Virginia Water Resources Research Center Annual Technical Report FY 2008

Introduction

The Virginia Water Resources Research Center (VWRRC) was established at Virginia Tech in 1965 as a federally authorized program. In 1982, the Virginia General Assembly authorized the VWRRC as a state agency under the Code of Virginia (§23-135.7:8).

Mission

The VWRRC provides research and educational opportunities to future water scientists; promotes research on practical solutions to water resources problems; and facilitates timely transfer of water science information to policy- and decision-makers and the general public.

Mission Elements

Research

Assisting university researchers in securing research support funds from public and private sources.

Assisting university researchers in initiating and executing water resources research.

Education

Advancing educational opportunities for students in water-resources fields by helping university researchers provide undergraduate and graduate research opportunities in water resources.

Outreach

Maintaining a publication series that synthesizes and reports on water resources science, engineering, and policy.

Securing academic advisors to work in an advisory capacity with the public and private sector.

Initiating and participating in the design and execution of conferences and symposia on Virginia, regional, and national water issues.

Program Administration

Administrative oversight is provided by the Dean of the College of Natural Resources. A Statewide Advisory Board appointed by the Governor advises the VWRRC director on state water research and information priorities. Because of its multiple legislative authorities and administrative responsibilities, the VWRRC has a number of reporting responsibilities. In addition to the annual reporting requirements to the USGS and the National Institutes for Water Resources (NIWR), it presents an annual report to the Virginia Tech administration. Five-year reports and reviews are presented to the USGS and the State Council on Higher Education for Virginia (SCHEV).

National Affiliations

The VWRRC is affiliated with NIWR and University Council on Water Resources (UCOWR).

Programs of the VWRRC

Programs are structured to meet strategic goals of the VWRRC and are consistent with the VWRRC mission as authorized by the U.S. Congress through the Water Resources Research Act of 1984, (Public Law 98-242) and Code of Virginia (§23-135.7:8). Programs in research and education are available to students and faculty at all Virginia colleges and universities. Outreach and collaborative programs include information transfer to policy/decision makers and citizens, and collaborative partnerships with state agencies and other water interest groups.

1) Research Programs

(a) The VWRRC's statewide competitive grants program provides research funds to find practical solutions to water problems in Virginia and the region. The grant period begins July 1 and ends June 30 of the following year. The review criteria include 1) technical merit of the proposed project, 2) relevance to Virginia and the region, 3) relevance to contemporary water issues, and 4) ability to provide research opportunities for graduate and undergraduate students. A priority listing of water research needs for this competitive grants program is updated annually in consultation with the VWRRC State Advisory Board. These grants are designed to initiate research efforts with a high potential for expanded funding from additional sources.

(b) The VWRRC applies for external grants and conducts in-house research.

(c) The VWRRC facilitates research team building and interdisciplinary, multi-institute collaborative research.

(d) The VWRRC facilitates research opportunities to other university faculty and external contractors through a partnership with federal agencies that provide targeted funding from the USGS.

2) Educational Programs

(a) The VWRRC provides research opportunities to undergraduate students and assistantships to graduate students who participate in sponsored research. Also, numerous graduate and undergraduate students are supported through the VWRRC's competitive grants program in Virginia Tech academic departments, and at Virginia's other colleges and universities.

(b) In 1999, the VWRRC established the William R. Walker Graduate Research Fellowship to honor the many contributions of Dr. William R. Walker, the VWRRC's first director. The $2,500 award is intended for individuals preparing for a professional career in water resources and is provided to a new graduate student each year. Details of the program can be found on the VWRRC website:
http://www.vwrrc.vt.edu/walker_fellowship.html

(c) The VWRRC coordinates the interdisciplinary Watershed Management Undergraduate Minor and a Watershed Management Graduate Certificate Program in collaboration with five colleges and ten departments at Virginia Tech.

(d) The VWRRC supports the Virginia Tech Chapter of the American Water Resources Association.

3) Outreach and Collaborative Programs

(a) The VWRRC provides administrative support for the Virginia Water Monitoring Council.

(b) The VWRRC publishes research reports, symposia proceedings and citizen education booklets. It provides funding for the publication of outreach efforts.

(c) The VWRRC publishes a quarterly newsletter, Virginia Water Central. It features scientific and educational articles, legislative information, and news of interest. The newsletter is available to the public at http://www.vwrrc.vt.edu/watercentral.html and electronic copies are provided via email to more than 500 people.

(d) The VWRRC sponsors or co-sponsors symposia, workshops, and seminars.

(e) The VWRRC facilitates peer reviews for state programs when requested.

(f) The VWRRC website (http://www.vwrrc.vt.edu/) serves as a repository of the Center's publications, houses an academic expert database, provides updated news and information relevant to water resources and manages website links for several collaborative partners including the Virginia Water Monitoring Council, the Virginia Department of Conservation and Recreation Stormwater BMP Clearinghouse, and the Clinch Powell Clean Rivers Initiative.

Research Program Introduction

Research Program

The research program of the VWRRC is supported through its Virginia state appropriation, external funding, and overhead generated by external funding. The 104 federal funds are not allocated to support research, but are used to support the outreach and information dissemination programs of the VWRRC. During FY2008, the VWRRC funded two research projects through its competitive grant program. For the USGS reporting period, funding for eight facilitated grants passed through USGS; projects were managed by the VWRRC. Basic information and resulting products are described in the following section.

Grant No. 06HQGR0189 Microtopography Effects on Vegetative and Biogeochemical Patterns in Created Wetlands: A Comparative Study to Provide Guidance for Wetland Creation and Restoration

Basic Information

Title:	Grant No. 06HQGR0189 Microtopography Effects on Vegetative and Biogeochemical Patterns in Created Wetlands: A Comparative Study to Provide Guidance for Wetland Creation and Restoration
Project Number:	2006VA105G
Start Date:	9/1/2006
End Date:	5/30/2009
Funding Source:	104G
Congressional District:	11
Research Category:	Biological Sciences
Focus Category:	Wetlands, Ecology, Hydrogeochemistry
Descriptors:	
Principal Investigators:	Changwoo Ahn, Gregory B. Noe

Publication

1. None
2. Moser, K. F. 2007. Characterization of microtopography and its influence on vegetation patterns and soil nutrients in created wetlands, M.S thesis. George Mason University.
3. Moser, K. F., C. Ahn, G. B. Noe. 2007. Characterization of microtopography and its influence on vegetation patterns in created wetlands. Wetlands 27: 1081-1097.
4. Moser, K. F., C. Ahn, G. B. Noe. 2009. The influence of microtopography on soil nutrients in created mitigation wetlands. Restoration Ecology (in press) online published.
5. Bhattarai, S, C. Ahn. 2008. Induced microtopography and its effects on the first year vegetation development in a mitigation wetland newly created in the piedmont region of Virginia, USA. Annual American Ecological Engineering Meeting Beyond Wetlands:Engineering the landscape, Virginia Tech, Blacksburg, Virginia.
6. Bhattarai, S, C. Ahn. 2008. Vegetation development patterns in wetland mitigation banks newly created in the piedmont region of Virginia, USA.. Society of Wetland Scientist 29th Annual Meeting, Capitalizing on Wetlands, International Conference, Washington, DC, USA.
7. Ahn, C., S. Bhattarai, R.M. Peralta, and K. L. Wolf. 2008. Functional assessment of compensatory wetland mitigation banks with varying design in the piedmont region of Virginia. Society of Wetland Scientist 29th Annual Meeting, Capitalizing on Wetlands, International Conference, Washington, DC, USA.

Final report for USGS –NIWR Project

Project title:
Microtopography effects on vegetative and biogeochemical patterns in created wetlands: a comparative study to provide guidance for wetland creation and restoration (2006-2008)

Changwoo Ahn, PI
4400 University Drive, MS5F2, Fairfax, Virginia 22030
Department of Environmental Science and Policy
George Mason University

** The final report consists of two manuscripts, one for vegetation patterns and the other for soil nutrients pattern as influenced by induced microtopography in created wetlands.*

1. CHARACTERIZATION OF MICROTOPOGRAPHY AND ITS INFLUENCE ON VEGETATION PATTERNS IN CREATED WETLANDS

INTRODUCTION

Microtopography, loosely defined as topographic variability on the scale of individual plants (Huenneke and Sharitz 1986, Titus 1990, Bledsoe and Shear 2000), describes soil surface variation within an elevation range from roughly one centimeter to as much as one meter, encompassing both vertical relief and surface roughness. Microtopography is included in the broader notion of topographic heterogeneity, which includes patterns of elevation at many spatial scales formed by geologic, hydrologic, physical, and biological processes (Larkin et al. 2006). Microtopography can influence wetland hydrology, physicochemistry, and habitat variability, and it is thus important in determining vegetation patterns and, ultimately, ecosystem function. Consequently, the manipulation of microtopography to promote plant community and ecosystem development has implications for wetland creation and restoration.

Created wetlands often show little evidence of ecosystem development comparable to that of their natural counterparts, and many wetlands created to mitigate wetland losses fail to meet basic success criteria within the time frame legally mandated for monitoring (National Research Council 2001, Spieles 2005). Although the legal framework may be insufficient to ensure that mitigation wetlands perform equivalent function to the wetlands they replace, identifying wetland creation methods that enhance ecosystem development might increase the probability of mitigation success, both legal and functional, thus increasing the likelihood that lost wetland ecosystem services will, in fact, be replaced. In the construction of mitigation wetlands, grading is ordinarily performed to assure surface variation within a centimeter or two of the site plan elevation, so the microtopographic variability more typical of natural settings is reduced (Stolt et al. 2000). Although not legally mandated, microtopography is sometimes adopted as a performance/monitoring criterion in compensatory mitigation since it is understood to promote floral and faunal diversity (Norfolk District Army Corps of Engineers and Virginia Department of Environmental Quality 2004). Thus microtopography is sometimes intentionally induced after wetland creation or restoration by a variety of techniques including bucket-mounding, hand-mounding, tire-rutting, and disking (or disk-harrowing).

Microtopographic relief affects the proximate hydrologic conditions experienced by an individual seed or plant (Pollock et al. 1998, Bledsoe and Shear 2000), but it may also affect wetland hydrology more broadly. Under conditions of standing water, microtopographic features may cause increased flow resistance (Harvey et al. 2003). The implication that increased microtopography enhances water retention in a wetland is supported by field experiments in which disked wetland restoration plots had higher water retention and higher water table levels than non-disked plots, whether for above or belowground water table conditions (Tweedy et al. 2001). Thus, roughing the surface (as by disking) may help in restoring wetland hydrology to agricultural lands, and it has been proposed as a way to reduce the amount of seeding needed (Bledsoe and Shear 2000).

Topographic heterogeneity on the scale of a few centimeters in relief has been shown to promote species richness and abundance in experimental wetland mesocosms (Vivian-Smith 1997). Surface variation on a similar scale also promoted differential germination of species in prepared-bed and pot experiments (Harper et al. 1965). Studies of woody seedling distributions support the importance of microtopography in determining wetland plant species distribution, with preferential establishment of species and growth forms (tree, shrub, vine) dependent on microtopographic setting (Collins et al. 1982, Huenneke and Sharitz 1986, Titus 1990). Furthermore, sedimentation has been linked to reductions in plant species richness through the loss of microtopographic features associated with *Carex* tussocks (Werner and Zedler 2002). Generally stated, processes explaining the effects of microtopography on wetland plant community structure may include: 1) water retention; 2) microsite variations in extent and frequency of inundation due to elevation; 3) propagule dispersal; 4) microsite variations in habitat (e.g., soil physicochemical properties, temperature, light penetration); 5) protection from erosion/deposition; and 6) increased surface area and exposure of soil to the atmosphere.

Because ecological phenomena may only be apparent at certain scales, it is important to recognize the significance of experimental scale; the notion of "micro"-topography itself demands that scale be considered. A proper investigation takes into account the *extent* (overall area of study) and *grain* of the study (i.e., *resolution*, the unit size of individual study plots), attempting to ensure that experimental results are not skewed by these scale-determining factors (Wiens 1989, Reed et al. 1993, Stohlgren et al. 1997). Only a few examples of multiscale microtopography studies have been published (Pollock et al. 1998, Morzaria-Luna et al. 2004), and these suggest that while there is greater variability at smaller scales, microtopographic effects are evident and consistent across scales from 0.1 m^2 to 1000 m^2.

Most ecological studies have categorized microtopography qualitatively with descriptors such as mound/pit or hummock/hollow/flat (Huenneke and Sharitz 1986, Paratley and Fahey 1986, Titus 1990, Bruland and Richardson 2005). Microtopography is difficult to measure and quantify, however, as it encompasses and combines elements of surface relief and surface roughness. Relief is the vertical extent of a topographic profile, whereas roughness is the extent of topographic variability (as opposed to smoothness), although the term roughness is also commonly used to refer to the combination of relief and roughness. Although relief can be measured and its variance quantified (Allmaras et al. 1966), it is an incomplete descriptor. Agricultural tillage studies have approached the quantification of topography formally, often in the context of erosion or depression storage, and typically at the clod or crumb scale (Romkens and Wang 1986, 1987, Potter and Zobeck 1990, Potter et al. 1990, Saleh 1993, Hansen et al. 1999, Kamphorst et al. 2000).

This study examined the effects of artificially induced microtopography on hydrologic conditions and vegetation patterns in non-tidal freshwater mitigation wetlands, with the goal of informing wetland creation practices. Several index measures were employed to quantify microtopography and separate out components of roughness and relief. A natural wetland was examined as a comparison to address how microtopography differs between created and natural wetlands. Our research hypotheses were: 1) that created and natural wetlands differ quantitatively in terms of microtopography, and that disked wetlands have greater microtopography than non-disked; 2) that increased microtopography is associated with a higher water table, and consequently with more hydrophytic vegetation; and 3) that increased microtopography is associated with greater species richness, diversity, and cover of vegetation, in both created and natural wetlands. Since disking is a method which can be used to rapidly and widely induce microtopography, the comparison of disked to non-disked created wetlands was of particular interest. Due to increased microtopography, disked wetlands were expected to have greater species richness, diversity, and plant cover, and a higher water table, than non-disked wetlands.

METHODS

Site Details

Field research was carried out at created and natural wetlands in Virginia, USA (mean annual precipitation 1085 mm, mean annual temperature min 7.0°C / max 19.3°C). Created wetlands were North Fork and Cedar Run mitigation banks in Prince William County; natural wetlands were at Huntley Meadows Park in Fairfax County (Figure 2). Within each wetland, sites were randomly selected, although for created wetlands where marked survey locations had been previously established, a survey marker was randomly selected and the study site established 3 m north of the marker.

The created wetlands were located in the Piedmont physiogeographic province, generally characterized by rolling terrain underlain by igneous and metamorphic rock, whereas the natural wetlands were in the Coastal Plain province, comparatively flat and underlain by unconsolidated sediment. North Fork Mitigation Wetland is a 125-acre wetland complex created on land formerly used as cattle pasture, graded in 1999-2000, and hydroseeded in fall 2000 and spring 2001. Study sites were located in the "Main Pod," surrounding an open water area fed by the North Fork of Broad Run, with vegetation in its fifth growing season (sites A, B, C, and D). Cedar Run Mitigation Bank is a 610-acre multiple-wetland complex developed on land formerly used for agriculture. Study sites E and F were located in Cedar Run 1, a 67-acre wetland created/graded in 2004, while sites G and H were in a portion of a smaller adjacent mitigation wetland which was graded in 2004; sites E, F, G, and H were hydroseeded in fall 2004, and were thus in their first growing season. While the mitigation projects at North Fork and Cedar Run sites E and F were disked prior to seeding to provide a more heterogeneous soil surface, the mitigation project at sites G and H was not. Owing to incomplete availability of data, direct comparison of the seed mixes actually used in the created wetlands was not possible. However, these wetlands were seeded with commercially available wetland plant seed mixes appropriate for the region and the intended hydrology (e.g., wetland meadow as opposed to obligate wetland). From the information available, these seed mixes would have included ~20 plant species, mostly within the genera *Carex* (Cyperaceae), *Juncus* (Juncaceae), and *Scirpus* (Cyperaceae). The 1425-acre Huntley Meadows Park prominently featured beaver-engineered wetlands, some of which were in existence before the park was established in 1975. Here, study sites J and L were in mature (> 30 years old) wetland, while sites I and K were in wetland adjacent to a more recently established (~10 years old) beaver pond.

Field data were collected throughout the growing season, between May and December. Each site was examined using a set of tangentially conjoined circular transects (hereafter referred to collectively as a *multiscale transect*), with field measurements and samples taken at regular intervals along the circular paths (Figure 3). The circular transect approach designed for this study was intended to be directionally unbiased; any confounding directional effects of disking orientation, wind, direction of hydrologic flows, orientation of incident sunlight, etc, were minimized. This approach covers a more limited spatial extent than do linear transects, and so reflect more localized conditions. Multiple scales were adopted to aid in identifying any scale-dependent effects; transects were laid out as 0.5 m-, 1 m-, 2 m-, and 4 m-diameter circles using crosslinked polyethylene tube hoops.

Microtopography

Field measurement of microtopography consisted of elevation measurements taken using conventional surveying equipment (Sokkia SET4110 total station). At the beginning of the study (between 23 June and 15 July), and prior to other measurements, elevations were measured at 10 cm intervals along the 0.5 m-, 1 m-, and 2 m-diameter transects (a total of 108 measurements) at each site and at 20 cm intervals along the 4 m-diameter transect (62 measurements per site) at half the sites in each wetland. Conditions for surveying were generally dry (with soil yielding minimally underfoot), although care was taken not to alter the existing microtopography during elevation measurement; likewise, throughout the study, field work was conducted as much as possible to minimize disturbance of microtopography in the vicinity of the multiscale transects. Coordinate data were recorded to the nearest millimeter, although at the distances used, the total station has nominal sub-millimeter precision for elevation (Sokkia Co. 1997). Measurement intervals were chosen as appropriate to the overall scale of interest (plant-scale), the equipment used (survey rod base diameter of ~6 cm), and the transect sizes.

Microtopography was quantified using three index measures. For a two-dimensional path, such as a cross-sectional elevation profile, the ratio of the over-surface distance to the corresponding straight-line path is referred to as “tortuosity” (Kamphorst et al. 2000), and it can either be calculated from elevation data (Werner and Zedler 2002) or measured directly (Saleh 1993, Merrill 1998). This unitless measure is a simple indicator of microtopography, sensitive to changes in both roughness and relief (Figure 1), but incapable of distinguishing a low-relief high-roughness surface (upper-right in Figure 1) from a high-relief low-roughness one (lower left in Figure 1). Although it appears to perform well for measuring the change in soil surface roughness due to rainfall

(Bertuzzi et al. 1990), tortuosity is not regarded as a good predictor for depression storage of runoff (Kamphorst et al. 2000). Tortuosity (T) was calculated using elevation data and the known transect lengths. Point-to-point distances were summed for each transect, then divided by the corresponding planar transect distance (Kamphorst et al. 2000). The method used was analogous to that of Werner and Zedler (2002), although the measurement intervals were finer (10-20 cm versus ~1 m) and transects were shorter and circular.

A geostatistical approach using a combination of limiting slope (LS) and limiting elevation difference (LD) was proposed by Linden and Van Doren (1986) to physically characterize soil surfaces. LS and LD are indices derived from the variogram of change in elevation versus the horizontal interval of measurement (lag distance). The LD index (in elevation units, cm in this study) represents the limit of elevation change approached for large intervals, thus expressing relief. It is somewhat comparable to the random roughness index of Allmaras et al. (1966) and can be used to estimate maximum depression storage (Bertuzzi et al. 1990, Kamphorst et al. 2000). The LS index (a unitless metric) represents the rate of change in elevation as the interval between measurements approaches zero, pertaining to microrelief at small sampling intervals (i.e., roughness); in tillage studies it has been correlated with tortuosity and fractal indices (Bertuzzi et al. 1990). LS and LD were adopted to distinguish roughness from relief. LS and LD were determined by mean absolute-elevation-difference analysis of the first-order variogram after correcting for slope (Linden and Van Doren 1986), treating change in elevation as a function of the distance between two points. Slope correction for elevation data was achieved by nonlinear (wave form) regression, with appropriate periodicity (i.e., 2π times the transect radius). The mean absolute elevation difference (ΔZ_h) is defined as

$$\Delta Z_h = \sum_{i=1}^{n} \left| Z_i - Z_{i+h} \right| / n$$

where:
Z_i is the slope-corrected elevation of a given point;
Z_{i+h} is the slope-corrected elevation of a point h intervals from Z_i; and
n is the number of pairs of points used in the calculation.

Linear regression was used to relate ΔZ_h to the lag distance X_h, the horizontal distance between a pair of points h intervals apart, fitting the equation

$$\Delta Z_h = 1/\left[\left(b\left(1/X_h\right)\right) + a\right]$$

and treating $1/\Delta Z_h$ as a function of $1/X_h$. LS and LD were calculated from the fitted-line parameters a and b (LS = $1/b$ and LD = $1/a$). This approach is equivalent to using Lineweaver-Burk (or double-reciprocal) plots to solve for Michaelis-Menten enzyme kinetics constants.

T, LS, and LD indices were calculated for each circular transect. For the LS and LD indices, lag intervals were considered for every point on the circular transect, with intervals continuing past the starting point on the transect as the last points on the transect were reached. Because the lag distances were chord distances, approaching as a limit the transect diameter, those used for regression differed for each scale: for the 0.5 m-diameter transects, three lag distances (10 cm, 20 cm, 30 cm as measured along the transect circle) were used for analysis; for 1 m-diameter transects, five (10-50 cm) lag distances were analyzed; for 2 m-diameter transects, ten (10-100 cm) lag distances were analyzed; and for 4 m-diameter transects, ten (20-200 cm) lag distances were analyzed. Since microtopography might vary within a circular transect, "proximal" values for T, LS, and LD indices were also calculated for each transect point based on near-neighbor points and used to express localized microtopography. These indices were proximal tortuosity (pT), proximal limiting slope (pLS), and proximal limiting elevation difference (pLD); they differ from their full-transect counterparts (T, LS, and LD) in that they are based on a small subset of points, with smaller lag intervals represented by more observations, whereas the full- transect indices were based on a larger set of elevation points with equal counts of elevation differences for each lag interval. Near-neighbor points were treated as those within 0.5 m of the point of interest, except for the case of 4 m transects, where, due to the 20 cm spacing between points, near-neighbor points were treated as within 0.6 m. In determining proximal indices, two guiding principles were applied: first, the points included should not account for more than half the data points in a circular transect; second, lag distances used should not exceed those used for transect-level indices. Consequently, the following lag distances were used: 0.5 m transects: 10-30 cm (measured along the transect circle); 1 m transects: 10-50 cm; 2 m transects: 10-60 cm; and 4 m transects: 20-60 cm. Index calculations were carried out using the mathematics application Maple version 10 (Maplesoft Inc. 2005).

Hydrology

Because installing wells/piezometers would have disrupted the surrounding microtopography, water table depth was estimated using 2.4 mm-gauge (3/32") steel welding rods (Bridgham et al. 1991). Rods were driven either to refusal or to a depth of approximately 80 cm and spaced at 80 cm intervals (total 28 measurements per multiscale transect; Figure 3). Rods were left in place for a minimum of 4 weeks, then removed and exchanged for new ones. A total of four deployments were performed beginning in June, with the final collections taking place in December. However, because sampling dates were staggered among study wetlands, there were a total of 12 sampling dates. Upon extraction, the below-surface depth beyond which no oxidation was apparent was recorded and interpreted as water table depth.

Vegetation

Macrophyte species composition and cover were sampled using 0.2 m^2 circular plots located at 160 cm intervals along each circular transect (Figure 3). Vegetation data were collected from 23 to 26 August 2005. Species were field-identified (Newcomb 1977, Brown 1979, Tiner et al. 1988) and percent cover visually estimated, with a minimum cover percentage of 1 percent. Visual estimates of less than 15% cover were reported in increments of 1%, while those of 15% or more were reported in 5% increments. Cover was also estimated for non-plant surface features, such as large rocks or logs. Due to multiple herbaceous canopy layers, the sum of species cover estimates could exceed 100%, even when visual estimate of total cover was less than 100%. Species were assigned a wetland indicator category (Reed et al. 1988, Pepin 2000).

Taxon counts for each vegetation plot, including unidentified taxa, were used to determine species richness (S) per plot, and, for multiscale transects, species richness per m^2. For the latter, taxon-sampling curves were used to derive S for n = 5 survey plots using EstimateS (Colwell 2005), based on the mean for 50 randomized runs without replacement. To characterize plant diversity, we used the Shannon diversity index (H'), which takes into account both the number of species and their relative abundances, without making assumptions about underlying distributions (Hayek and Buzas 1997, Jørgensen et al. 2005). For this study, H' was determined based on percent cover, rather than by count of individuals, similar to a method used to evaluate plant community diversity (Mitsch et al. 2005). Natural log Shannon diversity values were calculated for each sample plot and for each multiscale transect using EstimateS (Colwell 2005).

Vegetation plots were assigned a wetland prevalence index (P.I.) value according to the weighted average of indicator ranks, excluding unidentified and non-listed species (Wentworth et al. 1988). Under this classification, each wetland indicator category was assigned a rank value as follows: OBL (obligate wetland) = 1, FACW (facultative wetland) = 2, FAC (facultative) = 3, FACU (facultative upland) = 4, UPL (upland) = 5, with no adjustment for +/- designations. Rank values were weighted according to the associated percent cover, and the weighted ranks were averaged to reach an indicator rank for the sample area, with lower index values corresponding to prevalence of more hydrophytic vegetation. The prevalence index was calculated as:

$$\text{P.I.} = \frac{\sum A_i W_i}{\sum A_i}$$

where:
A_i = abundance of species i;
W_i = wetland indicator category for species i; and
i = individual species.

Statistical Analysis

Two separate parametric analyses were conducted to compare sites based on creation method (disked, non-disked, natural). First, to address how microtopography differed and to examine issues of scale, full-transect indices (LS, LD, T) were examined. Second, to address how disking relates to vegetation patterns and hydrology, proximal indices (pLS, pLD, pT) were considered in connection with vegetation parameters and steel rod oxidation measurements, using a nested design to partition out variance attributable to site. Two-way multivariate analysis of variance (MANOVA) was used to examine LS, LD, and T for differences attributable to creation method and transect scale (0.5 m, 1 m, 2 m, 4 m) for the combined dependent variable, followed by post-hoc Dunnett's T3 pairwise comparisons. A nested-design two-factor MANOVA (site nested within creation method) was used on the vegetation survey plot data to examine differences among creation methods as to the combined dependent variable of pLS, pLD, pT, H', S, P.I., percent cover, and steel rod oxidation depth, followed by post-hoc Dunnett's T3 pairwise comparisons. MANOVA analyses were conducted using Type IV sum-of-squares and an alpha level of 0.05 (due to unequal sample sizes, Pillai's Trace was adopted as a more robust alternative to Wilks' Λ), and performed using SPSS (SPSS Inc. 2004). A nested (site nested within creation method) two-factor nonparametric analysis of similarity (ANOSIM) was carried out for species assemblage data ($\alpha = 0.05$). Decomposition of the Bray-Curtis similarity used for ANOSIM was used to characterize within-site similarity and between-site dissimilarity, as well as to express the contributions of individual species to similarity/dissimilarity. ANOSIM and related routines were performed using PRIMER (PRIMER-E Ltd. 2006).

To better conform to the assumptions of MANOVA, appropriate transformations (Osborne 2002) were applied for tortuosity (T and pT, base 10 log), limiting elevation difference (LD and pLD, natural log), and wetland prevalence index (natural log). Multivariate outliers were identified by Mahalanobis distance, using the Chi-square critical value ($p < 0.001$, with df = number of dependent variables) as the criterion for exclusion of outliers from analysis. For ANOSIM of vegetation abundance data, square-root transformation was applied to the data matrix prior to Bray-Curtis ordination in order to downweight the influence of highly abundant species (Clarke and Warwick 2001, Clarke and Gorley 2006). For transformed variables, mean values reported in figures and tables are reported in original untransformed units. While the relationship between microtopographic indices and vegetation/hydrologic variables was conjectured to be monotonic (but not necessarily linear), and because the study design is observational, correlations were examined using non-parametric Spearman rank correlation coefficients ($\alpha = 0.05$) using untransformed variables, without excluding outliers.

RESULTS

Microtopography

Visual inspection of transect elevation profiles suggested empirically that disked, non-disked, and natural sites were microtopographically distinct (Figure 4), with more pronounced vertical relief evident in disked and natural wetlands than in non-disked wetlands. Circular transect microtopographic index values ranged from 0.06 to 1.7 for LS (excluding two negative values likely causing the two Mahalanobis outliers), from 0.4 to 12.4 cm for LD, and from 1.001 to 1.043 for T (Table 1). The combined dependent variable of LS, LD, and T indices differed among creation methods (Pillai's Trace = 0.460, $F_{6,54} = 2.69$, $p = 0.024$), while there were no significant differences for scale (Pillai's Trace = 0.254, $F_{9,84} = 0.86$, $p = 0.56$). Differences existed for LD ($F_{2,28} = 7.62$, $p = 0.002$) and T ($F_{2,28} = 3.47$, $p = 0.045$) indices, but not for LS ($F_{2,28} = 0.83$, $p = 0.45$). LD was significantly higher for disked ($p = 0.002$) and natural ($p = 0.026$) wetlands than for non-disked wetlands (disked [$\bar{x} = 3.4$] ≈ natural [$\bar{x} = 2.5$] > non-disked [$\bar{x} = 1.2$]). T was also higher for disked than for non-disked wetlands ($p < 0.001$), although neither differed significantly from natural wetlands (disked [$\bar{x} = 1.014$] > non-disked [$\bar{x} = 1.002$]; natural [$\bar{x} = 1.012$]). Excluding the one Mahalanobis outlier (n = 97), the plot-level combined dependent variable also differed among creation methods (Pillai's Trace = 0.924, $F_{16,158} = 8.48$, $p < 0.001$) (Figure 5), but only pLD differed significantly ($F_{2,85} = 3.88$, $p = 0.024$). Mean pLD was higher for disked than for either non-disked ($p = 0.018$) or natural wetlands ($p = 0.012$), while the latter two did not differ (disked [$\bar{x} = 3.5$] > non-disked [$\bar{x} = 2.1$] ≈ natural [$\bar{x} = 1.8$]).

Hydrology

The record of water table depths during the study period indicated that all the study sites met the legal assessment criteria for wetland hydrology (Federal Interagency Committee for Wetland Delineation 1989). Notwithstanding the drought period, the overall pattern of water table depth readings supported the notion that the study sites were hydrologically comparable, even though the created wetlands were perched, whereas the natural sites were groundwater-connected. Growing-season water table depths ranged from zero to > 69 cm, with a notable drop in depth coinciding with a period without precipitation in September (Figure 5a and b). Over the entire study period, site mean water table depths ranged from 3.4 to 29.2 cm below the surface (Table 1). However, the first two weeks of September were abnormally dry (drought severity D0), followed by three weeks of moderate drought (drought severity D1), which ended with heavy rains on 7 October (National Drought Mitigation Center 2005). Because the steel rod method is less reliable when the water table drops significantly (Bridgham et al. 1991), and since the steel rod data collection dates differed for each wetland, the steel rod oxidation depths used for analysis encompassed only those measurements taken between 19 August and 8 September, reflecting the water table for the pre-drought period (and peak growth). During this time, the mean daily precipitation for the antecedent 30 day period was comparable among study wetlands, averaging ~0.2 cm per day (Figure 5a). Steel rod oxidation depth differed by creation method ($F_{2,85} = 6.32$, $p = 0.003$), but the difference was significant only between disked and natural wetlands ($p = 0.047$, disked [$\bar{x} = 15.6$] < natural [$\bar{x} = 20.4$]; non-disked [$\bar{x} = 25.4$]). For steel rod observations across all sites ($n = 248$), no correlation was evident between rod oxidation depth and pLS ($r_{Sp} = -0.032$, $p = 0.61$), pLD ($r_{Sp} = 0.014$, $p = 0.83$), or pT ($r_{Sp} = 0.019$, $p = 0.76$). Nonetheless, the steel rod oxidation depth did correlate weakly, but positively with elevation (relative to the corresponding multiscale transect mean, $r_{Sp} = 0.16$, $p = 0.014$), validating the expectation that microtopographic high points lie higher in relation to the water table, and are thus drier.

Vegetation

Field identification of macrophytes resulted in a total count of 72 taxa, with 5 identified to genus and 60 identified to species. Accounting for a small proportion of cover were seven taxa that could not be field-identified, either because they were seedlings or because they lacked distinguishing morphologic characteristics. Twenty-seven species had average abundances exceeding 2 percent cover for at least one study location (Table 2). Although the disked sites appeared to have greater vegetation cover than non-disked sites, and although total percent cover differed by creation method ($F_{2,85} = 9.74$, $p < 0.001$), the difference between disked and non-disked sites was not significant ($p = 0.051$), although disked and natural sites differed ($p = 0.016$, disked [$\bar{x} = 125$] > natural [$\bar{x} = 103$]; non-disked [$\bar{x} = 84$]). Geographically, species richness (S) was highest for Cedar Run (42 species total, 30 for disked and 19 for non-disked sites), followed by North Fork (31 species) and Huntley Meadows (26 species). S ranged from 8 to 22.2 species among multiscale transects (Table 1). Considering survey plots across all sites ($n = 106$), S correlated with both pT ($r_{Sp} = 0.208$, $p = 0.032$) and pLD ($r_{Sp} = 0.235$, $p = 0.015$). Within Cedar Run ($n = 34$), the correlations were stronger, although again the correlation for pLD ($r_{Sp} = 0.533$, $p = 0.001$) was stronger than that for pT ($r_{Sp} = 0.424$, $p = 0.013$). Plot-level species richness differed by creation method ($F_{2,85} = 23.89$,

$p < 0.001$) and was higher for disked plots than for non-disked ($p = 0.009$) and natural ($p < 0.001$) plots (disked [$\bar{x} = 7.9$] > natural [$\bar{x} = 5.0$] ≈ non-disked [$\bar{x} = 4.8$]).

Plot-level Shannon diversity index (H') values ranged from zero to 2.13, while transect-level values ranged from zero to 2.56. Because the Shannon index increases with sampling effort (Hayek and Buzas 1997), transect-level values could not be compared across different scales. At the sample plot level, H' differed by creation method ($F_{2,85} = 19.01$, $p < 0.001$), with significant differences among all methods (disked [$\bar{x} = 1.38$] > natural [$\bar{x} = 0.96$] > non-disked [$\bar{x} = 0.72$]). Across all survey plots (n = 106), H' was significantly correlated with both pLD ($r_{Sp} = 0.32$, $p = 0.001$) and pT ($r_{Sp} = 0.31$, $p = 0.001$), although not with pLS ($r_{Sp} = -0.064$, $p = 0.51$). These general correlations were not observed consistently. While they were evident for Cedar Run for both pLD ($r_{Sp} = 0.57$, $p < 0.001$) and pT ($r_{Sp} = 0.45$, $p = 0.007$), they were not for North Fork (pLD $r_{Sp} = -0.15$, $p = 0.38$; pT $r_{Sp} = 0.10$, $p = 0.55$). Considering disked and non-disked created wetland survey plots as a pooled group (n = 70), H' was positively correlated with both pLD ($r_{Sp} = 0.27$, $p = 0.022$) and pT ($r_{Sp} = 0.30$, $p = 0.013$). At the natural wetland survey plots (n = 36), H' correlated with pT ($r_{Sp} = 0.33$, $p = 0.047$), but not pLD ($r_{Sp} = 0.28$, $p = 0.098$).

Although water table depth could affect S and H', particularly where conditions are relatively constant (e.g., inundation), and although it should largely determine the wetland prevalence index (P.I.), the steel rod oxidation depth was not correlated with S ($p = 0.68$), H' ($p = 0.87$), or P.I. ($p = 0.23$). The wetland prevalence index ranged from 1.0 to 4.0 (Table 1), although most were below 2.5, thus within the wetland vegetation range. An exception was site H at Cedar Run, where the vegetation was markedly different from that observed at other sites, with prevalence of non-hydrophytic vegetation (P.I. = 3.6, or FACU) and low percent cover ($\bar{x} = 44\%$). Prevalence indices differed by creation method ($F_{2,85} = 24.92$, $p < 0.001$), where disked and natural wetland plots had significantly lower P.I. values (i.e., prevalence of more hydrophytic vegetation) than non-disked wetland plots (disked [$\bar{x} = 1.4$] ≈ natural [$\bar{x} = 1.4$] < non-disked [$\bar{x} = 2.6$]). Even though hydrology should largely determine the prevalence of hydrophytes, the steel rod oxidation depth difference between disked and non-disked plots ($p = 0.099$, mean difference of 9.8 cm) appeared insufficient to explain the large difference in P.I. (a full indicator category, OBL versus FACW/FAC). The equivalence of P.I. between disked and natural wetlands suggested that their differing steel rod oxidation depths ($p = 0.047$, mean difference of 3.8 cm) did not affect the prevalence of hydrophytes.

Creation methods differed in community composition (Global R = 0.715, $p = 0.002$), while significant assemblage differences were also attributable to site (Global R = 0.634, $p = 0.001$). Pairwise comparisons showed that disked and natural wetlands differed (R = 0.921, $p = 0.005$), but that disked and non-disked wetlands did not (R = 0.396, $p = 0.11$). Although the test for difference between non-disked and natural wetlands was not significant (R = 0.786, $p = 0.067$), it likely reflects the small number of non-disked replicates. Clarke and Gorley (2006) emphasize that the R statistic is more important for interpretation than is the p-value when the number of replicates is small; the large R statistic here suggests significant differences between non-disked and natural sites.

Decomposition of Bray-Curtis similarity showed that the within-site similarity between samples was generally higher (i.e., greater homogeneity) for sites with less microtopography (Table 1). There were marked contrasts in within-site similarity between disked sites (E and F) and non-disked sites (G and H) at Cedar Run, and between beaver pond sites (I and K) and mature wetland sites (J and L) at Huntley Meadows. Decomposition of similarity percentages by species suggested that four common species were important contributors to within-site similarity (Table 3), as well as to difference between sites: barnyardgrass (*Echinochloa crus-galli* (L.) Beauv.), blunt spikerush (*Eleocharis obtusa* (Willd.) Schult.), rice cutgrass (*Leersia oryzoides* (L.) Sw.), and marsh seedbox (*Ludwigia palustris* (L.) Ell.). *Echinochloa crus-galli*, an annual graminoid often found in association with *E. obtusa* and *L. palustris* (Pepin 2000), was abundant at Cedar Run, and was the overwhelming component of cover observed in the non-disked wetland (and at site G).

DISCUSSION

Microtopography in Created and Natural Wetlands

The range of values obtained for T (Table 1) fell within a range overlapping that obtained by Werner and Zedler (2002) for *Phalaris*- and *Typha*-dominated wetlands (1.00 to 1.02), but considerably lower than those for *Carex*-dominated wetlands (1.06 to 1.16) (although the methods used in this study differed, particularly in terms of the interval of measurement). For created wetlands, the microtopography of disked sites differed from that of non-disked sites in terms of both tortuosity and relief, confirming our hypothesis that disked microtopography is greater than non-disked. The distinction was particularly apparent for relief; whereas disked sites had LD greater than 3 cm, non-disked sites had LD of 2 cm or less. Disked LD exceeded the relief of heterogeneous experimental treatments (Vivian-Smith 1997), while non-disked LD approached the condition of homogeneous treatments in that study. Disked relief is thus sufficient to affect the frequency and spatial variation of flooding (Pollock et al. 1998, Bledsoe and Shear 2000).

Although disked and non-disked microtopography clearly differed, created and natural microtopography did not, in contrast to our hypothesis that created and natural wetlands would differ quantitatively. While natural LD was similar to disked LD and larger than non-disked LD, natural pLD was similar to non-disked pLD and less than disked pLD, suggesting that natural microtopographic relief encompasses the range of relief found in both disked and non-disked created wetlands. Although this finding contrasts with that of Stolt et al. (2000), at the comparatively small scale of this investigation, the distinction between created and natural microtopography may be subtle. At our study's scale, disked relief was comparable to the high end of the relief found in natural wetlands, while non-disked relief fell at the low end.

Wetland microtopography has typically been examined at resolutions (or grain sizes) on the order of meters or square meters and/or spatial extents greater than 10 m or 100 m^2 (Pollock et al. 1998, Bledsoe and Shear 2000, Stolt et al. 2000, Werner and Zedler 2002, Bruland and Richardson 2005). The present study examines wetland microtopography at grain sizes of 10-20 cm (or 0.2 m^2 for vegetation plots) and spatial extent 4 m or less (~12.5 m^2), extents comparable to those of Morzaria-Luna et al. (2004), with the smaller transects comparable in extent to the experiment of Vivian-Smith (1997). At our study's resolution, wetland microtopography differed minimally between extents of 0.5 m to 4 m, a result echoing that of Morzaria-Luna et al. (2004).

This finding validates the use of the proximal indices pT and pLD, since these indices were calculated based on near-neighbor elevations. It also suggests that mesocosm-scale experiments in microtopography might be extrapolated at least as far as the 4 m transect spatial extent. At finer grain sizes (e.g., seed-scale as opposed to plant-scale), this may not necessarily be the case. Moreover, at larger spatial extents, broader-scale patterns in microtopography (such as hummock/hollow) may be more important.

Measures of Microtopography

Although LD and LS were adopted to quantify relief and roughness separately, LD proved more useful than LS. The LS and pLS indices failed to distinguish the study sites (Table 1, Figure 6c). Some of the regressions had negative slopes, resulting in negative (uninterpretable) LS values, implying a non-zero mean absolute elevation difference (ΔZ) as the lag distance approaches zero. In Lineweaver-Burk linear regression, however, such a result can occur when the smallest-interval ΔZ value exceeds those of larger intervals. While this provides qualitative information (i.e., microtopographic roughness more apparent at small intervals than at larger ones), it suggests caution in interpreting LS as a physical parameter, supporting the contention that LS only describes variogram slope, not surface slope (Kamphorst et al. 2000). Several differences in method may explain why LS and pLS results appeared less robust than those of Linden and Van Doren (1986). First, elevation data were collected along circular transects, rather than in oriented grids; second, the smallest interval used was 10 cm, as opposed to 5 cm. For the larger transects, the largest lag intervals exceeded those used in the original method. Moreover, the proximal indices were derived from a small number of elevation measurements. Whereas the original study reported that most regressions had close fits to the Lineweaver-Burk plots ($R^2 > 0.90$), in our study only half the regressions used to calculate LS and LD had comparable fit, and about a third had rather poor fit ($R^2 < 0.50$).

LD appeared to perform reasonably well as a measure of relief, with values appropriate to the respective elevation profiles (Figure 4); pLD values were clustered about their respective transect-level LD index values and produced few univariate outliers. LD and pLD were thus useful in characterizing microtopography, although they were computationally intensive. A simpler measure of relief, such as random roughness (Allmaras et al. 1966) might be more appropriate for future studies. Quantification of relief is essential, however; consideration of tortuosity alone would suggest that disked sites C and D were similar to non-disked sites G and H (Figure 6a), whereas these sites differed in relief, measured as pLD (Figure 6b).

Hydrology

The correlation between relative elevation and steel rod oxidation depth suggests that relief should affect proximate hydrologic conditions, but the weakness of the correlation may reflect steel rod oxidation depth variability. Indeed, steel rod depth standard deviations more than doubled those of elevation. However, the inferred variability in water table depth may reflect redoximorphic conditions independent of water table depth (e.g., soil texture, compaction, or organic/microbial content). As used in this study, the steel rod oxidation method had drawbacks, particularly for fine-scale measurement and comparison.

The lack of correlation between steel rod oxidation depth and any of the index measures of microtopography contrasts to the findings of Tweedy et al. (2001) relating higher water table to microtopography, as well as with our corresponding hypothesis. During the growing season, increased water retention may have been offset by increased evapotranspiration, thus masking microtopographic effects. Alternatively, steel rod oxidation may have been too coarse an approach to establish a meaningful correlation with water table depth, which might only vary on the order of 10 cm due to microtopography (Tweedy et al. 2001). Indeed, while the steel rod oxidation depth differed significantly among creation methods, the mean depths were within a 10 cm range. Moreover, since the steel rod analysis only covers a relatively brief period in late summer, it can not reflect seasonal aspects of the hydrology.

Microrelief appeared to increase water retention by storing water in small depressions. Rarely was standing water observed in the low-relief non-disked sites; when present, it was of less than 2 cm depth (Moser, personal observation). In contrast, standing water of several centimeters depth was frequently observed at disked and natural sites, suggesting that microrelief can affect hydroperiod, increasing inundation stress on plants and germinating seeds. The prevalence of hydrophytic vegetation may thus depend more on ephemeral inundation by perched pools than on water table depth.

Vegetation

The observed association between microtopography and both species richness and Shannon diversity in created wetlands confirms the notion that inducing microtopographic heterogeneity in created wetlands promotes diversity (Vivian-Smith 1997, Bruland and Richardson 2005, Larkin et al. 2006). Furthermore, this association mirrored the patterns observed in natural wetlands in our study and in others, (Huenneke and Sharitz 1986, Titus 1990, Werner and Zedler 2002), supporting our hypothesis that increased microtopography is associated with greater species richness and diversity.

Physiogeographic setting, as well as seed source, may explain some of the differences between created and natural wetland assemblages. The higher-elevation clay loams of the Piedmont likely support a vegetation community different from that supported by the lower, sandier soils of the Coastal Plain. Moreover, the even and abundant supply of seed provided for wetland creation contrasts with the spatially variable, population-dependent seed source and distribution in natural wetlands, possibly explaining the richness and diversity in the created wetlands. Spatial variability was more evident at Huntley Meadows, where numerous additional species were observed in the vicinity, whereas the created wetlands lacked such broader-scale diversity. Furthermore, since the created wetlands were in comparatively early successional stages, their plant communities may include species that will not persist in the long term.

A species-area relationship has been suggested as potentially explaining increased species richness with increased microtopography in tussock sedge meadows (Werner and Zedler 2002, Peach and Zedler 2006). In our study, vegetation effects (e.g., tussock effects) were not confounded with microtopographic effects (the Cedar Run sites had no pre-existing biogenic microtopography). Since tortuosity is the 2-dimensional analogue of surface area, the correlation between species richness and pT may support a species-area relationship. However, surface area may only reflect habitat heterogeneity, rather than being an influence itself (Brose 2001). As a measure of relief, the pLD index should more closely reflect habitat heterogeneity than does pT, whether considered in terms of hydrology (Pollock et al. 1998, Bledsoe and Shear 2000) or other factors, such as light penetration (Peach and Zedler 2006). At Cedar Run, the strength of correlation with pLD better supports the hypothesis that species richness is promoted by habitat heterogeneity than by a richness-area relationship.

The four species accounting for most of the assemblage similarities are generalists common in wetland plant communities in Virginia and highly tolerant of disturbance (Virginia FQAI Advisory Committee 2004). The distribution and abundance of these generalists within a site's plant community was important in distinguishing among assemblages. These assemblages varied less where microtopography was limited. It thus appears that increased microtopography reduces the importance of generalists and fosters the establishment of non-generalists, as would be expected through niche differentiation. It should also increase the evenness of species distribution, suggested in part by the correlations between H' and both pT and pLD. The higher within-site assemblage similarity for non-disked sites further supports the notion that decreased microtopography is associated with species dominance (Werner and Zedler 2002, Larkin et al. 2006).

CONCLUSIONS

As a practical consequence of engineering practices, created wetland ecosystems are relatively uniform at the outset, in contrast to natural wetland conditions. An area of concern for mitigation is the extent to which this uniformity may lead to the predominance of few species, diminishing ecosystem functions. Our study showed that disking clearly enhanced microtopography in created wetlands and the increased microtopography was associated with greater species richness, diversity, and percent cover, as well as with the prevalence of hydrophytic vegetation. However, it may represent topographic uniformity when considered at the full extent of a created wetland. Disked microtopography was thus qualitatively different from that induced by excavation (hummock/hollow or mound/pit), which provides greater magnitude of relief but is typically applied over a proportionally smaller area. Disking affects vegetation throughout a wetland, whereas hummocks/hollow creation yields localized benefits (e.g., pools of standing water, patches of vegetation) which may be more relevant to wetland fauna.

Disking appears to prevent the dominance of generalist species, some of which may be undesirable species in mitigation wetlands. Where generalist species were associated with the loss of microtopographic features and biodiversity, even the short-term plant community effects of disking, apparent in this study for Cedar Run, might help guarantee longer-term plant species richness and diversity. In terms of mitigation performance criteria (i.e. legal success), the disked sites clearly had the better prospects. The non-disked site H failed the basic performance criterion of prevalence of hydrophytic vegetation. Disking is therefore recommended as a relatively low-cost method of inducing microtopographic variation that could assist ecosystem development in created mitigation wetlands.

LITERATURE CITED

Allmaras, R. R., R. E. Burwell, W. E. Larson, and R. F. Holt. 1966. Total porosity and random roughness of the interrow zone as influenced by tillage. USDA Conservation Research Report no. 7.

Bertuzzi, P., G. Rauws, and D. Courault. 1990. Testing roughness indexes to estimate soil surface-roughness changes due to simulated rainfall. Soil & Tillage Research 17:87–99.

Bledsoe, B. P. and T. H. Shear. 2000. Vegetation along hydrologic and edaphic gradients in a North Carolina coastal plain creek bottom and implications for restoration. Wetlands 20:126–147.

Bridgham, S. D., S. P. Faulkner, and C. J. Richardson. 1991. Steel rod oxidation as a hydrologic indicator in wetland soils. Soil Science Society of America Journal 55:856–862.

Brose, U. 2001. Relative importance of isolation, area and habitat heterogeneity for vascular plant species richness of temporary wetlands in east-German farmland. Ecography 24:722–730.

Brown, L. 1979. Grasses, an Identification Guide. Houghton Mifflin, Boston, MA, USA.

Bruland, G. L. and C. J. Richardson. 2005. Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. Restoration Ecology 13:515–523.

Clarke, K. R. and R. N. Gorley. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E Ltd, Plymouth, UK.

Clarke, K. R. and R. M. Warwick. 2001. Change in Marine Communities: an Approach to Statistical Analysis and Interpretation, 2nd edition. PRIMER-E Ltd., Plymouth, UK.

Collins, S. L., J. V. Perino, and J. L. Vankat. 1982. Woody vegetation and microtopography in the bog meadow association of Cedar Bog, a west-central Ohio fen. American Midland Naturalist 108:245–249.

Colwell, R. K. 2005. EstimateS: Statistical Estimation of Species Richness and Shared Species from Samples. Software and user's guide, version 7.5, http://purl.oclc.org/estimates.

Federal Interagency Committee for Wetland Delineation. 1989. Federal manual for identifying and delineating jurisdictional wetlands. US Army Corps of Engineers, US Environmental Protection Agency, US Fish and Wildlife Service, USDA Soil Conservation Service, Washington, DC, USA.

Hansen, B., P. Schjonning, and E. Sibbesen. 1999. Roughness indices for estimation of depression storage capacity of tilled soil surfaces. Soil & Tillage Research 52:103–111.

Harper, J. L., J. T. Williams, and G. R. Sagar. 1965. The heterogeneity of soil surfaces and its role in determining the establishment of plants from seed. Journal of Ecology 53:273–286.

Harvey, J. W., J. T. Newlin, and J. Choi. 2003. Characterization of microtopography in the Everglades. p. 37–38. *In* G. R. Best (ed.) U.S. Geological Survey Greater Everglades Science Program: 2002 Biennial Report (open-file report 03-54). USGS, Tallahassee, FL, USA.

Hayek, L. C. and M. A. Buzas. 1997. Surveying Natural Populations. Columbia University Press, New York, NY, USA.

Huenneke, L. F. and R. R. Sharitz. 1986. Microsite abundance and distribution of woody seedlings in a South Carolina cypress-tupelo swamp. American Midland Naturalist 115:328–335.

Jørgensen, S. E., R. Costanza, and F.-L. Xu, (eds.). 2005. Handbook of Ecological Indicators for Assessment of Ecosystem Health. Taylor & Francis, Boca Raton, FL, USA.

Kamphorst, E. C., V. Jetten, J. Guerif, J. Pitkanen, B. V. Iversen, J. T. Douglas, and A. Paz. 2000. Predicting depressional storage from soil surface roughness. Soil Science Society of America Journal 64:1749–1758.

Larkin, D., G. Vivian-Smith, and J. B. Zedler. 2006. Topographic heterogeneity theory and ecological restoration. p. 142–164. *In* D. A. Falk, M. A. Palmer, and J. B. Zedler (eds.) Foundations of restoration ecology. Island Press, Washington, DC, USA.

Linden, D. R. and D. M. Van Doren. 1986. Parameters for characterizing tillage-induced soil surface-roughness. Soil Science Society of America Journal 50:1560–1565.

Maplesoft Inc. 2005. Maple 10. Waterloo Maple Inc., Waterloo, Canada.

Merrill, S. D. 1998. Comments on the chain method for measuring soil surface roughness: Use of the chain set. Soil Science Society of America Journal 62:1147–1149.

Mitsch, W. J., N. Wang, L. Zhang, R. Deal, X. Wu, and A. Zuwerink. 2005. Using ecological indicators in a whole-ecosystem wetland experiment. p. 213–237. *In* S. E. Jørgensen, R.

Costanza, and F.-L. Xu (eds.) Handbook of Ecological Indicators for Assessment of Ecosystem Health. Taylor & Francis, Boca Raton, FL, USA.

Morzaria-Luna, H., J. C. Callaway, G. Sullivan, and J. B. Zedler. 2004. Relationship between topographic heterogeneity and vegetation patterns in a Californian salt marsh. Journal of Vegetation Science 15:523–530.

National Drought Mitigation Center. 2005. U.S. Drought Monitor (accessed November 2006, available at http://drought.unl.edu/dm/).

National Research Council. 2001. Compensating for wetland losses under the Clean Water Act. National Academy Press, Washington, DC, USA.

Newcomb, L. 1977. Newcomb's Wildflower Guide : an Ingenious New Key System for Quick, Positive Field Identification of the Wildflowers, Flowering Shrubs and Vines of Northeastern and North Central North America, 1st edition. Little Brown, Boston, MA, USA.

Norfolk District Army Corps of Engineers and Virginia Department of Environmental Quality. 2004. Recommendations for wetland compensatory mitigation: including site design, permit conditions, performance and monitoring criteria (accessed 7/29/2006 as http://www.deq.state.va.us/wetlands/pdf/mitigationrecommendabbrevjuly2004.pdf).

Osborne, J. W. 2002. Notes on the use of data transformations. Practical Assessment, Research and Evaluation 8(6):1–9.

Paratley, R. D. and T. J. Fahey. 1986. Vegetation – environment relations in a conifer swamp in central New York. Bulletin of the Torrey Botanical Club 113:357–371.

Peach, M. and J. B. Zedler. 2006. How tussocks structure sedge meadow vegetation. Wetlands 26:322–335.

Pepin, A. L. 2000. Correction of indicator status for *Echinochloa crusgalli* (Barnyard Grass). Virginia Association of Wetland Professionals Update 7:4–5.

Pollock, M. M., R. J. Naiman, and T. A. Hanley. 1998. Plant species richness in riparian wetlands – A test of biodiversity theory. Ecology 79:94–105.

Potter, K. N. and T. M. Zobeck. 1990. Estimation of soil microrelief. Transactions of the ASAE 33:156–161.

Potter, K. N., T. M. Zobeck, and L. J. Hagan. 1990. A microrelief index to estimate soil erodibility by wind. Transactions of the ASAE 33:151–155.

PRIMER-E Ltd. 2006. PRIMER v6.1. PRIMER-E Ltd., Plymouth, UK.

Reed, P. B., National Wetlands Inventory (U.S.), U.S. Fish and Wildlife Service., United States. National Interagency Review Panel., and United States. Regional Interagency Review Panel: Northeast (Region 1). 1988. National list of plant species that occur in wetlands. Northeast (Region 1). U.S. Dept. of the Interior, Fish and Wildlife Service, Research and Development, Washington, DC, USA.

Reed, R. A., R. K. Peet, M. W. Palmer, and P. S. White. 1993. Scale dependence of vegetation-environment correlations – a case-study of a North Carolina piedmont woodland. Journal of Vegetation Science 4:329–340.

Romkens, M. J. M. and J. Y. Wang. 1986. Effect of tillage on surface-roughness. Transactions of the ASAE 29:429–433.

Romkens, M. J. M. and J. Y. Wang. 1987. Soil roughness changes from rainfall. Transactions of the ASAE 30:101–107.

Saleh, A. 1993. Soil roughness measurement – chain method. Journal of Soil and Water Conservation 48:527–529.

Sokkia Co. 1997. SET2110/SET3110/SET4110 Electronic Total Station : Operator's Manual. Sokkia Co., Ltd., Tokyo, Japan.

Spieles, D. J. 2005. Vegetation development in created, restored, and enhanced mitigation wetland banks of the United States. Wetlands 25:51–63.

SPSS Inc. 2004. SPSS 13.0 for Windows graduate student version. SPSS, Inc., Chicago, IL, USA.

Stohlgren, T. J., G. W. Chong, M. A. Kalkhan, and L. D. Schell. 1997. Multiscale sampling of plant diversity: Effects of minimum mapping unit size. Ecological Applications 7:1064–1074.

Stolt, M. H., M. H. Genthner, W. L. Daniels, V. A. Groover, S. Nagle, and K. C. Haering. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands 20:671–683.

Tiner, R. W., A. Rorer, R. H. Wiegand, Maryland. Dept. of Natural Resources., and U.S. Fish and Wildlife Service. 1988. Field Guide to Nontidal Wetland Identification. Maryland Dept. of Natural Resources Water Resources Administration ; U.S. Dept. of the Interior Fish and Wildlife Service, Annapolis, MD & Newton Corner, MA, USA.

Titus, J. H. 1990. Microtopography and woody plant regeneration in a hardwood floodplain swamp in Florida. Bulletin of the Torrey Botanical Club 117:429–437.

Tweedy, K. L., E. Scherrer, R. O. Evans, and T. H. Shear. 2001. Influence of microtopography on restored hydrology and other wetland functions (Meeting Paper No. 01-2061). *in* 2001 American Society of Agricultural Engineers Annual International Meeting. ASAE, St. Joseph, MI, USA.

Virginia FQAI Advisory Committee. 2004. Virginia wetlands plants C-value list (draft, accessed 7/26/2006 as http://www.deq.virginia.gov/wetlands/pdf/virginiacvaluescompleterev1.pdf). Commonwealth of Virginia, Office of Wetlands, Water Protection Compliance.

Vivian-Smith, G. 1997. Microtopographic heterogeneity and floristic diversity in experimental wetland communities. Journal of Ecology 85:71–82.

Wentworth, T. R., G. P. Johnson, and R. L. Kologiski. 1988. Designation of wetlands by weighted averages of vegetation data – a preliminary evaluation. Water Resources Bulletin 24:389–396.

Werner, K. J. and J. B. Zedler. 2002. How sedge meadow soils, microtopography, and vegetation respond to sedimentation. Wetlands 22:451–466.

Wiens, J. A. 1989. Spatial Scaling in Ecology. Functional Ecology 3:385–397.

Table 1. Tortuosity (T), limiting slope (LS), and limiting elevation difference (LD, cm) for each transect scale; mean water table depth (WTD ± SE, cm), mean percent cover (%Cover ± SE), mean wetland prevalence index (P.I. ± SE) and corresponding category, species richness (S_{obs}) as estimated from taxon-sampling curves for n = 5 samples (1 m^2), 50 randomized runs; mean Shannon diversity index (H' ± SE); percent within-site similarity as determined from decomposition of average within-group Bray-Curtis similarity. For LS index, "neg" indicates negative/uninterpretable values.

		North Fork (disked)				Cedar Run (disked)		Cedar Run (non-disked)		Huntley Meadows (natural)			
	Scale	A	B	C	D	E	F	G	H	I	J	K	L
T	0.5m	1.011	1.016	1.005	1.001	1.023	1.012	1.002	1.001	1.043	1.003	1.001	1.035
	1m	1.020	1.026	1.010	1.002	1.013	1.021	1.004	1.002	1.005	1.006	1.001	1.012
	2m	1.013	1.030	1.005	1.004	1.020	1.017	1.008	1.002	1.018	1.006	1.004	1.015
	4m	1.015	1.012	--	--	1.011	--	1.004	--	1.023	1.004	--	--
LS	0.5m	0.259	0.351	0.086	0.168	0.180	1.707	0.928	0.083	neg	0.086	neg	0.321
	1m	0.227	0.249	0.328	0.062	0.152	0.260	0.156	0.117	0.104	0.209	0.084	0.254
	2m	0.209	0.524	0.088	0.070	0.248	0.203	0.247	0.074	0.234	0.109	0.132	0.188
	4m	0.471	0.725	--	--	0.220	--	0.173	--	0.352	0.145	--	--
LD	0.5m	2.0	2.2	4.7	0.5	12.4	1.5	0.6	0.5	1.8	2.1	0.4	4.3
	1m	4.6	5.4	1.7	1.9	6.0	4.9	1.2	0.8	3.2	1.5	0.9	2.0
	2m	2.8	3.4	3.7	4.2	4.0	5.0	1.6	1.6	3.2	2.4	1.3	3.3
	4m	3.6	3.2	--	--	5.1	--	2.1	--	4.4	2.1	--	--
WTD		19.2±1.1	18.9±1.1	3.4±0.5	12.9±1.3	18.9±1.2	21.6±2.0	20.0±1.7	27.5±4.3	16.8±1.4	29.2±2.5	11.4±2.0	18.1±1.9
%Cover		129 ± 9	90 ± 14	114 ± 11	166 ± 12	132 ± 8	128 ±12	109 ± 4	44 ± 8	120 ± 4	87 ± 6	107 ± 4	101 ± 8
P.I.		1.4±0.1	1.7±0.1	1.3±0.2	1.3±0.1	1.7±0.1	1.4±0.1	2.0±0.1	3.6±0.2	2.0±0.1	1.1±0.1	1.4±0.2	1.6±0.1
		OBL	FACW	OBL	OBL	FACW	OBL	FACW	FACU	FACW	OBL	OBL	FACW
S_{obs}		11.2	19.5	18	19.2	22.2	16	14.4	10	9.3	13.3	8	14
H'		1.01±0.08	1.54±0.10	1.48±0.21	1.52±0.11	1.68±0.10	1.21±0.08	0.59±0.10	1.06±0.14	0.90±0.08	1.11±0.12	0.65±0.15	1.03±0.17
%Similarity		53	35	41	50	39	52	68	60	58	47	63	35

Table 2. Percent cover and wetland indicator category (Reed 1988) for common species (> 2% average cover at any location). Percent cover totals may exceed 100% due to multiple layers of cover. Mean ± one SE.

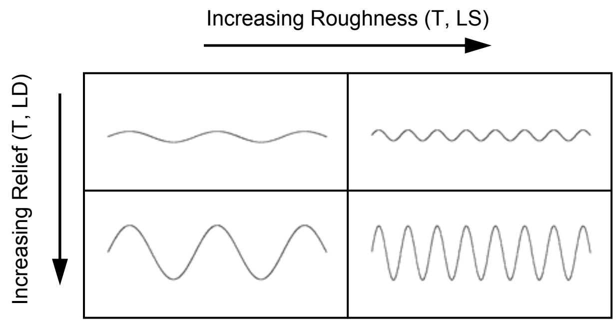
SPECIES	Indicator[1]	North Fork (disked)	Cedar Run (disked)	Cedar Run (non-disked)	Huntley Meadows (natural)
Alisma plantago-aquatica L.	OBL	1 ± 1	5 ± 3	0	0
Ambrosia artemisiifolia L.	FACU	0	0	4 ± 2	0
Bidens cernua L.	OBL	7 ± 3	1 ± 1	0	0
Carex frankii Kunth	OBL	10 ± 4	0	0	0
Carex lurida Wahlenb.	OBL	1 ± 1	0	0	4 ± 1.7
Carex vulpinoidea Michx.	OBL	6 ± 3	0	0	0
Carex sp.	--	0	3 ± 1	1 ± 0	0.1 ± 0.04
Cyperus strigosus L.	FACW	0	11 ± 3	0	0
Diodia virginiana L.	FACW	0	4 ± 1	0.1 ± 0.1	0
Echinochloa crus-galli (L.) Beauv.	FACW–[2]	22 ± 4	19 ± 7	61 ± 10	3 ± 1.9
Eleocharis obtusa (Willd.) Schult.	OBL	28 ± 5	29 ± 7	6 ± 3	1 ± 0.8
Juncus effusus L.	FACW+	3 ± 1	2 ± 1	0	6 ± 2.8
Juncus tenuis Willd.	FAC–	9 ± 3	10 ± 3	0	0
Leersia oryzoides (L.) Sw.	OBL	1 ± 0	17 ± 6	0	30 ± 5.0
Lindernia dubia (L.) Pennell	OBL	0	3 ± 1	0.3 ± 0.1	0.1 ± 0.1
Ludwigia alternifolia L.	FACW+	6 ± 3	2 ± 1	0	0
Ludwigia palustris (L.) Ell.	OBL	18 ± 4	12 ± 6	1 ± 0	1 ± 0.5
Microstegium vimineum (Trin.) A. Camus	FAC	0	0	0	24 ± 5.5
Panicum virgatum L.	FAC	0.3 ± 0.3	4 ± 2	6 ± 4	0
Polygonum hydropiper L.	OBL	4 ± 2	3 ± 2	0.1 ± 0.1	0
Polygonum punctatum Ell.	OBL	0.2 ± 0.1	1 ± 1	0	5 ± 2.3
Polygonum sagittatum L.	OBL	0.1 ± 0.1	0	0	4 ± 1.8
Saururus cernuus L.	OBL	0	0	0	12 ± 4.0
Scirpus atrovirens Willd.	OBL	2 ± 2	0	0	0
Scirpus cyperinus (L.) Kunth	FACW+	0	0	0	4 ± 2.9
Setaria glauca (L.) Beauv.	FAC	0	5 ± 2	0	0
Setaria viridis (L.) Beauv.	NL	0	0	6 ± 4	0

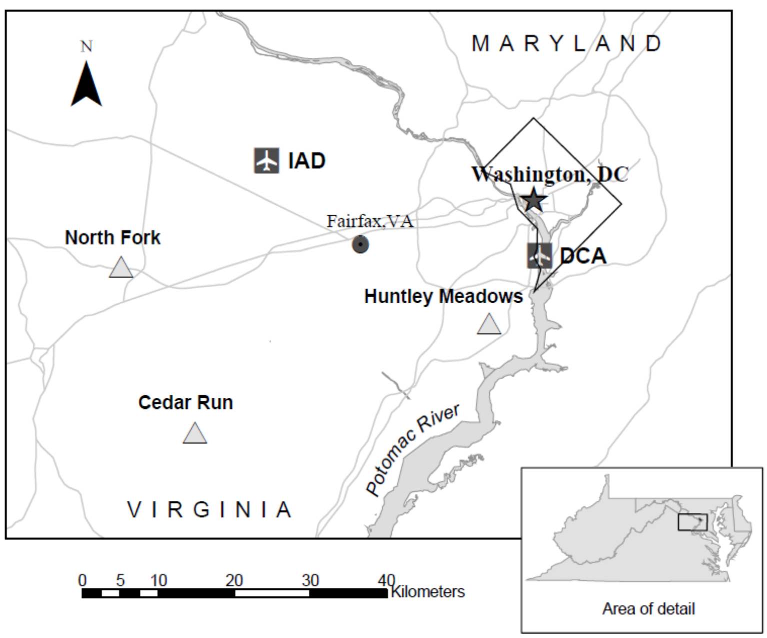
[1]OBL = obligate wetland; FACW = facultative wetland; FAC = facultative; FACU = facultative upland; NL = not listed. +/– indicates more/less frequently found in wetlands for a given indicator category.
[2]Indicator category reflects corrected status (Pepin 2000) for *E. crus-galli*.

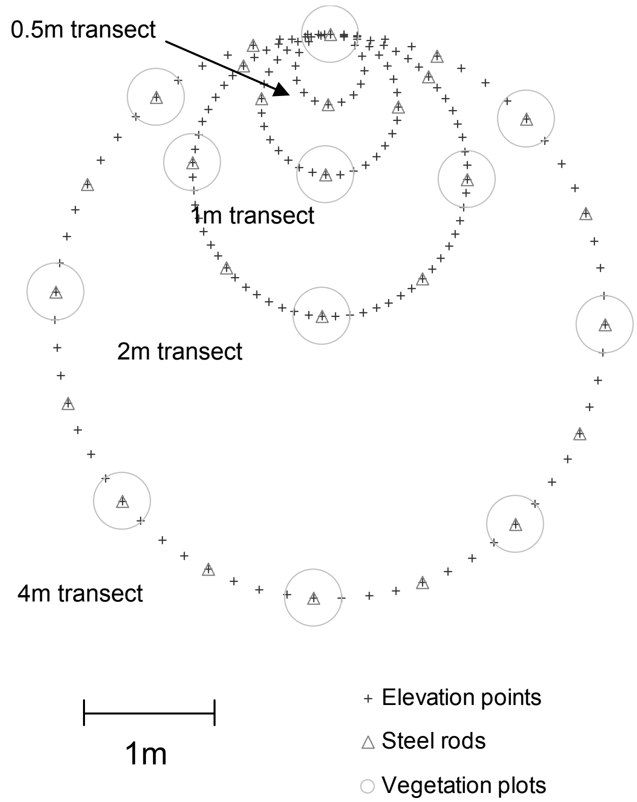
Table 3. Percent contribution to within-site similarity (from ANOSIM) for the four major contributors to similarity: barnyardgrass (*E. crus-galli*), blunt spikerush (*E. obtusa*), rice cutgrass (*L. oryzoides*), and marsh seedbox (*L. palustris*). Also given are overall mean percent and percentages for created and natural wetlands.

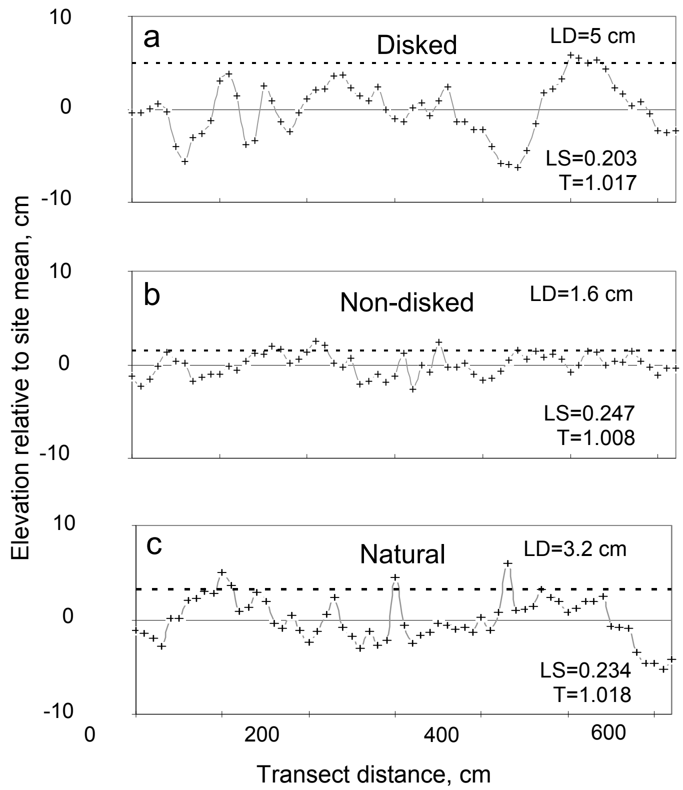
		E. crus-galli	*E. obtusa*	*L. oryzoides*	*L. palustris*
Created wetlands	A	42	36	<1	15
	B	16	2	2	22
	C	7	26	2	30
	D	12	29	<1	14
	E	2	8	7	11
	F	18	51	11	5
	G	75	13	0	2
	H	0	0	0	0
Natural wetlands	I	0	0	33	0
	J	1	1	20	13
	K	0	0	75	0
	L	19	0	8	2
Overall mean %		16.0	13.8	13.1	9.4
Created mean %		21.5	20.6	2.7	12.4
Natural mean %		4.9	0.4	34.0	3.6

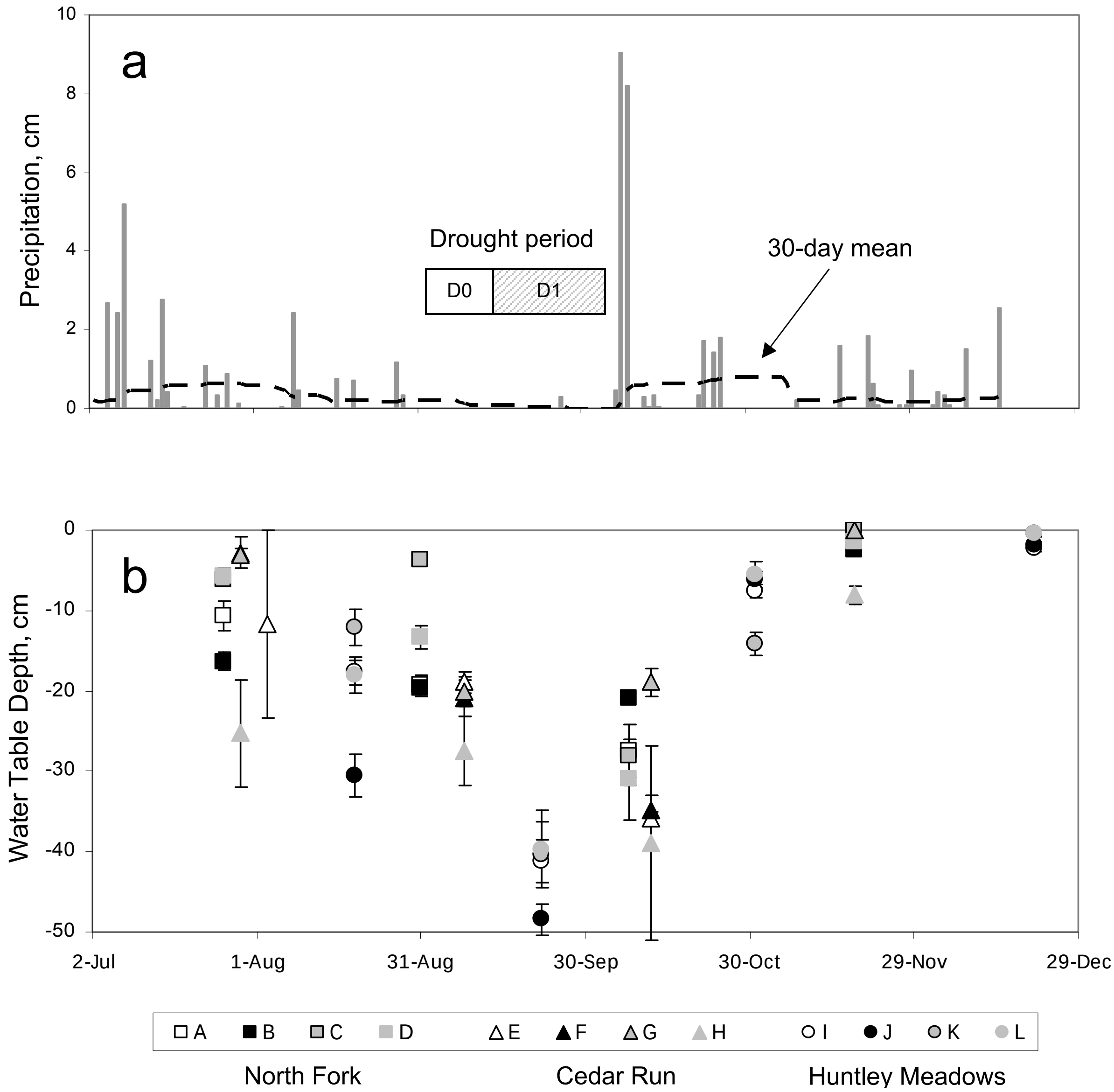
LIST OF FIGURES

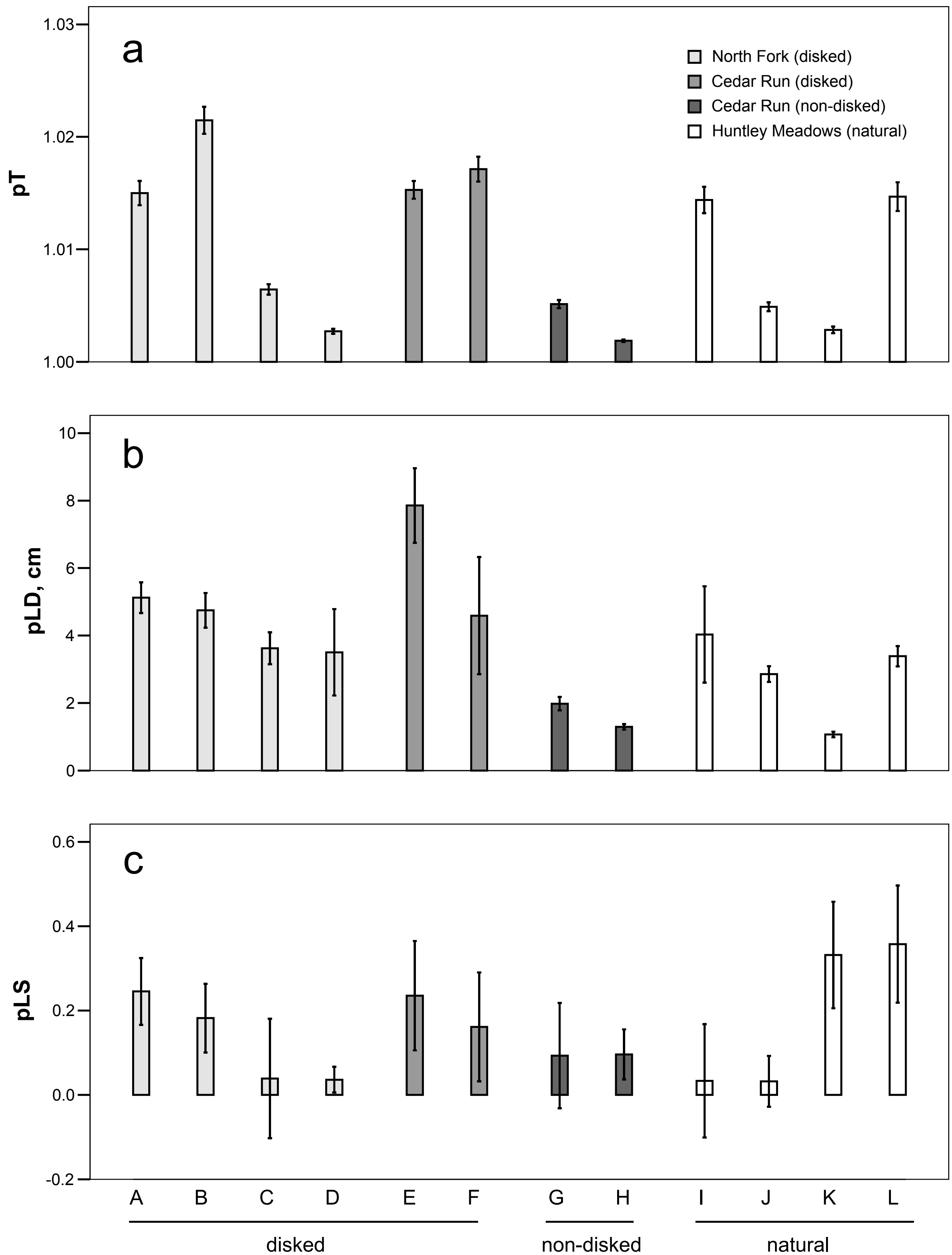

Increasing Roughness (T, LS)
Increasing Relief (T, LD)


N
MARYLAND
IAD
Washington, DC
Fairfax,VA
North Fork
DCA
Huntley Meadows
Cedar Run
Potomac River
VIRGINIA
0 5 10 20 30 40
Kilometers
Area of detail


0.5m transect
1m transect
2m transect
4m transect
1m
Elevation points
Steel rods
Vegetation plots


a
Disked
LD=5 cm
LS=0.203
T=1.017
b
Non-disked
LD=1.6 cm
LS=0.247
T=1.008
c
Natural
LD=3.2 cm
LS=0.234
T=1.018
10
0
-10
0
200
400
600
Elevation relative to site mean, cm
Transect distance, cm


10
8
6
4
2
0
Precipitation, cm
a
Drought period
D0
D1
30-day mean
0
-10
-20
-30
-40
-50
Water Table Depth, cm
b
2-Jul
1-Aug
31-Aug
30-Sep
30-Oct
29-Nov
29-Dec
A
B
C
D
E
F
G
H
I
J
K
L
North Fork
Cedar Run
Huntley Meadows


a
b
c
North Fork (disked)
Cedar Run (disked)
Cedar Run (non-disked)
Huntley Meadows (natural)
pT
pLD, cm
pLS
1.00
1.01
1.02
1.03
0
2
4
6
8
10
-0.2
0.0
0.2
0.4
0.6
A
B
C
D
E
F
G
H
I
J
K
L
disked
non-disked
natural

2. THE INFLUENCE OF MICROTOPOGRAPHY ON SOIL NUTRIENTS IN CREATED MITIGATION WETLANDS

Introduction

Plant-scale topographic variability, or microtopography, may influence wetland hydrology and physicochemistry, thus affecting the balance of plant nutrients in soil. Wetland plants vary in nutrient demands (McJannet et al. 1995; Güsewell & Koerselman 2002) and in adaptations to flooded soil conditions (Kozlowski 1984). Plants also differ in morphology and in their ability to exploit nutrients (Crick & Grime 1987; Hinsinger 2001). Individual species-level responses to soil conditions may determine community composition, richness, and diversity (Bedford et al. 1999; Güsewell et al. 2005), ultimately determining ecosystem functions.

The use of heavy machinery for grading during wetland creation tends to reduce the microtopographic variability commonly found in natural settings (Stolt et al. 2000). Created wetlands also tend to lack the spatial variability of nutrients and biogeochemical processes found in natural wetlands (Bruland et al. 2006). In spite of these characteristic failings, created wetlands are increasingly used to mitigate the loss of natural wetlands. An area of concern for mitigation is the extent to which uniformity of physicochemical conditions may lead to the predominance of few species, thus to a paucity of ecosystem functions (Hooper et al. 2002). In theory, greater variability in localized (plant-scale) nutrient or hydrologic/redoximorphic conditions should support greater plant diversity generally (Tilman 1997; Larkin et al. 2006), greater diversity of functional vegetation types and associated biota (Boutin & Keddy 1993; Grime et al. 1997), and greater ecosystem stability to disturbance (Chapin et al. 1997). Induced microtopography during the restoration or creation of wetlands may enhance such variability and benefit ecosystem development.

Substantial chemical heterogeneity exists at small (1.5 cm interval) vertical scales in wetland soils (Hunt et al. 1997), indicating that even small-scale variations in relief may meaningfully affect soil nutrients. Gradients of increasing moisture, substrate pH and exchangeable Ca and Mg, and decreasing inorganic N and total P, have been shown associated with a microtopographic gradient from higher to lower elevations (Karlin & Bliss 1984; Stoeckel & Miller-Goodman 2001; Bruland & Richardson 2005). Vertical relief may also affect the flux of nutrients with Mn, Fe, and P complexed to Fe accumulating above the water table in microhigh soils, a net upward translocation (Fiedler et al. 2004).

The relationship between microtopography and nutrient distribution is commonly explained in terms of hydrologic/redoximorphic regimes. For instance, in wetland soils, iron plays an important role in phosphorus adsorption, retention, and release (Patrick & Khalid 1974; Baldwin & Mitchell 2000; Aldous et al. 2005); aluminum can likewise affect P availability (Richardson 1985; Axt & Walbridge 1999; Darke & Walbridge 2000). While availability of aluminum-bound phosphate is unaffected by redox status, iron-bound phosphate becomes soluble and available under anaerobic conditions. Consequently, the redox status of Fe in flooded soils may determine P availability (Aldous et al. 2005). Microtopographic elevation affects the frequency, duration, and spatial variability of flooding (Pollock et al. 1998; Fiedler et al. 2004), so it may affect redox conditions and availability of redox-sensitive nutrients. Microtopography may also enhance water retention and soil moisture through increased depression storage (Kamphorst et al. 2000). Thus, roughing the surface (as by disking, the use of tractor-drawn offset disk or disk-harrower) may help establish wetland hydrology, in addition to promoting redoximorphic variability. Field experiments show higher water retention and water table levels for disked than for non-disked wetland restoration plots (Tweedy et al. 2001).

Within the time frame legally mandated for monitoring at mitigation sites, created wetlands show little evidence of ecosystem development comparable to that of natural wetlands, and many fail to meet basic success criteria (National Research Council 2001; Spieles 2005). Our hope is that wetland creation methods might be refined to enhance wetland ecosystem development and functional diversity, increasing the probability that lost wetland ecosystem services are actually replaced, as well as legally mitigated. This study examines the effects of artificially-induced microtopography on soil nutrients in non-tidal freshwater mitigation wetlands, supplementing a study that suggests disking quantitatively enhances created wetland microtopography and plant diversity (Moser et al. 2007). We investigate major limiting nutrients (N, P, K) and macronutrients (Ca, Mg), as well as micronutrients/trace elements involved in toxicity and P availability (Fe, Mn, Al). Broadly stated, our questions are: 1) How do created and natural wetlands differ in terms of soil nutrients/elements? 2) How do disked and non-disked created wetlands differ in terms of these nutrients/elements? and 3) How does microtopography relate to the distribution/abundance of soil nutrients?

Methods

Site Details

Field research was carried out in summer 2005 at 12 study sites in created and natural non-tidal freshwater wetlands in Virginia, USA. Created wetlands were North Fork (38°49.4' N, 77°40.2' W) and Cedar Run (38°37.6' N, 77°33.6' W) mitigation banks in Prince William County; natural wetlands were at Huntley Meadows Park (38°45.0' N, 77°06.8' W) in Fairfax County. While all study wetlands are located within 30 km of Fairfax, Virginia, the created wetlands are located in the Piedmont physiogeographic province, generally characterized by rolling terrain underlain by igneous and metamorphic rock, whereas the natural wetlands were in the Coastal Plain, comparatively flat and underlain by unconsolidated sediment. Although portions of the created wetlands were intended to mitigate the loss of palustrine forested wetlands, all planted trees were small saplings at the time of the study, and these wetlands could best be characterized as palustrine emergent, comparable to the natural wetlands.

North Fork is a 125-acre wetland/upland complex created in 1999-2000 on land formerly used as cattle pasture. Soils are generally silt loams and silty clay loams over Newark Supergroup basalt and sandstone/siltstone formations of the Culpeper Basin. Four study sites were located in a 51-acre wetland area surrounding open water, with vegetation in its fifth growing season following disked wetland creation. Cedar Run is a large multiple-wetland complex developed on land formerly used for agriculture. Soils are primarily silt loams over Newark Supergroup interbedded sandstone/siltstone/shale. Two study sites were located in a 67-acre wetland complex created and disked in 2004-5, while two sites were in a smaller adjacent wetland that was re-graded without disking and seeded in late 2004. All Cedar Run sites were thus in their first growing season. The 1425-acre Huntley Meadows Park prominently features beaver-engineered wetlands in an urbanized watershed. Soils are derived from gravel, sand, silt, and clay of the Shirley Formation, Pleistocene Epoch deposits of the Potomac River. Two study sites were in a mature (>30 years old) emergent wetland, while two were in an emergent wetland adjacent to a more recently established (~10 years old) beaver pond.

Microtopography

Each of the 12 study sites was examined using a single set of tangentially-conjoined circular transects, with field measurements and samples taken at regular intervals along the circular paths (Figure 1). The circular transect is an approach designed to be directionally-unbiased; any confounding directional effects of disking orientation, wind, direction of hydrologic flows, orientation of incident sunlight, etc, are thus minimized. Transects were laid out as 0.5 m-, 1 m-, and 4 m-diameter circles using polyethylene tubing hoops. Within each wetland, sites were randomly selected, although for created wetlands where marked survey locations had been previously established, a survey marker was randomly selected and the study site established 3 m to the north.

Each soil sampling location was associated with three microtopographic parameters determined from fine-scale survey of transect elevations (10-20cm interval, see Figure 1). These were the indices tortuosity (Kamphorst et al. 2000) and limiting elevation difference (Linden & Van Doren 1986), and elevation relative to the mean multiscale transect elevation. Whereas tortuosity is an overall measure of roughness akin to surface area, limiting elevation difference reflects the degree of topographic relief. Index measures were calculated for each soil sampling location based on transect data for near-neighbor data points (points within 30-60cm, depending on the transect scale) using a methodology developed for evaluating wetland microtopography (Moser et al. 2007). Subsets of the data from that study, these indices characterize the immediate surrounding microtopography, and are referred to as proximal tortuosity (pT) and proximal limiting elevation difference (pLD).

Soil sampling and analysis

Soil samples were collected at 80cm intervals along 0.5 m-, 1 m-, and 4 m-diameter transects (Figure 1, 162 samples total) between 26 July and 2 August 2005, at peak vegetation growth. A soil probe/auger (1.8 cm inner diameter) was used to collect the top 10 cm of soil, excluding surface litter. Samples were stored in polyethylene bags and transported on ice, then stored in the lab at -15°C pending analysis. Samples were thawed and homogenized by hand, with roots, recognizable plant material, and coarse gravel removed. Sub-samples were oven-dried at 105°C for 48 hours and used to determine moisture content for each sample (calculated as [wet weight – dry weight]/dry weight, expressed as a percentage). Dried sub-samples were passed through a 2 mm sieve and ground with a mortar and pestle before analysis for total C and N (percent dry weight) using a Perkin-Elmer 2400 Series II CHNS/O Analyzer (Perkin-Elmer Corporation, Norwalk, CT, USA). KCl extraction (Mulvaney 1996) was performed on field-moist samples to quantify available inorganic nitrogen (NO_3–N and NH_4-N, expressed as µg N/g dry weight), with 40 mL of 1M KCl added to 4 g dry-weight-equivalent soil, the mixture shaken at room temperature for 60 minutes on a reciprocating shaker table and allowed to settle for 30 minutes, and the supernatant passed through a 1.2 µm glass-fiber filter, followed by colorimetric analysis using an Astoria-Pacific segmented flow analyzer (Astoria-Pacific International, Clackamas, OR, USA). Mehlich-3 extraction (Mehlich 1984) was performed for field-moist samples to quantify available Al, Fe, P, Ca, Mg, Mn, and K, with 20 mL of Mehlich-3 extractant added to 2 g dry-weight equivalent of soil, the mixture shaken at room temperature for 5 minutes on a reciprocating shaker table and allowed to settle for 1 minute, then passed through a 0.45 µm polyethersulfone filter. 1:10 (v:v) dilutions were analyzed by inductively-coupled plasma optical emission spectroscopy (ICP-OES) using a Perkin-Elmer Optima 4300 DV analyzer (Perkin-Elmer Corporation, Norwalk, CT, USA), and also analyzed for inorganic P by colorimetry using a Technicon II Autoanalyzer (Bran+Luebbe GmbH, Norderstedt, Germany). Because P availability depends to a great extent on both Al and Fe, the molar ratio [P/(Al+Fe)] for Mehlich-3 extraction was used as a measure to predict P saturation (Kleinman & Sharpley 2002; Sims et al. 2002). Soil inorganic N:P (designated iN:iP) ratios [(NO_3-N + NH_4-N)/ortho-P], pertaining to the forms of N and P most available to plants, were also used to determine N or P limitation (Wassen et al. 1995; Koerselman & Meuleman 1996; Bedford et al. 1999). N limitation was inferred for iN:iP < 14, P-limitation for iN:iP > 16, and co-limitation for ratios in between.

Statistical analysis

Because the hydrogeomorphic settings differed among study localities, our analysis treats soil moisture as a covariate reflecting differences in both soil pore space attributable to soil composition and proximity to the water table (i.e., concentrations of mobile/water-soluble nutrients were expected to reflect water volume). Moreover, our analysis also stresses comparison of group variances, a necessity because topographic variability may influence nutrient distribution irrespective of nutrient abundance (i.e., group means). Our data conformed poorly to the implicit assumptions of multivariate analysis of variance and covariance (MANOVA/MANCOVA), so univariate analysis of covariance (ANCOVA) was carried out separately for each nutrient variable. Where the covariate was not significant ($\alpha=0.05$), or the assumption of homogeneity of regression slopes could not be met, univariate analysis of variance (ANOVA) was performed. Data were categorized into four wetland groups for ANOVA/ANCOVA: 1) disked, North Fork (n=52), 2) disked Cedar Run (n=26), 3) non-disked Cedar Run (n=28), and 4) natural, Huntley Meadows (n=52). ANOVA/ANCOVA analyses were nested two-factor analyses (site nested within wetland group) using Type III sums-of-squares and an alpha level of 0.05. Post-hoc Dunnett's T3 pairwise comparisons were performed for ANOVA; since this test is not appropriate for ANCOVA, Bonferroni adjustment was applied for ANCOVA pairwise comparisons. Comparisons of means for microtopographic index measures have previously been reported for the parent data set (Moser et al. 2007), so microtopographic index parameters were not compared here. The Levene test of equality of variance (commonly used to test the ANOVA/ANCOVA equality of variance assumption) was used as a more robust alternative to Bartlett's two-sample test for comparing group variances ($\alpha=0.05$); Bonferroni adjustment was applied for pairwise comparisons. Correlations among microtopographic indices and nutrient variables for each soil sample location were examined using non-parametric Spearman rank correlation coefficients, using untransformed variables. ANOVA/ANCOVA and Spearman correlation analyses were performed using SPSS (SPSS Inc. 2004). Correlation-based Principal Components Analysis (PCA, conducted using normalized variables) was also performed to reduce the number of nutrient variables to a small number of factors, using PRIMER (PRIMER-E Ltd. 2006). To better conform to the assumptions of ANOVA/ANCOVA and PCA, natural log transformations were applied for Ca, Mg, Mn, and NO_3-N. For log-transformed variables, reported values are converted back to original units.

Results

The microtopographic data supported the notion that though the microtopography of disked created wetlands is generally comparable to that of the natural wetlands at the extent and resolution examined, disked sites have more pronounced microtopography than do non-disked created wetland sites. Proximal limiting elevation difference (pLD) index means clearly distinguished disked and non-disked created wetland microtopography, with minimal relief evident in non-disked sites (Table 1); proximal tortuosity (pT) likewise showed non-disked sites to be comparatively low in microtopography (Table 1). While pLD and pT means for the natural sites were intermediate, their overall ranges encompassed the corresponding ranges of both disked and non-disked sites.

Soil moisture content, with an overall range between 12 and 44 percent (Table 1), correlated positively, but weakly, with the pLD index but not with the pT index or relative elevation (Table 2). Soil moisture differed significantly among wetlands ($F_{3,8.2} = 8.77$, $p = 0.006$, Figure 2), and it correlated significantly with most parameters (Table 2). Soil moisture was weakly/positively correlated with extractable Ca, Al, Fe, total P, ortho-P, NO_3-N, and NH_4-N, and weakly/negatively with Mn. Despite numerous correlations (r_{Sp}), moisture was a significant covariate only within created wetlands for total C, total N, and extractable Ca, and among all sites for total P and ortho-P (Table 3). For C, N, and Ca, the Huntley Meadows (natural) sites were excluded from ANCOVA in order to satisfy the assumption of homogeneity of regression slopes, enabling ANCOVA adjustment for means comparison among created wetlands. Moisture was not a significant covariate for K, Fe, or Mn, while the assumption of homogeneity of regression slopes could not be met for Mg, Al, and NH_4–N (for NH_4-N, the homogeneity assumption could be met within created wetlands, but moisture was nonetheless not a significant covariate). Thus, group means were selectively ANCOVA-adjusted (Table 3).

As measured, soil total C and N were lower for created (C range 0.3-4.0%, mean 1.2%; N range 0.01-0.26%, mean 0.11%) than for natural wetlands (C range 0.7-7.7%, mean 2.5%; N range 0.06-0.39%, mean 0.19%). Adjusted for the covariate moisture, however, no significant mean difference was evident either for total C ($F_{2,8.7} = 1.96$, $p = 0.20$, Figure 2) or for total N ($F_{2,6.9} = 2.64$, $p = 0.14$, Figure 2). NO_3-N concentrations ranged from zero to 11.7 µg N/g ($\bar{x}$ = 2.2 µg N/g), while NH_4-N concentrations ranged from 0.1 to 35 µg N/g ($\bar{x}$ = 8.4 µg N/g). Average NO_3-N concentrations differed among the study wetlands ($F_{3,9.0} = 6.22$, $p = 0.014$, Figure 2); concentrations were higher for North Fork than for Huntley Meadows. NO_3-N concentrations were lowest at Cedar Run, and there was no apparent difference between the disked and non-disked sites there. Average NH_4-N concentrations also differed ($F_{3,9.1} = 12.75$, $p = 0.001$, Figure 2), with higher concentrations for disked Cedar Run and Huntley Meadows than for North Fork and non-disked Cedar Run (the lowest).

Mehlich-3 extractable total P and orthophosphate-P had similar ranges (0-46 µg/g) and were highly correlated (Table 2), with linear regression slope approximating equality ([orthophosphate-P] = 0.998 * [total P] – 0.810; $R^2 = 0.917$); any extraction of organic P (non-molybdate-reactive P) was apparently minimal, and the two methods are thus essentially equivalent as used here. However, since ICP-OES resolution was limited at low P concentrations, the orthophosphate-P determinations are presumed more reliable; these alone were used for Principal Components Analysis. Mean extractable orthophosphate-P differed among the study wetlands ($F_{3,9.4} = 8.59$, $p = 0.005$, Figure 2). Similarly, mean extractable total P differed among the study wetlands ($F_{3,9.4} = 8.14$, $p = 0.006$, Huntley Meadows [$\bar{x}$ = 15.1 ug P/g] > disked Cedar Run [$\bar{x}$ = 8.6] ≈ non-disked Cedar Run [$\bar{x}$ = 6.1] ≈ North Fork [$\bar{x}$ = 2.1], with disked Cedar Run > North Fork).

Based on the Mehlich-3 [P/(Al+Fe)] molar ratios determined for this study, all sites fell within the "below optimum" category for P availability (Beegle et al. 1998; Sims et al. 2002). Ratios at North Fork and non-disked Cedar Run were especially low ($\bar{x}$ = 0.004 and $\bar{x}$ = 0.005, respectively), while those at disked Cedar Run ($\bar{x}$ = 0.023) compared to those for Huntley Meadows ($\bar{x}$ = 0.024). The iN:iP means suggested N limitation for disked Cedar Run and

Huntley Meadows ($\bar{x}$ = 2.5 and $\bar{x}$ = 1.2, respectively) and P limitation for North Fork and non-disked Cedar Run ($\bar{x}$ = 21.1 and $\bar{x}$ = 26.7, respectively); the distributions were severely skewed, however, and iN:iP medians were below 2, except for North Fork (*Mdn* = 11.4), where 34% of the iN:iP values (15/44) were higher than 16, evenly distributed among sites.

The ranges of Mehlich-3 extractable macronutrient concentrations were 29-3800 μg Ca/g, 10-427 μg Mg/g), and 0.1-32 μg K/g. Mean concentrations differed among wetland types for Ca ($F_{2,6.4} = 14.02$, $p = 0.005$, Figure 2) and Mg ($F_{3,8.1} = 11.44$, $p = 0.003$, Figure 2), but not for K ($F_{3,8.4} = 0.577$, $p = 0.65$, Figure 2). Concentrations of Mg were significantly lower in the disked compared to non-disked Cedar Run sites (Figure 2). Micronutrient concentrations ranged from 26 to 487 μg Fe/g and from 8 to 247 μg Mn/g. Mean Fe concentrations were higher for natural than for created wetlands, and also higher for disked than non-disked created wetlands ($F_{3,8.6} = 7.16$, $p = 0.010$, Figure 2); mean Mn concentrations were higher for created than for natural wetlands, but similarly higher for disked than for non-disked created wetlands ($F_{3,8.7} = 19.11$, $p < 0.001$, Figure 2). No significant mean difference among sites was apparent for Mehlich-3-extractable Al, which ranged from 80-770 μg/g.

The Levene test indicated inequality of variance for all but three of the measured soil parameters (Table 4); consequently, the *p*-values of the ANOVA/ANCOVA comparisons (α=0.05) should be understood as somewhat non-conservative (ANOVA/ANCOVA is fairly robust to violations of the homogeneity of variance assumption, however, and *p*-values were generally well below α). Neither created (disked/non-disked) nor natural wetlands had consistently higher or lower variance, but non-disked variances were consistently the lowest (Table 4).

Numerous significant intercorrelations were apparent among the soil nutrient parameters (Table 2). Strong correlations existed between C and N, extractable total P and orthophosphate-P, and extractable Fe and P (both total P and orthophosphate-P). Correlations with microtopographic parameters were very weak, and only evident for K, Mn, Al, and soil moisture. Extractable K and Al were lower, and Mn greater, with increasing proximal tortuosity.

Principal Components Analysis of the nutrient data identified three components with eigenvalues > 1 (4.6, 2.3, and 1.5), the first two of which accounted for 63% of the nutrient variability. Explaining 41% of the variability, PCA component 1 had highest factor loadings for orthophosphate-P (0.396), NH_4-N (0.339), and Fe (0.383), and a high negative loading for Ca (-0.358). Component 2 accounted for 21% of the nutrient variability, with highest component loadings for Mg (0.469) and NO_3-N (0.397), but also a fairly high loading for Ca (0.339). While the first component axis clearly separates the created from the natural wetlands, with higher component scores for the latter, the second component axis mainly distinguishes the two created wetland locations, with higher component scores for North Fork sites than for Cedar Run sites (Figure 3).

The first PCA factor suggests a gradient from the more mineral soils found in the Piedmont to more organic and comparatively nutrient-rich soil of the Coastal Plain (Figure 3, note also that the two Huntley Meadows wetland groups have similar spreads, but the older wetland is shifted right). Interpretation of the second component is less clear, especially because the Huntley Meadows sites span the range of component scores, but the created wetlands are clearly separated along the second axis by extractable soil cation concentrations. On either component axis a broader range is evident for the natural (Huntley Meadows) sites than for any of the created wetlands, and soil total C and N seems to increase along both axes. Though the third component axis accounted for 13% of the nutrient variability, this component axis (not shown) did not strongly differentiate the comparison groups.

Discussion

Created wetlands are commonly located on former agricultural lands, and thus tend to have mineral soils which gradually accumulate organic matter with age; the relatively low ranges of soil total C and N in this study were fairly typical for created wetlands (Stolt et al. 2000; Anderson et al. 2005). Natural wetlands feature comparatively organic soils, suggested in this study by greater soil C and N, and to some extent by greater soil moisture, reflecting lower bulk density and increased pore space. Created wetland soil moisture, C, and N also increased with both disking and age. Higher P and Fe in natural wetlands may reflect the presence of humic-Fe-P complexes, characteristic of more organic soil.

The contrast between the more recently-flooded mineral/clay soils of the created wetlands and the comparatively more developed organic sandy soils of the natural wetlands may explain the negative correlations between mineral cation elements (Ca, Mn) and C, N, and P. Chemical properties of the weathering rock substrate may be a source of site differences as well; North Fork, for instance, is geologically associated with extrusive basalt, so higher Ca and Mg should be expected. Greater groundwater connectivity and soil permeability also distinguishes the natural from the created wetland study sites. Thus, soil samples collected during a low-water season at a site with greater groundwater recession might be expected to have diminished quantities of soluble nutrients, as was observed for Ca, Mg, and Mn at Huntley Meadows. Current and prior land use may also be a factor; the agricultural/pastoral rural setting of the created wetland sites may promote nutrient depletion, while Huntley Meadows' urbanized setting may contribute nutrients to the wetland.

Due to cultivation and chemical soil amendments, agricultural lands tend to lack nutrient heterogeneity. The low microtopographic relief ordinarily imparted by wetland creation practices further imposes uniformity of soil conditions. Such conditions were apparent from low variances in nutrient concentrations and comparatively low microtopographic index values for non-disked sites, but not for the (disked) created wetlands generally. The PCA ordination suggests that two of the created wetlands (North Fork, all of which was disked, and non-disked Cedar Run) had less nutrient variability than natural wetlands, evident from the relative spread of points for each study location along the first two component axes. The variability within the disked Cedar Run wetland, however, was more comparable to that of the natural wetland.

Significant inequalities of variance were observed for inorganic N, orthophosphate-P, and K, nutrients that are critical for plant growth. While it is difficult to make a fair comparison between the created and natural wetlands based on gravimetric determinations of nutrient concentrations (Wheeler et al. 1992; Bridgham et al. 1998), it is worth noting that the variances of disked wetlands compared favorably with those of the natural wetlands. Non-disked wetlands had comparatively low variance, supporting the contention that nutrients are spatially homogeneous in microtopographically homogeneous created wetlands (Bruland et al. 2006).

The contrast between disked and non-disked created wetlands at Cedar Run is striking, given their shared setting and conditions. Although of the same age, disked sites had higher moisture content than non-disked, possibly attributable to increased storage in soil voids or to increased depression storage. The microrelief induced by disking also appears to enhance availability of certain nutrients, as well as nutrient variability. The disked Cedar Run sites had higher Mehlich-3 extractable Fe and Mn, and much (7x) higher NH_4-N than the non-disked sites, even though total soil N was comparable. The latter may indicate increased prevalence of nitrogen mineralization relative to nitrification, as might be expected when anaerobic conditions predominate due to greater soil moisture, or it might indicate better nutrient retention. However, it could be a consequence of soil inversion and consequent exposure of previously unavailable organic N substrate to microbial activity, particularly since disking was recent, within a year (Silgram & Shepherd 1999; Calderon & Jackson 2002).

The range of Mehlich-3 extractable P was somewhat lower than values reported for other created freshwater wetlands (Anderson et al. 2005). Available P was low or very low for created wetlands, and low to medium for natural wetlands (Tisdale 1993; Sims et al. 2002). North Fork, in particular, had very low P concentrations (moisture-adjusted $\bar{x}$ = 0.6 µg ortho-P/g) and iN:iP ratios suggesting P limitation. In this study, the iN:iP and [P/(Al+Fe)] ratios reflect soil conditions at peak growth, when a great extent of P cycling within the system might be expected to be in living plant material, as opposed to in the soil. However, low P availability and/or P limitation at this time could affect the growth of late-season developing plants (Boeye et al. 1999).

The intercorrelations among total C and N, P and Fe (and to a lesser extent Al) suggested the importance of humic-metal complexes with adsorbed P, potential sources of P for plant uptake. This result accords with other studies associating extractable Al and Fe with soil organic content (Axt & Walbridge 1999; Darke & Walbridge 2000). Since Fe appears to play a role in P availability, the significant differences in group variances for Fe take on greater importance than the somewhat less definitive differences in Al group variances. Although orthophosphate-P did not differ between disked and non-disked created wetlands, mean differences in Fe may be important both because P limitation was implicated and because P availability influences plant community composition (Güsewell & Koerselman 2002).

Differences in NH_4-N are also important, as this form of N is readily available to plants, and N is commonly limiting (or co-limiting) in freshwater wetlands (Bedford et al. 1999). At very low nutrient levels, vegetation diversity is likely to decline (Tilman 1997; Güsewell et al. 2005), so to the extent that disking enhances retention and variability of nutrients, it is likely to promote diversity as well as productivity during early ecosystem development; moreover, nutrient heterogeneity may also reduce competitive exclusion (Tilman 1997). The concomitant plant and functional diversity may enhance ecosystem stability and resilience (Tilman 1996; Loreau 200).

Explanatory mechanisms were not strongly evident from the study data. The correlation between moisture content and pLD confirmed our expectations based on the utility of limiting elevation difference in predicting depression storage (Kamphorst et al. 2000); it also comports well with the empirical observation that sites with greater microtopographic relief were often associated with the presence and persistence of standing water. If Fe concentrations are attributable to the microtopographic effects of disking, the results are consistent with net upward translocation of Fe and P from reducing to oxidizing soil layers (Fiedler et al. 2004). However, relative elevation correlated with neither Fe nor P, in contradiction. It has been suggested that upward transport of Fe and P is limited to recycling within the top 30 cm of soil (Hunt et al. 1997). As such, the effect of soil inversion by disking would not be expected to increase Fe or P. A possible explanation is that more pronounced flooding in the disked soils leads to development of poorly crystalline hydroxides of iron that are more easily extracted and enhanced release of phosphate to solution (Patrick & Khalid 1974; Gambrell & Patrick 1978); indeed, some evidence suggests Fe is less crystalline in microlows than in microhighs (Darke & Walbridge 2000). Alternatively, disked microtopography may simply prevent leaching of Fe to soil layers below the root zone and runoff-induced loss.

Conclusions

Though disking clearly provides microtopographic variability not otherwise evident in created wetlands, it does so at a specific scale, with vertical relief on the order of that shown to promote floristic diversity in controlled experiments (Vivian-Smith 1997). Measured as tortuosity or as limiting elevation difference, this effect was apparent at all the spatial extents (i.e., transect scales) in the companion study (Moser et al. 2007). Consequently, though disking promotes microtopographic heterogeneity evident at small spatial extents, it nonetheless represents topographic uniformity when considered at the full spatial extent of a created wetland. Disking-induced microtopography is thus qualitatively different from excavated hummocks/hollows, which provide greater topographic relief in distinct locations. Because disking covers a wide area, its effects apply broadly, whereas hummock/hollow topography yields localized benefits (e.g., pools, patches of vegetation).

A number of soil characteristics associated with disked microtopography are beneficial in wetland mitigation. Increased soil moisture with increasing microrelief suggests that microtopography enhances wetland hydrology, a legal and functional mitigation success criterion, and resulting anaerobic conditions may increase the prevalence of wetland plants. Increased variability of soil nutrients and hydrologic conditions are expected to promote plant diversity by catering to a wider spectrum of plant capabilities and demands. In terms of functional replacement of lost wetlands, the enhanced soil development and nutrient variability should promote a greater complexity of processes and interactions than might be supported by more typical wetland creation methods. As a relatively low-cost method to establish microtopography in mitigation wetlands, disking is recommended, though it should not preclude other methods of inducing microtopography.

Implications for Practice

- In contrast to natural wetlands they are intended to replace, created mitigation wetlands are often characterized by uniformity of soil conditions, including hydrology, nutrients, and microtopography.
- Created wetlands may be disked to establish microtopographic variability, affecting the distribution of nutrients as well as the frequency and duration of flooding, creating heterogeneous soil conditions comparable to those in natural wetlands
- Disking also appears to increase retention of soil nutrients and moisture, enhancing accumulation of organic material and promoting the development of organic soils from the mineral soils typically used to create mitigation wetlands.
- Disking-induced microtopography may help ensure that created mitigation wetlands adequately replace lost wetland functions, as well as meet criteria for legal mitigation success.

Literature Cited

Aldous, A., P. McCormick, C. Ferguson, S. Graham, and C. Craft. 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restoration Ecology **13**:341-347.

Anderson, C. J., W. J. Mitsch, and R. W. Nairn. 2005. Temporal and spatial development of surface soil conditions at two created riverine marshes. Journal of Environmental Quality **34**:2072-2081.

Axt, J. R., and M. R. Walbridge. 1999. Phosphate removal capacity of palustrine forested wetlands and adjacent uplands in Virginia. Soil Science Society of America Journal **63**:1019-1031.

Baldwin, D. S., and A. M. Mitchell. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: A synthesis. Regulated Rivers-Research & Management **16**:457-467.

Bedford, B. L., M. R. Walbridge, and A. Aldous. 1999. Patterns in nutrient availability and plant diversity of temperate North American wetlands. Ecology **80**:2151-2169.

Beegle, D. B., A. N. Sharpley, and D. Graetz. 1998. Interpreting soil test phosphorus for environmental purposes. *In* J. T. Sims (ed.) Soil testing for phosphorus: environmental uses and implications. Southern Cooperative Series Bulletin No. 389. University of Delaware, Newark, DE, USA.

Boeye, D., B. Verhagen, V. Van Haesebroeck, and M. El-Kahloun. 1999. Phosphorus fertilization in a phosphorus-limited fen: effects of timing. Applied Vegetation Science **2**:71-78.

Boutin, C., and P. A. Keddy. 1993. A functional classification of wetland plants. Journal of Vegetation Science **4**:591-600.

Bridgham, S. D., K. Updegraff, and J. Pastor. 1998. Carbon, nitrogen, and phosphorus mineralization in northern wetlands (vol 79, pg 1545, 1998). Ecology **79**:2571-2571.

Bruland, G. L., and C. J. Richardson. 2005. Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. Restoration Ecology **13**:515-523.

Bruland, G. L., C. J. Richardson, and S. C. Whalen. 2006. Spatial variability of denitrification potential and related soil properties in created, restored, and paired natural wetlands. Wetlands **26**:1042-1056.

Calderon, F. J., and L. E. Jackson. 2002. Rototillage, disking, and subsequent irrigation: Effects on soil nitrogen dynamics, microbial biomass, and carbon dioxide efflux. Journal of Environmental Quality **31**:752-758.

Crick, J. C., and J. P. Grime. 1987. Morphological plasticity and mineral nutrient capture in 2 herbaceous species of contrasted ecology. New Phytologist **107**:403-414.

Darke, A. K., and M. R. Walbridge. 2000. Al and Fe biogeochemistry in a floodplain forest: Implications for P retention. Biogeochemistry **51**:1-32.

Fiedler, S., D. Wagner, L. Kutzbach, and E. M. Pfeiffer. 2004. Element redistribution along hydraulic and redox gradients of low-centered polygons, Lena Delta, northern Siberia. Soil Science Society of America Journal **68**:1002-1011.

Gambrell, R. P., and W. H. Patrick, Jr. 1978. Chemical and microbiological properties of anaerobic soils and sediments. p. 375-423. *In* D. D. Hook and R. M. M. Crawford (eds.) Plant life in anaerobic environments. Ann Arbor Science Publishers, Ann Arbor, MI, USA.

Grime, J. P., K. Thompson, R. Hunt, J. G. Hodgson, J. H. C. Cornelissen, I. H. Rorison, G. A. F. Hendry, T. W. Ashenden, A. P. Askew, S. R. Band, R. E. Booth, C. C. Bossard, B. D. Campbell, J. E. L. Cooper, A. W. Davison, P. L. Gupta, W. Hall, D. W. Hand, M. A. Hannah, S. H. Hillier, D. J. Hodkinson, A. Jalili, Z. Liu, J. M. L. Mackey, N. Matthews, M. A. Mowforth, A. M. Neal, R. J. Reader, K. Reiling, W. RossFraser, R. E. Spencer, F. Sutton, D. E. Tasker, P. C. Thorpe, and J. Whitehouse. 1997. Integrated screening validates primary axes of specialisation in plants. Oikos **79**:259-281.

Güsewell, S., K. M. Bailey, W. J. Roem, and B. L. Bedford. 2005. Nutrient limitation and botanical diversity in wetlands: can fertilisation raise species richness? Oikos **109**:71-80.

Güsewell, S., and M. Koerselman. 2002. Variation in nitrogen and phosphorus concentrations of wetland plants. Perspectives in Plant Ecology Evolution and Systematics **5**:37-61.

Hinsinger, P. 2001. Bioavailability of soil inorganic P in the rhizosphere as affected by root-induced chemical changes: a review. Plant and Soil **237**:173-195.

Hooper, D. U., M. Solan, A. Symstad, S. Diaz, M. O. Gessner, N. Buchmann, V. Degrange, P. Grime, F. Hulot, F. Mermillod-Blondin, J. Roy, E. M. Spehn, and L. van Peer. 2002. Species diversity, functional diversity, and ecosystem functioning. *In* M. Loreau, S. Naeem, and P. Inchausti (eds.) Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. Oxford University Press, Oxford, UK.

Hunt, R. J., D. P. Krabbenhoft, and M. P. Anderson. 1997. Assessing hydrogeochemical heterogeneity in natural and constructed wetlands. Biogeochemistry **39**:271-293.

Kamphorst, E. C., V. Jetten, J. Guerif, J. Pitkanen, B. V. Iversen, J. T. Douglas, and A. Paz. 2000. Predicting depressional storage from soil surface roughness. Soil Science Society of America Journal **64**:1749-1758.

Karlin, E. F., and L. C. Bliss. 1984. Variation in substrate chemistry along microtopographical and water-chemistry gradients in peatlands. Canadian Journal of Botany **62**:142-153.

Kleinman, P. J. A., and A. N. Sharpley. 2002. Estimating soil phosphorus sorption saturation from Mehlich-3 data. Communications in Soil Science and Plant Analysis **33**:1825-1839.

Koerselman, W., and A. F. M. Meuleman. 1996. The vegetation N:P ratio: a new tool to detect the nature of nutrient limitation. Journal of Applied Ecology **33**:1441-1450.

Kozlowski, T. T. 1984. Plant responses to flooding of soil. Bioscience **34**:162-167.

Larkin, D., G. Vivian-Smith, and J. B. Zedler. 2006. Topographic heterogeneity theory and ecological restoration. p. 142-164. *In* D. A. Falk, M. A. Palmer, and J. B. Zedler (eds.) Foundations of restoration ecology. Island Press, Washington, DC, USA.

Linden, D. R., and D. M. Van Doren. 1986. Parameters for characterizing tillage-induced soil surface-roughness. Soil Science Society of America Journal **50**:1560-1565.

Loreau, M. 2000. Biodiversity and ecosystem functioning: recent theoretical advances. Oikos **91**:3-17.

McJannet, C. L., P. A. Keddy, and F. R. Pick. 1995. Nitrogen and phosphorus tissue concentrations in 41 wetland plants - a comparison across habitats and functional groups. Functional Ecology **9**:231-238.

Mehlich, A. 1984. Mehlich-3 soil test extractant - a modification of Mehlich-2 extractant. Communications in Soil Science and Plant Analysis **15**:1409-1416.

Moser, K., C. Ahn, and G. Noe. 2007. Characterization of microtopography and its influence on vegetation patterns in created wetlands. Wetlands **27**:1081-1097.

Mulvaney, R. L. 1996. Nitrogen - inorganic forms. p. 1123-1184. *In* D. L. Sparks (ed.) Methods of soil analysis. Part 3, Chemical methods. Soil Science Society of America : American Society of Agronomy, Madison, WI.

National Research Council. 2001. Compensating for wetland losses under the Clean Water Act. National Academy Press, Washington, DC, USA.

Patrick, W. H., Jr., and R. A. Khalid. 1974. Phosphate release and sorption by soils and sediments: effect of aerobic and anaerobic conditions. Science **186**:53-55.

Pollock, M. M., R. J. Naiman, and T. A. Hanley. 1998. Plant species richness in riparian wetlands - A test of biodiversity theory. Ecology **79**:94-105.

PRIMER-E Ltd. 2006. PRIMER v6.1. PRIMER-E Ltd., Plymouth, UK.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science **228**:1424-1427.

Silgram, M., and M. A. Shepherd. 1999. The effects of cultivation on soil nitrogen mineralization. p. 267-311. *In* Advances in agronomy, Vol 65. Academic Press Inc, San Diego.

Sims, J. T., R. O. Maguire, A. B. Leytem, K. L. Gartley, and M. C. Pautler. 2002. Evaluation of Mehlich 3 as an agri-environmental soil phosphorus test for the Mid-Atlantic United States of America. Soil Science Society of America Journal **66**:2016-2032.

Spieles, D. J. 2005. Vegetation development in created, restored, and enhanced mitigation wetland banks of the United States. Wetlands **25**:51-63.

SPSS Inc. 2004. SPSS 13.0 for Windows graduate student version. SPSS, Inc., Chicago, IL, USA.

Stoeckel, D. M., and M. S. Miller-Goodman. 2001. Seasonal nutrient dynamics of forested floodplain soil influenced by microtopography and depth. Soil Science Society of America Journal **65**:922-931.

Stolt, M. H., M. H. Genthner, W. L. Daniels, V. A. Groover, S. Nagle, and K. C. Haering. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands **20**:671-683.

Tilman, D. 1996. Biodiversity: population versus ecosystem stability. Ecology **77**:350-363.

Tilman, D. 1997. Mechanisms of plant competition. *In* M. J. Crawley (ed.) Plant ecology, 2nd edition. Blackwell Science, Inc., Cambridge, MA, USA.

Tisdale, S. L. 1993. Soil fertility and fertilizers, 5th edition. Prentice Hall, Upper Saddle River, NJ, USA.

Tweedy, K. L., E. Scherrer, R. O. Evans, and T. H. Shear. 2001. Influence of microtopography on restored hydrology and other wetland functions (Meeting Paper No. 01-2061). *in* 2001 American Society of Agricultural Engineers Annual International Meeting. ASAE, St. Joseph, MI, USA.

Vivian-Smith, G. 1997. Microtopographic heterogeneity and floristic diversity in experimental wetland communities. Journal of Ecology **85**:71-82.

Wassen, M. J., H. G. M. Olde Venterink, and E. O. A. M. de Swart. 1995. Nutrient concentrations in mire vegetation as a measure of nutrient limitation in mire ecosystems. Journal of Vegetation Science **6**:5-16.

Wheeler, B. D., S. C. Shaw, and R. E. D. Cook. 1992. Phytometric assessment of the fertility of undrained rich-fen soils. Journal of Applied Ecology **29**:466-475.

Table 1: Proximal microtopographic indices for tortuosity (pT), limiting elevation difference (pLD), and soil moisture content. Mean ± 1 SE.

	pT	pLD (cm)	Moisture %
North Fork (disked)	1.013 ± .002	5.0 ± .8	26.5 ± .5
Cedar Run (disked)	1.012 ± .001	6.1 ± .9	22.8 ± .8
Cedar Run (non-disked)	1.003 ± <.001	1.6 ± .2	16.8 ± .7
Huntley Meadows (natural)	1.011 ± .003	4.3 ± .7	32.4 ± .5

Table 2: Spearman rank correlations among measured soil/microtopographic parameters. Boldface indicates correlation is significant at the 0.01 level; underlining indicates correlation is significant at the 0.05 level.

	pT	pLD	Elev	% Moist	C	N	NO3	NH4	P	oP	K	Ca	Mg	Fe	Mn
pLD	**.616**														
Elev	-.018	-.058													
% Moist	.083	**.221**	-.012												
% C	-.027	.100	.040	**.836**											
% N	-.042	.076	.047	**.841**	**.974**										
NO3-N	-.058	-.015	-.008	**.411**	**.466**	**.524**									
NH4-N	.048	.102	-.128	**.444**	**.474**	**.426**	-.027								
P	-.109	.041	-.067	**.365**	**.385**	**.314**	-.176	**.711**							
oP	-.082	.027	-.091	**.355**	**.398**	**.336**	-.220	**.757**	**.901**						
K	**-.214**	-.122	**-.214**	.041	.174	.160	**.207**	**.252**	**.370**	**.324**					
Ca	.150	.044	.023	**-.286**	**-.244**	-.181	**.354**	**-.514**	**-.606**	**-.615**	.078				
Mg	.045	.007	.027	-.035	-.030	.054	**.489**	**-.545**	**-.595**	**-.632**	.075	**.877**			
Fe	-.003	.115	-.086	**.448**	**.513**	**.429**	-.085	**.692**	**.759**	**.763**	**.389**	**-.429**	**-.418**		
Mn	**.224**	.141	.057	**-.284**	**-.262**	**-.215**	.165	**-.312**	**-.477**	**-.436**	-.034	**.838**	**.658**	**-.329**	
Al	**-.214**	-.073	-.025	**.204**	**.333**	**.306**	.129	**.259**	**.542**	**.429**	**.636**	-.033	.076	**.552**	-.040

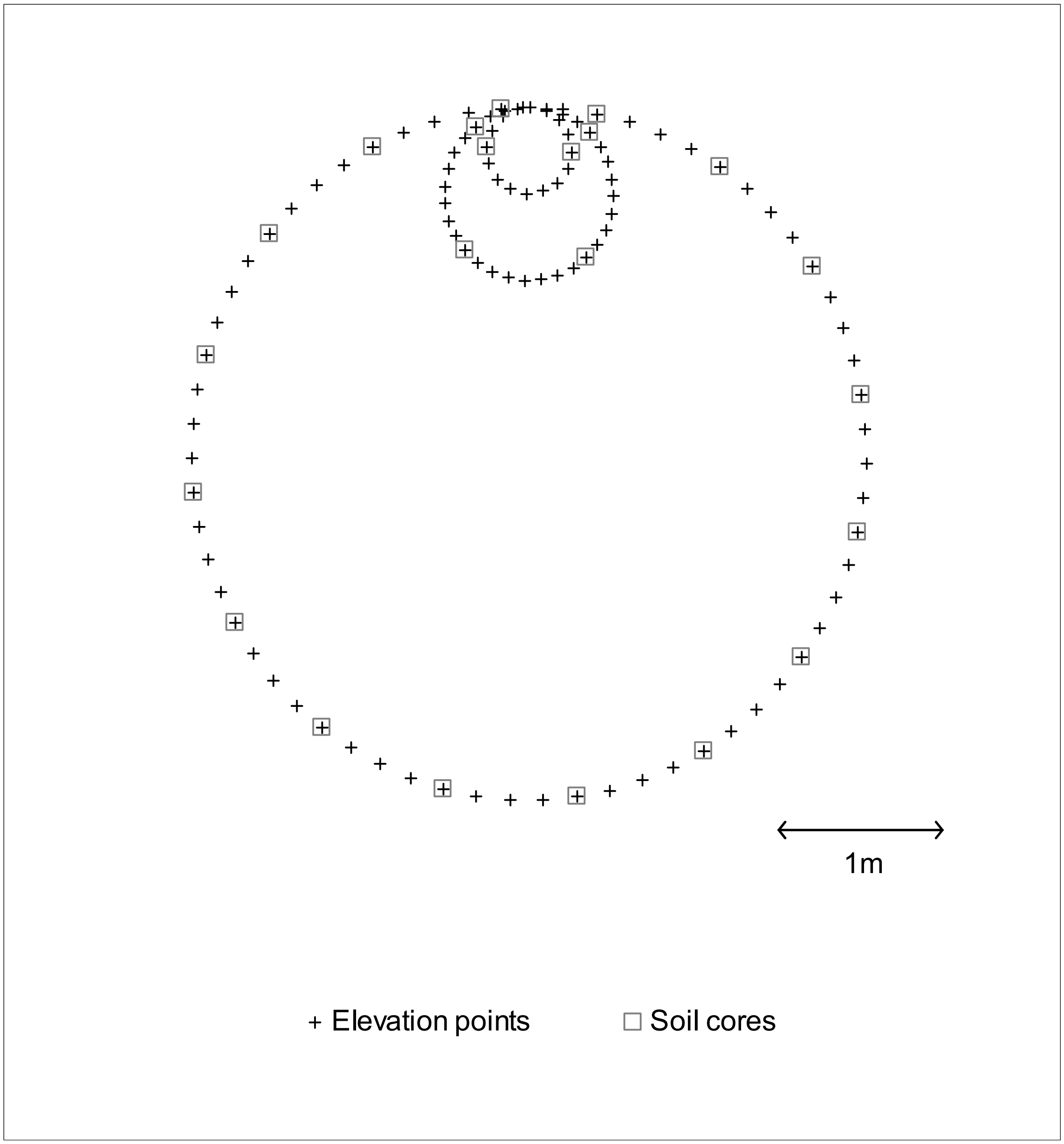
Table 3: Analysis of covariance (ANCOVA) adjustment of means for moisture content. Unadjusted and adjusted means; *p*-value for significance of covariate. na = not adjusted.

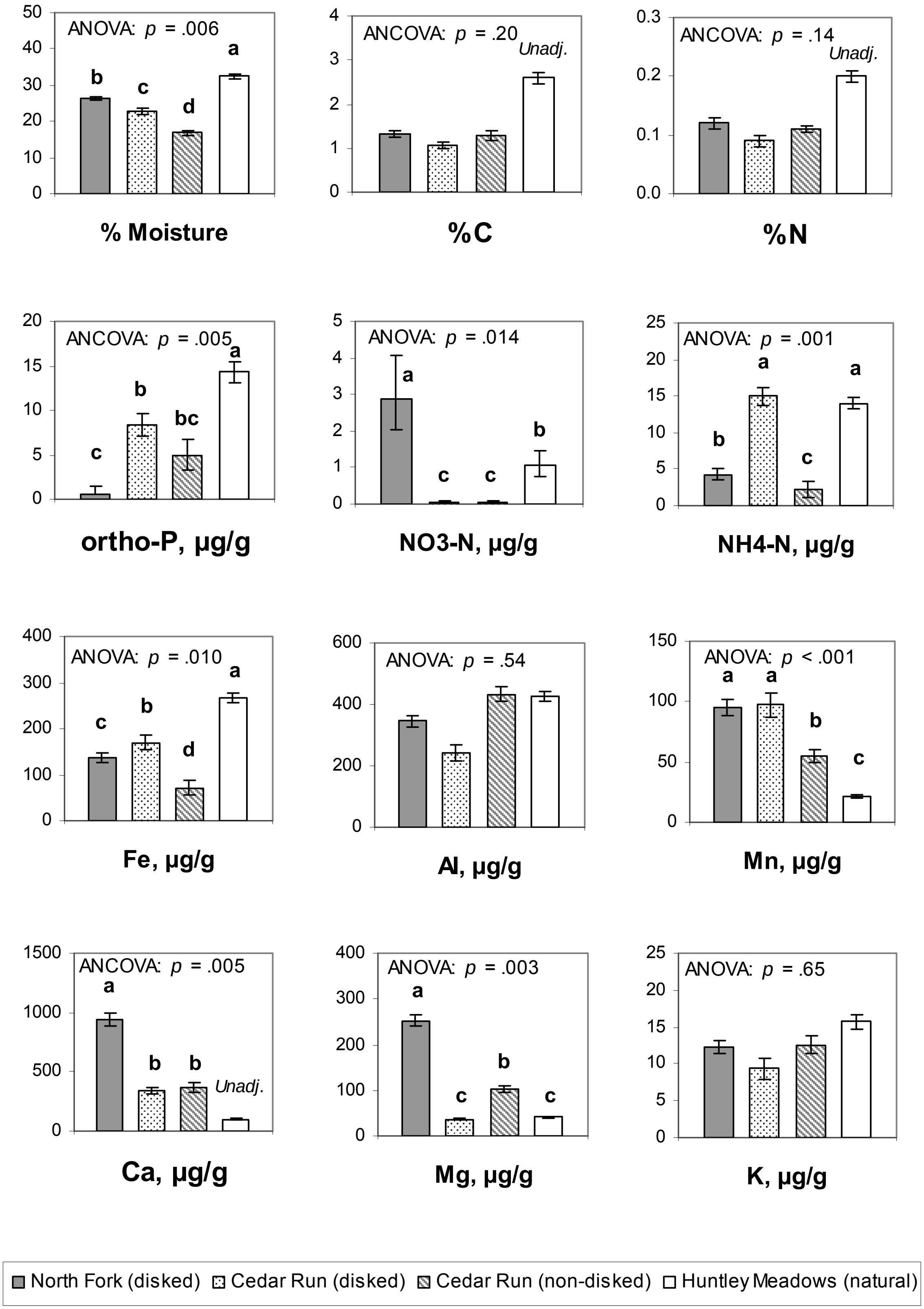
	North Fork (disked)		Cedar Run (disked)		Cedar Run (non-disked)		Huntley Meadows (natural)		
	Unadj	*Adj*	*Unadj*	*Adj*	*Unadj*	*Adj*	*Unadj*	*Adj*	*p*
C %	1.59	1.31	1.00	1.06	0.67	1.27	2.60	na	< .001
N %	0.14	0.12	0.09	0.09	0.07	0.11	0.20	na	< .001
P, µg/g	2.2	2.1	7.5	8.6	3.0	6.1	17.0	15.0	.009
ortho-P, µg/g	0.7	0.6	7.2	8.4	1.7	5.0	16.3	14.3	.011
Ca, µg/g	846	944	344	337	464	369	101	na	.004

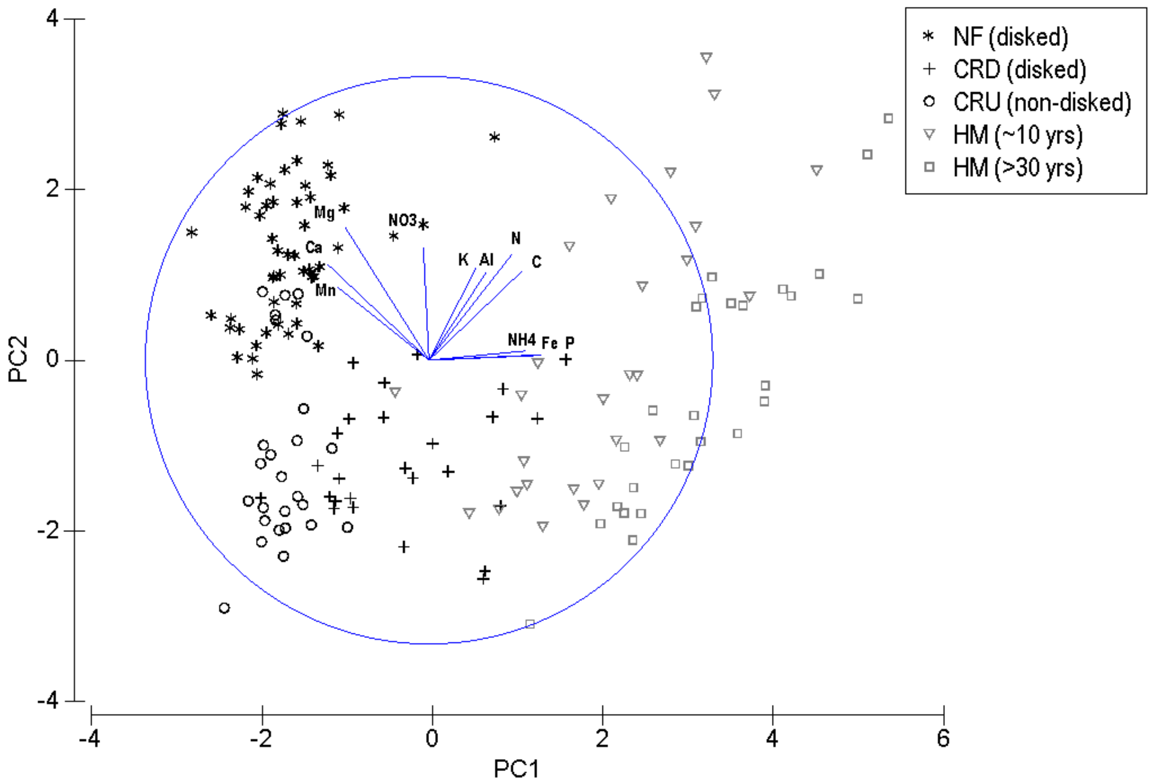
Table 4: Summary of multiple comparison groupings for equality of variance (α=0.05, Bonferroni-adjusted). Groups sharing the same letter within a row are statistically indistinguishable (A=highest variance, D=lowest). Overall test for total N was significant (α =0.05), but multiple comparisons were not. # indicates exclusion of sites from ANCOVA and associated Levene's test.

	Levene's test for equality of variance				North Fork (disked)	Cedar Run (disked)	Cedar Run (non-disked)	Huntley Meadows (natural)
	df1	*df2*	*F*	*p*				
Moisture	11	150	5.63	< .001	B	B	C	A
Total C #	7	97	1.91	.077	—	—	—	#
Total N #	7	97	2.62	.016	—	—	—	#
NO_3-N	11	147	9.70	< .001	A	C	C	B
NH_4-N	11	147	13.87	< .001	C	A	D	B
Total P (ICP)	11	149	4.09	< .001	C	A	BC	AB
o-P (colorimetry)	11	149	7.16	< .001	B	A	B	A
K	11	149	2.61	.005	A	A	AB	B
Ca #	7	98	1.16	.33	—	—	—	#
Mg	11	149	2.40	.009	A	AB	AB	B
Mn	11	149	0.61	.82	—	—	—	—
Fe	11	149	4.56	< .001	B	A	C	AB
Al	11	149	2.85	.002	B	AB	B	A

LIST OF FIGURES


1m
+ Elevation points
□ Soil cores


ANOVA: p = .006
% Moisture
ANCOVA: p = .20
Unadj.
%C
ANCOVA: p = .14
Unadj.
%N
ANCOVA: p = .005
ortho-P, µg/g
ANOVA: p = .014
NO3-N, µg/g
ANOVA: p = .001
NH4-N, µg/g
ANOVA: p = .010
Fe, µg/g
ANOVA: p = .54
Al, µg/g
ANOVA: p < .001
Mn, µg/g
ANCOVA: p = .005
Unadj.
Ca, µg/g
ANOVA: p = .003
Mg, µg/g
ANOVA: p = .65
K, µg/g
North Fork (disked)
Cedar Run (disked)
Cedar Run (non-disked)
Huntley Meadows (natural)


NF (disked)
CRD (disked)
CRU (non-disked)
HM (~10 yrs)
HM (>30 yrs)
Mg
NO3
Ca
Mn
K
Al
N
C
NH4
Fe
P
PC2
PC1
4
2
0
-2
-4
-4
-2
0
2
4
6

USGS Grant No. 08HQGR0004 A Weight of Evidence Screening Value Approach to Nutrient Criteria Development for Wadeable Streams

Basic Information

Title:	USGS Grant No. 08HQGR0004 A Weight of Evidence Screening Value Approach to Nutrient Criteria Development for Wadeable Streams
Project Number:	2007VA123S
Start Date:	1/1/2008
End Date:	6/30/2009
Funding Source:	Supplemental
Congressional District:	09
Research Category:	Water Quality
Focus Category:	Water Quality, Surface Water, Methods
Descriptors:	
Principal Investigators:	Tamim Younos

Publication

1. None

Progress Report 3

Project Title
A Weight of Evidence Screening Value Approach to Nutrient Criteria Development for Wadeable Streams

Submitted by
Dr. Tamim Younos

Date: April 23, 2009

Project Objective
The objective of this project is to conduct a trial run of a weight of evidence nutrient-criteria screening value approach for wadeable freshwater streams in Aggregate Nutrient Ecoregions IX and XI. The Virginia Department of Environmental Quality (DEQ) biologists work with the Academic Advisory Committee (AAC), a scientific advisory group, to develop and implement the proposed approach for the project. The DEQ will use AAC recommendations for initiating a notice of intended regulatory action for developing nutrient criteria for wadeable freshwater streams.

Summary of Accomplishments
In spring 2008 the Virginia Department of Environmental Quality (DEQ) initiated a pilot project involving stream sampling of benthic macroinvertebrate communities along with algae, chlorophyll and water samples at 29 selected sites. Additional sampling at 33 other sites occurred during the fall of 2008. The DEQ transferred data to the Academic Advisory Committee (AAC) for analysis. The AAC compiled the data and conducted a preliminary analysis. Two major findings are as follows:

- DEQ biologists were able to apply the visual assessment procedure to identify impaired sites successfully. However, attempted application of the visual assessment procedure to identify non-impaired sites was not successful.
- Results of the preliminary analysis indicated that pilot program data were not adequate for establishment of TN and TP screening values or critical values. An alternative procedure for identifying screening values was developed and applied using DEQ's probabilistic monitoring data (2001 – 2006). This application appeared to be successful.

The AAC submitted results of preliminary analysis to DEQ in early March 2009. A joint DEQ-AAC meeting was held on March 19, 2009 in Charlottesville, Virginia to discuss results of the preliminary analysis. Comments from the meeting will be incorporated in the draft final report that will be submitted to DEQ by June 1, 2009.

The project process and progress are fully explained in the four appendices attached to this report.

Report Appendices

Appendix I.
Graphic illustration of AAC recommended approach to nutrient criteria development.

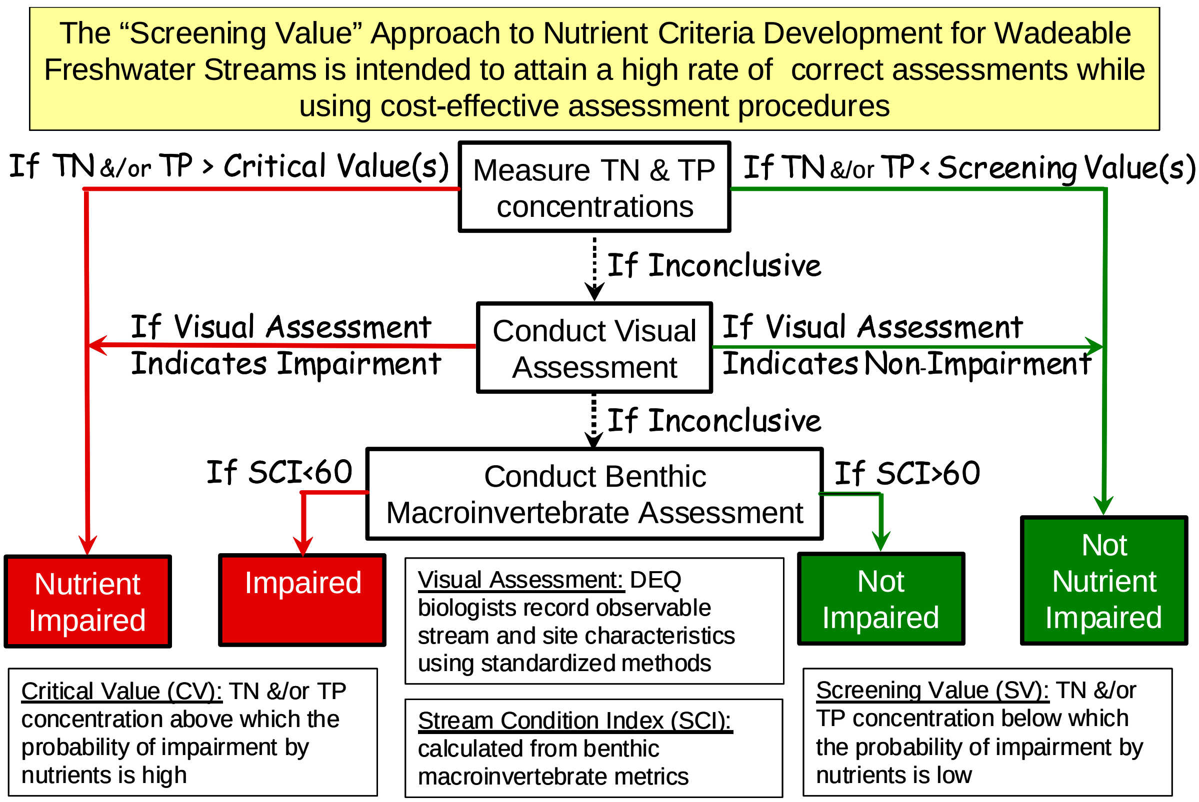
Appendix II.
Pilot Program Summary Report

Appendix III.
Pilot Program Summary Slide Presentation

Appendix IV.
Pilot Program Data.

Appendix I

Graphic illustration of AAC recommended approach to nutrient criteria development.



Appendix II: Pilot Program Summary Report

General Description: The Nutrient Criteria Pilot Program was conducted by Virginia DEQ regional biologists in spring and fall of 2008. Data from 62 sites were obtained, 29 in the spring and 33 in the fall. Benthic macroinvertebrate assessments were replicated at 1 spring and 3 fall sites. Impairment status (i.e. whether or not SCI<60) for replicate samples did not differ from the primary sample, so only primary sample results are used in the following analysis. Minor adjustments were made in the visual assessment form after the spring sampling, as several new assessment procedures were added in response to the spring experience. Sites were selected for inclusion in the study based on previously measured TN and TP concentrations, with the intention of ensuring sufficient representation of high-nutrient streams to allow characterization of the high-nutrient effects that are of primary interest in this study. Basic data for streams included in the study follow:

	SCI<60	SCI>60	all
n	36	26	62
TN (median, mg/L)	0.85*	0.47	0.61
NO3-N (median, mg/L)	0.54*	0.10	0.25
TKN (median, mg/L)	0.4	0.4	0.4
TP (median, mg/L)	0.045	0.03	0.04
Benthic algae: Ash free dry matter (AFDM, median, mg/m^2)	20.8	16.6	17.6
Benthic Algae: Chlorophyll a (Chl-a, median, mg/m^2).	56.8	27.0	39.5
SCI (mean)	47.5	68.3	57.3

* = significantly different (p<.05) vs. SCI>60 sites. Other water quality and benthic algae measures are not significantly different.

Biochemical relationships: In general, the biochemical relationships occurred as predicted: high nutrient concentrations, high algae/plant densities, and low SCI scores were all correlated; but those relationships, although often statistically significant and sometimes highly significant, did not provide a basis for development of predictive models with potential for precise application because of high variance. Generally speaking, relationships with benthic algae and SCI are stronger for N than for P, and are stronger for TN than for either of the two major TN components (TKN, NO3).

Visual assessments: Sites identified by biologists as having a “high probability” of being impaired (SCI<60) usually were impaired (8 of 9 total, and 6 of 7 rated as “high probability” for nutrient impairment), but the visual assessments were not as successful at the other end of the spectrum; i.e. a number of the sites identified as having a low probability of being impaired were, in fact, impaired (3 of 7 during the fall, when both nutrient and non-nutrient impairment probabilities were visually assessed). Nutrient effects were evident at the one non-impaired site rated as having as having high-probability of impairment, as it was visually assessed as having 40-70% of the stream bottom covered by algae (predominantly tall filamentous) and plants. Certainly, one reason for difficulty at the lower end is that non-nutrient stressors were also acting at a number of sites; we know that because of the comments the biologists wrote on the data sheets; by far the most common non-nutrient stressor cited was sediments. Data sheet comments indicated that the visual presence of plants and algae was a primary factor considered in estimating probabilities of impairment by nutrients. The biologists' visual assessments of algae presence tended to agree with the in-stream measurements but again with high variance. AFDM corresponded more closely with biologists' visual assessments of stream-bottom coverage by algae than did Chl-a.

Each fall stream was rated for probability of impairment by both nutrient and non-nutrient stressors; SCI tended to correspond with these ratings, on average (i.e. mean SCIs decreased along the progression of "low" to "medium" to "high" probability of impairment), but this result was not statistically significant.

Each Fall stream was rated visually for "total stream bottom coverage by algae and vascular plant growth;' this metric showed no statistically significant relationship with SCI, and nominal relationships did not confirm the expected trends; of the 4 stream-bottom coverage categories (<10%, 10-40%, 40-70%, and >70%), the <10% category showed the highest proportion of impairments (3 of 3) while the 40-70% category showed the lowest proportion (2 of 8). 11 of the 15 streams with >70% stream bottom coverage were impaired (SCI<60), but the two highest SCI's among fall-sampled streams were also within this (>70%) visual-assessment category.

<u>Critical Values:</u> "Critical values" are defined in the study plan as in-stream concentrations that allow the stream to be assessed for nutrient impairment. Critical values can be relatively high concentrations that allow sites to be identified as impaired and/or relatively low concentrations that allow sites to be identified as not impaired. No low-end critical values were evident from this dataset, possibly because the dataset does not allow discrimination of nutrient from non-nutrient impairment. High-end critical values (*i.e.* values above which all sites had SCI<60) were evident (see table below), but they would have been adequate to assess only a very small proportion of the tested sites. The adequacy of potential high-end critical values for water quality and benthic algae can be checked using the probabilistic monitoring dataset.

Parameter	**Critical Value (CV)**	**# sites > CV***
Benthic algae Chl-a	170 mg/m^2	4
Benthic algae AFDM	70 g/m^2	5
TN	2.6 mg/L	6
NO3-N	2.3 mg/L	6
TKN	0.9 mg/L	4
TP	0.4 mg/L	4
TN, TP, NO3, TKN (WQ)	Combined	10
WQ + benthic algae	Combined	13
Best Professional Judgment (BPJ)	High (nutrients only)	7 (6 SCI<60)
WQ + BPJ	Combined	13
WQ + Benthic Algae + BPJ	Combined	14

*Out of 62 total sites and 36 impaired (SCI<60) sites in pilot program. At 32 sites, SCI<57.5; of the 4 remaining sites ("borderline impaired"), 1 was caught by the AFDM screen but none were caught by the WQ or BPJ screens.

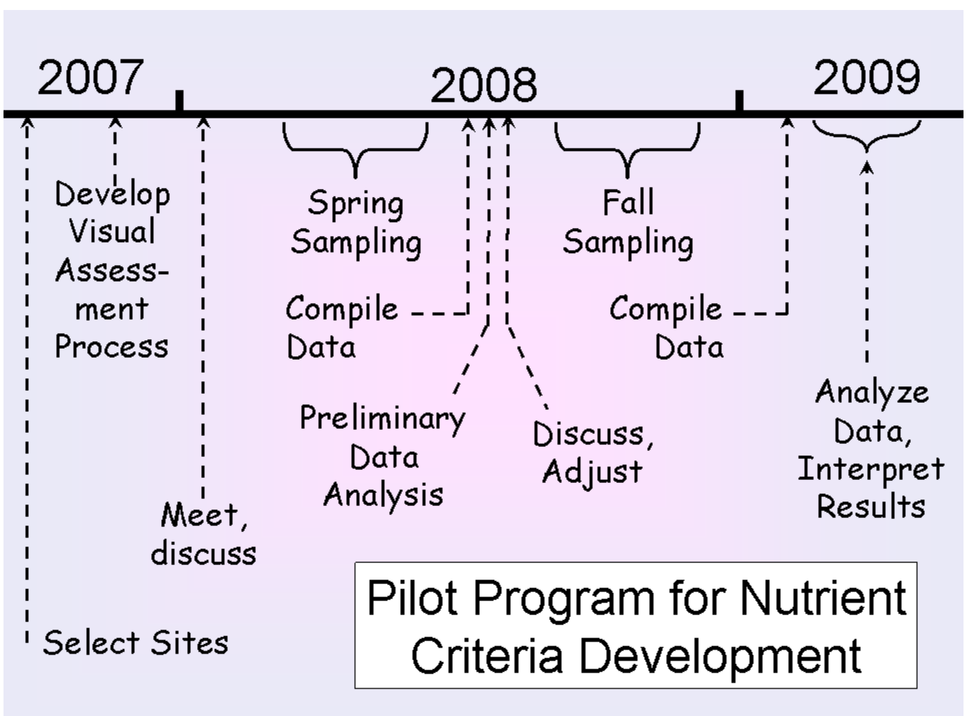
Appendix III

Pilot Program Summary Slide Presentation

"Pilot Program": Application of the full "Screening Value Approach" protocol (water quality measurement, visual assessment, benthic macroinvertebrate assessment) by DEQ; analysis of resulting data by AAC.

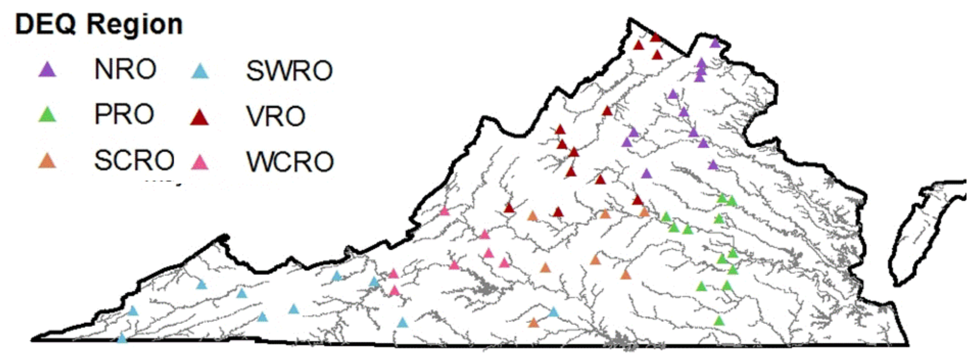
Pilot Program purpose: Evaluate the ability of the screening-value approach to achieve intended goals:

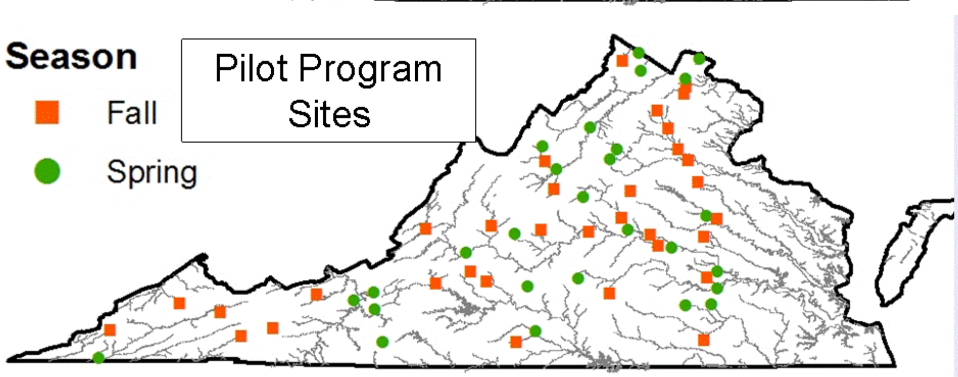
- Scientific and legally defensible criteria that protect water quality.
- Reduce error rates to acceptable levels.
- Can be implemented by DEQ with available resources.



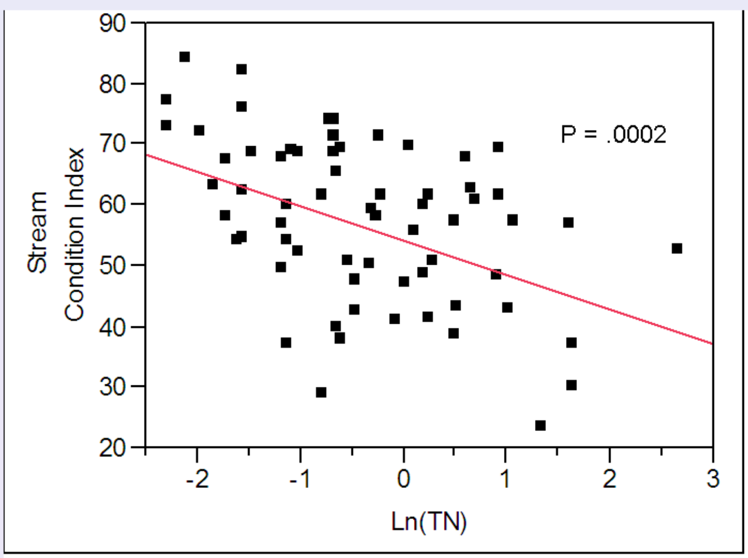
Pilot Program Site Selection Process

1. Wadeable, suitable for benthic collection.
2. Not known to be influenced by major urban inputs, toxic inputs likely to cause benthic impairments, point source discharges.
3. Sites distributed among TN and TP levels so as to represent a range of nutrient conditions.
4. Choose sites that are not clustered and are representative of the entire region.



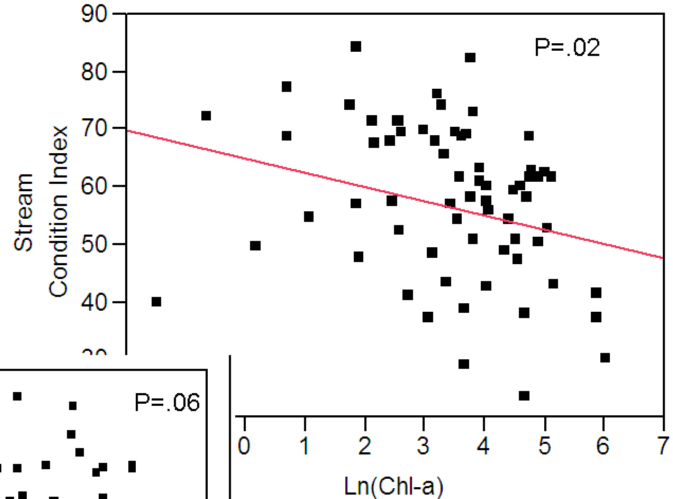


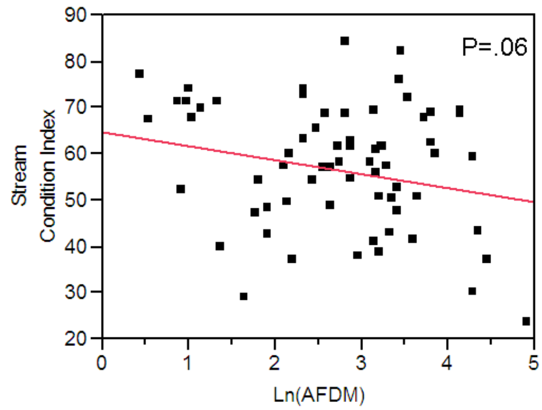
In general, the biochemical relationships occurred as predicted: high nutrient concentrations, high algal densities, and low SCI scores were all correlated; but those relationships, although often statistically significant and sometimes highly significant, did not provide a basis for development of predictive models with potential for precise application because of high variance.



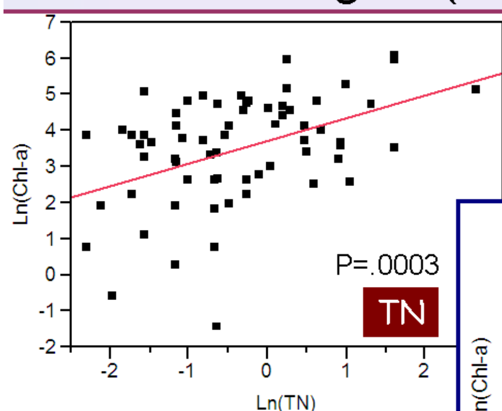
Influences of TN, NO3-N, and TKN concentrations on Stream Condition Index are all negative and statistically significant ($p<.05$). TN exhibits the strongest relationship (see above). Relationship of TP to SCI is not statistically significant. (Nutrient concentrations were log-transformed to near-normal distributions).

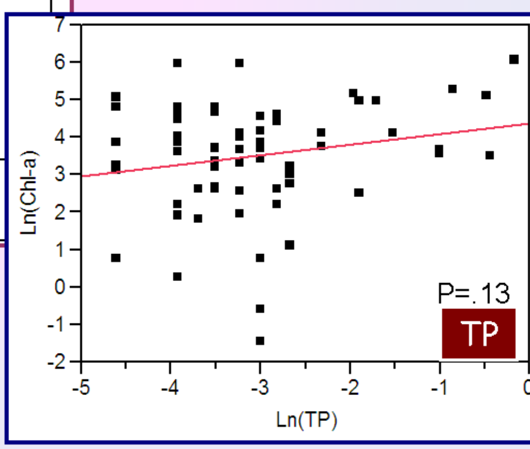
SCI vs. two measures of benthic algae biomass: Chlorophyll a (mg/m^2) and ash-free dry matter (g/m^2).





Benthic Algae (Chl-a) vs. in-stream concentrations





AFDM vs. TN, TP relationships are similar but not as strong

Were Regional Biologists Able to Determine Impairment/Non-Impairment Successfully Using Visual Assessments?

Biologists rated each stream for probability of impairment by nutrients during both spring and fall, and for impairment by non-nutrient stressors during fall only.

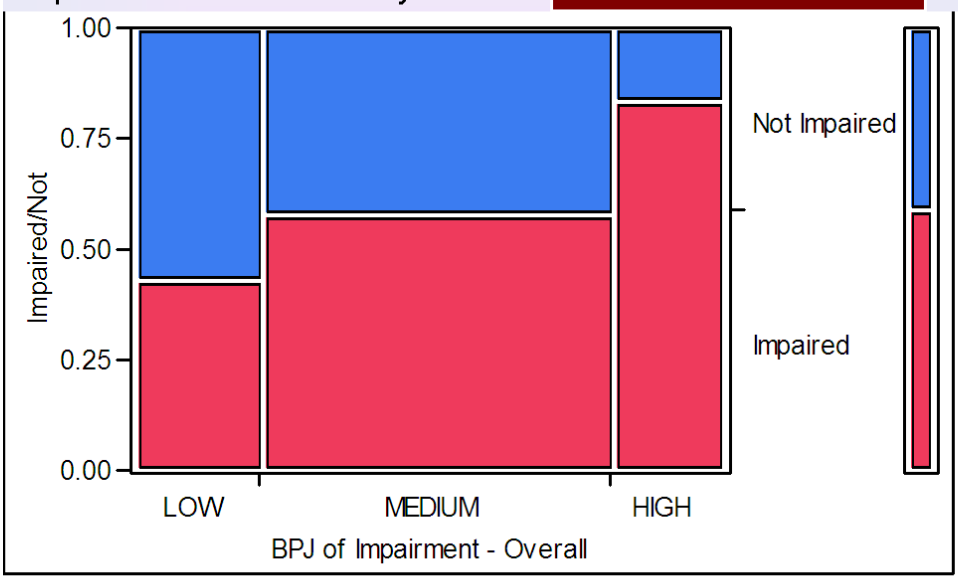
Biologists recorded estimates of stream-bottom coverage by benthic algae and vascular plant growth during fall only.

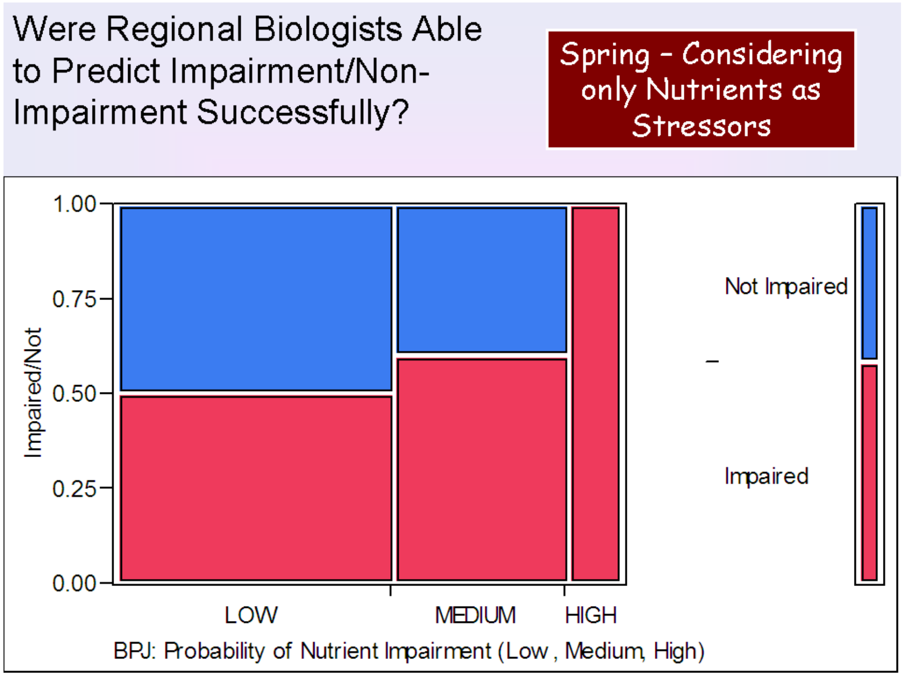
Biologists recorded visual perceptions of stream bottom coverage by specific algae types during spring and fall. An Algal index was computed for each site by summing those metrics as recorded, weighted as follows:

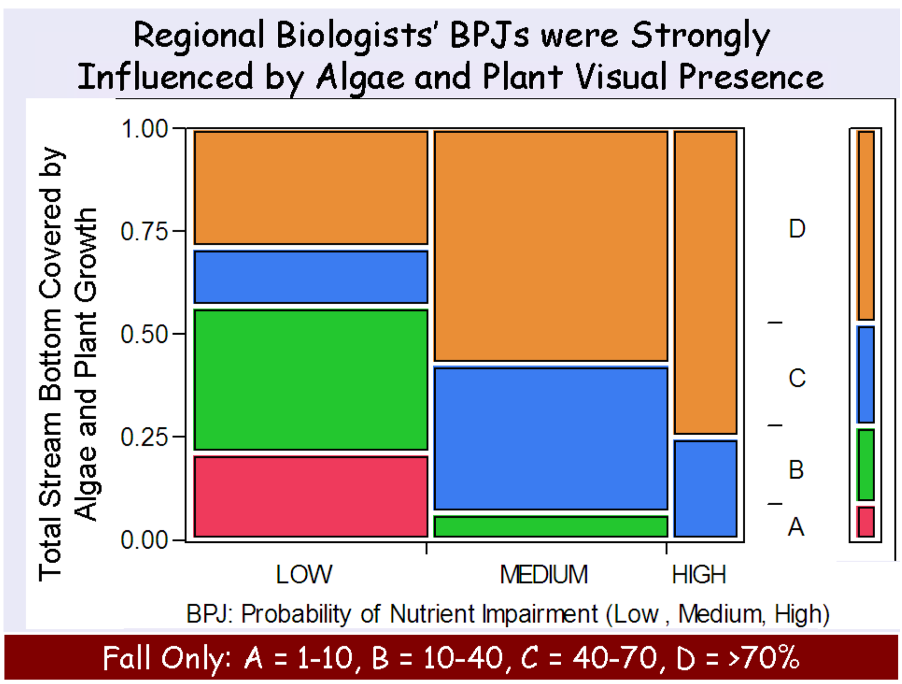
A = 1-10% stream bottom coverage, weighted as x1; B = 10-40% coverage, weighted as x3; C = 40-70% coverage, weighted as x6; D = >70% coverage, weighted as x10.

Were Regional Biologists Able to Predict Impairment/Non-Impairment Successfully?

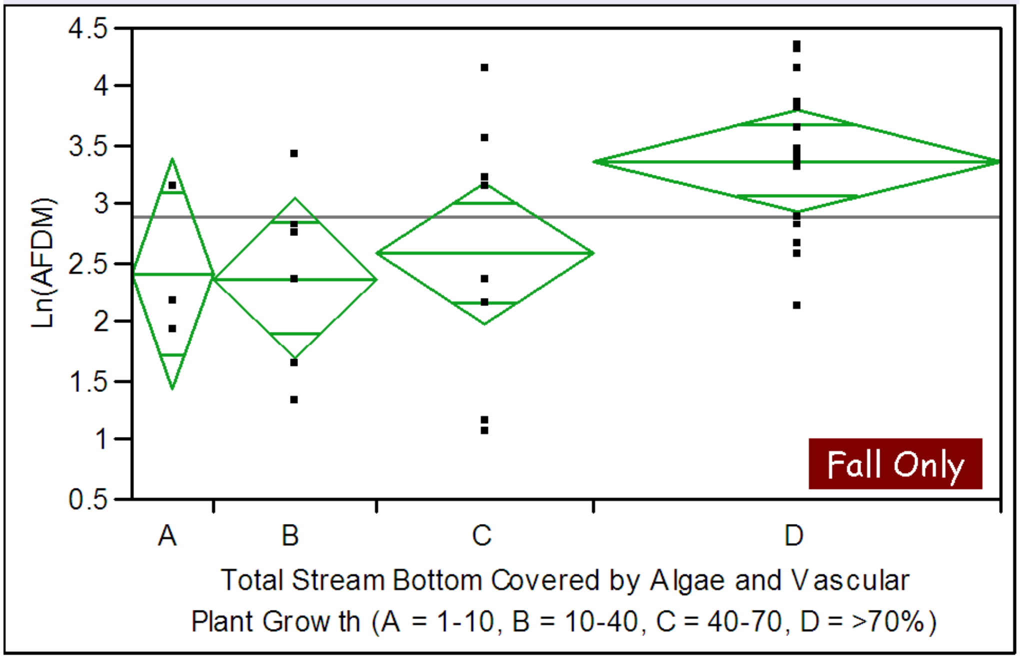
Fall Only, considering both Nutrient and Non-Nutrient Stressors

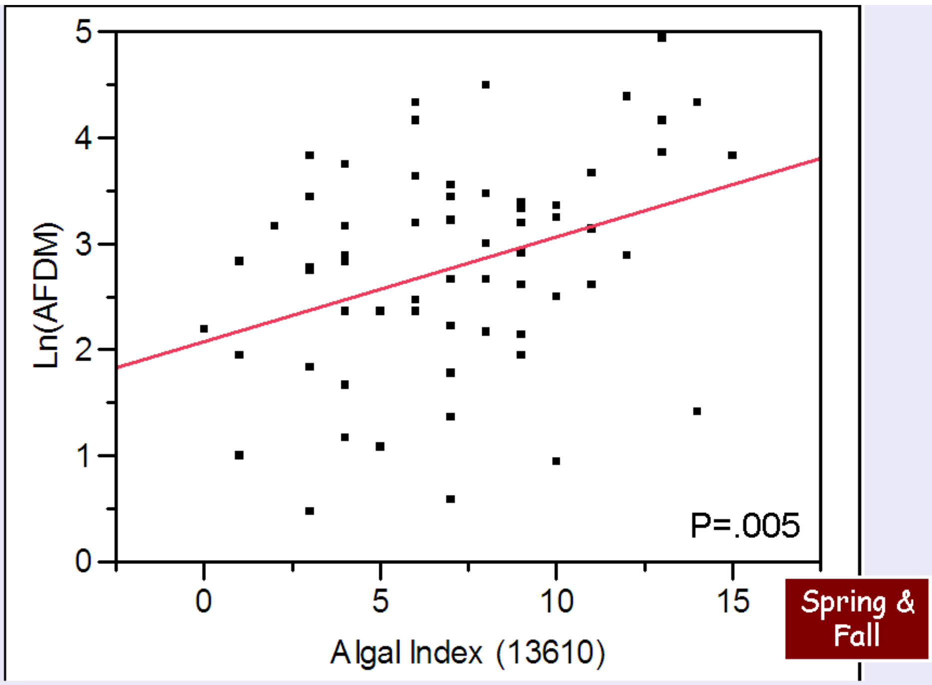



Were Regional Biologists Able to Predict Impairment/Non-Impairment Successfully?
Spring - Considering only Nutrients as Stressors
1.00
0.75
0.50
0.25
0.00
Impaired/Not
Not Impaired
Impaired
LOW
MEDIUM
HIGH
BPJ: Probability of Nutrient Impairment (Low , Medium, High)

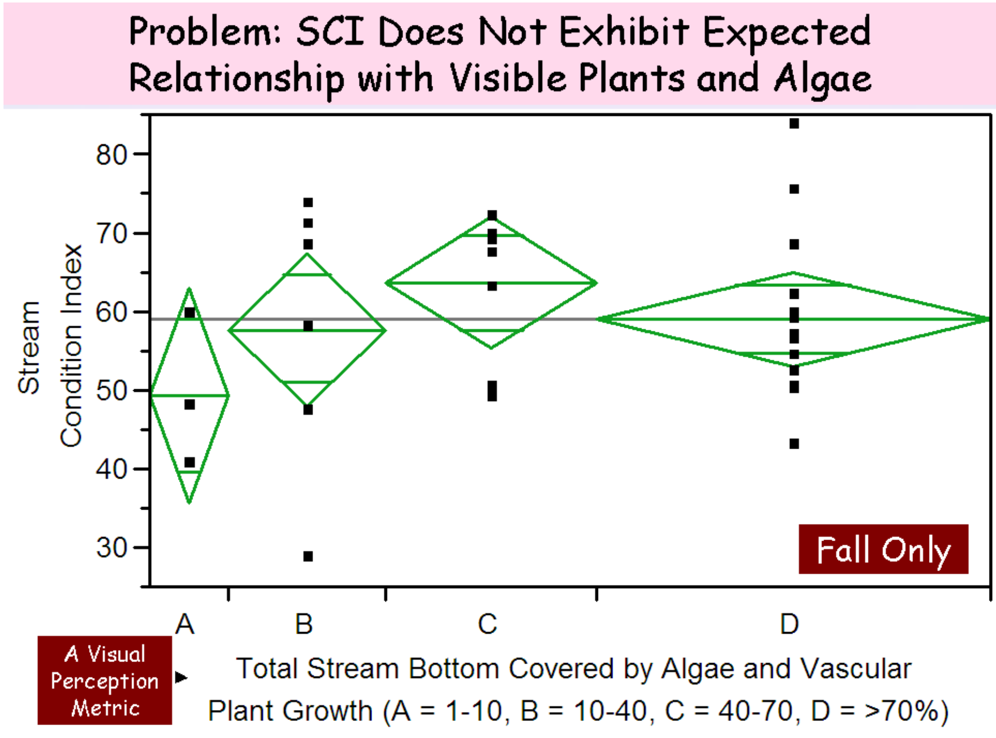

Regional Biologists' BPJs were Strongly Influenced by Algae and Plant Visual Presence
1.00
0.75
0.50
0.25
0.00
Total Stream Bottom Covered by Algae and Plant Growth
D
C
B
A
LOW
MEDIUM
HIGH
BPJ: Probability of Nutrient Impairment (Low , Medium, High)
Fall Only: A = 1-10, B = 10-40, C = 40-70, D = >70%

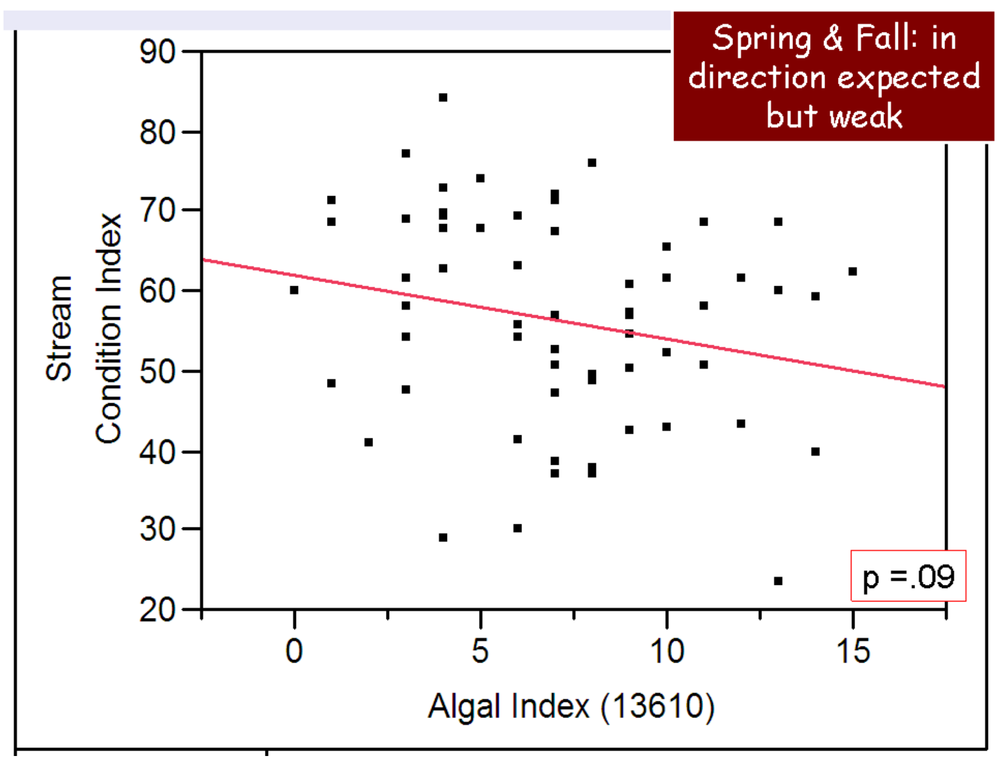
Regional Biologists' Perceptions of Plant/Algal Presence Were Consistent with Measured Algae.





Regional Biologists' Perceptions of Algal Presence Were Consistent with Benthic Algal Biomass Measurements.


Problem: SCI Does Not Exhibit Expected Relationship with Visible Plants and Algae
80
70
60
50
40
30
Stream Condition Index
Fall Only
A
B
C
D
A Visual Perception Metric
Total Stream Bottom Covered by Algae and Vascular Plant Growth (A = 1-10, B = 10-40, C = 40-70, D = >70%)


Spring & Fall: in direction expected but weak
90
80
70
60
50
40
30
20
Stream Condition Index
p =.09
0
5
10
15
Algal Index (13610)

Non-nutrient stressor effects were evident in Visual Assessment at Pilot Program sites

Sediments Cited as Non-Nutrient Stressor

	Sediments Cited	# of Sites
Spring	5	29
Fall	17	33
Total	23	62

Non-nutrient stressor effects were addressed specifically by the Visual Assessment data form during Fall only. In spring, sediment effects were noted as general comments.

Conclusions:

Biologists Applied Visual Assessment procedure successfully to identify SCI<60 sites.

Attempted identification of non-impaired sites was not successful, possibly because non-nutrient stressors were having an effect.

Data do not allow determinaton of whether "not nutrient impaired" designations are correct.

Appendix IV. Pilot Program Data.

Hypothetical application to Pilot Program sites of illustrative screening and critical values within AAC recommended approach.

StationID	Season	TN (mg/L)	TP (mg/L)	BPJ: Prob Nutrient Impairment	BPJ: Prob Non-Nutrient Impairment	CV: TN>1.8	CV: TP>0.1	SV: TN<0.81 & TP<0.05	BPJ	Outcome	Stream Condition Index
6BPLU002.15	Spr	1.27	0.04	MEDIUM	-	-	-	-	-	BenMac Assess	40.94
2-PCT002.46	Spr	0.62	0.10	MEDIUM	-	-	-	-	-	BenMac Assess	42.12
1ANOG005.69	Spr	1.02	0.06	MEDIUM	-	-	-	-	-	BenMac Assess	46.81
1BSSF053.09	Spr	1.22	0.06	LOW	-	-	-	-	-	BenMac Assess	48.28
4ATKR000.69	Fal	1.34	0.05	MEDIUM	MEDIUM	-	-	-	-	BenMac Assess	50.34
6CMFH055.88	Fal	0.59	0.05	MEDIUM	MEDIUM	-	-	-	-	BenMac Assess	50.35
4ASEE003.16	Fal	0.21	0.07	MEDIUM	LOW	-	-	-	-	BenMac Assess	54.37
2-CNE000.96	Spr	1.11	0.05	LOW	-	-	-	-	-	BenMac Assess	55.31
6BIDN000.69	Fal	1.22	0.03	MEDIUM	LOW	-	-	-	-	BenMac Assess	59.81
5AGRV000.08	Spr	0.51	0.05	LOW	-	-	-	-	-	BenMac Assess	68.20
2-NOR000.20	Spr	0.34	0.10	LOW	-	-	-	-	-	BenMac Assess	68.62
3-MTN000.59	Fal	1.05	0.07	MEDIUM	MEDIUM	-	-	-	-	BenMac Assess	69.51
3-RAP006.53 (S1)	Fal	0.78	0.06	LOW	MEDIUM	-	-	-	-	BenMac Assess	70.99
2-HAT000.14	Fal	0.14	0.05	LOW	MEDIUM	-	-	-	-	BenMac Assess	71.86
1ASYL000.02	Spr	3.77	0.02	MEDIUM	-	imp	-	-	-	Impaired	23.01
1AOPE036.13	Spr	5.13	0.84	LOW	-	imp	Imp	-	-	Impaired	29.78
1BMDD005.81	Spr	5.13	0.02	MEDIUM	-	imp	-	-	-	Impaired	36.66
9-STE007.29	Spr	1.63	0.05	HIGH	-	-	-	-	Imp	Impaired	38.18
5ABTR002.80	Spr	0.52	0.05	HIGH	-	-	-	-	Imp	Impaired	39.41

2-CHK079.23	Fal	0.92	0.07	LOW	HIGH	-	-	-	Imp	Impaired	40.62
4AMEY016.00	Spr	2.76	0.43	HIGH	-	imp	Imp	-	Imp	Impaired	42.66
1BCKS001.03	Fal	1.66	0.05	HIGH	HIGH	-	-	-	Imp	Impaired	42.98
3-THM001.40	Fal	2.48	0.07	LOW	MEDIUM	imp	-	-	-	Impaired	47.94
2-JKS018.68	Fal	0.72	0.15	HIGH	LOW	-	Imp	-	Imp	Impaired	50.13
3-GRT001.70	Fal	14.2	0.62	HIGH	LOW	imp	Imp	-	Imp	Impaired	52.43
4ALOR008.64	Fal	5.04	0.64	MEDIUM	MEDIUM	imp	Imp	-	-	Impaired	56.52
1BSTH019.52	Fal	1.62	0.22	LOW	LOW	-	Imp	-	-	Impaired	56.78
2-SOL001.00	Fal	2.86	0.04	LOW	MEDIUM	imp	-	-	-	Impaired	57.11
6BPOW179.20	Fal	0.74	0.02	MEDIUM	HIGH	-	-	NotNI	Imp	Impaired	59.03
9-DEN000.03	Spr	1.99	0.04	MEDIUM	-	imp	-	-	-	Impaired	60.44
1BSTH002.14	Spr	1.28	0.14	MEDIUM	-	-	Imp	-	-	Impaired	61.15
2-APP012.79	Spr	0.45	0.18	MEDIUM	-	-	Imp	-	-	Impaired	61.29
9-MLC005.44	Spr	1.91	0.03	LOW	-	imp	-	-	-	Impaired	62.34
6CMFH033.40	Fal	1.83	0.15	LOW	LOW	imp	Imp	-	-	Impaired	67.39
2-RVN015.97 (S1)*	Fal	2.54	0.37	HIGH	MEDIUM	imp	Imp	-	Imp	Impaired	69.06
1ALIV012.12	Fal	0.45	0.03	LOW	LOW	-	-	NotNI	-	Not Nut Imp	28.56
6ASAT000.26	Spr	0.32	0.01	LOW	-	-	-	NotNI	-	Not Nut Imp	36.70
2-IVC010.20	Spr	0.54	0.02	LOW	-	-	-	NotNI	-	Not Nut Imp	37.37
2-MTC001.24	Fal	0.62	0.04	LOW	MEDIUM	-	-	NotNI	-	Not Nut Imp	47.17
2-LIH005.28	Fal	0.31	0.02	MEDIUM	MEDIUM	-	-	NotNI	-	Not Nut Imp	49.07
8-LTL009.54	Spr	0.36	0.03	LOW	-	-	-	NotNI	-	Not Nut Imp	52.02
3-RAP077.28	Spr	0.32	0.02	LOW	-	-	-	NotNI	-	Not Nut Imp	53.66
9-LTL001.22	Spr	0.20	0.02	LOW	-	-	-	NotNI	-	Not Nut Imp	54.01
2-LIA000.50	Fal	0.31	0.02	MEDIUM	MEDIUM	-	-	NotNI	-	Not Nut Imp	56.40
6BWAL005.97	Spr	0.77	0.03	MEDIUM	-	-	-	NotNI	-	Not Nut Imp	57.56
6AIND000.52	Fal	0.18	0.02	MEDIUM	MEDIUM	-	-	NotNI	-	Not Nut Imp	57.88
8-SAR097.82	Fal	0.32	0.04	LOW	NO	-	-	NotNI	-	Not Nut Imp	59.81
1ACAX004.57	Spr	0.80	0.02	LOW	-	-	-	NotNI	-	Not Nut Imp	61.39
2-MIS000.04	Fal	0.21	0.01	LOW	MEDIUM	-	-	NotNI	-	Not Nut Imp	62.16

9-NBS000.70	Fal	0.16	0.02	MEDIUM	LOW	-	-	NotNI	-	Not Nut Imp	62.93
2-FIN000.81	Spr	0.52	0.03	MEDIUM	-	-	-	NotNI	-	Not Nut Imp	65.18
3-ROB023.06	Spr	0.18	0.02	LOW	-	-	-	NotNI	-	Not Nut Imp	66.98
4ASNA015.30	Fal	0.31	0.03	LOW	-	-	-	NotNI	-	Not Nut Imp	67.42
1AHOC006.23	Fal	0.36	0.01	MEDIUM	MEDIUM	-	-	NotNI	-	Not Nut Imp	68.21
1AGOO022.44	Fal	0.23	0.04	LOW	LOW	-	-	NotNI	-	Not Nut Imp	68.38
5ATRE038.07	Fal	0.54	0.03	MEDIUM	MEDIUM	-	-	NotNI	-	Not Nut Imp	68.91
6CSFH097.42 (S1)	Spr	0.52	0.03	LOW	-	-	-	NotNI	-	Not Nut Imp	71.04
2-JES000.80	Spr	0.10	0.02	LOW	-	-	-	NotNI	-	Not Nut Imp	72.61
8-POR008.97	Fal	0.49	0.04	LOW	LOW	-	-	NotNI	-	Not Nut Imp	73.77
8-NAR005.42 (S1)	Fal	0.21	0.01	MEDIUM	LOW	-	-	NotNI	-	Not Nut Imp	75.48
2-BNF003.52	Spr	0.10	0.01	LOW	-	-	-	NotNI	-	Not Nut Imp	76.91
2-RKI003.40	Fal	0.12	0.02	LOW	LOW	-	-	NotNI	-	Not Nut Imp	83.64

* SCI for 2-RVN015.97 (S2) was 61.22.

Information Transfer Program Introduction

The VWRRC supports timely dissemination of science-based information to policy and decision-making bodies and citizens. The VWRRC used its 104 funds to support expert personnel with responsibilities related to the VWRRC's outreach and collaborative programs. In FY 2008, the 104 funds supported:

1. Preparation and electronic publication of the newsletter Virginia Water Central

2. Partial support for organizing the 2008 Mid-Atlantic Regional Water Resources Research Conference

3. Partial administrative support for the Virginia Water Monitoring Council

4. Partial support for management of the VWRRC webpage

Information Dissemination

Basic Information

Title:	Information Dissemination
Project Number:	2006VA97B
Start Date:	3/1/2006
End Date:	2/28/2009
Funding Source:	104B
Congressional District:	9th
Research Category:	Not Applicable
Focus Category:	None, None, None
Descriptors:	
Principal Investigators:	Stephen H. Schoenholtz

Publication

1. Warren, P. M. and T. Younos. 2008. Analysis of Nutrient-Response Characteristics to Support Criteria Development for Constructed Reservoirs. VWRR SR37-2008. 38 pp. http://www.vwrrc.vt.edu/special_reports.html#2008
2. Gowland, D. and T. Younos. 2008. Feasibility of Rainwater Harvesting BMP for Stormwater Management. VWRRC SR38-2008. 23pp. http://www.vwrrc.vt.edu/special_reports.html#2008.
3. Grady, C. and T. Younos. 2008. SR39-2008. Analysis of Water and Energy Conservation of Rainwater Capture System on a Single Family Home. VWRRC SR39-2008. 23 pp. http://www.vwrrc.vt.edu/special_reports.html#2008
4. Adams, E. and T. Younos. 2008. Community-Based Sustainable Development Planning. VWRRC SR41-2008. 16 pp. http://www.vwrrc.vt.edu/special_reports.html#2008.
5. Young, K., T. Younos, R. Dymond and D. Kibler. 2009. Virginia's Stormwater Impact Evaluation Project: Developing an Optimization Tool for Stormwater Runoff BMPs. VWRRC SR44-2009. http://www.vwrrc.vt.edu/special_reports.html#2009
6. Lohani, V. K. and T. Younos. 2008. Implementation and Assessment of an Interdisciplinary NSF/REU Site in Watershed Sciences and Engineering. Full paper In: Proceedings of the American Society of Engineering Education (ASEE) Annual Conference, Pittsburg, PA, June 23-25, 2008.
7. Younos, T., C. Grady, T. Chen and T. Parece. 2009. Conventional and Decentralized Water Infrastructure: Energy Consumption and Carbon Footprint. Extended Abstract In: Proceedings of the American Water Resources Association 2009 Spring Specialty Conference Managing Water Resources and Development in a Changing Climate (Ed. H. Toniolo). 10pp. May 4-6, Anchorage, Alaska. ISBN: 1-882132-79-3.
8. Younos, T., V. K. Lohani, M. Licher (Editors). 2008. NSF REU 2008 Proceedings of Research: Research Opportunities in Interdisciplinary Watershed Sciences and Engineering, 90 pp. Virginia Water Resources Research Center, Virginia Tech, Blacksburg, Virginia http://www.vwrrc.vt.edu/nsf_reu.html
9. Lohani, V.K. and T. Younos. 2009. Assessment of Learning Experiences of Undergraduate Researchers in an NSF/REU site on watershed Sciences and Engineering. Abstract, In: Proceedings of the American Water Resources Association 2009 Spring Specialty Conference Managing Water Resources and Development in a Changing Climate. Anchorage, Alaska, May 4-6, 2009.
10. Jin Y, Hu Z, Wen Z. 2009. Enhancing anaerobic digestibility and phosphorus recovery of dairy manure through microwave-based thermochemical pretreatment. Water Research. (accepted)

11. Tang J, Jin Y, Wen Z. 2009. Effects of mixing on biogas production and pathogen reduction of dairy manure anaerobic digestion. March 9, 2009. In: Southwest Virginia Science Forum. Blacksburg, Virginia.
12. Jin Y, Mukhopadhyay B, Wen Z. 2009. Molecular Assessment of the Fate of Pathogenic Organisms in Dairy Manure during anaerobic treatment. In: ASABE (American Society of Agricultural and Biological Engineers) 2009 Annual Conference, Reno, NV. June 21 - 24, 2009.
13. Huntington, T.G., A.D. Richardson, K.J. McGuire, and K. Hayhoe. 2009. Climate and hydrological changes in the northeastern United States: recent trends and implications for forested and aquatic ecosystems. Canadian Journal of Forest Research 39: 199-212.
14. Tetzlaff, D., J. Seibert, K.J. McGuire, H. Laudon, D. Burns, S.M. Dunn, and C. Soulsby. 2009. How does landscape structure influence catchment transit time across different geomorphic provinces? Hydrological Processes 23(6): 945-953.
15. Manley, S.W., R.M. Kaminski, P.B. Rodrigue, J.C. Dewey, S.H. Schoenholtz, P.D. Gerard, and K.J. Reinecke. 2009. Soil and nutrient retention in winter-flooded ricefields with implications for watershed management. Journal of Soil and Water Conservation. 64(3):173-183. DOI: 10.2489/jswc 64.3.173.
16. McFarlane, K.J., S.H. Schoenholtz, and R.F. Powers. 2009. Plantation management intensity affects belowground carbon and nitrogen storage in northern California. Soil Science Society of America Journal. 73(3): 1020-1032.
17. Dent, L., D. Vick, K. Abraham, S.H. Schoenholtz, and S. Johnson. 2008. Summer temperature patterns in headwater streams of the Oregon Coast Range. Journal of American Water Resources Association. 44(4):803-813. DOI: 10.1111/j.1752-1688.2008.00204.x
18. Kelly, C.N., S.H. Schoenholtz, and M.B. Adams. 2008. Factors controlling soil nitrogen and carbon transport through riparian areas in Appalachian watersheds: role of vegetation in nutrient loss and water quality. In Proceedings of AWRA Summer Specialty Conference, Virginia Beach, VA, June 30-July 2, 2008. http://www.awra.org/proceedings/0806pro_toc.html
19. ***Virginia Water Central***

    Virginia Water Central, April 2008 (No. 44), 36pp.

    Virginia Water Central, June 2008 (No. 45), 42pp.

    Virginia Water Central, September 2008 (No. 46), 34pp.

    Virginia Water Central, December 2008 (No. 47), 31pp.

Outreach and Information Transfer Accomplishments

Newsletter -- Email distribution to 635 recipients and announcement/availability on VWRRC Web site.

Special Notifications to VWRRC List Serves

1. Virginia environmental lab certification regulation comment period (9/9/08)
2. Water-related bill inventory for 2009 General Assembly available on VWRRC Web site (1/27/09)
3. Commonwealth of Virginia Web site available to accept and inventory proposals for using federal economic stimulus funds 2/16/09

Notifications to Virginia Water Monitoring Council List Serve

Weekly water-related announcements via list serve are provided to 275 members of the VWMC. Announcements include information about conferences, workshops, total maximum daily load (TMDL) public meetings in Virginia, job openings, newly published reports, information posted on Web sites, and other pertinent information.

VWRRC Website (see www.vwrrc.vt.edu) is updated at least weekly and serves as the portal for three other websites that the VWRRC manages:

1. Virginia Water Monitoring Council (http://www.vwrrc.vt.edu/vwmc/default.asp)
2. VA DCR Stormwater BMP Clearinghouse (under development) (http://www.vwrrc.vt.edu/swc/)
3. Clinch-Powell Clean Rivers Initiative (under development)

VWRRC is now on Twitter at http://twitter.com/VaWaterCenter

In FY09 the VWRRC Web site added a Water News Grouper page which is updated daily.

Mid-Atlantic Regional Water Resources Research Institutes (WRRI) 2008 Conference

The Mid-Atlantic Regional Water Resources Research Institutes (WRRI) 2008 Conference was co-sponsored by the eight water research institutes in the Mid-Atlantic and was the first regional conference organized by WRRIs in the east. The conference was held November 17-19, 2008 at the U.S. Fish and Wildlife Service National Conservation Training Center in Shepherdstown, West Virginia. The conference theme was *The Water-Energy Nexus: a Necessary Synergy for the 21st Century* and featured 45 presentations and 112 attendees. The conference Web site is at http://wvwri.nrcce.wvu.edu/conferences/2008/WRRI/index.cfm.

VWRRC/ICTAS Water Seminar Series

Established, provided leadership, and organized a new Water Seminar Series on campus this spring in collaboration with the Institute for Critical Technology and Applied Science (ICTAS). See http://www.vwrrc.vt.edu/seminar.html. Four invited speakers will participate in the series in spring 2009:

1. Dr. Marc Edwards, VT-- *"Corrosion Control Hits Home: The Profound Implications of Premise Plumbing Corrosion."* (March 27, 2009)
2. Dr. Nicolas Zegre, West Virginia Univ. – *"In Lieu of the Paired-Catchment Approach: Hydrologic Model Change Detection at the Catchment Scale"* (April 6, 2009)
3. Dr. Jim Wigington, Western Ecology Division, US EPA – *"Prospects for Hydrologic Classification of Landscapes and Watersheds"* (April 17, 2009)
4. Dr. K. Ramesh Reddy, Univ. Florida – *"Coupled Biogeochemical Cycles in Wetlands: The Everglades as a Case Example"* (April 27, 2009)

International Outreach Activities

1. Universidad Austral de Chile

   October 29-30, 2008, Virginia Tech. Met with visiting faculty to discuss establishment of a water center, study abroad program, and student exchange opportunities.
2. State Parliament Environment & Natural Resources Committee, Parliament of Victoria, Australia

   November 22, 2008, Washington, DC. Met with Committee and made presentations on water challenges and desalination in Virginia and the mid-Atlantic Region.
3. Dominican Republic

   February-April, 2009 Assisted Virginia Tech Engineering without Borders Chapter/Dominican Republic group, including accompanying the group to the Dominican Republic for an assessment of water needs at a school and clinic near Veron.

Student Support					
Category	Section 104 Base Grant	Section 104 NCGP Award	NIWR-USGS Internship	Supplemental Awards	Total
Undergraduate	0	0	1	13	14
Masters	0	0	0	2	2
Ph.D.	0	2	0	2	4
Post-Doc.	0	0	0	0	0
Total	0	2	1	17	20

Notable Awards and Achievements

Publications from Prior Years