agronomically available P, as represented by the Mehlich-3 extractable fraction, was significantly higher in the prior converted cropland sites, comprising 12 %, 37 %, and 20 %, respectively, of the P_{AR} for the natural, prior converted cropland, and restored sites (Table 1). These differences likely reflect the historical addition of P in manures and/or mineral fertilizers at prior converted cropland and restored sites. Concentrations found in the prior converted cropland soils were well above crop sufficiency requirements (Beegle 2013), but common in agricultural soils in the Atlantic Coastal Plain (Sims and Vadas 1997). By contrast, agronomically available P in natural and restored sites was much lower, and did not differ significantly, likely reflecting the physical removal of accumulated P in surface soils during restoration.

Phosphorus extractable by acid ammonium oxalate (P_{OX}) provides insight into P that is associated with amorphous forms of Fe and Al (Fe_{OX} and Al_{OX}, respectively), which have been shown to be the primary P sorbing species in many soils (Freese et al. 1992; Lookman et al. 1995). This fraction represented 60 %, 91 %, and 57 % of the P_{AR} at natural, prior converted cropland, and restored sites, respectively. P_{OX} was significantly higher (almost twice as high) in prior converted cropland sites than it was in the other two categories (Table 1). Natural and restored sites, however, had about the same amount of P_{OX}, and there was no significant difference between them. Given these findings alone, one might presume that P had simply been leached from the natural and restored sites due to wetter conditions, but this is not the case, as reflected by P_{AR} concentrations.

Another indication that P has not simply been leached from the sites due to wetter soil conditions can be found in the topographical distribution of P_{OX} within the sites (Figure 2D). In natural and prior converted cropland sites P_{OX} was found to be significantly higher in the lowest landscape positions relative to the highest. The fact that the restored sites did not follow the same pattern is further evidence that excavation during restoration has removed surface soils, particularly in areas of low elevation. Similar patterns in ammonium oxalate extractable Al (Al_{OX}) in both the natural and prior converted cropland sites, and acid ammonium oxalate extractable Fe (Fe_{OX}) in the prior converted cropland sites compared to the restored sites also support this conclusion (Figure 2, E and F).

A direct example of the potential of wetlands to mitigate P loads to receiving water bodies can be found in the fraction of P_{AR} that is reversibly bound (P_{OX}) (Lookman, et al. 1995). While mean P_{AR} was not significantly higher in the prior converted cropland sites than in the natural sites (Table 1), 96 % of this value was found as P_{OX}, a form that could be mobilized under reducing conditions (i.e., iron associated P), changing pH (i.e., Al associated P), or desorption due to low aqueous P concentrations, while P_{OX} was equivalent to only 68 % of P_{AR} in soils of natural sites (Figure 3). This indicates that wetland soils likely play an important role in the immobilization of P under flooded conditions when the potential for leaching is higher, while prior converted cropland sites have a higher potential to be a source of P. In the restored sites, 83 % of the P had the potential to be mobilized, which may either be an indication that they are returning to natural conditions through biogeochemical processes or an artifact of wetland restoration implementation practices.

Trends in soil pH and extractable forms of Al and Fe provide insight into the controls of P sorption in soils of this study (Borggard 1983; van der Zee and van Riemsdijk 1988; Torrent 1987). Mean pH varied significantly between site types (Table 2). Mean pH values of natural wetland soils was below 4, while the pH of prior converted cropland and restored wetland soils ranged from 5 to 6. This is not surprising, as liming is central to the management of soil fertility and is part of the process of converting wetland soils to agriculture, in part to reduce aluminum toxicity by lowering its solubility (U.S. Department of Agriculture 1999). This lowering of Al solubility also results in lowering the concentration of amorphous forms of Al (as reflected in oxalate extractable Al values, Table 2) which, in turn, lowers the ability of the soil to act as a sink for P.

The consequence of the discrepancy between native and disturbed sites is best seen in the P sorption saturation (P-sat) values with respect to amorphous forms of Fe and Al (Figure 4). P sorption saturation is one of the most robust indicators of environmental availability of P (Freese et al. 1992; Lookman et al. 1995), and it is clear that while the prior converted cropland sites were significantly more P saturated (18 %), even the restored sites had P saturation values significantly higher than natural wetlands (10 % and 4 %, respectively).

Conclusions

Many studies have shown that it is the sorption capacity of soils that determines their effectiveness in removing dissolved P (e.g., Freese et al. 1992; Lookman et al. 1995; Heiberg et al. 2009). This study shows that, of the wetland types studied, natural wetlands have the greatest potential for P mitigation, while prior converted cropland sites have the lowest, and may even serve as sources of P to receiving water bodies due to their high P saturation. Restoration practices do appear to enhance P sorption capacity of these soils compared to prior converted cropland sites, but we hypothesize that this is primarily due to excavation, a practice which is likely to negatively impact the provision of non-P related ecosystem services. While P-sat values of restored site soils are half of prior converted cropland sites, there is still potential for P-sat values to be substantially lowered by natural weathering processes, which can create new A horizon soils and lower pH.

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Tables

Constituent	**NAT** n = 51	**PCC** n = 63	**RST** n = 71
Aqua Regia P	304.6 (35.3) a	333.8 (19.8) a	219.4 (13.0) b
CaCl P	2.60 (0.35) a	2.10 (0.24) a	2.2 (0.84) a
Mehlich P	37.9 (5.6) a	105.2 (8.0) b	46.6 (3.9) a
Oxalate P	181.9 (24.2) a	312.5 (21.6) b	179.3 (14.3) a

Notes: Standard error of means are given in parentheses next to the means. Values followed by differences identify groupings per ANOVA, Welch's Test, or the Brown-Forsythe test, as needed.

Table 1: Mean soil P (mg kg^{-1}) extracted by different reagents per site type (prior converted cropland [PCC], natural [NAT], and restored [RST]; adapted from Church et al. [in review]).

Constituent	**NAT** n = 51	**PCC** n = 63	**RST** n = 71
pH	3.97 (0.03) a	5.77 (0.08) b	5.30 (0.06) c
Aqua Regia Al	346 (28) a	315 (17) a	351 (25) a
Oxalate Al	126 (15) b	45.7 (6.4) c	46.8 (5.7) c
Aqua Regia Fe	43.8 (4.3) a	67.8 (4.5) b	69.2 (5.0) b
Oxalate Fe	27.3 (3.1) c	20.1 (1.5) d	21.4 (1.5) d

Notes: Standard error of means are given in parentheses next to the means. Values followed by differences identify groupings per ANOVA, Welch's Test, or the Brown-Forsythe test, as needed.

Table 2: Mean soil pH, Al and Fe (mmols kg^{-1}) extracted by different reagents per site type (prior converted cropland [PCC], natural [NAT], and restored [RST]; adapted from Church et al. [in review]).

Figures

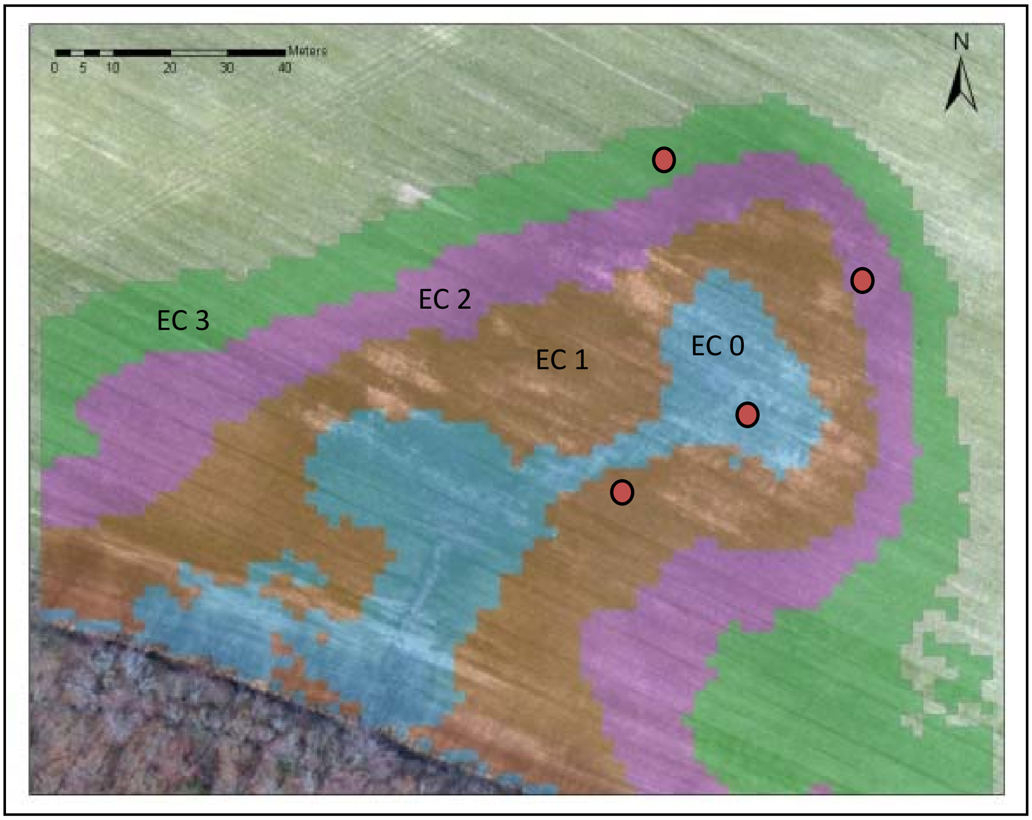

Figure 1: Schematic of sampling design. Elevation classes (EC) range from 3 (highest elevation) to 0 (lowest elevation) and red dots represent randomly select sampling points (Church et al. [in review]).

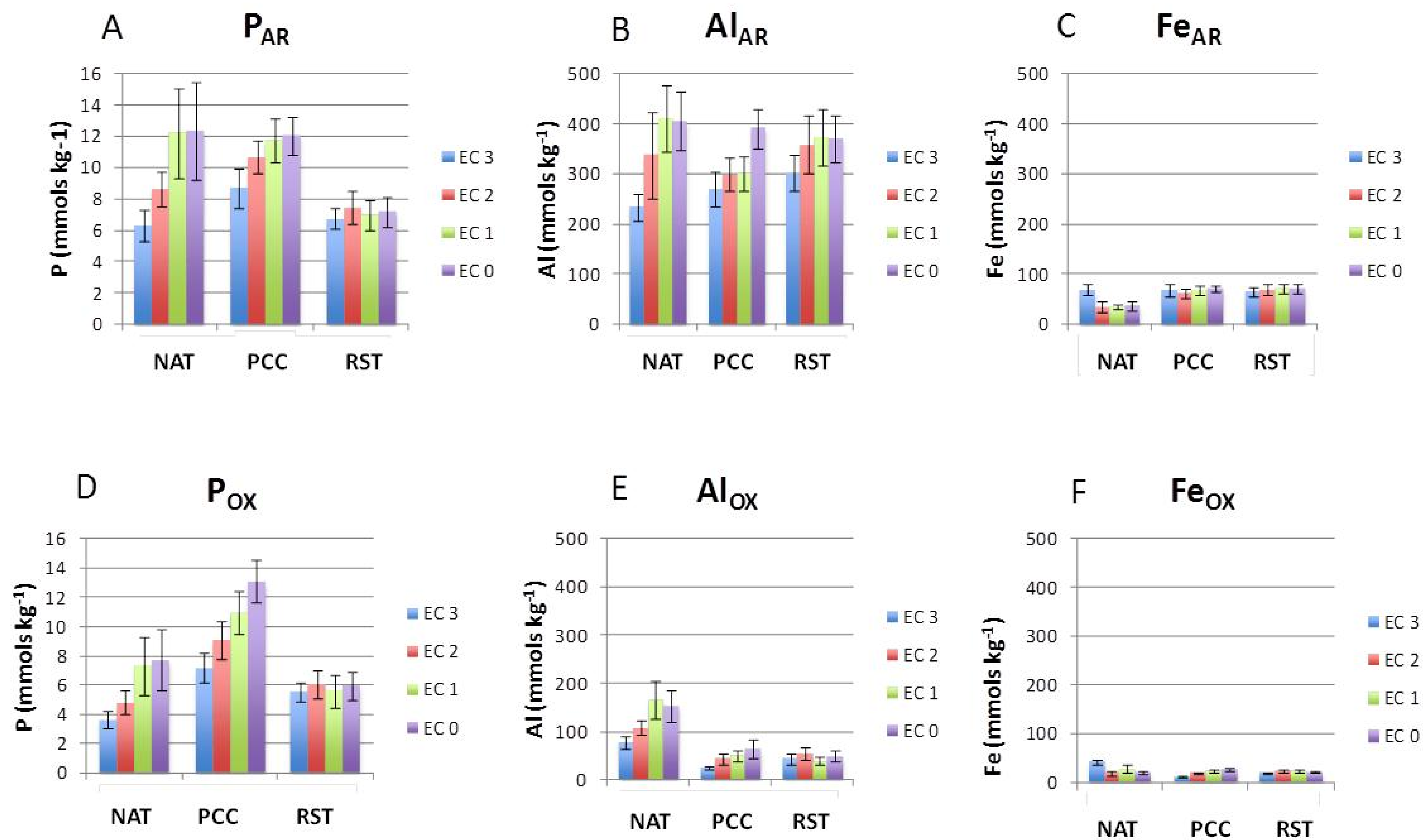

Figure 2: Aqua regia and ammonium oxalate extractable P, Al, and Fe for natural (NAT), prior converted cropland (PCC), and restored (RST) sites. EC 3 – 0 are relative elevation classes, highest (3) to lowest (0; adapted from Church et al. [in review]).

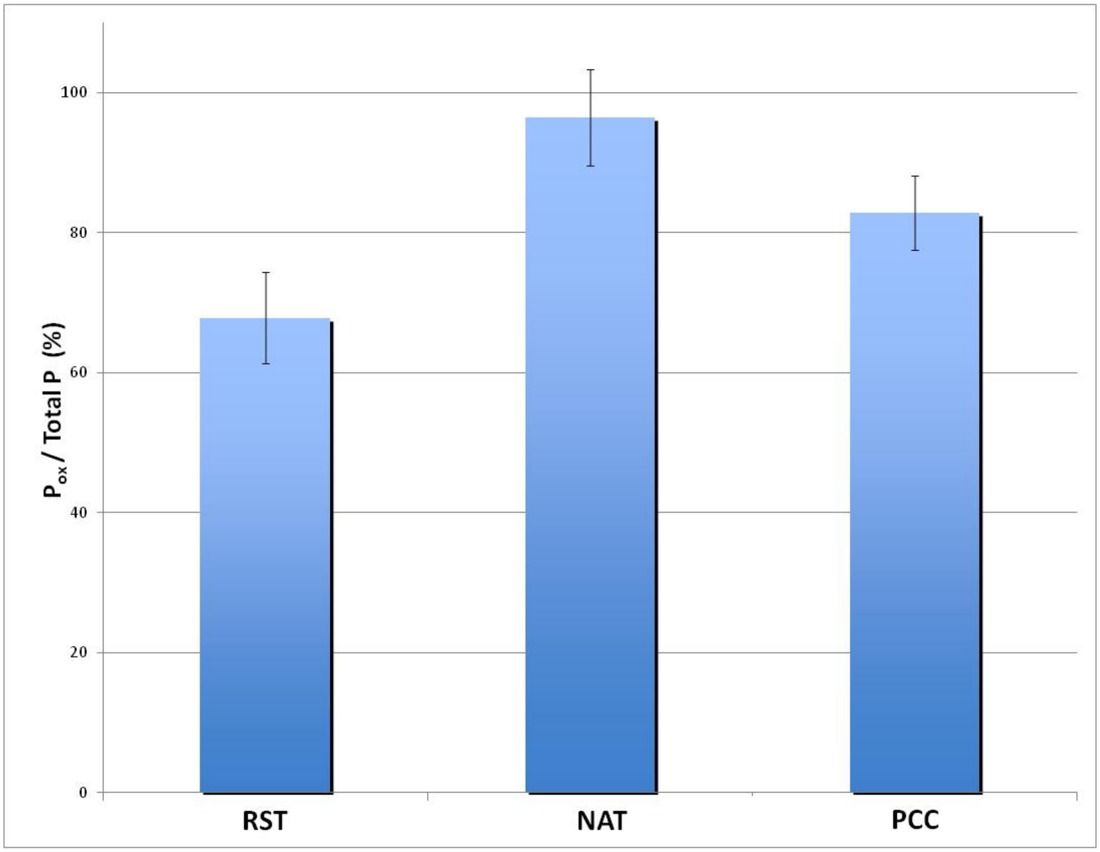

Figure 3: Percent ammonium oxalate extractable phosphorus (adapted from Church et al. [in review]).

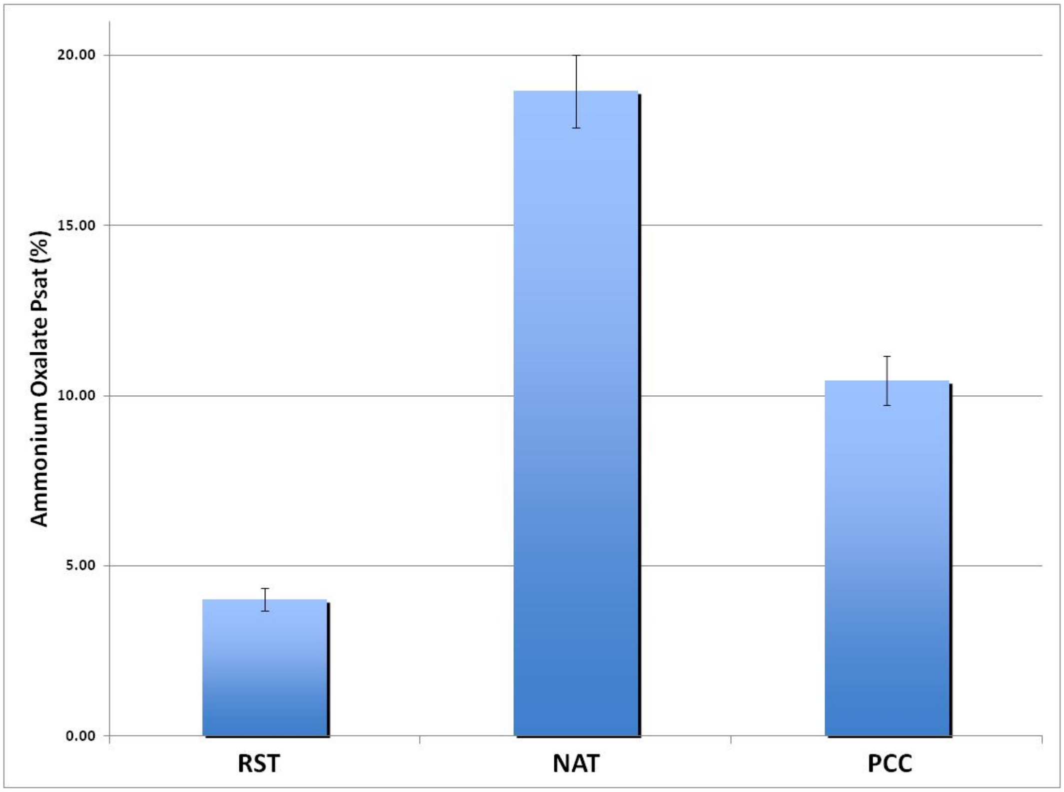

Figure 4: Ammonium oxalate P-sat with respect to Fe and Al **(**adapted from Church et al. [in review]).

5. Change in Depressional Wetland Water Volume Storage on the Delmarva Peninsula: Opportunities for Improved Storm Flow Mitigation

Key Findings

-This study validated the use of airborne LiDAR for accurate measurement of depressional wetland elevation and morphology in a low relief landscape.

-A majority (58 %) of the ~14,500 identified depressions on the Delmarva Peninsula are currently associated with cropland, indicating the magnitude of wetland loss since the introduction of agriculture on the Delmarva Peninsula.

-Another 18 % of total identified depressions were classified as mixed land use (i.e., cropland and forestland), and a large number of those depressions are also likely drained.

-Total estimated volume storage associated with identified depressions was 35,900 ha-m, including 16,900 ha-m on cropland, 12,400 ha-m on forestland, and 6,600 ha-m on mixed forest and cropland.

-MIAR restored wetland study sites had substantially less volume storage than average depressions located on forestland and cropland, indicating that there is potential to enhance performance of wetland restorations for improved volume storage on Delmarva landscapes.

-In general, the agricultural landscape of the Delmarva Peninsula has a very high capacity for increased surface water volume storage. Implementation of wetland restoration and drainage control structures can take advantage of the potential volume storage capacity on croplands.

***Recommended Practices*:** Wetlands should be restored, especially when prior converted croplands are found to be marginal for crop production. Restoration of larger wetland cells should be considered. Controlled drainage structures, on ditches and tile drains, should be used to increase seasonal water storage capacity within prior converted croplands that are currently productive cropland. Remaining natural wetlands are a substantial source of surface water volume storage, and should be preserved to retain regulation of natural hazards (e.g., flooding) and hydrologic flow services within agricultural landscapes.

Introduction

Wetlands provide an important ecosystem service by modulating storm flows and reducing the frequency of stream flood stage, and subsequent flooding of urban, suburban, and rural landscapes. Watersheds where wetlands have been drained have greatly reduced water storage capacity resulting in less modulated (i.e., spiky) stream flows, which are more subject to flooding (Miller and Nudds 1996; Miller and Frink 2000). However, relatively flat landscapes and an abundance of poorly drained soils (i.e., wetlands) requires mechanisms of accelerated removal of surficial waters before conversion to cropland. The vast majority of inland wetland loss within the United States has occurred through drainage. Organized ditch drainage on the Delmarva Peninsula dates back to the 17th century with formation of the first recognized public drainage association in North America (Bell et al. 2000). Depressional wetlands are a prominent feature of the Delmarva landscape, although they were once even more common. Many have been drained to allow for agricultural cultivation, primarily corn and soybean production, in support of a substantive poultry industry (McCarty et al. 2008). Estimates of the percent of wetland area that was lost between European colonization and the 1980s within the states that compose the Delmarva Peninsula range from 42 – 73 % (Dahl 1990).

Depressional features, similar to Carolina bays, are regionally known as Delmarva bays, and they occur primarily near the Maryland and Delaware state border in the northern and central portions of the Delmarva Peninsula (Tiner 2003; Fenstermacher et al. 2014; Chapter B9). A detailed geomorphometric analysis using a LiDAR derived digital elevation model (DEM) estimated that 17,000 depressional features exist on the Delmarva Peninsula, most of these being current or former Delmarva bays (Fenstermacher et al. 2014). This estimate was an order of magnitude higher than previous reports. The extensive drainage of Delmarva bays, primarily via ditches, to support agricultural activities has undoubtedly had marked effects on water storage capacity, which supports the provision of natural hazard and hydrologic flow regulation services. However, the current status of depressional wetland water storage and the potential for increased storage with wetland restoration on the Delmarva Peninsula are unknown, as well as the extent of volume storage loss due to historic conversion of natural wetlands to croplands. This MIAR CEAP-Wetland study component provides an estimate of surface water volume storage once associated with depressional wetlands on the Delmarva Peninsula, assess the proportion of volume storage loss in this landscape due to drainage for agricultural production, and compares this loss with the gain of volume storage associated with implementation of wetland restoration practices on cropland.

Methods

Study Area and Sites

The research area encompassed the entire Delmarva Peninsula with validation of extrapolation methodology occurring in New Castle and Kent counties in Delaware, and Dorchester, Talbot, Caroline, and Queen Anne’s counties in Maryland. For validation, representative subsets were selected from the known population of Delmarva Peninsula depressions, including natural and restored wetlands, as well as prior converted croplands. Natural wetlands were selected from the population of Delmarva bays on forestland. Restored wetlands included cropland areas that had been hydrologically restored to depressional wetlands through USDA conservation programs. Prior converted cropland sites included in the survey were located on active croplands containing roughly circular depressional areas with morphologies approximating those established for Delmarva bays (Fenstermacher et al. 2014).

Outline of Volume Storage Scaling Approach

A multistep calibration and validation process for scaling volume storage estimates across land cover within the Delmarva Peninsula was used for this study: 1) validate the use of LiDAR derived DEMs to estimate volume storage using a highly accurate field based approach at 20 depressional wetlands; 2) calibrate and validate a generalized formula to estimate depression volume based on surface area, relief, and a constant optimized for Delmarva bays using a set of 58 representative depressional wetlands; 3) test the utilization of median depression radius against measured radius for assigning land cover to wetlands within the set of 58 wetlands using high resolution imagery for validation; 4) characterize distributions of measured relief and radius for a random ~1,400 subset of the regional Delmarva bay population and further test use of median radius for land cover assignment using course resolution (30 m) NLCD; 5) use the distributions measured in the ~1,400 subset to randomly assign relief and radius to the full ~14,500 Delmarva bay population and calculate storage volume using the calibrated general formula; assign land cover to the full population using the intersection of a median radius polygon with the land cover map.

Comparing Volume Estimates: Ground based Surveys vs. Aircraft based LiDAR

A set of 20 depressions representing natural and restored wetlands and prior converted croplands were selected for comparison of volume storage estimates based on ground based surveys and airborne LiDAR. The ground based surveys took place during the summer and fall of 2012 (dry season), ensuring the greatest access to all sections of the wetland using construction grade robotic total station survey equipment (Make/Model: Sakkia SRX3X3 and ToHon GPT-8203). Elevation readings were taken in 0.3 m increments in areas of rapid change, such as ditches and berms, and 5 m increments in areas of minimal relief. Total station data points were corrected and processed with Trimble GPS Pathfinder and imported into ArcMap version 10.1 (ESRI, Redlands, CA). LiDAR data were collected from the Maryland Department of Natural Resources, the state of Delaware, or the Agricultural Research Service (ARS). All LiDAR data had a vertical accuracy of ≤ 18 cm RMSE and were designed to meet or exceed Federal Geographic Data Committee National Standards for Spatial Data Accuracy for data at a scale of 1:2,400. Estimated horizontal positional accuracy of point returns exceeds 50 cm. Triangulated Irregular Networks (TINs) were created for individual wetlands using the total station or LiDAR data points (Figure 1).

Volume calculations were performed in ArcMap using the hydrology function within the Spatial Analyst extension. The spill point, the elevation at which water exits the depression, was selected using the spill point function in 3D Analyst, and confirmed using multiple years of aerial imagery and DEMs. This elevation point was then used in the Volume and Surface Area tool in ArcMap. Volume calculations using both LiDAR derived DEMs and total station surveys were compared.

Pairing Restored Wetlands with Representative Natural Wetlands and Prior Converted Croplands

To assess the effectiveness of wetland restoration for the reestablishment of surface water volume storage, eleven wetland groups were formed. These groups contained one natural wetland, one restored wetland, and one prior converted cropland resulting in 33 representative sites across the wetland alteration gradient, most of which were also used to assess LiDAR reliability (above). The natural wetland and prior converted cropland pairs were selected to be within a 5 km buffer of each restored wetland. This approach minimized the influence of observed geographic gradients in depression morphology (Fenstermacher et al. 2014). Wetland morphologic characteristics (i.e., area, relief, and volume), typical for the region, were derived from the 33 representative sites using the LiDAR derived DEMs and volume calculation protocols described above.

Derivation of Equation for Calculating Regional Volume Storage

Hayashi et al. (2000) developed a generalized formula for deriving the volume of depressional wetlands, which was used to calculate volume storage in vernal pools (Brooks and Hayashi 2002). Vernal pools exhibit many of the same characteristics as Delmarva bays. The formula relates storage volume to surface area and relief by inclusion of a dimensionless constant P (see below).

Hayashi et al. Formula

A = Area, V = Volume, h = depth (relief), P = constant

$$V = (A * h)/(1 + 2/P)$$

Calibration of the formula for a given area requires calculation of a constant (P value), which varies from values of less than one to values greater than one for convex and concave depressions, respectively. To determine the average P value for Delmarva bays, 29 prior converted croplands and 29 natural bays (total 58) were characterized by measurement of volume storage, surface area, and relief using LiDAR derived DEMs and ArcMap (see above). The sites selected were within the interval between the 1st and 3rd quartile of the median size of depressional wetlands in the region (based on Fenstermacher et al. 2014). The distribution of P values was found to be normal, and average P value, along with measurements of area and relief, were used to estimate volume storage across the Delmarva Peninsula (see below).

Regional Estimation of Volume Storage

Using LiDAR coverage of the Delmarva, Fenstermacher et al. (2014) identified and hand digitized the point locations of nearly 15,000 Delmarva bays and then extrapolated findings to areas without LiDAR coverage for a total population estimate of 17,000 bays. Fenstermacher et al. (2014) further characterized the morphology of 1,494 depressions selected through stratified random selection (roughly 10 % of the measured population) by determining surface area, major/minor axis, and orientation by manual construction of morphometric polygons (Figure 2). The relief of these depressions was determined by measuring the elevation of three random points in the depression relative to rim locations. In the present study, a portion of this population subsample (n=1,372) was used to estimate volume storage for the full (~15,000) population set based on the Hayashi et al. (2000) equation.

Land Cover Assignment

Land cover data were retrieved from the 30 m resolution 2006 Multi-Resolution Land Characteristics (MRLC) Consortium National Land Cover Dataset (NLCD; Homer et al. 2015). Intersection of wetland polygons with the NLCD land cover map was used to classify land use for each depression. The threshold criteria used to designate a single land cover classification was 80 % of depression surface area. When less than 80 % of the depression area was occupied by one land cover class, the depression was classified as having a mixed land cover. In the case of the 58 wetland subset, land cover assignment based on NLCD was compared to that obtained using high resolution (~ 1 m) aerial photography. In the case of the ~1,400 wetland subset, NLCD was used to assign land cover based on both measured radii of the morphometric polygons and median radii of the subset population.

Results and Discussion

Comparing Volume Estimates: Ground based Surveys vs. Aircraft based LiDAR

Traditionally ground based surveys have been used to obtain accurate estimates of depressional storage volume, but the advent of aircraft based LiDAR systems holds promise for expanded coverage. Variable wetland characteristics, such as vegetation cover, can obstruct bare earth determinations required for accurate LiDAR DEM development, thus biasing LiDAR based estimates of volume storage. We explored this potential limitation by comparing estimates of volume storage derived using ground and LiDAR based methods. The depressions used in the analysis varied widely in ground based volume estimates (i.e., 79 m^3 to 26,700 m^3). Overall there was excellent agreement between volume estimates derived from ground and LiDAR based methodologies (Figure 3). On average, LiDAR derived volume estimates were within 3 % of those based on ground surveys. The largest discrepancies in terms of percent deviation occurred with two of the restored wetlands, perhaps due to the inability of ground surveys to capture the irregular shape (i.e., large islands and microtopography) of some restorations. Within the total study population, surface area to volume ratio trended downward with increasing surface area for both methods of assessment indicating that larger depressions were generally deeper. However depression size was not strongly predictive of the ratio ($r^2 = 0.11$).

Evaluating Ability to Estimate Storage Volume Based on Surface Area and Relief.
A population of depressions with similar geomorphic characteristics (i.e., Delmarva bays; Fenstermacher et al. 2014) raises the likelihood of being able to predict volume storage based on surface area and relief alone. This ability was enhanced by use of the generalized Hayashi et al. (2000) volume formula. The population of 58 depressional wetlands used to determine the range of Hayashi et al. (2000) P values for Delmarva depressional wetlands displayed a wide range of relief, volume, and surface area. All three of these parameters had skewed distributions with natural wetlands displaying greater skewness (Figure 4). By contrast P values were normally distributed for the total (natural + prior converted cropland) population with a mean value of 1.91, and there was no statistically significant difference in P values between natural wetlands and prior converted croplands. Moreover, within the study area P value was found to be independent of a considerable range in volume, relief, and surface area. An average P value of around 2 indicates a substantial concave morphology, which is in agreement with the findings of Fenstermacher et al. (2014). A more extensive assessment of morphometric attributes was assessed using 1,372 (~1,400) natural depressional wetlands and prior converted croplands that were probably former depressional wetlands whose perimeters had previously been hand delineated (Fenstermacher et al., 2014). Analyses of this population also found that both radius and relief were not normally distributed (Figure 5).

Regional Storage Volume Estimates
Results demonstrate that the coarser spatial resolution NLCD predicted prior converted croplands with 97 % accuracy and natural wetlands with 87 % accuracy when using actual digitized wetland boundaries. Accuracy was only slightly reduced (average 90 %) when median depressional radius (53 m) was used in conjunction with NLCD to determine land cover. Land cover was assigned to the ~1,400 previously delineated depressions using the NLCD based both on measured radii from the Fenstermacher et al. (2014) delineations and median radii (53 m), as well as the 80 % criteria (Figure 6). The classifications using these two methods were found to be in very good agreement (Table 1). Based on this approach, roughly 80 % of depressions fell within a single land type, including 53 % within the agricultural class and 27 % within the natural forestland class. These findings verified the suitability of using median radius and threshold values for scaling volume storage measurements per land cover type to the Delmarva Peninsula.

Fenstermacher et al. (2014) detected 14,492 (~14,500) depressions on cropland and forestland that were consistent with classification as a Delmarva bay. For these ~14,500 point locations, land cover was assigned using the NLCD and median radius. The ~14,500 depressions were then classified as either forestland, cropland, or mixed using the median radius and 80 %

inclusion rules. This assessment found that 81 % of the ~14,500 depressions fell within a single land cover class, which is in close agreement with the 80% finding pertaining to the ~1,400 depression subset. For the ~14,500 population, a majority (58 %) of the depressions were found to be located on cropland, whereas 23 % were estimated to be on forestland and 18% had mixed land cover (Figure 7). These results are again comparable to those obtained with use of the ~1,400 depression subset.

To account for skewness each depression was randomly assigned a surface area bin based on the distributions of depression radii for their respective land cover classes (see binned histograms in Figure 5). The mixed class locations were randomly assigned to bins in the combined (all) distribution. The depressions were then randomly assigned to relief histogram bins depicted in Figure 5 based on distributions of depression relief for their respective land cover classes. This use of double randomization was found to have validity because analysis of the ~ 1,500 delineated polygons demonstrated that surface area and relief were not correlated ($r = 0.08$). With completed attribute assignment for surface area and relief, volume was then calculated using the Brooks-Hayashi formula using a P value equal to 1.91, as determined using the 58 calibration sites (Figure 4).

Total volume storage of all identified depressions (~14,500) on cropland and forestland was determined to be 35,900 ha-m. Estimated water storage volume for depressions on cropland sites totaled 16,900 ha-m, compared to12,400 ha-m for natural sites and 6,600 ha-m for sites with mixed classification. A substantial portion of the mixed class depressions is also likely drained for crop production adding to loss of volume storage. The median storage capacities of depressions were 13,000, 17,100 and 15,300 m^3 for cropland, forestland, and mixed land covers, respectively. The forested depressions tended to have greater surface area (radius, Figure 5), which likely accounted for their greater storage capacity. Nevertheless, due to a larger number of depressions found on cropland, this land cover classification had the greatest potential for volume storage if not drained for agricultural purposes. These results reveal the great potential for volume storage gain with wetland restoration programs.

This Delmarva Peninsula study covered an area of approximately 1.55 million ha. By comparison, Gleason and Tangen (2007) assessed volume storage in the Prairie Pothole Region (PPR) over an area of 445,000 ha, and estimated a total depressional storage capacity of approximately 56,500 ha-m. The higher storage capacity per unit land area in the PPR is likely accounted for by a couple of factors. One factor is that landscape relief on the Delmarva is less than that of the PPR and another is that Delmarva bays are concentrated in the upper portion of the Peninsula with the southern portion having a low density of depressions (Fig. 7). If analysis was limited to the high density region of the Peninsula, regional estimates would become more comparable on a per area basis.

Volume Storage in Wetland Restorations

In the paired wetland assessment, the morphometric properties of 11 restored wetlands were compared to those of geographically similar natural wetlands and prior converted croplands

(Figure 8). Data showed that the range of restored wetland properties tended to be less than those observed for natural wetlands and prior converted croplands. This trend was particularly strong for area and volume. Median volume storage of restored wetlands (1,480 m^3) was between 53 to 60 % of that for prior converted croplands and natural wetlands, respectively. The measured volume storage for restored wetlands was also considerably smaller than median volumes estimated for wetlands in the 58 site calibration set (natural: 5,450 m^3 and prior converted cropland: 8,260 m^3) and almost a factor of 10 smaller than the population estimates of volume for depressions on cropland and forestland covering the Delmarva (see section above).This finding, in conjunction with the substantially lower number of restored wetlands on the Delmarva Peninsula relative to prior converted croplands and natural wetlands, indicates that current restorations likely have limited impact on volume storage within the region, but that the potential surface water volume storage gain from restoration is great.

Summary and Conclusions

A recent survey of the Delmarva Peninsula discovered nearly ~15,000 circular or semi-circular depressions with features consistent with the morphology of Delmarva bays. The MIAR study component described herein found that a majority of these depressions (58 %) are located on actively farmed cropland, representing a loss of 16,900 ha-m of potential volume storage to agricultural production. Furthermore, an additional 18 % identified depressions were found in areas of mixed cropland – forestland cover, representing a likely loss of potential volume storage of 6,600 ha-m. This combined estimated loss of 23,500 ha-m of potential surface water volume storage has likely had a substantial, negative impact on hydrologic flows, likely increasing the likelihood and duration of stream-overbank events and resultant floods.

Wetland restoration can aid in the re-establishment of these lost hydrologic flow and natural hazard mitigation services. An earlier Choptank CEAP-Wetland study that involved four natural and three restored wetlands as well as four prior converted cropland sites found that natural wetlands exhibited relatively continuous flow into adjacent streams in contrast to prior converted croplands, which provided flashier flows directly after precipitation events (McDonough et al. 2015). Restored wetlands exhibited surface water flows intermediate to natural wetlands and prior converted croplands. Wetland area was found to be significantly correlated with the periodicity of surface water flows. Even when depressional wetlands are not directly connected to streams via surface water flow, their size and arrangement has been found to be critical for supporting flow in adjacent streams (McLaughlin et al. 2014). Although wetland restoration has been found to exert a positive effect on the regulation of hydrologic flows and likely natural hazards, the extremely large volume of surface water storage that has been lost at a landscape scale relative to the modest gains in water storage made possible by restoration highlights the need for increased, sustained restoration. In general, the agricultural landscape of the Delmarva Peninsula has a very high capacity for increased surface water volume storage. Implementation of wetland restoration and drainage control structures can take advantage of potential volume storage capacity on croplands.

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Tables

Site Type	Original Polygons		Median Radius	
	Single Land Type	Mixed	Single Land Type	Mixed
All Sites	1,095	257	1,117	238
Cropland	732	118	729	107
Forestland	363	139	388	131

Table 1: Comparison of land cover classifications using hand delineated polygons to those based on median radius and 80 % area threshold for single land type classification.

Figures

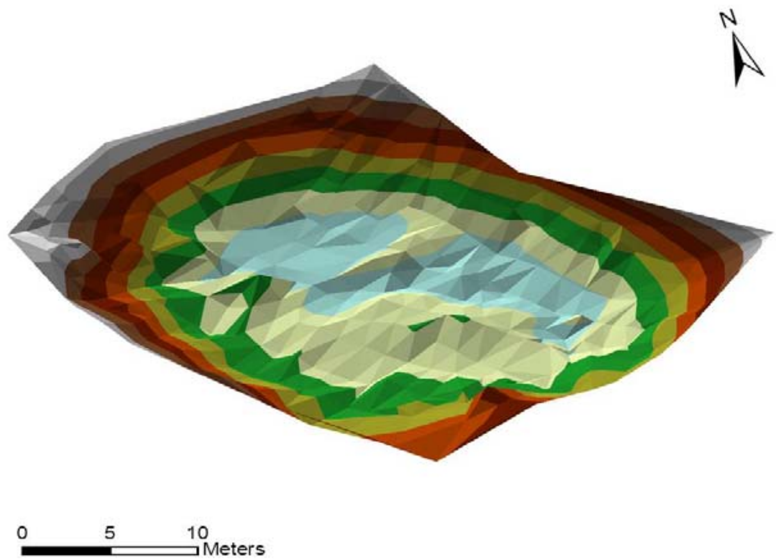

Figure 1: Example of a TIN created from ground based survey data.

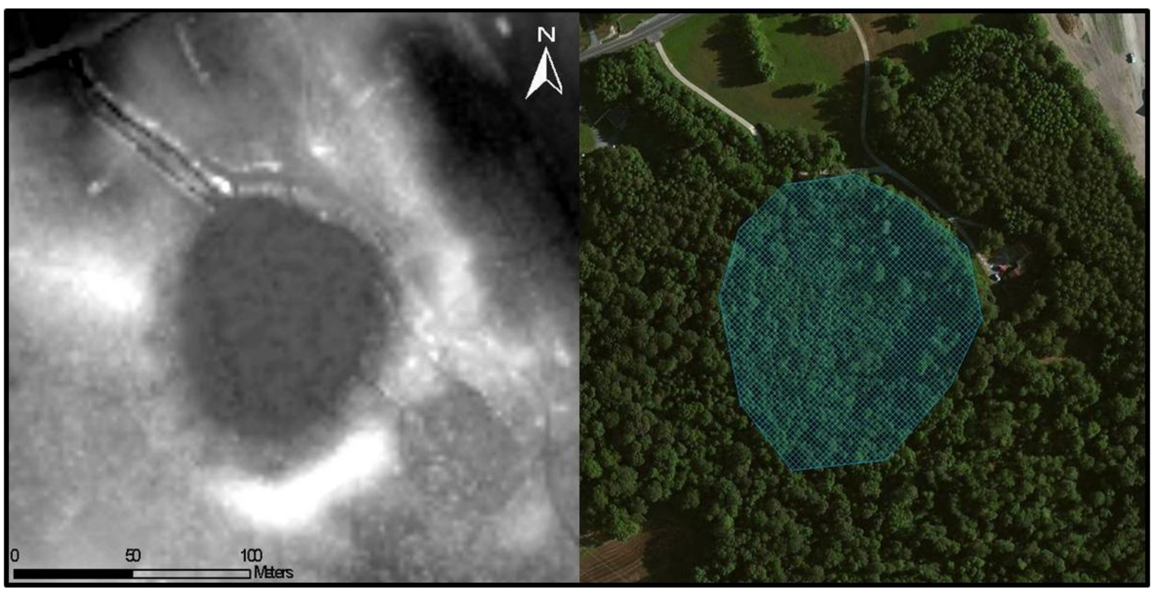

Figure 2: Example of a LiDAR derived DEM (left) depicting a depressional forested wetland, and the associated hand delineated polygon created to represent the depression surface area over-laid on an aerial photograph (right).

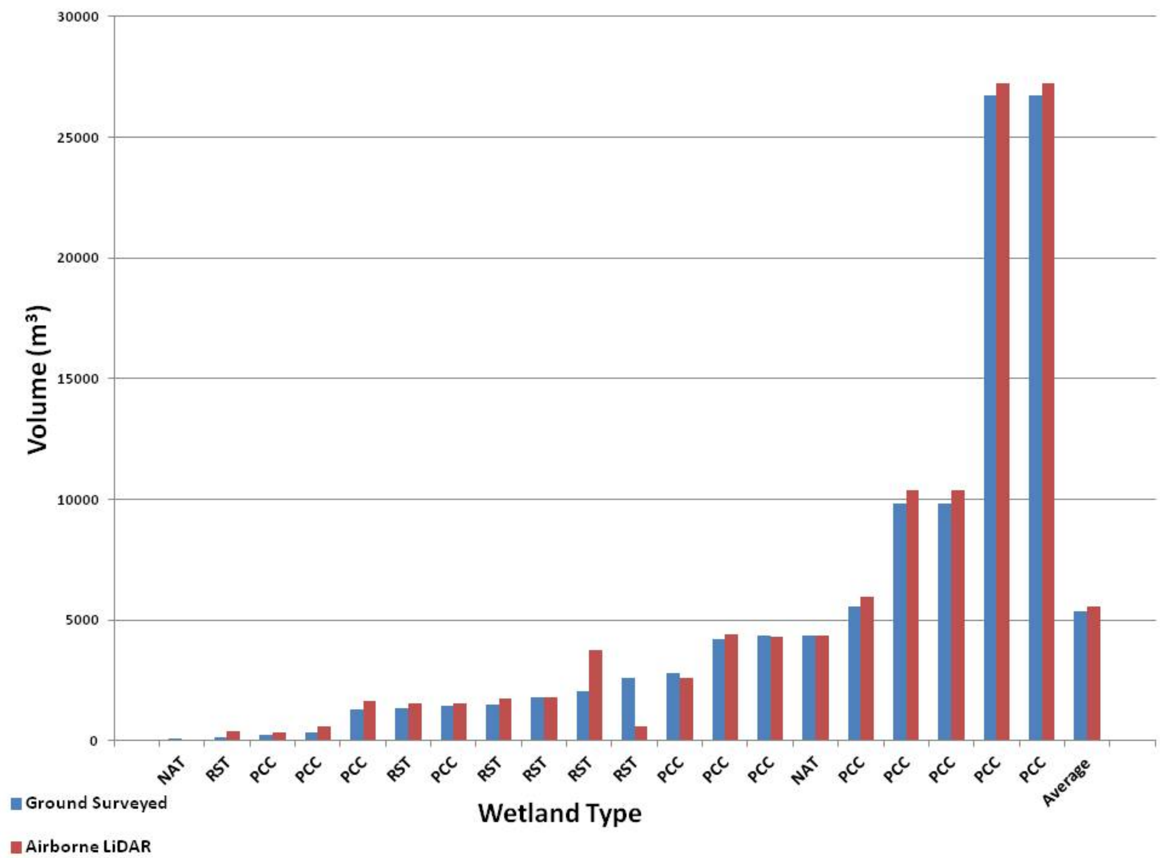

Figure 3: Comparison of volume estimates using ground surveys and airborne LiDAR digital elevation models at prior converted cropland [PCC], natural [NAT], and restored [RST] sites.

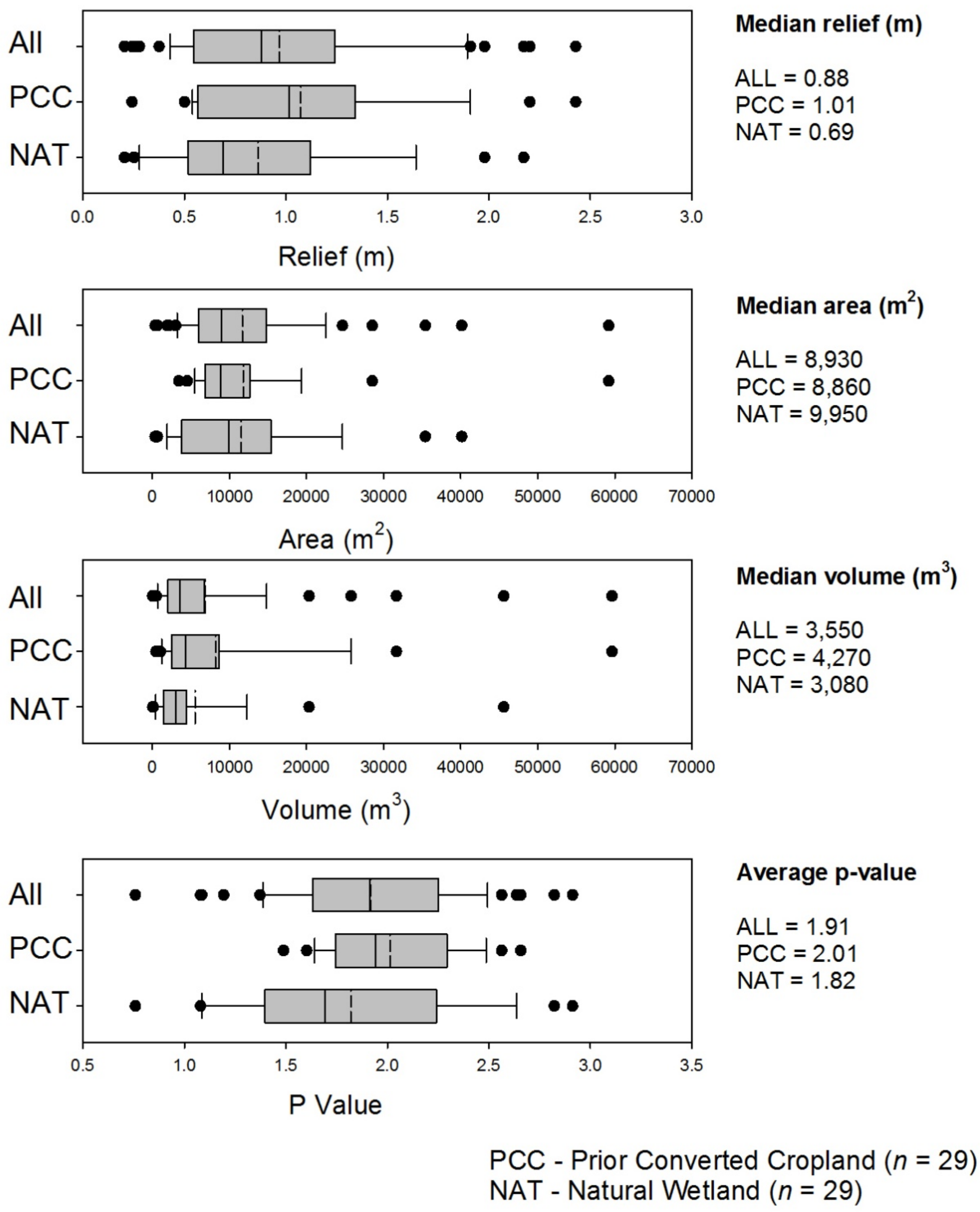

Figure 4: Range in morphometric properties for 29 prior converted croplands (PCC) and 29 natural wetlands (NAT) used for calibration. Graph representations: boxes = 25th and 75th percentiles; whiskers = 10th and 90th percentiles; dots = outliers; solid line = median; dashed line = mean.

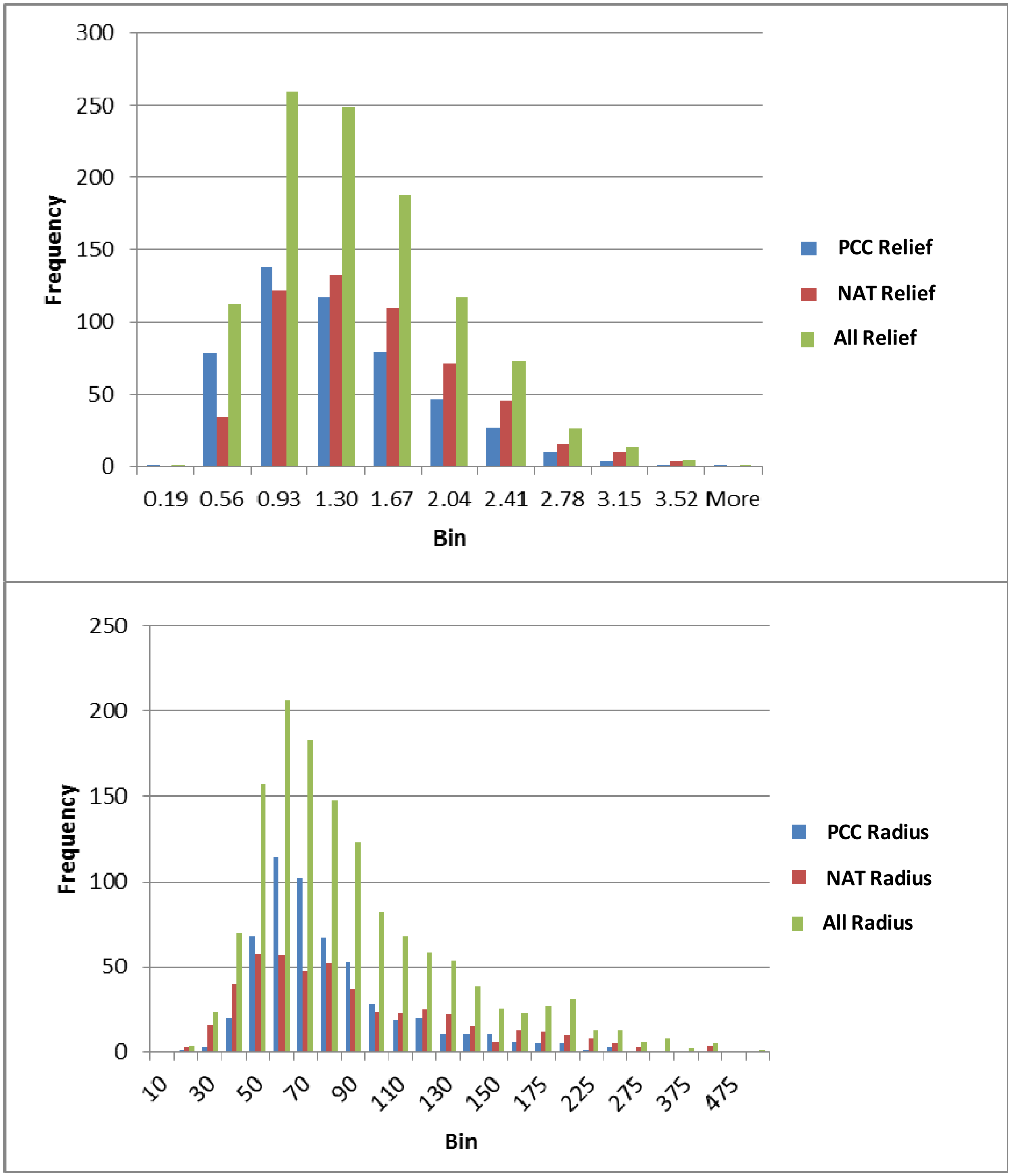

Figure 5: Distributions for relief and radius of ~1,400 natural wetlands and prior converted croplands. Histograms were created using 26 bins for depression radius and 11 bins for depression relief. Values on both x axes are in meters.

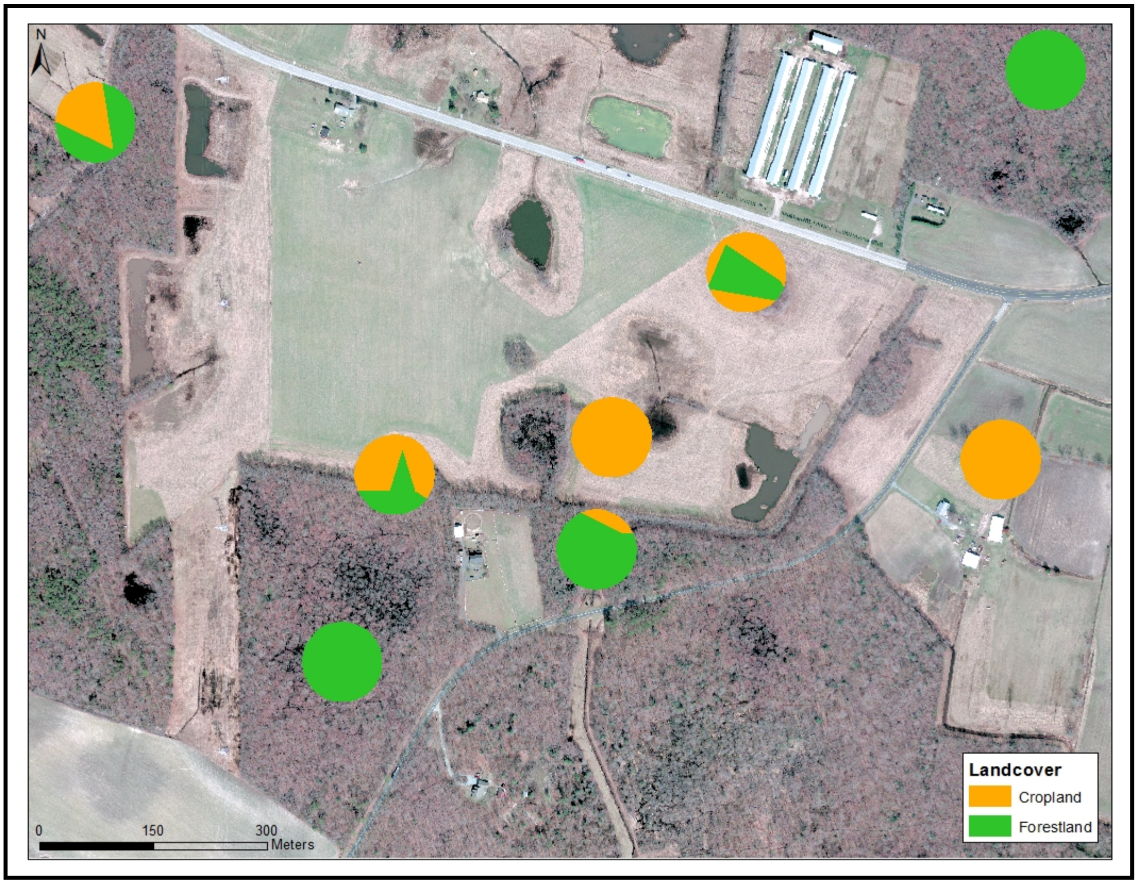

Figure 6: Median radius (53 m) polygons showing different degrees of mixed land cover.

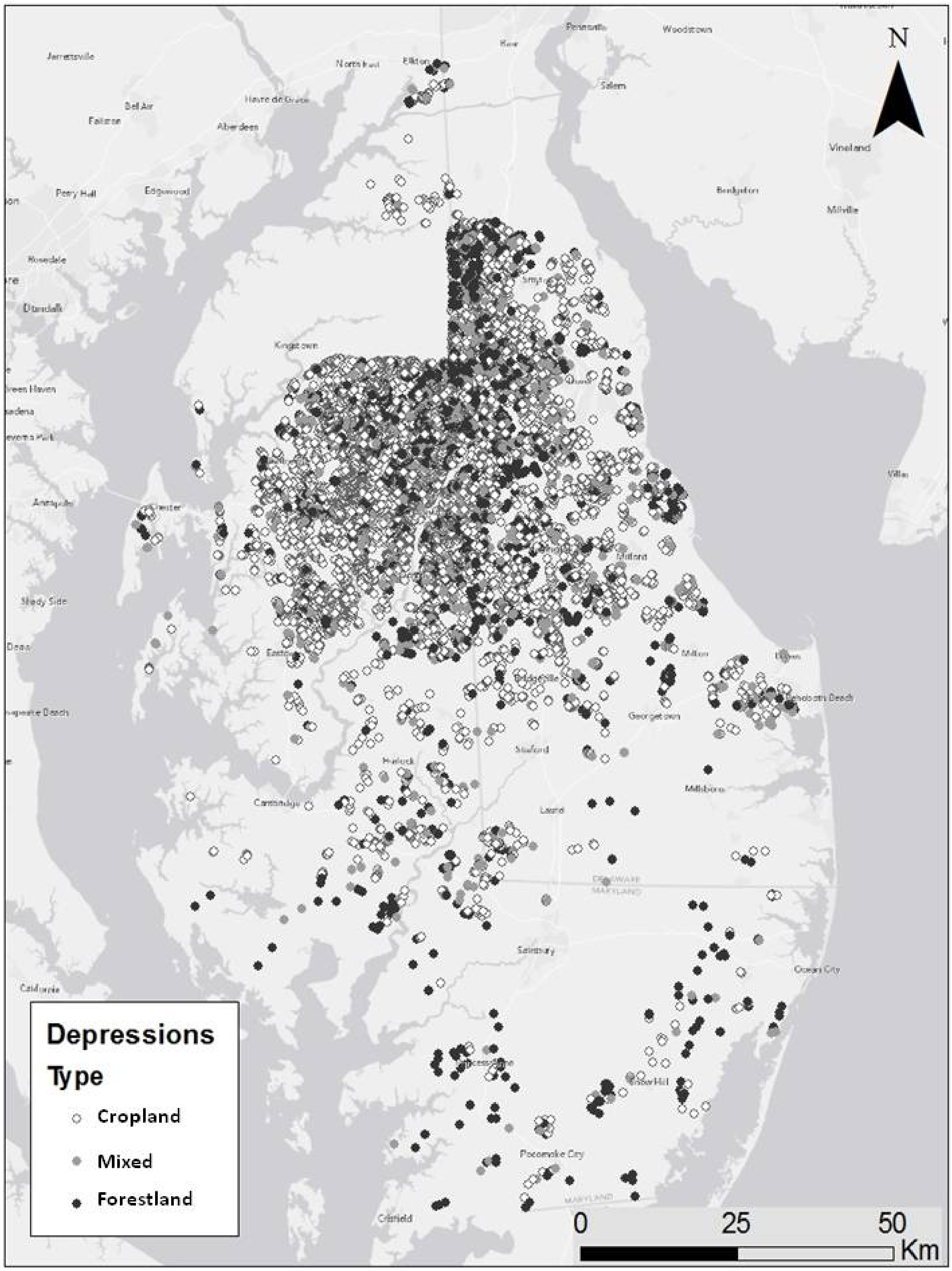

Figure 7: Distribution of depressions with cropland, forestland, and mixed land cover classifications.

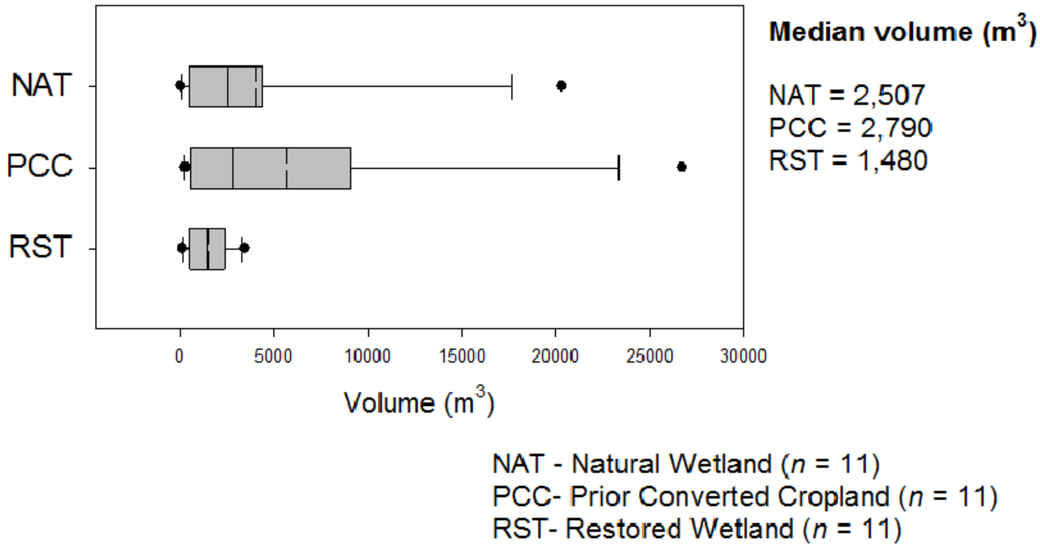

Figure 8: Range in volume storage for the paired sets of restored (RST), prior converted cropland (PCC) and natural (NAT) sites. Graph representations: boxes = 25th and 75th percentiles; whiskers = 10th and 90th percentiles; dots = outliers; solid line = median; dashed line = mean.

6. Plant Community Biodiversity and Quality

Key Findings

-While prior converted croplands were highly disturbed with low plant community biodiversity, restored wetlands were hotspots of diversity (i.e., species richness) – surpassing even natural wetlands. Species evenness was similar between restored and natural sites.

-Although natural and restored wetlands were both dominated by native species, their plant communities were primarily composed of different functional types. Whereas woody species composed > 70 % of the plant community at natural sites, they composed < 20 % at restored sites. Furthermore woody species accounted for > 70 % of plant cover in natural sites and only 10 % of cover in restored sites.

-Species found in natural sites were less associated with disturbed conditions than those found in restored sites, as indicated by coefficients of conservatism. This was also reflected in an anthropogenic activity index that indicated that restored sites were four times more impacted by human disturbance than natural sites.

-Floristic quality assessment index may be a more robust indicator of plant community integrity than the floristic assessment quotient for wetlands when comparing plant communities in natural and restored wetlands.

***Recommended Practices*:** Removal of the seed bank through excavation and intensive post-restoration management of the plant community (e.g., mowing) should be avoided, while hydroperiods and water depths characteristic of natural wetlands should be supported to encourage colonization and growth of species that are representative of more natural conditions. Planting of wetland species at restored sites does not appear to be necessary for producing a rich, native plant community. Placing restored wetlands near natural wetlands may encourage the dispersal of native species to restored sites, especially when multiple dispersal mechanisms exist (e.g., surface water connection and wind dispersal). Due to the significant amount of time required to develop a woody plant community more characteristic of natural wetlands longer contract or easement periods are desirable.

***Primary Chapter Source*:** Yepsen, M., Baldwin, A., Whigham, D., McFarland, E., LaForgia, M., and Lang, M. 2014. Agricultural wetland restorations achieve diverse native wetland plant communities but differ from undisturbed wetlands. *Agriculture, Ecosystems and Environment*, 197:11-20.

Introduction

Biological indicators of ecosystem integrity have been developed into rapid field assessment methods (Fennessy et al. 1998; Lopez and Fennessy 2002; Fennessy et al. 2004). These assessments can be used to describe overall ecosystem condition, suggest probable causes of poor conditions, identify human activities that contribute to degradation, monitor wetland restoration trajectories, and set and assess measureable goals (Galatowitsch et al. 1999; Cronk and Fennessy 2001). Karr and Dudley (1981) define ecosystem integrity as: "the capability of supporting and maintaining a balanced, integrated, adaptive, community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region." Ecosystem integrity is thought to be inversely related to human disturbance because disturbances can change nutrient cycling, photosynthesis, hydrology, competition, predation, and more.

Plants are one of the easiest and most frequently used factors for assessing the progress of wetland restoration (Mitsch and Wilson 1996). Plants are adapted to normal natural variations in physical conditions and plant communities reflect current as well as historic conditions (Bedford 1999; Cronk and Fennessy 2001). Some of the advantages of using plants as biological indicators include: 1) they are present in most wetland ecosystems; 2) they are relatively easy to identify; 3) established methods for sampling exist; and 4) their immobility creates a direct link between onsite environmental conditions and plant community characteristics (Cronk and Fennessy 2001). Because of these traits, plant communities are a robust mechanism for comparing the condition of wetlands along a human alteration gradient. This study component compared plant community biodiversity and quality across a wetland alteration gradient, including natural wetlands, prior converted croplands, and restored wetlands and considered the relationship between these metrics and human disturbance.

Methods

Plant community surveys were conducted at each of the MIAR sites during peak growing season (late June through September 2011) to minimize differences due to time of year. The areas sampled in natural and restored wetlands were within the wetland boundary, excluding ponded areas without vegetation. Prior converted cropland sites were sampled within approximately 25 m of the center of the wettest drained area. Given adequate area, three randomly placed 10 x 10 m quadrats were sampled per plant community at each site. Where space was limited, quadrat shape was modified, and if three sampling locations could not be accommodated per plant community, fewer than three locations were sampled. In order to ensure adequate sampling, quantitative cover data for all dominant plants and 90 % or more of the species in each site were captured in the quadrats, the latter based on surveys of species that were at each site but not encountered in the plots. Each species within a quadrat was assigned a percent cover class (Trace, 0-1, 1-2, 2-5, 5-10, 10-25, 25-50, 50–75, 75–95, or >95; Peet et al. 1998). Common plant species were defined as those that occurred within quadrats at 40 % or more of natural, restored, or prior converted cropland sites. Dominant plant species within each site type were defined as those that had cover of 20 % or higher in the sites in which they were found. Percent cover of a species and percent cover of woody and herbaceous species at each site were calculated as the average of the species' cover across all quadrats at the site, with 0 % cover assigned in quadrats where the species was not found.

Natural, restored, and prior converted cropland sites were compared using 9 vegetation indices commonly used to determine differences in wetland condition. The indices are described in Table 1 and include the Shannon-Weiner evenness index, species richness, wetness coefficient, coefficient of conservation, proportion of woody species, proportion of native species, the floristic quality assessment index, the floristic assessment quotient for wetlands, and the anthropogenic activity index. The indices were calculated using the quadrat cover data and the presence of species found inside and outside the quadrats. Indices were averaged for each site and then wetland type. Statistical analysis was used to compare site types according to index scores. Comparisons of natural, restored, and prior converted cropland sites were made using analysis of variance (ANOVA) performed using the MIXED procedure in SAS version 9.2 (SAS Institute, Cary, NC). Arithmetic mean and standard error were calculated using the MEANS procedure in SAS. Regressions were calculated using the REG procedure in SAS and SigmaPlot (Systat Software, San Jose, CA), comparing: 1) the anthropogenic activity index (AAI) to the floristic quality assessment index (FQAI), 2) AAI to floristic assessment quotient for wetlands (FAQWet) scores, and 3) time since restoration to percent cover by woody species in restored sites. Variation in plant communities between sites and their relation to metrics of diversity, quality, and disturbance were also examined using non-metric multidimensional (NMS) analysis. Sørenson distance measures were used and Beal's smoothing was applied to vegetation data to achieve a final stress value near 10 (McCune and Grace 2002). Significant difference between wetland types was tested using multi-response permutation procedures (MRPP). Both NMS and MRPP analyses were conducted using PC-ORD v. 6 (MjM Software, Gleneden, OR). Joint plots were prepared to visualize site scores relative to vectors of plant community and disturbance metrics.

Results

A total of 204 species were observed across the three site types with 71 species found in natural sites, 134 in restored sites, and 34 in prior converted cropland sites. Four species (*Hypericum mutilum, Phytolacca americana*, *Diospyros virginiana*, and *Liquidambar styraciflua*) were found at all three types of sites. There was no overlap in dominant species between site types and little overlap in common species (Figure 1). Woody species accounted for more than 70 % of cover in natural sites, typically from *Liquidambar styraciflua*, *Acer rubrum,* and *Nyssa biflora* (Figure 1A). These species, and others, shaded an understory shrub and small tree stratum, which contained species such as *Clethra alnifolia, Smilax* species, *Eubotrys racemosa, Magnolia virginiana,* and various *Vaccinium* species. By contrast, only 10 % of cover in restored sites was from woody species, and there was greater variation in plant community composition between sites. While woody species, such as *L. styraciflua* and *A. rubrum,* were found in 30 % to 50 % of restored sites, they averaged <1 % cover. *Echinochloa crus-galli, Xanthium strumarium*, *Scirpus purshianus*, *Phragmites australis*, and *Mollugo verticillata* were frequently found in restored sties and tended to have relatively high cover (Figure 1B). Only 7 herbaceous species were found in both restored and natural sites. Of those seven species, only *Scirpus cyperinus* and *Woodwardia virginica* were found in more than one site of each type. Prior converted cropland sites were dominated by conventional row crops of *Zea mays*, *Glycine max*, *Gossypium hirsutum*, or *Sorghum bicolor* (Figure 1C).

For most indices, restored sites were more similar to natural sites compared to prior converted cropland sites (Figure 2). Natural and restored sites had similar proportions of native species and evenness (Figure 2C and 2D). Natural sites, however, had a significantly higher proportion of woody species, higher FQAI scores, and lower anthropogenic disturbance (Figures 2B, 2G, and 2I). Restored sites, compared to natural sites, had significantly higher species richness and the plant species were more wetland specific (Figures 2A and 2E). The NMS ordination indicated that plant communities in the three types of sites were significantly separated from each other (MRPP A-statistic = 0.35, p <0.0001; Figure 3). Axes 1 and 2 cumulatively explained 87.6% of the variation in species composition.

There was a stronger negative correlation between AAI and FQAI than AAI and FAQWet scores, but both were significant (R^2 = 0.65 and 0.30 respectively, p< 0.001). The correlation between percent cover by woody species and number of years since restoration in restored sites was not significant (R^2 = 0.24, p = 0.17).

Discussion

Restored sites appeared to be following a trajectory of recovery, with indices generally more similar to natural than prior converted cropland sites. Given enough time and without intensive site management (e.g., mowing), the structure and function of restored wetlands should become more like natural sites, although completion of this process may take decades if not over a century (Mitsch and Wilson 1996; Moreno-Mateos et al. 2010). One notable exception to the greater similarity between restored and natural sites was the low proportion of woody species found in restored wetlands relative to natural wetlands. The restored wetlands sampled in this study were relatively young (i.e., sampled 3 to 11 years post restoration), but we anticipated that the oldest would have increased cover and density of woody species found in nearby reference sites. De Steven et al. (2006 and 2010), who conducted a similar study, found that cover of woody species in restored wetlands averaged 40 % after 5 years and that restored sites had 53 % of species in common with forested reference sites. In contrast, we found that restored sites had very few of the woody species found in natural sites and that there was no correlation between the time since restoration and percent cover of woody species 3 to 11 years post restoration.

Several explanations could account for the low number of woody species at our restored sites, including the lack of an adequate seedbank, dispersal limits, water depth and hydroperiod restrictions, and continued anthropogenic disturbance. Since few of the restored sites were planted with woody species and the majority of the trees that were planted were not the same species found in the natural sites (e.g., *Platanus occidentalis* and *Taxodium distichum* were not found in natural sites), seed banks and seed dispersal remain the major sources of propagules in restored wetlands. However, seed banks of farm fields tend not to contain woody species (De Steven et al. 2006; Middleton 2003). Even if viable seeds of native woody species were present in the seed bank at the time of restoration, many of them would have been removed through excavation of topsoil, which is often not replaced (Chapter B2). Because restored wetlands in agricultural areas are often surrounded by farm fields rather than forested wetlands, dispersal limitations may explain the lower abundance of woody species (Herault and Thoen 2009; Kettenring and Galatowitsch 2011; Middleton 1999 and 2003). Wetland depressions, which do not receive overland flow of water from other wetlands, would primarily be dependent on wind transport. Wind based dispersal of seeds from woody species declines exponentially from forest edges into clearings (Greene and Johnson 1996). Clewell and Lea (1990) suggested that wetland restoration sites within two tree heights of a forest composed of mature trees would have the most successful natural regeneration of early colonizers. However, even if propagules are present, environmental conditions must be compatible with germination and growth. Excavation often creates pond-like conditions, extending hydroperiod and increasing depth of inundation, both of which play an important role in determining species composition. Herbaceous wetland species are only abundant in natural wetlands in our study area when they have a deeper zone where water depths preclude the establishment of woody species (Tyndall et al. 1990; Verhoeven et al. 1994). Forested wetlands exist only where hydroperiod is long and deep enough to exclude upland species, but not so wet as to kill trees (Lugo 1990). In one study, higher cover by woody plant species was correlated with shorter hydroperiods 5 years post-restoration (De Steven et al. 2010). One final explanation as to why restored sites remained largely dominated by herbaceous species is continued anthropogenic disturbance. Many sites were regularly mowed and some had evidence of recent disturbance by heavy machinery. Regular mowing and soil disturbance are effective methods for preventing the establishment of woody species.

Restored sites had the highest species richness of all the site types, with evenness similar to natural sites. This finding was relatively consistent with other studies of recently restored freshwater wetlands (Balcombe et al. 2005; Ficken and Menges 2013; Gutrich et al. 2009; Matthews et al. 2009). Although matching species richness of reference (i.e., natural) sites is a common management goal, species identity should also be considered to determine whether richness indicates ecological health, degradation, or neither. Species richness may be increased by the presence of weedy and invasive species, removal of nutrient limitations through the addition of pollution, or by creating more micro-habitats where upland species can colonize (Ehrenfeld 2000; Ehrenfeld and Schneider 1993). Proportion of native species was similar between restored and natural sites, but restored sites hosted a greater proportion of species with lower coefficients of conservatism (e.g., generalists). This is likely due, at least in part, to the early successional state of restored sites.

Plant communities are biologic indicators of ecosystem integrity and have been used to determine restoration success. A number of indices have been developed to quantify plant community quality. Floristic quality indices incorporate plant species identity as well as richness, allowing for meaningful comparisons between sites (Lopez and Fennessy 2002). Although we recognize that coefficients of conservatism have not been developed for all regions of the United States and assignment of coefficients of conservation per species is somewhat subjective, our findings suggest that the FQAI index is preferable to the FAQWet index for comparing floristic quality between restored and natural wetland depressions. Because FAQWet scores are based on native species richness and species tolerance for wetland conditions a higher FAQWet score may indicate a more diverse and native wetland plant community or it may indicate longer hydroperiods (Ervin et al. 2006). Although not examined quantitatively, hydroperiods at our restored sites appeared to be longer than those at adjacent natural sites. Therefore the FAQWet scores may be a misleading indicator of biologic integrity. Similar to Tietjen and Ervin (2007), we found that FAQWet was only weakly correlated with AAI, also indicating that FAQWet may not be a good indicator of ecosystem integrity. By contrast, FQAI scores are related to species' habitat requirements and tolerance of disturbance, and it has been found by our study and others to be strongly correlated with anthropogenic disturbance (Lopez and Fennessy 2002).

Conclusions

Restored depressional wetlands in the MIAR have developed diverse native wetland plant communities that are likely to provide many of the broad functional benefits targeted by the USDA programs that supported their implementation. However, after 3 to 11 years post restoration many tree and shrub species that occurred in natural wetlands were not represented in restored wetlands. Although this may be due to multiple factors, some of which are not directly controlled by operational managers, actions can be taken by operational staff to increase the likelihood that restored wetland plant communities will be more similar to their natural counterparts. These actions include changes to targeting, implementation and management actions. Potential wetland sites that are adjacent to natural wetlands should be targeted to increase the likelihood of seed dispersal between natural and restored wetlands. To avoid the removal of existing seedbanks, sites which are likely to support wetland hydrology without excavation should be prioritized for restoration. Excavation should be avoided; but when this is not possible removed topsoil should be replaced. Wetland hydrology should be restored to a level similar to surrounding natural wetlands. Mowing should be discouraged. Finally, plant succession and related community composition changes take place over decades if not longer. Thus longer-term easements or contracts may be needed to fully restore a more natural plant community.

It is important to recognize that although wetland restorations provide important ecosystem services, wetlands with different plant communities (e.g., forested versus herbaceous) inherently provide different services. Furthermore, plant type and structure is an important component of animal habitat requirements. Thus, a gradual shift towards herbaceous wetland communities, away from forested communities, may result in a significant effect on ecosystem services and habitat provision at the landscape scale. This fact, and its consequences, should be considered when defining wetland restoration standards and goals.

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Tables

Index	
Shannon-Weiner evenness index	The Shannon-Weiner evenness index was calculated for a site based on the species richness and cover of the species found in quadrats. It is calculated as [sum(AC*ln(TC)]/ ln(SR); where AC is average cover of each species found in quadrats at the site with 0% cover assigned in quadrats where the species was not found; TC is the sum of all AC values found per site; and S is number of species found in quadrats at the site (Gurevich et al. 2006).
Species richness	The total number of species found in each site.
Wetness coefficient (WC)	Value assigned to plant species between -5 (always occurs in non-wetland upland areas) and +5 (always occurs in wetlands). Based on regional U.S. Fish and Wildlife Service Wetland Indicator Status (Ervin et al. 2006; Lichvar and Kartesz 2009).
Coefficient of conservatism (CC)	Value assigned to plant species between 0 (non-native or weedy species that tolerate disturbance and are found in wide variety of conditions) and 10 (native species that are only found in specific undisturbed conditions) (Chamberlain and Ingram 2012).
Proportion of woody species	The percent of species found in each site characterized as woody (as opposed to being an herb or vine) (USDA Plants Database).
Proportion of native species	The percent of plants identified to species found in each site that were native (USDA Plants Database).
Floristic Quality Assessment Index (FQAI)	Site wide index based on native species richness and coefficients of conservatism. A low score indicates low native species richness and/or species that tolerate a wide range of conditions and disturbance. A high score indicates high native species richness and plants that are only found under specific conditions and do not tolerate disturbance. Calculated as FQAI=$R/\sqrt{N}$; where R is the sum of the coefficients of conservatism for all species found at a site and N is the number of native plants identified to species in each site (Andeas and Lichvar 1995; Ervin et al. 2006; and Lopez and Fennessy 2002).
Floristic Assessment Quotient for Wetlands (FAQWet)	Site wide index based on species richness, species nativeness, and whether species are more commonly found in wetlands or uplands. A high score indicates high native species richness and plants that are found in wetlands. FAQWet uses wetness coefficients, which have been developed for the entire U.S. It places a heavier weight on non-native plant species than FQAI. It is calculated as FAQWet=(SumWC)/($\sqrt{S}$)(N/S); where WC is the wetness coefficient value assigned to each species, S is species richness, and N is number of native species (Ervin et al. 2006; Herman et al. 1997; Reed 1988).
Anthropogenic Activity Index (AAI)	A qualitative index for assessing human disturbance based on observations during site visits. AAI rates wetlands on a scale of 0-3 for five conditions: land use intensity in a 500-m buffer; intactness and effectiveness of a 50-m buffer; hydrologic alteration; habitat alteration; and habitat quality and microhabitat heterogeneity (Ervin et al. 2006).

Table 1: Summary of vegetation and disturbance indices used (adapted from Yepsen et al. 2014).

Figures

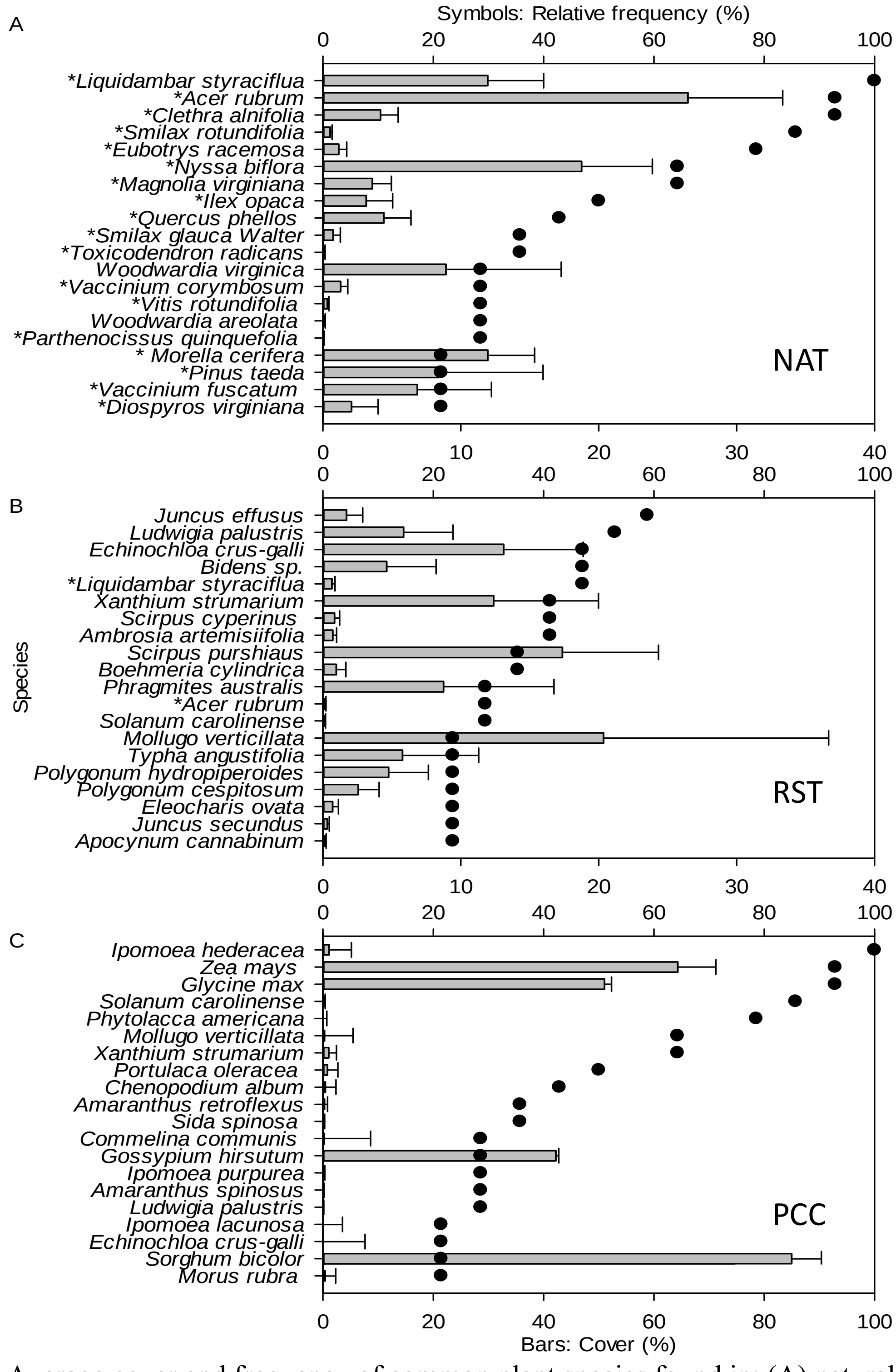

Figure 1: Average cover and frequency of common plant species found in: (A) natural (NAT), (B) restored (RST), and (C) prior converted cropland (PCC) sites in the MIAR. Plotted cover values are mean +1SE. * woody species. Dots represent frequency and bars represent average percent cover (adapted from Yepsen et al. 2014).

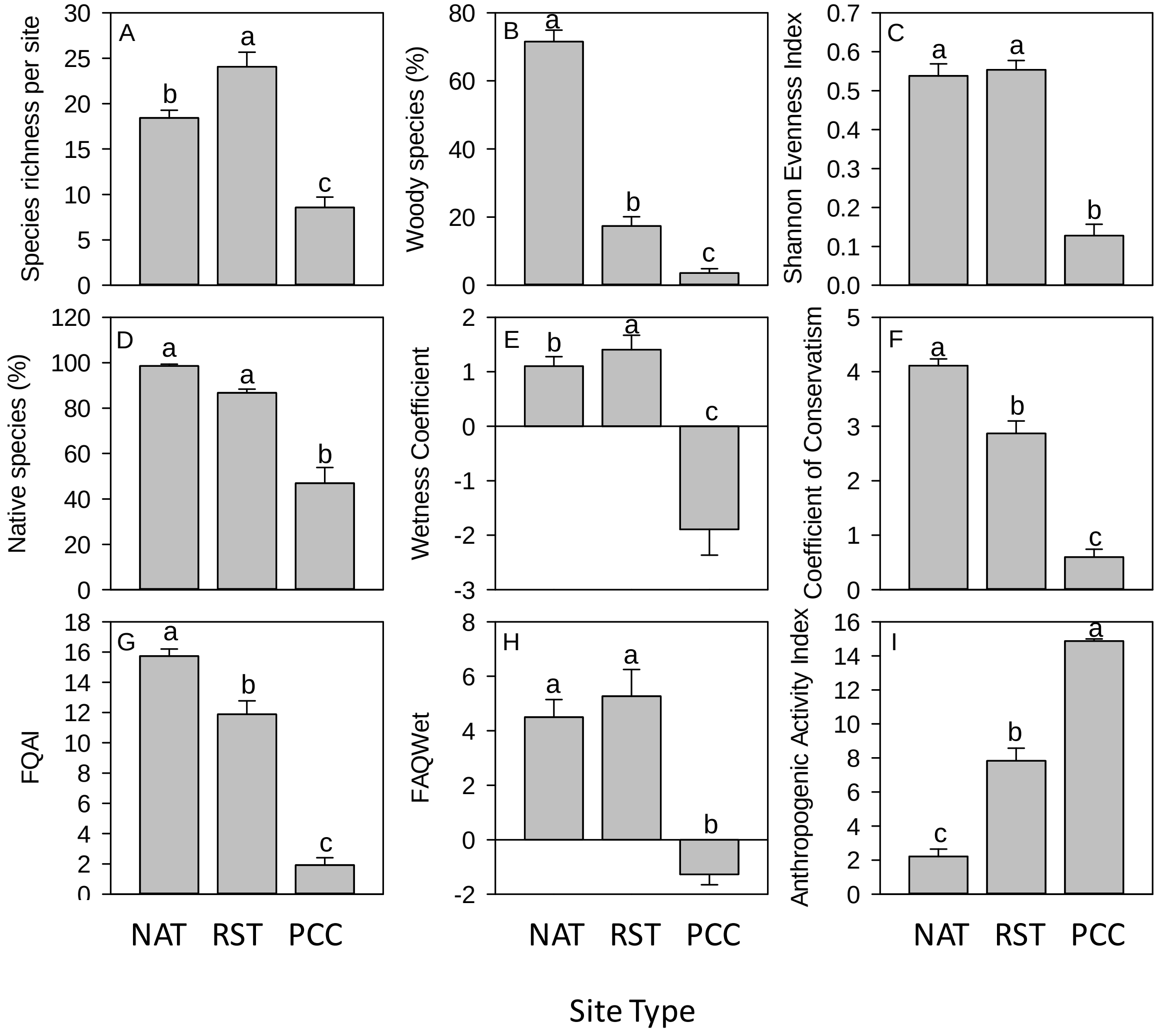

Figure 2: Plant community metrics for natural (NAT), restored (RST), and prior converted cropland (PCC) sites in the MIAR. Plotted values are mean + 1SE; means with different letters represent statistically significant differences (Tukey, $p < 0.05$). A: Species richness per site (number of species) ($F_{2,44} = 38.7$, $p < 0.0001$); B: Percent of species that were woody ($F_{2,44} = 183.8$, $p < 0.0001$); C: Shannon index of species evenness ($F_{2,44} = 77.15$, $p < 0.0001$); D: Percent of species that were native to the USA ($F_{2,44} = 35.66$, $p < 0.0001$); E: Wetness coefficient ($F_{2,44} = 35.66$, $p < 0.0001$); F: Coefficient of conservatism ($F_{2,44} = 97.50$, $p < 0.0001$); G: Floristic quality assessment index (FQAI) ($F_{2,44} = 115.99$, $p < 0.0001$; H: Floristic assessment quotient for wetlands index (FAQWet) ($F_{2,44} = 24.65$, $p < 0.0001$); I: Anthropogenic activity index ($F_{2,44} = 137.46$, $p < 0.0001$; adapted from Yepsen et al. 2014).

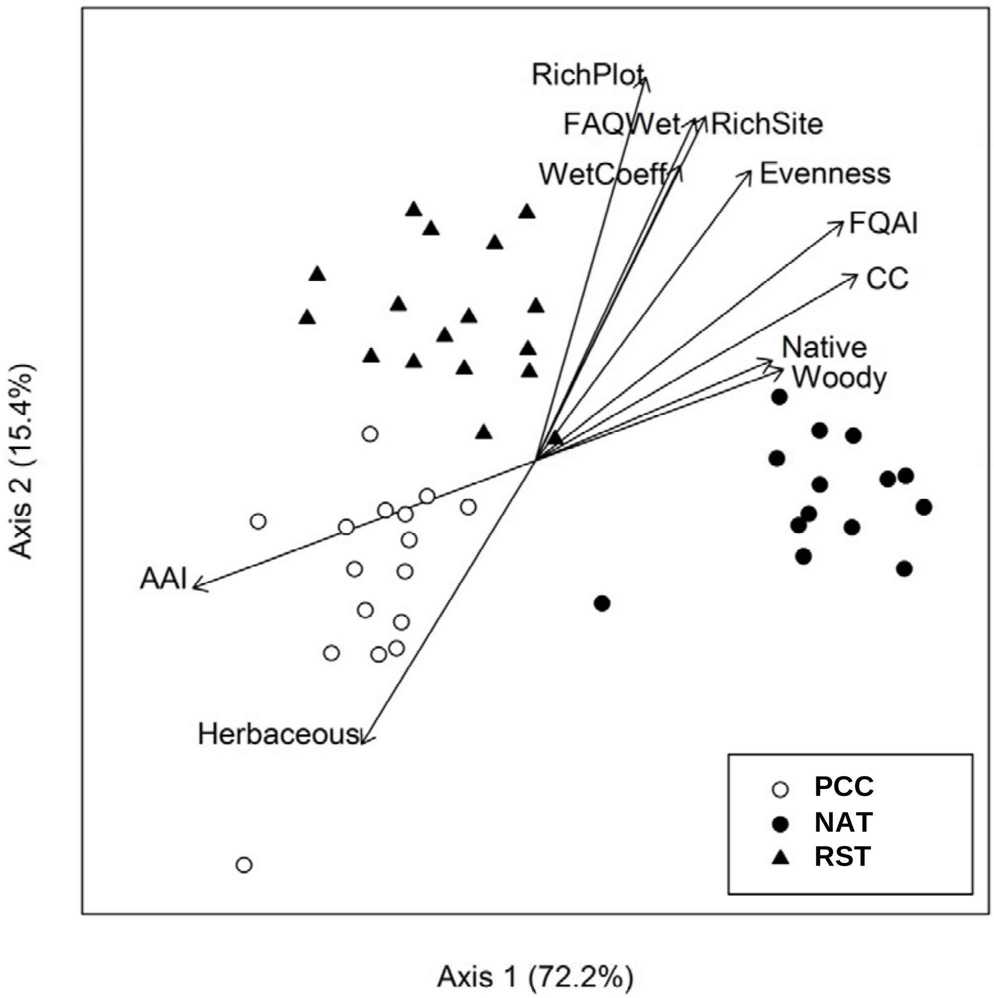

Figure 3: Results of non-metric multidimensional scaling (NMS) ordination of plant communities for natural (NAT), restored (RST), and prior converted cropland (PCC) sites. Plant communities of the three site types differed significantly (MRPP, A = 0.35, p <0.0001). Direction and length of vectors reflect the relationship of plant communities to community and disturbance metrics (AAI = Anthropogenic Activity Index; CC = Coefficient of Conservatism; Evenness = Shannon index of species evenness; FAQI = Floristic Quality Assessment Index; FAQWet = Floristic Assessment Quotient for Wetlands; Herbaceous = absolute cover of herbaceous plant (%); Native = Proportion of species that were native (%); RichPlot = number of species per 100-m^2 plot; RichSite = number of species in the entire wetland site; WetCoeff = Wetness Coefficient; Woody = absolute cover of woody plants (%; adapted from Yepsen et al. 2014).

7. Amphibian Community Biodiversity and Quality

Key Findings

-Eighty-one percent and 24 % of anurans and salamander species, respectively, known to occur in this region were found to inhabit various MIAR study sites.

-Total species and mean number of species based on all life history stages encountered was similar between restored and natural wetlands, but both wetland types contained twice the number of species detected at prior converted cropland sites. Total and mean number of species based only on larval occurrence showed comparable patterns.

-Restored and natural wetlands had approximately equal proportions of habitat generalists and specialists, but community similarity was relatively low. This indicates that USDA wetland restoration practices support amphibian biodiversity in the MIAR.

-Tree canopy closure was negatively correlated with larval species richness ($p = 0.058$).

Recommended Practices: In the MIAR, relatively high amphibian biodiversity can be preserved by protecting natural wetlands, and maintaining or increasing the number of restored wetlands. Restored wetlands with shallow basins and gently-sloping topographies that have open to low canopy cover, support aquatic vegetation, have no fish, and hold water from late-winter to early summer provide optimal habitats for amphibians in the MIAR.

***Primary Chapter Source*:** Mitchell, J. Amphibian use of restored wetlands in agricultural landscapes in the Mid-Atlantic Region, USA. *Journal of Fish and Wildlife Management*. (In Review)

Introduction

In addition to supporting cultural services related to recreation and aesthetics, amphibians are key sources of energy exchange between wetlands and their surrounding uplands. Therefore amphibians also support the provision of food, and other services directly related to energy exchange (Gibbons 2003; Davic and Welsh 2004; Gibbons et al. 2006; Faulkner et al. 2011; Hocking and Babbitt 2014)). Larval amphibians are herbivores, predators, and competitors that regulate the diversity and abundance of species at all trophic levels, including that of primary producers (Dickman 1968; Seale 1980; Petranka and Kennedy 1999; Register and Whiles 2006; Arribas et al. 2014). Keystone species, such as the Eastern Newt (*Notophthalmus viridescens*), regulate species diversity and abundance at lower trophic levels (Morin 1981; Fauth and Resetarits 1991; Fauth 1999). Adult frogs and salamanders are predators of many species and prey to many others. Unfortunately the role of amphibians in supporting the provision of ecosystem services has been diminished because of the dramatic loss of wetlands in the United States since initial colonization by Europeans (Tiner 1984; Dahl 1990; Noss et al. 1995; Piha et al. 2007). Negative associations often exist between agriculture and amphibian use of breeding wetlands (Babbitt et al. 2005; Anderson and Arruda 2006; Faulkner et al. 2011). Although 26 species of anurans and 17 species of salamanders occur in the MIAR (Conant and Collins 1998; Beane et al. 2010), agricultural lands are considered suitable habitat for only 42 % of these species, and this is only when breeding habitat is present (Mitchell et al. 2006). According to the Natural Resources Conservation Service (NRCS) Conservation Practice Standard, the purpose of wetland restoration is, in part, to restore wetland habitat and diversity to “a close approximation” of the pre-disturbance condition. The objective of this study component was to determine to what degree this goal was achieved relative to amphibians at MIAR wetland restoration study sites.

Methods

Wetlands were visited during the spring (April-May) and summer (June-July) of 2010 and 2011. Larval data were collected at all sites that were inundated. Sampling was conducted during the day. Focusing on aquatic larval communities provided a longer window of detection time, because the larvae are present and accessible during the daylight hours over several weeks to months. One restored wetland and two natural wetlands did not support standing water throughout the study and could not be sampled. Half of the prior converted croplands had no standing water during the entire study and were not sampled. The other half had one or more drainage ditches that served as larval sampling sites. These ditches varied from about 0.75 m to > 2 m deep, and held water for variable lengths of time. Five of these ditched sites were in the Delmarva and three were in southeastern Virginia and northeastern North Carolina.

Aquatic larval communities were assessed using samples collected with each 1 m sweep of an aquatic dip net. Sweeps were 5-10 m apart and wetland size determined the number of sweeps. Typically 10 to 20 sweeps were executed per site. The sum of the number of individuals for each species divided by number of sweeps provided an estimate of relative abundance for each wetland sampled. In addition to dipnet assessments, visual observations were made of eggs, juveniles, and adults during site visits, and males were also identified through species-specific vocalizations. Total species richness was defined as the sum of all species detected by these methods for both years. Larval occurrence provided evidence of species that used the wetlands for breeding. Relative abundance of larvae was based on average number of individuals per dipnet sweep. This estimate was based on the largest value obtained from each annual sampling season to reduce effects of annual weather variation. Percent canopy cover was visually estimated in 10 % increments for each wetland, but did not include surrounding land cover. Percent emergent aquatic vegetation was also visually estimated in 10 % increments for the total wetland area during spring sampling.

Total, mean total, larval, and mean larval species richness, and larval relative abundance were evaluated for the three wetland habitats in the MIAR. Northern (Maryland and Delaware) and southern (Virginia and North Carolina) areas were analyzed and compared separately due to substantial differences in climate, soil type, topography, and land cover. SYSTAT 11 was used for statistical tests following Zar (2009). Linear regression was used to compare relationships of species richness with environmental variables. Analysis of variance was used to compare species richness samples among the three habitat types. Nonparametric statistics (chi square) were used when assumptions of normality were not met.

Results

Twenty-one species of amphibians (17 anurans, 4 salamanders) were encountered at MIAR sites (Table 1), including fifteen species (13 anurans, 2 salamanders) in the northern sites and fifteen species (12 anurans, 3 salamanders) in the southern sites. Five frog species (i.e., Carpenter Frog, Green Treefrog, New Jersey Chorus Frog, Pickerel Frog, and Wood Frog) and one salamander species (Spotted Salamander) were only found in northern wetlands and four frogs (i.e., Pine Woods Treefrog, Southern Cricket Frog, Southern Toad, and Squirrel Treefrog) and two salamanders (Mabee's Salamander and Eastern Newt) were only found at southern wetlands (Table 1). Green Frogs exhibited the highest relative larval abundance at restored wetlands ($\bar{x}$ = 33.6/dipnet sweep), Southern Leopard Frogs (16.2/dipnet sweep) at natural sites, and Southern Toads (24.3/dipnet sweep) at prior converted cropland sites.

Total number of species recorded in all restored sites combined (16) was similar to the number detected in natural sites (15), whereas total number of species detected at prior converted cropland sites (7) was considerably less. For the entire study area, mean number of species detected at restored sites was greater than for natural sites, and prior converted cropland sites demonstrated lower values (Figure 2), but the difference between sites was not significant (p = 0.117; F = 2.272). Average number of species in restored and natural sites was similar, and both were higher than that for prior converted cropland sites.

Total number of species among the three wetland habitat types also varied within northern and southern regions, with restored sites exhibiting recovery of richness to near or even above the level of natural sites (Figure 3). For northern sites, differences in total species richness among the three wetland types were marginally significant (X^2 = 5.31, p = 0.07). In the south total species richness in restored and natural sites was significantly higher than species richness in prior converted cropland sites (Figure 3; X^2 = 6.08, p = 0.048). Mean number of species encountered in northern restored wetlands was higher than the mean for natural sites, and both were higher than the mean for prior converted cropland sites, but the difference was not significant (F = 2.283, p = 0.125). In the south, mean number of species that used restored sites ($\bar{x}$ = 3.63$\pm$1.84) was slightly lower than the mean for natural sites ($\bar{x}$ = 3.75$\pm$4.19), but both were higher than the mean for prior converted cropland sites ($\bar{x}$ = 2.33$\pm$2.87), although the differences were not statistically significant (F = 1.482, p = 0.278).

Mean number of amphibian species based solely on larval samples in MIAR restored sites was higher than the mean for natural sites, which was slightly higher than prior converted cropland site values, but differences were not significant (F = 0.030, p = 0.970; Figure 4). Mean number of larval species was not significantly different among the three habitat types in the north

(F = 0.359, P = 0.704) or south (F = 1.540, p = 0.279; Figure 5). Mean larval relative abundance in restored wetlands was higher than in natural wetlands and prior converted croplands in both 2010 and 2011 (Figure 6), however, differences were marginally significant at best (values for 2010 and 2011, respectively; F = 0.511-3.162, p = 0.070-0.624). None of the comparisons of the annual samples for northern and southern restored sites (t = -0.864- 0.910, p = 0.337-0.427), natural sites (t = 0.319-0.493, p = 0.652-0.762), or prior converted cropland sites (t = -0.529-1.481, p = 0.201-0.235) were significantly different. Relationships of total species richness (F = 0.886, p = 0.362) and larval species richness (F = 0.919, p = 0.354) with restored wetland age were not significant.

Total numbers of species in open and partial canopy natural sites was 3-7 and 4-8, respectively. One natural wetland with a full canopy supported three species of amphibians and one had none. A similar pattern occurred with amphibian larvae (open 3-4 species, partial 1-5, and full canopy 0-2). Relationship of larval species richness with percent forest canopy cover was marginally significant (P = 0.058). Number of larval species was not significantly related to amount of emergent vegetation (R^2 = 0.0972; P = 0.1937).

Discussion

A total of 43 species of amphibians (26 anurans and 17 salamanders) occur in the MIAR (Conant and Collins 1998; Dodd 2013), of which 23 anurans and 5 salamanders use depressional wetlands for reproduction. Eighty-one percent and 24 % of the expected number of anurans and salamanders, respectively, were found at study site wetlands. Distribution of species throughout the MIAR depends on species-specific geographic ranges (Bishop 1941; Conant and Collins 1998; Petranka 1998; Beane et al. 2010). Comparisons of species detected by call surveys to the presence of tadpoles in nearby wetlands demonstrated that not all species of anurans can be assumed to breed in the wetland under study, despite males vocalizing nearby (Mazanti 2000; Barlow 2006). Detection of anuran tadpoles and salamander larvae in wetlands is the only way to ensure accuracy when assessing species use of wetlands for breeding.

Restored and natural wetlands supported more species than documented in prior converted croplands with ditches containing water. Higher or equal species richness in restored sites compared to natural sites has been demonstrated in 89 % of published studies of amphibian use of created and restored wetlands worldwide (Brown et al. 2012). Although restored wetlands generally had levels of species richness and abundance similar to natural wetlands and considerably higher than prior converted croplands, these differences were often not statistically significant. This is likely due in large part to the considerable variation within site types and between years. This high level of variability is not surprising. The wide range of variation in number of species among wetlands detected in this study is consistent with results of other studies of amphibians, especially anurans (e.g., Rubbo and Kiescker 2005; Church 2008; Walls et al. 2013). Variation and turnover in species richness among wetlands is characteristic of amphibian populations in eastern North America (e.g., Petranka et. al. 2003a; Church 2008).

Agricultural practices, including wetland restoration implementation approaches and ditching, likely affected the number of species detected. Amphibian habitat provided by restored sites in North Carolina and other states were markedly different. Two out of three of the restored North Carolina sites contained areas of saturated soils and large, ditch-like open water areas that contained no larvae. Depressional restorations in other states were more likely to have larger areas of shallow water capable of supporting emergent vegetation, which has been found to be positively correlated with amphibian species richness (Brown et al. 2012). Low water pH may also have contributed to the lack of amphibian larvae at North Carolina sites. A pH below 6.3 has been found to severely limit anuran sperm modality (Feda and Taylor 1992), and organic acids, which are associated with Histosols, are known to reduce the pH of adjacent water. Ducey et al. (2015) found pH at the natural and restored MIAR sites to be well below the threshold of 6.3 (i.e., ~4.5). It should be noted that restoration sites were selected by state NRCS staff to be representative of restorations within the Coastal Plain Physiographic Province of their state. Although ditches originally served to drain wetlands and convert these areas to cropland, they now provide amphibian breeding sites. However, the communities supported by these ditches differ from those found in natural and restored wetlands.

Restored wetlands supported as many or more larvae than natural wetlands; however a number of factors influence larval abundance, and it can be challenging to properly account for these diverse, dynamic and often inter-related factors. These factors include number of breeding adults and aquatic predators, larval competition, nutrient levels, disease, and agrochemicals (Semlitsch 2000). Microdistribution patterns of amphibians in ephemerally inundated wetlands are determined, in part, by the occurrence of forest canopies (Werner and Glennemeier 1999; Skelly et al. 2002; Van Buskirk 2005). The importance of forest canopy closure was clear at our study sites. Life histories of several species with short larval periods (e.g., toads and chorus frogs [see species accounts in Lannoo 2005]) require open canopies that allow abundant sunlight, which hastens larval development though elevated temperatures. Number of species in natural wetlands with open or partial canopies were similar to numbers in restored wetlands. Natural wetlands with full canopies that shaded the wetland throughout the day supported the fewest species. Number of larval species was not significicntly related to the presence of emergent vegetation. However, emergent aquatic vegetation is an important microhabitat for several species (e.g., Spotted Salamanders and Chorus frogs) because egg masses need to be attached to underwater stems and because vegetation offers places for small larvae to hide from predators (see species accounts in Lannoo 2005). However, the large amount of variation in larval occurrence in this study prevented the detection of significant differences.

Implementation and management of wetland restoration projects should incorporate knowledge of amphibians that occur in the area and their habitat requirements (Brown et al. 2012). Species egg-laying site requirements and length of larval development time should be used to guide basin construction. In general, restored wetlands with shallow basins and gently-sloping topographies that have open to sparse canopy cover, support aquatic vegetation, have no fish, and hold water from late-winter to early summer provide optimal habitats for amphibians in the mid-Atlantic region (Shulse et al. 2010; Brown et al. 2012; this study). For amphibians, management plans should include guidelines for preventing establishment of predatory fish and invasive hardwoods once the restoration process has been completed. Although not always possible, long-term monitoring and habitat management are critical for ensuring regular amphibian use (Balcombe et al. 2005; Vasconcelos and Calhoun 2006; Brown et al. 2012). These and additional guidelines for management of native amphibian species are described in several publications and documents pertinent to the Mid-Atlantic, such as Semlitsch (2000), Bailey et al. (2006), and Mitchell et al. (2006).

Conclusions and Implications

Restored wetlands in the MIAR were found to support levels of diversity and community quality similar to natural wetlands. Relative to reference data collected at natural sites, these restorations can be considered to have successfully recreated an amphibian community that is a "close approximation" of the pre-disturbance condition. Furthermore, it is likely that these restorations enhance ecosystem services relative to amphibian community energy dynamics, as well as cultural services related to recreation and aesthetics. However, restored and natural community similarity was low. Therefore, amphibian diversity in this region can be best preserved if restored wetlands are maintained or increased and remaining natural depressional wetlands are protected from loss and degradation (Julian et al. 2013).

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Tables

Scientific Name	Common Name	North RST	North NAT	North PCC	South RST	South NAT	South PCC	Combined RST	Combined NAT	Combined PCC
Acris crepitans (G)	N Cricket Frog	4	---	1	2	1	---	6	1	1
Acris gryllus (G)	S Cricket Frog	---	---	---	1	---		---	---	---
Anaxyrus fowleri (G)	Fowler's Toad	10	3	3	3	---	---	13	3	3
Anaxrus terrestris (G)	Southern Toad	---	---	---	3	---	2	3	---	2
Hyla chrysoscelis (G)	Cope's Gray Treefrog	14	6	1	2	3	---	16	9	1
Hyla cinerea (G)	Green Treefrog	2	---	---	---	---	---	2	---	---
Hyla femoralis (S)	Pine Woods Treefrog	---	---	---	---	2	---	---	2	---
Hyla squirella (S)	Squirrel Treefrog	---	---	---	1	---	---	1	---	---
Lithobates catesbeianus (G)	American Bullfrog	6	4	4	2	1	2	8	5	6
Lithobates clamitans (G)	Green Frog	9	4	5	2	1	---	11	5	5
Lithobates palustris (G)	Pickerel Frog	2	---	---	---	---	---	2	---	---
Lithobates Sphenocephalus (G)	Southern Leopard Frog	10	6	3	6	3	2	16	9	5
Lithobates sylvaticus (S)	Wood Frog	---	1	---	---	---	---	---	1	---
Lithobates virgatipes (S)	Carpenter Frog	---	3	---	---	---	---	---	3	---
Pseudacris crucifer (G)	Spring Peeper	6	4	2	2	2	---	8	6	
Pseudacris kalmi (S)	New Jersey Chorus Frog	2	1	---	---	---	---	2	1	---
Scaphiopus holbrookii (S)	Eastern Spadefoot	1	1	---	1	---	---	2	1	---
Ambystoma maculatum (S)	Spotted Salamander	---	1	---	---	---	---	---	1	---
Ambystoma mabeei (G)	Mabee's Salamander	---	---	---	---	1	---	---	1	---
Ambystoma opacum (S)	Marbled Salamander	---	1	---	1	1	---	---	2	---
Notopthalmus viridescens (S)	Eastern Newt	---	---	---	1	---	---	1	---	---
Total Number of Species	**21**	**11**	**12**	**7**	**13**	**9**	**3**	**17**	**15**	**7**

Table 1: Occurrence of amphibian species in restored (RST), natural (NAT), and prior converted cropland (PCC) sites (North = DE/MD and South = NC/VA). Cells contain number of wetlands (column) in which species were detected. Abbreviations: G = habitat generalist, S = habitat specialist (Petranka 1998; Dodd 2013) (adapted from Mitchell [in review]).

Figures

Figure 1: Pictures of the three MIAR wetland types. Upper left: a restored wetland with inundation present (spring); upper right: the same restored wetland without inundation present (summer); lower right: a water-filled ditch draining a prior converted cropland; lower left: a natural wetland in the spring (Mitchell [in review]).

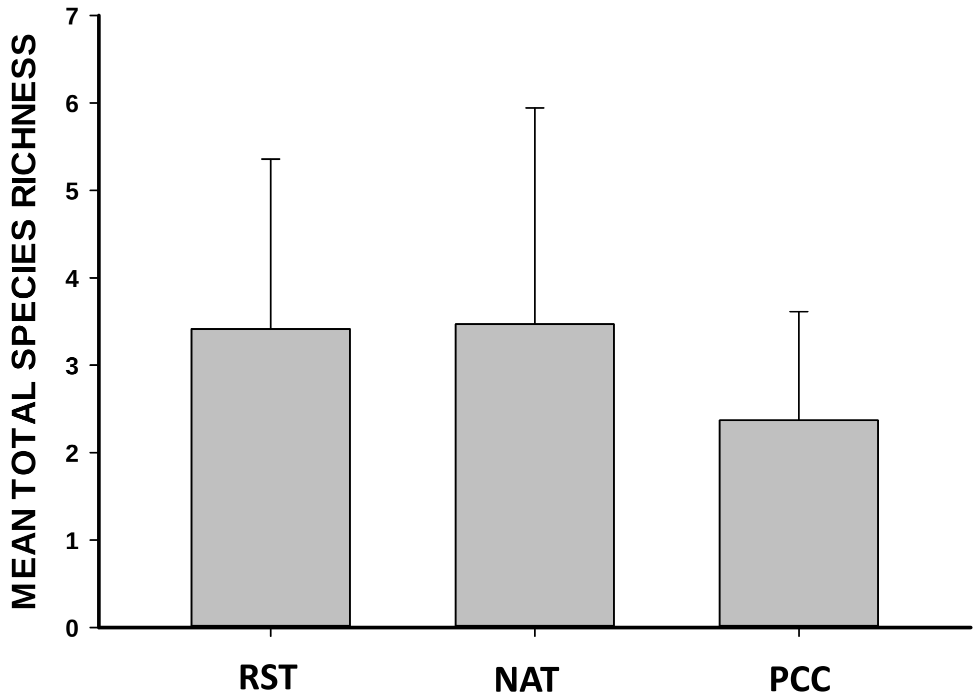

Figure 2: Mean total amphibian species richness in the three site types (restored [RST], natural [NAT], and prior converted cropland [PCC]). Bars are means and extensions are one standard deviation (adapted from Mitchell [in review]).

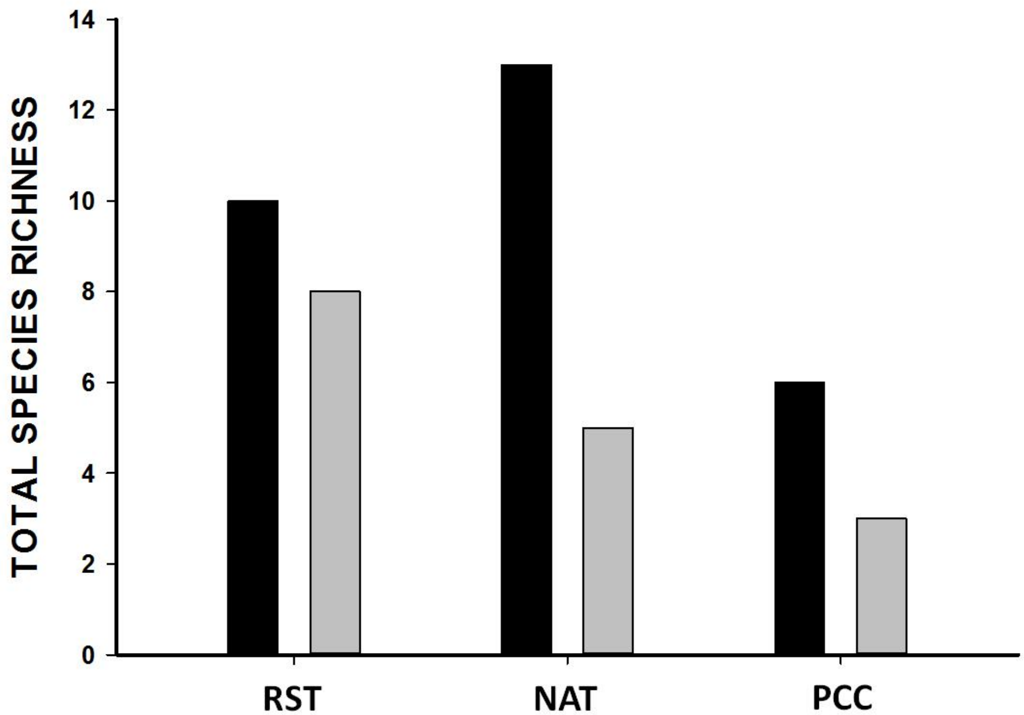

Figure 3: Total amphibian species richness per site type (restored [RST], natural [NAT], and prior converted cropland [PCC]) in the northern region (Delmarva; black bars) and the southern region (southeastern VA and northeastern NC; gray bars; adapted from Mitchell [in review]).

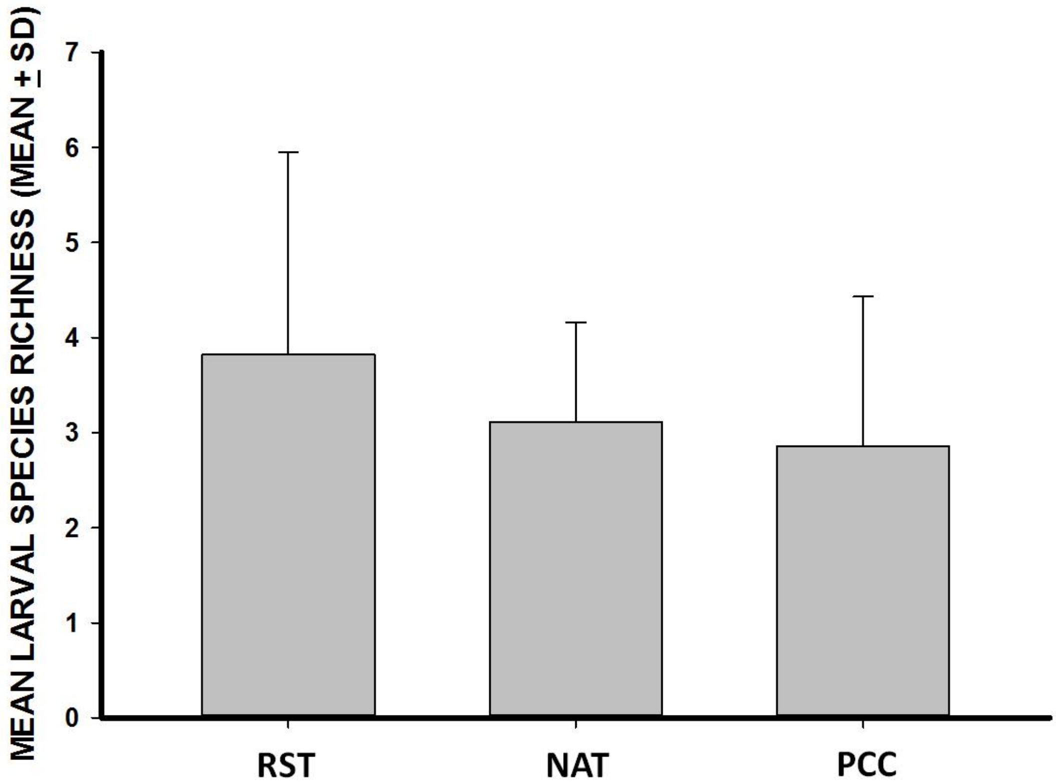

Figure 4: Mean larval amphibian species richness at the three site types (restored [RST], natural [NAT], and prior converted cropland [PCC]). Bars are means and extensions are one standard deviation (adapted from Mitchell [in review]).

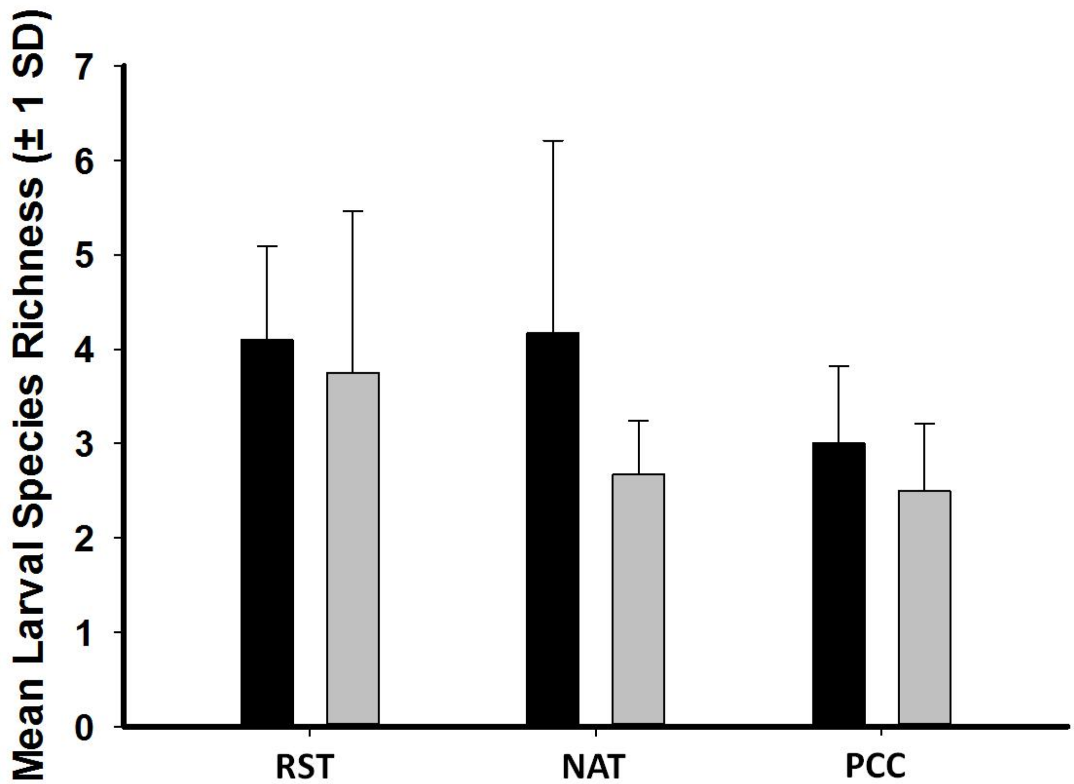

Figure 5: Mean larval amphibian species richness in the northern region (Delmarva; black bars) and the southern region (southeastern VA and northeastern NC; gray bars) per site type (restored [RST], natural [NAT], and prior converted cropland [PCC]). Bars are means and extensions are one standard deviation (adapted from Mitchell [in review]).

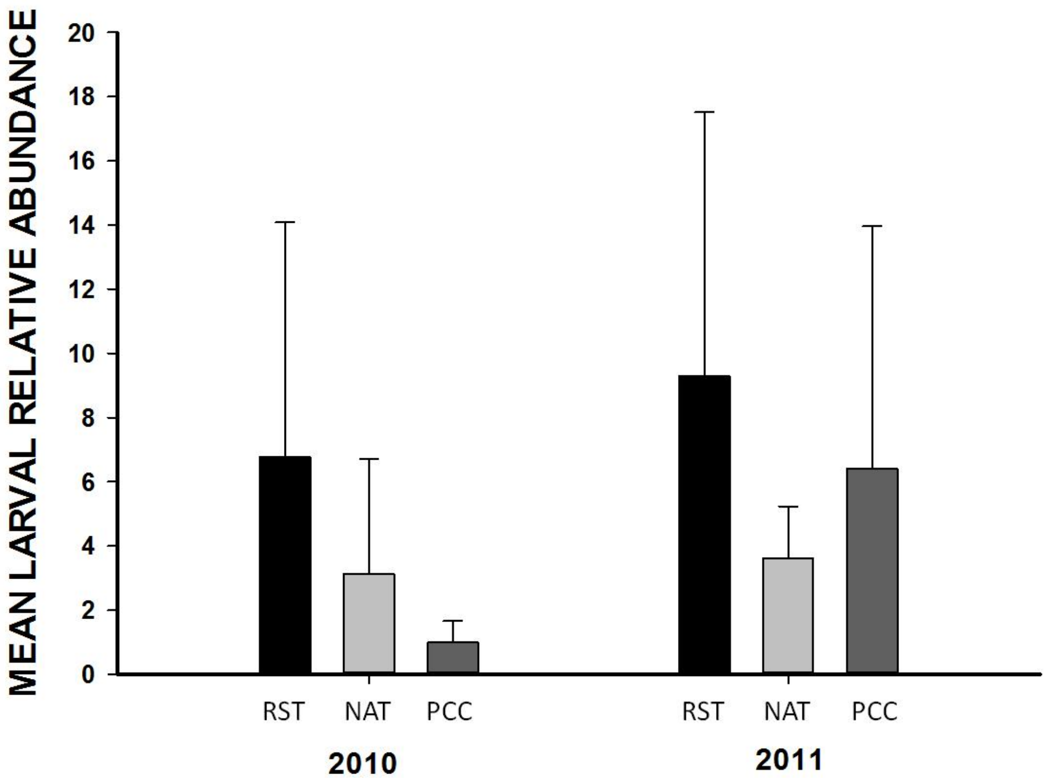

Figure 6: Mean relative abundance of amphibian larvae in restored (RST), natural (NAT), and prior converted cropland (PCC) sites during 2010 and 2011. Bars are means and extensions are one standard deviation (adapted from Mitchell [in review]).

8. Landscape Scale Nitrate Mitigation Potential

Key Findings
-The potential effectiveness of depressional wetlands for mitigating nitrogen transport varies substantially over different parts of the MIAR area. Sixteen commonly available geospatial metrics were found to be effective for classifying the MIAR area into eight wetland landscape regions (WLRs) that predict the likelihood of depressional wetlands and their effectiveness at mitigating nitrogen transport from upland source areas to surface waters.

-Three WLRs, covering 32 % of the MIAR area, were found to have a high potential for containing depressional wetlands that could mitigate nitrogen transport from nonpoint sources. These WLRs were concentrated in eastern North Carolina and southeast Virginia, as well as along the coast and central portion of the Delmarva Peninsula and in New Jersey.

-The lowest potential for the occurrence of depressional wetlands was found in two WLRs, covering 37 % of the study area, and located on the western edge of the Coastal Plain with rolling hills and relatively high slope and relief. Most of the western shore of the Chesapeake Bay and the Sand Hills of North Carolina are included in these WLRs.

-The three remaining WLRs had relatively flat catchments with moderate to low potential for nitrogen mitigation. These WLRs, found in 31 % of the area, typically had primarily well-drained to moderately well-drained sandy soils and few poorly drained upland areas where depressional wetlands would occur. They are located in large areas of the Delmarva Peninsula and North Carolina, and in southern New Jersey.

***Recommended Practices*:** The model predicts areas where depressional wetlands are likely to be more or less effective at mitigating nitrogen transport from nonpoint sources to streams, and therefore could be used to prioritize areas for wetland preservation or restoration or to guide the extrapolation of field-scale data by regression or process-based models. The WLRs also provide a framework for predicting regional effects of depressional wetlands on nitrogen yields that could be very useful for regional water quality modeling.

***Primary Chapter Source*:** Ator, S.W., Denver, J.M., LaMotte, A.E., and Sekellick, A.J. 2013. A regional classification of the effectiveness of depressional wetlands at mitigating nitrogen transport to surface waters in the northern Atlantic Coastal Plain: U.S. Geological Survey Scientific Investigations Report 2012–5266, 23 p.

Introduction

Agriculture is common in the MIAR Coastal Plain and nitrate contamination is typical in many parts of the surficial aquifer. Groundwater discharge provides the majority of flow in Coastal Plain streams, and ecological degradation due, in part, to excessive nitrogen is commonly found in streams and adjacent coastal estuaries, including Chesapeake Bay and Pamlico Sound. In the MIAR, groundwater can provide from 40 to 95 % of stream flow (Sinnott and Cushing 1978; Leahy and Martin 1993) and up to 70 % of the nitrogen load in streams (Domagalski et al. 2008; Ator and Denver 2012). Non-riparian wetlands, common in parts of the Coastal Plain, can be effective landscape sinks for nitrogen from nonpoint sources, such as fertilized croplands. Nitrate concentrations in poorly drained areas of the MIAR, such as those where depressional wetlands and wetland flats are found, are generally lower than in well drained areas (Ator 2008). The effectiveness of such wetlands at removing nitrogen, however, varies spatially with variability in hydrogeologic, pedologic, and other landscape conditions. Targeting wetland restoration to promote nitrogen removal over large regions requires understanding the spatial variability in landscape conditions that control local nitrogen fate and transport, including the flux of surface and groundwater and the occurrence of reducing conditions associated with wetlands that may promote nitrogen loss through denitrification. Geographic models using selected landscape metrics to identify relevant features coupled with conceptual models of nitrogen transport can be used to predict spatial variability in the effectiveness of wetlands at mitigating nitrogen transport from nonpoint sources to surface waters. The objective of this study component was to develop a geographic model delineating *Wetland Landscape Regions* (WLRs) or areas of the Mid-Atlantic Coastal Plain within which topographic, soil, and hydrogeologic conditions, and therefore the likely effectiveness of non-riparian wetlands at mitigating nitrogen movement from nonpoint sources to local streams are similar.

Methods

Study Area

The MIAR, as identified for this study component, covers approximately 114,000 square kilometers (km^2) in North Carolina, Virginia, Maryland, Delaware, New Jersey, and the District of Columbia. Due to a relatively humid climate and generally flat topography, the groundwater table is generally within a few meters of the land surface. Land cover is predominately forested, though areas of intensive agriculture and urbanization are found throughout the region. The study area included areas of 12-digit hydrologic units (HUC12s; as accessed from the Natural Resources Conservation Service [NRCS] Geospatial Gateway) that intersect MIAR boundaries. Catchments defined by the US Geological Survey (USGS) National Hydrography Dataset (NHD)Plus, a digital medium resolution (1:100,000-scale) stream hydrography network, were used as the landscape units for the model. Small catchments, with areas less than 0.5 km^2, and catchments that include more than 50 % open water were omitted. Tidal areas, assumed to be catchments with more than 75 % of land area below 2 m in elevation, were not included in the model. Catchments with missing Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO) soils data, which were generally heavily developed urban areas, were also excluded from consideration in the model. The resultant model consisted of 33,799 individual catchments in the MIAR with a median area of 2.2 km^2.

Landscape Metrics

Sixteen landscape metrics (Table 1) were selected for inclusion in the model, consisting of soils data and topographic measures that are particularly significant to the fate and transport of nitrogen in wetlands. These metrics were selected to characterize the likelihood of depressional wetlands and reducing conditions in each model catchment, as well as to determine their position in the local hydrologic flowpath. The likelihood of wetlands being present was represented by topographic variables indicative of landscape flatness and by soil variables representative of soil texture and drainage. Topographic metrics were computed from 30 m digital elevation models (DEMs) available from the USGS National Elevation Dataset website. Soil metrics were computed from SSURGO data in conformance with NRCS methods for aggregation based on area-depth weighted averaging (Natural Resources Conservation Service 2011). Topographic wetness index (TWI), a function of slope and upstream contributing area (Beven and Kirk 1979), was computed from NHDPlus attributes. High TWI values represent low slope (flat) areas with a large upslope contributing area. Also important to nitrate fate and transport in the surficial aquifer in the MIAR is the orientation of local groundwater flowpaths relative to wetlands and nitrogen sources. Metrics quantifying the percentage of each catchment that is flat (<1 % slope) upland or flat lowland were included to represent the likelihood of depressional wetlands in each catchment occurring hydrologically up-gradient or down-gradient, respectively, of agriculture or other potential nitrogen sources. Uplands (and lowlands) within each catchment are defined by areas above (or below) the catchment midpoint elevation or the mean of the maximum and minimum elevation.

Principal Components Analysis

Statistical techniques were used to delineate WLRs based on values of the 16 landscape metrics (Table 1) in each of the 33,799 individual catchments. Principle components analysis (PCA) was used to identify correlations and reduce redundancy among the landscape metrics. The first three principal components (PCs) were used for subsequent analysis and together explained 77 % of variability in the 16 landscape metrics. The first PC (PC1) explains nearly half (43 %) of the variability in the input data and represents a measure of the likelihood of non-riparian wetlands. Input landscape metrics with strong positive loading on PC1 include measures of overall catchment flatness and soil indicators of poor drainage (e.g., hydric soils). Landscape metrics with strong negative loadings for PC1 include slope and relief. PC2 represents a distinction between catchments with flat lowlands (positive values) and those with flat uplands (negative values), which can be interpreted as an indicator of where wetlands in each catchment may occur relative to nitrogen sources along local topographic gradients and presumably hydrologic flow paths. PC3 is an indicator of soil texture. Catchments with negative values of PC3 contain relatively sandy conductive soils, while those with positive values include less well-drained soils with higher available water capacity (Table 1).

Cluster Analysis

Catchments in the study area were classified into groups with similar PC scores using a cluster analysis. These groups contained similar values for the 3 PCs, and thus the 16 landscape characteristics deemed important to identify the potential for nitrogen mitigation by non-riparian wetlands. Clusters of catchments with similar PC values were identified using Ward's minimum variance method, a hierarchical agglomerative clustering technique (SAS Institute, Inc. 2009). The number of clusters selected for analysis is subjective. More clusters better represent the data, but are more difficult to interpret and may not be useful. The first eight clusters were selected and contain 68 % of the variability of the 3 PCs. The cluster analysis provided a classification of the MIAR based on the selected landscape metrics. There was significant variation in the landscape metrics as well as land covers between the eight clusters and these metrics were used to describe the landscape in each cluster and to develop hypotheses about nitrogen fate and transport, including likely losses in non-riparian wetlands.

Wetland Landscape Regions

The eight clusters (Table 2; Figure 1) can be interpreted as Wetland Landscape Regions (WLRs) with similar within region landscape metrics contributing to analogous effects of non-riparian wetlands on nitrogen fate and transport.

Very Flat Poorly Drained Uplands (VFPDU) are flat low-lying areas near the coast and are found in approximately 6,500 km^2 of mostly North Carolina and southern Virginia (Figure 1 and Table 2). These catchments have significantly less relief and a larger percentage of flat land than other WLRs. Catchments in this subregion are generally broad, flat plains likely including artificial ditches and incised streams. Much of the area can be considered flat uplands and the currently cultivated land in this WLR was likely wetland flats. Soils are fine textured, contain a significant amount of organic matter, have a high available water capacity (AWC), and are generally more poorly drained than any other WLR. Despite the flat topography and soil conditions, wetlands are less common than in other WLRs. This is likely due to artificial drainage to support agriculture. Natural or restored depressional wetlands in the VFPDU would have a high potential for nitrogen transport mitigation from nonpoint sources to local streams. The extremely flat topography and organic soils with high AWC and reducing conditions would allow water to move slowly through the landscape and provide ample prospects for denitrification. Ditching and artificial drainage would lower the local water table and induce greater flow from surrounding areas. This may increase the effectiveness of wetlands as nitrogen sinks, although it may also increase nitrogen transport from wetlands by reducing the length of groundwater flowpaths and the time available for denitrification to occur (Phillips and Donnelly, 2003). Artificial drainage may also oxidize upper soils, thus decreasing the area of likely denitrification.

Flat Poorly Drained Lowlands (FPDL) cover roughly 11,700 km^2 of the MIAR (Figure 1 and Table 2). These catchments are found interspersed with VFPDU catchments. FPDL catchments are generally extremely flat and likely formed in Carolina Bays and other landscape depressions. Flat land is typically located within catchment lowlands. Soils have a greater AWC and are more commonly hydric than in any other WLR except for the VFPDU. Soils are relatively fine grained and high in organic matter. Wetlands are more common than in other WLRs and non-riparian wetlands may have the highest potential for mitigating nitrogen from agriculture sources. Soil and topographic metrics in this WLR indicate slow movement of water and conditions ideal for denitrification. The larger amount of flat lowlands in this subregion (as opposed to uplands in the VFPDU) indicates that non-riparian wetlands may occur downgradient of nitrogen sources and be more likely to mitigate nitrogen transport.

Flat Sandy Lowlands (FSL) cover 16,700 km^2 of the MIAR and are often found in broad areas of the Delmarva Peninsula and southern New Jersey, as well as southern North Carolina (Figure 1 and Table 2). These catchments are characterized as moderately flat lowlands with sandy soils, a low mean AWC, and relatively high organic matter. Wetlands are found at a higher rate only in the FPDL and would likely have a high potential to mitigate nitrogen transport from upgradient agricultural sources. Ditching is common on agricultural areas and irrigation is generally not required. Ditching may increase nutrient transport in this subregion by decreasing the residence time of water in the organic-rich anoxic sediments that promote denitrification (Phillips and Donnelly, 2003).

Flat Mixed (FM) WLR covers 10,400 km^2 and is dispersed widely throughout the MIAR (Figure 1 and Table 2). These catchments are flat areas with similar amounts of local uplands and lowlands. Soils are moderately high in hydric properties. Agriculture is found at a higher rate in the FM than the VFPDU, FPDL, and FSL areas. Wetlands cover < 25 % of the area. Mixed and moderate landscape and soil conditions contribute to a variable potential for nitrogen mitigation from depressional wetlands. Wetlands will likely be effective as nitrogen sinks, but the movement of nitrogen will depend on varied local hydrologic conditions.

Flat Mixed Uplands (FMU) and Flat Sandy Uplands (FSU) WLRs cover 25,000 km^2 of the MIAR, particularly in North Carolina and the Delmarva Peninsula (Figure 1 and Table 2). The two WLRs have similar topographic settings, including low relief, incised streams, and relatively high amount of flat uplands. Soils in FSU catchments are generally sandier, have a lower AWC, contain higher amounts of organic matter, and are less commonly hydric than soils in FMU catchments. These WLRs contain the highest percent of agricultural land cover. Wetlands are found mostly in riparian zones and occupy 10 to 15 % of each WLR. There is a relatively low potential for nitrogen mitigation from depressional wetlands in the these WLRs. Wetlands are uncommon and would be relatively easy to drain for agricultural cultivation. The upland hydrologic position would indicate that relatively little drainage to wetlands is received from agricultural lands.

Rolling Hills with Mixed Soils (RMS) and Rolling Hills with Sandy Soils (RSS) WLRs are predominantly found on the western shore of the Chesapeake Bay (Figure 1 and Table 2). These WLRs have greater relief, less flat land, less hydric soils, and less soil organic matter than other WLRs. RSS soils are relatively sandy with a low mean AWC, while RMS soils are less sandy with a higher AWC. Both wetlands and agriculture are relatively uncommon in either WLR.

Applications and limitations

The geographic model comprised of the eight WLRs should be useful for numerous regional applications, but model use must be considered in light of inherent assumptions and limitations. The WLRs identify areas where nitrate is likely to be intercepted and reduced due to the presence of depressional wetlands. This can aid in targeting wetland conservation practices to maximize improvement to water quality. Targeting wetland restoration efforts towards lowland wetlands that have been artificially drained may result in enhanced water quality improvements through greater interception and transformation of agricultural nitrate. The geographic model may also be useful for improving models of nitrogen fate and transport. Such models often require spatial delineation of factors and classification of the potential for depressional wetlands to mitigate nitrogen. However, several limitations must be taken into consideration when using the model. Model predictions may be useful at broader regional scales, but predictions for small areas or individual catchments may be unreliable. Additionally, the model is limited by the availability and resolution of input data. Additional datasets, such as those depicting the thickness of the surficial aquifer, could significantly improve understanding of the potential for nitrate interception and model results. Currently these additional datasets are not available across the entire study area.

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Tables

Catchment Landscape Metric	(percent of variance)	PC1 (43)	PC2 (19)	PC3 (16)	Communality Estimate[1]
Percent flat land		**0.927**			0.890
Percent hydric soils		**0.890**			0.807
Mean soil hydric rating		**0.883**			0.855
Mean soil hydrologic group, undrained		0.661		0.559	0.761
Mean soil drainage class		**0.891**			0.869
Mean topographic wetness index		**0.866**			0.815
Mean soil percent organic matter		0.410			0.248
Mean watershed slope, in percent		**-0.857**			0.806
Watershed relief, in meters		**-0.817**			0.733
Percent lowland			**0.947**		0.962
Percent flat lowland		0.615	0.690		0.862
Percent upland			**-0.947**		0.962
Percent flat upland		0.660	-0.627		0.907
Mean soil available water capacity				**0.734**	0.710
Mean soil percent sand				**-0.798**	0.748
Mean soil saturated hydraulic conductivity				**-0.702**	0.673

[1]Communality estimates show the proportion of variance in each metric that is explained by the combination of PC1, PC2, and PC3 (SAS Institute Inc., 2009).

Table 1: Results of principal components analysis on 16 landscape metrics defined for 33,799 individual catchments in the MIAR [PC, principal component; loadings with absolute value greater than 0.7 are **bold**; loadings with absolute value less than 0.4 are omitted] (Ator et al. 2013).

Wetlands landscape	Area (km^2) [%]	Summary of setting and predicted depressional wetland landscape	Relative likelihood of nitrogen mitigation
Very Flat Poorly Drained Uplands (VFPDU)	(6,500) [6]	Mostly flat and poorly drained upland-dominated watersheds. Soils are mostly fine-grained and hydric with relatively high available water capacity. High potential for nitrogen losses in prior-converted cropland. Agricultural and timber harvesting common in drained former wetlands.	High
Flat Poorly Drained Lowlands (FPDL)	(11,700) [11]	Mostly flat and poorly drained lowland-dominated watersheds, including relatively large Carolina Bays to the south. Soils have mixed permeability but relatively high available water capacity. Wetlands are abundant and may be drained, but drainage would be difficult due to flat lowland topography. High potential for nitrogen losses in prior-converted cropland. Agriculture common in drained former wetlands.	High
Flat Sandy Lowlands (FSL)	(16,700) [15]	Typically flat lowland-dominated watersheds with mixed drainage. Soils are typically sandy and permeable with low available water capacity. Wetlands are common and often forested or drained for agriculture. Prior-converted cropland is likely to intercept nitrogen that is direcly applied, and remaining wetlands interspersed with agriculture may intercept nitrogen from adjacent uplands.	High
Flat Mixed (FM)	(10,400) [9]	Typically flat watersheds with mixed drainage, geomorphology, and soils. Mixed potential for wetlands and mitigating agricultural nitrogen.	Moderate
Flat Mixed Uplands (FMU)	(17,700) [16]	Typically flat upland-dominated watersheds. Mixed soils, potential for wetlands, and nitrogen losses. Wetlands may be drained for agriculture, although artificial drainage is likely less common than in the FSL, and irrigation may be required. Some potential for nitrogen losses in prior-converted cropland; however, adjacent agricultural areas contributing to wetlands may be very localized.	Low
Flat Sandy Uplands (FSU)	(7,300) [7]	Typically flat upland-dominated watersheds. Soils are typically sandy and permeable with relatively low available water capacity. Limited wetlands interspersed with well-drained areas. Artifical drainage is likely minimal and irrigation is often required for crop production. Nitrogen losses in prior-converted cropland may be limited by short hydroperiod, and adjacent agricultural areas contributing to wetlands may be very localized.	Low
Rolling Hills with Mixed Soils (RMS)	(32,900) [30]	Watersheds with relatively high slope and relief and good drainage. Geomorphology and soil conditions are mixed. Wetlands limited to riparian zones.	N/A
Rolling Hills with Sandy Soils (RSS)	(7,800) [7]	Watersheds with mostly relatively high slope and relief and relatively good drainage. Geomorphology is mixed. Soils are generally sandy and permeable with low available water capacity. Wetlands limited to riparian zone.	N/A

Table 2: Summary of depressional wetland landscape regions within the MIAR (Ator et al. 2013).

Figures

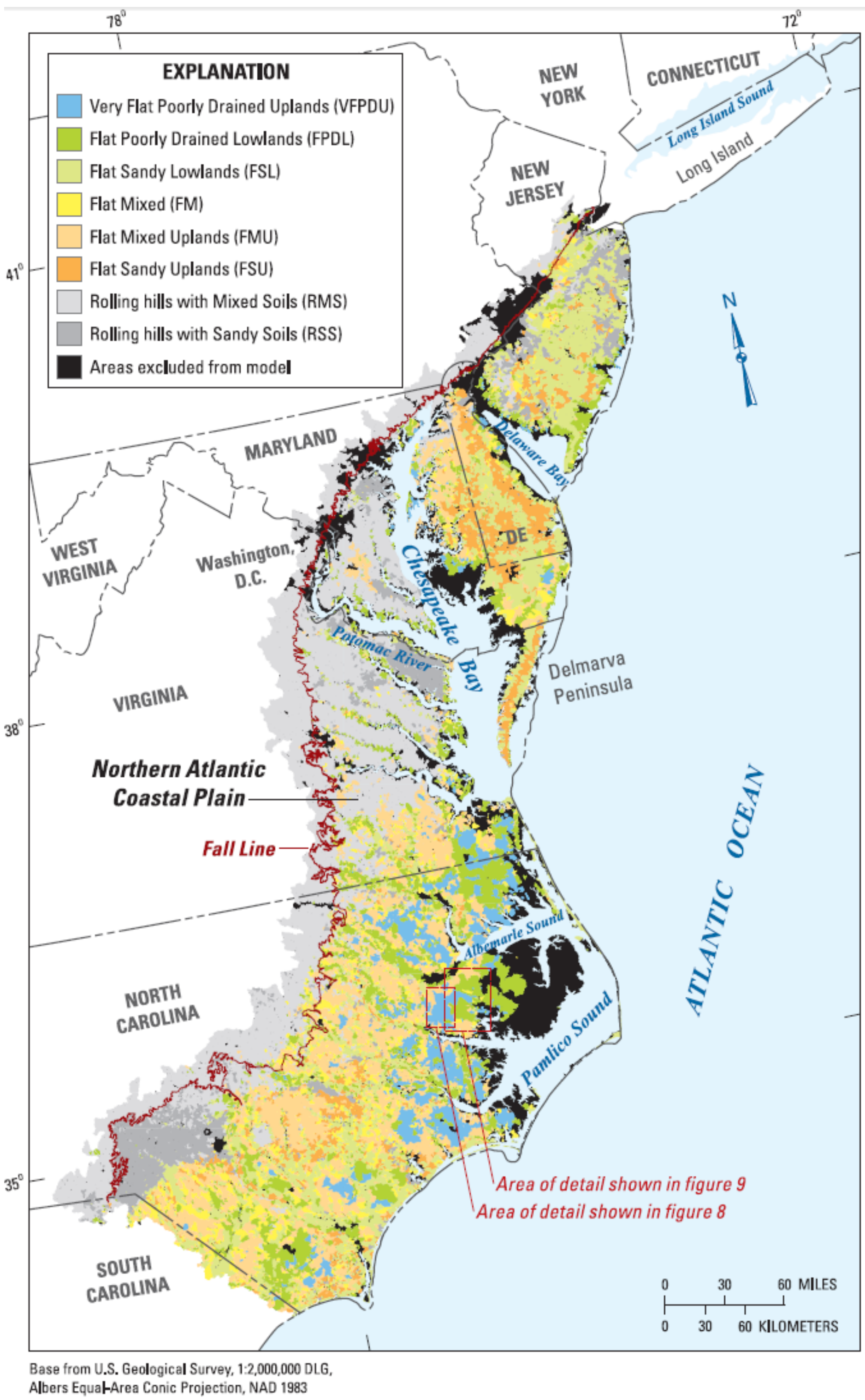

Figure 1: Wetland landscape regions within the MIAR (Ator et al. 2013).

9. Use of Remotely Sensed Data to Characterize Wetlands

Remotely sensed data constitute a powerful toolset for the characterization and monitoring of wetlands across the human alteration gradient, and the landscapes within which they function, as they are influenced by various change vectors (e.g., land use and climate). Remote observation of wetlands is particularly useful because they are often difficult to access on the ground, and on-site mapping at the landscape scale is usually cost prohibitive, especially at fine time scales. Wetland maps and the geospatial methods used to inventory and assess wetlands have evolved dramatically through the decades. Indeed imagery and techniques have changed substantially since the United States initiated the first comprehensive national wetland mapping effort, the U.S. Fish and Wildlife Service National Wetland Inventory, in the mid-1970s. These initial efforts were focused on the use of aerial photographs, which provide valuable information, but also present significant restrictions to the wetland mapping process (Lang et al. 2015a). While aerial photography has a proven operational wetland mapping track record, one type of imagery cannot be expected to map all wetland types accurately, nor can one type of remotely sensed data detect all environmental parameters that are indicative of wetland character or function. Instead, different types of remotely sensed data are sensitive to different biophysical parameters. These components (e.g., soil moisture, presence or absence of standing water, biomass, vegetation height, and plant cellular structure) can then be synthesized to gain a better understanding of wetland structure, function, and the ecosystem services that wetlands provide.

Newer remote sensing technologies and techniques have great potential to provide insights into not only wetland location and character, but also wetland function and ecosystem service provision. For example, synthetic aperture radar (SAR) data can be used to reveal subtle differences in inundation and soil moisture over time (i.e., hydroperiod) and space that are normally difficult to detect, but control the extent and functioning of wetlands. Hydroperiod is the most important abiotic factor controlling wetland function, and largely determines the level at which most wetland ecosystem services are provided. Light detection and ranging (LiDAR) data can be used to produce highly accurate maps of inundation below a vegetative canopy and predict hydroperiod and hydrologic fluxes based on modeling the movement of water across the landscape. These fluxes control not only the water budget of individual wetlands but also determine, in large part, their potential for providing pollutant regulation services (e.g., location of wetland relative to pollutant sources and flow pathways). Although multispectral data (e.g., Landsat) cannot be used to indicate hydroperiod as accurately as SAR and LiDAR data in many wetland environments, its historic record and temporal and spatial coverage cannot be matched. The entire Landsat historic record is now freely available for much of the globe, including the United States, and can be used to track the long-term effects of weather and land cover change on wetlands at the regional, national, or even global scale.

We are now entering a period which promises rapid, substantial improvements to the quality and availability of remotely sensed data and products. Landsat-8, launched in 2013, provides continued collection of Landsat data, initiated in 1972. The European Space Agency (ESA) recently launched the first satellite in the Sentinel-2 constellation, which in combination with Landsat-8 has the potential to collect moderate resolution multispectral data over the Earth's surface every few days. ESA is also collecting Sentinel-1 SAR data, a C-band Interferometric SAR constellation which will collect moderate spatial resolution (5 x 20 m) data across the globe approximately once a week. This and other SAR instruments are capable of generating valuable data regardless of cloud cover, and detecting hydroperiod below a plant canopy. Interferometric SAR sensors can also determine the elevation of land, water, and plant canopies. The joint National Aeronautics and Space Administration (NASA) - Indian Space Research Organization (ISRO) SAR (NISAR) mission is scheduled to be launched in 2020 and will provide 3 - 12 m L/S-band interferometric SAR data with a ~12 day repeat time. In the past SAR data were available only upon request and were not collected automatically, like Landsat. Sentinel-1 and NISAR will automatically collect data across the globe and be freely available. This will lead to a substantial increase in data use, and in turn, the development of refined techniques and integration into additional environmental implementation and management programs. Other substantial improvements to the availability of high quality, wetland relevant satellite data include NASA's Surface Water and Ocean Topography mission, Canada's Radarsat constellation, Japan's Advanced Land Observing Satellite-2, and NASA's Soil Moisture Active Passive sensor. Many more exist, but are not listed here for the sake of brevity. Finally, the availability of high quality elevation data, primarily LiDAR data, is rapidly increasing, in part through the efforts of the National Digital Elevation Program, which includes representatives from NRCS (Snyder and Lang 2012). In addition to the increased availability of high quality remotely sensed data, the operational use of remotely sensed data to map, monitor, and model wetlands will be greatly supported by the renewed emphasis of NASA, USGS, and other federal agencies on the production of map products, instead of just remotely sensed data. These products (e.g., USGS Dynamic Surface Water Extent) can be more readily incorporated into existing decision support systems. The advancements outlined above indicate the maturation of multiple remote sensing technologies that are highly valuable for the direct determination of wetland parameters, which are key to determining: 1) the effects of wetland restoration; 2) the effectiveness of wetland restoration relative to existing wetland ecosystems; 3) the effect of land use and climate change on the functioning of wetlands, including restored wetlands, at a landscape scale; and 4) mechanisms to enhance the provision of wetland ecosystem services through wetland conservation practices. Please see Lang et al. 2015a and Lang et al. 2015b for additional information regarding the utility of various types of remotely sensed imagery for wetland mapping and monitoring, and recent and future developments in the field of wetland mapping.

But these remote sensing technologies cannot be used by themselves; instead they are most powerful when combined with field data, such as the data described in earlier portions of this report, and used to guide statistical and process-based models. It is the synergistic use of these tools and techniques that will best position natural resource managers to respond to the challenges of climate and land use change, shifting budgets, and increased demand for accountability, while enhancing environmental outcomes and ecosystem service provision through wetland restoration. For this reason, in addition to assessing the effects and effectiveness of wetland restoration in the field, the MIAR CEAP-Wetland study was tasked with assessing the potential of and developing new techniques to use cutting-edge remote sensing technologies to support CEAP-Wetland goals. To date this assessment has covered a range of remotely sensed data types and applications, all focused on monitoring and/or characterizing wetland functions and services, and improving the capacity to do so. Chapter B5 describes the use of a combination of field measurements, LiDAR derived digital elevation models (DEMs), and land cover maps to estimate natural wetland and prior converted cropland surface water volume storage across the Delmarva Peninsula, and places these estimates in context with surface water volume storage gains offered by wetland restoration. Chapter B8 describes the creation of a regional map of wetland denitrification potential that can be used to target restoration of prior converted croplands that can best enhance water quality improvements through greater interception and transformation of agricultural nitrate. This effort took advantage of a range of geospatial datasets, including those created through the use of aerial photographs (i.e., Soil Survey Geographic Database) and DEMs. Two MIAR CEAP Science Notes (i.e., Lang and McCarty 2014 and Lang et al. 2015) are included below which detail the utility of LiDAR data for: 1) mapping portions of the landscape, including wetlands, that experience different patterns of inundation; 2) enhancing understanding of landscape controls (e.g., slope, contributing area, relief, and connectivity) on these patterns; and 3) identifying naturally occurring ranges in wetland relief, area, and shape that can be used to enhance wetland restoration implementation. Landscape scale estimates of naturally occurring wetland morphologies and landscape positions can be used to best support the stated goal of the USDA NRCS Wetland Restoration (657) Practice Standard, which is to "restore wetland function, value, habitat, diversity and capacity to a close approximation of the pre-disturbance conditions." Furthermore, the ability to locate and restore former depressional wetlands with sufficient relief and catchment area to volume ratio to support wetland hydrology without the need for excavation could be advantageous for the enhanced regulation of climate via carbon sequestration (Chapter B1).

Current MIAR efforts are focused, in part, on the creation of multi-temporal hydroperiod maps for the Delmarva Peninsula (Figure 1). Hydroperiod maps are being produced using yearly maps of subpixel inundation percent produced using Landsat 30 m multispectral satellite data and LiDAR return intensity data (Huang et al. 2014). Mapping of hydroperiod, a key wetland functional driver, is critical for modeling wetland functions across the human alteration gradient. Hydroperiod has been challenging to map in the MIAR due to the presence of plant canopies and relatively rapid changes in wetland hydropattern. However, recent advancements in remote sensing data availability and techniques, supported by CEAP-Wetland findings (e.g., Lang and McCarty 2009), have allowed the generation of a hydroperiod map for the majority of the Delmarva Peninsula. This map will be used to support a variety of complimentary modeling activities, including: 1) parameterization, calibration and validation of the Soil and Water Assessment Tool (SWAT) and the Agricultural Policy / Environmental eXtender (APEX); 2) ILM development within InVEST; and 3) creation of wetland functional classifications. Although hydroperiod maps are not currently available across the United States, MIAR team members are working with colleagues at the USGS and the University of Maryland to develop these types of maps elsewhere. Hydroperiod maps are currently being used to help parameterize a wetland module within SWAT, and additional opportunities are being explored with APEX and SWAT model developers. Wetland functional driver – structure relationships are being characterized through the integration of hydroperiod maps and field data. An ongoing MIAR CEAP-Wetland study found that maps of hydroperiod could be used to identify Delmarva bays that contained more than three times the amount of soil organic carbon stocks as other Delmarva bays, a substantial functional difference. These functional driver – structure relationships will be critical to the potential development of an National Resources Inventory (NRI) wetland functional classification, and are also vital to the development of the Integrated Landscape Model (ILM) (i.e., InVEST) within the MIAR.

This chapter highlights the significant and growing importance of remote sensing for wetland assessment and adaptive management, both through the extrapolation of field scale information and greater understanding of landscape scale processes that would have been costly and difficult to ascertain on the ground. The strong potential of remotely sensed data and products for improving conservation practice assessment and implementation paired with the rapid increase in the availability of such datasets highlights the critical role that this information source will play in future conservation efforts. The functions that occur within individual or groups of wetlands are unique to their placement on the landscape (Bedford 1999; Simenstad et al. 2006). Therefore the landscape perspective that remote sensing provides is critical to ensuring the optimum provision of wetland ecosystem services through restoration - at the individual wetland and watershed scale. The use of remotely sensed data can also provide temporal context, either directly though the use of historical images or indirectly through comparisons between wetland types (e.g., human alteration gradient). The importance of this historic perspective was emphasized by Bedford (1999): "By definition [wetland restoration] seeks to replace what has been lost. By definition then, it should be undertaken with knowledge of what has been lost." Remotely sensed images aid our understanding of wetlands within a wider landscape setting and help to ensure wetland preservation via an increased appreciation of the services that wetlands provide and more informed management practices. Doing so directly supports the vision for CEAP-Wetlands championed by Diane Eckles, the former national CEAP-Wetland component leader, who articulated a guiding concept for CEAP-Wetlands, including the development of landscape-scale conservation planning tools and implementation of an ecosystem services monitoring and reporting framework (Eckles 2011).

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Figures

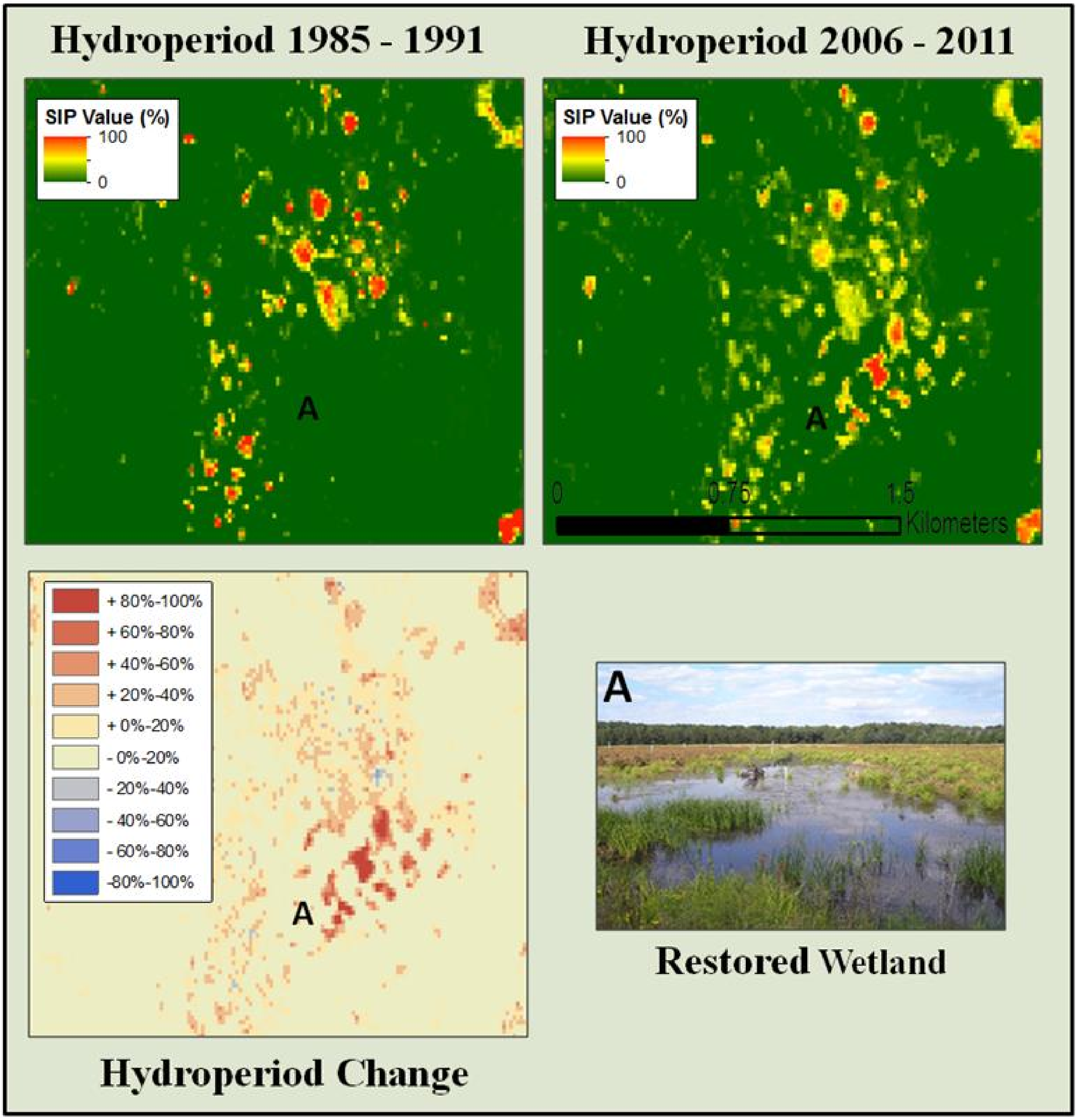


Figure 1: Hydroperiod through time within natural wetlands and prior converted croplands / restored wetlands. Hydroperiod maps were produced by summarizing Landsat based subpixel inundation percent (SIP) over five years. SIP was estimated during yearly maximum expression of surface water (early spring). Change in hydroperiod between 1985-1991 and 2006-2011 can be seen to the lower left. A large wetland restoration, including multiple wetland cells, can be seen at location A. Note the increase in inundation likelihood visible on the hydroperiod change map. A photograph is also provided for location A (subsequent to restoration). Maps are currently available for the Delmarva Peninsula.

United States Department of Agriculture

Light Detection and Ranging (LiDAR) for Improved Mapping of Wetland Resources and Assessment of Wetland Conservation Practices

Natural Resources Conservation Service
Conservation Effects Assessment Project

CEAP Science Note, September 2014

Summary Findings

LiDAR elevation data can be used to map the potential, static distribution of current and historic wetlands and key wetland functional drivers based on physical controls on water distribution. LiDAR intensity data can be used to map actual, dynamic variations in wetland inundation extent which can provide additional insights concerning key functional drivers.

LiDAR intensity data significantly improved the mapping of inundation below the forest canopy compared with using aerial photography. The accuracy of the LiDAR intensity based wetland inundation map was 97% versus 70% for the aerial photography based map, or about 30% more accurate.

Relief relative to a local elevation maximum provided a strong indicator of inundation dynamics (i.e., hydroperiod), but was less useful for mapping wetland boundaries. Combining local relief and an Enhanced Topographic Wetness Index produced a map that was well suited for mapping wetland extent and hydroperiod. Wetlands mapped using aerial photographs or LiDAR-derived digital elevation models (DEMs) contained a similar amount of inundated area, but the LiDAR-derived maps contained fewer errors of omission. For this reason, it was concluded that DEM based topographic metrics produced enhanced inundation maps relative to aerial photography derived maps.

When using LiDAR derived DEMs our results support the use of more distributed flow routing algorithms over algorithms that force greater flow convergence for the mapping of palustrine wetlands in areas with low topographic gradients. Accounting for water outflow as well as inflow is key to developing robust indicators of water accumulation potential.

A concerted effort is ongoing by NRCS and other federal agencies to hasten the collection of high quality LiDAR data throughout the entire United States and facilitate enhanced analyses of natural resources and ecosystems.

Remotely sensed data have long been an important tool for the assessment of land condition and the effects and effectiveness of land management. The USDA has an extensive history of remotely sensed data use, which has largely focused on aerial photography. Although the inherent benefits of aerial photography and established operational data processing structures merit the continued use of this data stream, newer types of remotely sensed data, including Light Detection and Ranging (LiDAR), have been shown to provide robust, synergistic information on conservation practices when used in conjunction with aerial photography. This includes, but is not limited to, the use of LiDAR data to improve the mapping and characterization of wetlands.

Although U.S. wetlands are currently mapped using aerial photography, these maps are often out of date and errors can be substantial (Stolt and Baker 1995; Kudray and Gale 2000), especially in difficult-to-map areas, which include wetlands with intermittent hydrology and forested wetlands. The Natural Resources Conservation Service (NRCS) is one of several Federal agencies that have expressed the importance of LiDAR data for improved wetland mapping and characterization (Snyder and Lang 2012).

Until recently, the spatial resolution of commonly available digital topographic data for the United States (vertical accuracies of ~3.3–32.8 ft [1–10 m]) was insufficient to map many geomorphologic features, including most wetlands. However, LiDAR-derived digital elevation models (DEMs) provide superior vertical accuracy (~3.9–5.9 in [~ 10–15 cm] and horizontal resolution (~39.4– 78.7 in [~100–200 cm] [Coren and Sterzai 2006]), allowing the enhanced mapping and characterization of existing, former, and restored wetlands, which can improve the implementation of wetland conservation practices. The use of LiDAR data can be especially vital in areas with low topographic variation, particularly when applied to mapping or monitoring wetlands that have previously been difficult to detect, such as forested wetlands.

This Conservation Effects Assessment Project (CEAP) Science Note briefly introduces discrete point return LiDAR technology, the most readily available type of LiDAR; describes multiple studies that have demonstrated the benefits of this technology for improved wetland mapping and characterization; and discusses the implications of these studies and others for improved wetland mapping and assessment of wetland conservation practices.

Light Detection and Ranging (LiDAR) Technology

LiDAR sensors provide detailed information on the elevation of the Earth's surface and objects on the landscape, such as vegetation and human -made structures. LiDAR sensors collect data through the use of an onboard laser system, which sends and receives laser energy. LiDAR sensors send frequent (hundreds of thousands per second) short pulses of laser energy, and a portion of that energy is reflected back to the sensor where it is recorded. Most LiDAR sensors used for land-based remote sensing operate in the near-infrared region of the electro-magnetic spectrum (commonly in the 0.90 to 1.55 μm wavelength range; Lemmens 2007), with 1.06 μm (near-infrared) being a commonly used laser wavelength (Goodwin et al. 2006). LiDAR data can be used to calculate precise x, y, z locations through the use of a highly accurate onboard Global Positioning and Inertial Navigation System and by calculating the distance to an object by recording the amount of time it takes for a pulse, or a portion of that pulse, to travel from the sensor to the target and back (Goodwin et al. 2006). LiDAR x, y, z points can be used to make DEMs through the interpolation of LiDAR point returns. The resolution of the resultant DEM is based largely upon the original density of LiDAR returns (point density) and user requirements. If only points originating from the Earth's surface, as opposed to points originating from above the Earth's surface (e.g., trees, grass, and buildings) are used for the interpolation, then the resultant image is called a bare earth DEM, and it represents topography. While return time provides information on location, LiDAR intensity, or the strength of the returned LiDAR signal relative to the amount of energy transmitted by the sensor per laser pulse (Chust et al. 2008), provides information regarding the identity of target materials which the LiDAR signal reflects from before returning to the sensor.

Wetland Applications of LiDAR

LiDAR Intensity

LiDAR intensity data are well suited for the identification of inundation, and possibly saturation, due to the strong absorption of near-infrared energy (the energy detected by most terrestrial LiDAR sensors) by water. Information derived from LiDAR intensity is complementary to LiDAR-based information on x, y, z location, and each LiDAR point return contains both types of information. The association of individual points of LiDAR intensity with precise x, y, and z values allows the selection and display of LiDAR intensity originating from the Earth's surface exclusively, in this way reducing the impact of a plant canopy or other vertical structures on the ability to discriminate inundated versus non-inundated areas on the ground. In this way, LiDAR intensity data can be readily filtered to remove the influence of the canopy. On the other hand, aerial photography cannot be similarly filtered and will contain a mix of information from the plant canopy and the ground.

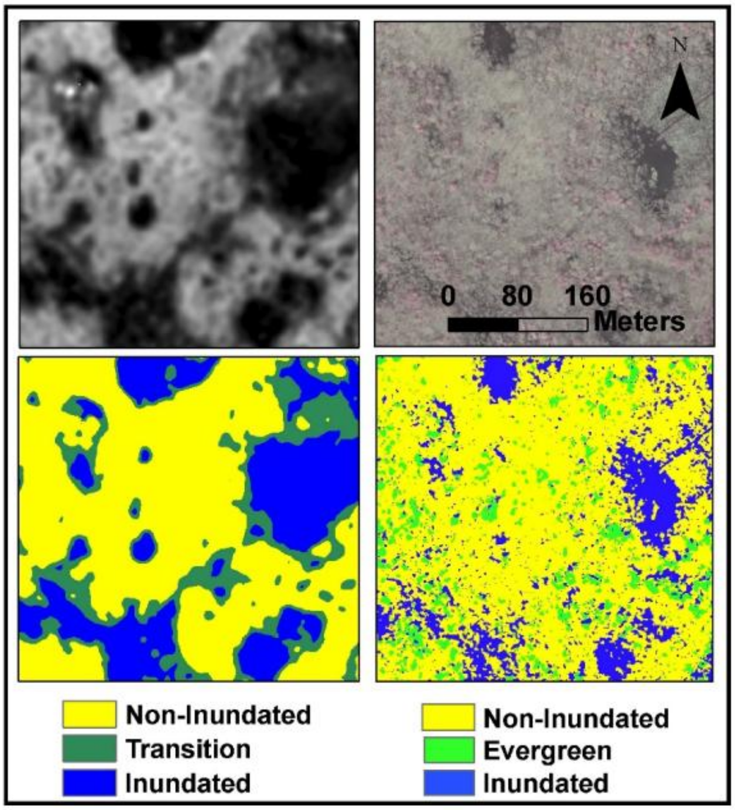
A study was conducted to determine the relative ability of LiDAR intensity and aerial photography to map inundation beneath the forest canopy in the Choptank River Watershed, an agricultural watershed on the Eastern Shore of Maryland (McCarty et al. 2008). Although inundation does not always equate with wetland status, data were collected during maximum hydrologic expression at the beginning of the growing season, March 27, 2007. Therefore, areas that were inundated during the study period were very likely to meet the hydrologic definition of a wetland and although areas that were not inundated during the study period could still meet this definition they were much less likely to do so. The mapping of forested wetlands is particularly important because these are the most common type of wetland in the United States and they are particularly difficult to map using existing technologies, such as aerial photography. This is especially true in areas of low topographic relief, such as the outer Coastal Plain of the Mid-Atlantic. Accurate maps of wetland extent and character are critical for a wide variety of natural resource management activities. For example, they can be used to assess the effects and effectiveness of forested wetland restoration and compare the level of ecosystem services provided by restored and less disturbed wetlands.

To meet the goal outlined above, LiDAR intensity data were collected using an Optech ALTM 3100 LiDAR sensor flown at 2,000 ft (~610 m) above the Earth's surface. Data were collected with a laser pulse frequency of 100,000 pulses of 1.06 μm wavelength energy per second at a scan angle of ± 20° using a scan frequency of 50 Hz and a 12-bit dynamic range. The resultant data had a vertical accuracy of ± 5.91 in (15 cm) and an average bare earth point density of ~0.23 pt ft^{-2} (2.5 pts m^{-2}). The sensor was coupled with a digital camera to capture coincident 4.72 in (12 cm) spatial resolution aerial photography in the near-infrared (0.72–0.92 μm), red (0.60–0.72 μm), and green (0.51–0.60 μm) bands (Lang and McCarty 2009).

The LiDAR intensity data were spatially filtered to reduce noise and a simple thresholding technique was used to create a map of inundation below the forest canopy. Prior to analysis, the aerial photograph was resampled to a spatial resolution of 1 m and an unsupervised isodata classification procedure was used to create a map of inundated and non-inundated forest using all bands of the digital image. The resultant inundation map was filtered to reduce error. The LiDAR intensity and aerial photography-based maps of inundation were validated with ground-based information on inundated and non-inundated areas collected using a highly accurate Trimble GeoXT global positioning system (GPS; Lang and McCarty 2009).

The study found that LiDAR intensity data significantly improved the mapping of inundation below the forest canopy relative to aerial photography (fig. 1). The LiDAR intensity-based inundation map was 97 percent versus 70 percent accurate, respectively or nearly 30 percent more accurate than the aerial photography-based map (Lang and McCarty 2009). Not unexpectedly, evergreen areas were found to influence the accuracy of both maps, although the impact appeared to be much greater on the aerial photography-based map. Tree canopy reflectance and shadow appeared to cause a large portion of the error contained within the aerial photography based-map. Since water is a strong absorber of visible and near-infrared energy, the expected low reflectance of

Figure 1. The original datasets (filtered intensity, top left, and aerial photography, top right) used to produce two different inundation maps (resultant map directly below parent dataset). Note that inundation patterns are more distinct in the LiDAR intensity image and resultant inundation map. Adapted from Lang and McCarty 2009.



water is easily confused with decreased reflectance in areas affected by shadow. Conversely, reflectance off of a tree canopy, even during the leaf-off period, is more similar to reflectance from non-inundated soils and organic debris (Lang and McCarty 2009). These influences are generally absent from or can be removed from LiDAR intensity data.

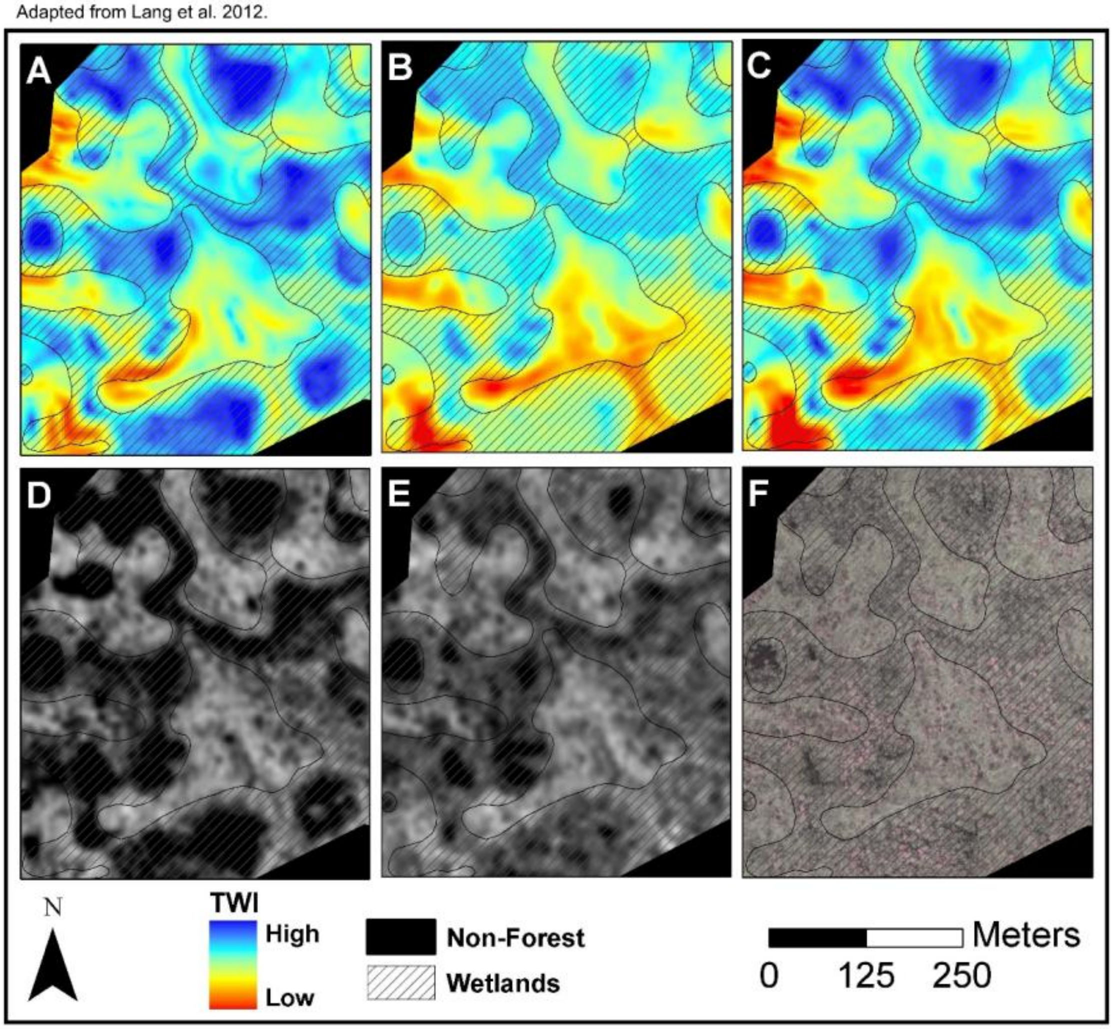
Although largely untapped, the potential of LiDAR intensity data to better understand fundamental ecosystem processes and improve land cover classification is strong. This was the first study to examine the ability of LiDAR intensity to map inundation below the forest canopy. A later study found that LiDAR intensity data have the potential to assist with the relative differentiation of deciduous forests with varying degrees of surface wetness and, therefore, wetland status within the coastal region of North Carolina (Newcomb and Lang 2011), supporting the conclusions drawn by Lang and McCarty (2009). Although there are inherent limitations of LiDAR intensity data, including the fact that the data are typically uncalibrated (i.e., standardized) between LiDAR collections and that they are sensitive to the angle at which the laser interacts with the Earth's surface, these weaknesses can be greatly reduced through the interpretation of LiDAR intensity data within one collection and the use of these data in areas of relatively low topographic variability, such as the Coastal Plain. Furthermore, intensity data are often included with LiDAR elevation data for low or no cost. Therefore, it makes sense to take advantage of this relatively untapped data stream when LiDAR intensity data are well suited for project needs. This statement is particularly relevant given the often limited availability of suitable imagery for wetland mapping and characterization.

LiDAR-Derived Topographic Metrics

DEMs can be used to predict the movement and distribution of water and thus relative wetness across the landscape. Whereas LiDAR intensity detects the presence of water, LiDAR based topographic metrics can predict the potential distribution of water accumulation across the landscape. Multiple types of topographic metrics can be produced using DEMs and used to infer relative wetness. These metrics relate to physical controls on water distribution. For example, the topographic wetness index is a commonly used topographic metric based on slope and contributing area and is expressed as $\ln(\alpha/\tan\beta)$, where α is the upslope contributing area per unit contour and $\tan\beta$ is the local topographic gradient (Beven et al. 1995). Although β has been calculated using a fairly consistent methodology, methods used to calculate α vary considerably based on the applied flow-routing algorithm (Lang et al. 2012). Numerous flow-routing algorithms are available, including the commonly used D8 (distribution of flow to one neighboring cell); the somewhat more distributed D∞ (distribution of flow to 1 or 2 neighboring cells); and FD8, which distributes flow to all neighboring pixels. These algorithms proportion flow according to slope with greater slope leading to increased allocations of water. The following section describes a study that investigated the ability of multiple LiDAR DEM-derived topographic metrics, including three topographic wetness indices computed using different flow routing algorithms, to map wetlands in the Choptank River Watershed on Maryland's Coastal Plain (Lang et al. 2012).

Topographic metrics were calculated using a DEM derived from LiDAR data that were collected when very little flooding was present within study area wetlands. It is critical to collect LiDAR data for topographic analysis when flooding is not present since flooding often leads to inaccurate and/or undependable elevation measurements. For this reason data were collected in December 2007 during a relatively dry period with very little wetland inundation on the landscape. The resultant LiDAR data were used to generate a 9.84 ft (3 m) gridded DEM which was subsequently filtered before applying multiple algorithms to produce five different topographic metrics (Lang et al. 2012). Topographic wetness indices were produced using the basic equation detailed above and the D8, D∞,

Figure 2: Topographic index products including the enhanced topographic wetness index (A), local terrain normalized relief (B), and the relief enhance topographic wetness index (C), LiDAR intensity during an average (D) and dry spring (E), and false color near-infrared aerial photograph (F; collected coincident to D) of a forested wetland complex. All images have been overlaid with a wetland map produced for the Maryland Department of Natural Resources. On the topographic index products, wetter areas are blue (more likely to be wetlands) while drier areas are red (less likely to be wetlands). Inundated areas are black on the LiDAR intensity images. Adapted from Lang et al. 2012.



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and FD8 flow-routing algorithms. A local terrain normalized relief (LTNR) map was created by subtracting a surface representing maximum elevation per 0.049 acre (200 m^2) from the original filtered 9.84 ft (3 m) DEM. An enhanced topographic wetness index (ETWI) was created by increasing FD8 based topographic wetness index values within depressions (i.e., pits or sinks). A Relief Enhanced TWI (RETWI) was created by adding the ETWI and LTNR metrics together after normalizing the metrics. The topographic metric-based wetland maps were compared with LiDAR intensity derived maps of inundation created to represent maximum yearly hydrologic expression during average weather (March 2007) and drought conditions (March 2009), and a wetland map produced by the Maryland Department of Natural Resources (MD DNR) (fig. 2)

The ability of the FD8 TWI to map inundation status, and therefore wetland status (see above), was superior to the D∞ and especially the D8 TWIs (Lang et al. 2012). The utility of the FD8 TWI was improved by increasing values within areas without a surface water outlet to create the ETWI. The outlet enhanced FD8 TWI (ETWI) performed well for wetland mapping but provided little information on hydroperiod. Local relief (LTNR) provided information on hydroperiod but was less capable of wetland mapping. Combining local relief and ETWI produced a map that was well suited for mapping wetland extent and hydroperiod. Wetlands mapped using aerial photographs and LiDAR-derived DEMs contained a similar amount of inundated area, but the LiDAR-derived maps contained fewer errors of omission.

Our results support the use of more distributed (FD8) flow routing algorithms over algorithms that encourage greater flow convergence (e.g., D8 and D∞) for the mapping of palustrine wetlands (Lang et al. 2012). This may be especially true in areas of low topographic relief. It is hypothesized that the ETWI map more completely represented the presence of surface water outlets from a given area to complement the input of surface water (i.e., specific catchment area). The ability of the local relief index (LTNR) to indicate temporal trends in flooding could support the use of this index to map hydroperiod and indicate critical zones associated with climate change. We hypothesize that LTNR and RETWI are dependent on two different physical drivers, surface expression of groundwater and lateral inflows and outflows, respectively (Lang et al. 2012). The metrics discussed above provide some degree of flexibility to best represent wetland distribution and boundaries within different study sites. Furthermore, topographic metrics illustrate gradual changes through space, which more accurately depict natural ecologic gradients, instead of the abrupt boundaries present on classified maps.

This study demonstrated that the predictive power and efficiency of wetland mapping efforts could be improved through the incorporation of LiDAR-derived DEMs (Lang et al. 2012). The use of LiDAR data will be especially vital in areas with low topographic variation or when applied to mapping wetlands that have previously been difficult to detect, such as forested wetlands. Optical (e.g., aerial photography) and LiDAR data are distinct remotely sensed datasets which offer unique benefits and limitations. The synergistic combination of these datasets has the potential to significantly improve not only the mapping of forested wetlands but also the mapping of historic wetlands (e.g., prior-converted croplands) within agricultural watersheds. These historic wetlands are critical agricultural management zones that can exert substantial control on crop productivity via nutrient processing (i.e., N and P) and water availability, especially during years of drought or flood.

Current and Future Availability of LiDAR Data and Specifications

Availability of LiDAR data has increased rapidly over the past 2 decades, but these data are not currently available for the entire United States. Although airborne LiDAR data are currently available for only about one-third of the conterminous United States, the spatial distribution of these data are advantageous for wetland mapping (Snyder and Lang 2012). LiDAR data happen to be available where wetlands are most common. A concerted effort is being made by NRCS and other Federal agencies to hasten the collection of high quality LiDAR data throughout the entire United States. The U.S. Geological Survey (USGS) recently conducted the National Enhanced Elevation Survey (NEEA) to assess the needs for, costs of, and best implementation scenarios for the collection of enhanced elevation data (Snyder and Lang 2012). As a result of the NEEA, the USGS has endorsed an implementation scenario focused on the collection of interferometric SAR data in Alaska and LiDAR data with a horizontal point spacing of 2.30 ft (0.70 m) and a vertical accuracy of 3.64 in (9.25 cm) throughout the rest of the United States (Snyder and Lang 2012). The NEEA concluded that there were no technical barriers or capacity issues that would prevent a national program, nor technical reasons to delay national program implementation (Snyder and Lang 2012). NRCS is currently working with USGS to develop a funding strategy and governance model to best assure the collection of the endorsed dataset.

The rapid evolution of LiDAR technology and growth in data availability and use led to a lag in developing LiDAR guidelines and, to some degree, applications. However, LiDAR guidelines were recently developed and are currently available to guide LiDAR collection and processing (e.g., http://pubs.usgs.gov/tm/11b4/). Continued application development is needed to fully realize the potential of LiDAR data for wetland mapping and assessment. This effort includes the development of optimal data collection specifications for different applications. LiDAR data should be collected to different specifications based on their intended application. For example, vegetation cover is known to reduce the spatial resolution and accuracy of bare earth DEMs. For that reason data are best collected for this purpose during the leaf-off period. Resolution can be further improved by collecting data at higher point densities. Fundamental research studies, such as those described in this document, have demonstrated the strong potential of LiDAR to support wetland assessment and management. Further advancements in LiDAR applications would greatly benefit from investigation of the suitability of developed techniques within an operational mapping and assessment framework. Perhaps most critical for wetland applications is consideration of ecosystem hydrologic state relative to the goal of the data collection. For example, obtaining detailed maps of actual and potential inundation extent from LiDAR requires contrasting hydrologic states and therefore careful planning of data acquisition within the hydrologic cycle.

Potential of LiDAR for Future Wetland Conservation and Management Efforts

The wetland science and management community has rapidly endorsed the use of LiDAR data for improved wetland mapping and characterization, which is likely attributable both to the considerable benefit of LiDAR and the poor suitability of older datasets for this application. Indeed, wetland-related applications were among the most commonly cited applications in the NEEA report (Snyder and Lang 2012). The future holds promise for increased data availability and consistency, more robust and accessible software and hardware processing capabilities, further development of applications, and increased integration of LiDAR data into the operational geospatial data -processing chain. This increased capability is well timed since it will become even more vital to map and monitor not only current wetland extent and function but also changes with predicted climate and land use change. LiDAR intensity and elevation data provide synergistic information that can be used for this purpose. LiDAR elevation data can be used to map the potential, static distribution of current and historic wetlands and key wetland functional drivers based on physical controls on water distribution. LiDAR intensity data can be used to map actual, dynamic variations in wetland extent and key functional drivers. The current use of LiDAR data, including the applications described in this CEAP Science Note, support the improved management of wetlands and serve as a foundation upon which to develop even more advanced LiDAR applications that would benefit from improvements in LiDAR technology and availability.

The Conservation Effects Assessment Project: Translating Science into Practice

The Conservation Effects Assessment Project (CEAP) is a multi-agency effort to build the science base for conservation. Project findings will help to guide USDA conservation policy and program development and help farmers and ranchers make informed conservation choices.

One of CEAP's objectives is to quantify the environmental benefits of conservation practices for reporting at the national and regional levels. Because wetlands are affected by conservation actions taken on a variety of landscapes, the wetlands national assessment complements the national assessments for cropland, wildlife, and grazing lands. The wetlands national assessment works through numerous partnerships to support relevant assessments and focuses on regional scientific priorities.

This assessment was conducted and this paper written by Dr. Megan Lang, University of Maryland, Department of Geographical Sciences, College Park, MD, and Dr. Greg McCarty, USDA Agricultural Research Service Hydrology and Remote Sensing Lab, Beltsville, MD

For more information: http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/nra/ceap, or contact Bill Effland at william.effland@wdc.usda.gov.

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Natural Resources Conservation Service
Conservation Effects Assessment Project

CEAP Science Note

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Assessing Wetland Morphometrics and Ecosystem Functions in Agricultural Landscapes of the Atlantic Coastal Plain Using Fine Scale Topographic Information

Summary

- Topography is a known control on multiple ecosystem processes, influencing the movement of water, soil, and other constituents.
- In the Atlantic Coastal Plain, even subtle differences in topography can lead to substantial variations in these processes, including those related to biogeochemistry (i.e., nutrient cycling), erosion/deposition, and surface and groundwater movement.
- Traditionally available digital elevation models (DEMs) created using aerial photography have much coarser vertical accuracies 3.28 – 32.81 feet (1 – 10 m) than those derived from LiDAR ~ 0.50 feet (~ 15 cm).
- LiDAR-derived DEMs also have relatively fine horizontal resolutions ~ 3.28 – 9.84 feet (~ 1 – 3 m).
- Using LiDAR, a total of 14,969 bays were visually identified, and it was estimated that areas without LiDAR data contained approximately 2,000 additional bays for a total of ~ 17,000 bays within the entire study area.
- Mean bay density was found to be approximately 5 bays per mi^2 (2 bays per km^2) ranging up to ~ 69 bays per mi^2.
- Bays had an average area of 6.99 ac (2.83 ha) with mean relief within bays of 3.97 feet (1.21 m; median 3.64 feet [1.11 m]).
- This study provided regional assessment of wetland landscape morphetric information to help identify local soil and hydrologic conditions suitable for supporting wetland functions and wetland restoration.
- For additional information regarding LiDAR technology and wetland conservation applications please see the CEAP Science Note: "Light Detection and Ranging (LiDAR) for Improved Mapping of Wetland Resources and Assessment of Wetland Conservation Practices" (Lang and McCarty 2014).

Background

Topography is a known control on multiple ecosystem processes, influencing the movement of water, soil, and other constituents. In the Atlantic Coastal Plain, even subtle differences in topography can lead to substantial variations in these processes, including those related to biogeochemistry (i.e., nutrient cycling), erosion/deposition, and surface water and groundwater movement. In turn, these processes influence a number of ecosystem services which are highly relevant in agricultural landscapes including the provision of clean water, the management of climate, mitigation of flood hazards, availability of fresh water, and support for soil character and function. In addition, the influence of topography on water flux and availability, soil quality, and nutrient cycling strongly affects crop production. The importance of topography is especially evident near the boundary between wetlands and uplands. This boundary was in large part established by scientists to identify the area at which water regimes, which are greatly influenced by elevation, produce markedly different plant communities and soils. However until recently the spatial resolution of commonly available topographic data were not sufficient for mapping the subtle changes in topography frequently associated with the presence of wetlands, especially in landscapes that are relatively flat, like the Atlantic Coastal Plain.

Light Detection and Ranging (LiDAR)-based digital elevation models (DEMs; see summary at left) enable mapping of landscape features that were previously difficult if not impossible to distinguish with commonly available DEMs produced using stereo-interpretation of aerial photographs. Fenstermacher et al. (2014) highlights the importance of LiDAR-based DEMs for mapping Delmarva bays, elliptical depressional landforms that are commonly found on the agriculturally dominated Delmarva Peninsula, including portions of Delaware, Maryland, and Virginia (Figure 1). Although not all Delmarva bays currently contain wetlands, it is likely that the vast majority did at one time. Furthermore, prior converted croplands (i.e., historical wetlands converted to upland cropland before 1985 and continuously used for agriculture through the present time) have been found to support some wetland characteristics and processes (Fenstermacher et al. 2011; Denver et al. 2014; Hunt et al. 2014; McCarty et al. 2014). Before publication of the Fenstermacher et al. (2014) study, Delmarva bay wetland studies focused on a small number of sites and little was known about the larger population of bays, including their morphology and spatial characteristics as well as their current land cover.

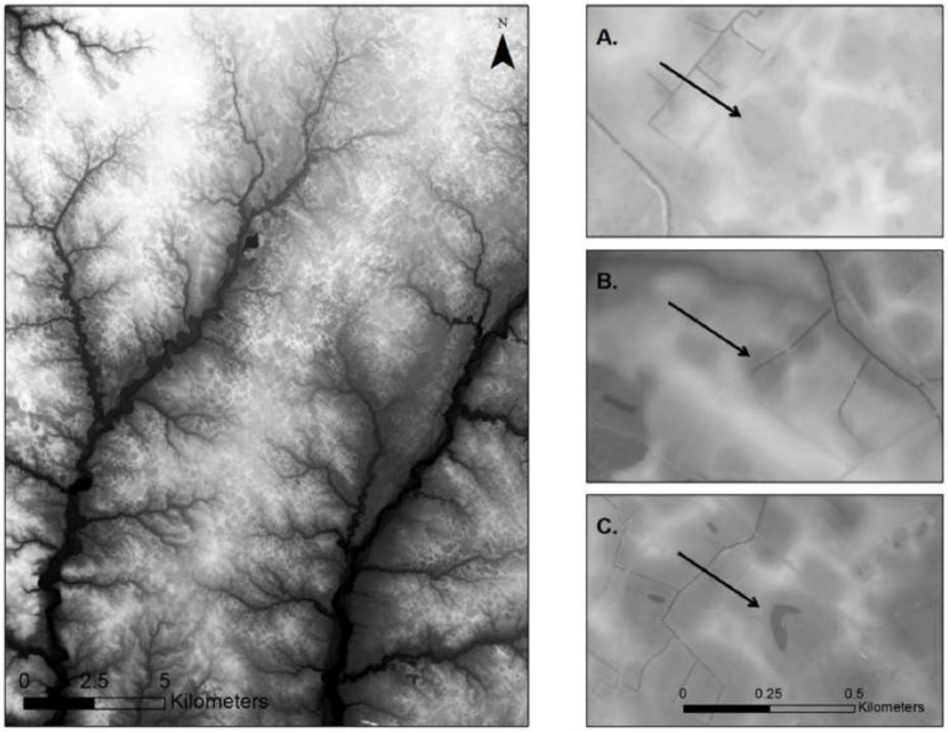

Figure 1. A LiDAR-based digital elevation model (DEM) for a portion of the Choptank Watershed, including areas within Maryland and Delaware. Three expanded views can be seen to the right. Note the abundance of relatively small circular depressions or Delmarva bays which are present across all land covers including both cropland and forest. The images to the right provide a more detailed look at (A) natural, (B) historical, and (C) restored Delmarva bays. Historical Delmarva bays are often drained by ditches and can be seen in all images. Note that only the northwest portion of the restored wetland (C) has been restored through excavation, while the southeastern portion of the bay was not converted to cropland.

This CEAP Science Note summarizes the Fenstermacher et al. (2014) study findings and highlights the importance of this type of morphometric assessment for the estimation of ecosystem services provided by natural, restored, and historical wetlands (i.e., prior converted croplands) and assessment of agricultural management practice effects.

LiDAR Reveals the Density, Distribution and Morphology of Delmarva Bays

Introduction to Delmarva Bays

Delmarva bays are believed to be a geographic subset of the depressional features that have more broadly been termed Carolina bays. Although the Carolina bays of North and South Carolina are the best known examples, natural depressions with a unique elliptical shape are found along the Atlantic Coastal Plain from New Jersey to Florida and along the Gulf of Mexico. In the Alabama and Georgia Coastal Plain areas, these depressions are locally known as "Grady ponds." Carolina bays are often oriented along a northwest – southeast major axis (Sharitz & Gibbons 1982; Stolt & Rabenhorst 1987b; Bruland et al. 2003) and typically have a sandy rim at their southeast end (Prouty 1952; Thom 1970; Stolt and Rabenhorst 1987a; Tiner 2003). Bays range in size from tens of meters to kilometers in length and cover as much as 50% of the land area where they are most abundant (Prouty 1952). Although there are a number of theories regarding the origin of Carolina and Delmarva bays, available field evidence suggests that they were wind blow-outs formed during the Pleistocene that filled with water and were elongated by wind-driven currents, resulting in their unique shape and characteristic sandy rim (Grant et al. 1998; Prouty 1952; Savage 1982; Stolt and Rabenhorst 1987b; French and Demitroff 2001). Less information is known about Delmarva bays than Carolina bays. Delmarva bays are generally smaller than Carolina bays, which are found to the south in North and South Carolina. Delmarva bays are known to be extremely common in portions of the Delmarva Peninsula including Queen Anne's, Caroline, and Talbot Counties in Maryland and New Castle and Kent Counties in Delaware. Their common occurrence exerts strong controls on field and landscape scale processes in this region (Figure 1). Delmarva bays that have not been drained for agricultural or urban development typically contain wetlands. These wetlands can act as both recharge wetlands that replenish groundwater and discharge wetlands that receive groundwater during different times of the year and in accordance with large or prolonged weather events. Where Delmarva bays are abundant, they constitute the majority of wetlands and provide important habitat to a disproportionately high number of rare and endangered species (Sharitz 2003; Olivero and Zankel 2000; Sharitz and Gibbons 1982).

Assessment Approach

Study Area

The Fenstermacher et al. (2014) study was conducted on the ~ 6,000 mi^2 (15,540 km^2) Delmarva Peninsula, in areas of Maryland and Delaware. The Delmarva Peninsula is located within the outer Coastal Plain Physiographic Province and has a humid subtropical climate with an average annual rainfall of 44 inches (Denver et al. 2004). The landscape is generally flat (elevation between 0 and 102 feet [0 and 31 m]) and is dominated by agriculture (48%), primarily corn and soybean fields, but also includes forests (33%), and a smaller amount of urban areas (7%)(Denver et al. 2004).

Publicly available LiDAR based DEMs with a spatial resolution between ~ 6.6 and 9.8 feet (2 and 3 m) and a vertical accuracy of approximately 7.1 inches (18 cm) were obtained from the USDA Geospatial Data Gateway and the Maryland Department of Natural Resources. These data were used to manually identify Delmarva bays based on their characteristic elliptical shape. Although automated processes are available to identify landscape features with distinct shapes, a manual process was selected due to the complex morphology of many Delmarva bays which have been superimposed upon each other, bisected by ditches, or otherwise modified. Bays with a continuous elliptical perimeter were identified as a single feature. Where the rims of overlapped bays were sufficiently distinct they were recognized and counted as separate features. Man-made depressions, such as ponds or reservoirs, which typically have a linear side for an earthen dam, were excluded from the study. When LiDAR-derived DEMs were not available for sites, their density was assumed to be similar to adjacent areas.

A stratified random approach based on bay density was used to select areas for more detailed morphologic analysis. Using this approach a total of 1,494 bays were selected, manually outlined, and their area, perimeter, major and minor axis, relief and land cover were determined using ArcGIS 9.2 (Environmental Systems Research Institute, Redlands, CA). Bays were categorized as having a natural, agricultural, residential, and/or fallow land cover class using false-color near-infrared aerial photography obtained from the USDA Geospatial Data Gateway. Additional information regarding the methods used to map and characterize Delmarva bays can be found in Fenstermacher et al. (2014).

Results and Discussion

A total of 14,969 bays were visually identified (Figure 2), and it was estimated that areas without LiDAR data contained approximately 2,000 bays for a total of ~ 17,000 bays within the entire study area (Fenstermacher et al. 2014). Previous estimates based on aerial photography are an order of magnitude less, including an estimate of 1,500 – 2,500 (Stolt and Rabenhorst 1987b) and an estimate of 10,000 to 20,000 for the entire Atlantic Coast (Richardson and Gibbons 1993). Mean bay density was found to be approximately 5 bays per mi^2 (2 bays per km^2) but was as high as ~ 69 bays per mi^2 (27 per km^2) accounting for over 50% of land area (Fenstermacher et al. 2014). Bays had a mean area of 6.99 ac (2.83 ha; median 3.58 ac [1.45 ha]), with 80% between 1.14 and 14.04 ac (0.46 and 5.68 ha).

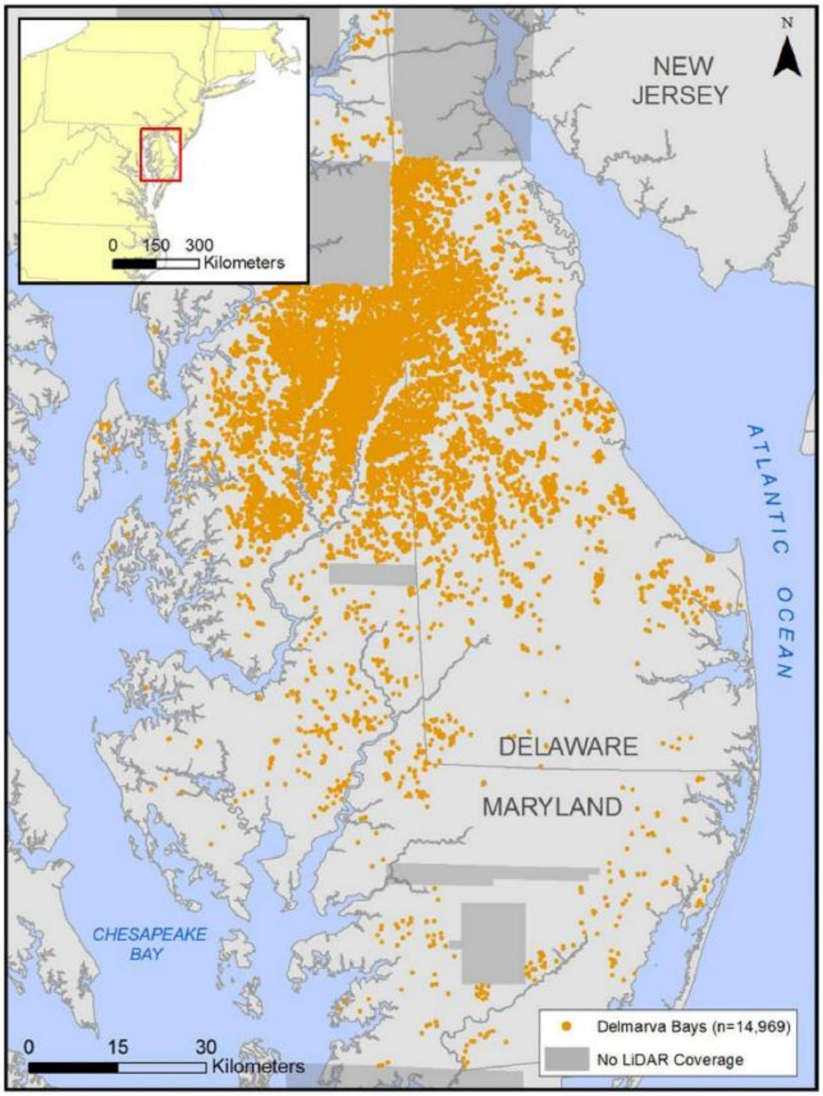

Figure 2. The abundance and distribution of Delmarva bays within the Delmarva Peninsula study area. Each dot represents one Delmarva bay. A total of 14,969 bays were manually identified using a LiDAR-based digital elevation model (DEM). Areas where LiDAR data were not available are marked in gray. Note that bays are concentrated in the northern portion of the Peninsula and are less likely to be found near large streams and the shoreline.

Mean relief within bays was 3.97 feet (1.21 m; median 3.64 feet [1.11 m]) with 80% falling within the range of 1.81 to 6.63 feet (0.55 to 2.02 m). Delmarva bays had an average major to minor axis ratio of 1.32 (median 1.26), with 80% falling within the range of 1.08 to 1.65 (Fenstermacher et al. 2014). Overall Delmarva bays were found to be smaller, shallower, and rounder than Carolina bays, which have been found to have a mean area of 113.67 ac (46 ha) (Bennett and Nelson 1991), relief of 5.94 feet (1.81 m)

(Prouty 1952; Thom 1970), and major-minor axis ratio of 1.51 (Melton and Scriever 1933). Fenstermacher et al. (2014) hypothesized that the difference between Delmarva and Carolina bay morphologies may be due to the relatively colder temperatures of the Pleistocene during the development of the higher latitude Delmarva bays. Frozen water would have been more common with the Delmarva bays and could have inhibited development of bay morphology due to wind driven waves, therefore limiting the size and elliptical shape of these features. This hypothesis is supported by the relatively large size of bays found in the southern portion of the Delmarva relative to the northern Delmarva.

The vast majority of Delmarva bays have been influenced by human development, mainly agriculture, with only 29% (4,930 out of the estimated 17,000 total; Fenstermacher et al. 2014) located within areas of natural vegetation. Many of these have been drained and all have likely been affected by regional declines in groundwater due to irrigation, human consumption, and other uses. Delmarva bays found entirely within natural land covers had significantly greater ($p < 0.001$) relief (4.17 feet or 1.27 m) than those in agriculture (3.54 feet or 1.08 m; Fenstermacher et al. 2014). This reduction in relief within cropland bays may have been caused by erosion and sedimentation following tillage or resulted from the preferential selection of shallower bays with better drainage for agricultural development. The average area of Delmarva bays found in natural and agricultural landscapes was not shown to be significantly different (Fenstermacher et al. 2014).

The Importance of Landscape-Scale Wetland Assessment

Although successful wetland restoration is generally considered to provide net benefits to society, the large investment that USDA has made in wetland restoration and increasing societal need for wetland ecosystem services highlight the importance of environmental research and monitoring. These efforts are needed to better understand the effects and effectiveness of conservation practices, such as wetland restoration, and to develop wetland restoration and agricultural management practices that result in greater societal benefits.

Fenstermacher et al. (2014) demonstrates the significant and growing importance of remote sensing for supporting these efforts, both through the extrapolation of field scale information and greater understanding of landscape scale processes that would have been costly and difficult to ascertain on the ground. The functions that occur within individual or groups of wetlands are unique to their placement on the landscape (Bedford 1999; Simenstad et al. 2006). Therefore the landscape perspective that remote sensing provides is critical to ensuring the optimum provision of wetland ecosystem services through restoration at the individual wetland and watershed scale. The use of remotely sensed data can also provide temporal context. The importance of this historic perspective was emphasized by Bedford (1999): "By definition [wetland restoration] seeks to replace what has been lost. By definition then, it should be undertaken with knowledge of what has been lost."

Wetland restoration has proven to be difficult, partly because wetlands are regionally and locally distinct (Zedler and Callaway 1999), and restoration of wetland hydrology is considered to be one of the most difficult and critical components of restoration. Lang et al. (2012) found relief to be well correlated with patterns of inundation on the Delmarva Peninsula and developed a LiDAR-based technique to map elevation driven controls on wetland distribution and hydroperiod. The link between hydroperiod and thus relief and the distribution of plant and animal species is well known (e.g., Pechmann et al. 1989; Corti et al. 1996; Snodgrass et al. 2000).

Current wetland restoration practices seek to mimic more natural variation in relief, making them generally shallower and adding micro-topography. The Fenstermacher et al. (2014) study provides a guideline as to variations in depressional wetland relief that are naturally occurring, thus supporting the stated goal of the USDA NRCS Wetland Restoration (657) Practice Standard to "restore wetland function, value, habitat, diversity and capacity to a close approximation of the pre-disturbance conditions." The ability to locate and restore former depressional wetlands with sufficient relief to support wetland hydrology without the need for excavation could be advantageous for the management of greenhouse gases and thus climate via carbon sequestration, since wetland excavation on the Delmarva Peninsula was found to lower soil organic carbon levels relative to even historical wetlands and this topsoil was found to be used in berms or other areas where oxidation and loss of carbon to the atmosphere was more likely (Fenstermacher 2011).

In conjunction with relief, depression size (i.e., volume) is also key to supporting wetland processes. McDonough et al. (2014) found wetland area to be correlated with flow in adjacent streams when depressional wetlands were connected to those streams via surface flows. Wetland volume relative to landscape position (e.g., catchment area) is considered to be critical to the establishment of wetland hydroperiod and therefore restoration success (Bedford 1999). Restored depressional wetlands have been found to generally be smaller than natural depressional wetlands (Galatowitsch and van der Valk 1996 [Prairie Pothole Region]; McDonough et al. 2014 [Delmarva Peninsula]; Mid-Atlantic CEAP-Wetlands *unpublished*). Thus larger wetland restorations may be needed to enhance the ability of restored wetlands to maintain surface water flows and likely to mitigate floods. Even when depressional wetlands are not directly connected to streams via surface water flow, their size and arrangement has been found to be critical for supporting flow in adjacent streams (McLaughlin et al. 2014). Remote-sensing based studies such as Fenstermacher et al. (2014) provide the context necessary to better approximate historical conditions, a USDA NRCS Wetland Restoration (657) Practice Standard goal, and wetland hydrology, a critical factor in restoration success.

Fenstermacher et al. (2014) provides insights regarding where local soil and hydrologic conditions may be suitable for supporting wetland function. These specific sites are more likely to be well suited for wetland restoration. This restoration information is especially critical considering the fact that on the Delmarva peninsula most wetland restorations have a depressional shape or morphometry although additional wetland types, including flats, and riparian wetland do occur there.

Information regarding the distribution, density, and morphology of Delmarva bays produced by Fenstermacher et al. (2014) is currently being analyzed to estimate the historical and current storage of surface water within Delmarva bays, as well as the contribution of USDA wetland restoration practices to enhanced wetland volume storage. This study was made possible by nationally available land cover maps produced using remotely sensed data and LiDAR-derived DEMs.

The wetland morphometric data (Fenstermacher et al., 2014) also support the extrapolation of results from a number of other studies supported by the Wetland Component of the National Conservation Effects Assessment Project, including studies documenting plant and amphibian biodiversity and abundance (Yepsen et al. 2014; Mitchell in review), carbon storage, quality, and movement (Fenstermacher 2011; McDonough et al. in review), and nutrient dynamics (Denver et al. 2014; Hunt et al. 2014) within natural, restored and historical wetlands in the Mid-Atlantic Region. Indeed, remotely sensed data greatly adds to wetland insights obtained on the ground and via modeling. CEAP team members are currently working to better incorporate remotely sensed data into process-based modeling, thus supporting the CEAP National Assessment for Cropland and Wetlands.

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The Conservation Effects Assessment Project (CEAP) is a multi-agency effort to build the science base for conservation. Project findings help to guide USDA conservation policy and program development and help farmers and ranchers make informed conservation choices.

One of CEAP's objectives is to quantify the environmental benefits of conservation practices for reporting at the national and regional levels. Because wetlands are affected by conservation actions taken on a variety of landscapes, the wetlands national assessment complements the national assessments for cropland, wildlife, and grazing lands. The wetlands national assessment works through numerous partnerships to support relevant assessments and focuses on regional scientific priorities.

This analysis was conducted by Daniel Fenstermacher, former UMD graduate student; supervised by Dr. Martin Rabenhorst, UMD. The Science Note was written by Dr. Megan Lang, UMD, Daniel Fenstermacher; Drs. Martin C. Rabenhorst and Brian Needelman at UMD, and Dr. Greg McCarty, USDA Agricultural Research Service Hydrology and Remote Sensing Lab, Beltsville, MD.

For more information on CEAP: www.nrcs.usda.gov/technical/NRI/ceap/ or contact William (Bill) Effland at william.effland@wdc.usda.gov.

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C. Importance of MIAR Field and Geospatial Data for Wetland Modeling

It is common knowledge that distributed, process-based models have a large number of parameters, partly due to the complex nature of the physical and biogeochemical processes that they characterize, and partly in an attempt to represent spatial heterogeneity (Whittaker et al., 2010). Most of these parameters are not directly measured *in situ* and need to be calibrated by matching modeled fluxes (e.g. discharge and nutrient loads) with monitored data (Beven 2000; Yen et al. 2014a). The calibration process is either performed manually, or using an autocalibration program that employs an optimization scheme for maximizing an objective function that is based on statistics that reflect goodness of fit between model results and field observations. In this conventional calibration practice, it is assumed that the model reflects true system behavior when the "global" model responses (e.g., discharge or nutrient fluxes) adequately match the field observations. However, it is quite feasible that adequately calibrated models, given the conventional standards described above, may contain input data errors not readily identifiable by model users (White et al. 2014), or may not realistically represent physical or biogeochemical reactions/mass exchanges in the environment (e.g. decomposition, denitrification, and NO_3 leaching) or other important fluxes (Yen et al. 2014b). Such flaws can be substantial, and lead to erroneous predictions when models are used to estimate the effects of conservation practices, other management efforts, or future climate scenarios. In response to the shortcomings of conventional calibration methods, some techniques and recommendations have recently been developed by the scientific community. Arnold et al. (2015) reviewed calibration strategies of 25 model application studies at different scales and provided recommendations for calibration/validation of hydrologic and water quality models. The recommendations include a four step process that embraces use of "hard data" – measured flow and nutrient fluxes – and "soft data" – estimations of physical and biogeochemical processes and exchanges (Yilmaz et al., 2008; Seibert and McDonnell, 2002).

The MIAR CEAP-Wetland study is currently working to use and enhance existing USDA process based models (i.e., APEX, SWAT, and ALMANAC) with the objective of accurately assessing the effects and effectiveness of wetland conservation practices. APEX and SWAT are process based field/watershed scale models that are subject to the negative effects described above, similar to many other models. It is known that wetland ecosystems are relatively complex and their biogeochemistry varies significantly from uplands. Therefore a model that incorporates wetlands, in addition to uplands, requires multifaceted relationships to describe the inherent complexity of these environments. In other words, more model parameters and variables are required to describe the complex relationships between various microbial and plant communities, nutrient pools, and hydrology. With every new parameter introduced, model uncertainty increases. One solution for reducing model uncertainty is to limit the range of each parameter using *in-situ* measurements. This is one important way in which existing MIAR field data directly benefit process-based modeling. In addition, much of the data described in Chapters B1-9 can be used to validate model fluxes, and avoid erroneous calibrations. For instance, estimates of surface water storage derived from remotely sensed data can be used to validate the hydrologic components of a model. Soil physicochemical parameters and microbial reaction rates, described in Chapters B2, B3 and B4, are vital to a successful model application. The field of wetland modeling generally suffers from a lack of available *in situ* data. The data highlighted in Chapters B1-9 will be used to reduce model prediction uncertainty, and increase prediction reliability.

D. Summary

Although successful wetland restoration is generally considered to provide net benefits to society, the large investment that the US Department of Agriculture (USDA) has made in wetland restoration emphasizes the importance of environmental research and monitoring in coordination with implementation. Increasing societal need for wetland ecosystem services further highlights the importance of this effort. Most comprehensive, field based studies of depressional wetland restoration effects have focused on a small number of sites. Robust consideration of restoration effects that addresses the ecological processes necessary to draw conclusions regarding multiple ecosystem services across a large geographic area has been uncommon; until now this type of study had not occurred in the Mid-Atlantic region of the US. The MIAR CEAP-Wetland study identified implementation and management actions influencing wetland restoration effects and effectiveness in agricultural landscapes. It is a common wetland restoration landscape; yet, it presents unique conditions and implications. These restorations will become increasing vital to both minimize potential negative environmental impacts of agricultural production and to maximize provision of ecosystem services from agricultural landscapes. Both functions will be critical as a growing human population simultaneously increases the demand for such services, including the provision of food, fiber, and bioenergy.

Overall study results indicate a trend of recovery, with restored wetlands taking on many of the attributes of natural wetlands. However, this trend was not shared amongst all wetland characteristics and functions, and intra-regional differences amongst certain criteria were significant. In terms of carbon related functions that primarily support the provision of climate regulation services, as well as help drive nutrient regulation services, some wetland restoration practices were found to cause an initial, significant decrease in function. Excavation, as a restoration practice, significantly reduced soil carbon stocks at sites with mineral soils, the majority of MIAR sites (Chapter B2). Excavated soil was found in berms and other landscape positions where oxidation and thus loss of carbon to the atmosphere is likely. This reduction in restored wetland carbon stocks is in addition to the very significant decrease in soil carbon stocks associated with cultivation of prior converted croplands. Furthermore, excavation is known to increase soil compaction, which in turn can lead to reduced carbon inputs (i.e., plant growth). Excavation has also redistributed nutrients and potentially soil microbial communities with potential impacts on nitrate removal and other biogeochemical processes. However, less invasive restoration approaches (e.g., ditch plugs and berms) did not have this effect. Above-ground plant biomass increased with time since restoration, indicating that carbon inputs, in addition to reduced oxidation through suspension of tillage and increased anaerobic conditions, should eventually compensate for losses. However, this could take decades, if not over a century. A MIAR CEAP-Wetland study by McFarland et al. (2015) found that after 25 years post-restoration annual herbaceous biomass and standing wood stocks were approaching levels found in natural wetlands, but annual tree leaf litter inputs were still significantly lower than that of natural wetlands. Increased accumulation of soil carbon was evident in the upper-most soil horizons at some sites. The effects of carbon loss through oxidation associated with drainage are pronounced in areas with Histosols, but recovery trends are evident post-restoration.

Soil physicochemical characteristics that control and indicate the regulation of pollutants (e.g., nitrogen and phosphorus), and climate appear to be recovering post-restoration, but have not achieved levels identical to natural wetlands within a decade post-restoration (Chapter B3). Within restored wetlands, soil physicochemical parameters, were found to generally be intermediate to those found in prior converted croplands and natural wetlands. However, the level of restored wetland soil parameters relative to natural wetlands and prior converted croplands, and therefore likely time necessary for recovery post-restoration, varied considerably between parameters. For example, phosphorus, zinc, pH, calcium, electrical conductivity, and soil moisture appear to recover faster than parameters like magnesium. The effects of fertilization and liming, as indicated by soil physicochemical parameters, are still evident within restored wetlands, but impacts of these management practices have decreased compared to prior converted croplands. This is also illustrated in the nutrient contents of herbaceous vegetation found across the alteration gradient (Chapter B1). Nutrient concentrations were significantly higher in prior converted cropland sites relative to natural sites, and restored sites demonstrated values intermediate to the other site types.

Wetland restoration practices appear to have enhanced phosphorus regulation capacity (Chapter B4). Our findings suggest that natural wetlands have the greatest potential for phosphorus uptake, while prior converted cropland sites have the lowest, and may even serve as a source of phosphorus to adjacent water bodies due to high phosphorus levels relative to binding sites (i.e., Phosphorus saturation). The capacity of restored wetlands to regulate phosphorus is in between that of prior converted croplands and natural wetlands, but there is still potential for the provision of this service to be substantially increased within restored wetlands through the natural weathering processes, which can create new soil A horizons and lower pH. It is notable that pH may have a significant effect on not only soil biogeochemistry, but also habitat for amphibians and other species. Although excavation can reduce the provision of some ecosystem services, it enhances Phosphorus mitigation services through the removal of Phosphorus rich topsoil, assuming the topsoil is placed in a landscape position less prone to anaerobic conditions, leaching, and erosion.

Soil nitrate removal potential and microbial communities that indicate the ability of wetlands to provide pollutant and climate regulation services also indicated a trend of recovery post-restoration, but intra-regional variations in implied service provision were also evident (Chapter B3). In areas dominated by mineral soils, potential nitrate removal in restored wetlands was more similar to natural wetlands, but in areas dominated by Histosols nitrate removal was intermediate to prior converted croplands and natural wetlands at lower landscape positions and surprisingly lower than other wetland types at higher landscape positions. The presence of a key functional gene (*NosZ*) within the microbial community that supports the conversion of an important greenhouse gas (i.e., nitrous oxide) to a harmless, common component of the atmosphere (i.e., dinitrogen gas) was found to be significantly different and intermediate to natural wetlands and prior converted croplands within restored wetlands on mineral soil. However, this gene was found to be significantly lower than both natural wetlands and prior converted croplands within restored wetlands on Histosols. These findings indicate a microbial community that has been influenced by restoration, and is likely still in flux. This transition may have been influenced by removal of topsoil (i.e., excavation), and thus microbial communities, during wetland restoration. Emissions of nitrous oxide and carbon dioxide were highly variable through time and between individual sites, but were not found to vary significantly between wetland type (Kluber et al. 2014). This implies that wetland restoration did not significantly increase the emission of greenhouse gases. It should be noted that soil biogeochemistry was found to vary significantly within sites based on elevation, with nitrate reduction potential generally increasing with decreasing elevation. Although our study stratified and thus accounted for these differences, this fact demonstrates not only the importance of such sampling protocols but also the potential impact of restoration implementation techniques which alter elevation and potentially hydroperiod.

The research described in this report, as well as an earlier CEAP-Wetland study conducted within the Choptank River watershed, indicates that restoration increases nitrate removal services, although nitrate removal services are also substantial within prior converted croplands (Chapter B3; Denver et al. 2014). Although potential nitrate removal is high in natural wetlands, these wetlands often receive less nitrate due to their landscape position and location relative to nitrate sources (Denver et al. 2014). Therefore nitrate regulation services provided by both restored wetlands and prior converted croplands are vital for maintaining the health of adjacent waters, including the Chesapeake Bay. Considering the significantly larger area that prior converted croplands occupy on the landscape relative to restored wetlands (e.g., Chapter B9; Fenstermacher et al. 2014; Lang et al. 2015), these areas merit special consideration. Conservation practices specific to these "critical zones" deserve increased attention, and include use of controlled drainage on ditches and tile drains. Although the benefits of all wetland types have been demonstrated in terms of localized biogeochemical processes, the level of nitrate pollutant regulation services ultimately provided by these areas will largely be determined by local groundwater flow pathways. Simply, nitrate rich groundwater must enter areas of anaerobic wetlands soils in order to be treated, and the volume of water entering these areas will, in large part, determine the ultimate level of water quality services being provided. In the Delmarva Peninsula, this volume of water is largely determined by the depth of anaerobic wetland soils relative to the depth of the surficial aquifer (Denver et al. 2014). Until very recently, the depth of the surficial aquifer could not be determined at a landscape scale for use in conservation practice targeting. Now that a map of surficial aquifer depth is available from USGS for the Delmarva Peninsula, its use in targeting should be strongly encouraged. Soil compaction will reduce the depth of anaerobic wetland soils and the flow of groundwater into these soils; compaction has been found to be common within restored wetlands (Palardy 2016).

Wetland restoration in the MIAR helps to support the regulation of hydrologic flows and mitigation of natural hazards (e.g., flooding). An earlier Choptank CEAP-Wetland study tracked surface water outflows at eleven sites, including four natural wetlands, three restored wetlands, and four prior converted croplands (McDonough et al. 2015). The study found that natural wetlands exhibited relatively continuous flow into adjacent streams in contrast to prior converted croplands, which provided flashier flows directly after precipitation events. Restored wetlands exhibited surface water flows intermediate to natural wetlands and prior converted croplands. Wetland area was found to be significantly correlated with the periodicity of surface water flows. This study has implications not only for the regulation of hydrologic flows and provision of freshwater but also for the health and productivity of downstream waters. A related MIAR CEAP-Wetland study (McDonough et al. [In Review]) found that the quality (e.g., bioavailability) of carbon in outflows from restored wetlands better approximated natural wetlands than prior converted croplands and postulated that stream food web dynamics and nutrient cycling may, in part, be restored through the restoration of nearby wetlands. Lang et al. (2012) demonstrated that surface hydrologic connections between various types of wetlands, including depressional wetlands that are often considered to lack these connections, and streams is higher than previously estimated using nationally available stream datasets (i.e., USGS National Hydrography Dataset). The article also infers that human altered wetlands are more likely to be connected to streams, and that the source of geospatial datasets can have a significant influence on assessment results. Even when depressional wetlands are not directly connected to streams via surface water flow, their size and arrangement has been found to be critical for determining flow in adjacent streams (McLaughlin et al. 2014). Although wetland restoration has been found to exert a positive effect on the regulation of hydrologic flows, and likely natural hazards, the extremely large volume of surface water storage that has likely been lost at a landscape scale (23,500 ha-m) relative to the modest gains in water storage made possible by restoration highlights the need for increased, sustained wetland restoration (Chapter B5).

Wetland restoration was found to have a strong, positive effect on plant and amphibian biodiversity and community quality, but restored communities were significantly different than those found in natural wetlands. Restored wetlands were hotspots of plant biodiversity, not only greatly surpassing biodiversity in prior converted croplands but also demonstrating greater biodiversity than natural wetlands (Chapter B6). Although natural and restored wetlands were both dominated by native species, their plant communities were primarily composed of different functional groups, and restored species were more associated with disturbed conditions. Metrics including plant biomass, percent cover, and richness all indicate that restored wetlands are early successional ecosystems dominated by herbaceous vegetation. On the other hand, natural wetlands are dominated by woody plants, in terms of both number of species and biomass/cover. This fundamental difference in plant functional types will undoubtedly lead to the preferential support of different fauna through provision of greatly different habitat types. However restored sites are following a trajectory of recovery and we predict that they will develop similarly to natural sites if succession is allowed to progress for decades. A later MIAR CEAP-Wetland study by McFarland et al. (2015) found that after 25 years post-restoration annual herbaceous biomass and standing wood stocks were approaching levels found in natural wetlands, but annual tree leaf litter inputs were still significantly lower than that of natural wetlands. Natural and restored wetlands supported a similar number of amphibian species (Chapter B7). Even prior converted croplands with requisite breeding habitat in the form of ditches supported about half the number of amphibian species found in natural and restored wetlands. Restored and natural wetlands supported approximately equal proportions of habitat generalists and specialists, but community similarity was relatively low. Thus restored and natural wetlands support diverse, high quality communities of plants and amphibians, but these communities contain different species. These differences likely have a significant effect on other, unstudied biota, through effects on habitat and food source. This is illustrated by the significant, negative effect of tree canopy closure on amphibian larval species richness. It implies that overall landscape scale biodiversity is enhanced through the presence of a combination of natural and restored ecosystems.

Landscape-scale analysis, including the use of remotely sensed imagery, was found to: 1) identify segments of the landscape that experience unique, wetland-related ecosystem processes; 2) help develop guidelines pertaining to naturally occurring morphologies; and 3) extrapolate field-based findings across the landscape. Topographic metrics were found to be correlated with different drivers of wetland hydroperiod (e.g., overland flow versus groundwater), and these metrics were found to predict the distribution of ponding across the landscape during periods with varying weather patterns (i.e., hydroperiod; Lang et al. 2013; Chapter B9). An ongoing MIAR CEAP-Wetland study found that maps of hydroperiod could be used to identify Delmarva bays that contained over three times the amount of soil organic carbon stocks as other Delmarva bays. Information regarding drivers of wetland function can be readily incorporated into regional maps, which can guide the implementation of conservation practices and enhance statistical and process-based models of wetland function. The regional map of denitrification potential described in Chapter B8 can be used to target restoration of prior converted croplands that can best enhance water quality improvements through greater interception and transformation of agricultural nitrate. Chapter B9, in part, describes naturally occurring ranges in wetland relief, area, and shape, whereas Chapter B5 describes naturally occurring ranges in wetland volume storage. These estimates can be used to best support the stated goal of the USDA NRCS Wetland Restoration (657) Practice Standard, which is to "restore wetland function, value, habitat, diversity and capacity to a close approximation of the pre-disturbance condition." The ability to locate and restore former depressional wetlands with sufficient relief and catchment to volume ratio to support wetland hydrology without the need for excavation could be advantageous for the enhanced regulation of climate via carbon sequestration. Chapter B5 demonstrates the utility of remotely sensed data for the extrapolation of field-based measurements (see above). The next several years will bring dramatic changes to the availability of not only remotely sensed images that are well suited for the mapping and monitoring of wetlands and wetland conservation practices, but also the availability of wetland related products (i.e., maps) that can be quickly incorporated into decision support systems (Chapter B9; Snyder and Lang 2012; Lang et al. 2015b; Lang et al. 2015a). The strong potential of remotely sensed data and products for improving conservation practice assessment and implementation paired with the rapid increase in the availability of such datasets highlights the critical role that this information source will play in future conservation efforts.

The MIAR CEAP-Wetlands study has developed a broad collaborative base, which has facilitated the collection and dissemination of novel integrative findings regarding wetlands in agricultural watersheds. In the past restoration of hydrology was considered to be the most challenging aspect of wetland restoration. Although restoration of natural wetland hydroperiod is essential to the success of any wetland restoration, complete restoration of natural physical conditions also necessitates restoration of wetland soil structure, chemistry, and biota. Doing so may turn out to be just as challenging, or perhaps more challenging, than restoring wetland hydrology. These challenges will be compounded by climate and land cover change, as well as shifts in socio-economic drivers, commodity prices, and policies. This study provides critical information that will advance our ability to restore all aspects of wetland ecosystems, thus enhancing the provision of ecosystem services. These findings not only help scientists, managers, and policy-makers better understand the impacts of wetland restoration relative to the existing wetland resource, but also support the improved allocation of resources and refinement of conservation implementation and management practices to optimize environmental outcomes. Insights gained from this study are particularly relevant to wetland restoration within agricultural landscapes, a practice that has been found to be altering the location and type of wetland ecosystems at a national scale (Dahl 2001). Specific recommendations regarding wetland restoration implementation and management can be found in the next section of this report.

E. Implications for Wetland Restoration Implementation and Management in the Mid-Atlantic Region

Overall study results indicate a trend of recovery, with restored wetlands taking on many of the attributes of natural wetlands. However, this trend was not shared amongst all wetland characteristics and functions, and actions can be taken by land managers to both encourage existing positive trends and better support functions that are lagging post restoration with the goal of enhanced provision of ecosystem services and environmental outcomes. A list of these potential actions can be found below, divided between recommendations that should generally support the provision of all or most ecosystem services, those that generally support ecosystem service provision but have notable trade-offs identified by this study, and actions that should be investigated to determine merit before implementation. These recommendations should be considered in the context of long-term conservation practice implementation objectives. As always, these objectives, along with logistical and resources considerations, should be used to guide the adoption of the targeting, implementation, and management practices.

General Recommendations

1) An expectation of many wetland restoration efforts is that ecological functions will resemble natural systems within a decade, but most functions have longer trajectories (e.g., Zedler and Callaway 1999; Ballantine and Schneider 2009; Suding 2011; Moreno-Mateos et al. 2012). Longer easement/contract periods should be promoted to allow time for slower environmental processes, including plant succession, soil carbon accumulation, and development of more natural soil biogeochemistry (e.g., microbial communities and pH).

2) Soil compaction should be avoided to encourage root growth and the movement of nitrate rich groundwater into wetland soils capable of nitrate removal (i.e., denitrification). Avoidance of soil compaction may make restoration of wetland hydrology more challenging, but improved targeting through the use of a LiDAR derived digital elevation model should, in part, assist with calculation of suitable wetland volume relative to catchment area, which is considered to be critical to the establishment of wetland hydroperiod (Bedford 1999).

3) Either a greater number of restored wetland cells and/or larger wetland cells should better support the regulation of hydrologic flows and groundwater levels, and the mitigation of natural hazards, such as flooding.

4) Natural wetlands should be conserved, not only due to the high level of ecosystem services that they provide and the fact that they harbor species that are not found in restored wetlands, but also because they serve to directly enhance the provision of ecosystem services from restored wetlands and prior converted croplands. Natural wetlands serve as sources of biota, thus assisting with the colonization of restored wetlands by desirable species, and when located near nitrate sources may encourage nitrate removal through the maintenance of anoxic sediments beneath adjacent ditches that intercept agricultural contaminants in groundwater (Denver et al. 2014).

5) Because local topographic relief does not predict groundwater flow pathways in flat landscapes, an effort should be made to restore wetlands in locations that are low relative to broader-scale topographic gradients, and therefore more likely to intercept upgradient groundwater containing agricultural contaminants.

6) Wetland basins should be relatively shallow with gently sloping topographies, such that they support hydroperiods and water depths characteristic of natural wetlands to encourage colonization and growth of species that are representative of more natural conditions. Water depths and hydroperiods should be deep/long enough to discourage colonization by upland

plants, reduce loss of carbon to the atmosphere (i.e., oxidation), and support the development of amphibian larvae but shallow/short enough to encourage plant growth and discourage the establishment of predatory fish populations.

7) Even within the MIAR, sub-areas with significantly different properties are found; these natural conditions influence the effects of wetland restoration on ecosystem service provision. These differences may include topographic, geologic, and climatic properties that result in considerable edaphic, hydrologic, and biologic variations. These variations should be considered when targeting, implementing, and managing wetland conservation practices.

8) Geospatial data, along with requisite hardware, software, and processing methods have matured considerably in the past decade, and greatly improved data and protocols will soon be developed. Examples of the utility of these tools and techniques can be found in Chapters B5, B8, and B9. Another example of note is the development of a depth of surficial aquifer map for the Delmarva Peninsula by the US Geological Survey. The greater incorporation of these datasets and techniques into precision conservation practice implementation and management strategies would serve to enhance not only ecosystem service provision but also the determination of derived benefits at a landscape scale, thus enhancing accountability.

Service Specific Recommendations

1) Overall our findings suggest that excavation should be minimized through enhanced targeting. Targeting should consider both landscape placement through the use of high resolution geospatial data sets, including digital elevation models, and direct examination of soils. When excavation cannot be avoided excavated topsoil should be replaced. Although excavation can reduce the provision of some ecosystem services, it enhances Phosphorus mitigation services through the removal of Phosphorus rich topsoil, assuming the topsoil is placed in a landscape position less prone to anaerobic conditions, leaching, and erosion. Therefore the merits of excavation should be considered relative to desired outcomes, and the practice maybe tailored to achieve desired outcomes by, for example, considering the degree of Phosphorus saturation in the surrounding landscape.

2) Overall findings suggest that the practice of mowing should undergo benefit analysis, since it prevents the establishment of woody species, which are more characteristic of natural wetlands in the MIAR. However, increased forest canopy cover has been found to reduce the abundance and diversity of amphibian larvae, or may impact suitability for migratory bird habitat. Therefore the merits of mowing should be considered relative to desired outcomes.

Recommendations Requiring Additional Research

1) It is possible that wetland restoration practitioners could hasten development of more natural soil conditions through the active lowering of pH via the direct application acidifying agents to counter the effects of lime used in agricultural production. However, additional research is needed to further evaluate the potential benefits of this practice and develop implementation techniques.

2) Due to the significant level of ecosystem services found to be provided by prior converted croplands (e.g., regulation of pollutants), and the large area that they occupy at the landscape scale, conservation practices should be considered that directly apply to prior converted croplands. Such practices would aim to generally maintain levels of crop productivity, at least during years with suitable moisture conditions, but would seek to reduce nutrient flux to

groundwater and/or retain surface water. These practices could include promoting implementation of controlled drainage, which restores additional (although limited) wetland functions to drained agricultural landscapes.

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