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In this study I propose a statistical model, the Riparian Nutrient Attenuation Model (RNAM), designed to quickly and accurately access the nutrient attenuation capability of riparian buffer zones. This information may help land managers evaluate riparian systems in terms of their capacity to retain nutrients.

Developed using data available in the literature, RNAM uses three physical properties of the riparian, including vegetation type, slope, and width, to estimate the retention of total N, NO_3^- and P. Three RNAM sub-models, RNAM-nitrogen, RNAM-nitrate and RNAM-phosphorus, were developed to handle each of the three nutrients. In developing RNAM, the relationships between the predictor variables and nutrient retention were examined.

A preliminary test of RNAM indicated that each of the sub-models is capable of producing reasonably accurate estimations of percent nutrient reduction. RNAM-nitrogen, however, produced inconsistent estimates of nitrogen reduction at higher levels. More data is needed to calibrate and validate RNAM.

MODELING NUTRIENT ATTENUATION BY

RIPARIAN BUFFER ZONES ALONG

HEADWATER STREAMS

by

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To Frank and Kathy Bereitschaft

APPROVAL PAGE

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CHAPTER I

INTRODUCTION

Riparian zones are transitional areas that lie between upland terrestrial and aquatic environments. Healthy riparian zones are often composed of unique and diverse ecosystems (Forman 1997; Xiong and Nilsson 1997; Sparovek et al. 2002). Due to their spatial and topographical position between the stream and upland matrix, riparian zones are capable of performing a variety of functions critical to the health and stability of the stream ecosystem. Riparian vegetation, for instance, helps to maintain stream bank stability, moderate stream water temperature, and provide habitats for terrestrial and aquatic organisms (Karr and Schlosser 1977; Sweeney 1992). In addition, a variety of riparian biotic and abiotic components interact with and retain sediment from upland erosion, and buffer excessive nutrient loads from upland surface runoff and groundwater flow (Hill 1996; Casey et al. 2001).

In the past, riparian research related to nutrient buffering has primarily focused on agricultural areas, which receive excessive loads of nutrients through agricultural activities, particularly, the overuse fertilizers. With the recognition that urban watersheds deliver significant volumes of NO_3^-, $NH4^+$, PO_4^{-3} and other nutrient pollutants to stream and coastal waters, riparian buffers are receiving more attention (Miller et al. 1997). Groffman and Crawford (2003) found that some urban riparian zones, after having undergone significant alteration (with a variety of litter, soil disturbance, and exotic plant species), were still capable of acting as effective biological sinks with denitrification potential (DNP) equal to that of much less disturbed rural riparian zones. Likewise, a recent study conducted by Bishop and Mou (2004) indicated that the top layer of soil within the urban riparian study sites (both forested and grassy) were able to filter out 99%, 95% and >85% of applied NH_4^+, PO_4^{-3} and NO_3^- carried by runoff, respectively. These and other similar studies (Peterjohn and Correll 1984; Hill 1996; Casey et al. 2001; Groffman et al. 2002) have clearly indicated that urban riparian zones are capable of effectively buffering pollutants.

Urban riparian zones, however, are uniquely vulnerable to a wide variety of anthropogenic alterations and disturbances that can influence the morphology and biological composition of the riparian, and ultimately decrease its ability to perform critical buffering functions. Perhaps the most influential – and obvious – anthropogenic alterations in urban areas are impervious surfaces, such as buildings, roads, parking lots, and other paved surfaces (Freeman and Schorr 2004). Impervious surfaces, along with unusually compact soils in urban areas, decrease the direct infiltration of water into the soil, thereby increasing stormwater runoff. Stormwater is typically channeled directly to the stream through pipes or drainage basins, often entirely bypassing riparian areas (Dunne and Leopold 1978; Rose and Peters 2001; Groffman and Crawford 2003).

These hydrological alterations, which focus the movement of large quantities of stormwater runoff quickly and frequently to the stream, have important consequences for both the stream and riparian environment (Paul and Meyer 2001; Groffman and Crawford 2003). The increase in stormwater volume and water flow speed in the stream often causes incision or "downcutting" of the stream channel (Henshaw and Booth 2000). Downcutting, produced by fast moving water scouring out sediment left behind by construction projects and (perhaps earlier) agricultural activity within the watershed, results in a widening and deepening of the stream channel and an increase in suspended sediment concentration (Booth and Jackson 1997; Henshaw and Booth 2000; Paul and Meyer 2001). As the stream water becomes more turbid, less light is able to penetrate the water column, reducing the photosynthetic capacity of benthic plants and the ability of fish and other aquatic organisms to visually forage for prey. Entire aquatic food chains can be disrupted in this manner (Lenat and Crawford 1994; Wood and Armitage 1997; Paul and Meyer 2001).

The combination of stream incision and reduced infiltration in urban watersheds can significantly reduce riparian water table levels. Groffman et al. (2002) observed that the water table depth at two suburban and one urban riparian site in the Baltimore, MD area (over a 2 year period) was significantly lower than at a forested reference site. Lowering urban riparian water table depth can alter the structure and function of these ecosystems. A decrease in soil water content, for instance, can significantly affect the composition of riparian plant communities, as well as the fauna that depend on them, that prefer wet, hydric soils. In addition, a change from hydric to mesic soils can have a dramatic impact on the microbial processes critical to the consumption of NO_3^- within the riparian. Groffman et al. (2002) found that the anaerobic process of denitrification was lower in urban riparian soils with relatively deep water tables than in forested riparian soils with shallow water tables. Finally, riparian zones directly bypassed by drainage systems cannot filter non-point source pollutants, such as NO_3^- and PO_4^{-3}, which are transported via surface water runoff. Although this may not harm the riparian, it can have a significant impact on stream water quality.

In order to help mitigate the consequences of altered urban watershed hydrology, and improve the general capability of riparian zones to act as non-point source pollution buffers, thereby protecting stream water quality, it has been proposed that surface water runoff from urban uplands be channeled to a source of broad application at the upper boundary of "high-quality" riparian corridors (Casey et al. 2001; Groffman and Crawford 2003). A simple form of this proposed broad application would consist of a storm drain pipe or basin that empties into a smaller pore-lined pipe lying perpendicular to the stream for some lateral distance. The application pipe, located upslope from the stream, would allow surface runoff to slowly drain down the slope and through the riparian zone. As the flow moves toward the stream, water and suspended pollutants can be physically slowed through their interaction with riparian vegetation, soil, and organic litter. This allows time for water and nutrients to infiltrate the soil and be taken up by vegetation or retained in the soil profile.

In addition to preventing excessive nutrient loads from reaching the stream, this course of action should also help reduce the frequency and severity of flash floods in lower order streams. Allowing surface runoff to drain slowly through riparian zones, instead of channeled directly into the stream, will decrease the amount of water being deposited into the stream following heavy precipitation. Lessening the frequency and severity of floods will decrease the current rate of stream bank incision. Consequently, urban stream banks will become more stable, further helping to decrease soil erosion, sediment deposition, and the eutrophication of downstream water bodies.

A variety of models currently simulate hydrology and nutrient cycling within and around riparian and aquatic ecosystems. Several of these models, including the Riparian Ecosystem Management Model (REMM), the Chemicals, Runoff, and Erosion from Agricultural Managements Systems (CREAMS) model, the Environmental Management Support System (EMSS), and the Soil and Water Assessment Tool (SWAT) were developed primarily to evaluate the potential impact of different land management practices on local water quality (Kinsel 1980; Lowrance et al. 1998; Vertessy et al. 2001; Neitsch et al. 2002). None of these complex process models, however, were designed specifically to estimate the nutrient attenuation capacity of riparian buffer zones. Therefore, the development of a new, less complex, statistical model is necessary to evaluate riparian zones in terms of their buffering capability, and quickly identify which buffers are most capable of accepting the broad application of storm water proposed in this study.

CHAPTER II

LITERATURE REVIEW

Riparian Functions

The critical functions performed by intact riparian zones can be divided into three major categories: protection of water quality, regulation of hydrology and geomorphology of the stream, and maintenance of natural biological structure and fitness. The health of the entire stream ecosystem depends on these interconnected functions.

Riparian zones are capable of protecting stream water quality by regulating three critical components: water temperature, nutrient and sediment load. Water temperature in lower-order streams is principally controlled by riparian vegetation along stream banks, which can provide considerable shade to the stream channel. Overhanging leaves and branches help to keep the water cool, especially in summer months. Consequently, the removal of riparian vegetation can significantly increase stream water temperature (Brown and Krygier 1970; Karr and Schlosser 1977; Holtby 1988). Loss of riparian stream bank vegetation can also increase daily water temperature fluctuations and reduce temperatures in winter months (Beschta et al. 1987). Water temperature is an important aspect of overall water quality, and can be the determining factor in whether cool water organisms are able to inhabit the stream.

Riparian zones are able to retain water-suspended nutrients, including various forms of nitrogen and phosphorous, before they reach the stream channel (Osborne and

Kovacic 1993; Hill 1996; Casey et al. 2001). Reducing the amount of inorganic nutrients that reach the stream helps maintain low water turbidity and limit the rise of harmful algal blooms in downstream lakes and estuaries (Nebel 1990; Burkholder and Glasgow 1997). Riparian vegetation physically slows down and takes up dissolved nutrients as they move, via surface runoff and shallow subsurface flows, from the upland to the stream (Tabacchi et al. 2000). Nutrients also leach into the riparian soil where the may be processed by bacteria, or eventually absorbed by plants. In this way, riparian zones act as effective nutrient sinks (Lowrance et al. 1997; Groffman 2002). Riparian vegetation, however, can also contribute nutrients to the stream in the form of leaves and other organic material (Gregory et al. 1991).

As with dissolved nutrients, riparian zones are effective at retaining sediment bound for the stream channel. Sediment, especially in urban areas, commonly originates from pockets of soil erosion (Waters 1995; Freeman and Schorr 2004). As sediment moves down the hillslopes and through the riparian zone, it comes into contact with a variety of vegetation and ground litter (leaves, branches, logs). This interaction slows down surface sediment flow, increasing infiltration of water and suspended sediment particles (Castelle et al. 1994).

In addition to water quality, riparian zones can significantly impact the hydrology and geomorphology of the stream. For instance, riparian soil, litter, and vegetation interrupt the flow of water that carries both nutrients and sediment from the upland matrix toward the stream. Infiltration of surface water into the soil profile, and subsequent uptake by riparian vegetation, significantly reduces the amount of water that reaches the stream channel. It also slows the rate at which storm water is released into the stream following periods of heavy precipitation (Booth 1990). This relatively slow release of storm water not only helps reduce the amount of nutrients and sediment that reaches the stream, it also decreases stream bank incision by moderating the volume and rate of in-stream flow (Paul and Meyer 2001; Henshaw and Booth 2000).

The geomorphology of the stream is strongly influenced by a variety of large woody debris (LWD), provided by riparian vegetation. LWD, which include large branches, tree trunks, and rootwads, are usually washed into the stream channel from the surrounding watershed during flood events, or fall directly into the channel from adjacent stream banks. LWD directly impacts the rate and path of in-stream flow by physically blocking the water, and creating small impoundments, such as pools and point bars (Robinson and Beschta 1990; Forman 1997). The speed of water flow, especially in lower order channels, is commonly reduced by LWD and the impoundments they create. By dissipating some of the hydrologic energy of the stream flow, LWD helps to decrease incision and stabilize the stream bank (Bilby 1988).

The riparian zone helps sustain the biological structure and fitness of the stream ecosystem by performing the functions necessary to maintain water quality, and preserving natural geomorphic and hydrological conditions. Without riparian vegetation, for example, the water temperature of the stream would increase, leading to a dramatic decrease in dissolved oxygen concentration. Many aquatic organisms, including salmon and steelhead, require high levels of dissolved oxygen (> 4 mg/l), and would be negatively affected by an increase in water temperature (Alabaster and Gough 1986;

Mueller and Stadelmann 2004). Furthermore, without the filtering capacity of adjacent riparian zones, the diversity and composition of the entire aquatic community may be disrupted by the excessive influx of sediment and nutrients. Although the overall biomass of the stream may actually increase as result, the composition of organisms will change dramatically as turbidity increases and oxygen levels drop. Overall species diversity generally decreases as many oxygen-dependent fish and benthic macroinvertebrates and plants are replaced by species tolerant of eutrophic conditions (Sand-Jensen and Riis 2001).

Another critical biological function of riparian vegetation is the contribution of LWD to the stream channel. Not only does LWD stabilize the stream bank, it provides shelter for a variety of fish and other aquatic organisms. Deep pools that rarely form without the input of LWD are frequently the habitat of many species of fish (Keller 1971; Forman 1997; Opperman and Merenlender 2004). Trout density and biomass, for instance, was found to be greater in stream segments with LWD dams then in control segments without impoundments (Lehane et al. 2002).

Riparian vegetation is also an important source of food at the base of the aquatic food web, especially for low order streams. A substantial amount of organic matter is deposited into the stream from falling leaves and other plant material. Bacteria, fungi, and protozoa break down the organic material into detritus, which is consumed by microcrustacea and a variety of aquatic organisms (Forman 1997; Ribblett et al. 2005). Microcrustacea are typically consumed by macroinvertebrates, which are themselves food for fish, amphibians, and other members of the stream community. When the input of organic material from riparian vegetation is diminished or removed, the entire aquatic food web can be disrupted (Johnson and Wallace 2005).

Riparian Zones as Nutrient Buffers

The role of riparian zones as "nutrient buffers" has been documented by numerous studies conducted throughout the past few decades. Collectively, these studies have shown that most riparian zones can filter a variety of nutrient pollutants moving from the upland matrix to the stream via surface runoff and subsurface water flow (Casey et al. 2001). A pioneering study by Peterjohn and Correll (1984) found that a riparian forest in Maryland filtered 45 kg nitrate per ha, per year, from subsurface water moving from agricultural land towards the stream. Pinay (1986) estimated that riparian forest vegetation retained 75% of the N and 45% of the P applied to nearby cropland. Several hundred studies have since been conducted, firmly establishing the riparian as a critical sink for several nutrients, especially nitrate (NO_3^-), ammonium ($NH4^+$), and phosphorus (P) (e.g., Hill 1996; Lowrance et al. 1997; Casey et al. 2001; Groffman et al. 2002; Hill et al. 2004).

Nitrate

Nitrate, the most common pollutant found in U.S. drinking water, is a common derivative of anthropogenic activities, including fertilizer application, waste water treatment, and fossil fuel combustion (USEPA 1990; Groffman et al. 2002). Excessive nitrate levels in both fresh water and estuarine ecosystems can facilitate eutrophication (Vitousek et al. 1997). Under eutrophic conditions, algal blooms greatly disturb the ecosystems of lakes, rivers, and coastal waters by increasing turbidity and decreasing dissolved oxygen content. Benthic plants die off as a consequence of reduced sunlight infiltration, and a variety of fish suffocate from lack of oxygen (Nebel 1990). High levels of nitrate in wetlands can also decrease native species richness by enhancing the colonization and growth of invasive vegetation (Green and Galatowitsch 2002).

Due to its potential to undermine the water quality and integrity of wetland ecosystems, the majority of riparian-oriented research has focused on the movement and biogeochemical processing of nitrate in agricultural watersheds (Peterjohn and Correll 1984; Vought et al. 1995; Lowrance et al. 1997). This research has yielded several important insights into the movement of NO_3^- and the processes capable of mitigating its release into freshwater streams and marine waters. First, nitrate anion moves freely with surface and groundwater flow. Thus, it can easily move from source locations (usually upland crop fields) to nearby streams either with surface runoff or by leaching into groundwater (Peterjohn and Correll 1984; Vought et al. 1995; Lowrance et al. 1997). Second, the concentration of NO_3^- in water may decrease substantially as it moves along riparian flow paths (Hill 1996; Gold et al. 1998; Casey et al. 2001). Third, the capacity of riparian zones to attenuate NO_3^- is highly dependent upon the ability of riparian vegetation and soil microbes to capture and process the nutrient. Anaerobic microbes convert NO_3^- to N gases (N_2O, N_2O, N_2) through denitrification (Groffman and Crawford 2003). The N gases are then released into the atmosphere and liberated from the ecosystem. In addition, most riparian vegetation can readily take up nitrate from the soil, incorporating the nutrient into various tissues (Firestone, 1982; Groffman 2002).

Ammonium

Ammonium (NH_4^+) is another prevalent form of inorganic nitrogen utilized by plants. Most NH_4^+ enters the ecosystem through organic matter decomposition, and as a relatively minor source, fixation by soil-dwelling organisms, which reduce atmospheric N_2 to NH_4^+ (Nebel 1990; Kimmins 1997). Like nitrate (NO_3^-), ammonium is also deposited as a component of several common fertilizers, such as ammonium sulfate ($(NH_4)_2SO_4$) and ammonium nitrate (NH_4NO_3) (Aldrich 1980).

Negatively charged soil particles, including clay and humus, attract NH_4^+ electromagnetically to their surface. As a result, NH_4^+ is typically bound to the surfaces of soil particles, and may be less likely to leach into groundwater or move via water flow in soil (Aldrich 1980). Nonetheless, NH_4^+ is readily taken up by the root systems of plants and a wide variety of soil microbes. The decay of these organisms returns organic nitrogen to the soil (Aldrich 1980; Nebel 1990; Paul and Clark 1996).

The rate at which NH_4^+ is absorbed by plants depends on several factors, including soil composition, temperature, pH, and availability of other nutrients. An increase in pH (up to 7.4), temperature (up to 30° C), soil Ca content, and organic matter content typically increases the rate of NH_4^+ absorption (Marcus-Wyner and Rains 1982; Singh et al. 1984; Fenn and Taylor 1990).

In addition to absorption by plant and soil microbes, NH_4^+ may also be oxidized to NO_3^- by nitrifying bacteria, becoming free anions in soil water, or reduced to anhydrous ammonia when soil pH is high, and released to the atmosphere via volatilization (Aldrich 1980; Pinay et al. 1993). Each of these processes may occur as NH_4^+ transverses the riparian zone, making it an effective sink for soil NH_4^+.

Phosphorus

Phosphorus (P) is a highly limiting macronutrient for plant growth in many freshwater ecosystems, second only to nitrogen (Schachtman et al. 1998). For this reason, phosphorus in the form of phosphate (PO_4^{-3}) is a common component of many fertilizers applied to agricultural fields and urban lawns. Phosphates are also routinely used in laundry detergents as a water softener, although this practice has been strictly regulated in many areas and prohibited in at least 27 states (Nebel 1990; Litke 1999). Nevertheless, large quantities of P are continuously transported with sediment via surface runoff to lakes, streams, and rivers where it contributes to (primarily freshwater) eutrophication (Carpenter et al. 1998).

Although soil usually contains a large quantity of phosphorus, most of it exists in unavailable forms (Troeh and Thompson 1993). The phosphorus contained in organic matter is firmly held within the structure of that material until it fully decomposes. Inorganic phosphorus, which comes from the mineral apatite, is typically adsorbed by iron and aluminum compounds in the soil (Troeh and Thompson 1993). Dissolved inorganic phosphorus, usually in the form $H_2PO_4^-$ or HPO_4^{-2}, are readily taken up by plant roots via active or passive transport (Troeh and Thompson 1993; Schachtman et al. 1998). Symbiotic mycorrhizal fungi that grow out into the soil from plant roots often enhance uptake of phosphorus (Brady and Weil 2001). However, excessive soluble inorganic P, i.e., $H_2PO_4^-$ and HPO_4^{-2} can easily react with soil Al, Fe, and Mn, and precipitate in the soil profile (Brady and Weil 2001). Large quantities of phosphorus adsorbed to soil particles may be transported by soil erosion. Unlike nitrogen, however, phosphorus cannot to be liberated from the ecosystem in a gaseous phase. Therefore, riparian zones may be characterized as areas of phosphorus retention rather than removal (Casey et al. 2001).

Riparian Models

Several models have been developed to describe and predict riparian buffer zone filtering capacity. Philips (1989) developed two physics-based models to describe buffer zone effectiveness: the Hydraulic Model and Detention Model. Both models compare the buffer zone under study with an arbitrary reference buffer. The Hydraulic model deals exclusively with the surface flow of sediments and sediment-bound materials, whereas the Detention Model also takes into consideration subsurface flow. The Philips Models have yet to be calibrated or validated experimentally using field data. In addition, Philips' reference buffers were chosen, in his own words, "somewhat arbitrarily," which may limit the accuracy of the models (Philips 1989).

The Riparian Ecosystem Management Model (REMM) simulates daily surface and subsurface water movement, nutrient cycling, sediment transport and deposition, and plant growth within a three-zone riparian buffer running parallel to the stream (Lowrance et al. 1998). Each zone is classified by degree of management, with the least managed zone, consisting of undisturbed forest, closest to the stream bank. The REMM computer program is designed to allow riparian buffer managers to determine the impact of several physical and climatic variables, including slope, vegetation, precipitation, and soil, on the water quality of the adjacent stream (Lowrance et al. 1998).

Initial testing of the model's hydrological component at the Southeast Watershed Research Laboratory in Tifton, GA showed that REMM was capable of producing a reasonable estimation of water table levels in a three-zone riparian buffer **(**Inamdar et al. 1998a). A separate study was carried out to evaluate the nutrient component of REMM. Nitrate concentrations in subsurface flow calculated by REMM were significantly close approximations in two out of the three zones of a single riparian buffer. REMM predictions of overall soil water nitrate concentrations within each zone, however, were significantly lower than observed (Inamdar et al. 1998b).

Major features of the REMM model include its complexity and data-intensive requirements. Therefore, its applicability is likely limited to organizations capable of conducting data intensive studies. For those few with the necessary resources, however, REMM may prove to be a useful and realistic riparian management tool once calibrated to local conditions.

Another well-known riparian model is the Chemical, Runoff, and Erosion from Agricultural Management Systems (CREAMS) model. CREAMS simulates runoff, erosion, and chemical transport in field-sized agricultural watersheds (Kinsel 1980). Several studies have been conducted to test and evaluate the model. Silburn and Freebairn (1992) found that CREAMS produced reasonable estimates of total soil moisture content, but consistently overestimated runoff volumes by 1 to 39 percent, in two areas with self-mulching Vertisol soil. In addition, CREAMS accurately estimated the removal of sediment by grass filter strips (Flannagan et al. 1989), as well as nitrogen and phosphorus losses from agricultural fields undergoing different tillage practices (Yoon et al., 1992).

Several catchment-scale models, including the Environmental Management Support System (EMSS) (Vertessy et al. 2001) and the Soil and Water Assessment Tool (SWAT) (Neitsch et al. 2002), contain components that model riparian buffer performance. The ability of riparian zones to affect the water quality of entire watersheds provides the basis for their incorporation into these broad-scale simulations. EMSS is a composite of several other models that collectively simulate daily transport and storage of suspended sediment, nitrogen, and phosphorus at scales of mid-sized watersheds consisting of a number of sub-catchments.

SWAT, like EMSS, is a physically based process model that operates on a daily time step (Arnold et al. 1998; Neitsch et al. 2002). In general, the SWAT model is primarily used to evaluate the long-term consequences of different agricultural management practices on large watersheds and river basins. Components of EMSS simulate a wide variety of watershed-scale parameters, including soil temperature, crop growth, weather, and the daily movement of nutrients, pesticides, and sediment. Surface runoff is calculated using a function that varies non-linearly with soil moisture content (Arnold et al. 1998; Neitsch et al. 2002).

These models, with the possible exception of the Philips Model, are highly complex and data-intensive. They are designed to produce estimations of a wide variety of physical parameters at different scales. CREAMS, SWAT and EMSS are specifically used to simulate conditions in catchments and watershed-scale areas. Their suitability as riparian-scale models has not been confirmed. REMM and the Philips Model operate on a site-level, riparian scale, however REMM is too cumbersome for many practical applications, and the Philips Model has yet to be sufficiently validated. Each of these models is also process-based, and not developed from field data. The model proposed in this study, the Riparian Nutrient Attenuation Model (RNAM), is unique in that it is a statistical model based upon field data, is simple to use, and is designed specifically to estimate nutrient retention in riparian systems.

CHAPTER III

PURPOSE OF STUDY

The purpose of this study was to develop a simple statistical model capable of evaluating riparian zones in terms of their relative capacity to buffer common nutrient pollutants, including total nitrogen (N), nitrate (NO_3^{-1}), total phosphorus (P), and phosphate (PO_4^{-3}). By accurately estimating the percent nutrient attenuation of select riparian zones, the Riparian Nutrient Attenuation Model (RNAM), will enable the quantitative comparison of multiple riparian zones within a watershed, or along a particular stream. This will allow land managers to select only the optimal riparian zones for the broad application of stormwater, designed to reestablish the natural role of riparian corridors as nutrient buffers in urban areas.

In developing RNAM, the relationships between individual variables used in the model, including width, slope, vegetation, and soil, and riparian nutrient retention were also evaluated. The nature and extent of these relationships may be useful in further improving the RNAM model, as well as developing other relevant models and best land management strategies.

Research Objectives

1) Develop a simple, yet robust, riparian nutrient attenuation model (RNAM) to predict the percent reduction of four common nutrients (nitrogen, phosphorus, nitrate, phosphate) by riparian buffer zones

2) Test the accuracy of the model using randomly selected field data.

3) Determine degree of correlation, and potential relationship, between width, slope, vegetation type, soil type and riparian buffer zone nutrient reduction.

CHAPTER IV

MATERIALS AND METHODS

Data obtained from previously conducted research on nutrient retention in riparian buffer zones (Appendix A) will be used to develop a riparian nutrient attenuation model (RNAM) for total nitrogen, phosphorus, nitrate, and phosphate. RNAM, as a relatively simple statistical mode, will represent a practical alternative to more complex process-based models, including REMM and CREAMS. Values of nutrient retention obtained from RNAM will constitute rough preliminary estimations, particularly useful to land managers interested in quickly gauging the nutrient retention potential of different riparian systems.

Basis for Variable Selection

As a natural system, riparian zones, and the catchments in which they exist, are highly complex. There are countless variables, many spatially heterogeneous and temporally dynamic, which influence the ability of riparian buffer zones to trap and process nutrients as they transverse the system. Some variables, however, contribute to overall riparian buffer effectiveness more than others. Decades of research have provided a relatively short list of key variables that are known to contribute significantly to riparian buffer effectiveness. These variables include buffer width (perpendicular from the stream to edge of riparian), slope, soil characteristics (moisture content, organic matter content, infiltration rates, water retention capacity, roughness coefficient, structure, composition, temperature, pH, and nutrient loading rates), vegetation type and density/cover, catchment size, land use, and precipitation and climate.

Complex riparian models, such as REMM, attempt to incorporate each of these variables in great detail to simulate the buffer processes mechanistically. In order to construct less data-intensive models, only four key variables will be used in RNAM: riparian width, slope, soil type, and vegetation type. These variables were chosen primarily because: 1) they contribute directly or indirectly to the ability of riparian buffers to remove key nutrients, such as nitrogen and phosphorus, 2) they are recorded in nearly all riparian-nutrient studies, allowing RNAM to be constructed using a diverse collection of pre-existing data, and 3) they are relatively easy to measure and evaluate in the field, providing convenience to land managers.

Width

Several studies have demonstrated a strong correlation between riparian buffer width and nutrient retention. Dillaha et al. (1989) demonstrated that grass buffer strips of 9.1m wide could remove significantly more nitrogen, phosphorus, and nitrate than those of 4.6m wide. In at least two trials involving the 4.6m buffer strip, percent nitrate retention was negative, indicating that the short buffer released more nitrate than entered. Similarly, Chaubey et al. (1995) tested the nitrogen and phosphorus removal effectiveness of five grassy riparian strips of varying width (3.1, 6.1, 9.2, 15.2, and 21.4m). All other variables, including slope, were kept constant. Both nitrogen and phosphorus retention increased with increasing buffer width. The most significant increases in nutrient retention occurred between 3.1m and 6.1m, while the least significant increases occurred between 15.2m and 21.4m. Additional studies have revealed a similar pattern, indicating that the majority of nutrient retention by riparian buffers occurs within the first 5 to 10m (Patty et al. 1997; Schmitt et al. 1999; Uusi-Kamppa et al. 2000; Abu-Zreig et al. 2003; Blanco-Canqui et al. 2004).

Slope

The slope of the land bordering either side of the stream is considered one of the most important variables influencing the speed at which runoff passes through riparian buffers. The steeper the slope, the faster runoff moves through the riparian buffer, and the less time there is for soluble nutrients to infiltrate riparian soils to be processed. Despite this widely accepted relationship, few studies have directly investigated the effect of varying slope on riparian nutrient removal.

Patty et al. (1997) examined the effect of width on the removal of several agricultural pollutants, including nitrate and phosphorus, in 12 riparian field plots. Four plots of varying widths were each constructed at three experimental sites throughout France. The three sites were recorded as having different average slopes: 7%, 10%, and 15%. Although not discussed by Patty et al. (1997), there appears to be a limited relationship between slope and percent nitrate removal. The experimental site with 7% slope was consistently more effective at retaining nitrate than the site with 10% slope. However, there does not appear to be a significant difference in nitrate retention between the sites with 7% and 15% slope. Furthermore, a greater percentage of soluble phosphorus was retained at the site with 15% slope than at either of the two other sites. Differences in site conditions, such as soil texture and structure, watershed size and topography, and land management practices, make it difficult to compare the effects of slope on nutrient removal across the three experimental sites.

The experimental riparian plots used in Patty et al. (1997) had moderate slopes (15% or less) and a wide range of nutrient removal efficiencies. It is generally believed that riparian buffers with steep to very steep slopes (perhaps 25% or greater) are largely inadequate for removing contaminants (Cohen et al. 1989; Schueler 1995). Furthermore, it is well documented that significant increases in percent slope reduce soil infiltration rates, a critical step in nutrient retention (USDA-SCS 1984).

Vegetation

Riparian vegetation has a significant impact on both surface and subsurface flow. High stem density reduces the velocity of surface water, allowing more time for dissolved nutrients and sediment to infiltrate the soil. Plant roots can also increase infiltration rates by creating root channels, and reducing soil bulk density. Nutrients in subsurface flow are primarily retained via plant root uptake and denitrification by soil microbes (Tabacchi et al. 2000). Over time, vegetation increases the organic content of the soil, leading to higher denitrification rates (Osborne and Kovacic 1993).

The density, age, and type of vegetation can all effect riparian nutrient retention. Riparian buffers covered primarily with grass and other herbaceous vegetation with high stem densities typically trap sediment and nutrients faster than mixed hardwood and pine forests. Due to the presence of more extensive root systems, however, the soils of riparian forests are often less compact, and thus more favorable to infiltration (Bharati et al. 2002; Schoonover et al. 2005). The age of vegetation tends to have a greater impact on nutrient retention in forested riparian zones. Compared with mature forests, younger forests may sequester more nitrogen and phosphorus through additional growth (Mander et al. 1997).

Several researchers have suggested periodic harvesting of riparian vegetation to maintain nutrient retention rates (Kimmins 1977; Groffman et al. 1991; Vought et al. 1994). Phosphorus, which cannot be liberated to the atmosphere like nitrogen, is commonly released from older, phosphorus-saturated forests, which may lead to a net increase in phosphorus release to the stream (Osborne and Kovacic 1993).

Despite their differences, grass, forest, and mixed grass-shrub-forest riparian are all reasonably effective at retaining nutrients (Schmitt et al. 1999; Lee et al. 2000; Lee et al. 2003). By directly comparing forested and herbaceous riparian buffers, Schoonover et al. (2004) found significantly higher nitrate retention in the herbaceous (predominantly giant cane, *Arundinaria gigantea*) riparian buffer at 3.3m. By 10 meters, however, both riparian plots demonstrated near 100% retention. High stem density and a thick litter layer are believed responsible for the more rapid nitrate retention in the cane-dominated riparian buffer (Schoonover et al. 2004).

Other studies have suggested that forested riparian zones may be equally as capable of attenuating nutrients. Schmitt et al. (1999) found no significant difference in percent mass reduction of contaminants, including total phosphorus and total nitrogen, among 7.5m and 15m wide 2-yr-old grass and grass-shrub-forest riparian plots. Interestingly, there was a noticeable difference in nutrient retention between the 2-yr-old and 25-yr-old grass plots; the 25-yr-old grass plots tended to be more effective at removing nearly every contaminant. This directly opposes the trend observed between forest age and nutrient retention (Osborne and Kovacic 1993, Mander et al. 1997). As demonstrated by Abu-Zreig et al. (2003), riparian plots with no vegetation (bare soil) perform considerably worse at retaining nutrients than either grass or forest buffers. Although intuitive, this finding does provide evidence for the role of vegetation in riparian nutrient retention.

Soil

The relationship between soil and riparian nutrient retention is highly complex. Soil texture, structure, moisture content, organic matter content, temperature, CEC, pH, etc., can all effect nutrient retention. Several studies have demonstrated the relationship between denitrification rates and certain soil properties. Denitrification is generally higher in soils with more organic matter and moisture, higher temperatures, and less dissolved oxygen (Groffman et al. 1991; Hanson et al. 1994; Willems et al 1996; Schnabel et al. 1997; Cey et al. 1999). Changes in these soil conditions may therefore have a significant impact on overall nitrogen attenuation.

Soil texture and composition as outlined by the US Department of Agriculture (USDA-SCS 1984) has not been shown to have a direct impact on riparian nutrient retention. However, soil texture and composition clearly affect rate of surface water infiltration, water holding capacity, and nutrient sorption by soil particles (USDA-SCS 1984; Troeh and Thompson 1993; Gerrard 2000).

Model Development

Categorization of Vegetation and Soil

It was first necessary to categorize the two qualitative variables in RNAM, vegetation type and soil type. Vegetation was classified into three broad groups for incorporation into RNAM: grass, forest, and bare ground. Soil, as described by their USDA texture class, was then divided into four major soil groups based upon their infiltration rate and runoff potential. This categorization system is based directly upon the USDA's four hydrologic soil groups (USDA 1986). Soil category A contains sand, loamy sandy, and sandy loam soils. These soils, commonly found along stream banks, exhibit high infiltration rates and a low runoff potential. Soil category B consists of the soils silt loam and silt. Soil profiles composed primarily of silt have moderate infiltration rates and a moderate runoff potential. They are also moderately to well drained and have a fine to somewhat coarse texture. Sandy clay loam is the only soil texture in category C. This soil is characterized by a layer of moderately fine to fine particles that hinder the downward movement of water. Infiltration rates are low, and runoff potential is moderate to high. Soil category C was not represented in the study; there was no soil with a sandy clay loam texture. Finally, soil category D contains the soils clay loam, silty clay loam, sandy clay, silty clay, and clay. These clay soils have very low infiltration rates and the highest runoff potential (USDA 1986).

Creation of Dummy Variables

Following the categorization of vegetation and soil, the two nominal variables were translated into multiple dummy variables for incorporation into RNAM. This translation allowed the two nominal variables to be treated as interval variables in the linear regression model. Where only two classifications were present, such as vegetation types grass and forest for nitrogen and nitrate, a 0 and 1 were used to delineate the two options. In cases where variables contained more than two classifications, such as soil categories A, B, and D for all nutrients, or vegetation types grass, forest, and bare ground for phosphorus, the following coding schemes were used:

vegetation	dv1	dv2
Forest	1	0
Bare Ground	0	1
Grass	0	0

soil	ds1	ds2
Soil Cat. A	1	0
Soil Cat. D	0	1
Soil Cat. B	0	0

In the above table, dv1 represents the dummy variable for forest vegetation, dv2 the dummy variable for bare ground (no vegetation), ds1 the dummy variable for soil category A, and ds2 the dummy variable for soil category D. Grass and soil category B, which contained the greatest number of data points, were chosen as the reference categories against which all other categories within that variable would be measured.

Additional vegetation and soil dummy variables may be created in the future to expand the applicability of RNAM. The availability of data for soil category C, for example, would allow for the creation of a third dummy variable for soil (ds3). This expanded RNAM equation could therefore accommodate riparian zones with sandy clay loam soil (category C).

Linear Transformation of Slope and Width

The two quantitative variables, riparian width and average slope, underwent standard linear transformations. Riparian width, which exhibited a typical logarithmic curve when plotted against percent nutrient reduction, was linearized using a log10 transformation (Figure 4). Slope, which appeared to have an irregular quadratic relationship with percent nutrient reduced, was transformed by squaring each value (Figure 5).

RNAM Equations

A final review of collected data indicated that there were too few phosphate-related data points (n = 27) to both construct and test a RNAM-phosphate model. Phosphate data was therefore eliminated from the study. The largest data set for RNAM calculation was available to phosphorus (n = 75), followed by nitrogen (n = 51) and nitrate (n = 46). Approximately one third of all data points collected for each nutrient were randomly selected and set aside to test RNAM. This left greater than 30 data points for the computation of each model.

Using SPSS statistical software, multiple linear regressions were carried out to produce regression (B) coefficients for each regular and dummy variable. These regression coefficients were then used to form an individual nutrient attenuation model (RNAM) for nitrogen, nitrate, and phosphorus.

The RNAM equation takes the general form:

$$Y = a + \beta_1 \log_{10} x_1 + \beta_2 x_2^2 + \beta_3 x_3 + \beta_4 ds_1 + \beta_5 ds_2 \tag{1}$$

where Y is the percent reduction of nutrient, β the regression coefficient, x_1 the riparian width, x_2 the riparian percent slope, x_3 the dummy variable for vegetation ($x_3 = 0$ if grass and 1 if forest), ds_1 the first dummy variable for soil ($ds_1 = 1$ for soil category A and 0 for all others), and ds_2 the second dummy variable for soil ($ds_2 = 1$ for soil category D and 0 for all others).

The exception to this general form is RNAM-phosphorus, which contains multiple dummy variables for vegetation as well as soil:

$$Y = a + \beta_1 \log_{10} x_1 + \beta_2 x_2^2 + \beta_3 \mathrm{ds}_1 + \beta_4 d\mathrm{s}_2 + \beta_5 dv_1 + \beta_6 dv_2 \qquad (2)$$

where dv_1 is the first dummy variable for vegetation ($dv_1 = 1$ for forest and 0 for all others), dv_2 the second dummy variable for vegetation ($dv_2 = 1$ for bare ground and 0 for all others), and all other variables the same as Eq.1.

Elimination of Non-Significant Variables

Variables that are found to have no significant relationship or correlation with nutrient retention, and do not contribute significantly ($p >> 0.05$) to the predictive capability of each RNAM model, may be dropped from the regression equations. Therefore, Eq.1 and Eq.2 should be considered preliminary base equations from which variables may later be removed.

Testing RNAM

Data points set aside to test RNAM (nitrogen: n = 15, phosphorus: n = 24, nitrate: n = 14), with actual percent nutrient reduction values, were then plotted against model predicted values. Linear regressions were used to indicate correlation between model predicted and actual nutrient reduction values.

Additional Analyses

T-tests and one-way ANOVAs were carried out to determine if nutrient reduction varied significantly between soil and vegetation categories. Regression analyses were also performed to assess the level of correlation between the two continuous variables, slope and width, and nutrient reduction.

CHAPTER V

RESULTS

Individual Variables and Nutrient Reduction

An ANOVA indicated no significant difference in the percent reduction of nitrogen ($p = 0.528$) and nitrate ($p = 0.536$) between vegetation categories (Table 1A and 1B). A significant difference in percent reduction of phosphorus, however, was observed between vegetation categories ($p = 0.007$) (Table 1C). Specifically, there was a significant difference in phosphorus reduction between grassy vegetation and bare ground ($p = 0.009$) (Table 1D). Greater average retention was found, for all nutrients, among riparian buffers with grass vegetation. There was no significant difference in nutrient retention between the three soil categories (nitrogen: $p = 0.340$, nitrate: $p = 0.790$, phosphorus: $p = 0.430$) (Table 2). In addition, no one soil category was associated with the highest average nutrient retention across all three nutrients.

Scatterplots further elucidated the potential effects of vegetation and soil type on percent nutrient reduction. In plots of percent nutrient reduction versus riparian width, vegetation type had a limited effect on nitrogen and phosphorus reduction, as indicated by slight variations in slope and intercept (Figure 1A, 3A). Vegetation type had no noticeable effect on reduction of nitrate (Figure 2A). Mostly small variations in slope and intercept were observed between soil categories A, B, and D for nitrogen, nitrate and phosphorus (Figure 1C, 2C, and 3C). Significantly fewer data points among soil categories A and D, however, makes interpreting the effects of soil difficult. Similarly, scatterplots of percent nutrient reduction versus riparian slope did not yield interpretable results due to a lack of vegetation and soil data for steeper-sloped buffers (>7%) (Figure 1, 2, and 3).

A test of correlation indicated a marginally significant negative correlation between riparian percent slope and nitrate reduction ($r = -0.343$, $p = 0.055$). Non-significant correlation was observed between slope and reduction of total nitrogen ($r = -0.207$, $p = 0.241$), and phosphorus ($r = 0.193$, $p = 0.174$) (Table 3). A significant correlation was found between riparian width and percent reduction of all three nutrients: nitrogen ($r = 0.630$, $p < 0.001$), nitrate ($r = 0.564$, $p = 0.001$), and phosphorus ($r = 0.603$, $p < 0.001$). Scatterplots of percent nutrient reduction versus riparian width and slope indicated a general increase in nutrient reduction with increasing width (Figure 4), and a decrease in nutrient reduction with increasing slope (phosphorus exhibited a decline in reduction only after 10%) (Figure 5).

RNAM Computation

RNAM-nitrogen

Multiple linear regressions were used to produce separate linear models for each nutrient (Table 4, 5 and 6). Nitrogen model regression (B) coefficients indicated a negative relationship between slope (squared) and percent nitrogen reduction (-0.05), and a positive relationship between width (post-logarithmic transformation) and percent nitrogen reduction (55.3) (Table 4C). Thus, for every one percent squared increase in slope, model predicted nutrient reduction (Yn) decreases by 0.05 percent, and for each 10-fold increase in width, e.g., from 1 to 10 meters, model predicted nitrogen reduction increases by 55.3 percent. The regression coefficient for vegetation was -14.4, indicating that forested riparian zones retain 14.4 percent less nitrogen than grassy riparian zones in the RNAM-nitrogen model. Grass is therefore the favored vegetation type for nitrogen retention when combined with all other variables.

Width and vegetation type had statistically significant predictive capability, i.e., the regression coefficient associated with width and vegetation type was significantly different from zero ($p < 0.001$ and $p = 0.045$, respectively). Slope was non-significant ($p = 0.150$). Soil type (represented as two dummy variables) was removed from the model for having substantially non-significant predictive capability (ds_1: $p = 0.697$, ds_2: $p = 0.700$). The overall model was significant ($r = 0.698$, $p < 0.001$). The RNAM equation for nitrogen ((Eq. (3), RNAM-nitrogen) incorporates an intercept of 24.6, and regression coefficients for each variable:

$$Y\mathrm{n} = 24.6 + 55.3 \log_{10} x_1 - 0.05 x_2^2 - 14.4 x_3 \qquad (3)$$

where Yn is the percent reduction of nitrogen, x_1 the riparian width, x_2 the riparian percent slope, and x_3 the dummy variable for vegetation ($x_3 = 0$ if grass and $x_3 = 1$ if forest).

RNAM-nitrate

A multiple regression of nitrate data produced a pattern of regression coefficients similar to that of nitrogen (Table 5). Regression coefficients were negative for slope (-0.22), and positive for width (85.1). Thus, for every one percent squared increase in slope, model predicted nitrate reduction ($Y\text{no}_3$) decreases by 0.20 percent, and for each 10-fold increase in width, model predicted nutrient reduction increases by 82.6 percent. The regression coefficient for vegetation was -23.7, indicating that forested riparian zones retain 23.7 percent less nitrogen than grassy riparian zones in the RNAM-nitrate model.

Slope and width both had significant predictive capability ($p = 0.004$ and $p = 0.001$ respectively), while vegetation was marginally significant ($p = 0.057$). Soil type was once again removed from the model for having substantially non-significant predictive capability (ds_1: $p = 0.310$, ds_2: $p = 0.586$). The overall model was significant ($r = 0.706$, $p < 0.001$). The RNAM equation for nitrate (Eq. (4), RNAM-nitrate) incorporates an intercept of 12.1, and regression coefficients for each variable:

$$Y\text{no}_3 = 12.1 + 82.6\log_{10}x_1 - 0.20x_2^2 - 23.7x_3 \qquad (4)$$

where $Y\text{no}_3$ is the percent reduction of nitrate, and all other variables the same as Eq.3.

RNAM-phosphorus

A final multiple regression for phosphorus data yielded a positive regression coefficient for width (41.3), indicating that for each 10-fold increase in width, model predicted phosphorus reduction increases by 41.3 percent (Table 6). The two dummy variables for vegetation, dv_1 representing forest vegetation, and dv_2 representing no vegetation (or bare ground), each had a negative regression coefficient (-6.76 and -30.9 respectively). This suggests that grass, the base vegetation category (with a coefficient of 0), was the favored condition for total phosphorus reduction.

Riparian width ($p < 0.001$) and the second dummy variable for vegetation, dv_2 ($p = 0.003$) both exhibited significant predictive capability. The non-significance of the first dummy variable for vegetation, dv_1 ($p = 0.258$), indicates that a significant difference in riparian nutrient reduction exists between vegetation categories grass and bare ground, but not grass and forest. Riparian percent slope ($p = 0.241$), the dummy variable for soil category A (ds_1) ($p = 0.774$), and the dummy variable for soil category D (ds_2) ($p = 0.464$) were eliminated from the RNAM-phosphorus equation for having non-significant predictive capability. The overall model was significant ($r = 0.694$, $p < 0.001$). The RNAM equation for phosphorus (Eq. (5), RNAM-phosphorus) incorporates an intercept of 34.5 and regression coefficients for each variable:

$$Y\text{p} = 34.5 + 41\log_{10}x_1 - 6.67dv_1 - 41.3dv_2 \quad (5)$$

where Yp is the percent reduction of phosphorus, dv_1 the first dummy variable for vegetation ($dv_1 = 1$ for forest and 0 for all others), dv_2 the second dummy variable for vegetation ($dv_2 = 1$ for bare ground and 0 for all others), and all other variables the same as Eq.3.

RNAM Evaluation

Significant ($p < 0.001$) linear correlations between model-predicted and actual percent nutrient reduction values were observed for all nutrients. Correlation was highest for RNAM-nitrogen ($n = 15$, $r = 0.850$), followed by RNAM-phosphorus ($n = 24$, $r = 0.840$) and RNAM-nitrate ($n = 14$, $r = 0.825$) (Table 7, Figure 6). Furthermore, RNAM-nitrogen predicted most nitrogen reduction values within an error range of ± (10 – 20)% of their actual value, indicating relatively high accuracy (Figure 7). Over and under estimates of nitrogen retention are also fairly well balanced, with a slight tendency towards overestimation at high percentages (Figure 7A). RNAM-nitrate had the least overall accuracy, with two predicted nitrate reduction values nearly ± 40% from the actual. Most predictions, however, were also within ± (10 – 20)% of the actual. The number of over and under estimates produced by RNAM-nitrate were random (Figure 7B). The accuracy of RNAM-phosphorus was between that of RNAM-nitrogen and RNAM-nitrate. Over and under estimates by RNAM-phosphorus were also nearly even (Figure 7C).

CHAPTER VI

DISCUSSION

Potential Effects of Individual Variables

The ability of riparian buffer zones to reduce nitrogen, nitrate, phosphorus, and other nutrient chemicals has been associated with a wide variety of biological, chemical, and physical factors (Peterjohn and Correll 1984; Hill 1996; Hedin et al. 1998). In this study, significant associations were found between the reduction of at least one nutrient and the individual riparian variables width, slope, and vegetation type. Across all three nutrients, the most significant relationship was observed between riparian width and percent nutrient reduction, followed by slope and vegetation type. As a single variable, soil type was not significantly associated with nutrient reduction.

The strong positive correlation between riparian width and percent nutrient reduction is well supported by the literature (Dillaha et al. 1989; Vought et al. 1994; Chauby et al. 1995). In a manner similar to this study, Vought et al. (1994) collected and plotted 34 data points of riparian width versus percent nitrate reduction. The data indicated that rapid reduction of nitrate occurred within the first 10 meters of the riparian buffer. By 20 to 25 meters, the increase in percent nitrate reduction leveled off and remained near 90 – 100 percent through the maximum recorded width of 70 meters. This relationship was immediately discernable in this study as well. Although the collected data included a maximum riparian width of only 26 meters, percent reduction of each nutrient increased dramatically between 0 and 15 meters, and leveled off entirely by 15 to 20 meters (Figure 4). These data suggest that riparian zones need not be wider than 25 meters to achieve maximum nutrient retention.

Slope had minimal impact on nutrient retention up to 10 to 12 percent. Beyond this point, however, the reduction of nitrogen, nitrate, and phosphorus all began to decrease (Figure 5). No riparian buffer included in this study had an average slope greater than 16 percent. This severely limits the ability of the current RNAM model to accurately predict nutrient reduction values in relation to steep slopes of 20 percent or more. In fact, a linear regression of percent phosphorus reduction and percent riparian slope squared (quadratic transformation) indicated a general rise in the reduction of phosphorus with increasing slope (Figure 5F). This unexpected trend may be misleading because it is likely the product of an uneven distribution of slope data, the vast majority of which is within the 0 - 5 percent range. Over this range of slopes, percent reduction of phosphorus, as well as nitrogen and nitrate, changes very little. Between 5 and 10 percent slope, nutrient reduction increases slightly, then decreases by approximately the same amount between 10 and 16 percent slope. This suggests that, although an increase in slope generally decreases nutrient retention, it may have little effect on nutrient retention when less than 10 (or 15) percent. Therefore, it is recommended that additional nutrient retention data be obtained from riparian zones with moderate to steep slopes (> 15 percent) in order to more accurately represent, in the RNAM models, the linear relationship between nutrient retention and percent riparian slope.

The effect of vegetation type on nutrient retention was significant for phosphorus only (Table 1C). The phosphorus data contained two riparian buffers with no vegetation (bare ground). The significant difference in phosphorus retention between grass vegetation and bare ground produced an overall significant difference between vegetation types. The difference in phosphorus retention between the two more numerous vegetation types, grass and forest, however, was essentially the same as for nitrogen and nitrate. The vast majority of riparian systems used in this study had predominately grassy vegetation. A more even distribution of data may produce different results.

The length and intensity of natural and simulated precipitation events and surface water flow varied between studies. The incorporation of data obtained from high intensity water applications may have played a role in the significant difference in retention of phosphorus between vegetation types. In surface runoff, phosphorus clings tightly to sediment particles, often making it more susceptible to interception by vegetation and other surface obstructions than nitrogen. In this way, it is possible that during periods of high intensity rainfall, phosphorus may be affected by surface vegetation to a greater extent than other common nutrients (Owens and Shipitalo 2006).

Nutrient retention was greatest, though non-significant, among grassy riparian zones for all nutrients. This was not unexpected given that a majority of nutrient samples were collected within a few centimeters of the riparian surface. With a significantly higher stem density, grass is generally capable of trapping water and dissolved nutrients more efficiently than trees and other less dense vegetation. Grassy riparian zones in urban areas, however, are often utilized as city parks, and undergo significant soil compaction.

The compaction of soil reduces infiltration, and hinders the buffering capacity of the riparian zone. As a result, more surface runoff reaches the stream, though soil water nutrient concentrations remain low (Bishop and Mou 2004). If a greater proportion of samples had been taken several meters below the surface, it is possible that forested riparian zones, with deeper root systems and less compact soil, would have exhibited higher levels of nutrient retention.

Scatterplots further demonstrate the potential effect of vegetation and soil on nutrient retention (Figure 1, 2 and 3). They also illustrate the serious problem that arises when too few data points are available for analysis. The insufficient number of data points for forest and bare ground makes it difficult to compare the effects of all three vegetation categories. This is especially obvious in plots of slope versus percent nutrient reduction, in which all forest, soil category A, and soil category D data points have less then 10 percent slope (Figure 1, 2 and 3). Without a distribution of data points throughout the complete range of slopes, the associated regressions cannot be considered accurate. Additional data could give a much clearer picture as to potential effects of vegetation and soil type on nutrient retention.

Although there was no significant difference in nutrient retention between the three soil categories, it cannot be concluded that soil texture has no bearing on nutrient retention. As with vegetation type, the soil categories in this study contained a very uneven number of data points that are limited within a narrow range of soil variation. Out of 12 possible texture classes, a vast majority of soil data was of the texture type silt loam. This necessitated limiting the number of soil categories in order to maintain a workable number of data points in each category. With only four categories (three of which are currently used in RNAM: A, B, D), the hydrologic soil groups developed by the USDA provided a sensible means of placing the soil data into a limited number of groups. The USDA groups also divided the soil texture classes in terms of their runoff potential and associated infiltration rates, both of which are expected to have an impact on nutrient retention. A larger pool of soil data, with a more even number of data points in each category, may produce results different from those reported in this study.

Variable Predictive Capabilities

The predictive capability of each of the variables in the RNAM linear regression equations largely mirrored that of the extent of relationship, or correlation, between the variable and riparian nutrient retention. As expected, soil type had very low predictive capability in each RNAM model ($p \geq 0.310$). The variable was therefore excluded from each equation. No definite cut-off point in significance level, however, was established for the elimination of variables from RNAM. Therefore, the decision to also eliminate slope ($p = 0.241$) from RNAM-phosphorus and not from RNAM-nitrogen ($p = 0.150$) was a difficult one. Clearly, the available data suggests that soil type adds unduly to the complexity of RNAM without adding significantly to its predictive capability.
Slope, however, exhibits a pronounced polynomial relationship with riparian nutrient retention (Figure 5), suggesting that the variable has the potential to add significantly to the accuracy of RNAM, if not assumed linear within the model equation.

RNAM Performance

Despite the limitations imposed by individual variables, the three RNAM models achieved reasonably accurate estimates of percent nutrient reduction. There was a significant correlation between model predicted and actual nutrient reduction values for each nutrient (Table 7, Figure 6). With at least 14 extra data points to examine each model, the models demonstrated robust predictive power. These results suggest that the RNAM models have the accuracy and precision required to serve as a quick, preliminary tool for estimating the nitrogen, nitrate, and phosphorus reduction potential of riparian buffer zones.

Model Strengths and Limitations

When compared to other riparian models, RNAM appears to have three main advantages: 1) it is derived directly from field data rather than second-hand equations, 2) it is not data intensive or time consuming to use, and 3) it can easily be recalculated, modified and updated by including or replacing additional variables or data points. Complicated mechanistic models, such as REMM and CREAMS, are primarily used to simulate a wide variety of physical, chemical, and biological riparian processes (Kinsel 1980, Lowrance et al. 1998). Though they may have the capacity to do so, they are not specifically designed to estimate the nutrient retention of particular nutrients. In its current state, RNAM is built for this purpose exclusively. Furthermore, because RNAM requires very little data input, nutrient reduction estimations may be calculated in the field without the aid of a computer program.

Unfortunately, the derivation of RNAM directly from field data is currently as much a limitation as it is a strength. There is simply not enough data at this point to make conclusive judgments concerning the predictive capability of the model's two categorical variables, vegetation and soil. Both variables were divided into a small number of restrictive groups, one of which contained only two data points. Many of the groups were also highly variable. The vegetation category forest, for instance, contained riparian zones with a wide variety and degree of forest cover, including many with varying mixtures of forest, grass, and shrubs. Had more data been available, it may have been possible to create additional vegetation categories. Similarly, riparian buffers may contain many different types of soil textures and structures, which made the assignment of one texture class to each riparian buffer a significant oversimplification. In addition, soil category B was predominately composed of a single soil texture class, silt loam. Riparian buffer zones classified as having this texture class represented 80 – 90 percent of all riparian plots used in the study. Many of these riparian buffers may have contained any number of other soil textures. Due to the variability within and between vegetation and soil categories, it is clear that additional data will be necessary to better gauge the potential effects of these variables on nutrient retention.

Recommendations for Future Research

The addition of alternative variables may improve the accuracy of RNAM without adding significantly to its complexity. Such variables may include soil moisture content, soil organic matter content, soil oxygen levels, age of vegetation, and vegetation density.

Research has indicated that each of these variables is capable of influencing the retention of various nutrients (Muscutt et al. 1993; Osborne and Kovacic 1993; Schnabel et al. 1997; Cey et al. 1999). Soil texture, or soil hydrological classification, may also be reinstituted as a variable if additional data indicates a more significant relationship with nutrient retention. Beyond simply adding or replacing variables, more data is needed to better understand the effect of each variable on riparian nutrient retention. It is suggested that additional data either be collected in the field, or extracted from untapped pre-existing records. If feasible, field data collection is the preferred method, as it will allow for the inclusion of variables not commonly described in the literature.

CHAPTER VII

CONCLUSION

Among the many environmentally beneficial natural functions riparian zones perform, the attenuation of potentially harmful nutrient loads is considered one of the most vital. Unfortunately, this critical function is often unrecognized and underutilized in urban environments. The natural hydrological and biogeochemical processes that mitigate the release of nutrients and sediment to the stream are negated through the use of pipes and channels. These anthropogenic alterations undermine water quality and pose a serious threat to the health of aquatic ecosystems. In many regions around the world, the influx of unrestricted nutrient loads can also damage fishing industries, and restrict the access of entire populaces to fresh water.

As part of a comprehensive plan to reinstate the natural role of riparian zones as nutrient buffers, thereby protecting stream water quality, land managers and environmental engineers will require an accurate means of predicting the nutrient retention capacity of riparian systems. A simple model, such as RNAM, may be used to provide a side-by-side comparison of multiple riparian zones, from which one or a few may be selected as having the highest nutrient "fitness." Prior to the use of RNAM in this capacity, however, it is suggested that additional data be used to fully validate each RNAM model, as well as determine the appropriateness of additional variables.

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APPENDIX A

TABLES AND FIGURES

Table 1. Results of t-tests for percent reduction of nitrogen (1A) and nitrate (1B) between riparian vegetation categories. ANOVA results for percent reduction of phosphorus (1C) between vegetation categories. Tukey pairwise comparison (1D) indicating level of difference in phosphorus reduction between vegetation categories.

1A

Veg. Type	N	Mean	Std. Deviation	df	t	Sig.
Grass	27	68.9148	19.30894	32	0.638	0.528
Forest	7	63.3143	25.83973			

1B

Veg. Type	N	Mean	Std. Deviation	df	t	Sig.
Grass	23	64.3696	36.86611	30	0.625	0.536
Forest	9	55.9111	26.45479			

1C

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	3048.034	2	1524.017	5.455	0.007
Within Groups	13410.879	48	279.393		
Total	16458.913	50			

1D

Veg. Type		Mean Difference	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
Grass	Forest	9.98915	7.28444	0.364	-7.6282	27.6065
	No Veg.	37.45581*	12.09108	**0.009**	8.2137	66.6979
Forest	Grass	-9.98915	7.28444	0.364	-27.6065	7.6282
	No Veg.	27.46667	13.64779	0.120	-5.5403	60.4737
No Veg.	Grass	-37.4558*	12.09108	0.009	-66.6979	-8.2137
	Forest	-27.46667	13.64779	0.120	-60.4737	5.5403

*The mean difference is significant at the .05 level.

Table 2A. ANOVA results for percent reduction of nitrogen (2A), nitrate (2B), and phosphorus (2C) between riparian soil categories.

2A

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	932.524	2	466.262	1.117	0.340
Within Groups	12941.696	31	417.474		
Total	13874.220	33			

2B

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	580.493	2	290.246	0.238	0.790
Within Groups	35381.594	29	1220.055		
Total	35962.087	31			

2C

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	624.829	2	312.414	0.858	0.430
Within Groups	17847.756	49	364.240		
Total	18472.585	51			

Table 3. Results of Pearson's correlation between riparian percent slope and width, and percent reduction of nitrogen (3A), nitrate (3B), and phosphorus (3C).

3A

		slope	width
slope	r	1	-0.139
	Sig.		0.433
	N	34	34
width	r	-0.139	1
	Sig.	0.433	
	N	34	34
N reduction	r	**-0.207**	**0.630****
	Sig.	**0.241**	**0.000**
	N	34	34

* Correlation is significant at the 0.01 level (2-tailed).

3B

		slope	width
slope	r	1	-0.020
	Sig.		0.913
	N	32	32
width	r	-0.020	1
	Sig.	0.913	
	N	32	32
NO_3 reduction	r	**-0.343**	**0.564****
	Sig.	**0.055**	**0.001**
	N	32	32

* Correlation is significant at the 0.01 level (2-tailed).

3C

		slope	width
slope	r	1	0.014
	Sig.		0.924
	N	51	51
width	r	0.014	1
	Sig.	0.924	
	N	51	51
P reduction	r	**0.193**	**0.603****
	Sig.	**0.174**	**0.000**
	N	51	51

* Correlation is significant at the 0.01 level (2-tailed).

Table 4. Results of multiple linear regression, including overall correlation (4A), model significance (4B), and regression coefficients (4C) used in RNAM-nitrogen. 4C also indicates the significance of each predictor variable.

4A

Model	r	r^2	Adjusted r^2	Std. Error
N	0.698	0.488	0.436	15.39258

4B

Model		Sum of Squares	df	Mean Square	F	Sig.
N	Regression	6766.275	3	2255.425	9.519	0.000
	Residual	7107.946	30	236.932		
	Total	13874.220	33			

4C

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
N	Constant	24.614	10.841		2.270	0.031
	veg	-14.433	6.911	-0.289	-2.088	0.045
	slope	-.047	.032	-0.203	-1.478	0.150
	width	55.321	11.265	0.655	4.911	0.000

Table 5. Results of multiple linear regression, including overall correlation (5A), model significance (5B), and regression coefficients (5C) used in RNAM-nitrate. 5C also indicates the significance of each predictor variable.

5A

Model	r	r^2	Adjusted r^2	Std. Error
N	0.706	0.499	.445	25.37060

5B

Model		Sum of Squares	df	Mean Square	F	Sig.
N	Regression	17939.397	3	5979.799	9.290	0.000
	Residual	18022.690	28	643.668		
	Total	35962.087	31			

5C

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
N	Constant	12.068	22.522		0.536	0.596
	veg	-23.731	11.949	-0.318	-1.986	0.057
	slope	-0.198	0.063	-0.502	-3.158	0.004
	width	82.643	21.753	0.515	3.799	0.001

Table 6. Results of multiple linear regression, including overall correlation (6A), model significance (6B), and regression coefficients (6C) used in RNAM-phosphorus. 6C also indicates the significance of each predictor variable.

6A

Model	r	r^2	Adjusted r^2	Std. Error
N	0.694	0.482	0.449	13.47241

6B

Model		Sum of Squares	df	Mean Square	F	Sig.
N	Regression	7928.140	3	2642.713	14.56	0.000
	Residual	8530.772	47	181.506		
	Total	16458.913	50			

6C

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
N	Constant	34.501	7.140		4.832	0.000
	dv1*	-6.761	5.904	-0.121	-1.145	0.258
	dv2	-30.922	9.827	-0.334	-3.147	0.003
	width	41.316	7.968	0.552	5.185	0.000

*dv1and dv2 are dummy variables for vegetation categories grass and bare ground (no vegetation) respectively.

Table 7. Results of RNAM tests. Correlation between model predicted and actual percent nitrogen (7A), nitrate (7B), and phosphorus (7C) reduction values.

7A

		predicted
actual	r	**0.863***
	Sig.	**0.000**
	N	15

* Correlation is significant at the 0.01 level (2-tailed).

7B

		predicted
actual	r	**0.761***
	Sig.	**0.002**
	N	14

* Correlation is significant at the 0.01 level (2-tailed).

7C

		predicted
actual	r	**0.843***
	Sig.	**0.000**
	N	24

* Correlation is significant at the 0.01 level (2-tailed).

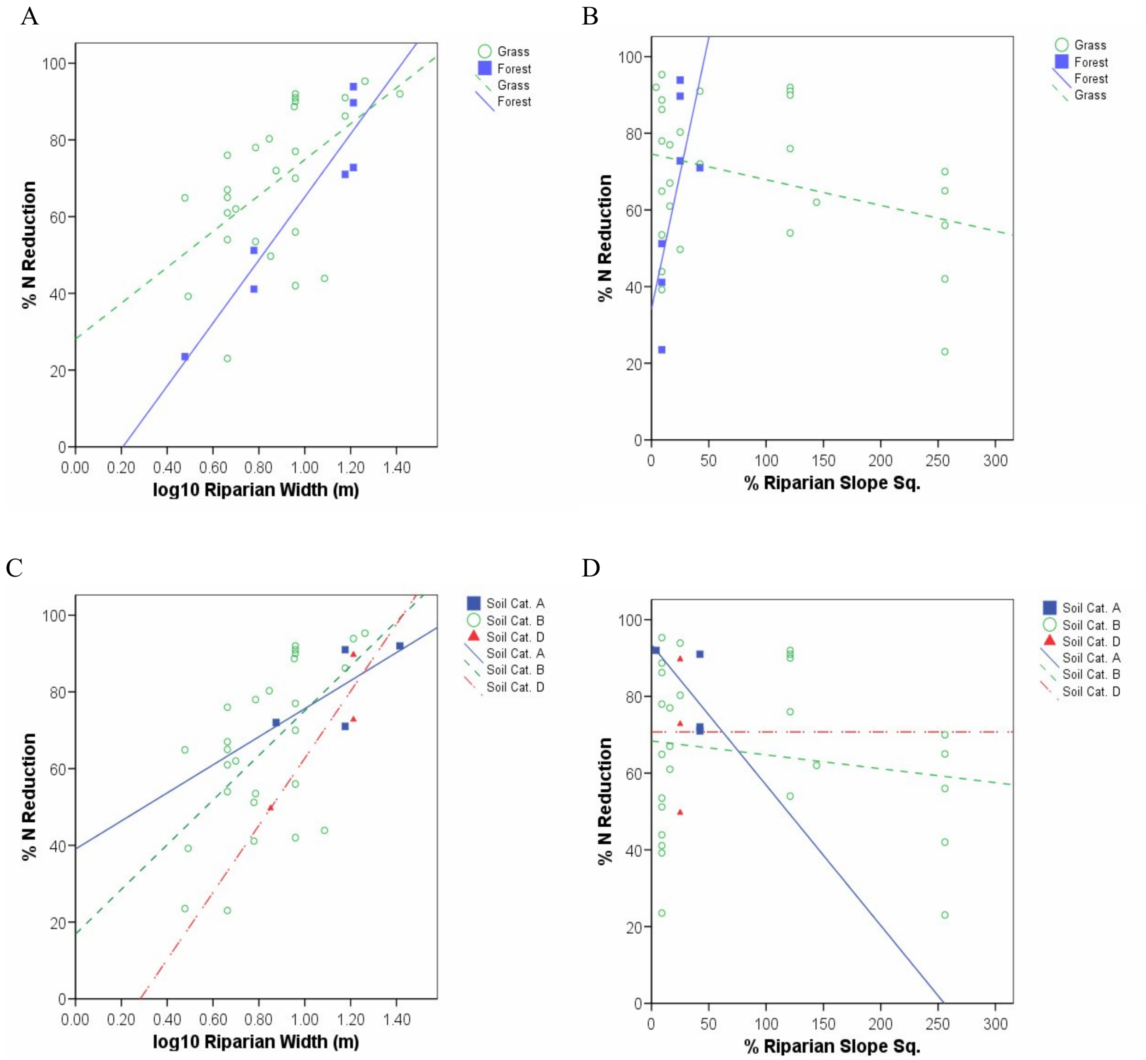


Figure 1. Percent nitrogen reduction in riparian buffers with grass and forest vegetation as separate functions of log10 riparian width (1A) and percent riparian slope squared (1B). Percent nitrogen reduction in riparian buffers with soil categories A, B, and D as separate functions of log10 riparian width (1C) and percent riparian slope squared (1D).

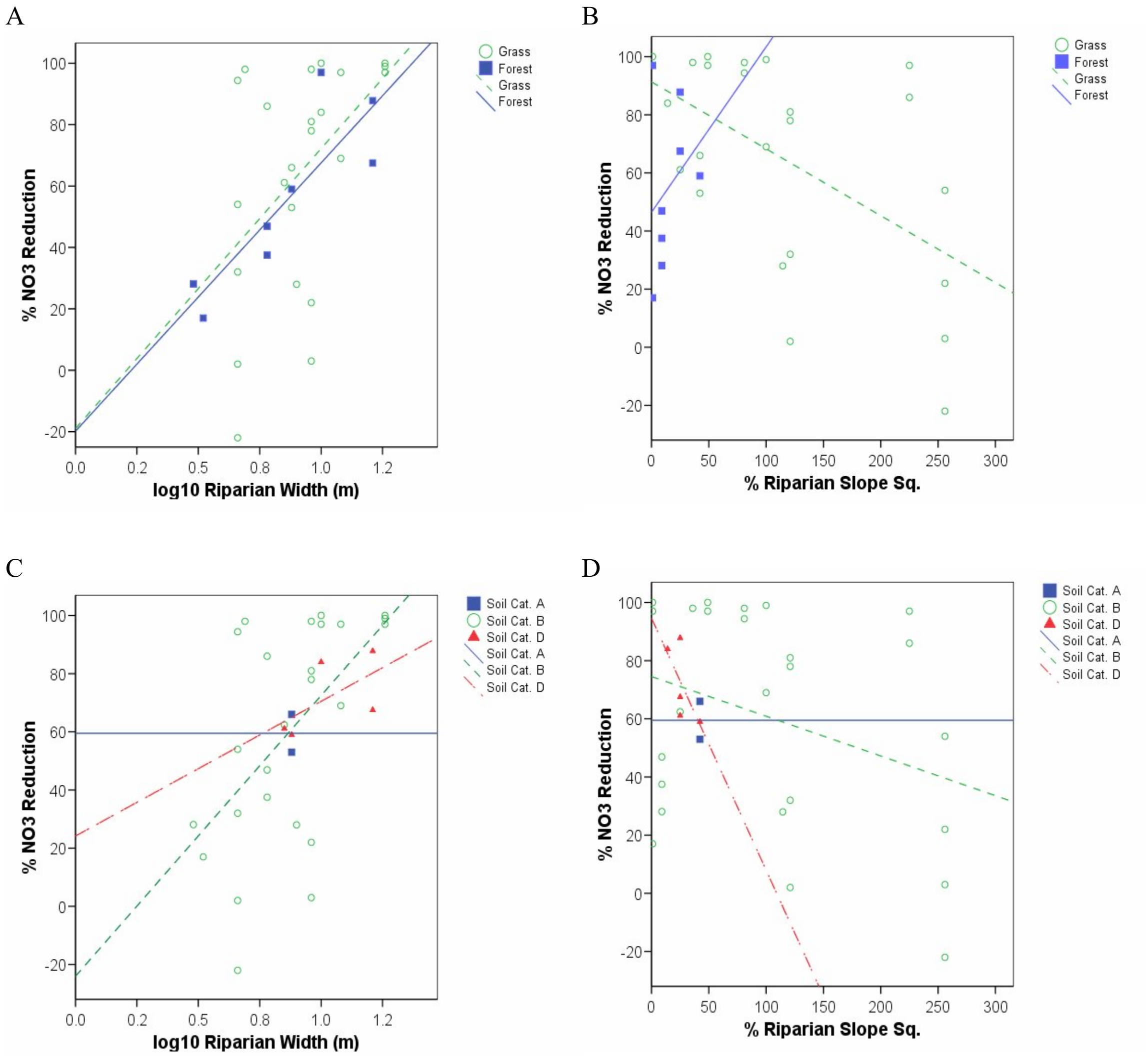


Figure 2. Percent nitrate reduction in riparian buffers with grass and forest vegetation as separate functions of log10 riparian width (2A) and percent riparian slope squared (2B). Percent nitrate reduction in riparian buffers with soil categories A, B, and D as separate functions of log10 riparian width (2C) and percent riparian slope squared (2D).

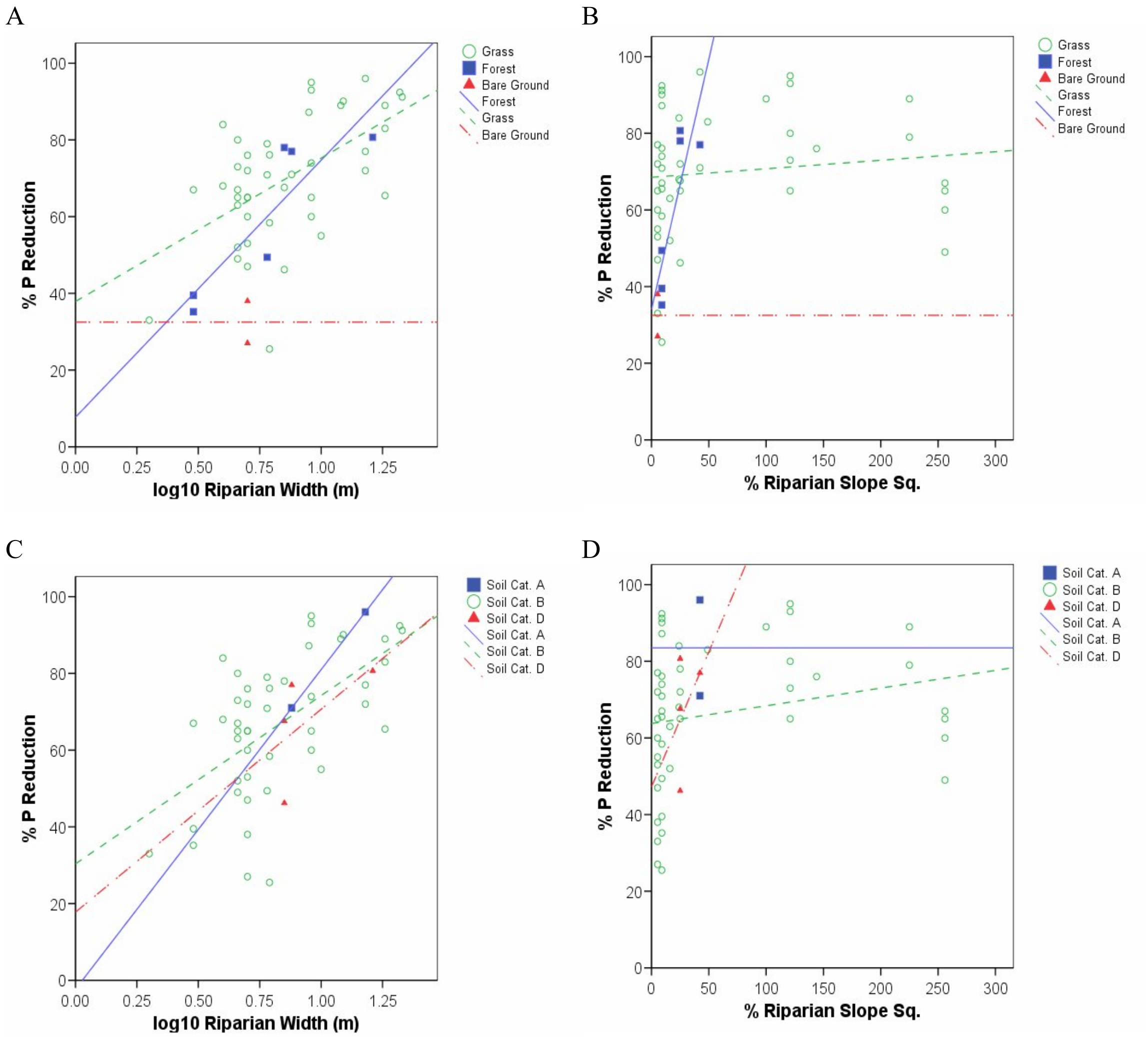


Figure 3. Percent phosphorus reduction in riparian buffers with grass and forest vegetation as separate functions of log10 riparian width (3A) and percent riparian slope squared (3B). Percent phosphorus reduction in riparian buffers with soil categories A, B, and D as separate functions of log10 riparian width (3C) and percent riparian slope squared (3D).

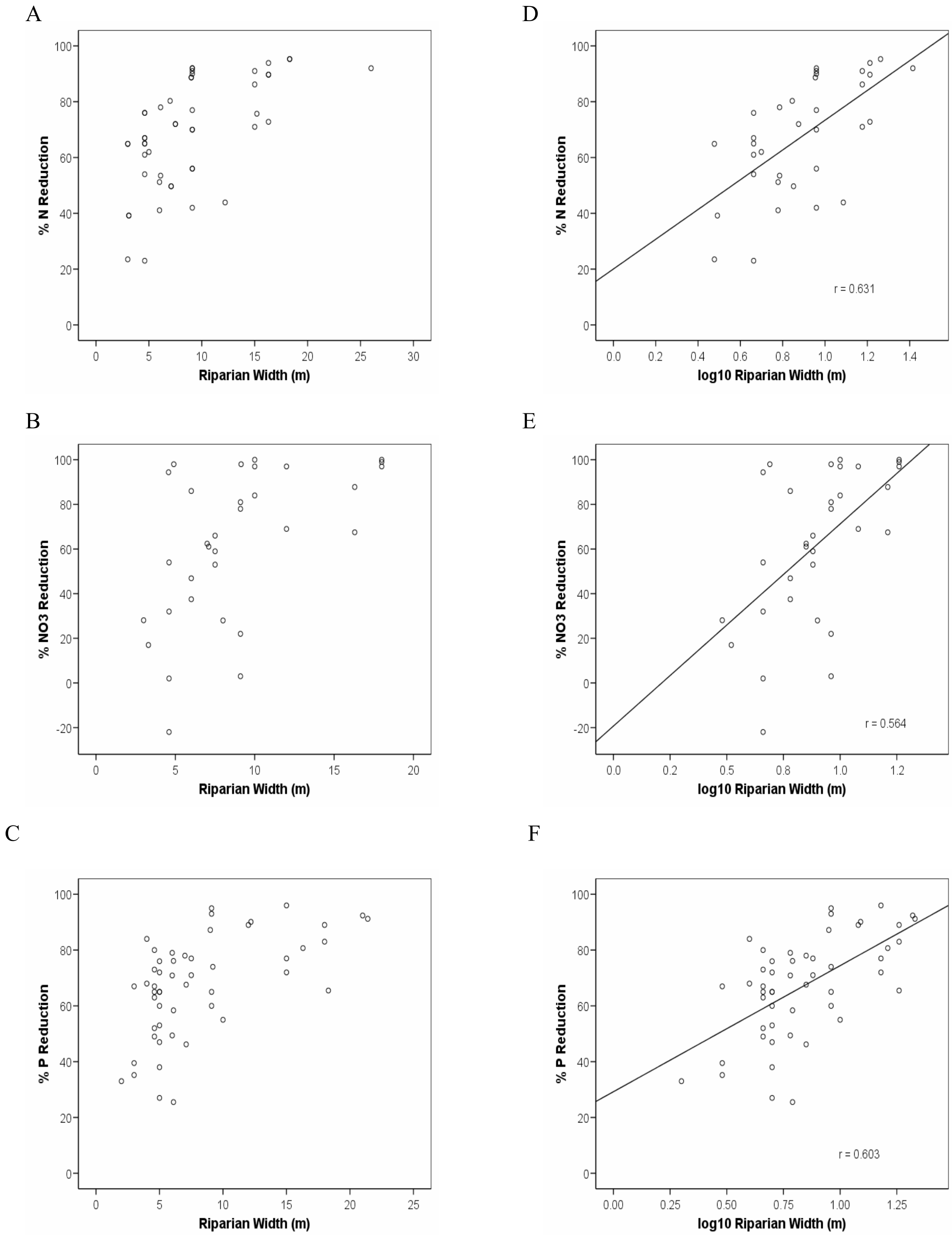


Figure 4. Percent reduction of nitrogen (4A), nitrate (4B), and phosphorus (4C) as a function of riparian width. Figures 4D, 4E, and 4F are corresponding functions using log10 transformed riparian width.

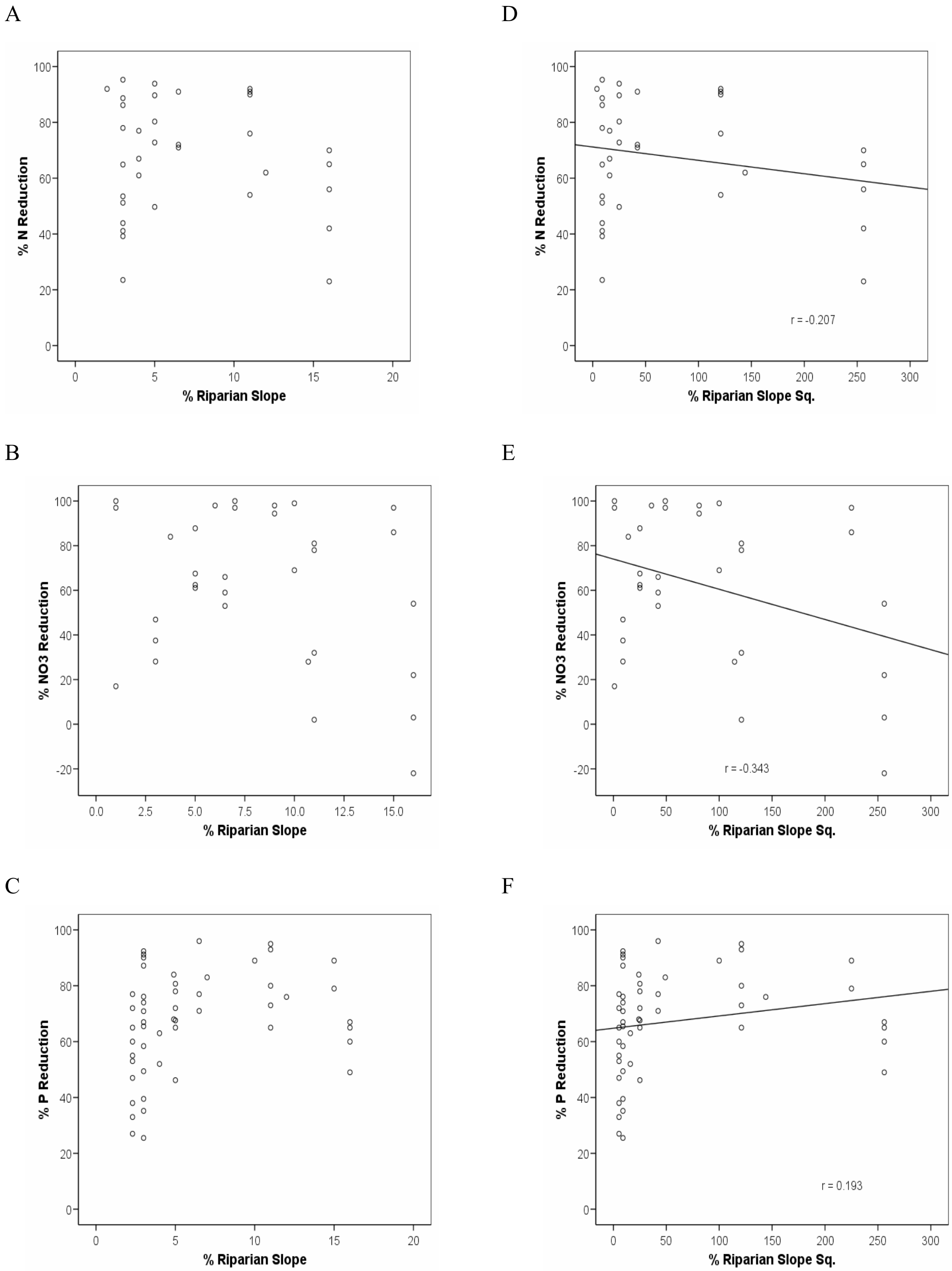


Figure 5. Percent reduction of nitrogen (5A), nitrate (5B), and phosphorus (5C) as a function of riparian average slope. Figures 5D, 5E, and 5F are corresponding functions using riparian slope squared (quadratic transformation).

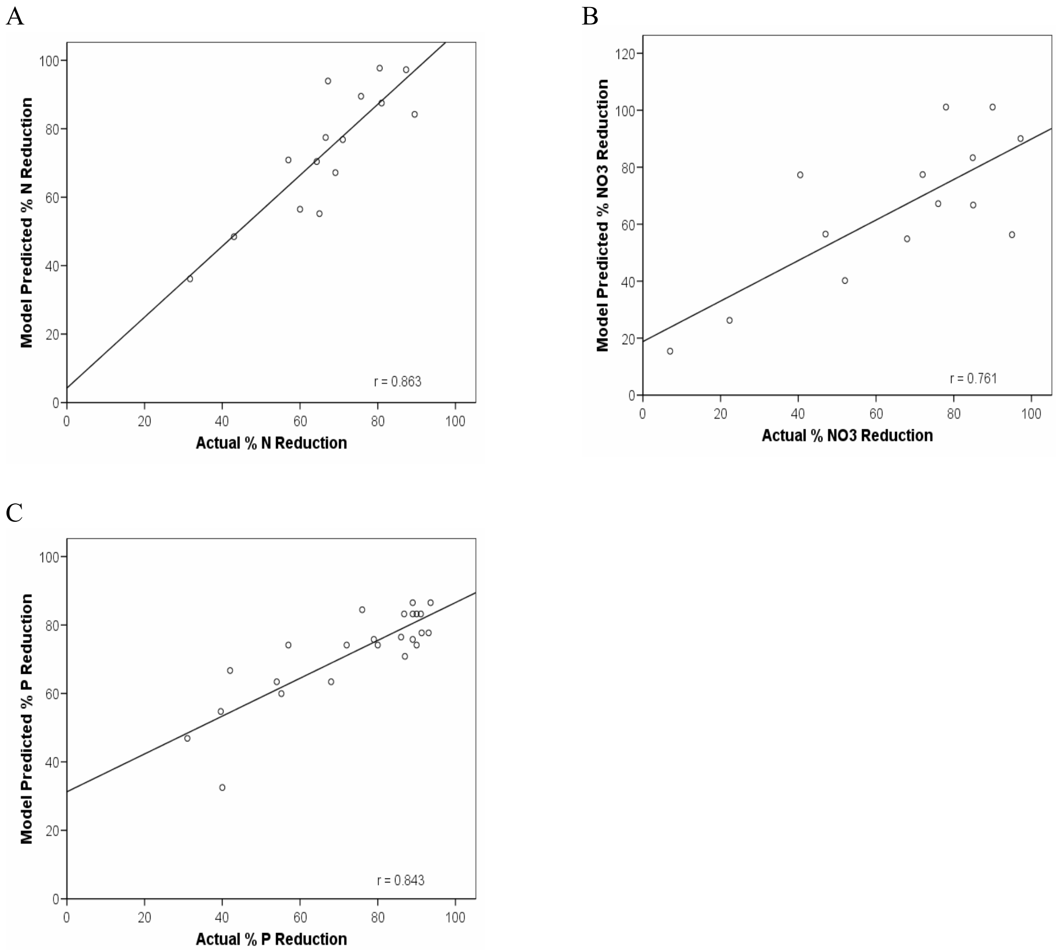
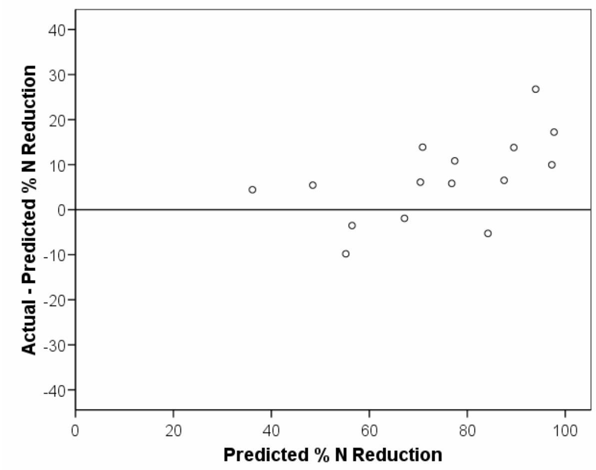
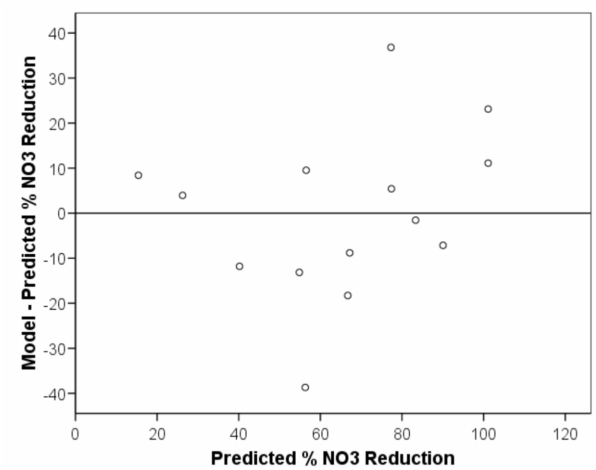


Figure 6. RNAM Predicted percent nutrient reduction versus actual percent nutrient reduction for nitrogen (6A), nitrate (6B), and phosphorus (6C).

A



B



C

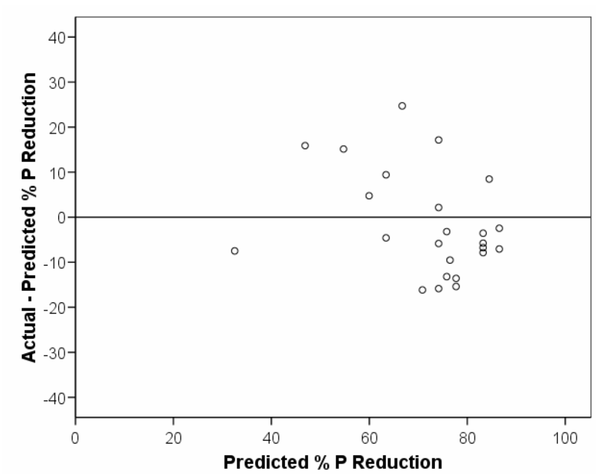


Figure 7. Residual plots indicating differences in actual and RNAM-predicted nitrogen (7A), nitrate (7B), and phosphorus (7C) reduction values.

APPENDIX B

NUTRIENT RETENTION DATA SOURCES

							% Retained			
Author/ (Year)	Location	Precipitation Type/ (Land Use)	Veg.	Soil	% Slope	Width (m)	N	P	NO_3	PO_4
Abu-Zreig et al. (2003)	Canada	simulated (bare field)	grass	silt loam	2.3	2		31		
			grass	silt loam	2.3	2		33		
			grass	silt loam	2.3	5		47		
			grass	silt loam	2.3	5		54		
			grass	silt loam	2.3	5		60		
			grass	silt loam	2.3	5		53		
			grass	silt loam	2.3	10		79		
			grass	silt loam	2.3	10		55		
			grass	silt loam	2.3	15		77		
			grass	silt loam	2.3	15		89		
			grass	silt loam	2.3	15		72		
			grass	silt loam	5	5		68		
			grass	silt loam	5	5		72		
			grass	silt loam	5	5		65		
			grass	silt loam	2.3	5		65		
			n/a*	silt loam	2.3	5		40		
			n/a	silt loam	2.3	5		27		
			n/a	silt loam	2.3	5		38		
Bedard-Haughn et al. (2004)	California, US	simulated (pasture)	grass	loam	10.7	8			28	
			grass	loam	10.7	16			42	
Blanco-Canqui et al. (2004)	Missouri, US	simulated	grass	silt loam	4.9	0.7	55	36	27	37
			grass	silt loam	4.9	0.7	67	53	68	54
			grass	silt loam	4.9	4		68		62
			grass	silt loam	4.9	4		84		72
Chaubey et al. (1994)	Arkansas, US	simulated	grass	silt loam	3	3	64.9	67		
			grass	silt loam	3	6	69.1	70.9		
			grass	silt loam	3	9	88.7	87.2		
			grass	silt loam	3	15	86.2	91.1		
			grass	silt loam	3	21	87.3	92.4		
			grass	silt loam	3	3.1	39.2	39.6		
Chaubey et al. (1995)	Arkansas, US	simulated	grass	silt loam	3	3.1	39.2	39.6		
			grass	silt loam	3	6.1	53.5	58.4		
			grass	silt loam	3	9.2	66.6	74		

			grass	silt loam	3	15.2	75.7	86.8		
			grass	silt loam	3	21.4	80.5	91.2		
Cole et al. (1997)	Oklahoma, US	simulated	grass	silt loam	6	4.9			98	
Corely et al. (1999)	Colorado, US	simulated (forest)	grass	clay loam	3.75	10			84	79
Dillaha et al. (1988)	Virginia, US	simulated (bare field)	grass	silt loam	4	9.1	77	80		
			grass	silt loam	4	4.6	61	63		
			grass	silt loam	4	9.1	71	57		
			grass	silt loam	4	4.6	67	52		
Dillaha et al. (1989)	Virginia, US	simulated (crop field)	grass	silt loam	11	9.1	92	95	81	
			grass	silt loam	11	9.1	90	90	76	
			grass	silt loam	11	9.1	91	93	78	
			grass	silt loam	11	4.6	76	80	32	
			grass	silt loam	11	4.6	54	65	-13	
			grass	silt loam	11	4.6	65	73	2	
			grass	silt loam	16	9.1	70	72	52	
			grass	silt loam	16	9.1	42	60	3	
			grass	silt loam	16	9.1	56	65	22	
			grass	silt loam	16	4.6	65	67	54	
			grass	silt loam	16	4.6	23	35	-22	
			grass	silt loam	16	4.6	43	49	7	
Eghball et al. 2000	Iowa, US	simulated (crop field)	grass	silt loam	12	0.75		40		
Fogle et al. (1994)	Kentucky, US	simulated (crop field)	grass	silt loam	9	4.57			94.4	89.1
			grass	silt loam	9	9.14			98	98.1
			grass	silt loam	9	13.72			97.2	97.3
Lee et al. (1999)	Iowa, US	simulated (pasture)	forest mix	loam	3	6	51.2	55.2	46.9	46
			forest mix	loam	3	6	41.1	49.4	37.5	39.4
			forest mix	loam	3	3	31.7	39.5	28.1	38.1
			forest mix	loam	3	3	23.5	35.2	22.3	29.8
Lee et al. (2000)	Iowa, US	simulated (crop field)	grass	silty clay loam	5	7.1	64.3	67.6	61.1	43.7
			forest mix	silty clay loam	5	16.3	89.7	93.1	87.8	85.3
			grass	silty clay loam	5	7.1	49.7	46.2	40.5	27.6
			forest mix	silty clay loam	5	16.3	72.8	80.7	67.5	34.7
Lee et al. (2003)	Iowa, US	natural (crop field)	grass	loam	5	7	80.3	78	62.4	57.5
			forest mix	loam	5	16.3	93.9	91.3	84.9	79.8
Lim et al. (1998)	Kentucky, US	simulated (pasture)	grass	silt loam	3	6.1	78	76.1		74.5
			grass	silt loam	3	12.2	89.5	90.1		87.8
			grass	silt loam	3	18.3	95.3	93.6		93
Patty et al. (1997)	France	natural (crop field)	grass	silt loam	10	6		42	47	

			grass	silt loam	10	12		22	69	
			grass	silt loam	10	18		89	99	
			grass	silt loam	7	6		0	85	
			grass	silt loam	7	12		46	97	
			grass	silt loam	7	18		83	100	
			grass	silt loam	15	6		79	86	
			grass	silt loam	15	12		89	95	
			grass	silt loam	15	18		89	97	
Payer and Weil (1987)	Maryland, US	simulated	grass	silt loam	2	30		72		
			grass	silt loam	2	30		64		
			grass	silt loam	2	30		62		
Sanderson et al. (2001)	Texas, US	simulated	grass	sandy loam	1	16.4		76		
Schmitt et al. (1999)	Nebraska, US	simulated	forest mix	silty clay loam	6.5	7.5	60	77	59	57
			forest mix	silty clay loam	6.5	15	71	86	72	68
			grass	silty clay loam	6.5	7.5	72	87	66	67
			grass	silty clay loam	6.5	15	91	96	90	90
			grass	silty clay loam	6.5	7.5	57	71	53	50
			grass	silty clay loam	6.5	15	81	90	78	76
Schoonover et al. (2004)	Illinois, US	natural (crop field)	grass	silt loam	1	10			100	
			forest	silt loam	1	10			97	
			grass	silt loam	1	3.3			68	
			forest	silt loam	1	3.3			17	
Schwer and Clausen (1989)	Vermont, US		grass	sandy loam	2	26	92			
Srivastava et al. (1996)	Arkansas, US	simulated	grass	silt loam	3	6.1	21.4	25.5		
			grass	silt loam	3	12.2	43.9	36		
			grass	silt loam	3	18.3	67.2	65.5		
Syversen (2002)	Norway	natural (crop field)	grass	silty loam	12	5	62	76		
			grass	silty loam	12	10	81	89		