Investigating the Spatial and Quantitative Impacts of Stream Restoration on Riparian Soil Properties in the North Carolina Piedmont

by

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Abstract

One of the prime objectives of restoration is to alter the biotic and abiotic components of a system in a way so as to promote the revitalization of ecosystem functions and characteristics similar to those of undisturbed ecosystems of the same type. In stream restoration, this involves reestablishing a hydrologic regime favorable to the colonization hydrophytic vegetation and the development of hydric soils. Soil properties of riparian floodplains are largely influenced by connectivity with the stream channel, but can also be affected by the physical process of restoration itself. The objective of this study was to quantify the spatial impacts of restoration efforts on soil properties by comparing soils collected before and four years after a riparian restoration in the piedmont of North Carolina. Few studies have assessed spatial variability both before and after restoration. We used a spatially discrete sampling design which allowed for the assessment of the spatial variability of soil properties: soil organic matter content (SOM), extractable inorganic Nitrogen (NO_2-NO_3) and extractable inorganic Phosphorus (P_{ex}). The spatial patterns were modeled with semi-variance analysis and kriging. We also used statistical analysis to compare the changes in abundance of soil properties. The mean SOM significantly decreased after restoration, whereas the mean P_{ex} significantly increased. Concentrations of NO_2-NO_3 were not significantly different in the post-restoration sampling compared to pre-restoration levels. Our results indicate that restoration processes have resulted in the spatial homogenization of SOM and $P_{ex,}$ removing intrinsic soil patchiness. The loss of this spatial patterning along with soil organic matter pools represent a negative impact of restoration on important ecosystem characteristics which may have taken extensive lengths of time to develop. Further research over longer time scales will be needed to assess whether these losses represent a short-term setback in the development of the ecosystem, or a long-term alteration of the ecosystems characteristics. While disturbance from restoration processes may be unavoidable to some extent, the potential negative impact of these activities is important to understand as to avoid excessive disturbance.

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Introduction

Riparian corridors are valuable landscape features which provide a variety of ecosystem services. Flood control, nutrient retention or removal, erosion control, water quality maintenance, carbon storage, and open space are just a few of the services provided by streams and their associated wetlands. Riparian corridors contribute significantly to regional ecology by providing habitat which both preserves rare plant species and promotes regional biodiversity. The maintenance of many of these ecological services is dependant on the ecosystem functioning properly (Zedler 2003). Urbanization is one of the major causes of riparian degradation and loss of ecosystem function (Groffman et al. 2003). Societal problems arise, when urbanization degrades the ecosystem services on which the society depends, such as clean water and natural hazard protection. While substantial efforts are being made to revitalize the natural ecosystem functions of degraded riparian systems, many of these efforts are proving unsuccessful due to a lack of understanding of the complexities of riparian systems. The key to improving restoration lies in understanding the spatial and temporal development of ecosystem functions such as; what ecosystem functions can be restored, how long functional development will take, and how development will be affected by various conditions(Henry and Amoros 1995).

Urbanization can degrade riparian corridors in a variety of ways. Apart from the physical destruction which results from development in riparian corridors, urbanization can have indirect effects on streams. Severe channel incision is a common characteristic of streams in an urban setting which results from the increase in volume and energy of runoff water coming from impervious surfaces. The increased volume and energy of runoff water scours the channel bottom leading to severe downward erosion. As the elevation of the stream channel decreases, so too does the elevation of ground water. This eventually leads to the complete separation of the stream channel from its floodplain, resulting in "hydraulic drought" (Groffman et al. 2003). In the absence of a natural hydrologic regime, the wetlands of the floodplain are unable to function and provide the ecosystem services they once did.

A proper functioning floodplain can act as a sink for pollutants through the processes of biogeochemical cycling (Reddy and Patrick 1984, Groffman et al. 2002). Riparian wetlands, which are hot spots for biogeochemical processes, act as natural filters to remove nutrients preventing the degradation of adjacent stream water quality (Richardson 1994, Zedler 2003, Mitsch and Day 2006, Mitsch and Gosselink 2007). They are also sinks for nitrogen, phosphorus, and sediment, however, the degree to which they are able to immobilize such contaminants varies depending on region, hydrology, soil type, site history, and other environmental factors (Groffman et al. 2002). As the appreciation for the importance and value of naturally functioning wetlands is growing, so to is the use of wetland restoration as a tool to reduce the impact of urbanization and agriculture on rivers and promote higher levels of water quality, as in the Mississippi river basin (Mitsch and Day 2006). When stream channels become separated from their floodplains, they fail to support the wetlands which exist there and assist in the immobilization of contaminants. Therefore, maintaining and restoring functional streams is being recognized as an important factor in promoting water quality both on small and large scales. Stream restoration is also beneficial to regional biodiversity as it creates habitat for flora and fauna dependant on riparian environments, and enhances connectivity between separate populations. Yet, attempts to successfully regain these functions through ecological restorations have proved difficult due to the biotic and abiotic complexities of ecosystems (Zedler 2000). In order to improve the quality of restoration, a better understanding of ecological rates of development over time is needed (Henry and Amoros 1995). The dynamics of soil development are of particular importance because soils are indicators of wetland functions such as biogeochemical cycling, as well as hydro-period, and they provide the direct ecosystem service of carbon storage as soil organic matter. Soils influence the development of the plant community in a wetland which in turn ultimately affects the hydrology and topology of the system. A better understanding of the development of ecosystem properties and processes in restored wetlands is crucial to assessing, designing, and managing wetland restoration projects.

Soils of wetlands and riparian areas are often characterized by a high degree of spatial variability, and therefore it is important to account for this spatial variability in any assessment of wetland characteristics (Stolt et al. 2000, Ettema and Wardle 2002, Bruland and Richardson 2004). This spatial variability is at least partially due to the combination of physical and biological processes operating at different scales. Features such as differences in topography, intensity of water regime, and vegetation communities, are just a few of the things which can contribute to the spatial variability of soil properties in a riparian area. Constructed and restored wetlands have been shown to lack the spatial variability exhibited by natural wetlands, because the activity of heavy machinery involved in grading and contouring restored wetlands can compact and homogenize soils (Stolt et al. 2000, Bruland and Richardson 2005b). Spatial variability of soil characteristics is thought to be an important factor which influences the plant communities as well as ecosystem function (Ettema and Wardle 2002). Furthermore, it is considered to be a key factor in promoting species richness as well as wetland function following restoration (Tweedy et al. 2001, Bruland and Richardson 2005a). Wetlands with variable microtopography, for instance, are believed to promote a more efficient removal of nitrogen because of the presence of both aerobic and anaerobic conditions within close proximity, allowing for rapid rates of biogeochemical cycling (Reddy and Patrick 1984, Bruland et al. 2006). This theory has been supported by studies documenting nitrate and ammonia concentrations varying with changes in microtopographic setting (Bruland and Richardson 2005a). It is also logical that an area with a diverse mosaic of soil conditions will support a diverse plant community with an assortment of individual species each best fit to survive within a unique niche of soil conditions. Conversely, an area with uniform soil characteristics will likely be dominated by a smaller number of species that are the most competitive under those conditions. With this in mind, addressing spatial variability should be considered to be extremely important part of restoration projects intending to promote ecological function and species richness (Bruland and Richardson 2005a). Unfortunately, research examining the effects of restoration activities on the spatial variability of soil properties in a riparian wetland is limited.

The goal of this project was to quantify the changes in soil characteristics that have occurred within the 4 years following a stream restoration. To quantify this, we compared soil properties and spatial characteristics from the flood plain of Sandy Creek in Durham, NC, USA, measured before restoration and again four years following restoration. The floodplains from which the baseline pre-restoration soils were collected were considered to be isolated from the stream channel due to "hydraulic draught" resulting from severe stream channel incision. One of the main goals of the stream restoration was to reconnect Sandy Creek with its floodplain thereby enhancing riparian function. While there is not one single parameter we which can be used to assess whether the restoration was successful in this, an integrated analysis of the difference in soil characteristics between the years was considered to be a useful method of understanding the complex manner in which soils develop in a riparian area following restoration. Furthermore, quantifying the acute impacts to soils which have resulted from the physical processes involved in restoration is important to determining if there are unanticipated negative consequences of restoration activities.

Soil organic matter content was expected to increase following restoration due to wetter conditions on the floodplain limiting soil respiration and decomposition rates. Soil organic matter pools in both restored and natural wetlands have been shown to be highest in settings where soil saturation is most frequent or continuous (Stolt et al. 2000, Craft et al. 2002). We hypothesized that the reinstated flood regime, resulting from the restoration, would cause soil nutrient concentrations (i.e. nitrogen and phosphorus) on the floodplain to increase as sediments and nutrient enriched storm waters would regularly come in contact with the soil of the floodplain. Sedimentation has been shown to be an important input of nitrogen and phosphorus to floodplains as organic nutrients, as well as clay sized particles which increase nutrient retention through the mechanism of adsorption, are deposited on the floodplain during periods of flooding (Bowden 1987, Craft and Casey 2000). Furthermore, we expected soil properties would exhibit a stronger spatial structure following restoration due to a revitalized hydrologic regime. This is because the impact of flooding on soil properties is inherently a spatially structured process. The nature of flood effects on soils will vary depending on the relative distance of a particular soil to the stream channel as well as its relative elevation. This study provides insight into how the physical process of restoration can alter the soil properties of a restored riparian area. Specifically we examine differences pre- and post-restoration in both the properties and the spatial variability of soil resources which are necessary to support microbial and plant communities and biogeochemical functioning.

Materials and Methods

Site Description

The study area is the Duke University Stream and Wetland Assessment Management Park (SWAMP), located in Durham, NC USA. Measuring approximately 8.47 hectares in size, the study site includes the riparian areas of Sandy Creek, as well as some areas which are the fringe between the upland lowland (Fig. 1). This includes areas under forest canopy, a man-made lake, the flood plain of Sandy Creek, riparian wetlands connected to Sandy creek, and six wetland treatment cells. Sandy Creek bisects the study site centrally, flowing from a box culvert under Cameron Blvd and continuing to the southwest, where it feeds into a retention pond which was created during the restoration. Water levels of the retention pond and entire backwater area can be controlled by a weir located on the dam. The topography of the study site is variable; however the area immediately around the stream channel maintains a constant elevation. The topology around the pond and the fringes of the study site are steeply sloped in some places.

The floodplain of Sandy Creek is wide and level, and does not have a well developed levee or terrace. This is most likely due to geologically recent deposition of alluvium resulting from upland erosion, coupled with subsequent channel incision. According to soil maps (Kirby and Station 1976), the most widespread soil series present at the site is the Cartecay(coarse-loamy, mixed semiactive, non-acid, thermic Aquic Udifluvent)/Chewacla(fine-loamy, mixed, active, thermic Fluvaquentic Endoaquepts) soil series. These sandy loamy soils are formed on floodplains through alluvial sedimentation and are considered to be hydric soils (National Resources Conservation Service 2009). Soil series on adjacent uplands include Mayodan, White Store, and Pinkstone series. These soils are mostly restricted to the upland areas which are outside the influence of sedimentation from flooding, and are all non-hydric.

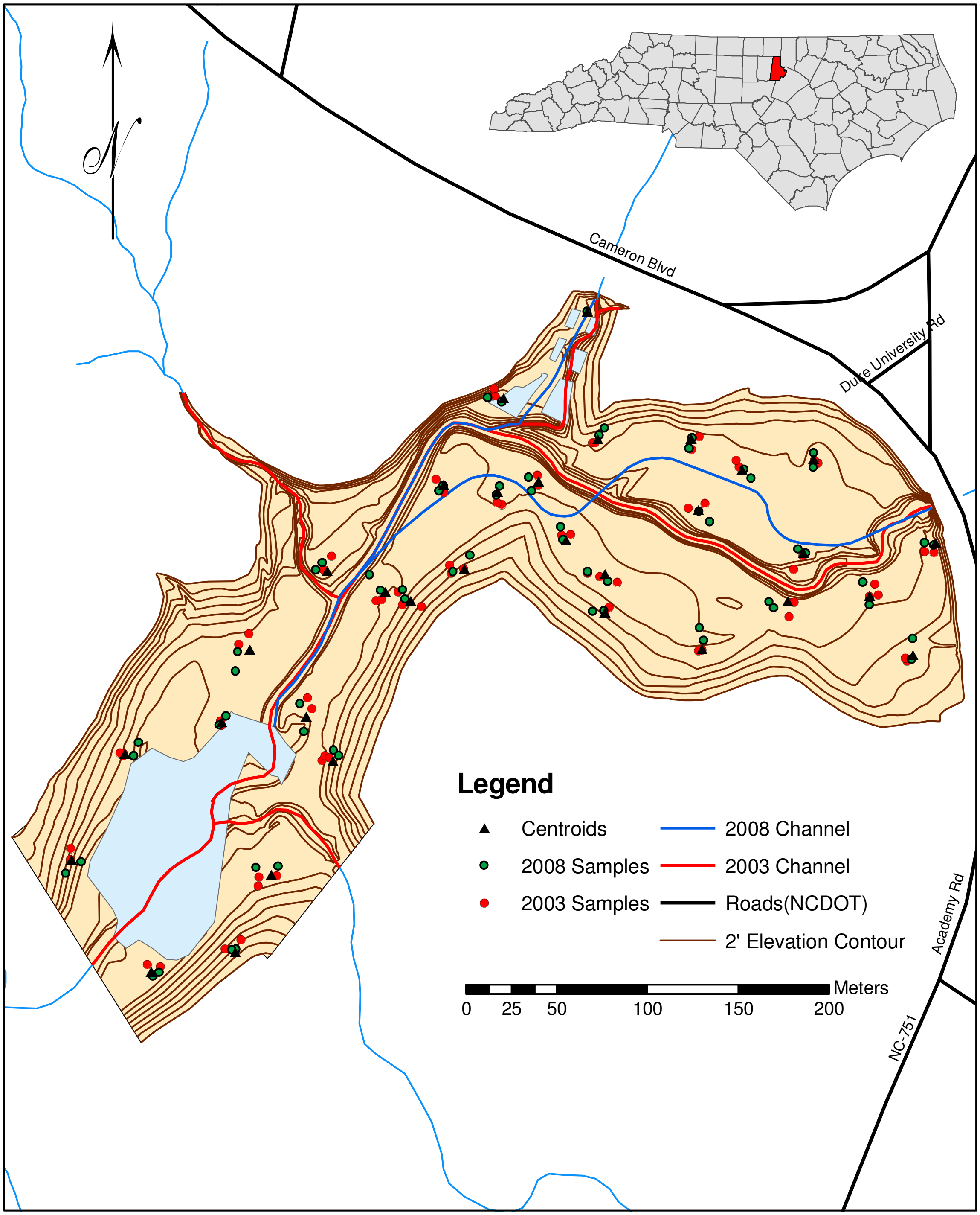

Figure 1. Sandy Creek and study site. Red channel indicates the location of the stream channel prior to restoration, blue represents channel following restoration. The pond and rectangular cells were a product of the restoration. Durham county is highlighted in red in the inset

The post-restoration floodplain of Sandy Creek regularly receives flood water during moderate to high rainfall events, however, the volume and frequency of overbank flow can vary at different locations along the stream channel (personal observation). This connection between the floodplain and stream channel was not always persistent. Prior to the restoration of Sandy Creek in 2004-2005, the stream was completely isolated from its floodplain due to a severely incised channel. This incised channel resulted from an increased volume and intensity of water associated with urban build up on and around Duke Universities Campus. It has been well documented that high percentages of impermeable surfaces alter the hydrology of a watershed, leading to high rates of stream bank erosion and the overall degradation of streams channels (Groffman et al. 2003). The portion of Sandy Creek which is contained in the study area underwent a stream restoration in 2004-2005. A main activity of the restoration, involved engineering a new stream channel of a suitable morphology to allow for overbank flooding during high volume discharge events, without compromising the stability of the stream banks itself. This new stream channel was also designed to follow a more sinuous path in order to reduce flow velocities as well as increase the contact areas of floodwaters with the floodplain. Soil excavated from the new channel was used to fill in the old stream channel. The new floodplain was graded to an appropriate elevation and topography, suitable to allow for the dispersal of flood waters. Six depressional wetland cells were engineered in the northernmost area of the study site along a tributary which receives a majority of runoff from the duke athletic fields for the purpose of reducing nutrient levels to Sandy Creek. An upper and lower bench area was created off of the floodplain at slightly different elevations for the purpose of a vegetation study. Other small wetlands were also created at various locations along the stream channel. A majority of the restoration involved the use of heavy machinery for earth moving activities. This disturbance is likely to have impacted areas of the floodplain differentially. The areas immediately adjacent to the new stream channel and around the retention pond would have been the most affected areas on the floodplain.

Sample Design and Collection Methods

In June of 2008, the 8 ha area around Sandy Creek was assessed to quantify the variability of soil characteristics following stream restoration. A spatially discreet sampling design was used to account for both large scale and fine scale patterns of variation, by randomly clustering sample points around centroids. Centroid locations were designated in 2003 for a base-line study (Sutton-Grier et al. 2003), and reused in the 2008 study. Random coordinates were generated for nineteen of the centroids using GIS. The other 14 centroids were previously established hydraulic monitoring wells. Random sample locations were established around the 31 centroids by generating a random value from 1-360 to represent bearing and a value from 1-1000 to represent distance in centimeters from the centroid location. Two samples were collected around each centroid. The coordinates for all centroid locations and all sample locations were acquired using a Trimble GPS (GeoExplorer 2005). A total of 62 samples were collected in 2008.

Soil cores were collected at each sampling location by driving a 5.5cm diameter steel sampling spoon to a depth of 10 cm below soil surface using a slide hammer. This depth was considered to be sufficient to capture developments of soil properties due to the influence of sedimentation and organic matter accrual occurring in the upper horizon (Mitsch et al. 2005). The steel spoon was lined with a plastic sleeve, which allowed for removal and storage of the soil core without disturbing it. Unincorporated duff and organic matter were brushed away from the soil surface prior to sampling so that only mineral soils were collected. After extracting a core of soil, the plastic sleeves were removed from the steel core and fitted with a rubber caps. Soil cores were kept on ice while in the field, and transferred directly into a cold room at the end of each day.

Laboratory Analysis

Soil cores were evenly divided into two halves of equal weight, by severing them longitudinally. Split plastic sleeves were weighed prior to sampling allowing the cores to be accurately separated into two evenly weighed ½ cores, with a minimal loss of soil. Half of the core (wet core) was stored at 4º C for later use in extractable phosphorus and nitrogen analysis. The soil of the other ½ core (dry core) was weighed, passed through a 2 mm mesh sieve (# 10) to remove large rocks and roots, and dried at 100° C until a constant weight was achieved.

Soil Bulk density was obtained by dividing the weight of the dried soil by the initial volume, which was corrected for the rocks and roots removed during sieving by subtracting the volume of rocks and roots from the volume of the half core. The water content of each soil sample was estimated by dividing the wet weight of the soil after it had been adjusted for the volume of roots and rock, by the dry weight of the soil.

Soil organic matter (SOM) was determined by loss on ignition (Nelson and Sommers 1996). Soils were ground with a mortar and pestle and five grams of the ground soil were weighed into pre-weighed ceramic crucibles. The crucibles were placed in a muffle furnace for 4 hours at 450° C then weighed after being transferred to a desiccator and allowed to cool. The difference in sample weights before and after the oven, divided by the dry weight of the soil, is equivalent to the percentage of organic matter in the soil.

Inorganic nutrients were extracted using similar techniques to measure extractable nitrogen (NO_2-NO_3), ammonia (NH_4), and extractable phosphorus (P_{ex}). NO_2-NO_3 as well as NH_4 was extracted from 2 g of soil with 20 milliliters of 2M KCl (Maynard and Kalra, 1993). Twenty milliliters of distilled water was used to extract inorganic phosphate from 2g of soil (Kuo 1996). The soil/extractant slurry was shaken by hand and machine for an established amount of time to allow for complete extraction of chemicals. The extractant was removed by first centrifuging, and then pipetting off the top 10 ml of extractant. A Lachat "Quick-Chem" was used to measure NO_2-NO_3 by cadmium column reduction (method 10-107-04-1-B), NH_4 by the Bertholet reaction (method 10-107-06-1-J), and P_{ex} by the Murphy-Riley (method 10-115-01-1-F). The field and analytical methods stated above were meant to replicate as closely as possible those methods used in 2003 to collect the pre-restoration data (Sutton-Grier et al. 2003).

Statistical Analysis

Tests of significance were performed on all variables, for the two years. Variables NO_2-NO_3, NH_4, and P_{ex} were normalized using log transformation, whereas a square root transformation was used for percent moisture. A Wilcoxon test (Gehan 1965) was used to determine if the difference in soil properties between years was statistically significant. This test of significance was preferred over the standard t-test, because it is a non-parametric test of significance which does not require assumptions of independence to be met. This is an important issue to address when analyzing samples taken from a clustered pattern which may be autocorrelated. In other words, samples located close to each other are likely to have more similar soil properties than samples which are far apart. This is recognized as a violation of the independence assumption, and therefore a standard parametric t-test may be inappropriate.

A statistical analysis consisting of two parts was used to compare soil properties measured in four years after restoration in 2008, to soil properties measured before restoration in 2003 (Sutton-Grier et al. 2003). The first focused on describing the difference in soil properties between years, whereas the second focused strictly on describing the spatial characteristics of soils from the two years. The first part of the analysis consisted of fitting a regression to SOM, NO_2-NO_3, and P_{ex}, in order to model the relationships between soil characteristics, and how the restoration affected these relationships. These soil properties we chosen for analysis because of their important to biogeochemical cycling as well as wetland functions. The second part of the analysis used geostatistics to examine how restoration affected spatial patterns in soil variability. To do this we used the residuals of our regression models in semivariance analysis. This allowed us to describe the spatial variability of soil properties in terms of autocorrelation, which is defined as the degree to which soil characteristics of points which are close to each other are more similar than those which are farther apart due to the influence of spatial processes. This was important for understanding the effect restoration may have had upon spatial characteristics of a riparian soil, and is indicative of the ecological impacts to the system. Lastly, we used universal kriging to interpolate soil properties across the whole study area for a visual comparison of the differences in spatial distributions between the two years. Statistical analysis were completed using the R v. 2.8.1 core package (R Development Core Team 2008), RPART package (Therneau et al. 2008) and GSTAT package (Pebesma and Wesseling 1998).

Part 1:Regression Analysis/Detrending

The goal of regression analysis was two-fold. First, using predictor variables to fit a linear model allowed us to understand which predictor variables are most important in explaining the variance of the soil variable of interest. By doing this we identified the strong statistical relationships between the soil variables, and compared the strength of the relationships between the years. The appropriateness of predictor variables was difficult to ascertain due to the complexities of soil chemistry, so we used a classification and regression tree (CART) analysis (Therneau et al. 2008) to assist in the selection of the strongest predictor variables. CART analysis is a tool which is useful in determining the importance of predictor variables in explaining the variability of a response variable. CART is a multivariate regression tool, which works by recursive partitioning to divide groups into smaller groups based on the variable which best reduces variability within the two groups, and is often used for determining the most suitable variables to include in regression model (Qian et al. 2001). CART analysis was performed for SOM, NO_2-NO_3, and P_{ex} using predictor variables SOM, NO_2-NO_3, P_{ex}, as well as bulk density (BD), distance from the stream channel (distance), and year. The variables selected through CART were subsequently used to fit a regression which modeled the relationship between soil variables. The regression analysis was used to compare the strength and type of the relationships between soil characteristics for each year. By taking the residuals of these regression models, we also detrended the data, which was important for the geostatistical portion of this analysis. The residuals could then be interpreted as representing only the variability in the data which was not explained by soil relationships, and could therefore be considered to be due to spatial variability.

Part 2: Geostatistical Analysis

The importance of spatial structure in SOM, inorganic NO_2-NO_3, and P_{ex} was evaluated by semivariance analysis using the GSTAT package in R v2.8.1 (Pebesma and Wesseling 1998). Semivariance analysis was used to describe the degree to which soil properties are autocorrelated. Semivariograms were created by plotting the average distance of paired samples separated by similar distances against the average semivariance of that group of paired samples. Semivariograms are useful for estimating autocorrelation. Empirical semivariograms are created directly from the data and then fit with a semivariogram model (Legendre and Fortin 1989). Spherical or exponential semivariograms are most commonly used in modeling semivariance. The significance of the type of semivariogram model used is discretionary and is generally inconsequential when used merely to describe spatial dependency. More care is needed in selecting an appropriate model if it will be used to make predictions (Qian 1997). For descriptive purposes, we are interested in three characteristics of the semivariogram; the nugget, the sill, and the range. These characteristics can be directly used to quantify the degree of autocorrelation the variable in question exhibits. The "nugget", the y value at distance 0, indicates the background variance of the data. This could be due to intrinsic random variability, or insufficiently capturing the fine scale spatial variability. The "sill", the y value at which the variogram flattens out, represents the total variance in the data. The "range", the x value at which the variogram flattens out, represents the distance beyond which samples are spatially independent. Range is an important characteristic of spatial patterns because it describes extent of spatial autocorrelation which can be ecologically viewed as the size of the patches of similar characteristics. The ratio of nugget to sill is generally representative of the degree of spatial dependency which is present (Cambardella et al. 1994), and is used as a means by which to compare spatial dependency between different properties and different years. Different classes of spatial dependence for the soil variables were grouped as follows: a ratio of 25% or less was considered strongly spatially dependent; a ratio between 25% and 75% was considered moderately spatially dependent; and a ratio greater than 75% was considered weakly spatially dependent (Cambardella et al. 1994).

Fitting a variogram which accurately describes the spatial characteristic at the site requires the assumption of stationarity. This assumption implies that the mean and variance of the data are the same throughout the entire study area (Legendre and Fortin 1989). It is unlikely that this is the case for our study sire, as our study site includes areas of different elevation, distance from the stream, vegetation cover. Therefore, the mean of any soil variable is likely to vary throughout different areas of the site. Large scale patterns or trends in the data will confound the semivariance analysis if not accounted for and are therefore removed by detrending prior to semivariance analysis. Commonly, spatially correlated data are detrended by fitting a polynomial of the latitude(X) and longitude(Y) coordinates to make a trend model, the residuals of which are used in modeling the semivariogram. This may not effectively remove the stationary residual if the variable of interest varies due to environmentally related factors other than spatial location. In such a case, it may be more effective to fit a regression that removes the overall trends due to these related environmental factors (Qian et al. 2001). In research of wetlands and riparian systems, environmental variables, such as distance to a water source, have been used to remove the large scale trend (Gallardo 2003).

The geographic location of a sample may have a large influence on soil properties, and therefore may explain a large amount of variability in soil properties. By fitting a regression to detrend the data, we were attempting to explain the degree of variability in the data which was due to one or many other related soil variables or covariates. Therefore, that which is modeled by the semivariogram of the residuals is only the influence of spatial patterns. For instance, consider two samples taken a relatively moderate distance apart from each other, and therefore it is uncertain whether or not they are influenced by the same spatial process. Also consider that the samples have similar organic matter and soil moisture content. In a characterization of spatial structure we are most interested in determining if the samples have similar organic matter because of their spatial relationship to each other (i.e. are they part of the same patch or pattern), or are they spatially independent. We would want to be sure when modeling the spatial relationship, that we don't mistake the influence of the potentially correlated environmental variable, moisture content, with the influence of spatial structure. To avoid this, we use the residuals of a regression between organic matter and moisture content in semivariance analysis, thereby modeling the true effect of spatial structure. This example reflects the theories and assumptions used in this geostatistical analysis.

We believe that detrending by fitting a regression of related variables is at least comparable if not better than modeling the trend by fitting a polynomial equation to the XY coordinates. The latter may yield similar results as the former if, for instance, the related variables exhibit some degree patchiness on the site. The polynomial trend may be able to account for variables which were not measured or could not be measured, and will essentially represent the overall trends or patchiness in the data. However, it may be difficult to differentiate what the polynomial equation is actually describing. Where as in regression by related variables, we know exactly what we are removing from the variance.

Kriging

Universal kriging (Krige 1966) was used to interpolate soil properties across the entire study area for the purpose of visually representing the spatial patterns. Universal kriging is an interpolation method which uses the components of a semivariogram to adjust the patterns in interpolation (Krige 1966, Legendre and Fortin 1989, Rossi et al. 1992). The steps involved in universal kriging are similar to what was done previously for semivariance analysis. Universal kriging uses a semivariogram model, derived from the residuals of a surface trend based upon x and y coordinates, to estimate the fine scale trends, and then refits the global trend surface to areas which fall outside the range of autocorrelation. Universal kriging using coordinates was preferred over ordinary kriging using the regression of soil variables because it is able to interpolate the entire study area, rather than just the areas within the range of sample point autocorrelation. Although more appropriate for visual representation, this method of universal kriging may be less useful for characterizing spatial dependency than the semivariance analysis as completed above. This is because it is difficult to determine the power of the polynomial to fit to the x and y coordinates. Too strong of a trend, may unintentionally account for spatial structure, while to weak of a trend may not sufficiently capture the global trend and satisfy the stationary assumption.

Results

Restoration appears to have resulted in significant changes to the soils of Sandy Creek (Table 1). The soil constituents which exhibited the most significant differences between pre- and post- restoration are percent soil organic matter (SOM), inorganic Phosphorus (P_{ex}), and ammonia (NH_4) (Table 1). Our results suggest that SOM significantly decreased following restoration. Conversely, P_{ex} and NH_4 concentrations showed a significant increase following restoration. Significance testing did not find the differences in bulk density, percent moisture, and NO_2-NO_3 to be statistically significant. The mean distance of sample locations to the stream decreased post-restoration likely resulting from the increased sinuosity of the channel (Table 1). Although this comparison is useful to show the general direction of the changes in soil characteristics it is only partially useful in describing the change in soil properties at the sites.

Table 1. Soil properties before and after restoration.

Parameter	2003 (n=66) Mean	SE	2008 (n=62) Mean	SE	Wilcoxon test p-value
Moisture, %	24.33	0.94	29.4	1.90	0.1320
Bulk Density, g cm^{-3}	1.037	0.02	1.044	0.03	0.3947
Soil organic matter, %	9.619	0.46	6.892	0.30	2.677e-05*
P_{ex}, ug g^{-1}	0.615	0.10	1.532	0.12	6.658 e-11*
$NO2-NO3_{ex}$ ug g^{-1}	2.472	0.21	3.617	0.73	0.578
$NH4_{ex}$ ug g^{-1}	2.367	0.18	5.574	2.48	7.805e-08*
Distance ft	103.31	6.33	67.68	7.47	4.978 e-05*

*** - a statistically significant p-value**

Model Fitting

Regression was used to model the relationships of SOM, P_{ex}, and NO_2-NO_3 with other soil properties measured during sampling. Predictor variables were selected through the use of a CART model (Appendix). The CART model for SOM determined BD and year to be the variables which explained a large degree of variability. The importance of year in this CART model suggests the restoration has had a substantial effect on SOM. The CART model also identified P_{ex} as a variable which had a strong relationship to SOM in 2003. Although not appearing important in the CART model, distance was expected to be related to SOM following restoration, because of the impact that an active flood regime can have in promoting SOM accrual. This relationship was confirmed by a plot of distance (square root transformed) vs. SOM which indicated a positive relationship (Fig. 2). Furthermore, when BD was fit into a model of

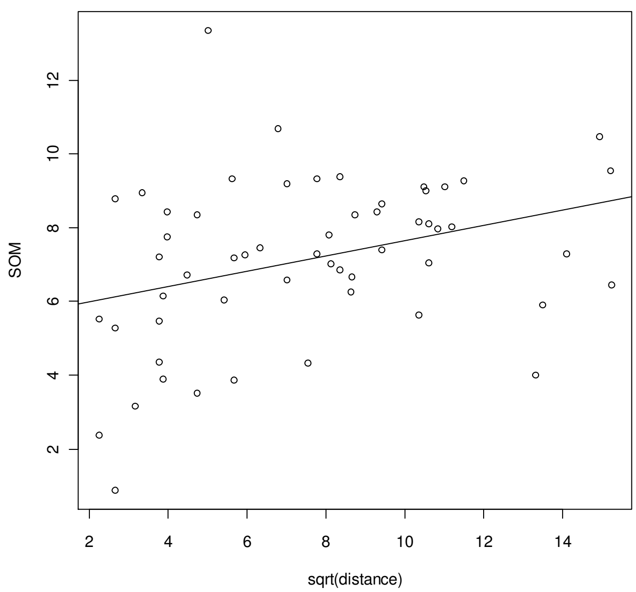

Figure 2. Plot of distance from stream channel vs % SOM after restoration. R^2 0.11.

SOM which included year as a variable, the relationship between SOM and BD was not significantly different for the years, which indicated that the relationship of BD and SOM was relatively unchanged by restoration. Conversely, the CART model identified P_{ex} (log transformed) as a variable which was strongly related to the variability of SOM in 2003 only, and exploratory analysis indicated distance to be related to SOM in 2008. For this reason, two different models were used to fit a regression for each year. The 2003 model used BD and P_{ex} variables to predict SOM, whereas the 2008 model used BD and distance as predictor variables (Table 2).

The CART model for P_{ex} indicated year as being the most important predictor of variation in sampled data, which suggested a significant impact of restoration on soil P. SOM and BD were also identified in the CART model. Distance was not particularly valuable in explaining variability, however it was included in the regression because of the logical relationship between distance to a source of flooding, and accrual of sediment bound P. A similar linear model was fit for both years with BD, SOM and distance to the stream as predictor variables (Table 2).

The CART model for NO_2-NO_3 suggested relationships between NO_2-NO_3 and BD, SOM and distance (Appendix). BD was indicated as the most important variable in explaining NO_2-NO_3 variation, followed by BD and SOM. To further investigate the significance of these relationships we used regression and tested if the slopes between years were different. It is important to note that year was not selected in the CART model, reflecting the insignificant difference in means between the two years. Although, the means were relatively unchanged, it was apparent from the regression analysis that the relationships between NO_2-NO_3 and BD, SOM, and distance were affected by the restoration. For instance, although BD was selected as the most important predictor, regression analysis indicated that it was only strongly related to NO_2-NO_3 in 2008. Regression analysis, also suggested that distance was the only variable related to NO_2-NO_3 in both years, although the slopes of these relationships differed. NO_2-NO_3, showed a negative relationship with distance which was greatly decreased in 2008. SOM was not found to exhibit a strong relationship with NO_2-NO_3, in either year. Therefore taking into account the CART and regression analysis, the linear model for NO_2-NO_3 included distance alone as a predictor variable in 2003, while distance and bulk density where used as predictors in 2008 (Table 2). This resulted in relatively low r^2 values (r^2 = 0.12 for 2003 and 0.28 for 2008), which may suggest that a large proportion of the variability was due to the spatial variability.

Table 2. Summary of linear models. Estimated coefficients are bold, with standard errors in brackets.

		2003			
y	Intercept (x_0)	x_1	x_2	x_3	r^2
SOM		BD	P_{ex}	BD: P_{ex}	0.400
	9.618 (.375)	**-12.554** (2.338)	(**1.661**) 0.392	**4.767** (2.192)	
P_{ex}		BD	SOM	BD:SOM	
	-0.892 (.112)	**1.846** (0.762)	**0.152** (0.035)	**0.201** (0.117)	0.249
NO_2-NO_3		Distance			
	2.070 (0.543)	**-0.166** (0.053)	-	-	0.129
		2008			
SOM		BD	distance	BD: distance	0.515
	6.401 (0.519)	**-13.072** (2.491)	**0.117** (0.062)	**1.002** (0.378)	
P_{ex}		BD	SOM	BD:SOM	
	0.255 (0.088)	**1.08** (0.509)	**0.093** (0.0420)	**-0.006** (0.137)	0.095
NO_2-NO_3		BD	Distance		
	0.910 (0.269)	**-2.100** (0.509)	**-0.029** (0.329)	-	0.224

P_{ex} and NO_2-NO_3 – log transformed
Distance – square root transformed

Semivariance Analysis and Kriging

The residuals from the selected models were used to create semivariograms which described the spatial autocorrelation of the samples. The semivariogram characteristics are substantially different between pre- and post- restoration years (Table 3). The class of spatial dependency was different after restoration compared to before restoration for each variable, however the direction and intensity of the difference was dissimilar for the variables. SOM and P_{ex}, for instance, showed a decrease in the degree of spatial structure between pre- and post-restoration, while the opposite is seen for NO_2-NO_3 which showed a higher degree of spatial structure after restoration (Table 3). Among all variables spatial dependency varied from no spatial structure, as exhibited by SOM in 2008, to strong spatial structure, as exhibited by P_{ex} in 2003 and NO_2-NO_3 in 2008. Our analysis suggested that the highest degree of spatial dependency was observed for NO_2-NO_3 in 2008. The range, also referred to as the lag distance, was also highly variable for the soil properties. It showed neither an overall increase nor decrease for all soil properties as a result of restoration. The shortest range observed was 23m for P_{ex} in 2003, while the largest range observed was 183.01 m for NO_2-NO_3 in 2008. It is important to note that although the range of SOM in 2008 appeared to be the largest, it is not directly comparable to the other semivariogram models because it was fit with a linear model rather than a spherical one.

Table 3. Components of Semivariogram

Semivariance							
variable	year	Model	Nugget	Sill	Nugget/Sill	Range	Spatial Dependency class
SOM	2003	Spherical	5.289	8.937	59.1%	42.91	Moderate
SOM	2008	Linear	3.298	3.304	99.8%	224.00	None
P_{ex}	2003	Spherical	0.160	0.763	20.9%	23.00	Strong
P_{ex}	2008	Spherical	0.122	0.356	52.1%	60.00	Moderate
NO_2–NO_3	2003	Spherical	0.317	0.985	32.0%	183.01	Moderate
NO_2–NO_3	2008	Spherical	0.135	0.685	19.8%	37.35	Strong

Soil Organic Matter

The semivariograms for SOM reflected the difference in attributes as described in Table 3. SOM exhibited a moderate spatial structure prior to restoration, with a range of 42.91 m. For 2003, the nugget to sill ratio was approximately 59% indicating “moderate” spatial structure. The nugget for 2003 was quite high, suggesting a high degree of intrinsic variability even at points which are spatially close together. Still, the “moderate” ranking implied that SOM was autocorrelated on a scale of 42.91m, which suggests that SOM in 2003 likely exhibited a moderate degree of “patchiness” (Fig.3a). Strikingly, the semivariogram for SOM in 2008 did not show any evidence of autocorrelation or spatial structure (Fig. 3b). The range in 2008, although much larger than that of 2003, was essentially arbitrary, as the difference in variance in data remained relatively constant throughout all distances. This implies that SOM after restoration was not spatially structured, and was distributed more or less homogeneously throughout the site without any patterning or patchiness. The insignificant difference between nugget and sill values indicated that SOM variability across the site was unaffected by the distance of separation between the measured points. The characteristics of the semivariogram implied an overall homogenization of the site. Essentially, the variability in SOM was not related to the location of a sample following restoration.

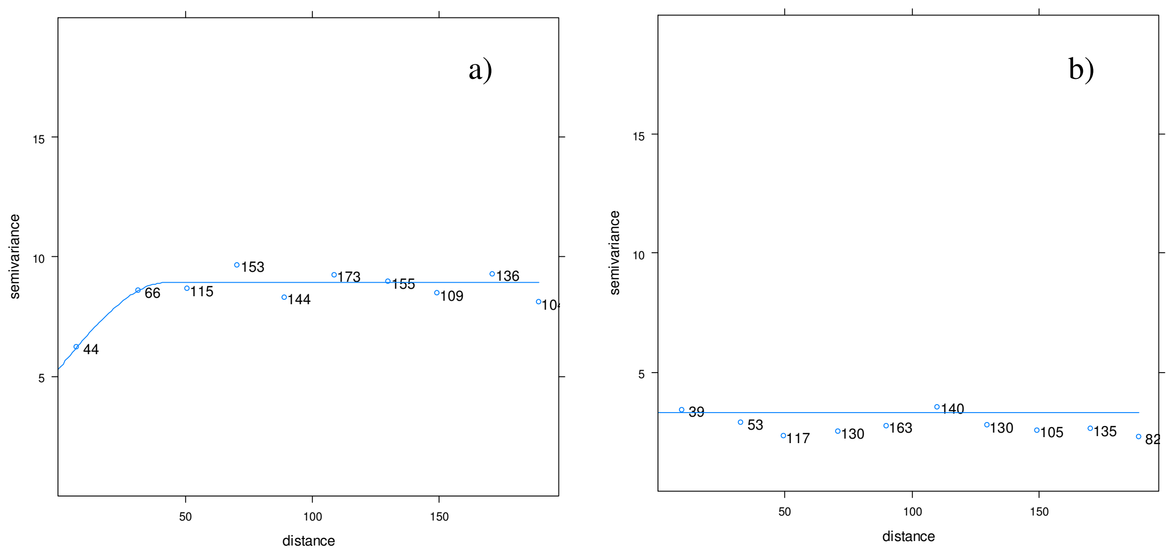

Figure 3. Semivariograms for Soil Organic Matter(SOM) a) pre-restoration 2003; b) and post-restoration 2008.

The spatial relationships are represented most clearly through the use of Universal kriging. The kriged surface of SOM in 2003 appears drastically different than that of 2008 (Fig. 4). The kriging model visually represents the large variability or SOM in 2003. It also confirms the "moderate" degree of spatial dependency, as explained by the semivariogram, by the presence of the few but significant patches of high and low SOM. The distribution of SOM in 2008 (Fig. 4b) is starkly different than that of 2003 (Fig 4a). The majority of the study area has

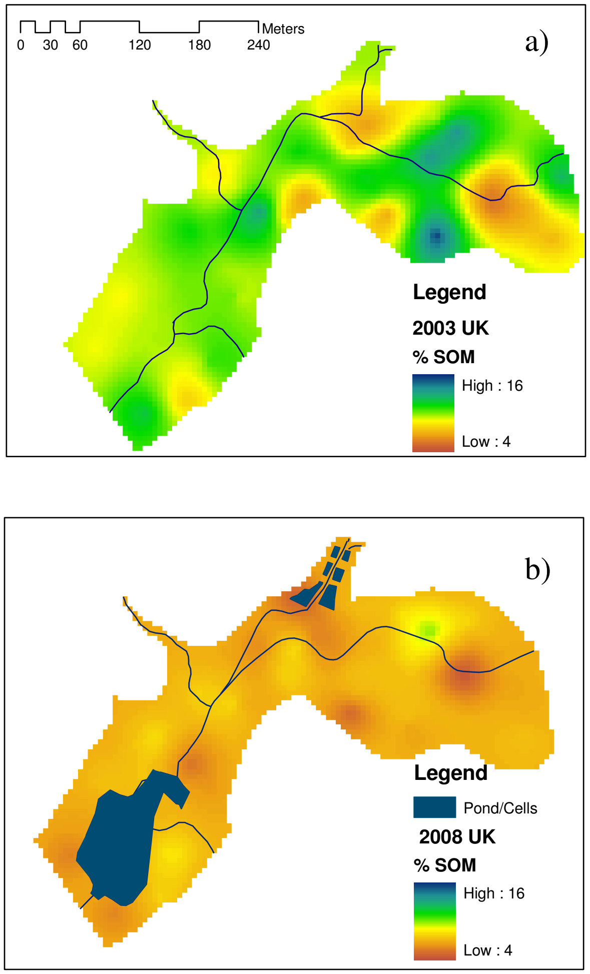

Figure 4. Kriged Map of SOM % for years a) 2003 and b) 2008.

been estimated to have lower SOM. The kriged model reflects the findings of the previously discussed analysis, suggesting that SOM distributions at the site are more or less homogenous. It is clear from this map that the restoration has had a large impact on the spatial characteristics of SOM distribution resulting in much more homogenous distribution.

Extractable Phosphorus

The response in spatial structure of P_{ex} to restoration was similar to that of SOM although not as extreme. Although exhibiting the shortest of ranges, spatial structure for P_{ex} in 2003 was considered “strong”. This was due to the low nugget value in comparison to the sill value. P_{ex} in 2008 had a similar nugget value comparatively, however the magnitude of the sill was much less, meaning overall variability had been reduced, and therefore spatial structure was only considered to be “moderate” (Fig. 5). The range value had increased substantially from 24m pre-restoration to approximately 60m post-restoration. This suggests that the distance at which autocorrelation influenced P_{ex} values had increased. The “strong” spatial dependency class as well as the low range value prior to restoration suggested that P_{ex} demonstrated small scale patchiness prior to restoration. The results of our 2008 semivariogram (Fig. 5b) suggest that much of this tight patchiness had dissipated. Although soils retained a “moderate” degree of spatial structure in terms of P_{ex}, restoration had substantially reduced the patchiness and overall variability in 2008.

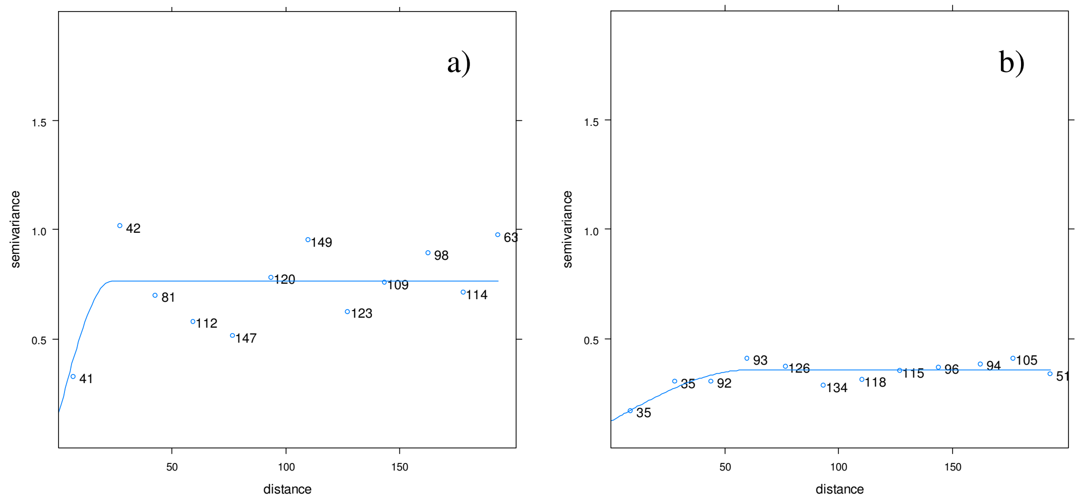

Figure 5. Semivariograms for Phosphate(P) a) pre-restoration 2003; b) and post- restoration 2008.

The results of kriging P_{ex} concentrations across the site from the two years (Fig. 7a,b), are similar to those seen for SOM. Strong patches of high and low P_{ex} concentration are frequently present throughout the study site in 2003, representing a high degree of variability, and a “strong” spatial structure (Fig. 7a). The kriged map of soil P_{ex} distribution in 2008 shows much less variability in P_{ex} concentrations, and a greatly reduced degree of patchiness (Fig. 7b). The model depicts that the areas of highest P_{ex} concentration are close to the stream channel and around the mid-portion of the site in 2003. There does not appear to be any ascertainable trend of P_{ex} distribution in 2008. These models suggest that restoration has had a similar spatially homogenizing affect on soil P_{ex}, as was seen for SOM at the study site. From the figure it appears, that the areas of lowest P concentration in the mid portion of the stream.

Extractable Nitrogen

Contrary to what was observed for SOM and P_{ex}, NO_2-NO_3 exhibited an increase in the degree of spatial dependency following restoration. The class of spatial dependency increased from “moderate” in 2003 to “strong” in 2008 (Fig. 6). This can be attributed to the nugget value estimated in 2008 being lower than that of 2003, as the difference between nugget and sill is relatively similar between the two years. The most obvious difference between the NO_2-NO_3 semivariograms is the range, which was much larger for 2003. This suggests that NO_2-NO_3 samples were autocorrelated at distances less than 183.01 m in 2008. Comparatively, NO_2-NO_3 semivariograms suggest that autocorrelation only influenced samples within 37m. This indicates that a clear change in NO_2-NO_3 patterning resulted due to restoration. Prior to restoration NO_2-NO_3 exhibited moderate large scale patchiness whereas patches of NO_2-NO_3 following restoration were much smaller but less variable.

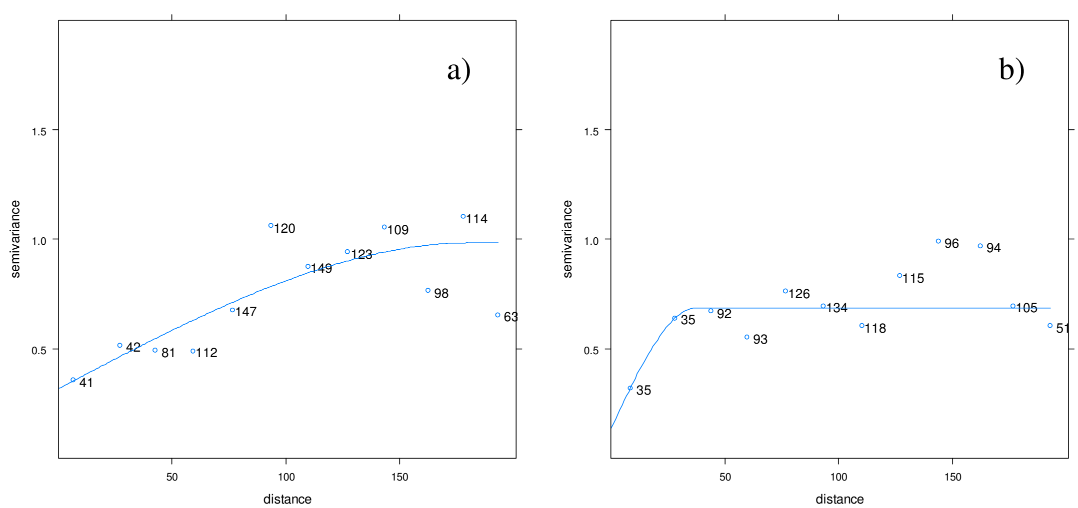

Figure 6. Semivariograms for extractable nitrogen (NO2-NO3) a) pre-restoration 2003; b) and post- restoration 2008.

The kriged models of NO_2-NO_3 (Fig. 7 c,d) exhibit an increase in the degree of spatial dependency following restoration. Although, NO_2-NO_3 has a relatively similar variability for both years, the post-restoration distribution (Fig. 7d) appears to be somewhat patchier than that of pre-restoration (Fig. 7c). This reflects the stronger spatial dependence of NO_2-NO_3 in 2008 which was described in the semivariance analysis. NO_2-NO_3 concentration is generally lower for most areas of the sample site prior to restoration, with the exception of a few small patches of high NO_2-NO_3 concentration. Conversely, there are many patches of both high and low NO_2-NO_3 concentration scattered throughout the post restoration sample site. It would appear that some areas of the highest NO_2-NO_3 concentration are adjacent to the stream channel. Although, this is not consistent for all areas adjacent to the stream channel, it may reflect the influence of active biogeochemical cycling due to hydrologic connectivity in those areas where NO_2-NO_3 is highest and adjacent to the stream channel.

Interestingly, the kriged maps of pre-restoration P_{ex} (Fig. 7a) and pre-restoration NO_2-NO_3 (Fig. 7c) are similar in terms of the locations of patches with high concentrations. This may reflect the presence of biogeochemical hotspots prior to restoration.

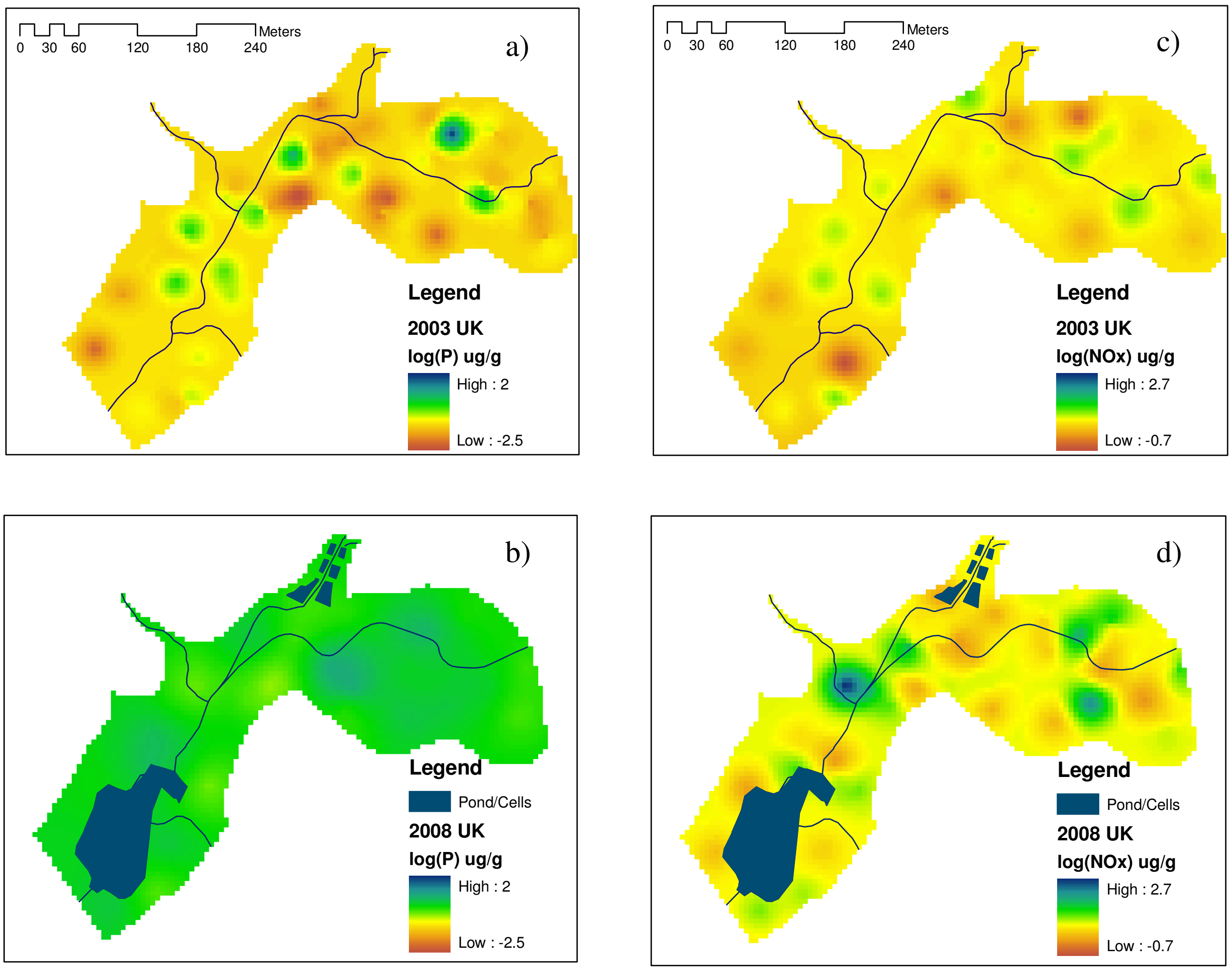

Figure 7. Kriging models of the study site showing a) P_{ex} in 2003, b)P_{ex} in 2008, c)NO_2-NO_3 in 2003 and d) NO_2-NO_3 in 2008.

Discussion

Collectively, our study suggests that the process of restoration has resulted in significant alterations to soils, and particularly the spatial distribution of soil resources, of the riparian area of Sandy Creek. Restoration affected the spatial patterns of soil properties in ways which differed in degree and direction. Spatial dependency for SOM, which was “moderate” prior to restoration, was absent in 2008. Similar affects were noted for P_{ex}, which transitioned from exhibiting a “strong” spatial dependency before restoration, to a “moderate” spatial dependency following restoration. However, it is important to note that the semivariogram fit for P_{ex} in 2003 was less than ideal, as the points were not tightly clustered around the spherical model, and the location of the nugget was determined solely by the first point. It does not demonstrate a gradual increase with distance which is ideal when fitting a spherical model to an empirical variogram. Although, we are confident that we fitted the most accurate model to the data, the value of the nugget, and therefore degree spatial dependency may have been overestimated. The difference between years remains clear though, SOM and P exhibit much less patchy distributions in 2008. Conversely, the spatial dependency of NO_2-NO_3 increased from “moderate” to “strong” following restoration. Therefore, NO_2-NO_3 exhibits a slightly stronger, though less extensive, patchiness after restoration.

We believe that these results primarily reflect the impacts of restoration on soil properties through physical disturbance of restoration including the movement of earth, the clearing of vegetation, and the compaction of soil through the use of heavy machinery. These disturbances led to significant changes in average resources levels in the soil (such as the decreases in SOM and increase in P_{ex}) and to changes in spatial variability (such as the decreases in the patchiness of SOM and P_{ex}). Our findings stress the importance of the rate of ecosystem development following restoration. Ecosystem properties, particularly soils, may be slow to develop post-restoration (Craft et al. 2002, Craft et al. 2003). It is our opinion that the four to five years which have elapsed since the restoration may not have been long enough for the restored ecosystem to recover from the disturbance due to restoration activities and to begin to develop natural levels of variability and spatial structure.

Our findings for SOM are contrary to what we might have expected on the outset of this study. It was our expectation that the reestablished flood regime would result in wetter soil conditions which would be conducive to decreased decomposition and hence the build up of SOM in soils. Furthermore, as the flood waters would likely influence areas on the floodplain differently as a result of variable floodplain microtopography and channel morphology, we expected that SOM might accrue faster in some areas than in others leading to a strong spatial patterning. Rather than exhibiting an increase of SOM, the mean and total variability of the study area has significantly decreased following restoration. This reduction in variance suggested by the regression analysis hints at an overall homogenization, and is confirmed by the semivariance analysis. The “moderate” degree of spatial dependency exhibited by SOM prior to restoration is completely absent following the restoration. This suggests that the distribution of SOM does not occur as patches of high and low concentration; rather SOM is much less variable around the site. The figure of the kriged surface (Fig. 6b) for 2008 reflects the overall lack of variability present at the site following restoration. We believe the reduction in average SOM and SOM spatial structure following restoration is a consequence of physical disturbance which accompanied the restoration process. The physical process of grading and contouring a new stream channel and floodplain required a high degree of earthmoving by heavy machinery which likely resulted in a mixing and homogenization of soils of the riparian area. This finding is consistent with other research which found restoration processes to result in soil homogenization (Bruland and Richardson 2005b). During these processes, SOM rich upper horizons may have been turned under or mixed with the lower mineral soils thereby reducing the average SOM concentrations on the surface. Soil mixing may also have increased decomposition rates in the upper soil profile as a result of it being exposed to aerobic conditions. Furthermore prior to restoration, the site existed under a forested canopy which contributed a large degree of SOM to the soil in the form of leaf detrital matter. Much of this tree canopy around the stream channel has been replaced with an emergent macrophyte community and young tree community which is still in development. The amount and nature of detrital matter being contributed to the soil is therefore substantially different post-restoration compared to pre-restoration. The climate in recent years has been exceptionally dry resulting in a drought in 2008. SOM accrual may have been substantially slowed if not reversed due to these dry conditions in recent years as a result of SOM oxidation. In light of these conditions we interpret the results of this study to suggest that restoration processes result in the net loss of SOM in the short term. The accrual of SOM moves at extremely slow rates, and it is likely that 4-5 years is not a sufficient time in which to buildup equivalent levels of SOM as were lost during restoration. This suggests spatial processes which act upon a riparian system may take longer than four to five years to be observed. Other research has estimated the amount of time needed to accumulate pools of soil organic matter equivalent to that of natural wetlands to be between 30 and 150 years, even in very productive and regularly inundated systems such as salt marshes (Craft et al. 2002, 2003). Estimating the rate at which these patterns develop will require further research over longer time scales.

Inorganic nutrients NO_2-NO_3 and P_{ex} responded differently to restoration activities. Firstly, comparison of the means suggested that P_{ex} was significantly increased site-wide following restoration, whereas the average NO_2-NO_3 concentration was relatively unchanged. The results of semivariance analysis also identified dissimilar responses to restoration. NO_2-NO_3 exhibited a “moderate” degree of spatial dependency pre-restoration, which increased substantially following restoration. On the other hand, P_{ex} transitioned from a “strong” spatial dependency pre-restoration to a “moderate” spatial dependency decreasing in response to restoration. The reason why these nutrients were affected so differently by restoration is not immediately clear. It is likely that the physical disturbance (i.e. soil moving) which took place during restoration had a large effect on the spatial distribution and concentrations of inorganic nutrients similar to that which was seen for SOM. The explanation for the elevated average P_{ex} concentration in 2008 is difficult to ascertain and may be due to multiple mechanisms. The most likely explanation for the elevated levels of P_{ex} is through the mechanism of mineralization. The breakdown of soil organic matter releases organic phosphorus as inorganic phosphorus, and is likely a reason for elevated P_{ex} after restoration. This explanation is supported by our observation of an overall loss of SOM site wide, as well as the strong relationship of P_{ex} and SOM in 2003. It is also plausible that soil homogenization has caused areas of higher P_{ex} concentrations to be mixed with areas of lower P_{ex} concentration resulting in an overall elevated level of P_{ex} when considered site-wide. The elevated and homogenous distribution of P_{ex} is reflected by the kriged surface map of 2008 (Fig 7 b) which indicates that, although less variable, the P_{ex} concentrations are elevated throughout the site compared to pre-restoration P_{ex} distribution (Fig 7 a). Conversely, the distribution of NO_2-NO_3 post-restoration appears to have increased and is much patchier in comparison with the distribution of NO_2-NO_3 prior to restoration, even though the average level of NO_2-NO_3 has not significantly changed. This may reflect the transient characteristics of NO_2-NO_3 in an ecosystem. Characteristically, NO_2-NO_3 is far more mobile than inorganic phosphorus in an ecosystem as it is connected to both biological and geochemical processes (Craft 1996). NO_2-NO_3 is more readily dissolved in solution and leached from a system than inorganic P and is also subject to a much faster biogeochemical turnover rate.

Therefore, the reason why we see a difference in the responses of P_{ex} and NO_2-NO_3, may be explained by the rapid rate at which excess N is taken up by plants, cycled through biogeochemical processes, or removed from the site via runoff or leaching. This may explain why the distribution of NO_2-NO_3 doesn't exhibit the same effects of homogenization as observed in SOM and P. While it may have been homogenized immediately following the restoration, the different rates at which it has been cycled through biogeochemical and plant processes the site have resulted in the observed patchy distribution.

Similar results were found in a chronosequence study in land used for agriculture (Boerner et al. 1998) where SOM, inorganic P and inorganic N all demonstrated unique spatial and temporal patterns in their development following disturbance. Similar to our analysis, Boerner et al.(1998) found that inorganic N was more spatially structured immediately following disturbance, than compared to mid-successional or mature systems. However, the spatial patterning in their study was due to the patterns of plowing and planting as the disturbance was agriculture.

We found soil properties of SOM, BD, and P to operate on spatial scales ranging from 23 to 183m. Other studies have found soil properties (SOM, inorganic N, inorganic P), to range from less than 5m (Boerner et al. 1998) or 8 to 10 m for floodplain forests (Gallardo 2003), and up to 104 to 201m (Cambardella et al. 1994) in agricultural fields. The variation in the range of spatial autocorrelation can be quite different for soil characteristics across different sites (Cohen et al. 2008). This is most likely a result of the unique spatial processes which operate at the site, which may vary substantially based on climate, elevation, slope, vegetation, or a variety of other environmental factors. Therefore it is difficult, if not impossible to make a general assumption about what the range of spatial dependency will be in a certain system. Furthermore, we have demonstrated that disturbance, even in the form of restoration processes, can greatly alter the range and degree of spatial influence for a given soil variable at the same site.

While the majority of our discussion has attributed the changes in soil properties to the disturbance of restoration, there may be some evidence of ecosystem function present in our data. The kriged surface maps identify that some of the areas of highest NO_2-NO_3 occur at locations close to the stream channel. These may represent areas which are hydraulically connected with the stream and thus are hotspots biogeochemical processes and nutrient transformations. It will be important to repeat similar studies of this area in the future which address the spatial distribution of soil properties and thus will allow for a further assessment on the development of ecosystem processes following restoration.

Conclusion

It is clear from this study that restoration has significantly affected both the spatial distribution of soil properties as well as the levels of soil properties. It is likely that these changes will influence the trajectory of ecosystem development following restoration. Other research has highlighted the importance of generating microhabitats and increasing overall variability as a means to promote diversity of biota as well as function (Bruland and Richardson 2005a). Based on the results of this study, modern restoration practices may be negatively impacting the intrinsic variability already present at the site, by homogenizing the soil, resulting in a temporal lag in realized functional benefits of restoration. A reduction in the variability of soils is likely to impact biogeochemical cycles particularly, as areas which are subject to both oxic and anoxic conditions tend to be hotspots for biogeochemical cycling (Baldwin and Mitchell 2000). But this reduction in variability may have additional consequences for ecosystem development. For instance, changes in the spatial distribution of soil resources may lead to a reduction in diversity of flora which colonize the site as those with competitive abilities best fit to exploit those given conditions will be able to proliferate across the entire site. These conditions may be competitively favorable to the takeover of invasive species (Zedler and Kercher 2004). The overall homogenization of SOM and P distributions leading to a decrease in the patchiness of these soil characteristics has likely reduced the variability of soil properties, and therefore the diversity in microhabitats which may benefit biodiversity. Our results suggest that restoration practitioners should consider the impacts of soil disturbance during restoration activities. While the goal of restoration is to revitalize the functional attributes of the system, which will in turn generate ecosystem services, restoration may be unknowingly removing variability which has been generated over long expanses of time and is important to ecosystem function itself. We suggest that restoration practitioners consider adopting practices that could mitigate the effects of restoration activities on soil properties. For instance, amending soil with carbon following restoration may help account for the any losses of SOM, while helping to jumpstart spatial patterns (Sutton-Grier A.E. *In Press*). Creating variable microtopography following restoration may be important to the development of spatial patchiness(Bruland and Richardson 2005a). The use of machinery which avoids extensive soil disturbance may reduce the impact to intrinsic soil characteristics. Understanding the full effects, both positive and negative, of restoration is an important step to increasing its effectiveness and the overall success of restoration activities.

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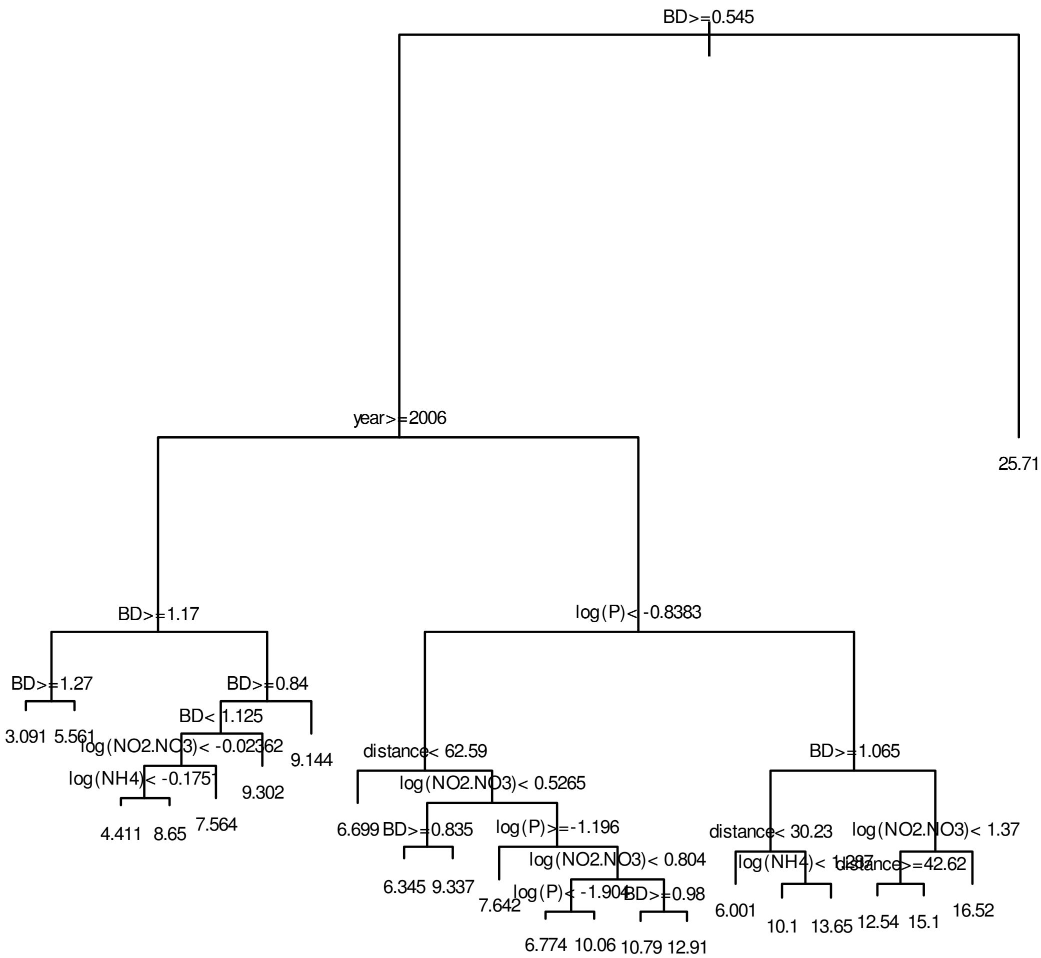
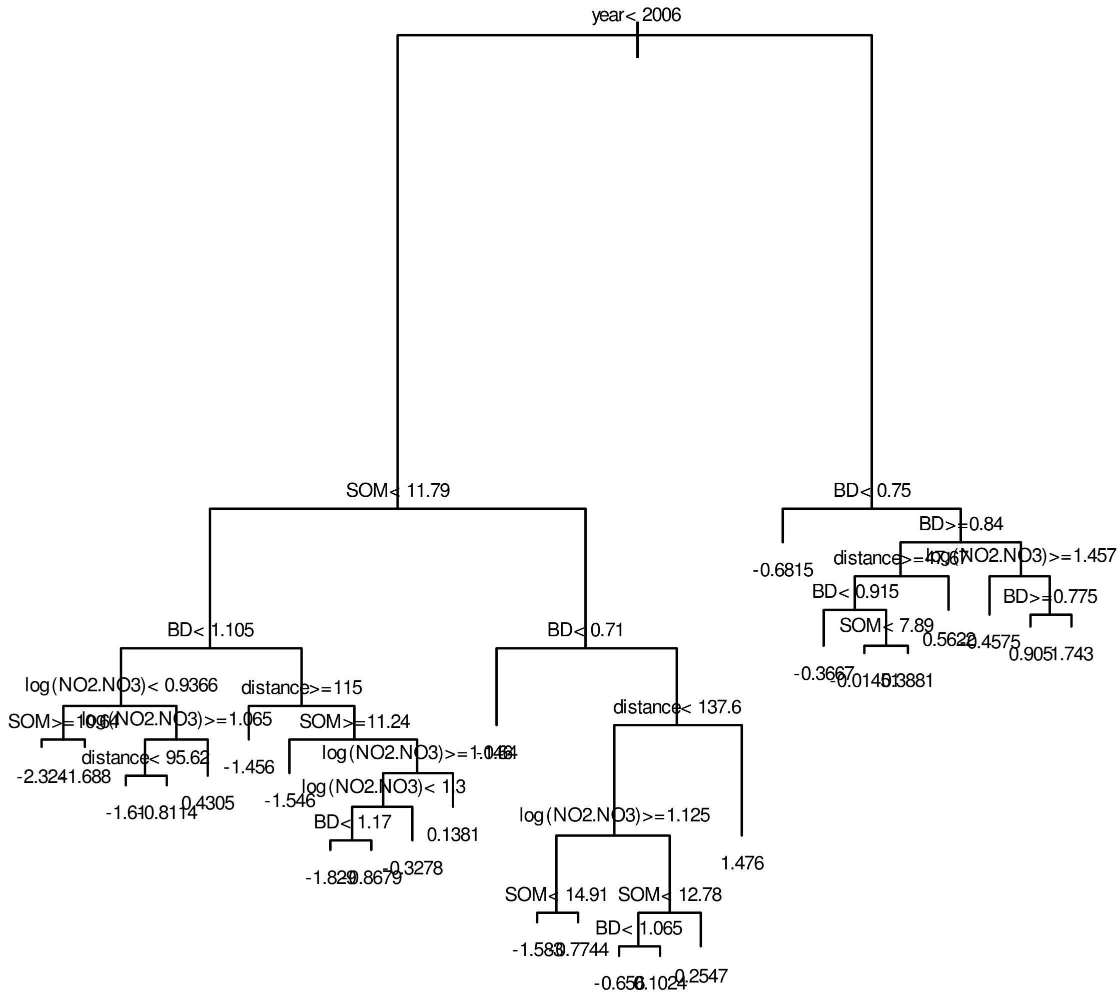
Appendix

Cart Models

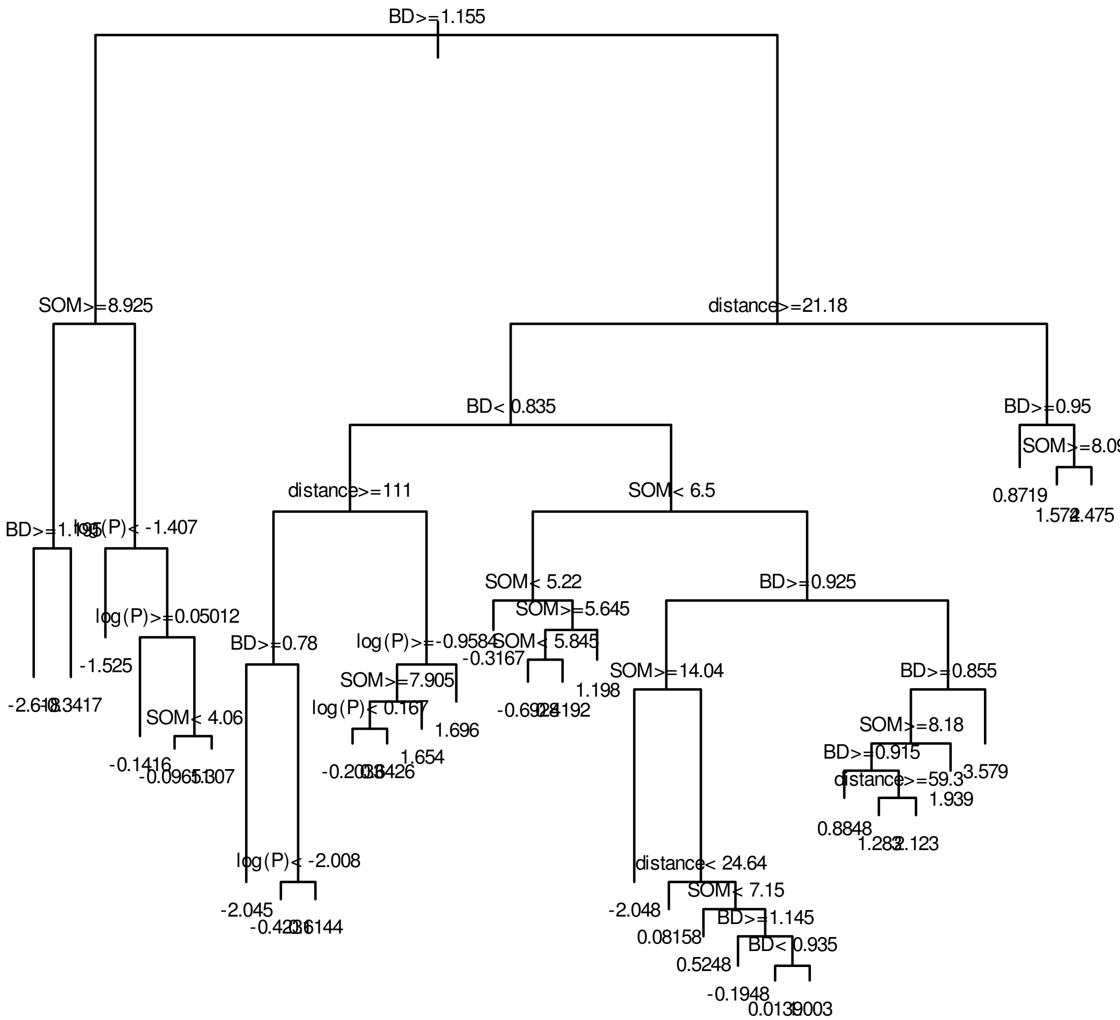
SOM



P

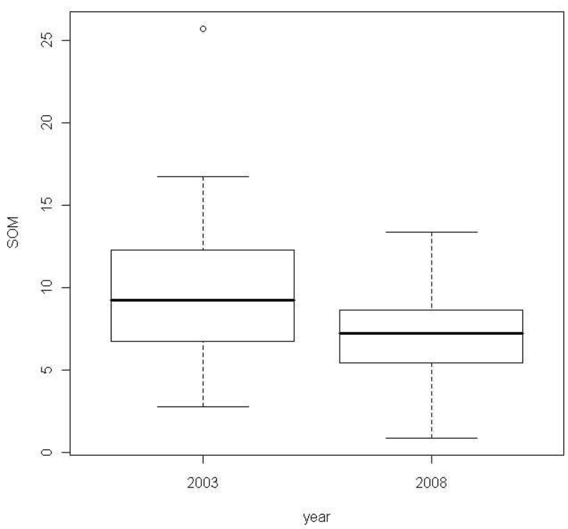

year< 2006
SOM< 11.79
BD< 0.75
BD>=0.84
-0.6815
distance>=47.67
log(NO2.NO3)>=1.457
BD< 0.915
BD>=0.775
SOM< 7.89
0.5622
0.4575
0.905
1.743
-0.3667
-0.01451
0.3881
BD< 1.105
BD< 0.71
log(NO2.NO3)< 0.9366
distance>=115
SOM>=10.64
log(NO2.NO3)>=1.065
SOM>=11.24
distance< 137.6
distance< 95.62
-1.456
log(NO2.NO3)>=1.046
-1.64
-2.324
1.688
log(NO2.NO3)< 1.3
-1.546
-1.61
0.8114
0.4305
BD< 1.17
0.1381
log(NO2.NO3)>=1.125
1.476
-1.829
0.8679
-0.3278
SOM< 14.91
SOM< 12.78
BD< 1.065
-1.583
0.7744
-0.656
0.1024
0.2547

NOX

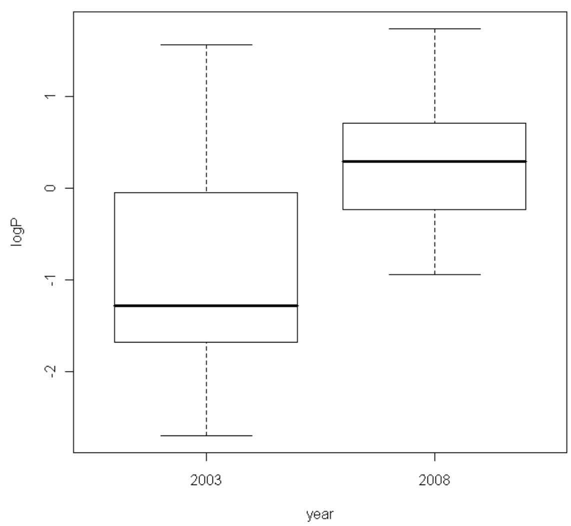

BD>=1.155
SOM>=8.925
distance>=21.18
BD< 0.835
BD>=0.95
SOM>=8.09
0.8719
1.574
2.475
distance>=111
SOM< 6.5
BD>=1.195
log(P)< -1.407
SOM< 5.22
BD>=0.925
SOM>=5.645
log(P)>=0.05012
-1.525
BD>=0.78
log(P)>=-0.9584
SOM< 5.845
-0.3167
SOM>=14.04
BD>=0.855
1.198
SOM>=7.905
-2.618
0.3417
log(P)< 0.167
-0.6924
0.8192
SOM< 4.06
1.696
SOM>=8.18
BD>=0.915
1.654
-0.1416
-0.0965
1.307
-0.2038
0.8426
distance>=59.3
3.579
1.939
0.8848
1.283
2.123
log(P)< -2.008
distance< 24.64
SOM< 7.15
-2.045
-2.048
BD>=1.145
-0.4201
0.6144
0.08158
BD< 0.935
0.5248
-0.1948
0.0139
1.003

Boxplots

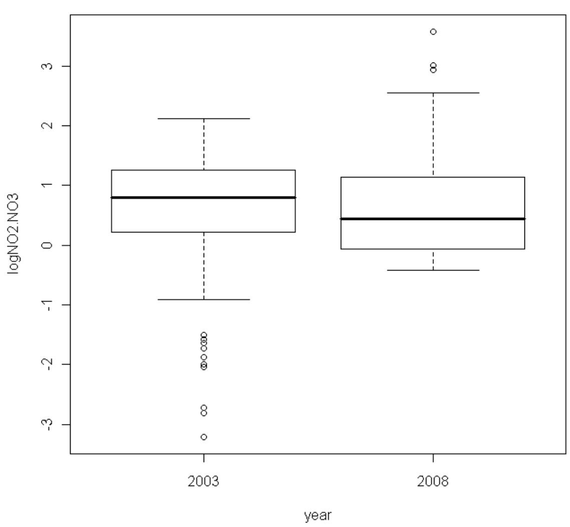
Soil Organic Matter



Inorganic Phosphorus



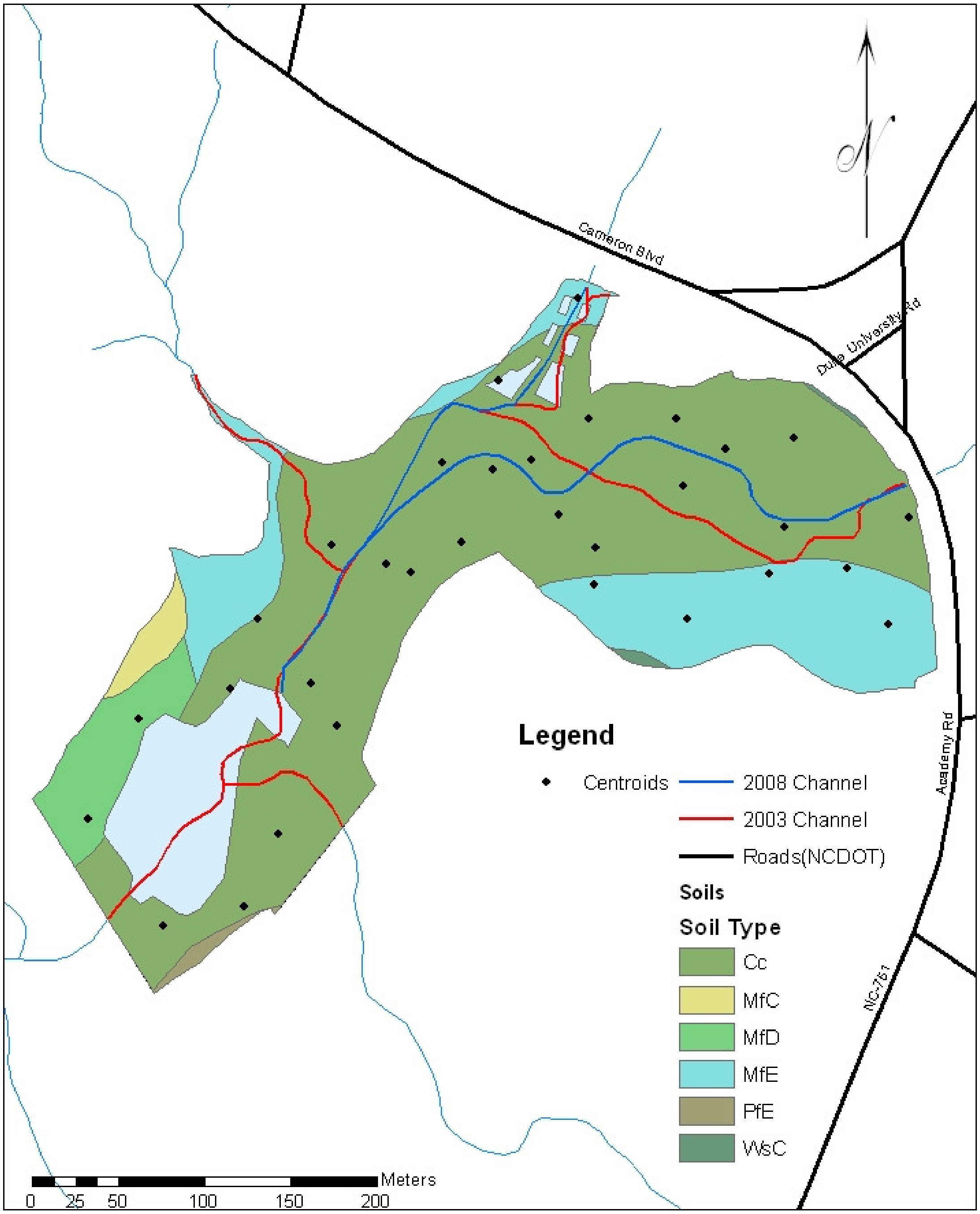
NO_2-NO_3



Maps

Aerial Photo

Soils Map


Cameron Blvd
Duke University Rd
Academy Rd
NC-751
Legend
Centroids
2008 Channel
2003 Channel
Roads(NCDOT)
Soils
Soil Type
Cc
MfC
MfD
MfE
PfE
WsC
Meters
0 25 50 100 150 200

Site Photos

Pre- Restoration

Post-Restoration