2013

INFLUENCE OF RIPARIAN BUFFER MANAGEMENT STRATEGIES ON SOIL PROPERTIES

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INFLUENCE OF RIPARIAN BUFFER MANAGEMENT STRATEGIES ON SOIL PROPERTIES

DISSERTATION

A dissertation submitted in partial fulfillment of the
requirements for the degree of Doctor of Philosophy in the
College of Agriculture
at the University of Kentucky

By
Amanda Abnee Gumbert

Lexington, Kentucky

Director: Dr. Mark S. Coyne, Professor of Plant and Soil Science

Lexington, Kentucky

2013

ABSTRACT OF DISSERTATION

INFLUENCE OF RIPARIAN BUFFER MANAGEMENT STRATEGIES ON SOIL PROPERTIES

The Kentucky Division of Water indicates that agriculture is responsible for 55% of the Commonwealth's assessed streams not supporting their designated uses. Riparian buffers reduce nonpoint source pollution in agroecosystems by storing and cycling nutrients, stabilizing streambanks, increasing infiltration, and storing water. Specific information regarding riparian buffer management is needed for land managers to maximize buffer effectiveness at reducing agricultural contaminants impairing water quality.

Baseline soil properties (texture, pH, C and nutrients) of the riparian buffer surrounding a tributary of Cane Run Creek in Fayette County, KY were characterized prior to imposing three mowing regimes (intense, moderate, and no mow treatments) and one native grass regime. Measurements were made along parallel transects located 2-m and 8-m distances from the stream. Root biomass, aggregate distribution, and saturated hydraulic conductivity were measured along the 2-m transect in two consecutive years following treatment establishment. The 2-m transect soils had the highest C, pH, Ca, Zn, and sand content. The 8-m transect had the highest P, K, Mg, and clay content. Semivariogram analysis of C content indicated slight to moderate spatial dependency along the 2m transect and moderate to strong spatial dependency along the 8m transect. Root biomass increased with decreased mowing frequency at the surface depth after one year; the native grass treatment had significantly less root biomass in both years compared to mowing treatments. There was no significant treatment effect on aggregate size distribution at the surface depth in either year. Mean weight diameter and large macroaggregates decreased from 2011 to 2012. Vegetation treatment had no statistically significant effect on water stable aggregates or saturated hydraulic conductivity. Experimental semivariograms provided evidence of spatial structure at multiple scales in root biomass, aggregates, and soil C. Spatial variability occurred over a shorter lag distance in 2012 than 2011, suggesting an effect of imposed treatments slowly developing over time.

This study provides important insights on riparian buffer soil properties, soil sampling strategies to detect spatial variability in riparian buffers, and length of time needed to assess effects of vegetation management regimes on riparian root biomass, soil aggregates, and hydraulic conductivity.

KEYWORDS: Riparian Zone Management, Spatial Variability, Soil Aggregates, Vegetation Management, Water Quality Protection

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July 22, 2013

INFLUENCE OF RIPARIAN BUFFER MANAGEMENT
STRATEGIES ON SOIL PROPERTIES

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July 22, 2013

To Pappy, who saw no reason why I could not “get that doctor’s degree.”

ACKNOWLEDGMENTS

No individual can accomplish an endeavor such as this without the assistance and support of many. Through these acknowledgments, I will humbly attempt to express my gratitude to those who have surrounded me during this process.

First, my appreciation is extended to Dr. Mark Coyne for accepting me as a student and serving as the Chair of my committee. He provided much scientific guidance, tireless editing, assistance in the field, and impromptu advice, all while maintaining a sense of calm. I would also like to thank my committee members Dr. Carmen Agouridis, Dr. Ole Wendroth, Dr. Chris Barton, and Dr. Steve Workman for their expertise, sound advice, and words of encouragement.

Next, I would like to thank the United States Department of Agriculture Natural Resources Conservation Service for funding this project, Roundstone Seed Company for seed donation, and the Kentucky Department of Fish and Wildlife Resources and Dr. David Ditsch for equipment use.

The following individuals deserve recognition and a great deal of thanks for their assistance in the field and the laboratory: Ann Freytag, Donnie Stamper, Russ Turpin, Brett May, Amber Goff, Stephanie Zapata, Wilson Markle, Tyler Sanderson, Casey Muncy, KD Eaton, Kenton Sena, Sam Lee, Kristi Meier, Heather Jordan, Martin Vandeviere, and Riley Hunter. I am appreciative of the Plant and Soil Sciences faculty for encouragement, as well as answers to my random yet sometimes frantic questions. Thanks also go to Ed Hutchens and Lee Rechtin for repairing broken equipment, Sarah Wightman and Stephanie Mehlhope for map making, and Kristen McQuerry for statistical assistance.

In addition, my Extension supervisors and colleagues are owed gratitude for their support and patience during the six years of simultaneous work and school. Doug McLaren, Ashley Osborne, and Stephanie Jenkins – thank you for your endless encouragement; you have kept me grounded. And a special thank you to Dr. Steve Higgins, who set the bar high, challenged me beyond measure, never doubted my abilities, and provided an endless supply of peanut M&Ms along the way.

My friends and family deserve special recognition for their support and patience during this process. I would like to thank my parents for teaching me the simple value of always doing my best, and to keep going even when things get tough. To Danielle Dove, Jennifer Hunter, and Candace Gilbert – you girls are simply the best. To Anne Anderson (Nan) – you are an exquisite editor.

And finally, heartfelt gratitude goes to my husband, Mark, for his support, patience, encouragement, and reminders to enjoy life along the way. I hope I can do more of that now.

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Riparian Buffers in the Agricultural Landscape: An Overview

Background

Kentucky has over 90,000 miles (144,841 km) of streams and rivers, much of which intersect Kentucky's 14 million acres (5,665,600 ha) of farmland (KEEC, 2010; USDA-NASS, 2011). Production agriculture is an integral component of Kentucky's economy; agricultural cash receipts totaled more than $4.4 billion in 2010 (USDA-NASS, 2011). However, agricultural practices are ranked as the number one cause of the state's impaired streams not supporting their designated uses (KEEC, 2010).

Sediment, pathogens, and nutrients are components of nonpoint source (NPS) pollution and have been associated with crop and livestock enterprises (KEEC, 2010; USEPA, 2009; Wilcock, 2008). Streamside grazing and unrestricted livestock access to streams can result in significant pollutant loads to streams over time (Sharpley and West, 2008). The demand for global food supplies, accompanied by rising commodity prices, will likely result in an increase in crop production acres and an accompanying increased likelihood of NPS pollution. Furthermore, agriculture has been identified as a major source of excessive nutrients in the Mississippi River basin, contributing to hypoxic conditions in the Gulf of Mexico (USEPA, 2011). These issues emphasize the need for strategies to reduce agricultural NPS.

In 1994 the Kentucky General Assembly passed the Kentucky Agriculture Water Quality Act (KAWQA) to provide guidance to agriculture and silviculture industries for pollution prevention and protection of the waters of the Commonwealth through the use of Best Management Practices (BMPs) (KEPPC, 2008). The KAWQA includes BMPs in the areas of silviculture, livestock, crops, pesticides and fertilizers, farmstead, and streams and other water bodies. Within these areas are recommended conservation practices such as riparian area protection, filter strips, and grassed waterways, yet specific implementation and management strategies are vague. Aboveground vegetation and root systems of riparian areas can intercept sediment, pathogens, and nutrients before they reach a water body; nevertheless, specific guidelines for riparian vegetation maintenance to maximize pollutant reduction remain unclear. Land managers need information to best utilize these conservation practices that will lead to reduced agricultural NPS pollution.

Riparian Buffers Defined

Riparian buffers (interchangeably referred to as riparian zones and riparian areas) are a subset of conservation buffers that broadly includes grassed waterways, filter strips, and vegetative barriers. Riparian buffers are three-dimensional zones of direct interaction between terrestrial and aquatic ecosystems (Gregory et al., 1991) and separate upland land uses from an intermittent or permanent water body, including areas associated with groundwater recharge. Subject to disturbances from upland and fluvial processes because of their prominent transition location, riparian buffers can be characterized as diverse plant communities that encompass ground and surface water interactions in soils subjected to fluctuating water levels. These buffers play a key role in landscapes by providing ecosystem services such as NPS pollution control, water storage and flood reduction, nutrient storage and cycling, streambank stability, and wildlife habitat (English et al., 2004). Accomplishing any of the listed riparian functions depends on the type of vegetation, buffer dimensions, and maintenance (USDA-NRCS, 2005).

Vegetation

Plant communities enhance riparian buffers by regulating water temperature, reducing erosive forces on stream banks, and contributing carbon to the ecosystem. Vegetated streamside buffers provide foliage and stems that increase surface roughness, as well as a dense network of roots that bind riparian substrates for increased streambank resistance to erosion, with root exudates playing a role in soil cohesion (Kiley and Schneider, 2005; Wynn et al., 2004). Additionally, riparian plants take up nutrients and contribute soil organic C needed to promote denitrification, two processes important in reducing nutrient losses to adjacent water bodies.

Riparian buffer vegetation may be naturally occurring or planted by a land manager. Riparian vegetation varies widely, may consist of woody (trees and shrubs) or herbaceous (grasses and forbs) plants, and can be manipulated easily through selection and management. Woody plants have larger, taller stems than herbaceous plants, and woody litter tends to decompose more slowly than herbaceous litter, with large debris from woody plants taking decades or more to decompose.

Trees and shrubs provide perennial root systems that stabilize stream banks and provide long-term nutrient storage while herbaceous, warm-season grasses have a high stem density and an annual root system that provides large amounts of organic matter to the soil (Schultz et al., 1997). Kiley and Schneider (2005) found a significant positive correlation between root biomass in the top 30-cm of soil and herbaceous plant density, and Wynn et al. (2004) found that herbaceous buffers had greater root length density than forested buffers.

Switchgrass (*Panicum virgatum*) is a native warm-season grass often recommended for the grass zone in riparian buffers. Switchgrass has dense, stiff stems that slow surface runoff and promote infiltration prior to the runoff reaching a stream. Cool-season grasses such as fescue (*Festuca arundinacea*) are not recommended for riparian buffers that experience overland flow because their stems do not remain upright under surface runoff; in addition, they produce eight times less root mass than native grasses (Schultz et al., 1997). Other research (Lowrance et al., 2002), however, has shown that cool-season grass filters have twice as much carbon in the upper 20 inches of soil compared to switchgrass, with corresponding higher rates of denitrification. Additionally, Self-Davis et al. (2003) found fescue to have greater infiltration and lower runoff compared to native warm-season grasses. This may indicate that native warm-season grasses are more effective at slowing overland flow, while cool-season grass filters might be more effective at below-ground processes.

Native grass strips increase infiltration rates and microbial activity and might be more effective at providing deeper soil carbon contents over longer periods than cool-season grasses such as fescue (Lowrance et al., 2002; Schultz et al., 1997). Compared to alfalfa (*Medicago sativa*) and smooth brome (*Bromus inermis*), switchgrass produced the highest level of root surface area after three growing seasons (Kelly et al., 2007), with root surface area remaining constant during May-August, and increasing significantly in September during one of those growing seasons.

Dimensions and Spatial Considerations

Typical design recommendations for conservation buffers such as filter strips consist of a uniform buffer width along a field margin to capture uniform surface runoff (Dosskey et al., 2011). Riparian buffer width plays a role in determining specific benefits. For example, wide buffers (> 160 feet) tend to be more efficient in removing nitrogen from water, whereas narrow buffers (<160 feet) may not remove significant amounts of nitrogen (Mayer, 2005). There are conflicting recommendations for buffer width. Schultz et al. (1997) recommend buffers of at least 66 feet, while other research has shown that the most effective buffers are at least 100 feet wide and composed of native forest (Wenger and Fowler, 2000).

The United States Department of Agriculture Natural Resources Conservation Service (USDA-NRCS) promotes the use of three zones when establishing and maintaining riparian buffers, with zone width determined by stream order. Stream order is a hierarchical system for classifying streams in which the smallest channels (which may only carry wet-weather flows and be otherwise dry) are designated first order; a second order stream is formed by the junction of two first order streams, a third order stream is formed by two second order streams, and so on (Strahler, 1952). For first through third order streams, Zone 1 is an undisturbed forested (80% hard mast trees) area that begins at the top of the stream bank and continues up gradient for about 15 feet. Zone 2 begins at the ending edge of Zone 1, and continues away from the stream for a width of 35-165 feet, with a wider Zone 2 as stream order increases. Zone 2 is often a managed forest area. The widths of Zones 1 and 2 are measured horizontally on a line perpendicular to the water body. Zone 3 is considered the runoff control zone, providing the primary sediment retention function for the buffer. Zone 3 is approximately 20 feet wide and is managed in permanent grass/legume/forb cover by mowing or rotational livestock grazing (USDA-NRCS, 2005). For first through third order streams, the USDA-NRCS prescription for riparian buffers recommends a minimum of 70 feet between the water body and the adjacent upland land use.

Spatial variability in the landscape needs consideration when prescribing conservation buffers, specifically in riparian areas where disturbances from upland and fluvial processes may influence soil properties. Characterizing soil properties along riparian areas may aid in prescribing appropriate conservation practices at appropriate scales. Variable-width buffers would place wider buffers in field locations where concentrated runoff occurs, providing the opportunity for enhanced treatment of the runoff load (Dosskey et al., 2005).

Geographical patchiness of natural ecological phenomena as reported by Legendre (1993), the concept that nature is neither uniformly nor randomly distributed, is likely applicable to riparian areas. Elements in nature closer to one another may tend to present a greater degree of likeness than those farther apart (Fernandes et al., 2011). Furthermore, spatial autocorrelation suggests that elements closer together in an ecosystem tend to be influenced by the same processes, unlike those farther away. The spatial component can be described by a mathematical function such as a semivariogram, which allows for the study of autocorrelation as a function of distance. Predicting values of a variable from known values at other sampling points is possible if the spatial positions are known and the values are autocorrelated (Legendre and Fortin, 1989).

Maintenance

Restoring and managing vegetation in riparian buffers is widely recommended to protect water resources in agricultural settings (English et al., 2004; NRC, 2002; Schultz et al., 1997; USDA-NRCS, 2000). However, riparian buffer maintenance is a multi-faceted issue, with respect to aesthetics, loss of agriculturally productive land, colonization by invasive species, and buffer function in terms of regulating nutrient and contaminant flow from adjacent lands. While some programs recommend mowing buffers for weed control during initial establishment (Schultz et al., 1997), others indicate that mowing and periodic burning can reduce infiltration rates over a longer period (Schacht et al., 1996). Periodic removal of above-ground biomass has been recommended to maximize nutrient uptake from the soil and runoff water (Kelly et al., 2007; Schultz et al., 1997). Hefting et al. (2005) found that mowing herbaceous sites in Europe removed up to 93% of nitrogen taken up each year by grasses; the addition of a fast-growing woody species (e.g. cottonwood) to herbaceous buffer systems can reduce phosphorus in riparian buffer soils when periodically harvested (Kelly et al., 2007).

Little research is available on the effects of mowing riparian buffers on soil physical, chemical, or biological properties, although studies have shown varying results of mowing effects on below-ground biomass in non-riparian landscape positions (Dickinson and Polwart, 1982; Kitchen et al., 2009; Todd et al., 1992). Mowing and imposed compacting forces (e.g. equipment used in the moist conditions of riparian buffers) have been shown to significantly increase bulk density (Carrow, 1980), especially in wheel tracks (Flannagan and Bartlett, 1961) in turf grass studies. Other research has focused on groundwater nitrate removal in mowed riparian zones (Addy et al., 1999), although the study area was not representative of large contiguous mowed areas.

Kentucky landowners employ various strategies in riparian buffer maintenance. Some landowners regularly mow riparian buffers to achieve a tidy appearance, while others view riparian buffer mowing as a burdensome task that falls behind income-producing activities on the priority list. Still others may manage riparian areas as part of pastures, allowing livestock access for grazing. Aesthetic characteristics of buffers should be considered in addition to their effectiveness at providing ecosystem services. Riparian buffers impact the visual appearance of agricultural landscapes by contrasting with row crops and livestock pastures (Lovell and Sullivan, 2006). Naturalized riparian buffers that have an unkempt appearance are not widely accepted in agricultural settings. Farmers and rural landowners often associate the clean or neat appearance of the farm as a measure of the farmer's work ethic (Lovell and Sullivan, 2006).

Some landowners express concern about taking land out of production, especially during times of high-value commodities, even though studies have shown conservation buffers are cost-effective in terms of environmental benefits returned for investment (Helmers et al., 2006; Lovell and Sullivan, 2006). Still others may participate in government-sponsored conservation programs that allow seasonal grazing or harvesting of buffer vegetation. The KAWQA recommends restricting livestock access to streams and maintaining grass on these restricted areas, but no specific maintenance recommendations are provided (KEPPC, 2008). Furthermore, following the USDA-NRCS riparian buffer prescription could take much land out of production, when considering the 70-ft (21.3-m) minimum buffer along water bodies.

Cost-share opportunities exist to create or enhance conservation buffers on farms;   costs for establishing and maintaining buffers influence the willingness of landowners to implement these practices (Lovell and Sullivan, 2006). The Conservation Reserve Program (CRP), the Conservation Reserve Enhancement Program (CREP), Conservation Security Program, and the Environmental Quality Incentives Program (EQIP) are examples of federal conservation programs that pay producers either through cost-share dollars or annual rental payments to convert highly erodible and environmentally-sensitive land into riparian areas. These programs are administered by the Farm Service Agency (FSA) and the NRCS (Buckloh et al., 2004). State and local cost-share programs are also available in Kentucky through local conservation districts and the Kentucky Soil and Water Conservation Commission. The National Conservation Buffer Initiative, launched in 1997 under the leadership of the USDA-NRCS, had a goal of 2 million miles (3.2 million km) on private land by 2002, and an economic evaluation of implementing the initiative showed buffer programs to be cost-effective in terms of economic and environmental benefits (Helmers et al., 2006). Dosskey et al. (2005)

suggest that precision conservation (e.g. site-specific filter strips) may cost more to implement but has a greater water quality benefit.

Riparian Buffer Function and Assessment

Soil water passes through riparian buffers before entering streams, and riparian vegetation may significantly modify the amount of dissolved nutrients entering streams by plant uptake (Gregory et al., 1991). Conservation buffers have been shown to increase infiltration rates (Bharati et al., 2002), remove sediment and nutrients from surface runoff (Lowrance et al., 2002), and increase soil organic matter. Efficacy of riparian buffers may also be tied to soil quality.

Assessing overall conservation buffer effectiveness can be complex and *in situ* data on specific ecosystem services provided by riparian buffers is lacking (Jones et al., 2010). Measuring indicators of ecosystem services (e.g. water and nutrient cycling, soil stability) along riparian landscape gradients by identifying specific soil and vegetative properties that measure or indicate desirable buffer behaviors is important. Lowrance et al. (2002) suggest that a buffer's ability to reduce NPS pollutants could be indirectly assessed over time by measuring aggregate structure, infiltration rate, soil carbon and microbial biomass, and denitrification rates.

Soil Aggregates

Soil particles are bound together by soil organic matter into aggregates. Soil aggregate formation is influenced by biotic (e.g. microbial activity, root exudates, etc.) and abiotic factors (e.g. clay content), and aggregates can be classified into the following size orders: clay micro-structures (<2 µm diameter); microaggregates (2-250 µm diameter); and macroaggregates (>250 µm diameter) (Carter, 2004). The aggregate hierarchy concept proposed by Tisdall and Oades (1982) suggested that aggregates form in sequence such that primary particles and silt-sized aggregates are bound together into stable microaggregates (20-250 µm) which in turn are bound together into macroaggregates (>250 µm). This concept further suggests that microaggregates are held together by persistent binding agents while macroaggregates are held together by more transient binding agents (i.e. root exudates), leaving macroaggregates subject to stability or vulnerable to destruction as a result of agricultural management techniques that affect root development.

Aggregates are significant features of soil structure (Babel et al., 1995), and their size and distribution determine the pore space geometry of the soil matrix. Subsequently, soil aggregates highly influence hydraulic conductivity as water movement through soil is primarily a function of pore size and distribution (Ehlers et al., 1995).

Infiltration and Hydraulic Conductivity

Infiltration is the entry of water into the soil surface, and maximizing infiltration of runoff water is expected to decrease the export of soluble pollutants (Helmers et al., 2006) by allowing sediments to deposit on the ground surface prior to reaching surface waters. Soils with a high infiltration capacity are likely to have a greater sediment trapping capability in addition to reducing the release of soluble pollutants to nearby waterways. Hydraulic conductivity is the measure of a soil’s ability to transmit water through pores (Klute and Dirksen, 1986) and is directly related to pore geometry and organization. The saturated hydraulic conductivity (K_{sat}) describes the maximum capacity of soils to conduct water and can be calculated from infiltration flow rates measured in the field based on Wooding's (1968) work. The hydraulic conductivity of soils is heavily influenced by soil texture and structure, with sandy soils generally having higher saturated conductivities than finer-textured soils. In a study sizing filter strips for effectiveness, Dosskey et al. (2011) reported a fine sandy loam soil able to trap nearly 100% of sediment in runoff while a silty clay loam soil was only able to trap 35% of runoff sediment.

Vegetated buffers improve soil quality in the riparian zone and may be effective in reducing NPS pollution in agroecosystems by increasing infiltration (Bharati et al., 2002) as plant stems slow overland flow. Vegetation type can influence infiltration of surface runoff by variations in stem density, amount of litter, and, subsequently, degree of roughness, yet Dosskey et al. (2010) report differing evidence from the literature as to the superiority of forested buffers versus grass buffers to increase soil porosity. Trees generally grow larger and more widely spaced than herbaceous plants; woody plants generally produce larger roots that decompose more slowly than the smaller, yet more numerous, shorter-lived roots of herbaceous vegetation. Soil permeability is increased by root growth and decay and burrowing by macroinvertebrates grazing on roots and litter (Dosskey et al., 2010), thus creating large pores that enhance water movement. Soils in naturalized areas have shown increased water infiltration rates as a result of accumulation and on-site decomposition of leaves and associated increases in earthworm and macroarthropod activity (Millward et al., 2011).

Roots

Information regarding the impact of aboveground management on root systems of riparian vegetation is limited. Researchers have reported variability in root density and root biomass in herbaceous versus forested riparian areas (Piercy and Wynn, 2008; Wynn et al., 2004) and in a multi-species agricultural riparian buffer (Tufekcioglu et al., 1998), but these studies did not focus on specific vegetation management techniques. Studies investigating the effects of aboveground vegetation removal on root systems (whether by mowing, grazing, or burning) have been predominantly focused in upland prairie or pasture systems (Johnson and Matchett, 2001; Kitchen et al., 2009; Todd et al., 1992), or simulated field conditions (Neigebauer et al., 2000) and report mixed results. Annual mowing has been reported to have no net effect on total root biomass, although it did significantly increase root biomass in the upper 10-cm compared to unmowed treatments (Kitchen et al., 2009). Todd et al. (1992) reported decreased live root biomass after two years of mowing treatments while others reported an increase in below-ground biomass in the second year of mowing (Dickinson and Polwart, 1982). Neigebauer et al. (2000) investigated the effects of mowing heights on the roots of wildflower (black-eyed Susan [*Rudbeckia hirta* L.]) sod and found that mowing significantly reduced total root biomass, while total depth of rooting increased linearly with mowing height. Chaieb et al. (1996) report that increased cuttings of perennial grasses resulted in a more superficial root system, concentrated in the upper 15-cm of soil, and found little difference in root systems of non-mowed plots and those mowed once. Although Kitchen et al. (2009) reported an increase in root biomass in the upper

10-cm of an annually mowed (with clippings removed) prairie, they reported a decrease in root biomass in a similar prairie that also received a prescribed burn treatment; mowing concentrated roots in the upper 20-cm in both burned and unburned prairie. Multiple studies have reported significant fire and mowing interaction effects on roots (Benning and Seastedt, 1997; Johnson and Matchett, 2001; Kitchen et al., 2009). Grazing studies have shown decreases in root growth with grazing pressure (Johnson and Matchett, 2001), no inhibition of root growth (McNaughton et al., 1998), and both positive and negative root responses to grazing (Milchunas and Lauenroth, 1993). Other grazing studies have shown more below-ground plant material under light pasture grazing (15-25% usage) (Johnston, 1961) and greater soil organic carbon under low grazing pressure (Franzluebbers and Stuedemann, 2010) compared to ungrazed pastures. In contrast, Johnston (1961) showed decreased root production with increased vegetation removal in a companion greenhouse clipping study.

Research Objectives

Specific information on riparian buffer maintenance is needed for agricultural producers to maximize the potential benefits to water quality through the utilization of riparian buffers. The goals of this study are: 1) to characterize baseline soil physical and chemical properties of a riparian buffer in grassland management prior to implementing vegetation management strategies; 2) to explore and assess spatial processes in a riparian buffer; and 3) to evaluate the influence of mowing and vegetation management strategies on root biomass, soil aggregate size distribution and stability, hydraulic conductivity, and soil carbon in this riparian buffer. This research includes exploration of spatial variability within the buffer and how this variability may influence management strategies to maximize buffer efficacy.

The dissertation is organized as the presentation of specific research objectives in two chapters as follows:

1) Spatial Variability of Soil Properties in a Central Kentucky Riparian Buffer
2) Riparian Buffer Management Influences on Roots, Soil Structural Properties, and Hydraulic Conductivity

Chapter Two

Spatial Variability of Soil Properties in a Central Kentucky Riparian Buffer

Introduction

Kentucky has over 6,985 miles (11,241 km) of impaired waters (KEEC, 2010), and nonpoint source (NPS) pollution is the leading source of impairment. The Kentucky Division of Water indicates that agriculture, a form of NPS, is responsible for 55% of assessed streams not supporting their designated uses (KEEC, 2010). NPS pollution is difficult to regulate, but state laws provide guidance to specific activities that may create NPS pollution. The Kentucky Agriculture Water Quality Act (KAWQA) was passed in 1994 to reduce NPS pollution from agriculture and silviculture through the use of Best Management Practices (BMPs) (KEPPC, 2008). Suggested BMPs for agriculture in the KAWQA include riparian zone protection, filter strips, and grassed waterways.

Riparian zones are three-dimensional zones of direct interaction between terrestrial and aquatic ecosystems (Gregory et al., 1991). Because of their prominent transition location, riparian zones are subject to disturbances from upland and fluvial processes. By providing ecosystem services such as control of NPS pollution, water storage and flood reduction, nutrient storage and cycling, streambank stability, and wildlife habitat (English et al., 2004), riparian zones play a key role in landscapes. Soil water passes through riparian zones before entering streams, and riparian vegetation may significantly modify the amount of dissolved nutrients entering streams by plant uptake (Gregory et al., 1991).

Conservation buffers (including riparian buffer zones, filter strips, etc.) can increase infiltration rates (Bharati et al., 2002), remove sediment and nutrients from surface runoff (Lowrance et al., 2002), and increase soil organic matter. Vegetated streamside buffers provide foliage and stems that increase surface roughness, and a dense network of roots that bind riparian substrates, which increases streambank resistance to erosion (Kiley and Schneider, 2005). Vegetated buffers improve soil quality in the riparian zone and may be effective in reducing NPS pollution in agroecosystems by increasing infiltration (Bharati et al., 2002).

Assessing overall conservation buffer effectiveness can be complex; therefore, identifying specific soil and vegetative properties that measure or indicate desirable buffer behaviors is important. One desirable riparian buffer behavior is the capacity to infiltrate runoff water. Maximizing infiltration is expected to decrease the export of soluble pollutants (Helmers et al., 2006). Soils with a high water infiltration capacity are likely to have enhanced sediment trapping capability in addition to reducing the release of soluble pollutants to nearby waterways. The hydraulic conductivity of soils is most directly related to soil texture and structure, with sandy soils generally having higher saturated conductivities than finer-textured soils. In a study sizing filter strips for effectiveness, a filter strip situated on a fine sandy loam soil trapped nearly 100% of sediment in runoff while a filter strip on silty clay loam soil only trapped 35% of runoff sediment, which was attributed to the finer textured soil experiencing less infiltration and thus fewer fine particles deposited in the filter strip (Dosskey et al., 2011). Efficacy of riparian buffers may also be tied to soil quality, with infiltration rate, aggregate structure, and soil carbon serving as indicators of a buffer's ability to reduce nonpoint source pollution (Lowrance et al., 2002).

Spatial variability in the landscape may be important when prescribing conservation buffers, specifically in riparian areas where disturbances from upland and fluvial processes can influence soil properties. Geographical patchiness of natural ecological phenomena as described by Legendre (1993) is likely applicable to riparian areas; nature is neither uniformly nor randomly distributed, but elements closer to one another tend to present a greater degree of likeness (Fernandes et al., 2011; Legendre and Fortin, 1989). Spatial autocorrelation suggests that elements closer together in an ecosystem tend to be influenced by the same processes as opposed to those farther away and can be described by a mathematical function such as a semivariogram. Further, predicting values of a variable from known values at other sampling points is possible if the spatial positions are known and the values are autocorrelated (Legendre and Fortin, 1989). Typical design recommendations for conservation buffers such as filter strips consist of uniform buffer width along a field margin to capture surface runoff, although Dosskey et al. (2011) suggest that variable width buffers may be more effective at intercepting concentrated flows. Understanding the range over which spatial structure exists among riparian buffer soil properties will aid in developing design recommendations for riparian buffers.

Specific information about riparian buffer maintenance is needed for agricultural producers to maximize the potential benefits to water quality from using riparian buffers. The purpose of this study was to characterize baseline soil physical and chemical properties prior to implementing vegetation management strategies in a riparian buffer. This characterization includes exploring spatial processes within the buffer, how these processes are linked to one another, and how spatial variation may influence management strategies to maximize buffer efficacy. While other researchers have employed traditional statistics to describe the spatial distribution of riparian soil properties (Blazejewski et al., 2009; Kang and Lin, 2009), few have utilized semivariogram analysis to describe riparian soil variability, making this study a novel approach to riparian buffer assessment and management.

Methods

Field

The study site is at the University of Kentucky Agriculture Experiment Station, near Lexington, KY (N 38°07'23.98", W 84°29'50.04"). The site is a riparian buffer of an unnamed tributary to the Cane Run Creek. Soils are classified as fine, mixed, active, mesic Fluvaquentic Endoaquolls and mapped as Lanton silty clay loam (dunning) series with fine-textured alluvium parent material derived from limestone (USDA-NRCS, 2011). Sections of the stream were channelized for agricultural purposes in the 1970s and the surrounding buffer was maintained as mowed grassland until 2010 (Calvert, 2011). The stream is not incised and thus can access the floodplain during high flow events. The streambanks display little evidence of erosion and the channel bed material is predominantly limestone bedrock. At the time of sampling, the riparian buffer consisted of mixed grassland vegetation (e.g. fescue [*Festuca arundinaceae*], bluegrass [*Poa pratensis*], broadleaf weeds) mowed every four to six weeks during the growing season.

For this study, parallel transects were established south of the stream at 2-m and 8-m distances from top-of-bank along a 650-m straightened stream section (Figures 2.1a and 2.1b).

One soil core was collected every 10-m (avoiding an improved stream crossing and a channelized drainageway) along each transect in July 2010 using an ATV-mounted hydraulic soil corer with 5-cm diameter (Giddings Machine Company, Windsor, CO). Cores were divided into 10-cm depth increments, up to a maximum sampling depth of 70-cm. For the purposes of this study, sampling locations are numbered 1-40, with location 1 situated at the most downstream location and location 40 at the most upstream location.

Soil Physical and Chemical Properties

Soil texture was determined using the micropipette method (Burt et al., 1993; Miller and Miller, 1987). Soil pH was analyzed in 1M KCl (SPAC, 2000b) and nutrient content (P, K, Ca, Mg, and Zn) was determined using Mehlich III extraction (SPAC, 2000a; SPAC, 2000c). Soil organic carbon was determined via LECO combustion (Nelson and Sommers, 1996) and is reported as % C.

Statistical Analysis

Means of soil chemical and physical properties were compared for differences between transects and among depths using the LSMEANS statement in the GLM procedure in SAS 9.2 (SAS, 2010). Least squares means were used to compare means among depths due to missing data from some sampling locations. Spearman correlation

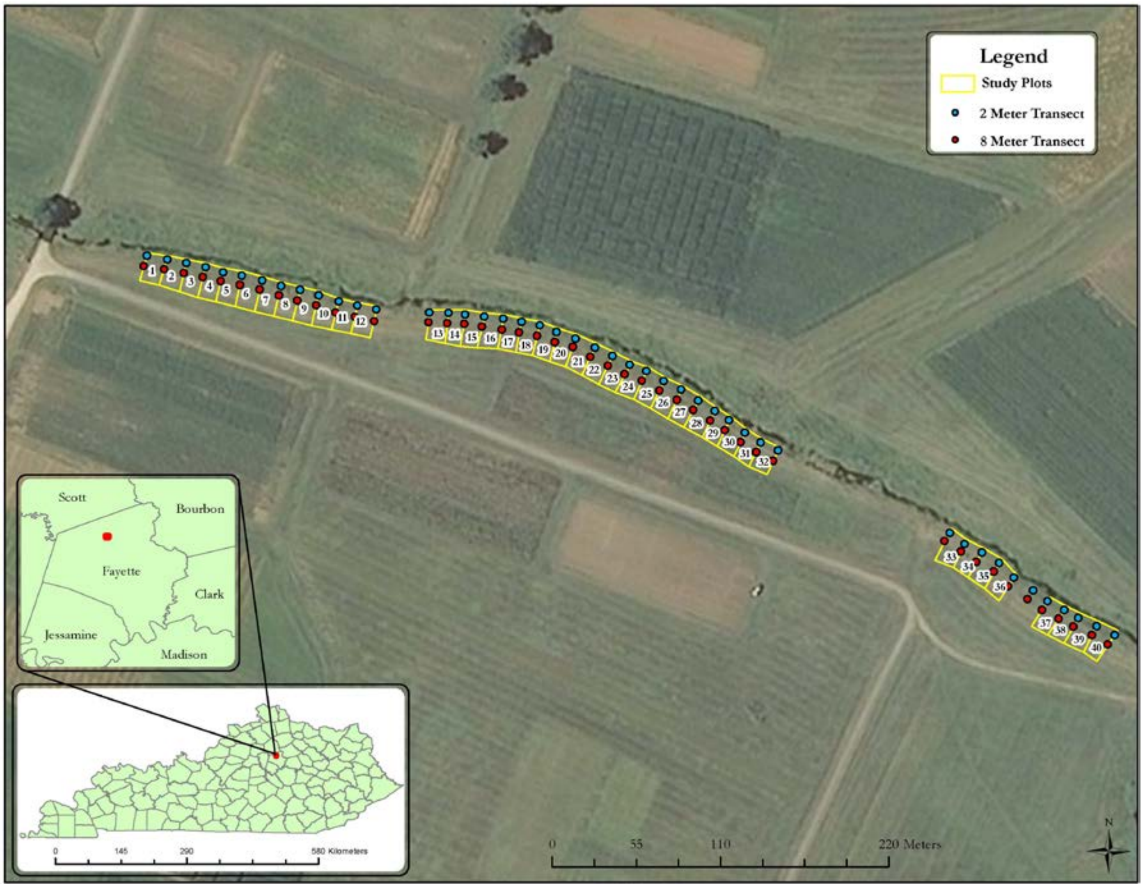


Figure 2.1a. University of Kentucky Agriculture Experiment Station study site in Fayette County, Kentucky (inset). Blue dots indicate transect located 2-m from top-of-bank and red dots indicate transect located 8-m from top-of-bank.

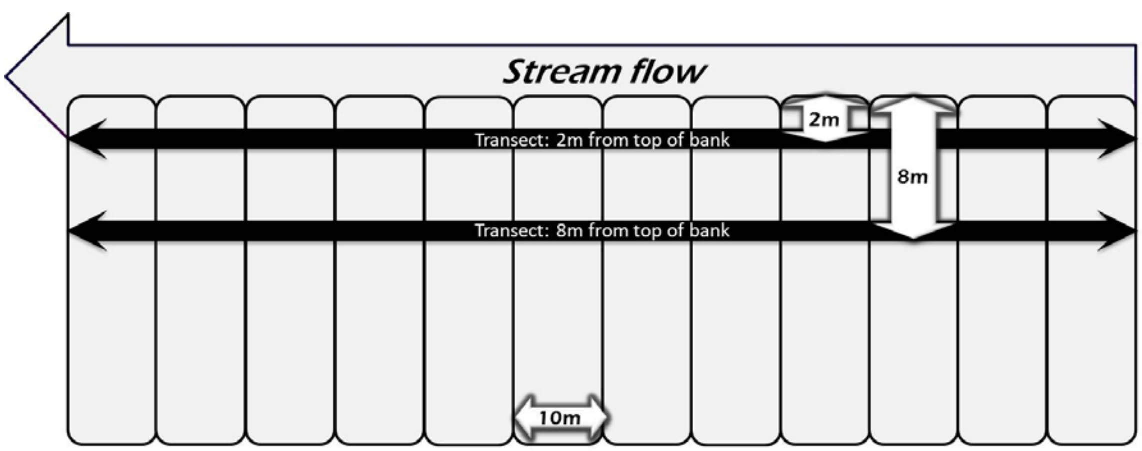


Figure 2.1b. Detail of sampling strategy.

analysis was used to examine relationships between transects and among depths. Analyses were conducted at the $\alpha = 0.05$ significance level.

To identify and describe spatial structure that may be present along the buffer transects, experimental semivariograms were computed using SGeMS software (http://sgems.sourceforge.net/). The solver function in Microsoft Excel was used to fit range, sill, and nugget parameters for semivariogram models. Spherical semivariogram models were fit using the equations

$$\gamma(h) = C_o + C\left[\frac{3h}{2a} - \frac{1}{2}\left(\frac{h}{a}\right)^3\right], \text{when } 0 \leq h \leq a$$

$$\gamma(h) = C_o + C, \text{when } h > a$$

for spherical models, where h=lag distance, C_o=nugget, C=structural component, C_o+C=sill, and a=range (Legendre and Fortin, 1989; Nielsen and Wendroth, 2003).

Results and Discussion

Soils in the 2-m transect fall into the loam, sandy loam, and silt loam textural classes. Soils in the 8-m transect fall into the clay, clay loam, loam, sandy loam, silt loam, and silty clay textural classes. When considering all depths, soils in the 2-m transect sampling locations had greater sand content and lower clay content than soils in the 8-m transect (Table 2.1). Greater sand content in the transect nearer the stream (2-m) may be a result of sediment from flooding, or a remnant of sediments deposited during the channelization of this stream segment. The greater sand content may result in soils along the 2-m transect having higher saturated hydraulic conductivities than the finer-textured soils in the 8-m transect. A closer examination of each sampling depth

Table 2.1. Soil physical properties. Values reported as least squares means; n = number of samples. For the same property different letters indicate significant differences ($P<0.05$). *Transect measurements compared among all samples at 2-m and 8-m, respectively.

Depth	n	% Sand	% Silt	% Clay
2-m transect				
0-10 cm	40	35.40	50.18	14.43
10-20 cm	40	34.86	49.48	15.65
20-30 cm	36	32.52	52.58	14.90
30-40 cm	29	32.99	51.76	15.26
40-50 cm	18	34.44	50.03	15.53
50-60 cm	4	36.65	46.82	16.52
60-70 cm	1	25.55	59.04	15.41
8-m transect				
0-10 cm	40	29.83	51.84	18.33
10-20 cm	40	30.30	51.39	18.30
20-30 cm	40	29.75	52.20	18.05
30-40 cm	38	29.02	52.87	18.11
40-50 cm	27	32.71	48.77	18.52
50-60 cm	13	32.09	51.37	16.54
60-70 cm	4	38.13	44.99	16.88
All depths*				
2 m	168	35.19[a]	50.17	14.64[a]
8 m	202	31.84[b]	50.33	17.83[b]

indicated the 2-m transect had greater sand content than the 8-m transect at all depths except 60-70 cm (Table 2.1). Considering that this stream segment was straightened, it is possible that the sampling at the 60-70 cm depth along the 8-m transect location may have encountered historical stream channel material that would likely have a higher sand content than native soils. Sand and silt content followed an opposing pattern at all depths for both the 2-m and 8-m transects (Figures 2.2 and 2.3, 2.5 and 2.6, respectively), such that as sand increased silt decreased, and vice versa in a somewhat erratic pattern across the sampling locations. Surface (0-10 cm) clay content in the 2-m transect ranged from 10-15% over most sampling locations without dramatic variation (Figure 2.4) while clay content in the 8-m transect ranged from 15-25% over most of the sampling locations, except for a spike at location 21 (Figure 2.7). Clay content in both the 2-m and 8-m transects displayed more variation with depth than at the surface; subsurface clay content in the 2-m transect was in the 10-20% range while clay content in the 8-m transect remained in the 15-25% range (Figures 2.4 and 2.7). There was no significant difference in silt content between transects (Table 2.1).

Spearman correlation coefficients were examined to detect relationships in sand, silt, and clay contents between transects and among depths. No consistent significant correlations were found (Appendix, Table A).

Soil pH in the 2-m transect did not show a consistent trend by depth (Table 2.2), but there was a trend in the 8-m transect of increasing pH with increasing depth. Both transect locations showed increasing levels of Ca with depth, which is likely driving soil pH and is attributable to limestone parent material. A lower pH in the upper depths of

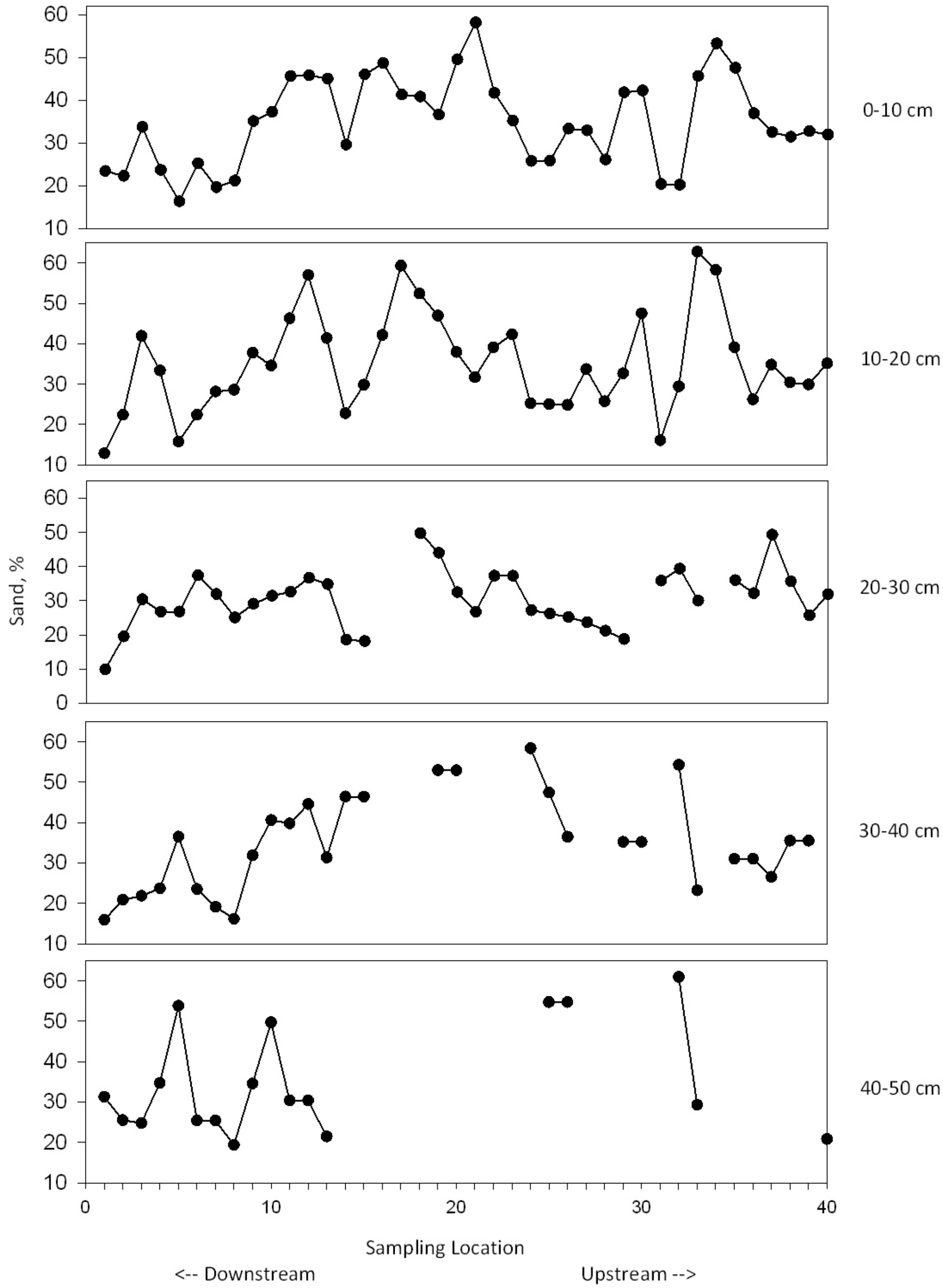


Figure 2.2. Sand content (%) along transect 2-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

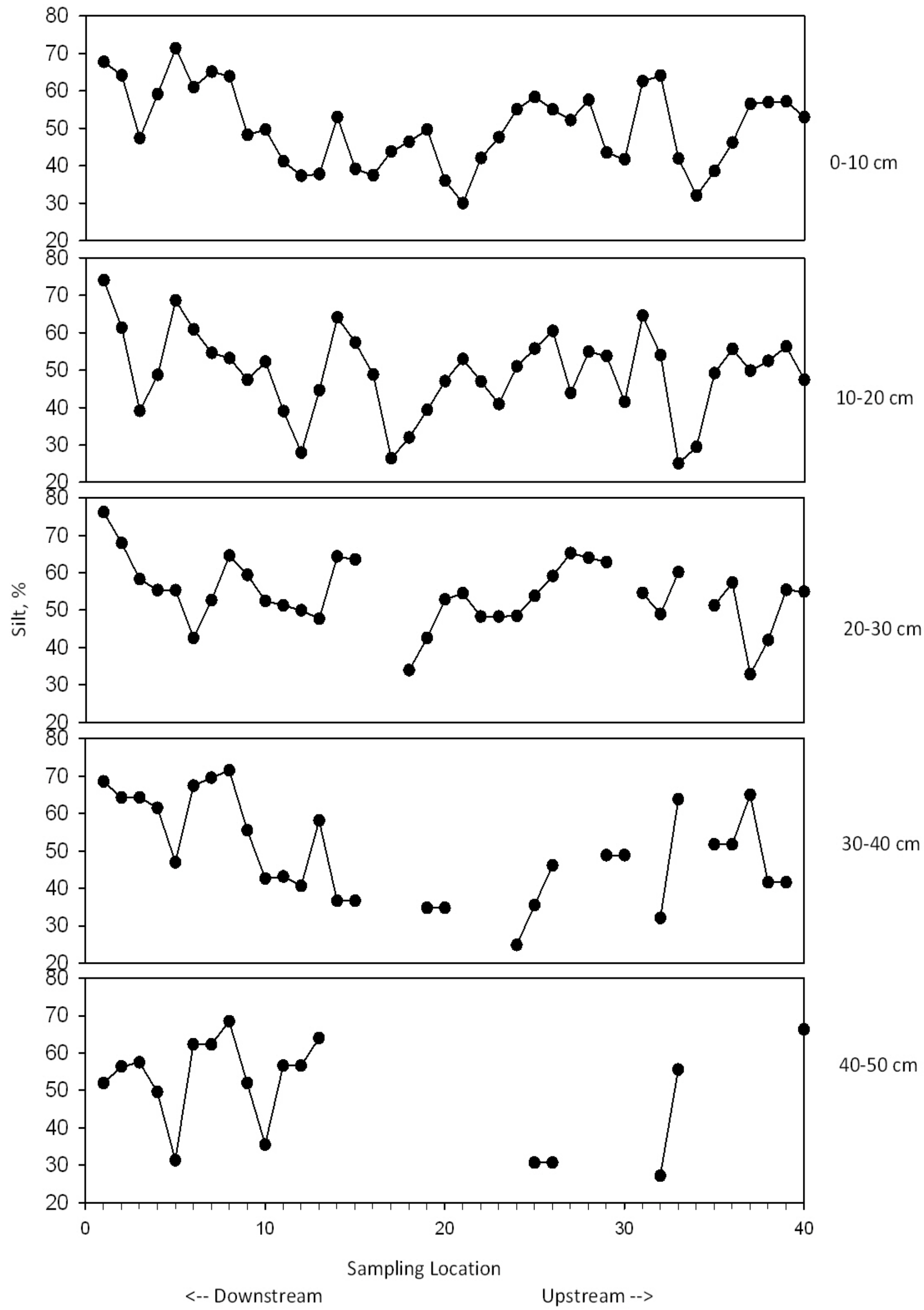


Figure 2.3. Silt content (%) along transect 2-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

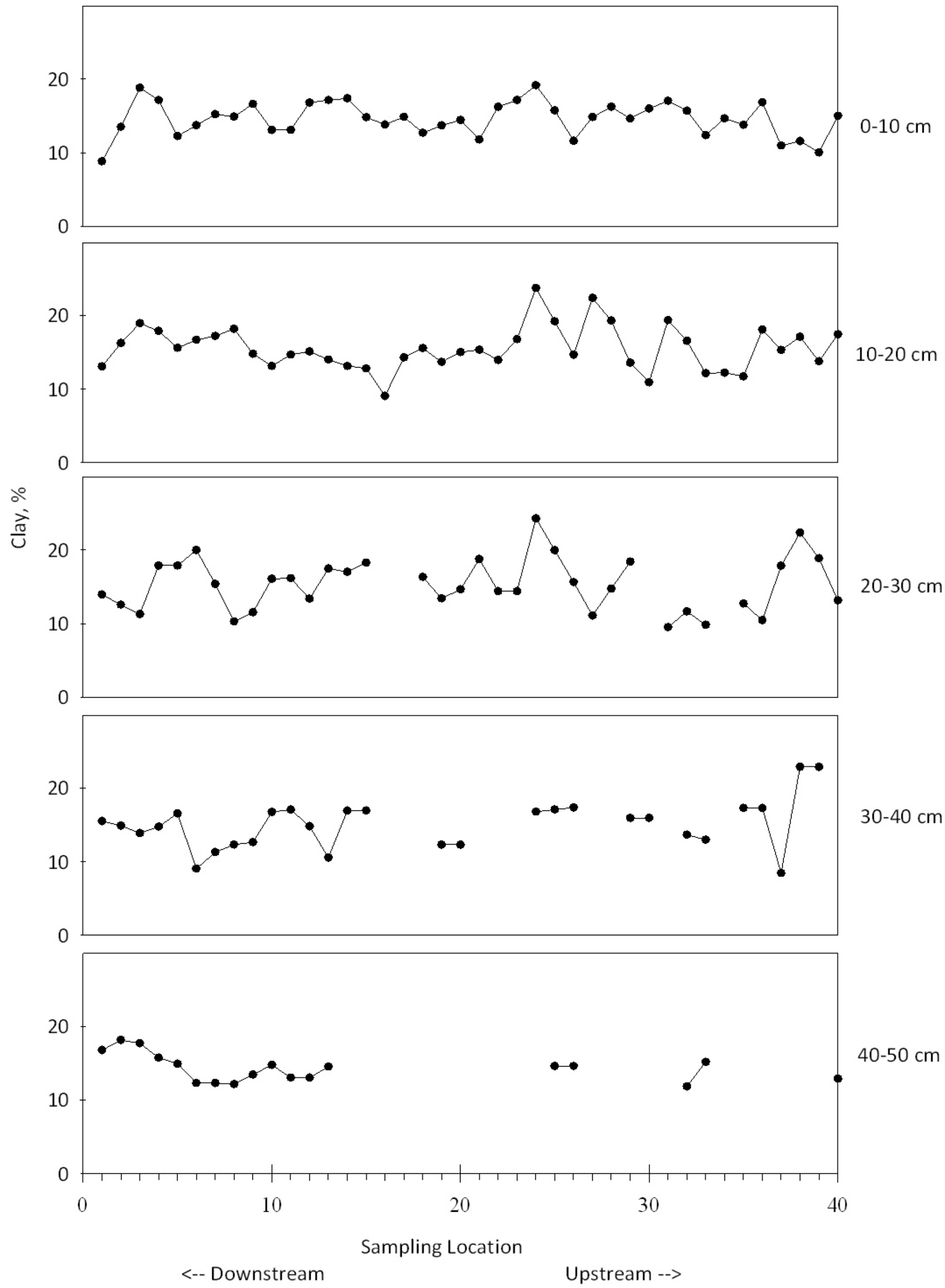


Figure 2.4. Clay content (%) along transect 2-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

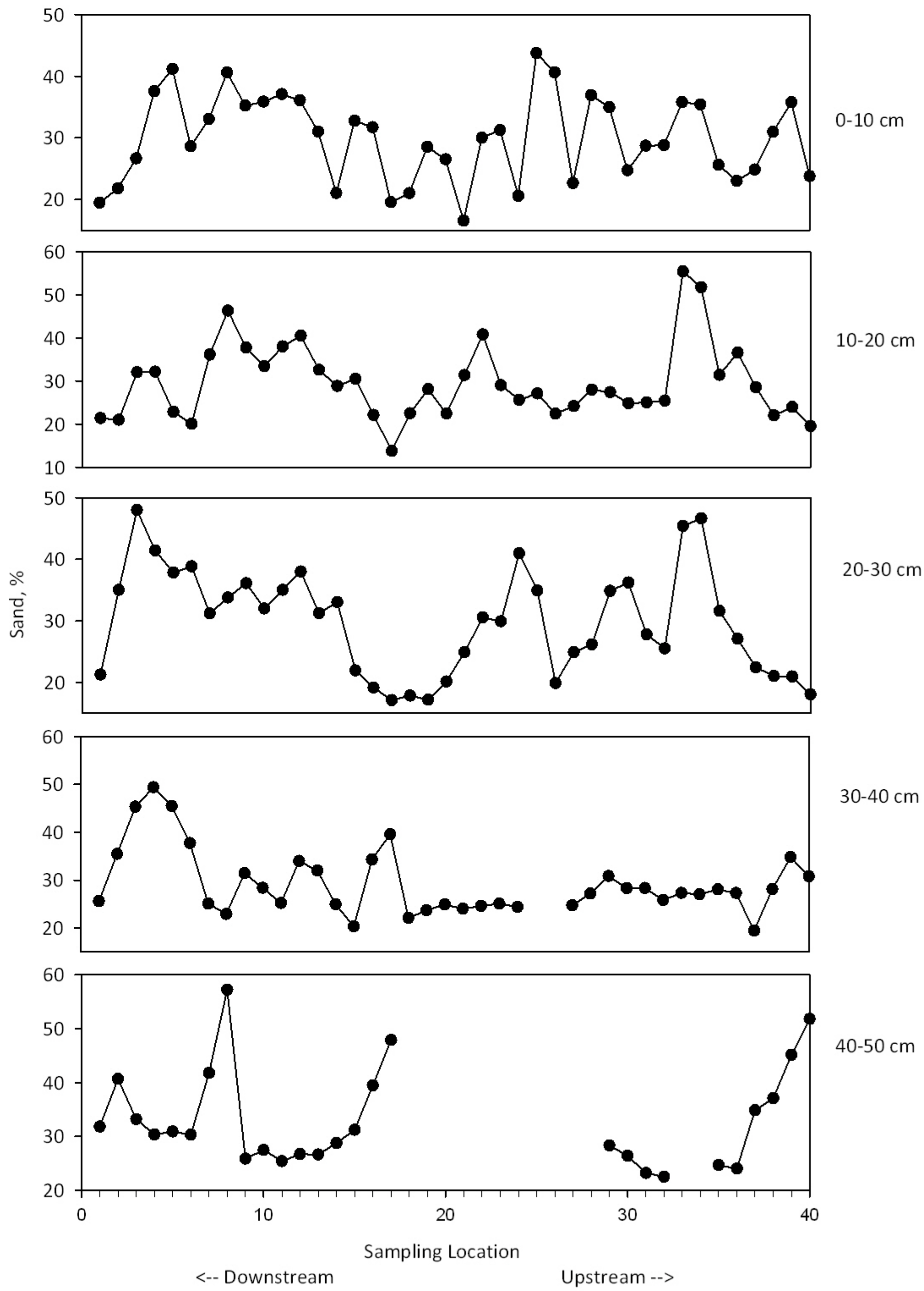


Figure 2.5. Sand content (%) along transect 8-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

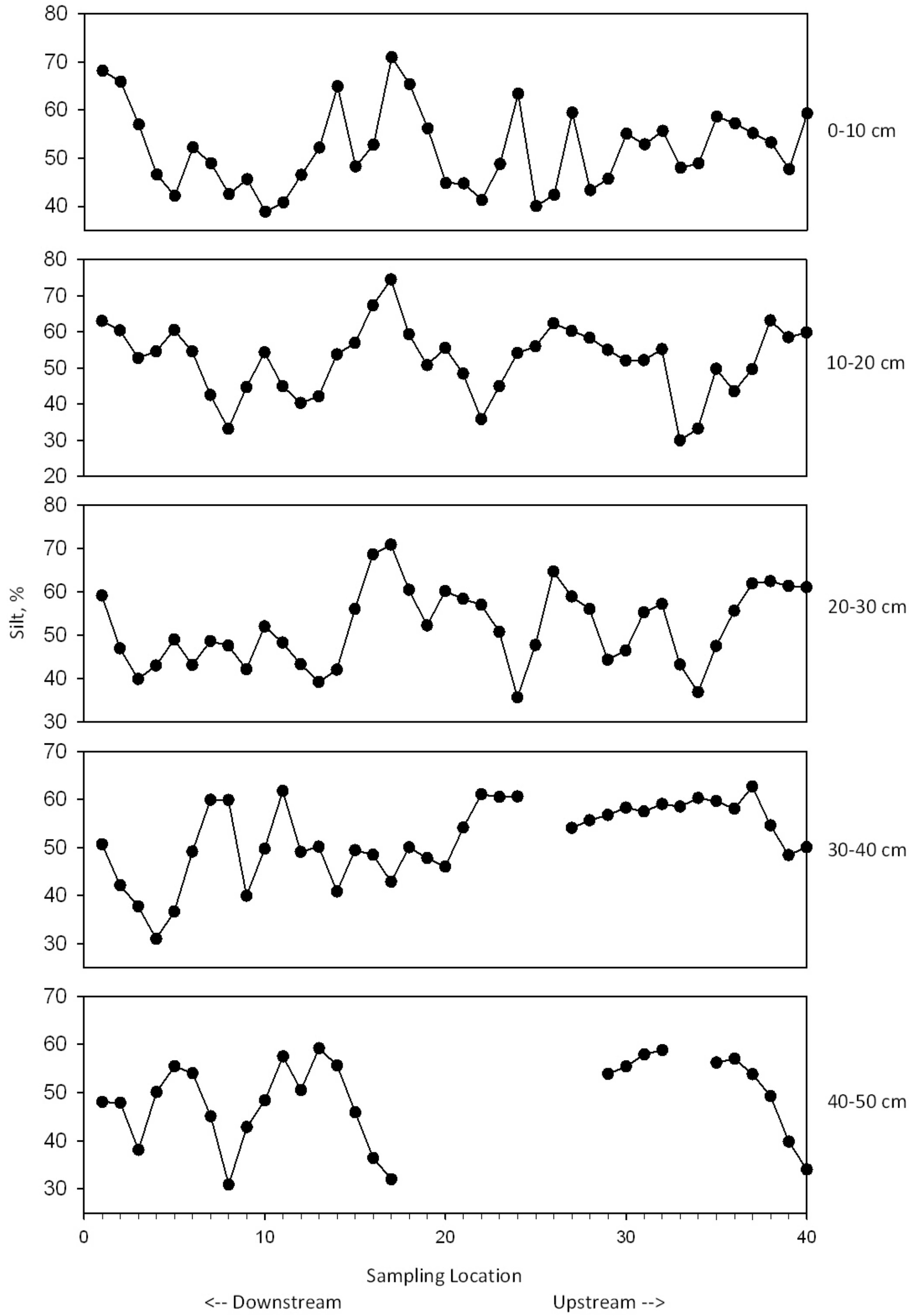


Figure 2.6. Silt content (%) along transect 8-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

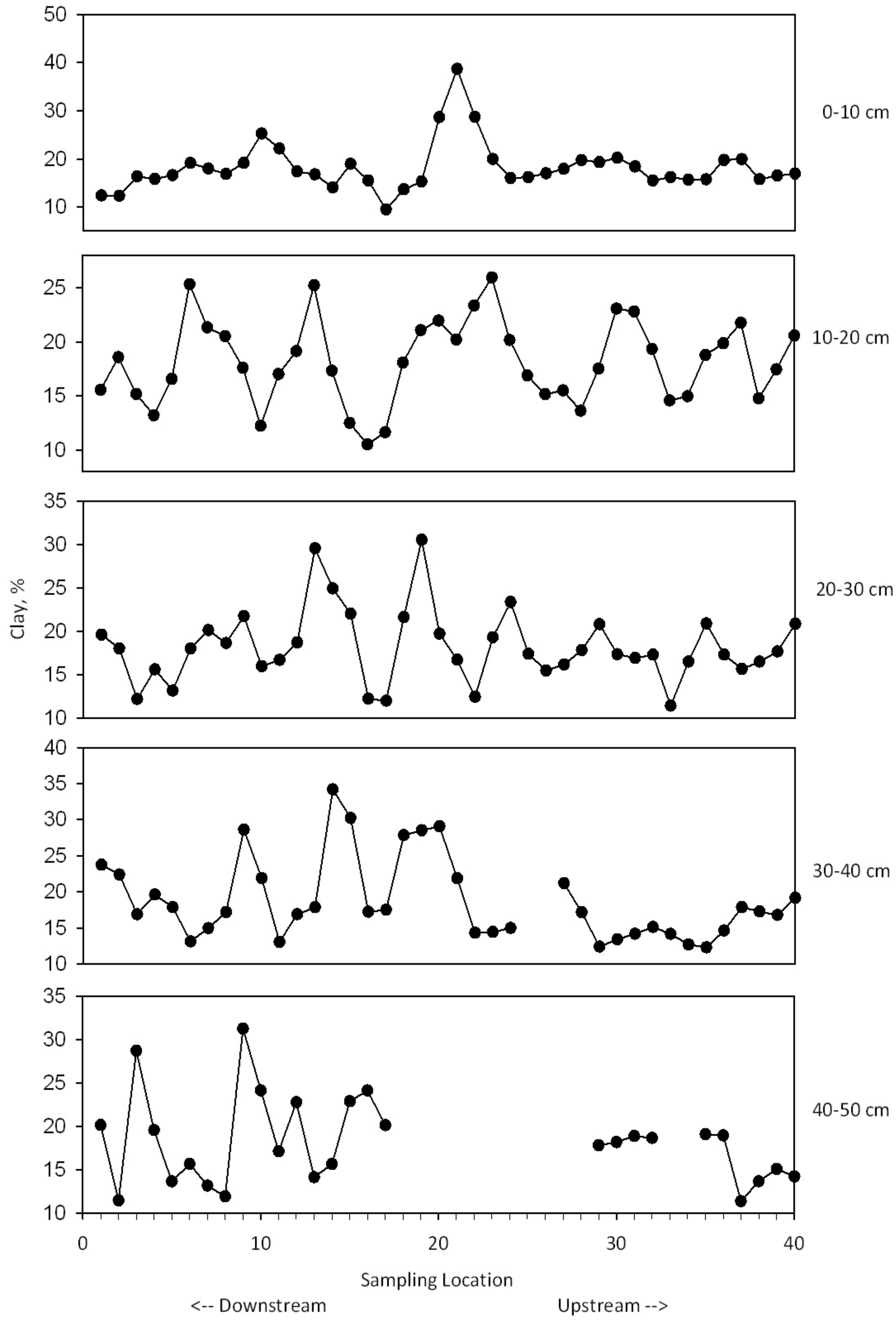


Figure 2.7. Clay content (%) along transect 8-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

Table 2.2. Soil chemical properties. Values reported as least squares means; n = number of samples. For the same property different letters indicate significant differences (P<0.05). *Transect measurements compared among all samples at 2-m and 8-m, respectively.

Depth	n	% C	pH	P mg kg^{-1}	K mg kg^{-1}	Ca mg kg^{-1}	Mg mg kg^{-1}	Zn mg kg^{-1}
2-m transect								
0-10 cm	40	4.93^{a}	6.2^{a}	225^{a}	237^{a}	5031^{a}	169^{a}	5.73^{a}
10-20 cm	40	2.37^{b}	$6.27^{a,b}$	212^{b}	131^{b}	5058^{a}	115^{b}	2.58^{b}
20-30 cm	37	1.95^{c}	$6.34^{b,c}$	212^{b}	$108^{b,c}$	5179^{a}	110^{b}	1.78^{c}
30-40 cm	30	$1.81^{c,d}$	6.43^{c}	$216^{a,b}$	101^{c}	5871^{b}	118^{b}	1.72^{c}
40-50 cm	18	1.44^{d}	6.44^{c}	$219^{a,b}$	91^{c}	$5414^{a,b}$	115^{b}	1.65^{c}
50-60 cm	4	$1.34^{c,d}$	$6.28^{a,c}$	$218^{a,b}$	$98^{b,c}$	$5093^{a,b}$	115^{b}	$1.55^{b,c}$
60-70 cm	1	$1.54^{b,c,d}$	$6^{a,b}$	$211^{a,b}$	$141^{a,b,c}$	$6046^{a,b}$	167^{a}	$1.13^{b,c}$
8-m transect								
0-10 cm	40	4.14^{a}	5.59^{a}	239^{a}	304^{a}	3903^{a}	180^{a}	4.38^{a}
10-20 cm	40	1.9^{b}	5.54^{a}	244^{a}	176^{b}	3887^{a}	139^{b}	1.74^{b}
20-30 cm	40	1.59^{c}	5.6^{a}	240^{a}	126^{c}	4054^{a}	134^{b}	1.23^{c}
30-40 cm	38	1.22^{d}	5.72^{b}	$231^{a,b}$	$106^{c,d}$	4367^{b}	134^{b}	1.26^{c}
40-50 cm	27	1.07^{d}	5.81^{b}	216^{b}	132^{c}	4769^{c}	146^{b}	$1.42^{b,c}$
50-60 cm	13	0.94^{d}	6.06^{c}	211^{b}	$84^{c,d}$	5265^{d}	140^{b}	0.95^{c}
60-70 cm	4	0.85^{d}	6.38^{c}	$234^{a,b}$	34^{d}	5453^{d}	117^{b}	$1.22^{b,c}$
All*								
2 m	170	2.15^{a}	6.36^{a}	217^{a}	112^{a}	5463^{a}	124^{a}	2.39^{a}
8 m	202	1.81^{b}	5.8^{b}	233^{b}	150^{b}	4467^{b}	142^{b}	1.76^{b}

the 8-m transect could be influenced by higher C content in the upper 30-cm, although this trend is not reflected in the 2-m transect. Overall, the 2-m transect locations had a significantly higher soil pH and greater C content than the 8-m transect locations, which suggests that soil pH is not determined by organic matter in this sampling area but instead driven by parent material.

The 2-m transect locations had lower P, K, and Mg but higher Ca and Zn than the 8-m transect. K, Mg, and Zn concentrations were greatest in the surface depths for both transects. Soils in both transects contain P and K levels sufficient for establishing vegetation for riparian buffers and filter strips (UK-CES, 2012); this characterization also provides an inventory of the potential nutrient load entering streams from sloughing stream banks.

Soil P values along the 2-m transect exhibited few distinct spatial patterns with two exceptions: 1) the 10-20 cm depth had a slight increase in soil P at sampling locations 25-28 and the 20-30 cm depth had a dramatic decrease in soil P at sampling locations 18-21 (Table 2.2, Figure 2.8). Soil P values along the 8-m transect had a greater range and exhibited a more pronounced large scale pattern than the 2-m transect. Sampling locations 4-5 and 15-18 had a spike in soil P at all depths (Table 2.2, Figure 2.9), possibly a concentration from weathered phosphatic limestone parent material or depositional area of P-rich sediments.

Both transects showed a decreasing trend of K with depth. The mean surface soil K value along the 2-m transect was 237 mg kg^{-1} with wide spatial variation (Table 2.2,

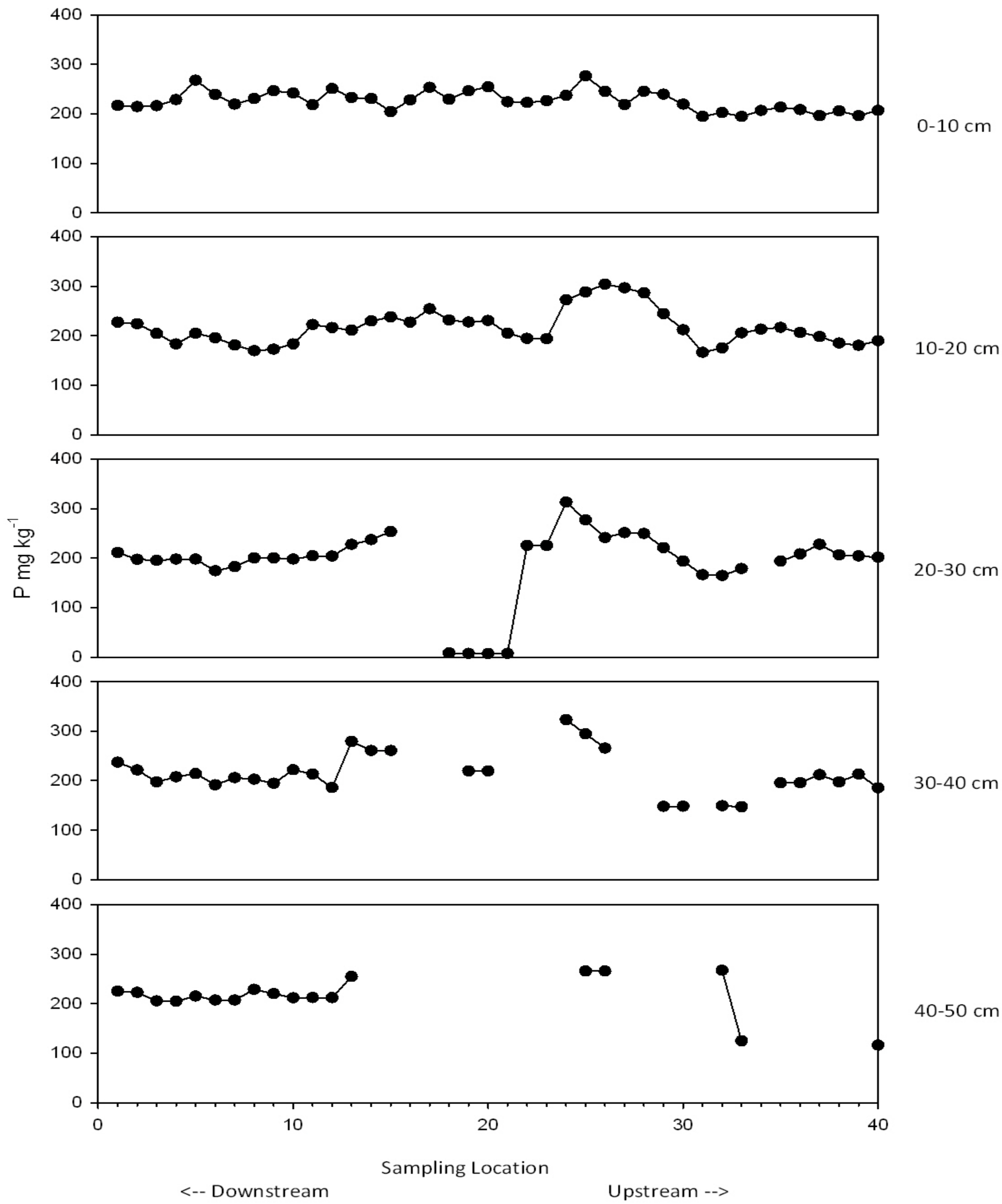


Figure 2.8. Soil P (mg kg^{-1}) along transect 2-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

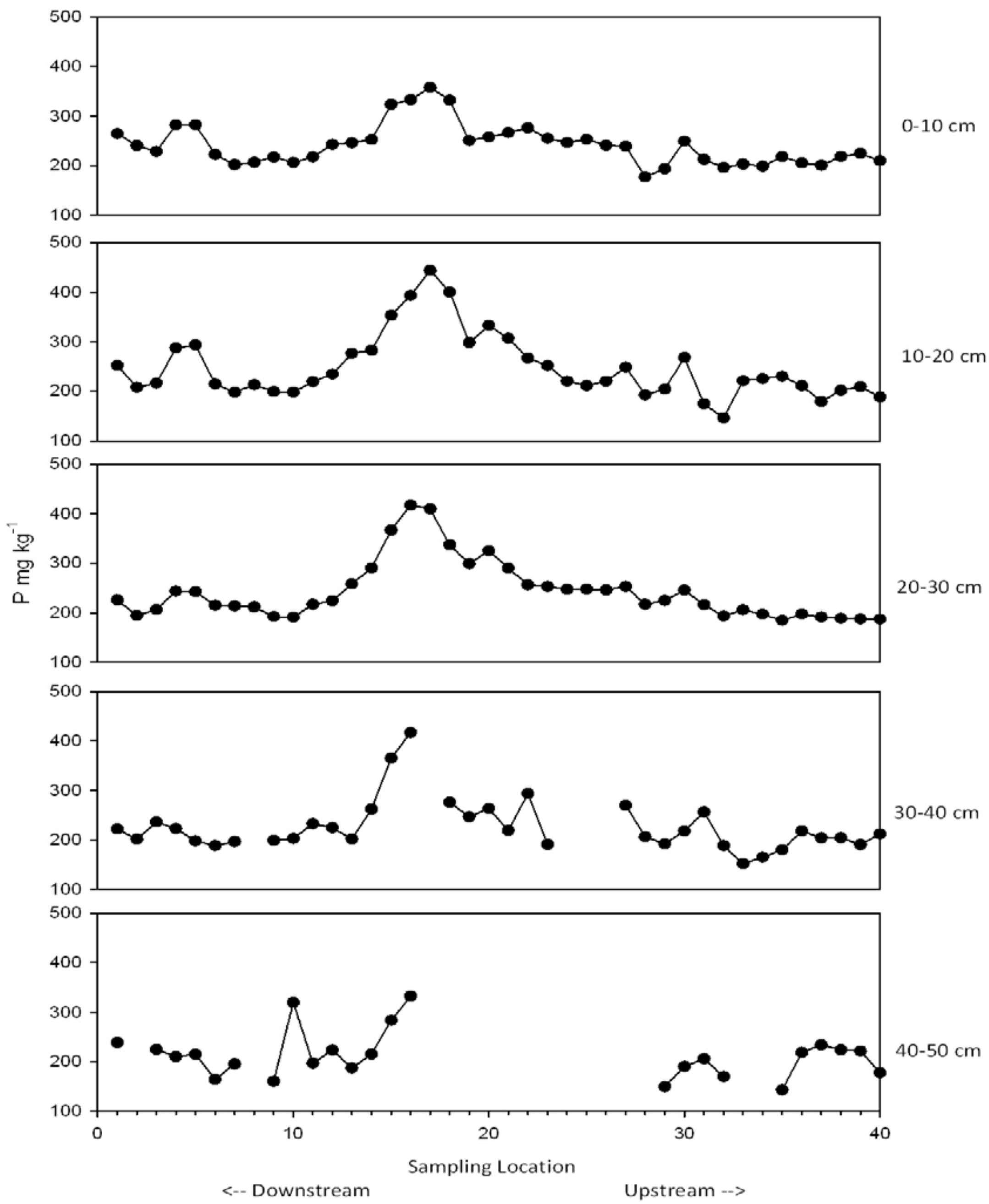


Figure 2.9. Soil P (mg kg^{-1}) along transect 8-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

Figure 2.10). Depths below 20-cm exhibited little spatial variation. Surface soil K values along the 8-m transect showed spatial variation similar to that in the 2-m transect (Figure 2.11). Soil K transformations can be very complex; soil K can be in solution or exchangeable, but more often may be in the mineral form or fixed to soil mineral particles (Helmke and Sparks, 1996). Soil moisture conditions play a role in K availability, with soil wetness associated with less available K (Chen et al., 1987; Winzeler et al., 2008) although the opposite is apparent in this case, considering the visually observed seasonal wet soil conditions in locations 33-35. Increased soil K in these locations may simply be K associated with alluvial soil minerals.

Percent C significantly decreased with depth to the 20-30 cm depth along the 2-m transect and to the 30-40 cm depth along the 8-m transect (Table 2.2). When considering all depths and all locations, soils along the 2-m transect location had significantly greater C content than soils along the 8-m transect location. A simple visualization of the surface depth (0-10 cm) C content along both transects reveals a spatial relationship between the two transects (Figure 2.12). The C content in both transects follow a similar pattern of variation in locations 1-8 (Figures 2.12, 2.13, and 2.14, *0-10 cm depth only*) and in locations 19-24, such that as C content increased or decreased in the 2-m transect so did the C content in the 8-m transect, with the magnitude of C content generally being higher in the 2-m transect. The same relationship is shown in locations 33-40, although the magnitude of C content was

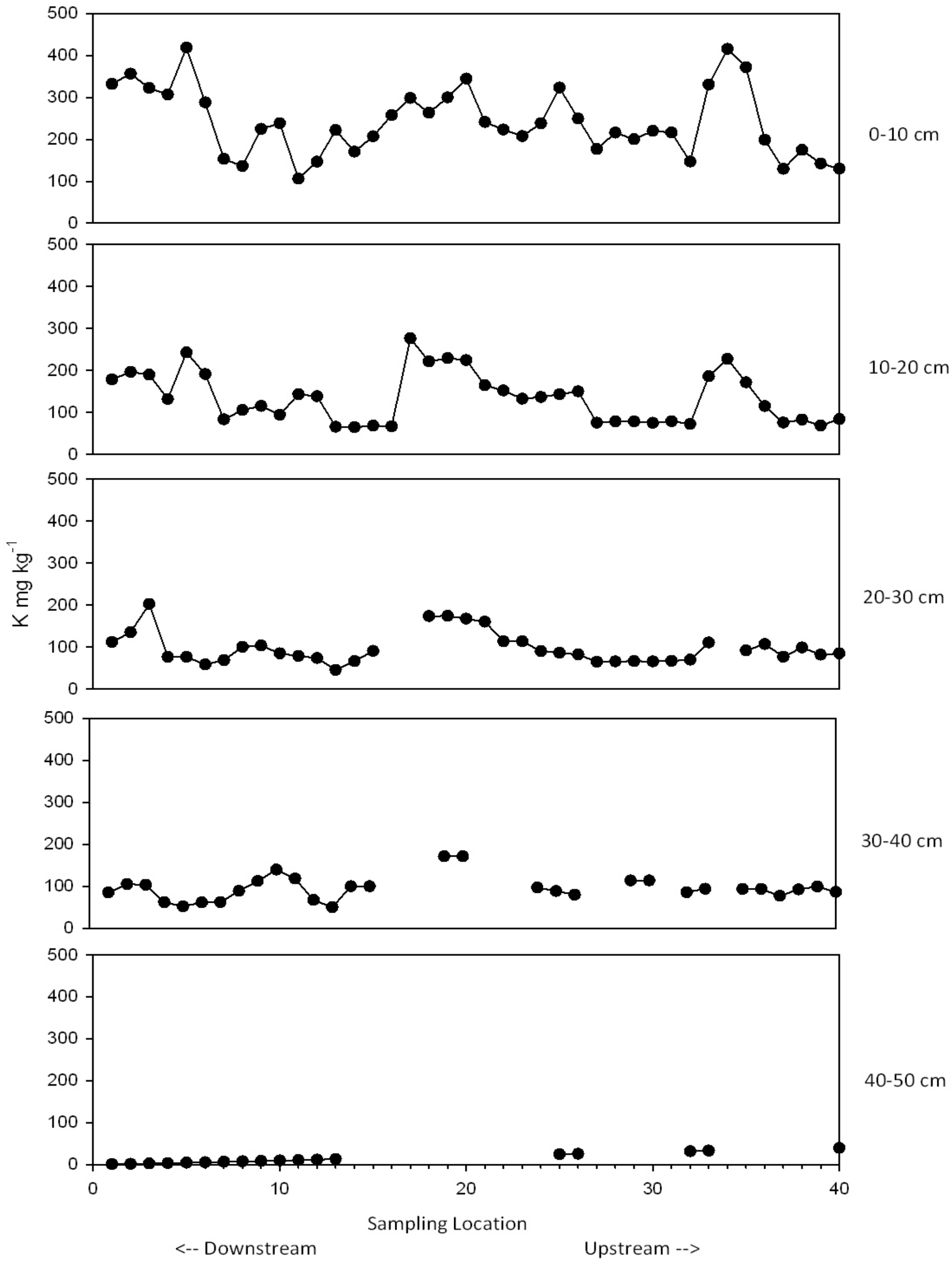


Figure 2.10. Soil K (mg kg^{-1}) along transect 2-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

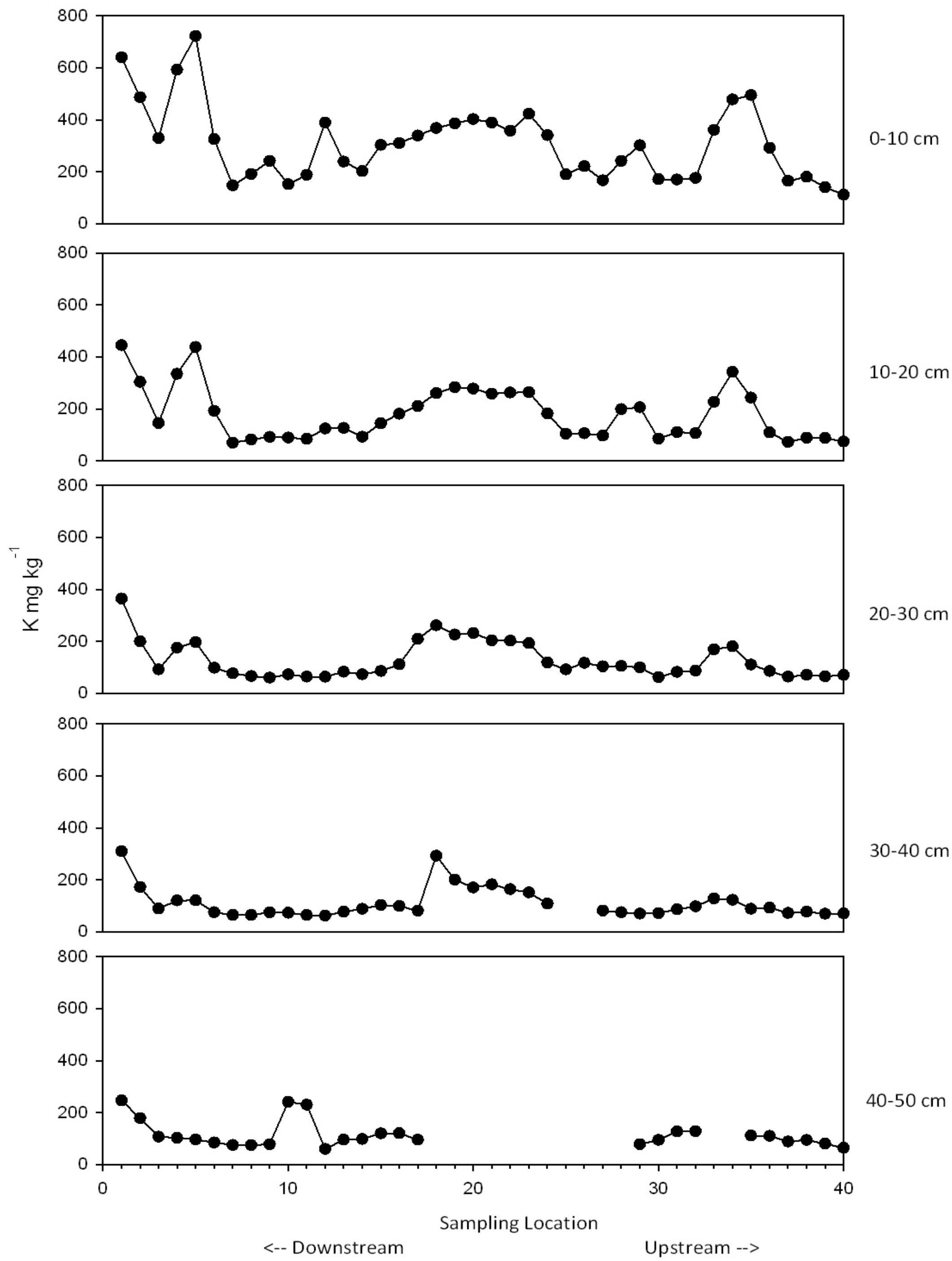
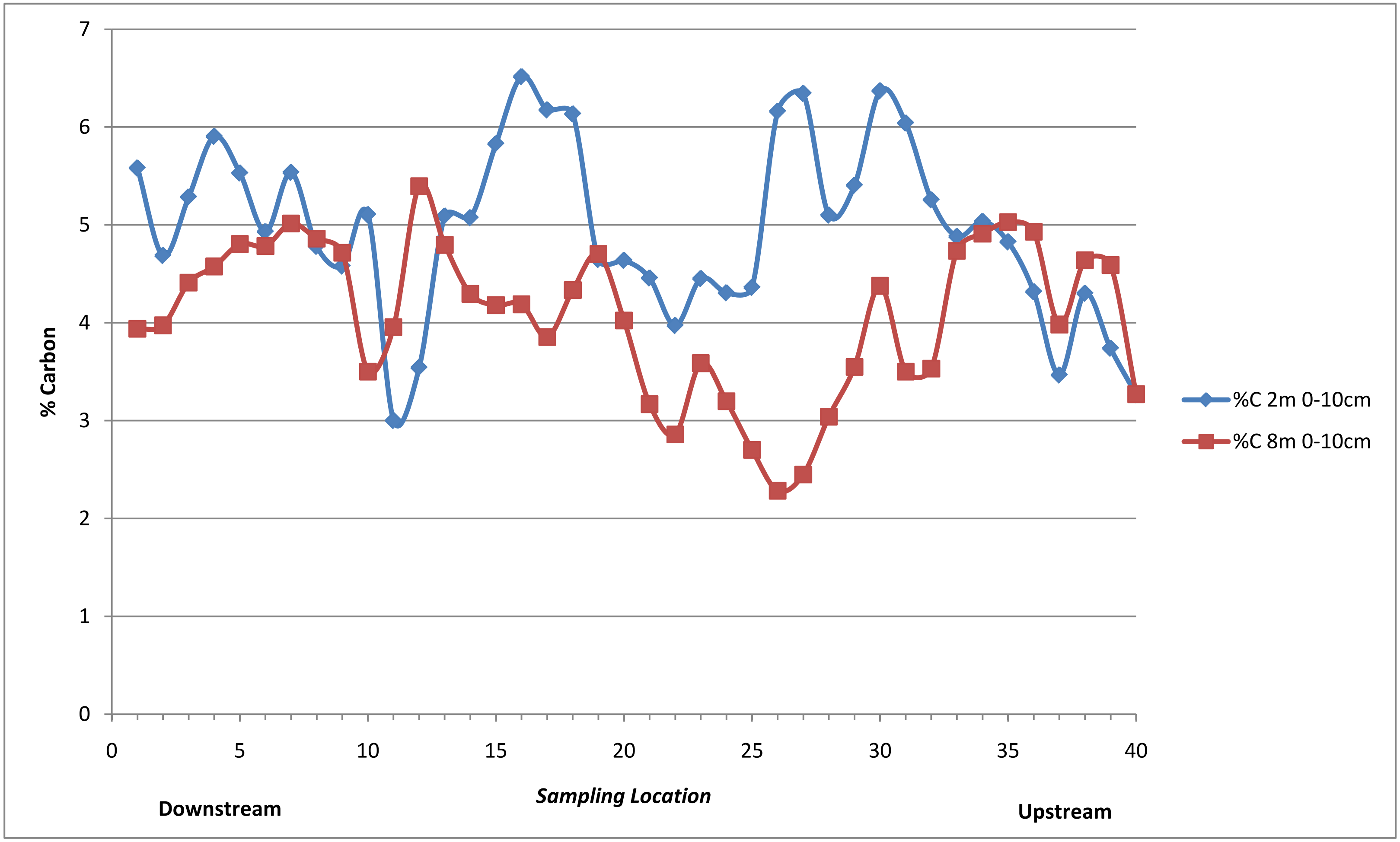


Figure 2.11. Soil K (mg kg^{-1}) along transect 8-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

Figure 2.12. Soil carbon (%) at 0-10cm depth along both the 2-m and 8-m transects.



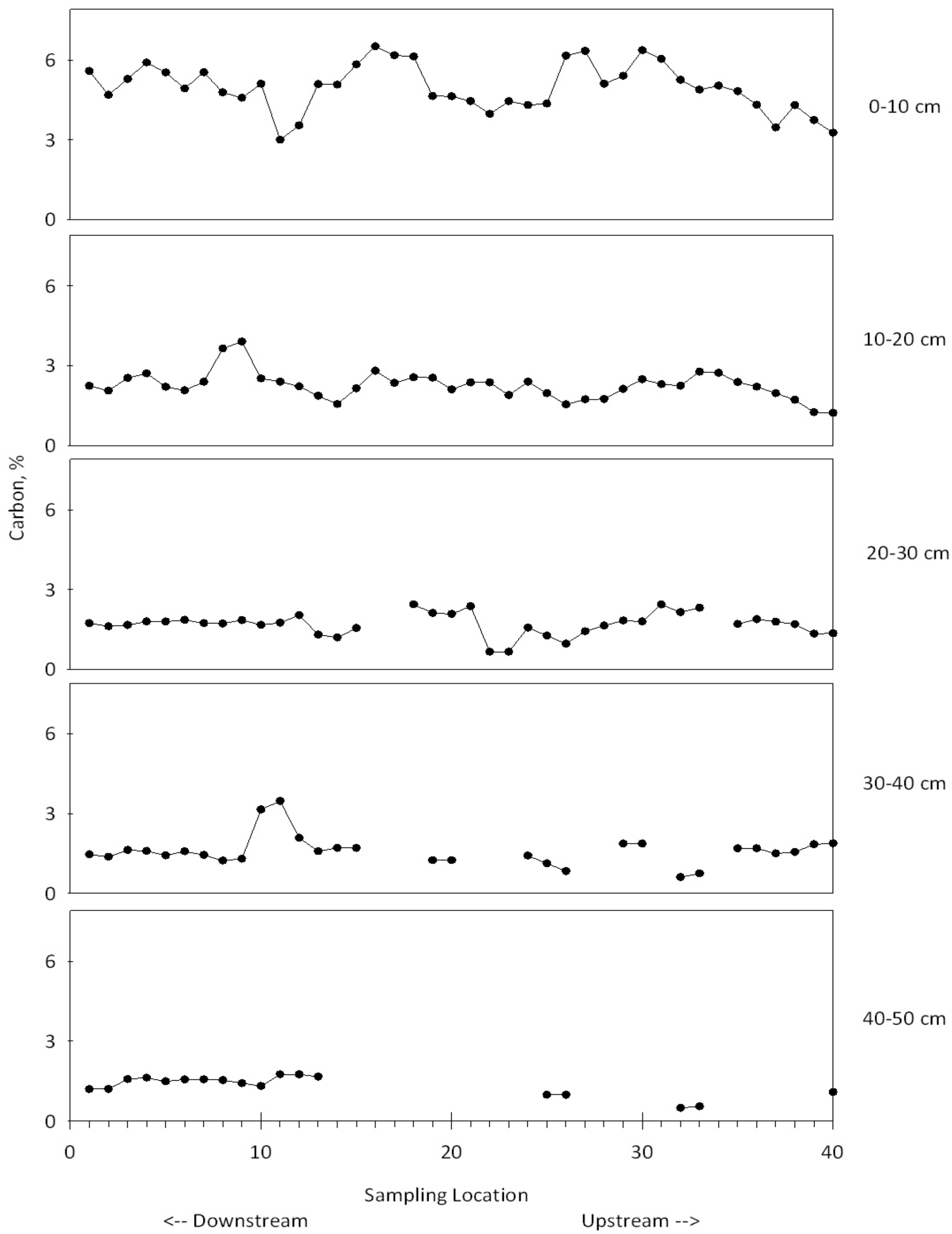


Figure 2.13. Soil carbon (%) along transect 2-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

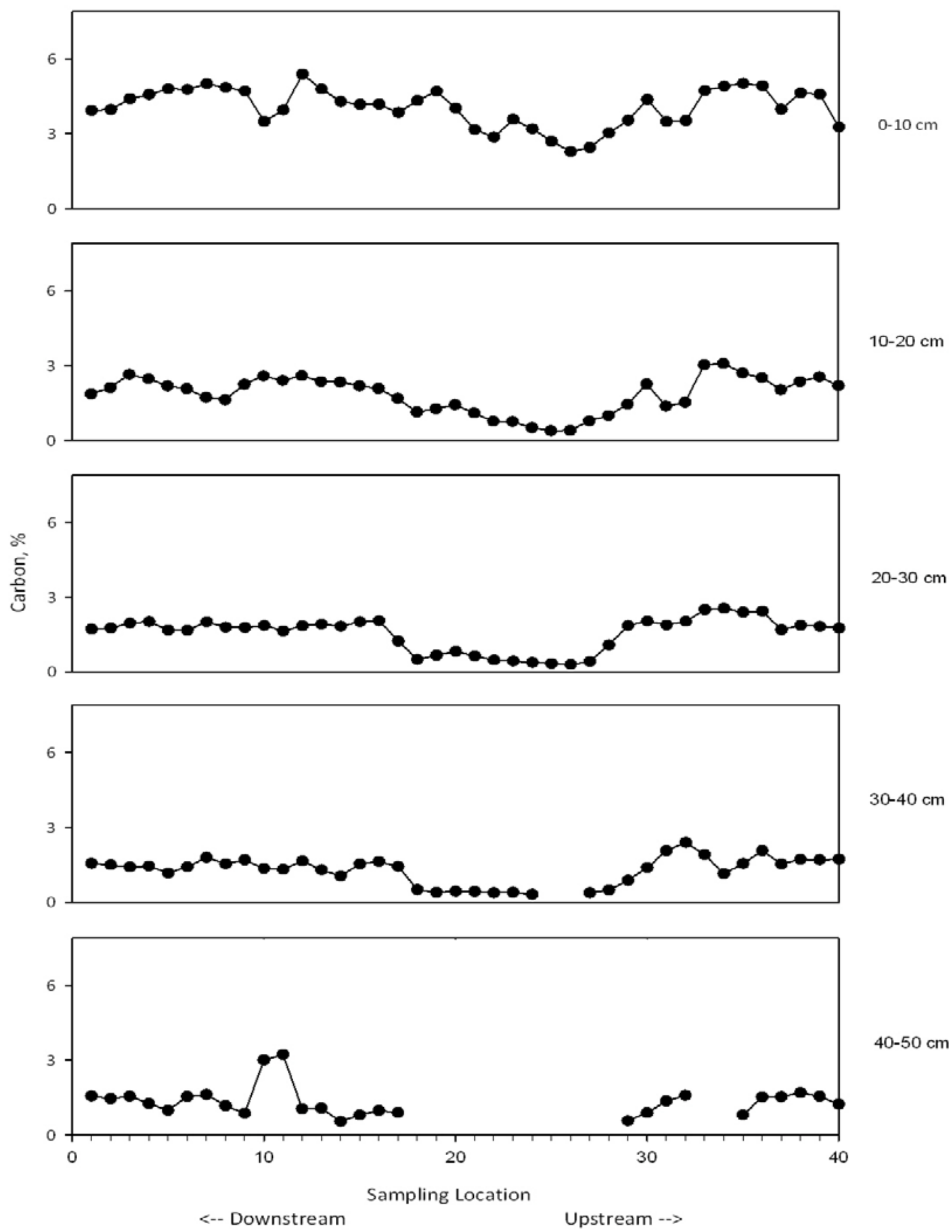


Figure 2.14. Soil carbon (%) along transect 8-m from stream, in 10-cm depth increments. Sampling location at 0 equals most downstream location of sampling. Gaps in the data indicate a restrictive layer preventing sample collection.

higher in the 8-m transect. This higher C content in the 8-m transect at locations 33-40 may be a result of wetter soils at the 8-m transect than at the 2-m transect, as observed during field sampling. The soils in this area of the study site tend to have standing water longer after rain events than the other sampled locations. The wetter soils in the 8-m transect could have decreased decomposition rates compared to the 2-m soils, resulting in C accumulation. Locations 9-18 and 24-32 appear to have opposing trends in C content, with the 8-m transect having noticeably lower C content than the 2-m transect except at locations 11 and 12. Locations 12 and 13 are separated by an improved equipment crossing, although there is no indication of this feature influencing soil C. Locations 25-32 have a steeper slope between the two transects than in other locations and a convex landform, possibly attributing to increased drainage and subsequently a lower water content, increased soil respiration, or both, at the 8-m location and a lack of accumulation of soil C (Abnee et al., 2004; Gessler et al., 2000). In addition, the 2-m transect at locations 25-32 tend to be locations of flooding debris deposition. These results concur with observations made by Blazejewski et al. (2009), who suggest that geomorphic processes such as flooding and deposition significantly influence soil C distribution in first- through fourth-order stream riparian areas.

Spatial patterns in C content at the subsurface sampling depths are less distinct than the surface depth along the 2-m transect and generally reflect lower soil C content (Figure 2.13). Subsurface C along the 8-m transect reflects the pattern of the surface C, with a noticeable decrease at locations 17-28 (Figure 2.14). Spearman correlation coefficients were examined to detect C content relationships between transects and among depths. No consistent significant correlations were found (Appendix, Table A).

Semivariogram analysis focused on soil C because it is a soil property that can be influenced by management and may be an indicator of riparian buffer function (Lowrance et al., 2002). Experimental semivariograms of percent C were calculated for both transects to a maximum depth of 40-cm (no significant difference in C content was shown below this depth). A semivariogram is a depiction of the average of squared differences between observations separated by a given lag distance. In this case, the semivariogram depicts the average variance in percent C of pairs of observations separated by a lag distance of 10-m. Semivariogram models consist of three components: 1) the nugget variance, which is the extrapolated intercept at lag = 0; 2) a structural component, which when added to the nugget variance makes up the sill or total semivariance; and 3) the range, the distance over which spatial dependence is observed for the spherical model chosen here.

The spatial dependence of percent C between the two transects and among depths is shown in Table 2.3. The nugget:sill ratio was calculated to express the nugget semivariance as a percentage of the total semivariance (Cambardella et al., 1994). This value was used to define classes of spatial dependence, similar to those described in Cambardella et al. (1994), with the addition of one class as follows: if ratio was ≤ 25, the relationship was considered to have strong spatial dependency; if ratio was between 25 and 50, the relationship was considered to have moderate spatial dependency; if the ratio was between 50 and 75, the relationship was considered to have slight spatial

Table 2.3. Parameters for % carbon semivariogram models.

Depth	Model	*Semivariance* Nugget	Total	Range (m)	Nugget[1] (%)	Spatial class[2]	Spatial class[3] (Cambardella et al., 1994)
2-m transect							
0-10 cm	Spherical	1.10	1.58	70.01	69.21	Slight	Moderate
10-20 cm	Spherical	0.25	0.36	32.82	68.86	Slight	Moderate
20-30 cm	Spherical	0.06	0.22	62.52	25.65	Moderate	Moderate
30-40 cm	Nugget	0.78	0.78	--	--	--	--
8-m transect							
0-10 cm	Spherical	0.46	1.23	129.83	37.76	Moderate	Moderate
10-20 cm	Spherical	0.06	0.98	140.70	5.76	Strong	Strong
20-30 cm	Spherical	0.00	0.81	133.80	0.00	Strong	Strong
30-40 cm	Spherical	0.18	0.60	150.86	29.37	Moderate	Moderate

[1] *Nugget = Nugget semivariance/total semivariance x 100.*

[2] *Spatial classes: Strong = strong spatial dependency (%Nugget<25); Moderate = moderate spatial dependency (%Nugget 25-50); Slight = slight spatial dependency (50-75); Weak = weak spatial dependency (%Nugget>75).*

[3] *Spatial classes (Cambardella et al., 1994): Strong = strong spatial dependency (%Nugget<25); Moderate = moderate spatial dependency (%Nugget 25-75); Weak = weak spatial dependency (%Nugget>75).*

dependency; and if the ratio was >75, the relationship was considered to have weak spatial dependency.

Semivariogram models shown here for C content in the 2-m transect indicate spatial structure in the upper 30-cm, but ranges vary (Table 2.3, Figure 2.15). The surface depth (0-10cm) indicates spatial structure to a range of 70-m (Figure 2.15a), meaning that samples taken within 70-m of each other show similar percent C content. This range drops to 33-m at the 10-20 cm depth (Figure 2.15b), then increases to 63-m at the 20-30 cm depth (Figure 2.15c). Carbon content measurements separated by distances shorter than these ranges are considered to be related spatially. The 30-40 cm depth variogram model showed a pure nugget effect (Figure 2.15d), indicating strong variability even over very short distances although the raw data show little variation over the sampling distance compared to the surface layer (Figure 2.13); the semivariance at the shorter lag distance is as large as the larger scale variance. The sampling distance employed in this study likely exceeded that required to detect spatial dependence in soil C at this depth. Fluctuations of range among depths at the 2-m transect may be a result of extrinsic variability caused by deposition of flood materials containing C. Our spatial classes show slight spatial dependence in the 0-10 cm and 10-20 cm depths, but moderate dependence at the 20-30 cm depth. Comparatively, Cambardella et al. (1994) would have classified all three models to have moderate spatial dependence.

Semivariogram models for the 8-m transect indicate moderate or strong spatial structure in percent C for all depths (Table 2.3, Figure 2.16), with ranges of 130-150 m.

Figure 2.15. Semivariograms for percent C along 2-m transect.

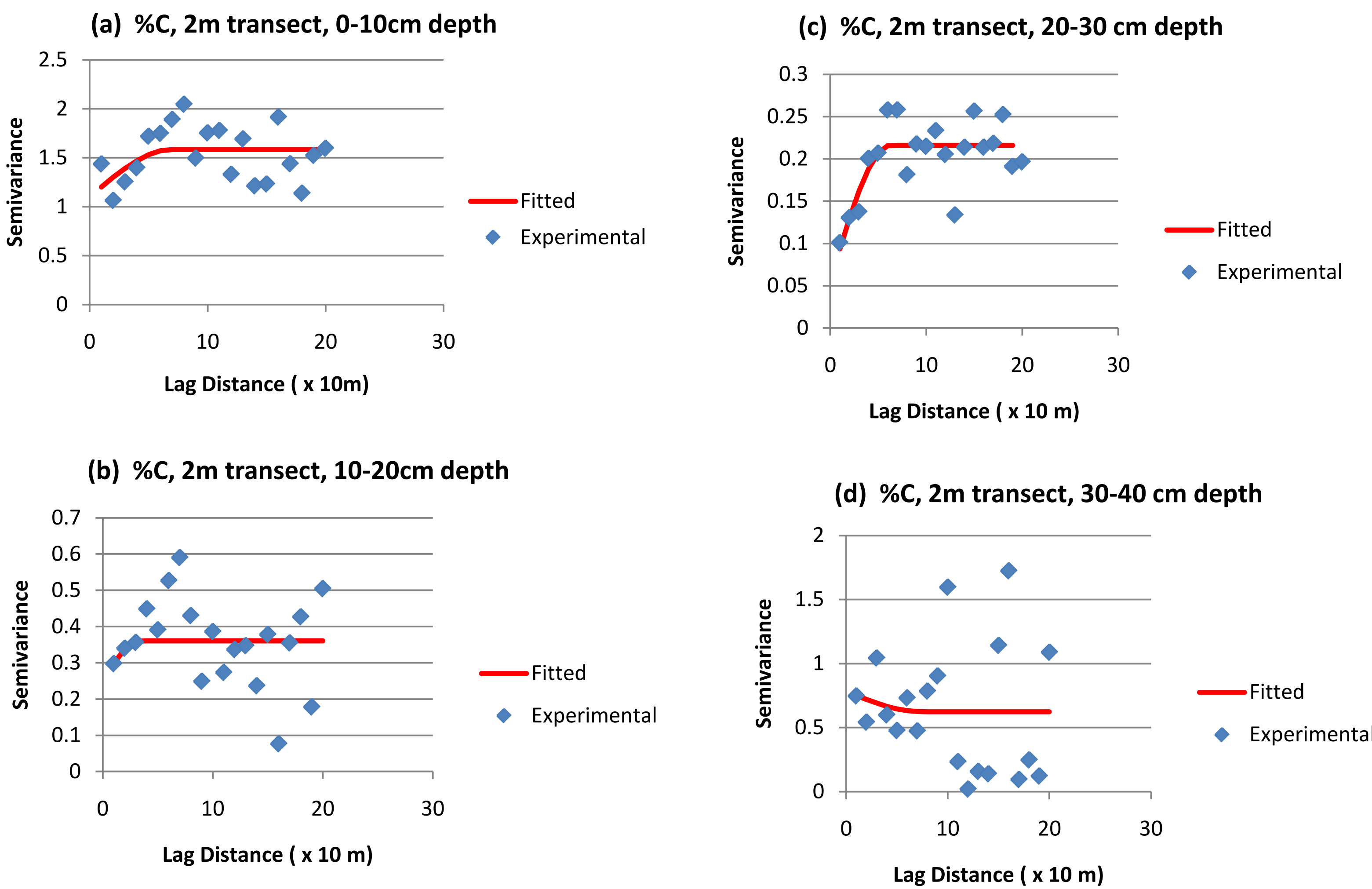
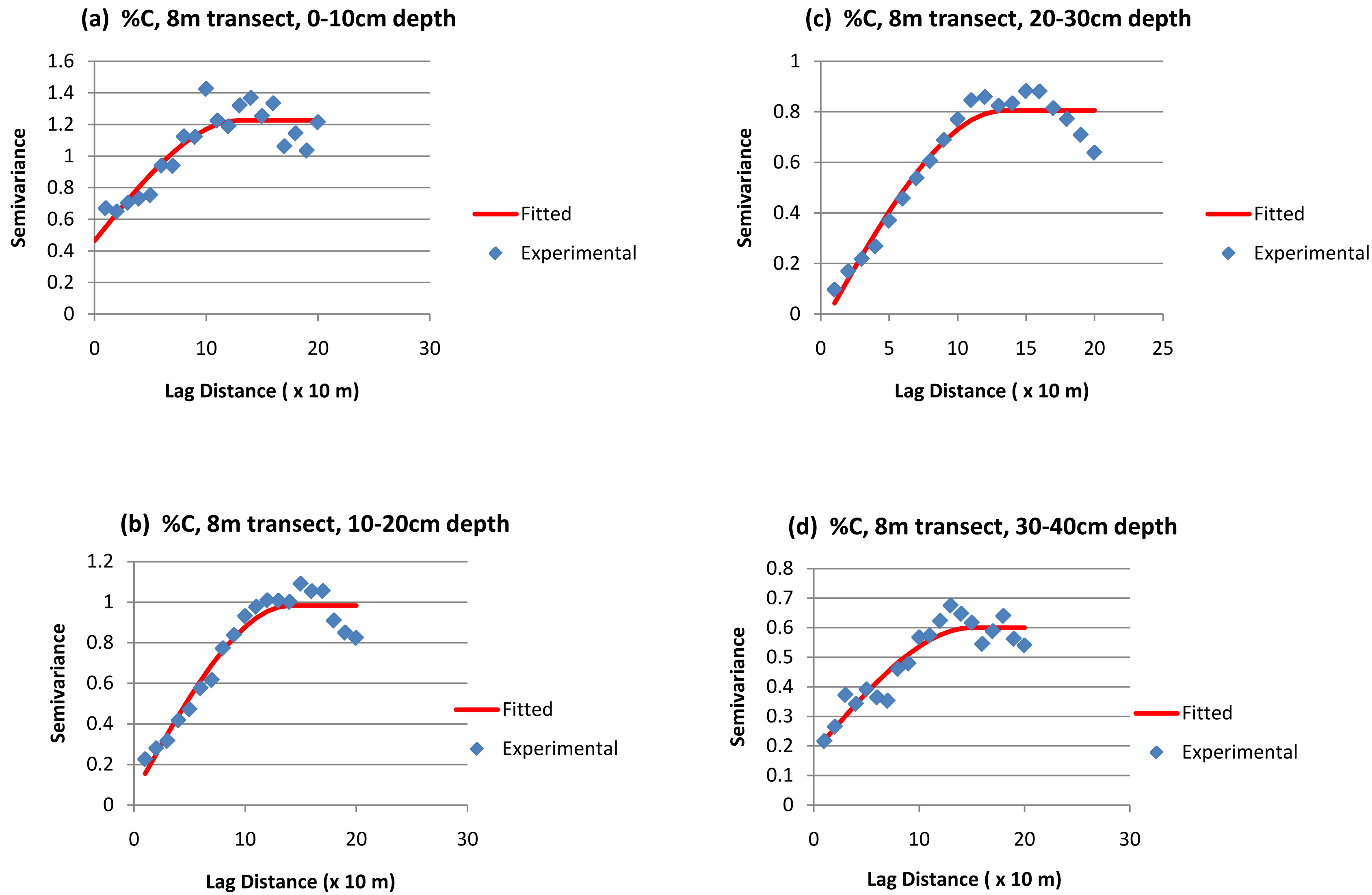


Figure 2.16. Semivariograms for percent C along 8-m transect.



Graphs of the raw data reflect these moderate to strong spatial relationships in soil C along the 8-m transect (Figure 2.14). These results suggest that riparian buffer soil C content farther from the stream is less variable and has a stronger spatial relationship than that closer to the water body. The ranges found for the 8-m transect are similar to those reported by Cambardella et al. (1994) for total organic carbon in two crop fields.

If soil carbon content is considered an indirect assessment of soil quality, and subsequently soil quality as an indicator of a buffer's ability to reduce nonpoint source pollution (Lowrance et al., 2002), a buffer's ability to reduce nonpoint source pollution would be easier to predict at a distance of 8-m from the stream than at 2-m distance from the stream based on soil C distributions. In addition, microscale investigations of landform (concave, convex) and slope influences on soil moisture and soil C dynamics in riparian buffers may also contribute to assessing buffer function in regulating pollutants, reflecting Dosskey et al.'s (2005) suggestion that precision conservation (site-specific filter strips, etc.) may have a greater water quality benefit but with an increased implementation cost.

Conclusions

Soils along the 2-m transect location differed significantly from the soils along the 8-m transect. The 2-m transect soils had greater C content, higher pH, higher Ca and Zn, lower P, K, and Mg, greater sand content, and lower clay content than soils along the 8-m transect location. Some of these differences (C and sand content) are likely a direct result of flooding and debris deposition from the stream while others may simply reflect inherent soil variation. This characterization of riparian buffer soil properties provides insight into the potential nutrient loading of Central Kentucky streams as a result of future sediment loss from sloughing stream banks.

The utilization of semivariogram analysis to describe soil C variability provided explicit information about the range of spatial autocorrelation in this riparian environment. Differences in soil C spatial variation between the two transects indicate that soil properties closer to the water body may be more variable than those further from the stream. Spatial relationships of soil C were stronger along the 8-m transect than along the 2-m transect, suggesting that managing for soil C farther from the stream would be less intensive than managing closer to the stream. If soil C is used as an indicator of buffer function, management intensity to maximize buffer efficacy may vary depending on distance from the stream. Uniform buffer function as a result of landscape management cannot be expected if underlying conditions are influencing soil properties. This relationship proposes a choice for riparian buffer managers: 1) retain more land area for production activities but increase the maintenance intensity for a 2-m wide buffer or, 2) reduce land area for production activities and reduce the maintenance intensity for an 8-m wide buffer.

Establishing baseline soil characteristics, including spatial variability, in riparian areas is an important first step in the process of determining how above-ground management techniques affect soil physical and chemical properties. Further study of these management strategies and their subsequent impact on soil properties will aid in assessing buffer efficacy and the development of site-specific conservation strategies.

Chapter Three

Riparian Buffer Management Influences on Roots, Soil Structural Properties, and Hydraulic Conductivity

Introduction

Approximately 55% of Kentucky's impaired stream miles do not support their designated uses due to agriculture (KEEC, 2010). Farmers have been encouraged to use conservation buffers such as riparian areas, grassed waterways, filter strips, and vegetative barriers as effective means to control agricultural nonpoint source (NPS) pollution for years (Helmers et al., 2006; Schultz et al., 1997; USDA-NRCS, 2000; USDA-NRCS, 2005). However, farmers have been reluctant to remove land from production because of potential income loss. Government-sponsored conservation programs have attempted to compensate agricultural producers for lost income and incentivize the implementation of conservation buffers (KEPPC, 2008; USDA-NRCS, 2000), but the aesthetics and perceptions of unkempt areas on the farm are still a challenge to widespread adoption (Lovell and Sullivan, 2006).

The United States Department of Agriculture (USDA) defines conservation buffers as strips of vegetation placed in the landscape to influence ecological processes and provide various goods and services (Bentrup, 2008); "conservation buffer" is a broad term that includes riparian areas, grassed waterways, filter strips, vegetative barriers, windbreaks, shelterbelts, and wildlife corridors (Bentrup, 2008; Helmers et al., 2006). Riparian areas can be maintained as conservation buffers in the agricultural landscape; they play a key role on the farm because they serve as a transition zone between terrestrial and aquatic ecosystems. Soil water passes through riparian areas before reaching streams and riparian vegetation, by uptake, may significantly modify the amount of dissolved nutrients entering streams (Gregory et al., 1991). Further, vegetation in the riparian zone may reduce NPS pollution in agroecosystems and improve soil quality by increasing infiltration (Bharati et al., 2002), removing sediment and nutrients from surface runoff (Lowrance et al., 2002), and increasing soil organic matter.

The goal of incorporating conservation buffers into agricultural landscapes is to improve ecosystem health (Lovell and Sullivan, 2006), but assessing overall conservation buffer effectiveness can be complex when buffers are located in transitional areas subject to disturbance. Upland and fluvial disturbances include surface and subsurface water movement (Dosskey et al., 2010), water table fluctuations, erosion and deposition of sediments (Wynn and Mostaghimi, 2006), and changes in vegetation as a result of these disturbances (Dosskey et al., 2010; Gregory et al., 1991; Helmers et al., 2006). Therefore, identifying specific soil and/or vegetative properties that measure or indicate desirable buffer behaviors is important. Lowrance et al. (2002) suggested that a buffer's ability to reduce NPS pollutants could be indirectly assessed with time by measuring infiltration rate, aggregate structure, and soil carbon.

Maximizing infiltration of runoff water is expected to decrease the export of soluble and adsorbed pollutants (Helmers et al., 2006) by allowing uptake by vegetation and deposition of sediments prior to reaching surface waters. Soils with a high infiltration capacity are likely to have a greater sediment trapping capability in addition to reducing the release of soluble pollutants to nearby waterways. Soils in naturalized areas have shown increased water infiltration rates because of accumulation and on-site decomposition of leaves and associated increases in earthworm and macroarthropod activity (Millward et al., 2011). In addition, soil permeability is increased by root growth and decay and burrowing by macroinvertebrates grazing on roots and litter (Dosskey et al., 2010), thus creating large pores that enhance water movement.

Hydraulic conductivity is a measure of soil's ability to transmit water (Klute and Dirksen, 1986) and is directly related to pore geometry and organization, which in turn is influenced by soil texture and structure (Ehlers et al., 1995). Soil particles are bound together by soil organic matter into aggregates that provide soil its structure; soil structure is heavily influenced by tillage, traffic (equipment and animal), and soil biology (Oades, 1993). Soil biology has been identified as the driving force of the evolution and maintenance of soil structure (Dick, 1992), and increased microbial activity in soil and related soil aggregate stability may indicate the potential for soil water infiltration, soil sustainability, and soil and ecosystem functions (Paudel et al., 2011).

Plant communities enhance riparian zones by regulating water temperature (English et al., 2004), reducing erosive forces on stream banks (Wynn et al., 2004) by reducing flow velocities (Helmers et al., 2006), and contributing carbon to the ecosystem (Dosskey et al., 2010). Vegetated streamside buffers provide foliage and stems that increase surface roughness and a dense network of roots that bind riparian substrates to increase streambank resistance to erosion (Kiley and Schneider, 2005;

Wynn et al., 2004). Veum et al. (2012), in a 10-year study, demonstrated that vegetated filter strips induced positive changes to soil carbon and aggregate stability compared to no-till row crop production.

Information about the impact of aboveground management on root systems of riparian vegetation is limited. Studies investigating the effects of aboveground vegetation removal (whether by mowing, grazing, or burning) on root systems have predominantly focused on upland prairie or pasture systems (Johnson and Matchett, 2001; Kitchen et al., 2009; Todd et al., 1992), or simulated field conditions (Neigebauer et al., 2000) and report mixed results. Annual mowing has been reported to have no net effect on total root biomass, although it did significantly increase root biomass in the upper 10 cm compared to unmowed treatments (Kitchen et al., 2009). Todd et al. (1992) reported decreased live root biomass under mowing treatments while Dickinson and Polwart (1982) reported an increase in below-ground biomass in the second year of mowing. Considering that mowing is a typical riparian buffer management strategy utilized by Kentucky land managers, research is needed to understand the effects of mowing on riparian buffer plant communities.

The type of vegetation established in riparian zones can influence overall function (Schultz et al., 1997). Switchgrass (*Panicum virgatum*) is a native warm-season grass often recommended for the grass zone in riparian buffers. It has dense, stiff stems that slow surface runoff and promote infiltration. Cool-season grasses such as fescue (*Festuca arundinacea*) are not recommended for riparian buffers that experience overland flow because their stems do not remain upright under surface runoff and they produce eight times less root mass than native grasses (Schultz et al., 1997). However, other research (Lowrance et al., 2002) has shown that cool-season grass filters have twice as much carbon in the upper 20 inches (50.8 cm) of soil as switchgrass, with corresponding higher rates of denitrification. Additionally, Self-Davis et al. (2003) found fescue to have greater infiltration and lower runoff compared to native warm-season grasses. This may indicate that native warm-season grasses are more effective at slowing overland flow, but cool-season grass filters might be more effective at below-ground processes. Native grass strips increase infiltration rates and microbial activity, and might be more effective at providing soil carbon deeper in the soil profile over longer periods than cool-season grasses such as fescue (Lowrance et al., 2002; Schultz et al., 1997).

At present we do not fully understand the interaction between management of above-ground plants and below-ground processes in riparian buffers. Specifically, study is needed to assess how above-ground treatment affects root biomass, the size and stability of soil aggregates, and related hydraulic conductivity. These attributes may influence riparian buffer function and effectiveness in trapping nutrients from surface runoff. Specific information on establishing and maintaining riparian buffers will assist agricultural producers in maximizing the potential for water quality protection by using riparian buffers. The purpose of this study was to assess root biomass, soil aggregate size distribution, water-stable aggregates, hydraulic conductivity, and soil carbon in a riparian buffer after imposing vegetation management variables that included three mowing regimes and one native grass regime.

Methods

Field

The study site is at the University of Kentucky Agriculture Experiment Station, near Lexington, KY (N 38°07'23.98", W 84°29'50.04") (Figure 3.1a). The site is a riparian zone of an unnamed tributary to the Cane Run Creek. Normal annual precipitation for Fayette County (Lexington, KY) is 124-cm (Priddy, 2012); annual precipitation for the duration of this study ranged from 102-166 cm (Table 3.1), including the wettest year on record for Kentucky (2011) (Priddy, 2012). Soils are classified as fine, mixed, active, mesic Fluvaquentic Endoaquolls and mapped as Lanton silty clay loam (dunning) series with fine-textured alluvium parent material derived from limestone (USDA-NRCS, 2011) (Figure 3.1b). Sections of the stream were channelized for agricultural purposes in the 1970s and the surrounding buffer was maintained as mowed grassland (Calvert, 2011) until research plots were established in July 2010. The stream is not incised and thus can access the floodplain during high flow events. The streambanks display little evidence of erosion and the channel bed material is predominantly bedrock. Prior to establishing the treatment plots, the riparian zone consisted of mixed grassland vegetation (e.g. fescue [*Festuca arundinaceae*], bluegrass [*Poa pratensis*], broadleaf weeds) mowed every four to six weeks.

The treatment plots measured approximately 10-m x 15-m, with the 10-m distance parallel to the stream. The experiment design consisted of ten replications of

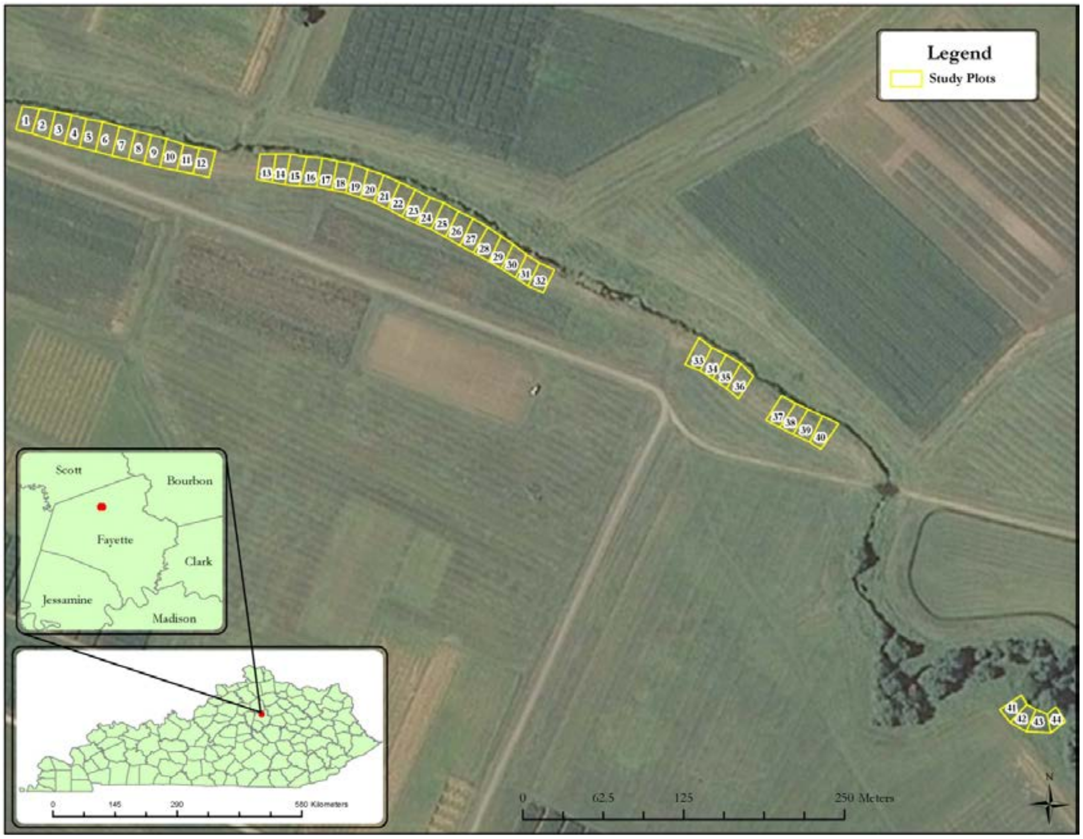


Figure 3.1a. University of Kentucky Agriculture Experiment Station study site in Fayette County, Kentucky (inset). Numbers indicate individual plots.

Figure 3.1b. Aerial view of study site with soil series detail.

Table 3.1. Rainfall totals (cm) for Fayette County, KY.

	2010	2011	2012
January	7.59	5.16	9.45
February	4.47	17.68	6.38
March	3.20	12.42	11.02
April	7.90	33.71	5.00
May	24.97	14.81	8.81
June	10.87	5.89	3.15
July	20.47	8.89	14.99
August	3.30	10.31	3.05
September	1.65	16.38	16.66
October	3.07	12.32	3.07
November	11.53	18.69	3.38
December	5.79	9.37	16.87
Total	104.83	165.63	101.83

Data reported at KY Mesonet station for Fayette County, http://kymesonet.org, accessed December 31, 2012.

four treatments in a repeating pattern to consider spatial variation along the length of the stream. Plot treatments were: 1) intensive mowing (mowed to 15-cm (6-inch) height every four weeks during the growing season); 2) moderate mowing (mowed to 15-cm (6-inch) height twice during the growing season); 3) no mow; and 4) native grass transition. Plots receiving mowing treatments were mowed with a 1.8-m (6-foot) mower attached to a 29,828-watt (40-horse power) tractor perpendicular to the stream to avoid influencing adjacent treatment plots. Native grass transition plots received glyphosate herbicide treatment in Fall 2010 and Spring 2011 to eliminate existing vegetation, and drill-seeded with a native grass-forb mixture (Roundstone Native Seed, Upton, KY) in June 2011. The native grass-forb mixture contained the following species: big bluestem (*Andropogon gerardii*), Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), partridge pea (*Cassia fasciculata*), Illinois bundleflower (*Desmanthus illinoensis*), black-eyed Susan (*Rudbeckia hirta*), and purple coneflower (*Echinacea purpurea*). Native grass plots were mowed once post-planting (May 2012) to reduce weed pressure. Four undisturbed plots located upstream of the study site served as a control. In this case, the term undisturbed is used to describe a riparian area receiving no mowing treatment with some trees present. Undisturbed plots were of the same dimensions as treatment plots, dominated by mixed grassland vegetation at the 2-m distance from the stream and shaded by mature trees adjacent to the stream channel.

A sampling transect located 2-m from top-of-bank was established along all plot locations. Soil samples were collected in May-June 2011 and May-June 2012 at 1, 3, 5, 7, and 9 m locations along the 2-m transect (Figure 3.2) within each plot using a JMC Environmentalist's Sub-Soil Probe Sampling System. Soil cores measuring 2-cm diameter were collected to a depth of 30-cm, divided into 10-cm increments, and stored at 3°C until processed for root biomass.

Soil surface infiltration was measured along the 2-m transect at the 5-m location (center) within each plot using a tension infiltrometer (Soil Measurement Systems, ND). Sampling locations were prepared by inserting a 20-cm PVC ring into the soil surface, removing all above-ground vegetation inside the ring with clippers, placing a piece of polyamide membrane (31-µm mesh opening) fabric over the soil surface, and applying a layer of fine sand (119-µm) evenly spread over the surface of the membrane to create an even contact surface (Figure 3.3). Flow-rate measurements were taken at -10 cm, -5 cm, and -1 cm tensions until steady-state conditions were reached. These sets of flow-rate measurements were used to calculate saturated hydraulic conductivity (K_{sat}) using Wooding's (1968) equation for approximating steady-state unconfined infiltration rates into soil from a circular interface with radius *r*:

$$Q=\pi r^2 K[1+(4/\pi r\alpha)]$$

where Q is water flux (in cubic length units per time), K is hydraulic conductivity (length units per time), and α is a constant from the expression by Gardner (1958):

$$K(\psi)= K_{sat} \exp(\alpha\psi)$$

where K_{sat} is the saturated hydraulic conductivity and is a constant fitted to the K(ψ) data pairs. Using two soil water potentials (ψ_1 and ψ_2), steady state infiltration fluxes

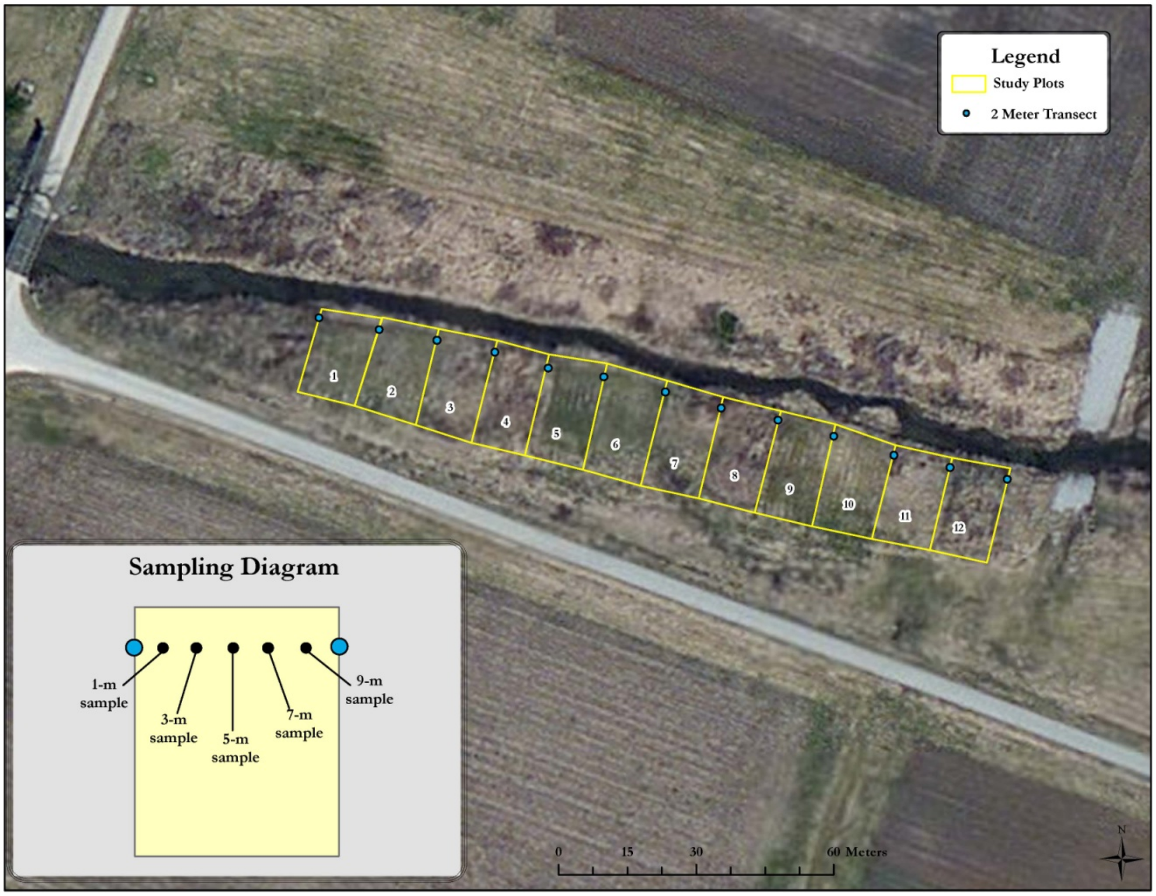


Figure 3.2. Study plots. Example plot layout is shown with detailed within-plot sampling scheme (inset).

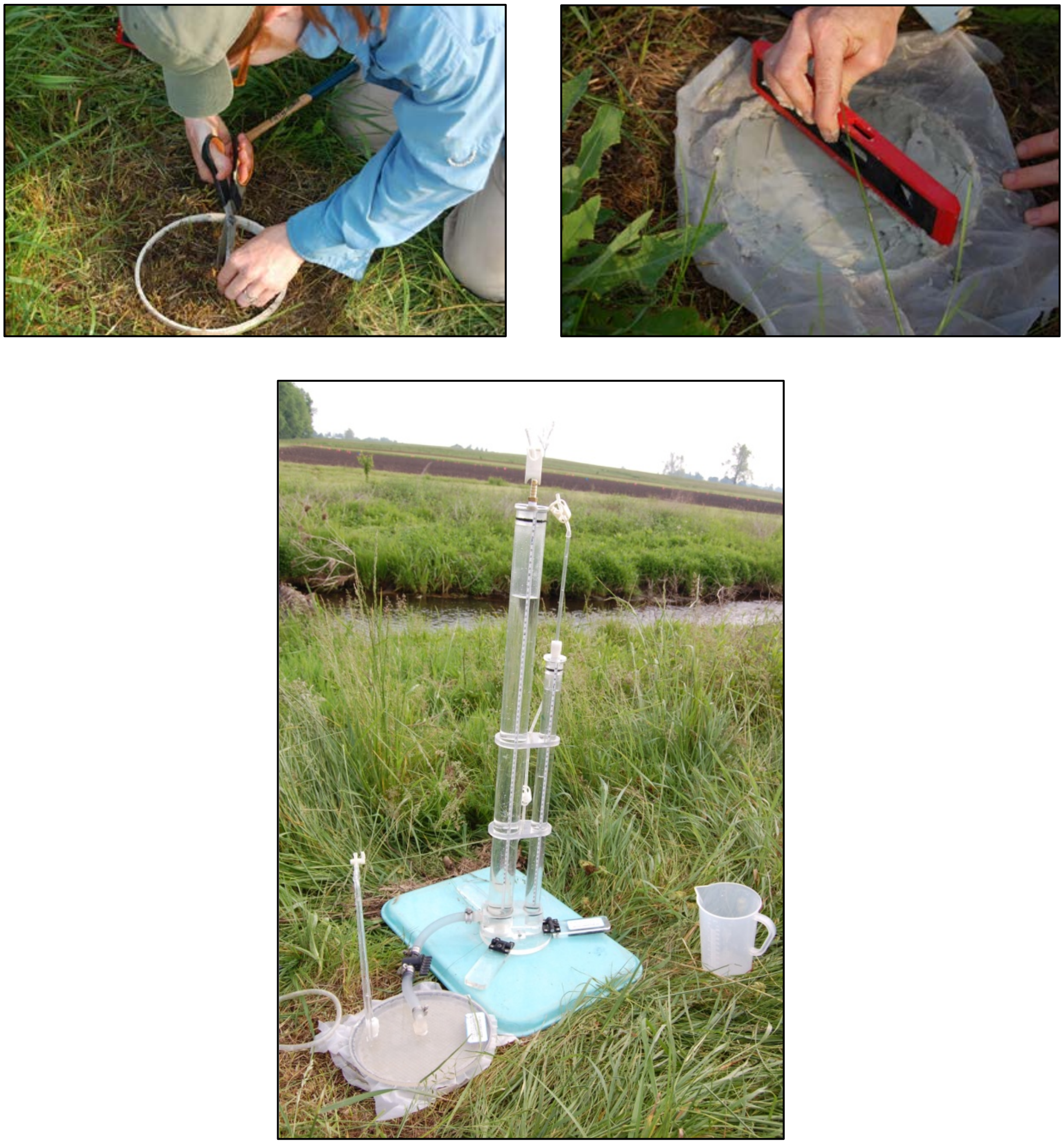


Figure 3.3. Infiltration data collection. (Clockwise from top left) Removing vegetation from the surface; applying a layer of fine sand to create an even contact surface; and tension infiltrometer in use.

$Q(\psi_1)$ and $Q(\psi_2)$ are measured, resulting in the following two equations with two unknowns:

$$Q(\psi_1)= \pi r^2 K_{sat} \exp(\alpha\ \psi_1)\ [1+(4/\pi r\alpha)]$$

and

$$Q(\psi_2)= \pi r^2 K_{sat} \exp(\alpha\ \psi_2)\ [1+(4/\pi r\alpha)]\ .$$

From these equations, α was determined by the following equation:

$$\alpha = (\ln[Q(\psi_2)/Q(\psi_1)])/(\psi_2-\psi_1)\ ,$$

and a value for α was determined for each pair of soil water potentials and water flux measurements. Respective coefficients for $K(\psi_1)$ and $K(\psi_2)$ were calculated by using α values in the equation:

$$K(\psi_{1/2})= Q(\psi_{1/2})/\ (\pi r^2\ [1+(4/\pi r\alpha)])\ .$$

Values for K_{sat} were determined by the following equation:

$$K_{sat}= K(\psi_{1/2})/\exp(\alpha\ \psi_{1/2})\ .$$

Laboratory

Roots were manually picked from each soil sample for 15 minutes, rinsed twice with deionized water, weighed, dried at 60°C for 24 hours, and weighed again to determine root biomass as described by Gift et al. (2010). Following root extraction, soil samples were air dried at room temperature (24°C) for approximately 5 days. Air dried soil was separated to determine aggregate size distribution by placing samples in a nest of sieves with openings of 4 mm, 2 mm, 1 mm, 0.25 mm, and 0.053 mm. Sieves were shaken at an amplitude of 2.5 cm for 1 minute. Mean weight diameter (MWD) was calculated by the methods described in Kemper and Rosenau (1986) using the equation

$$MWD = \sum_{i=1}^{n} \overline{x_i}\ w_i$$

where $\bar{x}_i$ is the mean diameter of each size fraction and w_i is the proportion of the total sample weight occurring in the corresponding size fraction.

After samples were sieved and weighed, one-half of each aggregate class size was combined to create a representative sample for chemical and physical property evaluation. Soil texture was determined using the micropipette method (Burt et al., 1993; Miller and Miller, 1987). Soil organic carbon was determined via LECO combustion (Nelson and Sommers, 1996) and is reported as % C. Wet aggregate stability was determined from 1-2 mm-size aggregates using the wet sieving procedure (Kemper and Rosenau, 1986).

Statistical Analysis

The Shapiro-Wilk test indicated a normal distribution of surface soil C prior to treatment implementation; therefore, subsequent statistical analyses were performed using original data sets without transformation with the exception of K_{sat}, which was log-transformed to simplify reporting. Analysis of variance (ANOVA) was conducted using the mixed procedure in SAS (Cary, NC) to determine statistical differences in response variables as a result of imposed treatments with year, depth, and treatment as main effects. An autoregressive covariance structure of sections was used in the ANOVA to account for underlying correlation considering that plots were organized in a linear repeating pattern. Each section contained one plot of each treatment in sequence, beginning with the most downstream plot. Means were compared among treatments and depths using the LSMEANS statement in the mixed procedure (SAS, 2010). Spearman correlation coefficients were used to examine relationships between response variables by treatment. Statistical differences were considered significant at $\alpha = 0.15$.

Spatial relationships of response variables were examined using the experimental semivariogram

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [A_i(x_i) - A_i(x_i + h)]^2$$

where $A_i x_i$ denotes observation A_i at location x_i and $N(h)$ denotes the number of observation pairs in lag class h (Wendroth et al., 2011).

Results and Discussion

This study examined root biomass, soil aggregate size distribution, aggregate stability, hydraulic conductivity, and soil C after imposing three mowing regimes and one native grass regime on a riparian buffer. The native grass treatment had no established above-ground vegetation at the time of sampling in 2011 due to the transition from existing grassland vegetation to a native grass-forb mix; above-ground vegetation was present in the native grass treatment plots by July 2011. Significance was determined at $\alpha = 0.15$ rather than $\alpha = 0.05$ because the risk of failure to detect real differences was greater than the risks associated with detecting differences that did not occur (Type II error) (Carmer and Walker, 1988). Failing to detect real differences in a conservation practice study such as the riparian buffers examined here could translate to a failure to utilize effective management strategies to protect water quality, whereas detecting differences that really do not occur will have minimal negative consequences.

Root Biomass

Root biomass values are reported as means from 32.4 cm^3 soil sample volume. Root biomass was compared among all treatments for both years of this study. With one exception, for both years and all treatments, root biomass was significantly greater ($p<0.15$) in surface soil samples (0-10 cm) than sub-surface depths (10-20 cm and 20-30 cm) (Table 3.2). These results are consistent with other studies reporting root biomass concentrated in the upper soil profile in riparian systems (Gift et al., 2010; Kiley and Schneider, 2005; Wynn et al., 2004). Wynn et al. (2004) reported significantly greater root length density (calculated as total length of all roots within a unit soil volume) in the top 30-cm of herbaceous buffers compared to forested buffers, but a greater root volume ratio (total volume of roots per unit soil volume) below 15-cm depth in forested buffers. The undisturbed sites in this study may be exhibiting characteristics of both herbaceous and forested buffers because both types of vegetation are present in these sites.

There were no statistically significant differences ($p<0.15$) in root biomass among imposed treatments below the 10-cm depth (Table 3.2) two years after treatment imposition; therefore, further discussion of root biomass will focus on the 0-10 cm sampling depth. After one year of treatment (2011), root biomass in intense mow, moderate mow, and no mow treatments were not significantly different ($p<0.15$) although the trend indicated an increase in root biomass with decreased mowing

Table 3.2. Dry root biomass for all study soils by treatment and depth. Std err = Standard error of the mean; n = number of samples. Values followed by a different lowercase letter within columns and depths are significantly different at α = 0.15.

Treatment	n	Mean (mg cm^{-3})	Std Err	Mean (mg cm^{-3})	Std Err
(0-10 cm)		*2011*		*2012*	
Intense Mow	50	8.32^{bc}	0.65	6.68^{bc}	0.43
Moderate Mow	50	8.66^{c}	0.78	8.15^{c}	0.67
No Mow	50	9.12^{c}	0.94	6.10^{ab}	0.49
Native Grass	50	6.96^{a}	0.71	4.57^{a}	0.72
Undisturbed	20	6.78^{ab}	0.90	10.88^{d}	1.61
(10-20 cm)					
Intense Mow	50	0.92^{a}	0.13	1.37^{a}	0.41
Moderate Mow	50	1.18^{a}	0.23	0.99^{a}	0.14
No Mow	50	0.82^{a}	0.12	0.79^{a}	0.16
Native Grass	50	0.57^{a}	0.09	1.09^{a}	0.32
Undisturbed	20	3.51^{b}	0.52	13.14^{b}	3.74
(20-30 cm)					
Intense Mow	44	0.27^{a}	0.07	0.33^{a}	0.07
Moderate Mow	43	0.76^{a}	0.29	0.36^{a}	0.06
No Mow	48	0.26^{a}	0.06	0.29^{a}	0.05
Native Grass	49	0.20^{a}	0.05	0.20^{a}	0.04
Undisturbed	20	2.36^{b}	0.41	5.61^{b}	1.30

frequency (Table 3.2, Figure 3.4). In 2012, the undisturbed sites had statistically greater ($p<0.15$) root biomass than all treatments (Table 3.2, Figure 3.4). For all imposed treatments, root biomass was greater in 2011 than 2012, with the intense mow, no mow and native grass transition treatments having significantly ($p<0.15$) less root biomass in 2012 (Figure 3.4).

The native grass treatment had less root biomass than all other imposed treatments in both years. Considering that native grass transition plots were planted with warm season grasses and forbs, two factors likely contributed to lower root biomass in these plots compared to the other mowing treatments: 1) native grass plots had been recently established (June 2011); and 2) root systems in the native plots were not in the same growth phase in 2012 compared to the other imposed treatments, which were dominated by cool season grasses and weeds.

Researchers have reported mixed results from the influence of mowing or clipping on root biomass. Kitchen et al. (2009) reported mowing significantly increased root biomass in the upper 10-cm of unburned tallgrass prairie compared to unmowed treatments, with little impact to roots in lower depths and no net effect on total root biomass. Todd et al. (1992) report decreased live root biomass of native grasses under mowing treatments to a height of 5-cm at 3-week and 6-week intervals; others report an increase in below-ground biomass of European lawn grasses in the second year of intense mowing (Dickinson and Polwart, 1982) similar to the intense mowing treatment in the present study. In a pasture study on Vancouver Island, Canada, Ziter and MacDougall (2013) reported a burst of short-lived roots in the upper 20-cm after

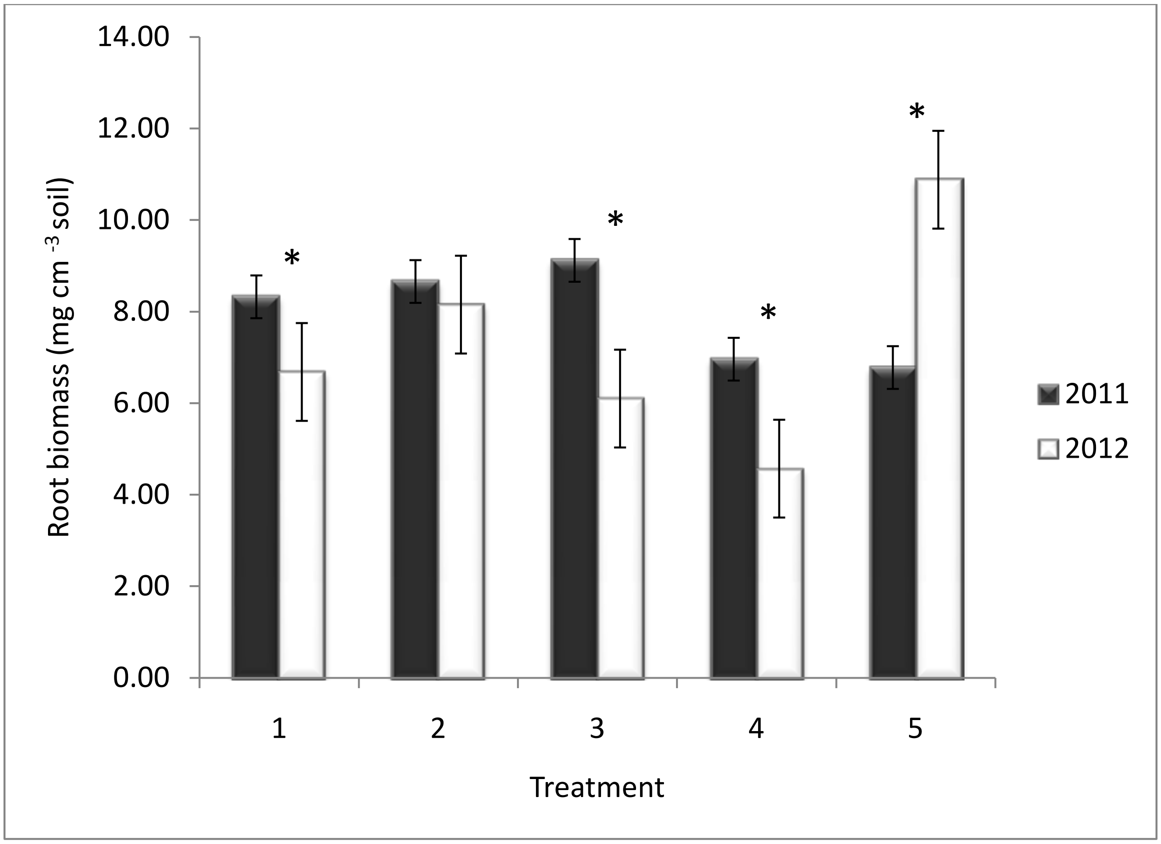


Figure 3.4. Root biomass at 0-10 cm depth by treatment. Treatments: 1=Intense Mow; 2=Moderate Mow; 3=No Mow; 4=Native Grass Transition; 5=Undisturbed. Standard error bars at 95% confidence intervals. Asterisks denote significant differences ($p<0.15$) between years within treatment.

clipping grasses to 5-cm (clippings removed); roots subsequently died during the remainder of the growing season and would have gone unnoticed without continuous sampling.

Although root biomass in the no mow treatment was significantly greater than the undisturbed site after one year of treatment (Table 3.2, Figure 3.4), the following year (2012) root biomass in the undisturbed site was statistically greater than ($p<0.15$) any mowing treatments. Chaieb et al. (1996) reported increased cuttings of perennial grasses resulted in a more superficial root system, concentrated in the upper 15-cm of soil, and found little difference in root systems of non-mowed plots and those mowed once. Although Kitchen et al. (2009) reported an increase in root biomass in the upper 10-cm of an annually mowed (with clippings removed) prairie, they reported a decrease in root biomass in a similar prairie that also received a prescribed burn treatment; mowing resulted in a concentration of roots in the upper 20-cm in both burned and unburned prairie.

Clippings were not removed from mowing treatments in the present study, but vegetation harvesting may play a role in root response to mowing. In a greenhouse clipping study, Johnston (1961) showed that as more aboveground vegetation was removed root growth correspondingly decreased; Neigebauer et al. (2000) found total rooting depth increasing linearly with increasing mowing height in a wildflower sod production study, with mowing (plant material removed) significantly reducing total root biomass. Dickinson and Polwart (1982) imply that terminating mowing affects grasslands in two ways: 1) less mowing depletes underground reserves previously stored in grass tillers and rhizomes as a means of recovery from defoliation, favoring an environment more suitable to forbs than grasses; and 2) surface accumulations of litter from unmown aboveground growth (and subsequent local shading) also favors forbs over grasses. This may explain the significant decrease in root biomass in the no mow treatments from 2011 to 2012; the lack of mowing may be inducing a successional transition from a grass dominated plant community to a forb-dominated community.

Due to the size and orientation of the study area, plots were mowed in a repetitive pattern perpendicular to the stream to avoid adjacent plots receiving a different mowing treatment. Visual observation indicated grass clippings of the intense mow treatments were consistently deposited in the same locations. This repetitive deposition of mowed material could have resulted in some areas of matted grass that was not clipped during the next mowing event. Furthermore, mowing and imposed compacting forces (e.g. equipment) have been shown to significantly increase bulk density (Carrow, 1980), especially in wheel tracks of turf grass studies (Flannagan and Bartlett, 1961). In crop fields, root growth has been shown to proliferate in the planted row, but be restricted by compaction within wheel-trafficked areas (Barnes et al., 1971). Compacted surface soils can restrict root growth (Carrow, 1980), decrease infiltration, and increase runoff (Barnes et al., 1971; Batey, 2009). In the present study, wheel traffic was not a consideration due to the mowing pattern (tractor was backed in and pulled forward so that the 2-m transect never received the heavy wheel traffic of the front of the tractor), but future studies may want to consider compaction effects on root biomass.

Seasonal differences in root biomass were not considered in this study, but may play an important role in assessing root systems in riparian buffers. Tufekcioglu et al. (1998) reported maximum live fine root biomass from August to October and minimum live fine root biomass in May in a multispecies buffer in Iowa. Kiley and Schneider (2005) found maximum root biomass during August in a forested buffer in New York. Root samples in this study were collected in late May-early June, and consequently may not have captured a representation of maximum root biomass.

Aggregates

Soil aggregation studies often look at long-term (more than six years) pasture or row crop land uses (Barto et al., 2010; Franzluebbers et al., 2000; Six et al., 2000) but changes in soil aggregate size class distribution can occur rapidly in restored stream corridors and quantifying these changes will be important in assessing stream restoration success (Handayani et al., 2008).

Soil aggregate size distribution and stability were compared among all treatments for both years of this study. Vegetation treatment had no statistically significant effect on MWD of soil aggregates at the 0-10 cm depth in 2011, although the trend indicated an increase in MWD with decreasing mowing intensity (Table 3.3, Figure 3.5); this trend was not evident in 2012 (Figure 3.6). MWD was significantly greater ($p<0.15$) under the no mow treatment compared to the intense mow treatment at the 10-20 cm depth in 2011; no other statistically significant differences occurred among treatments in the 10-20 cm depth for either treatment years. In contrast, Barto et al. (2010) found an increase in MWD with increasing land use intensity; specifically,

Table 3.3. Mean weight diameter of aggregates for all study soils by treatment and depth. Std err = Standard error of the mean; CV = Coefficient of variation (%); n = number of samples. Values within columns and depths followed by a different lowercase letter are significantly different at $\alpha = 0.15$.

Treatment	n	Mean (mm)	Std Err	CV	Mean (mm)	Std Err	CV
(0-10 cm)			*2011*			*2012*	
Intense Mow	50	5.76	0.22	27.0	4.90	0.15	21.9
Moderate Mow	50	5.64	0.23	28.3	4.72	0.09	13.7
No Mow	50	6.12	0.19	21.4	4.64	0.08	12.3
Native Grass	50	6.22	0.21	24.2	4.85	0.15	22.6
Undisturbed	20	6.27	0.29	20.4	4.52	0.06	6.4
(10-20 cm)							
Intense Mow	50	7.13^{a}	0.20	20.2	5.94	0.17	19.9
Moderate Mow	50	7.64^{ab}	0.19	17.8	5.99	0.15	17.9
No Mow	50	7.81^{b}	0.18	16.2	6.09	0.13	15.4
Native Grass	50	7.79^{ab}	0.22	20.4	6.00	0.15	17.7
Undisturbed	20	7.12^{ab}	0.43	27.0	5.03	0.07	6.3
(20-30 cm)							
Intense Mow	44	7.74^{a}	0.21	19.1	7.01^{b}	0.25	23.4
Moderate Mow	43	7.94^{a}	0.27	23.6	8.50^{c}	1.45	113
No Mow	48	8.60^{b}	0.22	18.1	7.17^{b}	0.22	21.1
Native Grass	49	8.32^{ab}	0.25	20.8	7.08^{b}	0.19	19.0
Undisturbed	20	8.24^{ab}	0.37	20.1	5.24^{a}	0.15	12.6

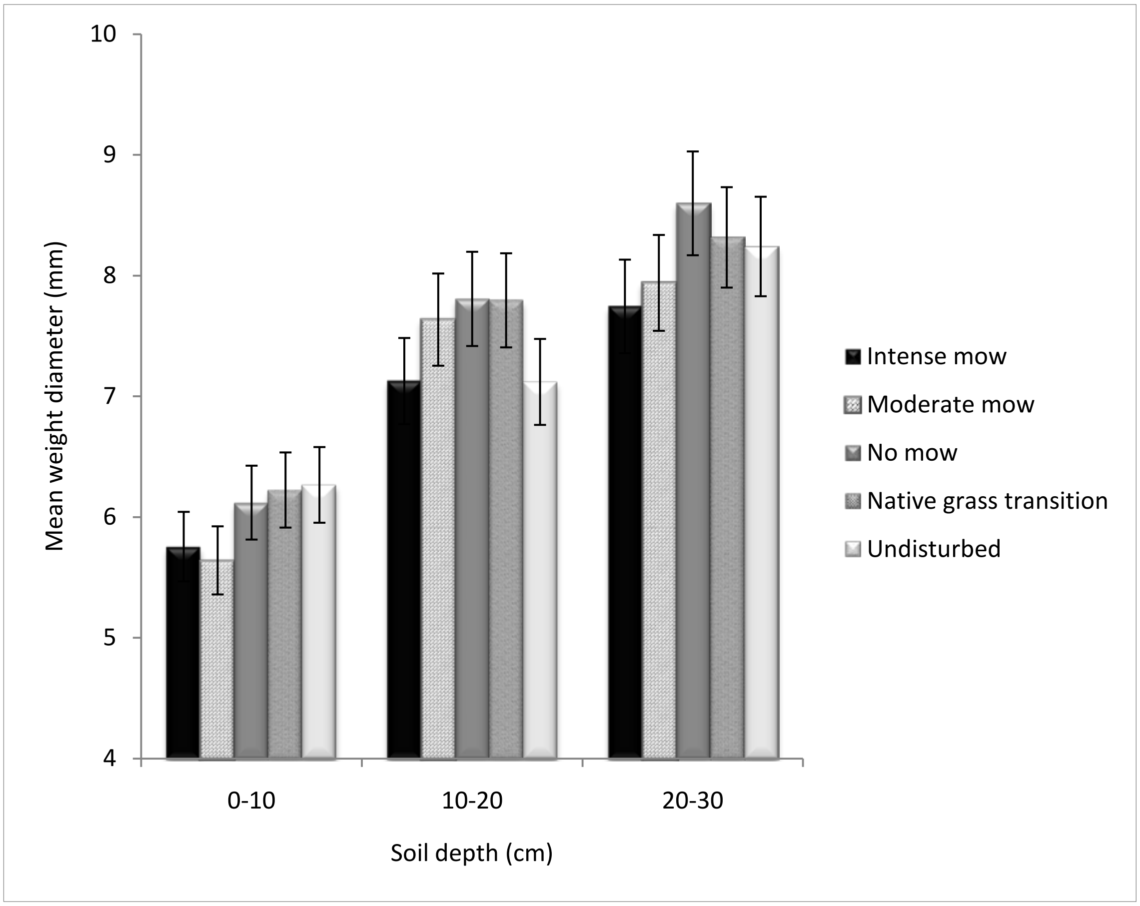


Figure 3.5. Mean weight diameter (MWD) of soil aggregates after one year of treatment (2011). Standard error bars at 95% confidence intervals.

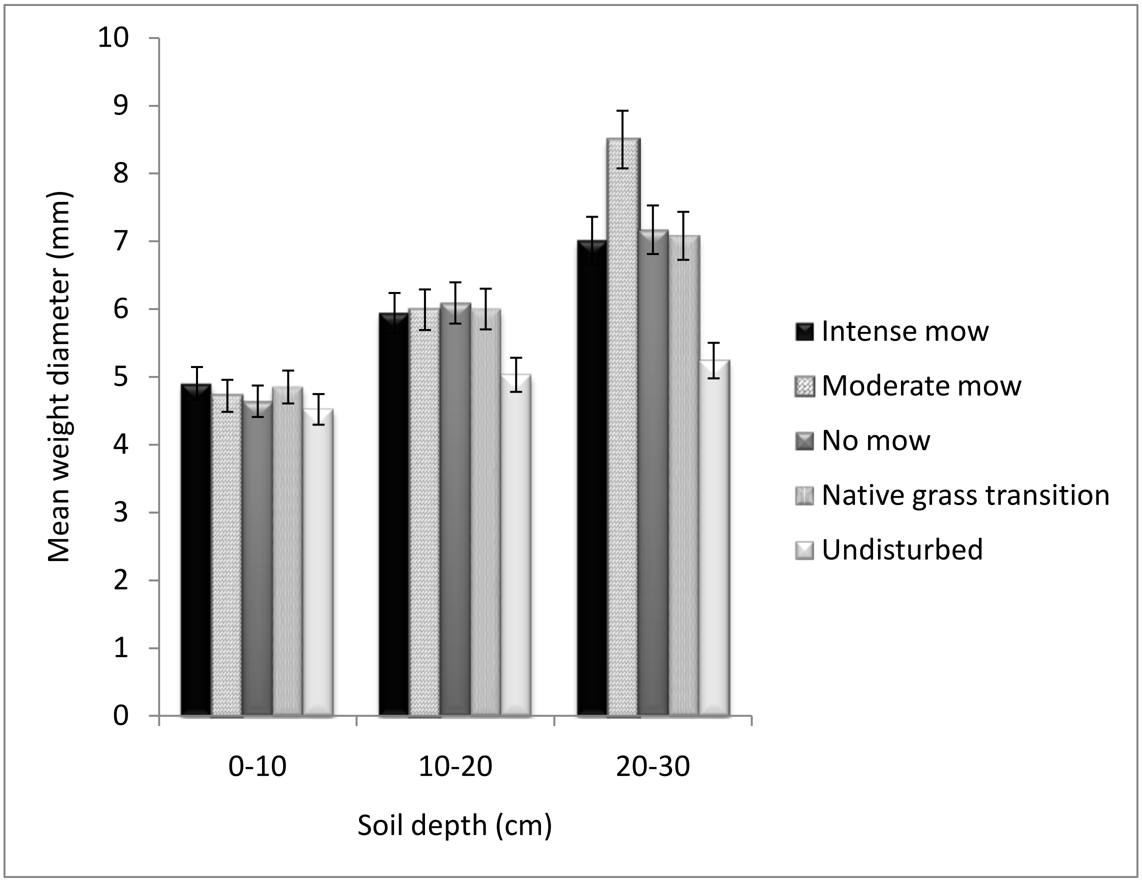


Figure 3.6. Mean weight diameter (MWD) of soil aggregates after two years of treatment (2012). Standard error bars at 95% confidence intervals.

mowing had a positive effect on MWD. In 2012, treatment had no statistically significant effect on MWD of soil aggregates at any sampled depth with one exception: at the 20-30 cm depth, the moderate mow treatment had a significantly higher ($p<0.15$) MWD than the other imposed treatments (Table 3.3).

Considering all treatments and both years, MWD significantly ($p<0.15$) increased with depth (Table 3.3), which is consistent with other studies (Franzluebbers et al., 2000; Sainju, 2006), and may be attributed to a slight increase in clay content with depth (Table 3.4). Considering all treatments and all depths, MWD in 2011 was significantly higher ($p<0.15$) than 2012. This response was not expected. Samples obtained in 2011 were collected following an extremely wet spring (Table 3.1). Some compaction of soil cores may have occurred during sample collection, resulting in artificially high MWD measurements. For all treatments in both study years, MWD fell within the 1-10 mm diameter range needed for crop growth (Tisdall and Oades, 1982). This range of aggregation provides sufficient pore space for infiltration and drainage, while retaining enough water for plant growth.

Comparing MWD values among studies is difficult due to various sieve sizes researchers may select for aggregate separation. Therefore, aggregates are often examined in three size classes: large macroaggregates (>2 mm), small macroaggregates (0.25-2 mm), and microaggregates (<0.25 mm) (Handayani et al., 2008; Kong et al., 2005; Six et al., 2004). Considering all depths and all treatments, large macroaggregates significantly decreased ($p<0.15$) from 2011 to 2012 while small macroaggregates significantly increased ($p<0.15$) (Table 3.5). Large macroaggregates increased with depth

Table 3.4. General soil properties of the study site (n = number of samples).

	n	% C	% Sand	% Silt	% Clay
Depth					
0-10 cm	40	4.9	35.4	50.2	14.4
10-20 cm	40	2.4	34.9	49.5	15.7
20-30 cm	37	2.0	32.5	52.6	14.9

Table 3.5. Percentage of aggregate size classes reported by treatment, year, and depth.

	Treatment									
	Intense Mow		Moderate Mow		No Mow		Native Grass		Undisturbed	
	2011	2012	2011	2012	2011	2012	2011	2012	2011	2012
Soil depth (cm)										
	% Large macroaggregates (>2mm)									
0-10	62.7	47.4	62.3	45.2	66.0	45.0	67.2	48.6	69.5	41.0
10-20	73.6	59.0	76.9	58.7	78.1	60.1	77.5	59.5	75.2	40.0
20-30	77.7	68.1	77.4	80.3	85.9	71.0	80.1	69.7	82.3	43.2
	% Small macroaggregates (0.25-2.00mm)									
0-10	32.6	41.3	32.4	43.8	29.6	44.3	28.7	42.0	27.2	47.3
10-20	23.0	34.5	19.9	34.5	19.2	33.9	19.5	34.4	21.6	48.1
20-30	19.0	27.4	18.6	26.4	15.7	25.1	17.1	26.0	15.4	45.0
	% Microaggregates (<0.25mm)									
0-10	4.5	10.0	4.8	10.8	3.9	10.6	4.1	9.2	2.4	12.2
10-20	2.9	6.4	2.9	6.6	2.4	5.8	2.6	5.9	2.1	12.1
20-30	3.0	4.3	3.8	4.6	2.6	3.7	2.5	4.2	1.7	10.6

in both years for all treatments, while small macroaggregates decreased with depth for all treatments. In 2011, the no mow treatment had significantly more ($p<0.15$) large macroaggregates than the intense and moderate mow treatments; the no mow treatment also had greater root biomass in the 0-10 cm depth in 2011 than both the intense and moderate mow treatments (Table 3.2), although the difference was not statistically significant. The native grass treatment had significantly more ($p<0.15$) large macroaggregates than the intense mow treatment in 2011. There were no statistically significant treatment effects on large macroaggregates in 2012. In 2011, the intense mow treatment had significantly more ($p<0.15$) small macroaggregates than the no mow and native grass treatments, and the moderate mow treatment had significantly more ($p<0.15$) small macroaggregates than the no mow treatment. There were no treatment effects on small macroaggregates in 2012.

Macroaggregate stability increases under pasture due to the presence of roots and hyphae, both of which may persist in the soil for years (Tisdall and Oades, 1982). For all treatments in both years, there was a larger proportion of macroaggregates than microaggregates. The mixed results seen in 2011 and lack of treatment effect seen in 2011 on large and small macroaggregates may be attributed to the historical grassland conditions in the study area providing adequate roots and hyphae that protect macroaggregates from disintegration regardless of imposed mowing treatment. The decrease in large macroaggregates from 2011 to 2012 may be the result of higher rainfall in 2011 (Table 3.1) causing wet soil conditions during sampling, and subsequent compaction creating an artificially large amount of large macroaggregates in 2011.

Six et al. (2004) suggest that macroaggregate turnover may occur as frequently as every five to twenty-seven days in agroecosystems because of seasonal dynamics; a representative quantity of macroaggregates may be difficult to capture with annual sampling. The aggregate class distributions for 2012 are close to the range reported by Handayani et al. (2011) for a nearby fescue pasture, although the percentage of large macroaggregates for both years of the present study is greater at all depths than the pasture study, suggesting that this riparian buffer may be reaching a maximum aggregation level.

A modification to the aggregate hierarchy concept presented by Tisdall and Oades (1982) suggests that transient binding agents holding together macroaggregates form a nucleus inside which microaggregates form (Oades, 1984); the transient binding agents subsequently decompose, leaving behind a microaggregate within a macroaggregate. Regardless of the mechanism or hierarchy of aggregate formation, both concepts support the importance of microaggregate formation. For all treatments and both years, microaggregates represented the smallest class size fraction and generally decreased with depth.

Considering all depths, microaggregates significantly increased ($p<0.15$) from 2011 to 2012 within all treatments (Table 3.5). In 2011, the moderate mow treatment contained significantly more ($p<0.15$) microaggregates than the no mow and native grass treatments. In 2012, the moderate mow treatment had significantly more ($p<0.15$) microaggregates than any other imposed treatment; all treatments contained significantly ($p<0.15$) fewer microaggregates than the undisturbed treatment. According to Oades' (1984) concept, this result indicates that the moderate mow treatment may have fewer transient binding agents (roots); however, this is not reflected by significantly lower root biomass in the moderate mow treatment (Table 3.2, Figure 3.5). In this case, microaggregate distribution is likely more a result of inherent mineral soil characteristics than imposed treatments.

The ability of soil aggregates to resist destruction by the disruptive force of water in the soil matrix may be an indication of the soil's ability to resist erosion. The proportion of water stable aggregates (WSA) has been associated with reduced soil disturbance and perennial vegetation and suggests the potential for soil water infiltration, soil sustainability, and soil and ecosystem functions (Paudel et al., 2011). Two years after establishment, imposed treatments had no significant effect on WSA at the 0-10 cm or 10-20 cm depths (Table 3.6). At the 20-30 cm depth, there was no imposed treatment effect on WSA in 2011, but in 2012 the no mow treatment had significantly lower WSA than the native grass treatment. The lack of treatment influence reflected in WSA is not surprising considering the high percentage of WSA in all sampled soils. The study site was historically (30+ years) mowed grassland, which has likely contributed to stable conditions creating high concentrations of WSA. In both study years, the WSA content was at least 95% in the surface soils and no less than 85% in the 10-20 cm depth. Other studies sampling to a depth of 20-cm have reported 78.4 and 78.3% WSA for grass buffers and agroforestry buffers, respectively (Paudel et al., 2011), and 84 and 85% for tall fescue pasture and hayed hybrid bermudagrass, respectively (Franzluebbers et al., 2000). Barto et al. (2010) also found a high abundance of WSA

Table 3.6. Percentage of water stable aggregates for all study soils by treatment and depth. Std err = Standard error of the mean; CV = Coefficient of variation; n = number of samples. Values within columns and depths followed by a different lowercase letter are significantly different at α = 0.15.

Treatment	n	Mean (%)	Std Err	CV	Mean (%)	Std Err	CV
(0-10 cm)			*2011*			*2012*	
Intense Mow	10	97.8	0.4	1.4	95.3	1.0	3.4
Moderate Mow	10	97.4	0.4	1.2	96.2	0.7	2.4
No Mow	10	97.5	0.4	1.4	95.2	1.0	3.2
Native Grass	10	98.3	0.3	0.9	96.4	0.7	2.1
Undisturbed	4	98.2	0.4	0.9	95.1	2.4	5.0
(10-20 cm)							
Intense Mow	10	92.5	0.9	3.1	85.0	4.4	16.3
Moderate Mow	10	89.2	2.7	9.5	87.1	1.6	5.7
No Mow	10	90.8	1.5	5.3	86.6	2.1	7.6
Native Grass	10	89.6	3.0	9.4	88.9	2.1	7.4
Undisturbed	4	87.8	3.8	8.6	89.6	3.2	7.1
(20-30 cm)							
Intense Mow	10	86.9^{b}	2.3	8.0	78.8^{ab}	6.8	24.3
Moderate Mow	10	86.1^{b}	2.6	8.9	79.6^{ab}	3.3	13.2
No Mow	10	88.4^{b}	2.2	7.0	76.0^{a}	2.9	12.1
Native Grass	10	88.0^{b}	2.9	9.9	83.9^{b}	4.0	14.9
Undisturbed	4	80.7^{a}	6.9	17.2	77.9^{ab}	5.8	15.0

(92%) in grassland soils of varying land use intensities, and Veum et al. (2012) demonstrated that vegetated filter strips induced positive changes to aggregate stability in a 10-year study.

All of the imposed treatments appear to maintain adequate soil aggregation for plant growth, infiltration, and drainage, indicating no negative consequences of reduced management (e.g. less frequent mowing) on soil structure in the short term.

Hydraulic Conductivity

The saturated hydraulic conductivity (K_{sat}) describes the maximum capacity of soils to conduct water and is primarily a function of pore size and distribution as influenced by soil structure (Ehlers et al., 1995; Lauren et al., 1988). Some researchers (Anderson et al., 2009; Bharati et al., 2002) have used field-based methods to measure infiltration (rings or permeameters) while others (Kumar et al., 2008; Zeleke and Si, 2005) employed laboratory-based soil core methods to determine hydraulic conductivity. Field methods were utilized in this study for multiple reasons: 1) the study site was easily accessible for field measurements; 2) core extraction can result in soil disturbance; and 3) disk permeameters have shown similar results to soil cores (White et al., 1992).

K_{sat} within treatments were not significantly different between 2011 and 2012; although no change in K_{sat} was seen from year 1 to year 2 following treatment, the data indicate consistency in field methodology over two sampling seasons. In 2011, the intense mow and no mow treatments had significantly higher ($p<0.15$) K_{sat} than the native grass treatment (Figure 3.7). The intense mow treatment had a higher K_{sat} than

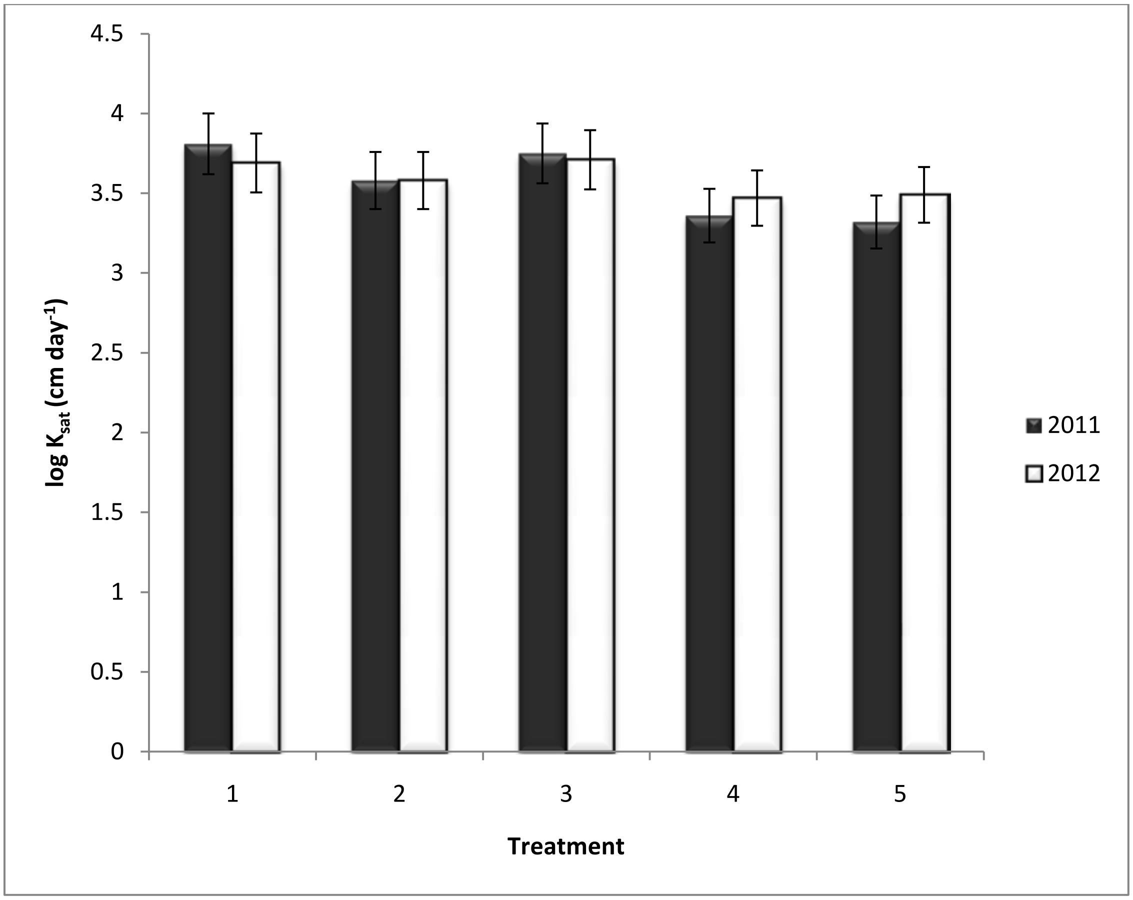


Figure 3.7. Saturated hydraulic conductivity by treatment. Treatments: 1=Intense Mow; 2=Moderate Mow; 3=No Mow; 4=Native Grass Transition; 5=Undisturbed. Standard error bars at 95% confidence intervals.

the no mow in 2011, but the two were nearly the same in 2012. This contrasts with results reported by Schacht et al. (1996), who found mowing treatments applied every fourth year reduced infiltration rates in a Nebraska grassland study. The moderate mow treatment had lower K_{sat} than the intense mow and no mow treatments both years. K_{sat} was nearly identical for the native grass transition treatment and the undisturbed treatment for both years.

K_{sat} values found in this study ranged 3.32-3.81 cm day^{-1} and were higher than those reported in other conservation buffer studies (data scaled and transformed). Comparatively, two Missouri studies reported K_{sat} values in agroforestry and grassed buffers in the 0.88-2.17 and 0.91-2.14 cm day^{-1} range, respectively (Anderson et al., 2009; Kumar et al., 2008), five years or more after establishment. Bharati et al. (2002) reported infiltration values of 3.05 cm day^{-1} and 2.86 cm day^{-1} for silver maple and grass filters, respectively, in a multi-species buffer after six growing seasons.

The lack of significant differences in K_{sat} among treatments is likely due to the immature age of imposed treatments and the previous management of the study site. Multi-species buffers have been shown to increase infiltration capacity by as much as five times compared to cultivated crop fields and pastures, but this was six growing seasons following establishment (Bharati et al., 2002). In contrast, Anderson et al. (2009) found no significant difference among K_{sat} rates for row crop, grass buffer, or agroforestry buffer after six years of treatment, suggesting significant changes in soil properties affecting infiltration may take long periods to develop. As previously stated, K_{sat} values found in this study were higher than those reported by Bharati et al. (2002), Kumar et al. (2008), and Anderson et al. (2009). This phenomenon presents two possible explanations: 1) the field method used in this study overestimated K_{sat} rates; or 2) previous management has created an environment of relatively high K_{sat} rates and considerable time must pass before a treatment effect is discerned.

While K_{sat} did not respond significantly to treatment implementation, neither did MWD, macroaggregates, or WSA, all of which influence hydraulic conductivity. Considering Lowrance et al. (2002)'s suggestion that infiltration may be a means of assessing riparian buffer function, and no adverse effect to K_{sat} was shown as a result of reduced management intensity or native grass transition treatment, this study indicates that practices other than routine mowing maintain riparian function in the short term.

Soil Carbon

Plant roots contribute carbon to riparian ecosystems (Dosskey et al., 2010) and soil organic carbon plays a key role in regulating denitrification (Tiedje, 1994). Based on a factor analysis of agricultural soils, Shukla et al. (2006) suggested that soil organic carbon should be used to monitor soil quality. Soil organic carbon (soil C) was measured in years one and two following treatment implementation (Table 3.7). In 2011, no treatment effects are seen in the 0-10 cm or 10-20 cm depth. In the 20-30 cm depth, the moderate mow treatment had significantly higher ($p<0.15$) soil C than the other imposed treatments. In 2012, soil C in the 0-10 cm depth increased with mowing intensity, with the intense mow treatment having significantly higher ($p<0.15$) soil C than the other mowing treatments; the native grass treatment had statistically similar ($p<0.15$) soil C as the intense mow treatment. The 2012 results support the findings of

Table 3.7. Soil carbon for all study soils by treatment and depth. Std err = Standard error of the mean; n = number of samples. Differences between years for given treatment and depth are reflected by p-value. Values followed by a different lowercase letter within columns and depths are significantly different at α = 0.15.

Treatment	n	Mean (%)	Std Err	Mean (%)	Std Err	p-value
(0-10 cm)		*2011*		*2012*		
Intense Mow	50	4.23	0.36	5.22[c]	0.20	0.0022
Moderate Mow	50	4.05	0.40	4.73[b]	0.14	0.0342
No Mow	50	3.93	0.31	4.36[a]	0.23	0.1707
Native Grass	50	4.23	0.28	4.84[bc]	0.13	0.0567
Undisturbed	20	4.10	0.12	4.44[ab]	0.08	0.5058
(10-20 cm)						
Intense Mow	50	2.08	0.20	2.14	0.12	0.8649
Moderate Mow	50	2.02	0.15	2.05	0.13	0.9225
No Mow	50	2.07	0.14	2.18	0.21	0.7342
Native Grass	50	1.95	0.07	2.15	0.07	0.5169
Undisturbed	20	2.48	0.15	2.50	0.02	0.9885
(20-30 cm)						
Intense Mow	44	1.96[a]	0.39	1.79	0.17	0.6023
Moderate Mow	43	2.45[b]	0.46	1.56	0.13	0.0056
No Mow	48	2.08[a]	0.35	1.72	0.14	0.2698
Native Grass	49	1.89[a]	0.35	1.73	0.06	0.6052
Undisturbed	20	1.89[ab]	0.10	2.09	0.05	0.7899

Ziter and MacDougall (2013), who saw a burst of root growth as a response to defoliation of pasture grasses. Although the long-term effects of defoliation on soil C were not measured, they reported root and shoot tissue production, mortality, and chemistry trends that would increase soil C in the short term in response to clipping. Shahzad et al. (2012), however, saw no difference in soil C among clipped, unclipped, and bare soil treatments although they did find clipping reduced C mineralization among seven grass species in a container study.

There were no significant treatment effects on soil C below 10-cm in 2012. Soil C increased from 2011 to 2012 in both the 0-10 cm and 10-20 cm depths; the increase was significant ($p<0.15$) at the surface depth in the intense mow, moderate mow, and native grass treatments. This increase in soil C over the short term may be a result of root growth and subsequent mortality stimulated by mowing. Soil C decreased from 2011 to 2012 in the 20-30 cm depth; the decrease was significant ($p<0.15$) in the moderate mow treatment.

Interactions

The relationships between soil properties were investigated by treatment in the 0-10 cm depth using Spearman correlation coefficients (Table 3.8). There was a significantly ($p<0.15$) negative correlation between MWD and root biomass in both the moderate and no mow treatments, as well as a significantly ($p<0.15$) negative correlation between WSA and root biomass in the intense mow treatment. A significantly ($p<0.15$) negative correlation was found between MWD and soil C in the moderate mow and native grass treatments.

Table 3.8. Correlation matrix for 0-10 cm depth after two years of treatment (2012).

		Roots	MWD	WSA	K_{sat}	C
	Roots		0.18	-0.52*	0.20	0.13
	MWD	0.18		-0.27	-0.48	-0.17
Intense Mow	WSA	-0.52*	-0.27		-0.24	0.00
	K_{sat}	0.20	-0.48	-0.24		0.45
	C	0.13	-0.17	0.00	0.45	
	Roots		-0.29*	-0.28	-0.21	0.18
	MWD	-0.29*		0.35	-0.15	-0.48*
Moderate Mow	WSA	-0.28	0.35		0.38	0.19
	K_{sat}	-0.21	-0.15	0.38		-0.21
	C	0.18	-0.48*	0.19	-0.21	
	Roots		-0.22*	0.27	0.31	0.05
	MWD	-0.22*		-0.33	-0.43	-0.07
No Mow	WSA	0.27	-0.33		0.33	0.40
	K_{sat}	0.31	-0.43	0.33		0.05
	C	0.05	-0.07	0.40	0.05	
	Roots		-0.15	-0.14	-0.16	0.35*
	MWD	-0.15		0.15	0.65*	-0.28*
Native Grass	WSA	-0.14	0.15		-0.05	0.50*
	K_{sat}	-0.16	0.65*	-0.05		-0.54*
	C	0.35*	-0.28*	0.50*	-0.54*	
	Roots		0.16	-0.40	-0.80	0.01
	MWD	0.16		0.40	0.80	-0.33
Undisturbed	WSA	-0.40	0.40		0.20	1.00*
	K_{sat}	-0.80	0.80	0.20		0.20
	C	0.01	-0.33	1.00*	0.20	

*Correlation is significant at 0.15 level; Roots=root biomass (mg cm^{-3}); MWD=mean weight diameter (mm); WSA=water stable aggregates (%); K_{sat} (cm day^{-1}); C=carbon (%).

These negative relationships are somewhat surprising considering the well documented connection of soil biology (e.g. roots, hyphae, and organic matter) to soil structure (Carter, 2004; Oades, 1993; Six et al., 2004; Tisdall and Oades, 1982). Other researchers (Barto et al., 2010) have reported no organic C effect on MWD.

In contrast to the mowing treatments, root biomass correlated significantly ($p<0.15$) with soil C in the native grass transition treatment. Because roots are a key mechanism for building soil C, it is expected that root biomass and soil C would have a strong positive relationship, as seen by Gift et al. (2010), regardless of treatment. K_{sat} correlated positively ($p<0.15$) with MWD but negatively ($p<0.15$) with soil C in the native grass transition treatments. K_{sat} was not significantly correlated with root biomass in any imposed treatment, which matches a similar study by Halabuk (2006) investigating wet meadow vegetation influences on K_{sat}, although preferential flow through macropores is often attributed to root activity (Dosskey et al., 2010). Soil C in the native grass treatment was positively correlated ($p<0.15$) with WSA. In the undisturbed sites soil C and WSA were strongly correlated ($p<0.15$), an indication that soil C may be the driving factor of aggregate stability in the undisturbed sites. Veum et al. (2012) also found WSA moderately correlated with soil C in the surface layer of vegetated filter strips and no-till corn-soybean production.

Spatial Variability

The spatial variability in the study area was investigated to identify potential soil property responses to imposed treatments that were undetectable in the overall comparison of means. Furthermore, this is an additional effort to more clearly define spatial relationships described in Chapter 2. Root biomass along the sampling transect at all depths is shown for 2011 and 2012 (Figures 3.8a, 3.8b, 3.9a, and 3.9b). Root biomass is higher in the surface depth (0-10 cm) than the other depths during both treatment years, as was seen in the previous comparison of means. No obvious treatment response is evident in the raw data for either treatment years at any depth. Experimental semivariograms of root biomass were calculated from 200 data points taken at 2-m sampling intervals for each treatment year (Figures 3.10, 3.11, and 3.12). Similar cycles of semivariance occur in the 0-10 cm depth for both treatment years, such that cycles of increasing semivariance occur at lag distances of approximately 1-35 m and 35-60 m, followed by cycles of decreasing semivariance at lag distances of 60-90 m and 90-130 m (Figure 3.10). Beyond a lag distance of 130-m the semivariance cycles differ between the treatment years. This indicates that spatial variability occurs at a range of approximately 30-40 m, which corresponds to one nest (one plot of each experimental treatment) of experimental plots. Semivariance at the 10-20 and 20-30 cm depths is an order of magnitude lower than the 0-10 cm depth for both treatment years. This is likely attributable to the sharp decrease in overall root biomass at depths below 10-cm. Both treatment years exhibit similar cycles of semivariance at the 10-20 cm depth up to the 115-m lag distance, with one cycle occurring at approximately 1-80 m and another at 80-115 m; cycles beyond the 115-m lag distance differ for the two treatment years (Figure 3.11). Semivariance occurred at a lower magnitude in 2011 than 2012. The cycles suggest that spatial variability occurs at both long (80-m) and short (35-m) ranges. The long ranges may be indicative of underlying topography while the short

Figure 3.8a. Raw root biomass after one year of treatment (2011). Five original data points are shown per plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

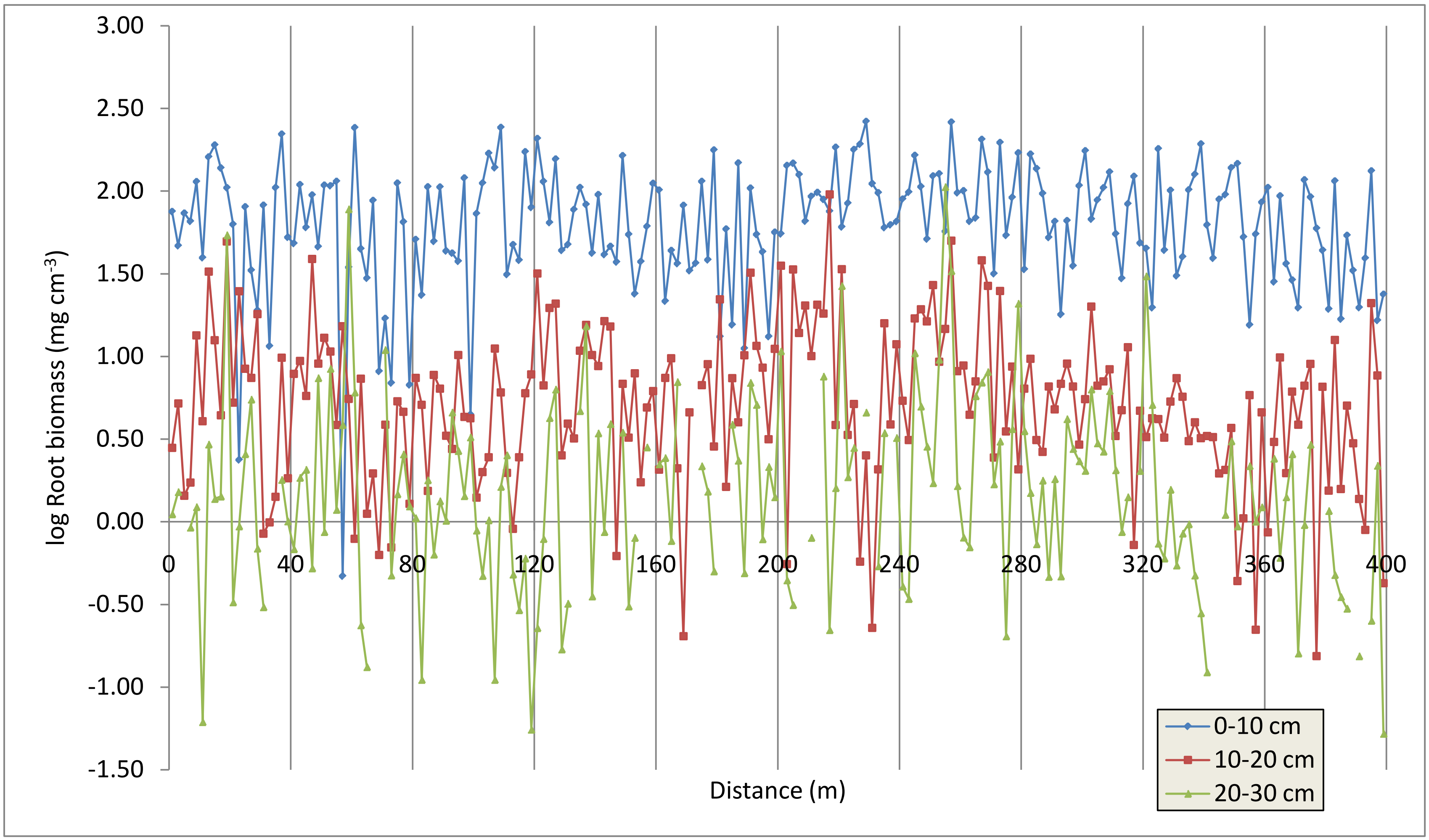


Figure 3.8b. Mean root biomass after one year of treatment (2011). Data points represent one treatment plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

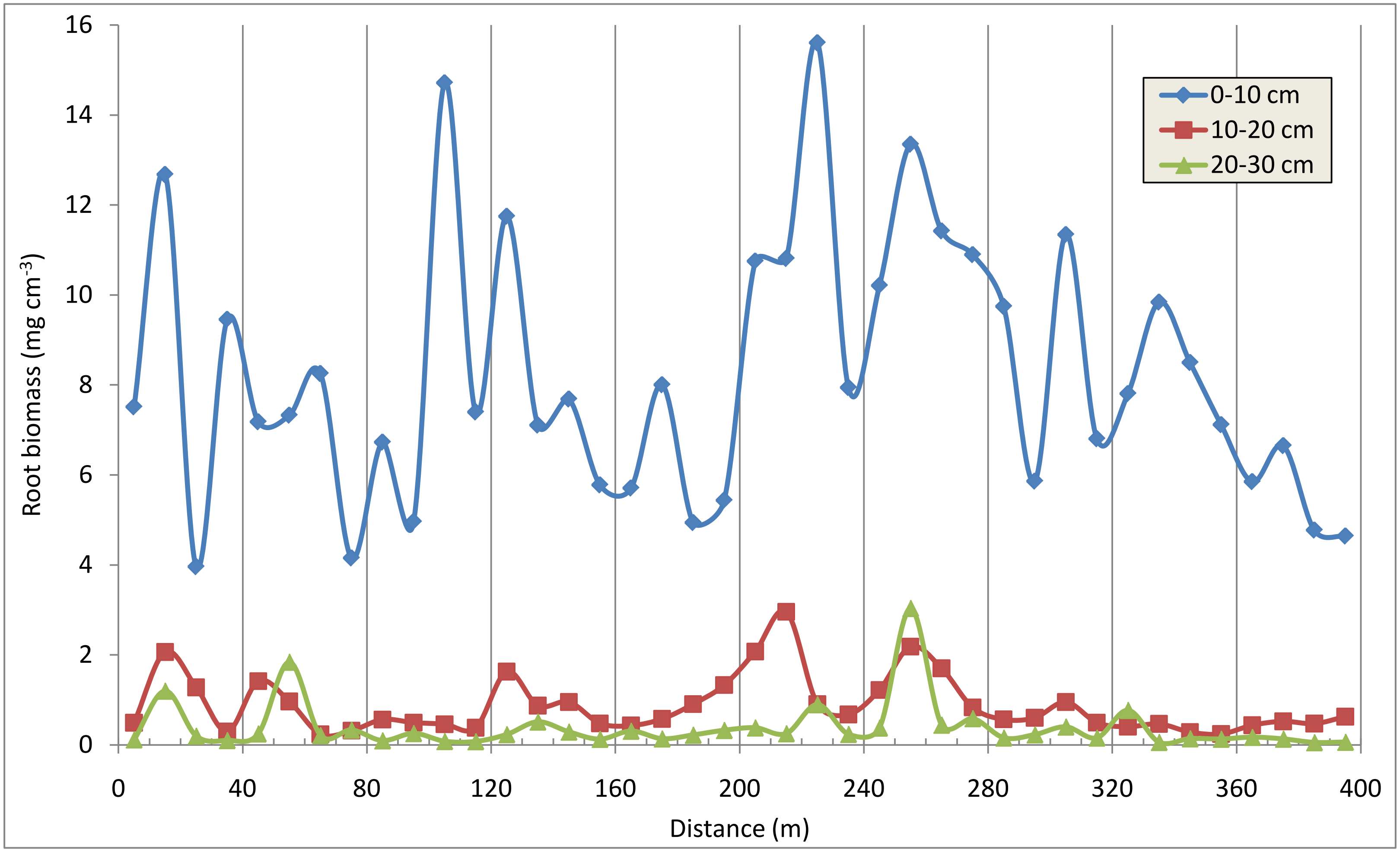


Figure 3.9a. Raw root biomass after two years of treatment (2012). Five original data points are shown per plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

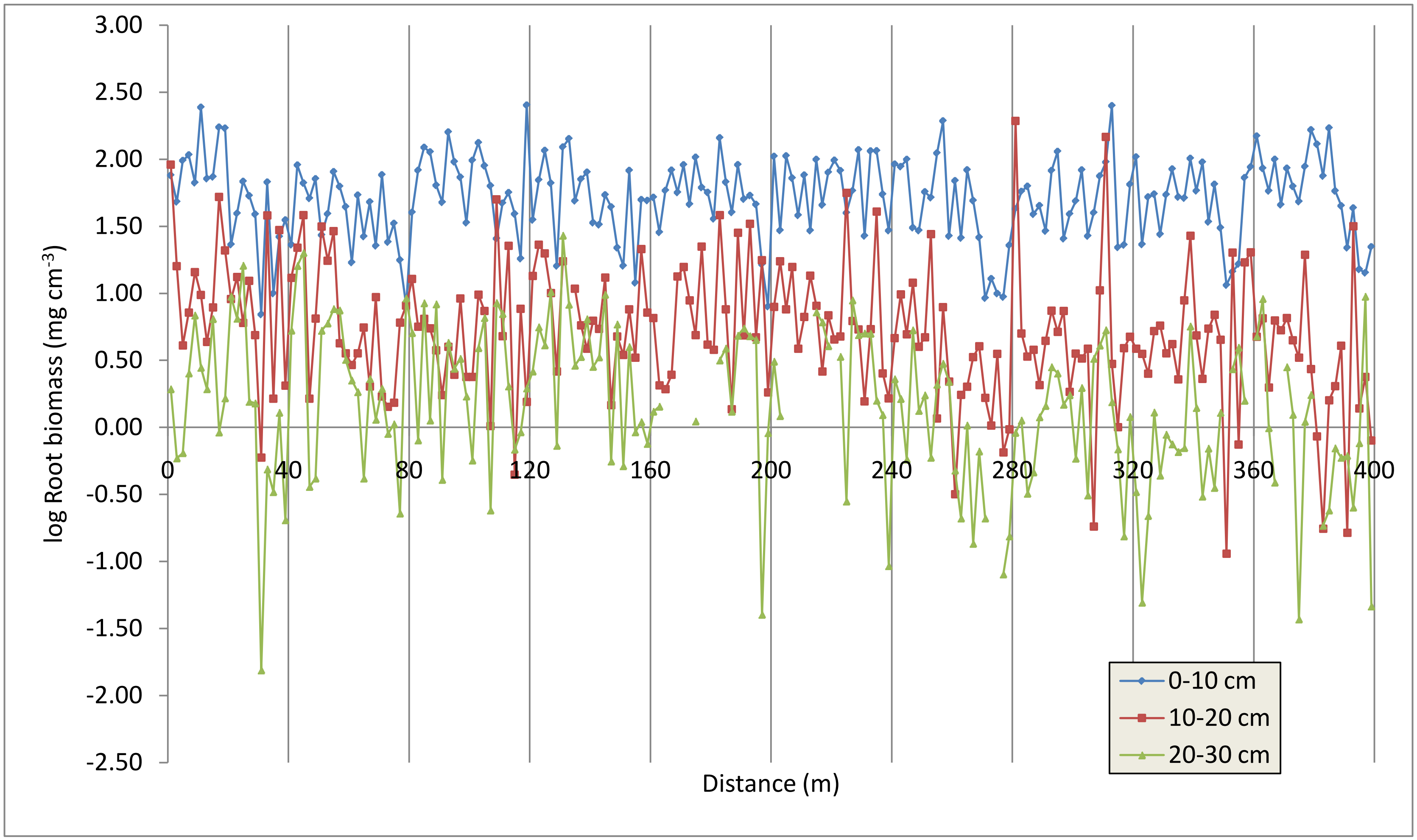


Figure 3.9b. Mean root biomass after two years of treatment (2012). Data points represent one treatment plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

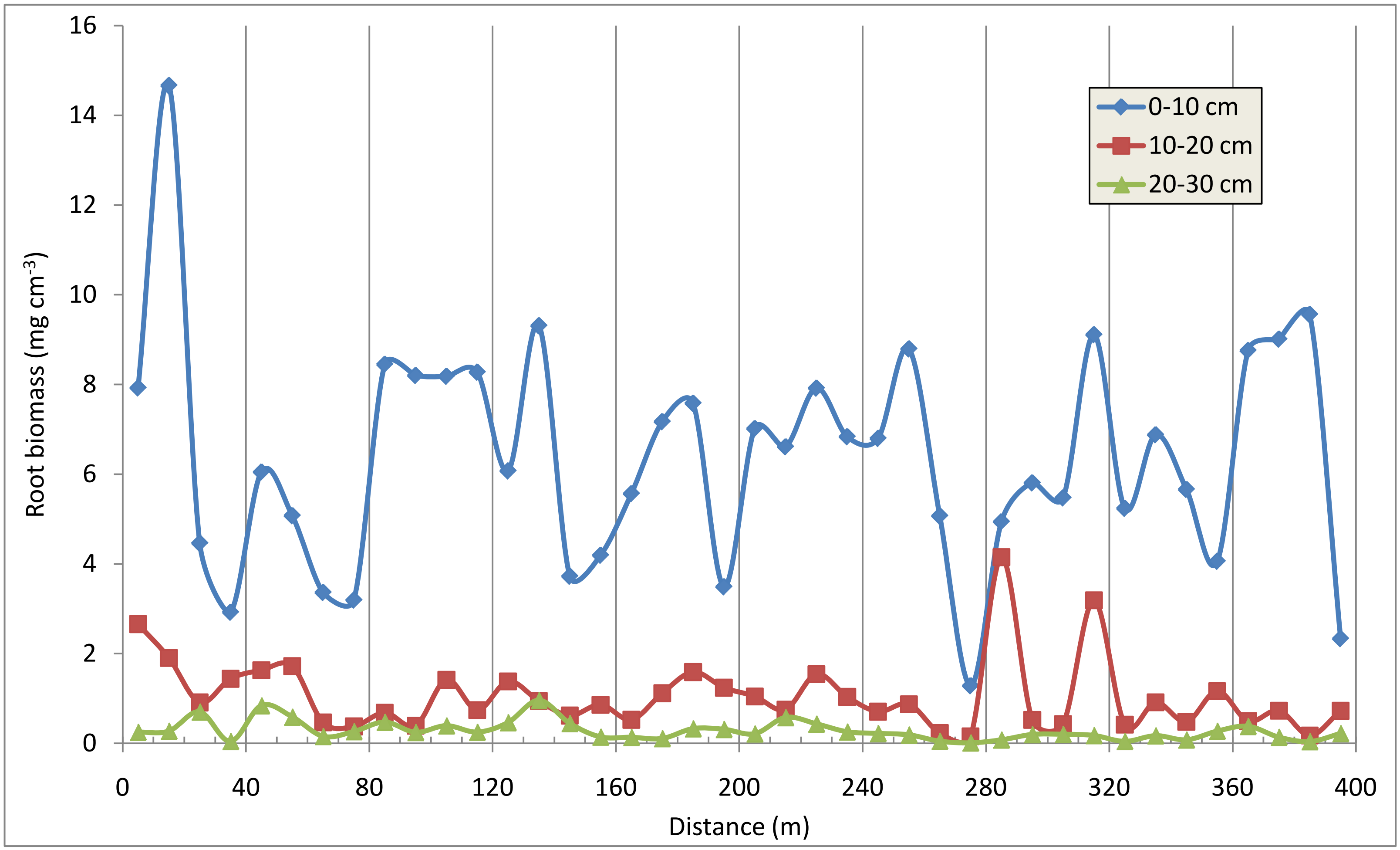


Figure 3.10. Experimental semivariograms for root biomass for both treatment years, 0-10 cm depth.

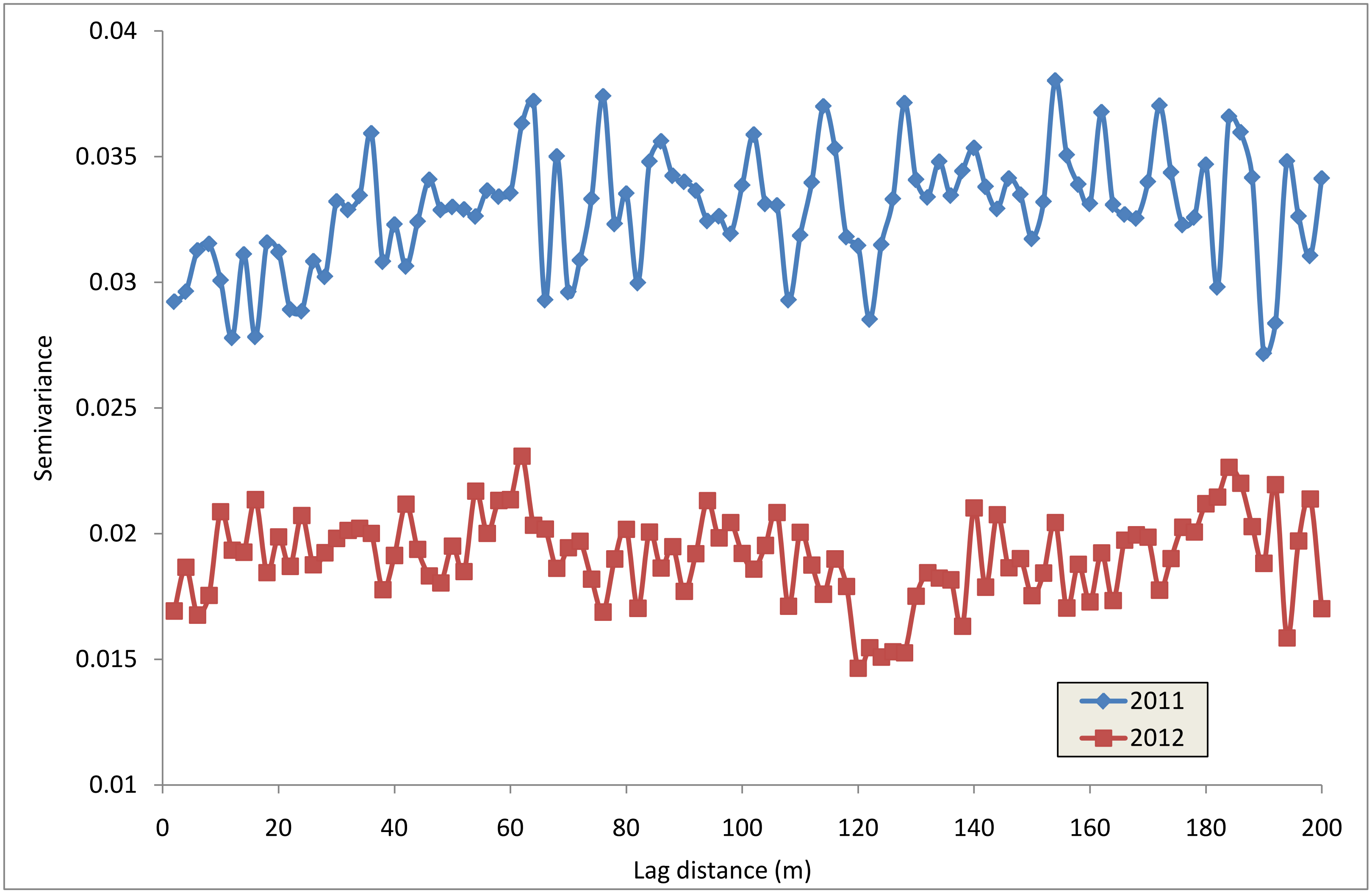


Figure 3.11. Experimental semivariograms for root biomass for both treatment years, 10-20 cm depth.

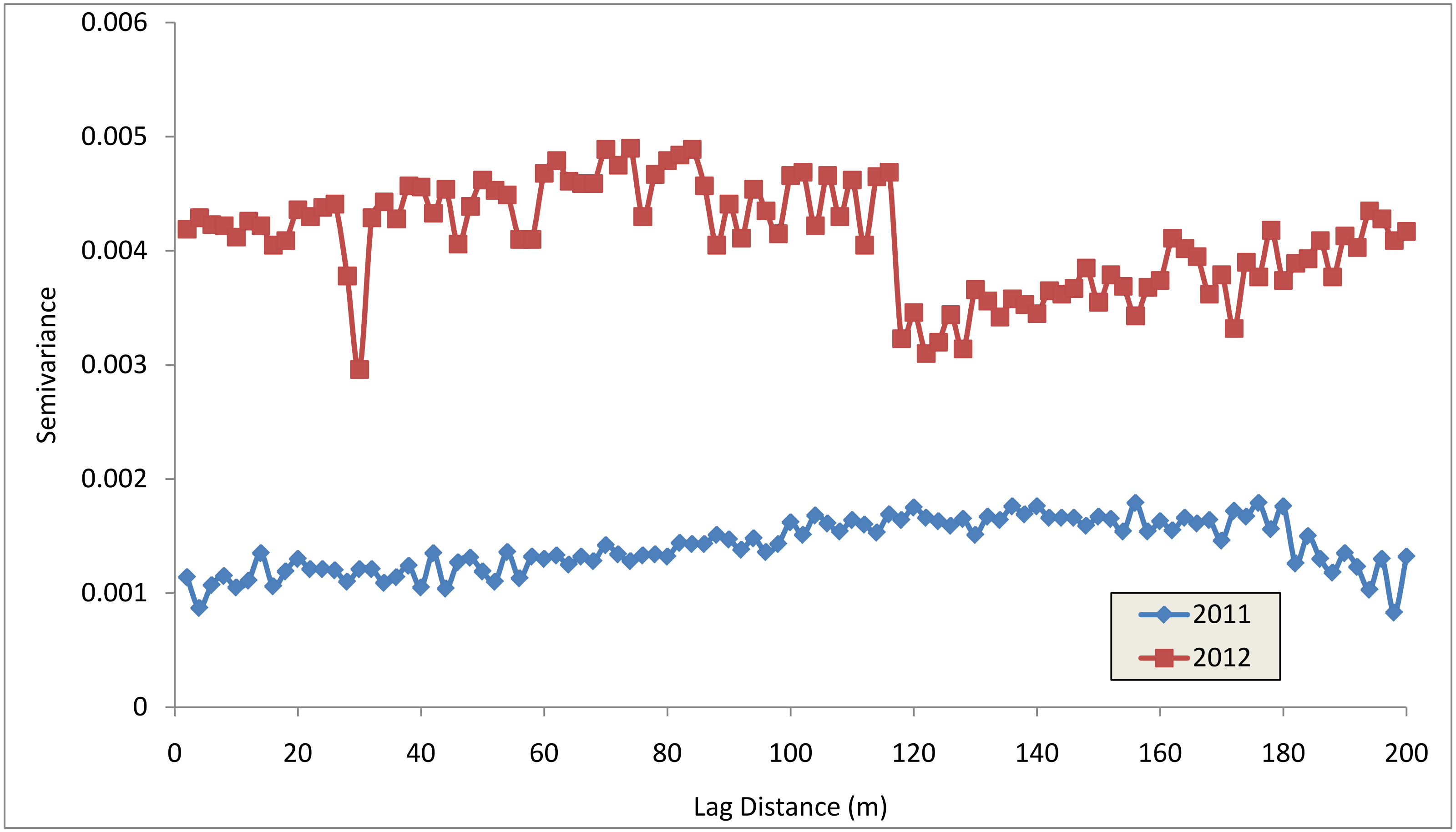
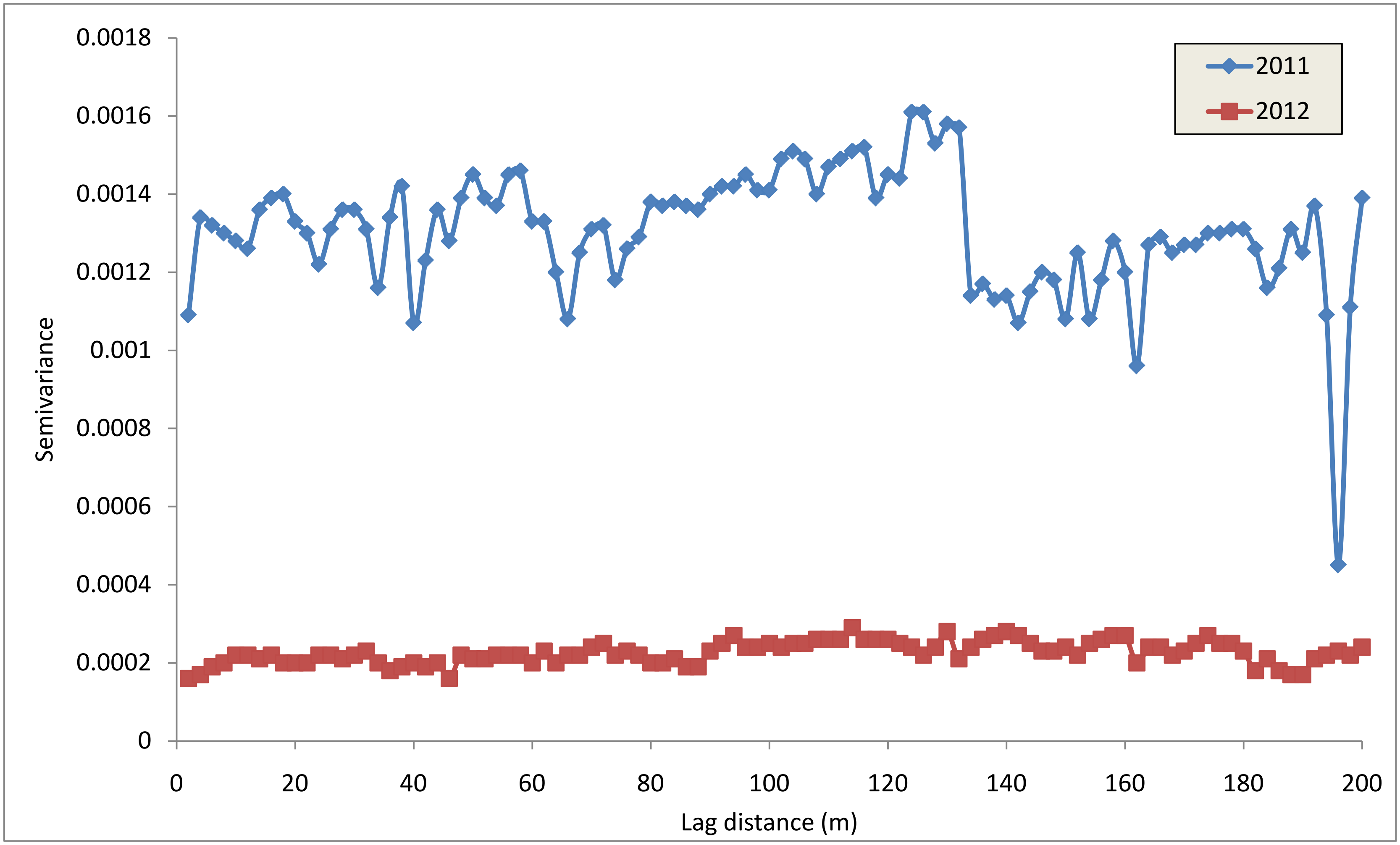


Figure 3.12. Experimental semivariograms for root biomass for both treatment years, 20-30 cm depth.



ranges may be indicative of imposed management. At the 20-30 cm depth, 2011 data show cycles of increasing semivariance at approximate lag distances of 1-60 m, 60-130 m, and 130-195 m. The 2012 data indicate overall less variability than 2011, with shorter cycles of semivariance at a lower magnitude than 2011. These data indicate a spatial variability range of approximately 60-m at the 20-30 m depth.

MWD at all sampling depths is shown for 2011 and 2012 (Figures 3.13a, 3.13b, 3.14a, and 3.14b). All depths have a similar spatial pattern, but no obvious treatment response is manifested in MWD. Experimental semivariograms of MWD were calculated from 200 data points taken at 2-m sampling intervals for each treatment year (Figures 3.15, 3.16, and 3.17). At the surface depth (0-10 cm), the nugget variance, represented by the point at which the semivariogram crosses the y-axis, is low for both treatment years (Figure 3.15), indicating low local variation compared to the overall variation in MWD. Multiple scales of variation are evident in 2011. The first small plateau at a lag distance of approximately 35-m indicates small scale variation while the second plateau at a lag distance of approximately 115-m represents a larger scale variation. Multiple scales of variation are also evident in 2012, but at a smaller magnitude. The depicted variation indicates spatial processes over the distance of the experimental plots and a further indication that the variation in MWD does not randomly occur. A similar pattern of variation occurred in the 10-20 cm depth (Figure 3.16) in 2011, but not in 2012. The 20-30 cm depth indicates more variation in 2012 than 2011 (Figure 3.17), which is opposite the trends in the other sampling depths. Furthermore, cycles in semivariance

Figure 3.13a. Raw mean weight diameter after one year of treatment (2011). Five original data points are shown per plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

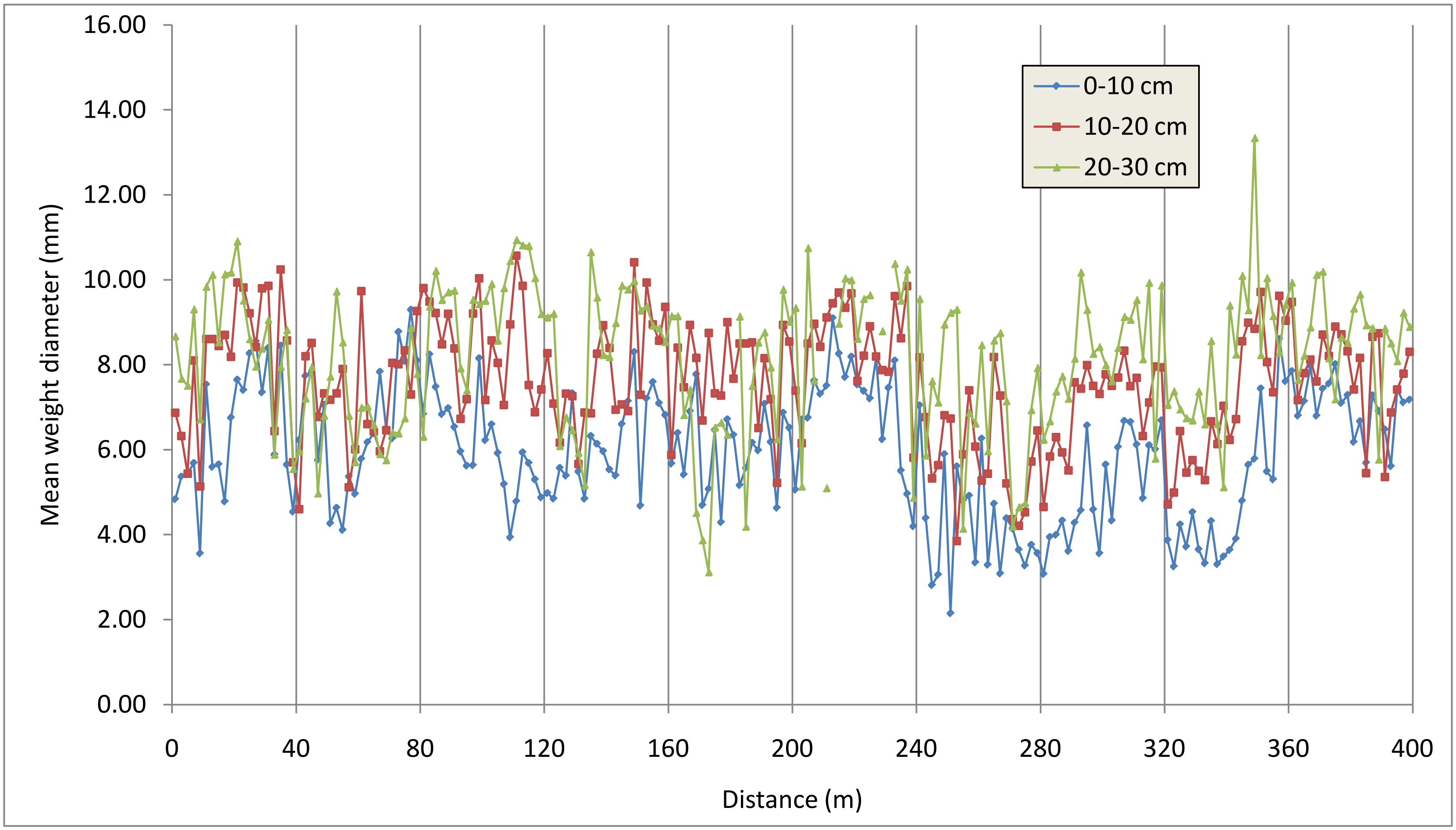


Figure 3.13b. Average mean weight diameter after one year of treatment (2011). Data points represent one treatment plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

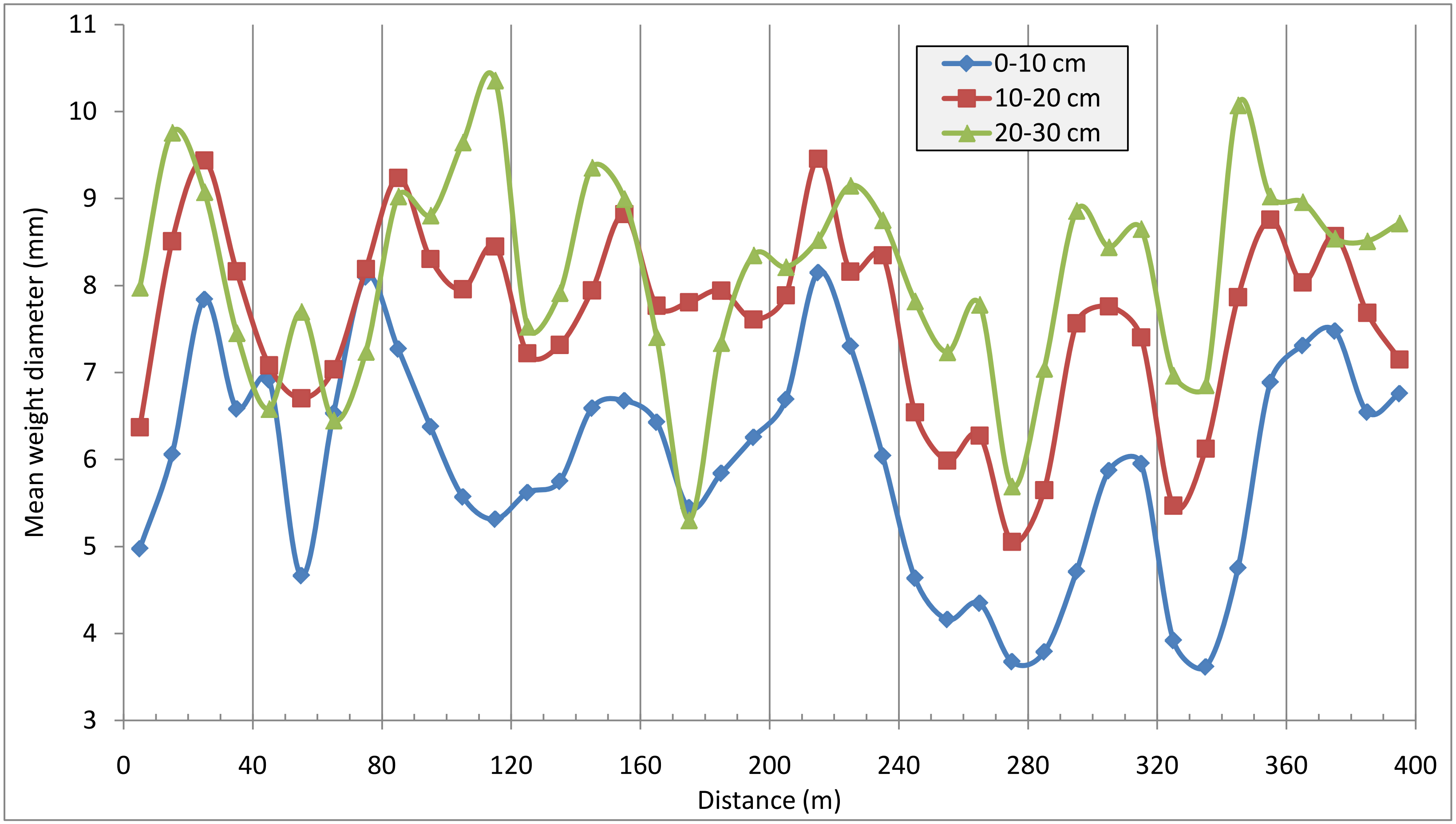


Figure 3.14a. Raw mean weight diameter after two years of treatment (2012). Five original data points are shown per plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

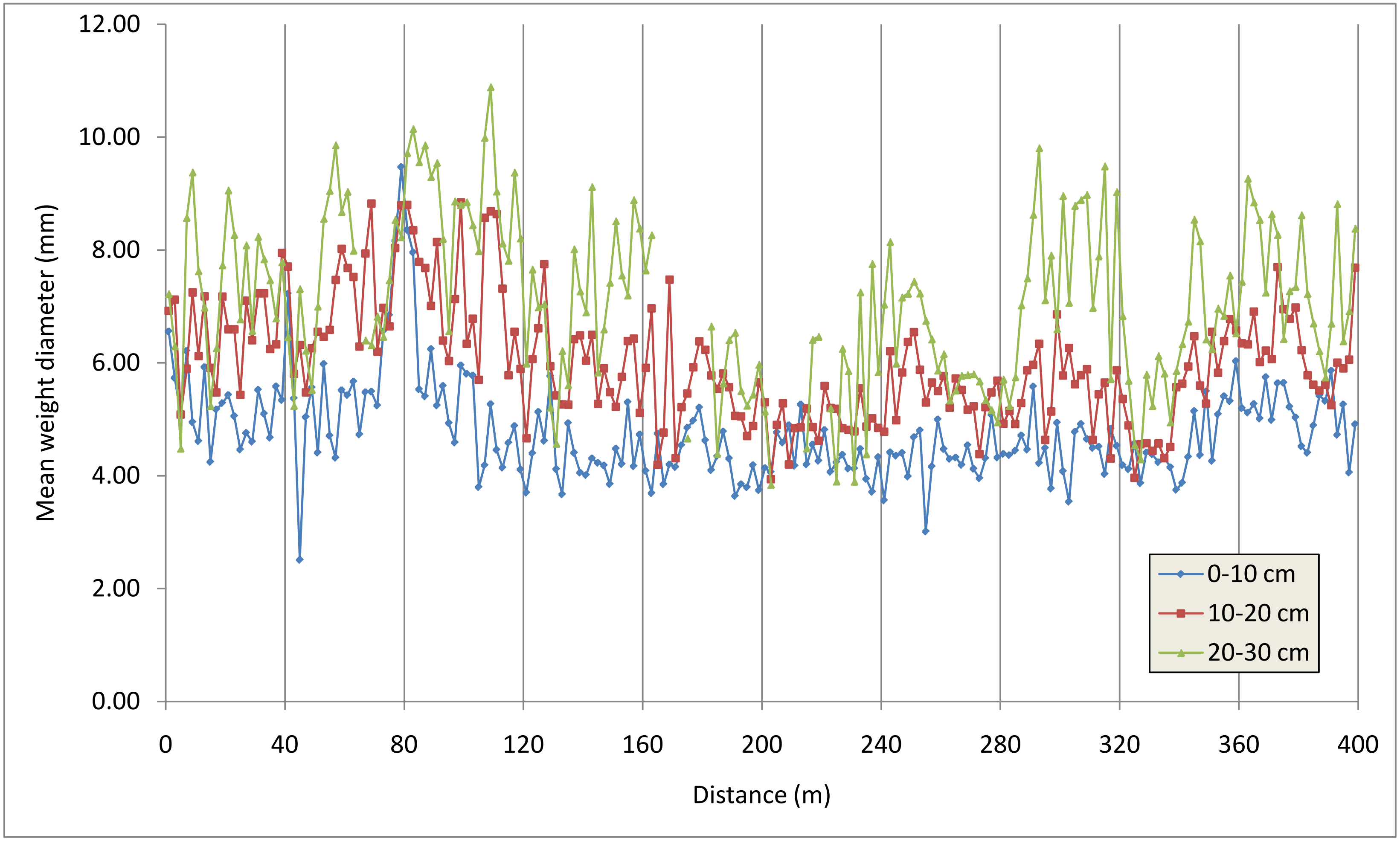


Figure 3.14b. Average mean weight diameter after two years of treatment (2012). Data points represent one treatment plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

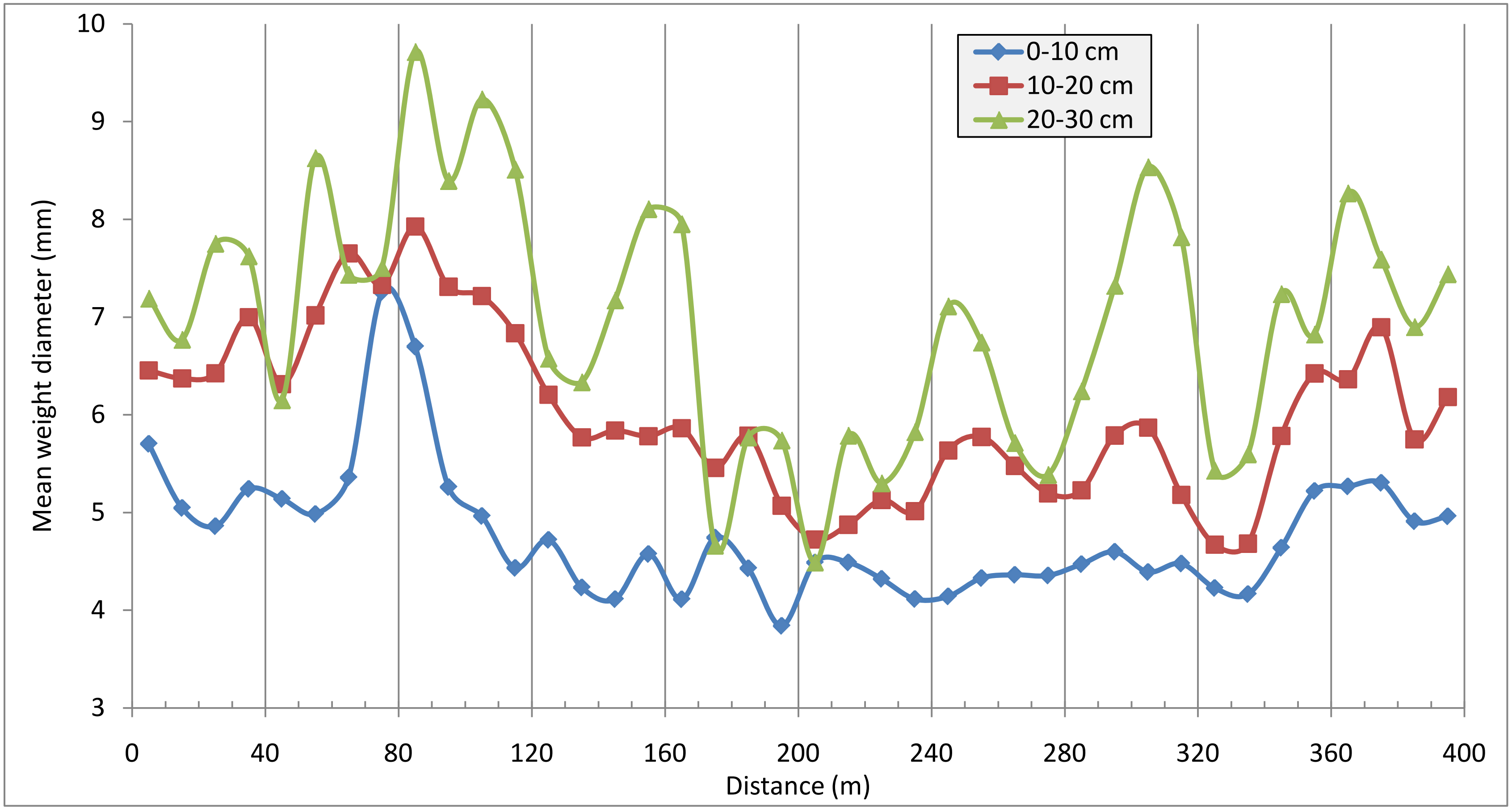


Figure 3.15. Experimental semivariograms for MWD for both treatment years, 0-10 cm depth.

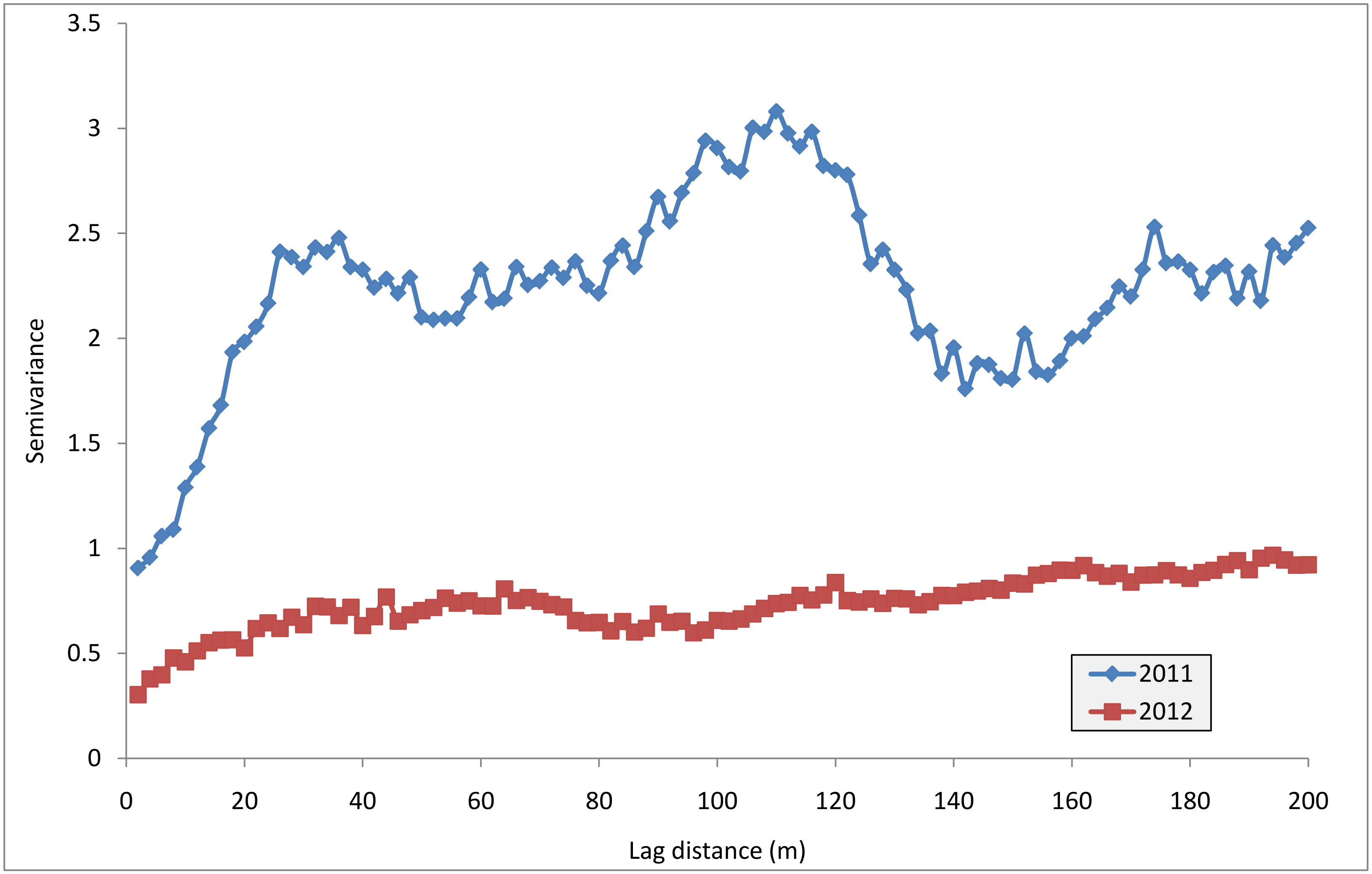


Figure 3.16. Experimental semivariograms for MWD for both treatment years, 10-20 cm depth.

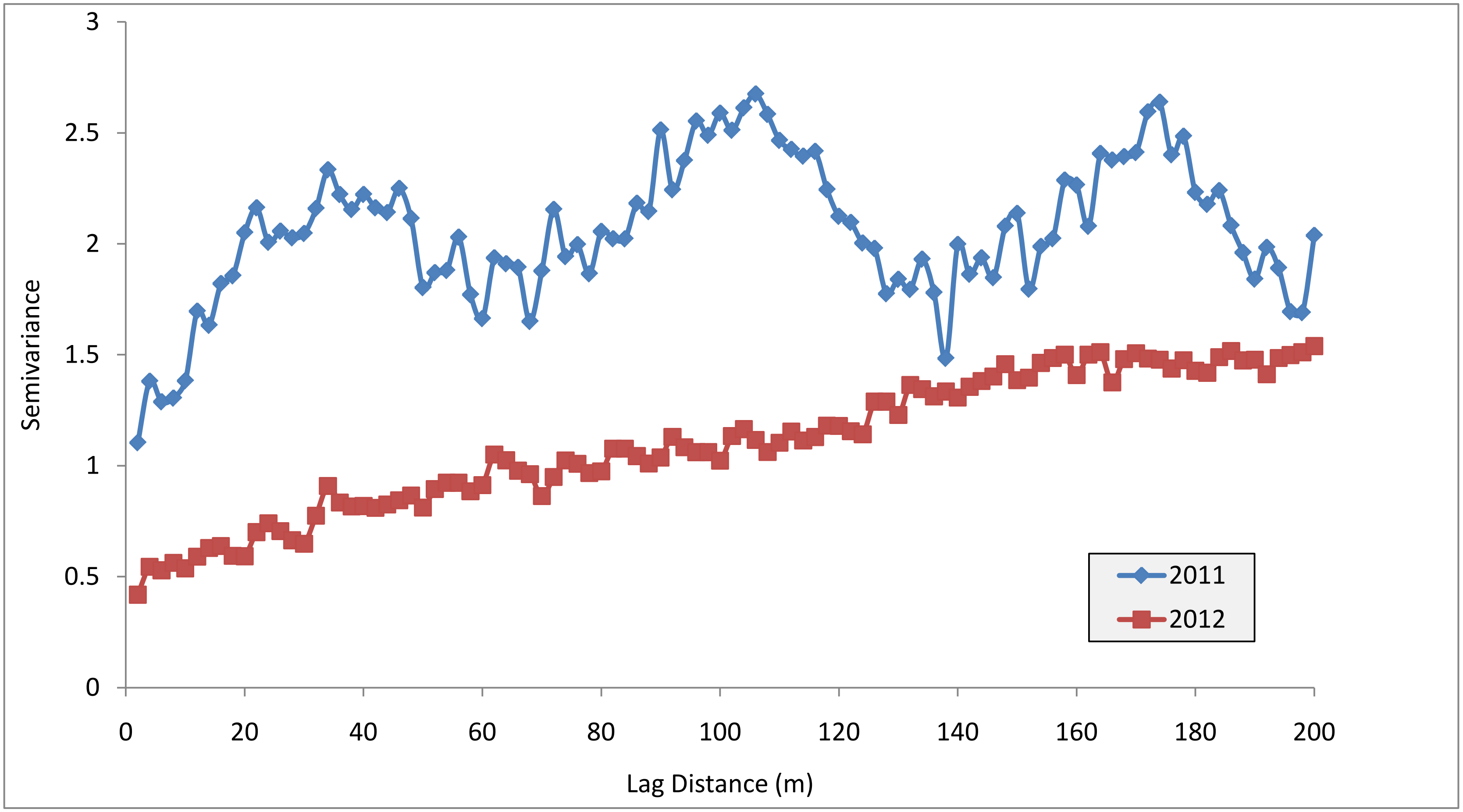
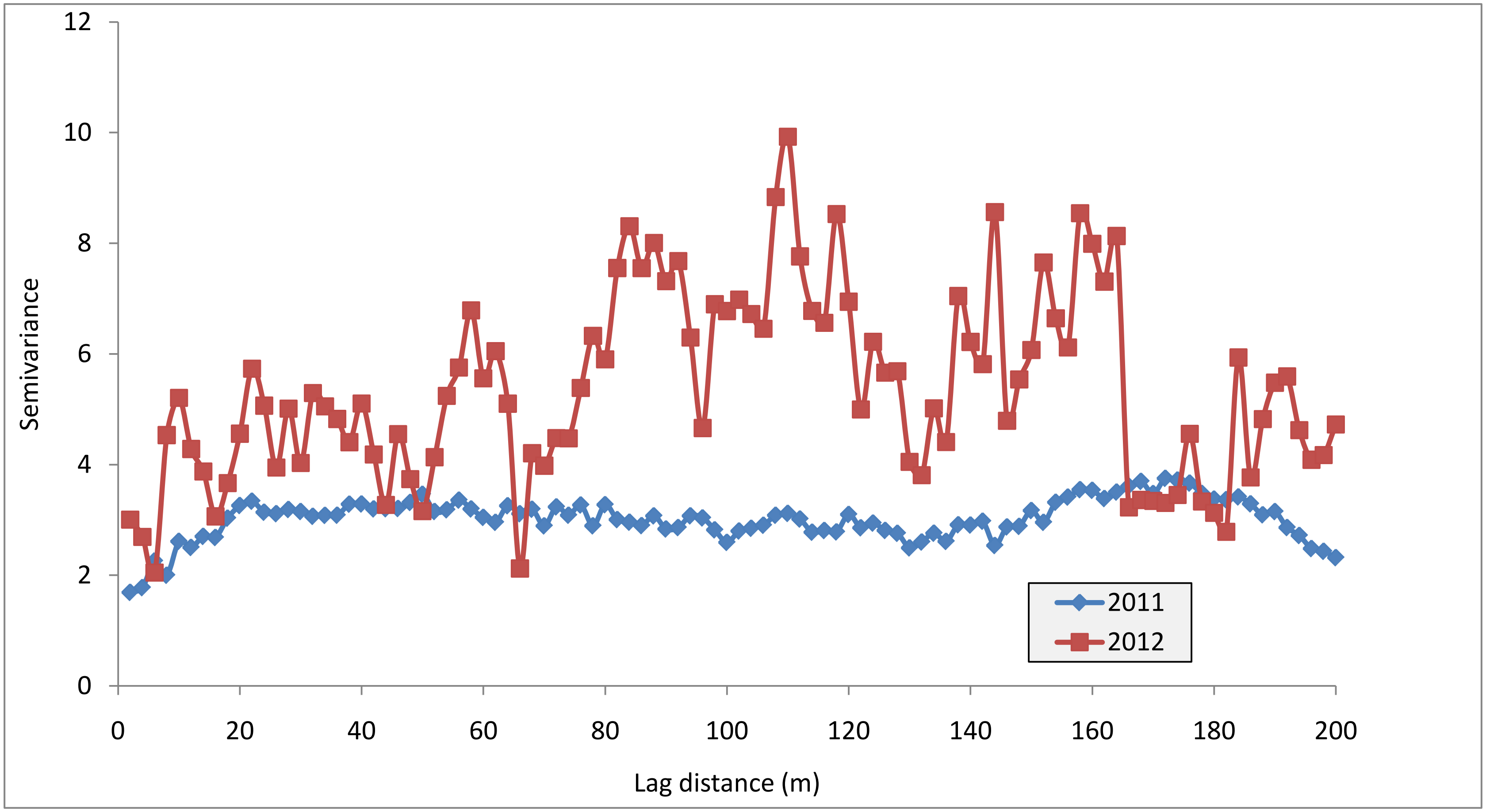


Figure 3.17. Experimental semivariograms for MWD for both treatment years, 20-30 cm depth.



at this depth suggest spatial variability occurs at shorter ranges than the upper two sampling depths.

K_{sat} measurements are shown for both treatment years (Figure 3.18). Experimental semivariograms of K_{sat} were calculated from 38 data points for 2011 and 40 data points for 2012, each taken at 10-m sampling intervals (Figure 3.19). K_{sat} showed no evidence of spatial structure for either treatment years. Because the driving forces of hydraulic conductivity (soil structure and subsequent pore geometry) can vary over space and time, it is not surprising that K_{sat} can be highly variable. Bormann and Klaassen (2008) found K_{sat} to vary seasonally and with land use, and Zeleke and Si (2005) found nested scales of variability in K_{sat} along a north-south transect in glacial sandy loam soil in Saskatchewan, Canada.

Soil C content is shown for all sampling depths for both treatment years (Figures 3.20a, 3.20b, 3.21a, and 3.21b). Experimental semivariograms of soil C were calculated from 200 data points taken at 2-m sampling intervals for each treatment year (Figures 3.22, 3.23, and 3.24). Elevated levels of soil C at the 20-30 cm depth occurred at the 180-m and 260-300 m sampling distances in 2011 (Figure 3.20b). Field observations of vegetative cover (data not shown) indicate grass debris and storm debris cover ranging from 5-65% in 2011; these elevated soil C levels in the subsurface do not occur in 2012, although similar debris deposition was observed. Alluvial deposition has been shown to create C-rich buried soil layers (Blazejewski et al., 2009; Cambardella et al., 1994) and this C could be an important factor in riparian processes such as denitrification. Considering that the data are inconsistent from 2011 to 2012, however, the elevated

Figure 3.18. K_{sat} for both treatment years (2011 and 2012). Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

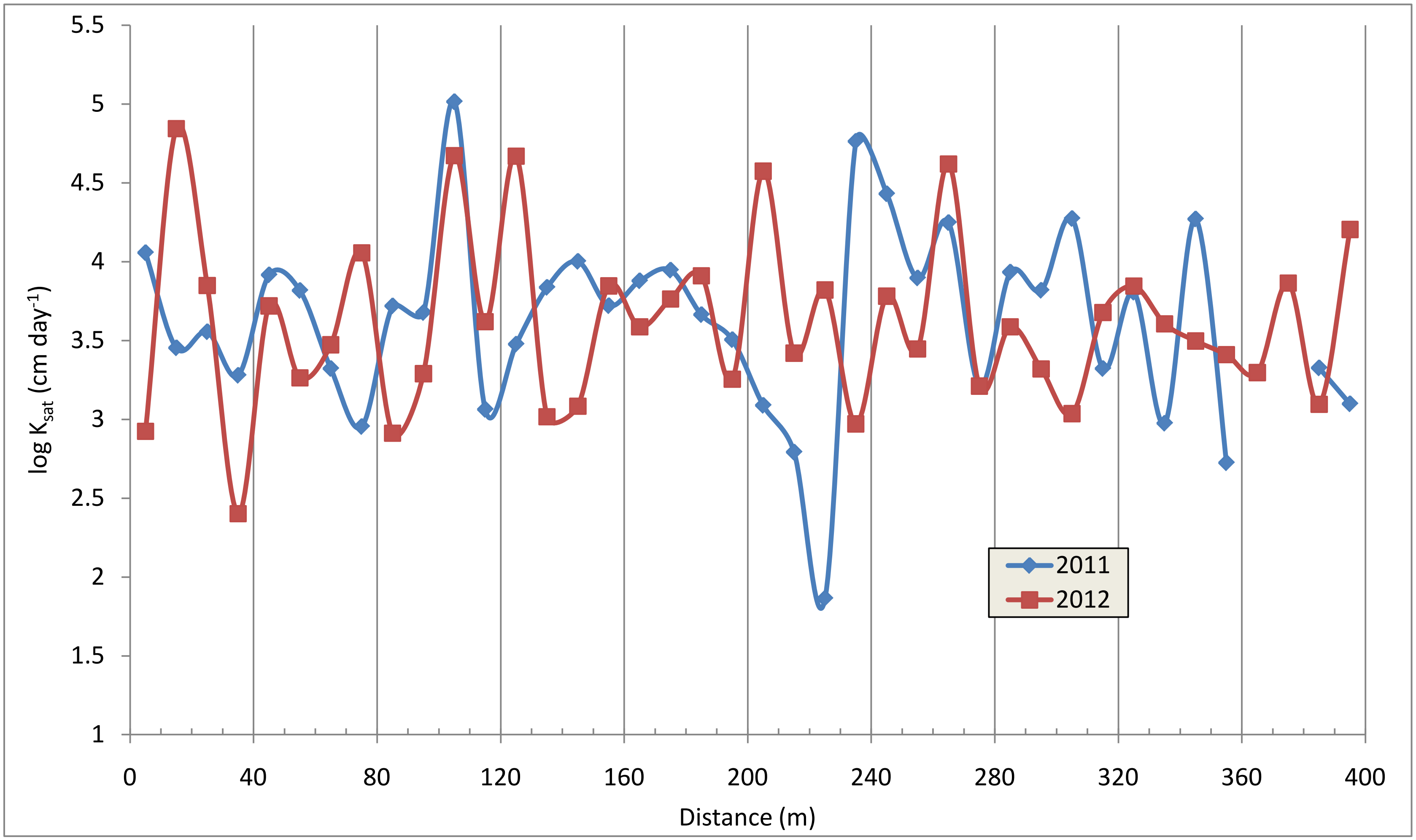


Figure 3.19. Experimental semivariograms for K_{sat} for both treatment years.

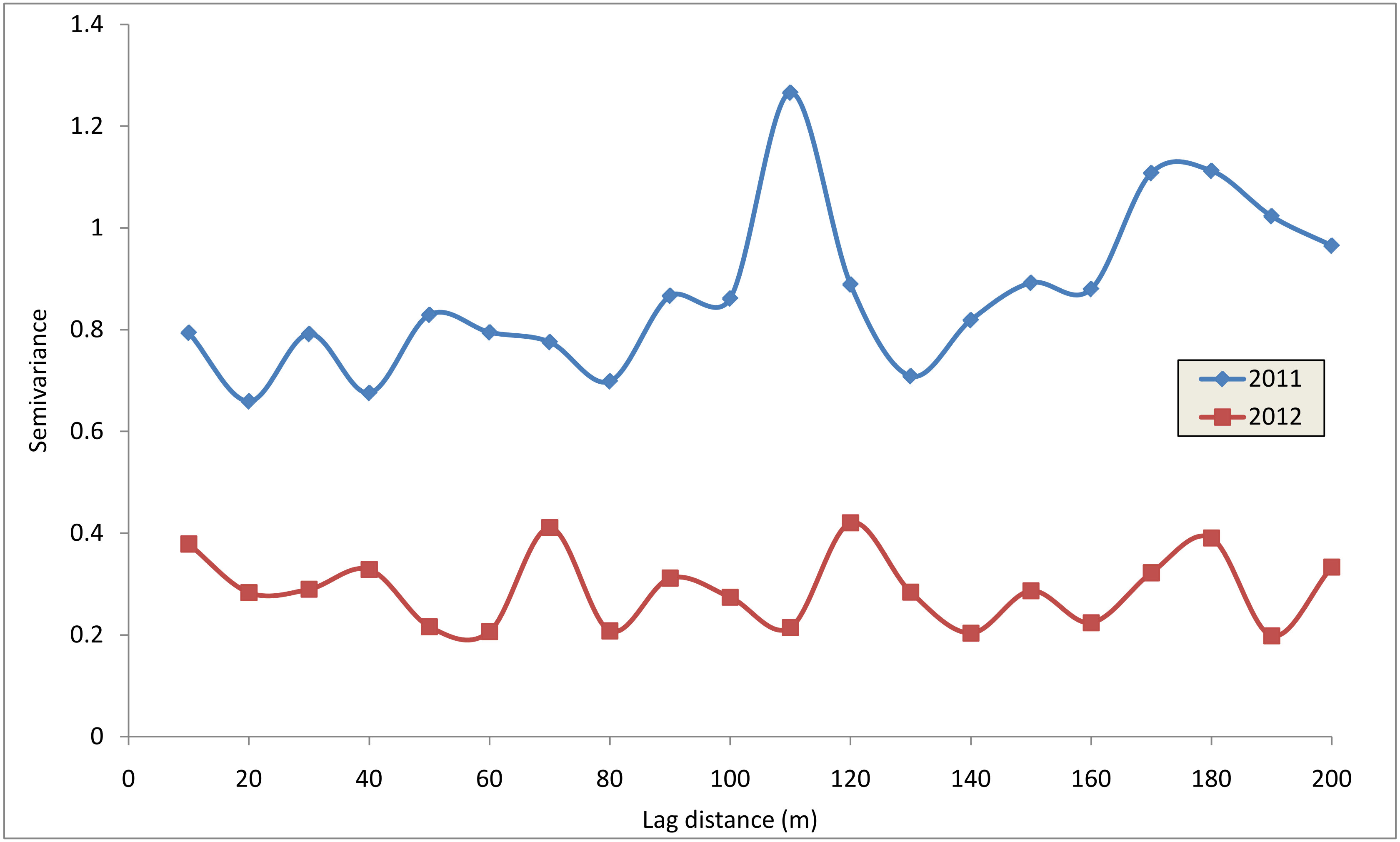


Figure 3.20a. Raw soil carbon after one year of treatment (2011). Five original data points are shown per plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

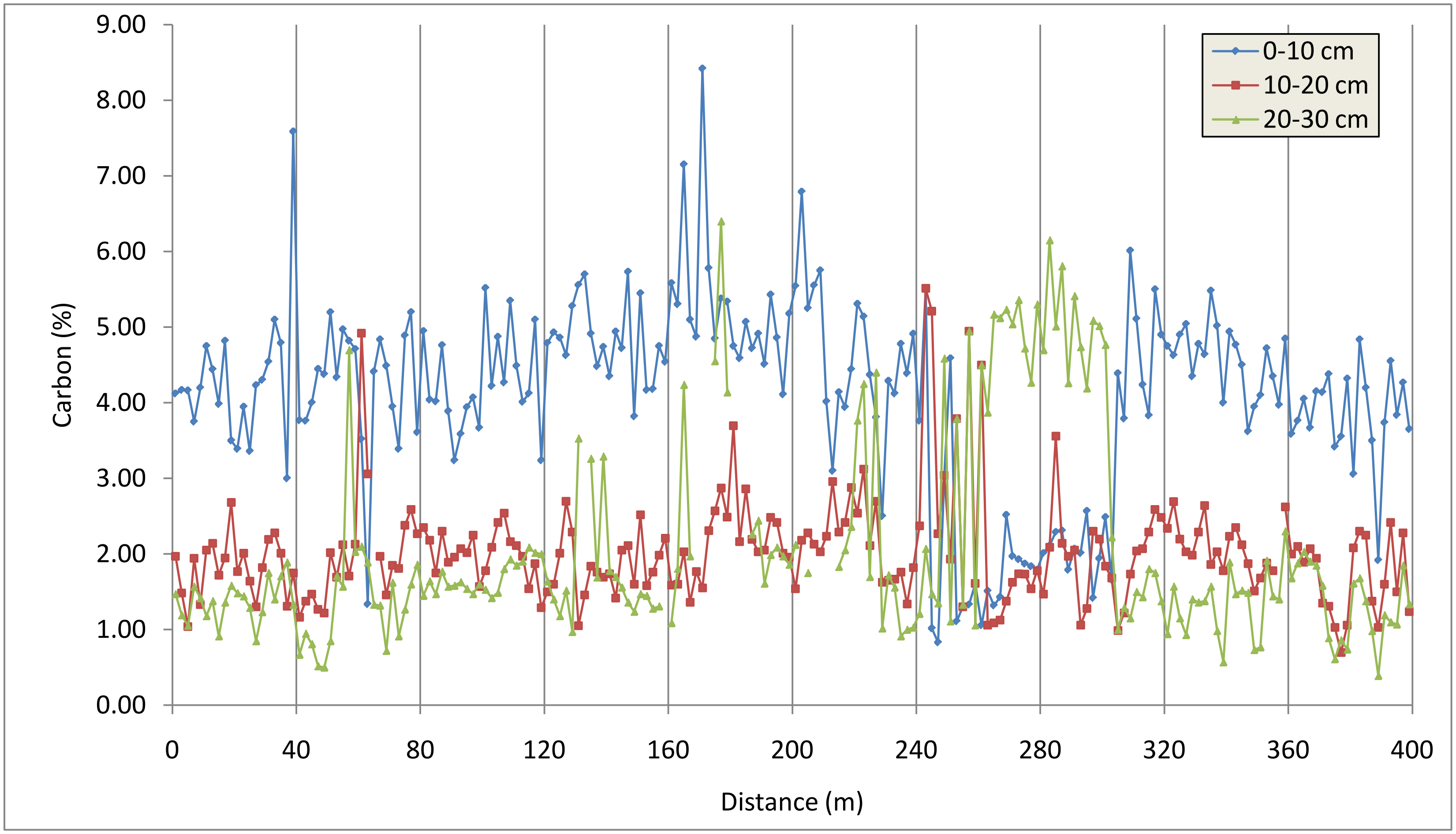


Figure 3.20b. Mean soil carbon after one year of treatment (2011). Data points represent one treatment plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

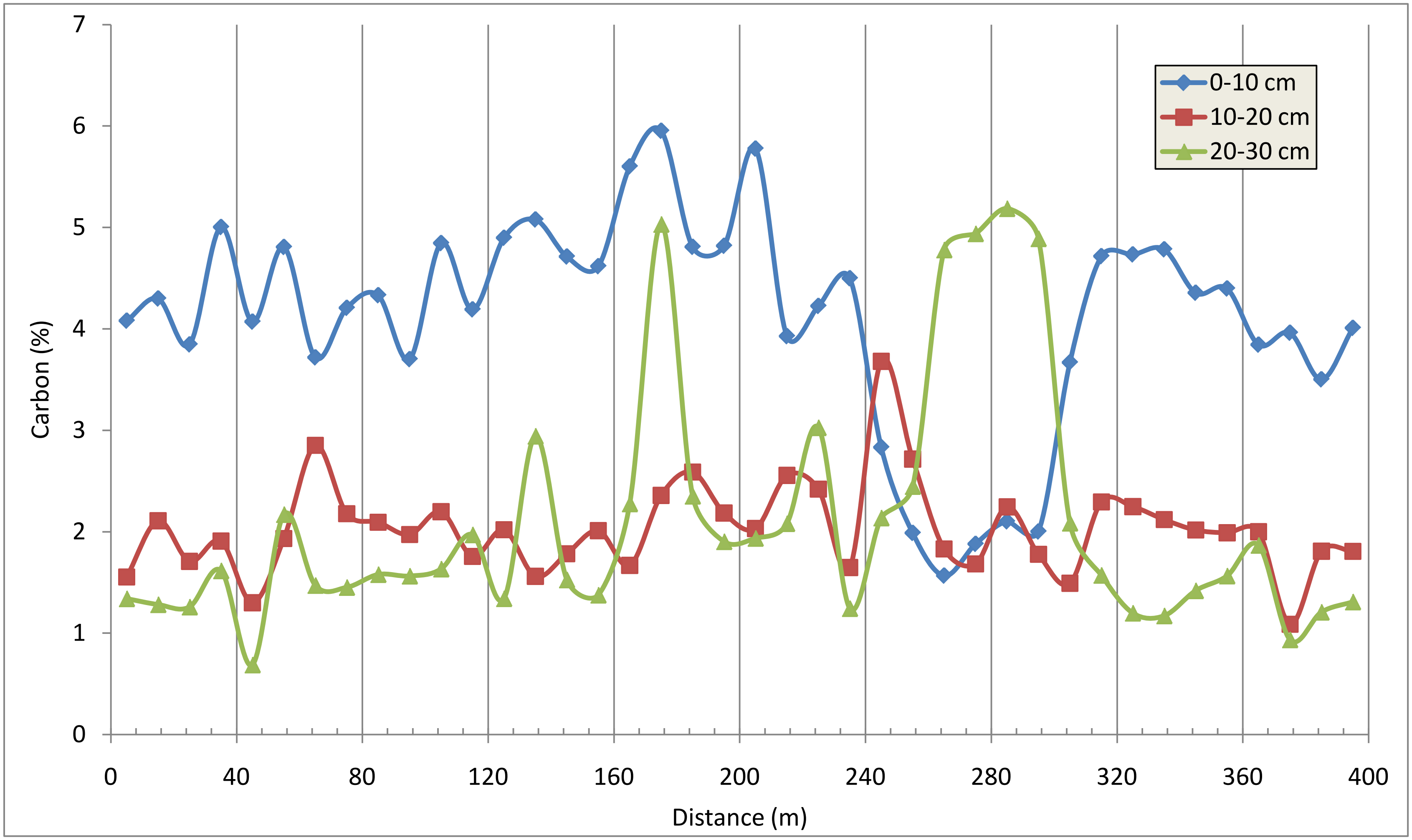


Figure 3.21a. Raw soil carbon after two years of treatment (2012). Five original data points are shown per plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

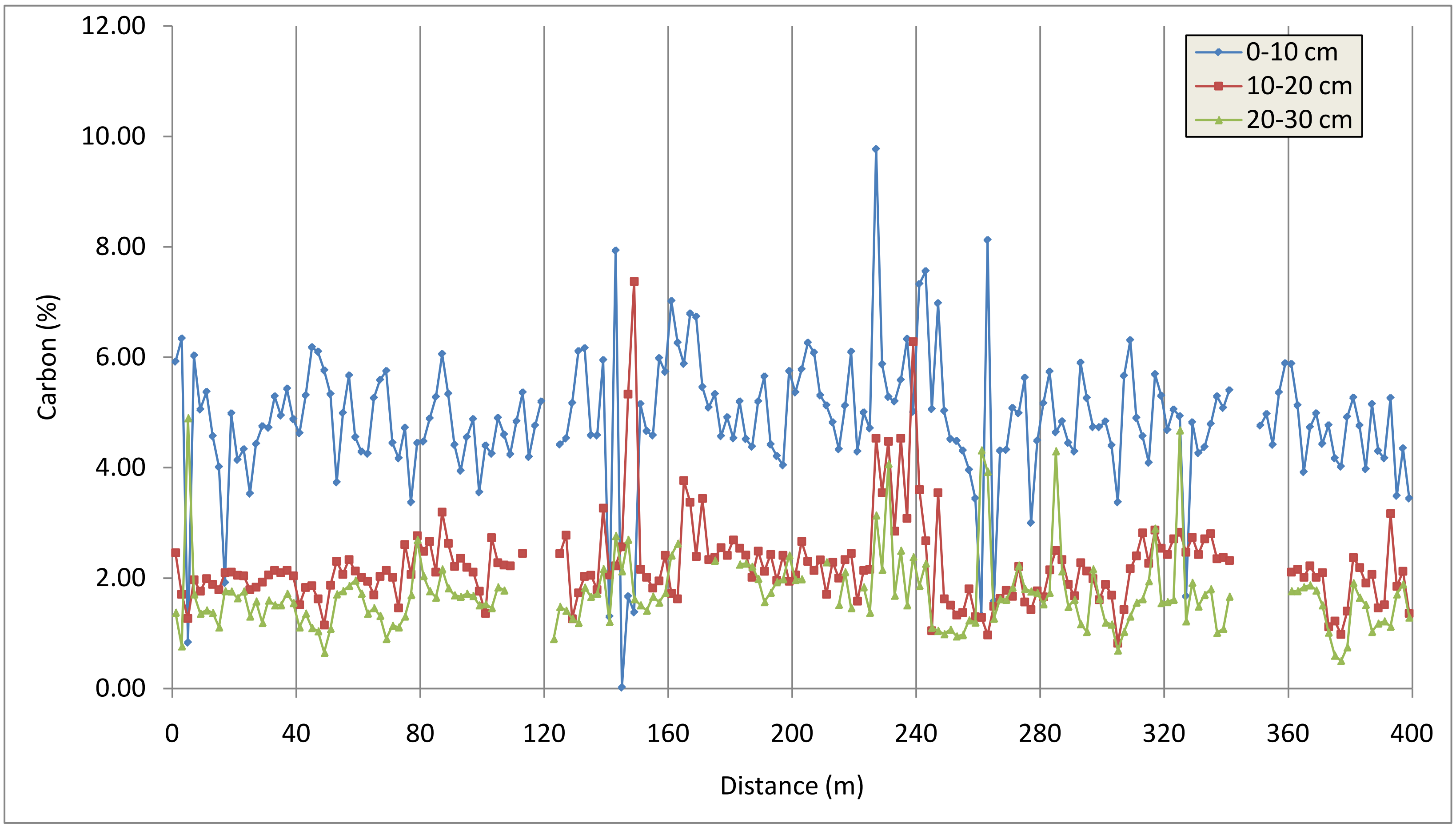


Figure 3.21b. Mean soil carbon after two years of treatment (2012). Data points represent one treatment plot. Beginning at 0-m, each 40-m distance increment (denoted by vertical gridlines) represents one sequence of treatments, such that from left to right, the sequence repeats intense mow, moderate mow, no mow, and native grass.

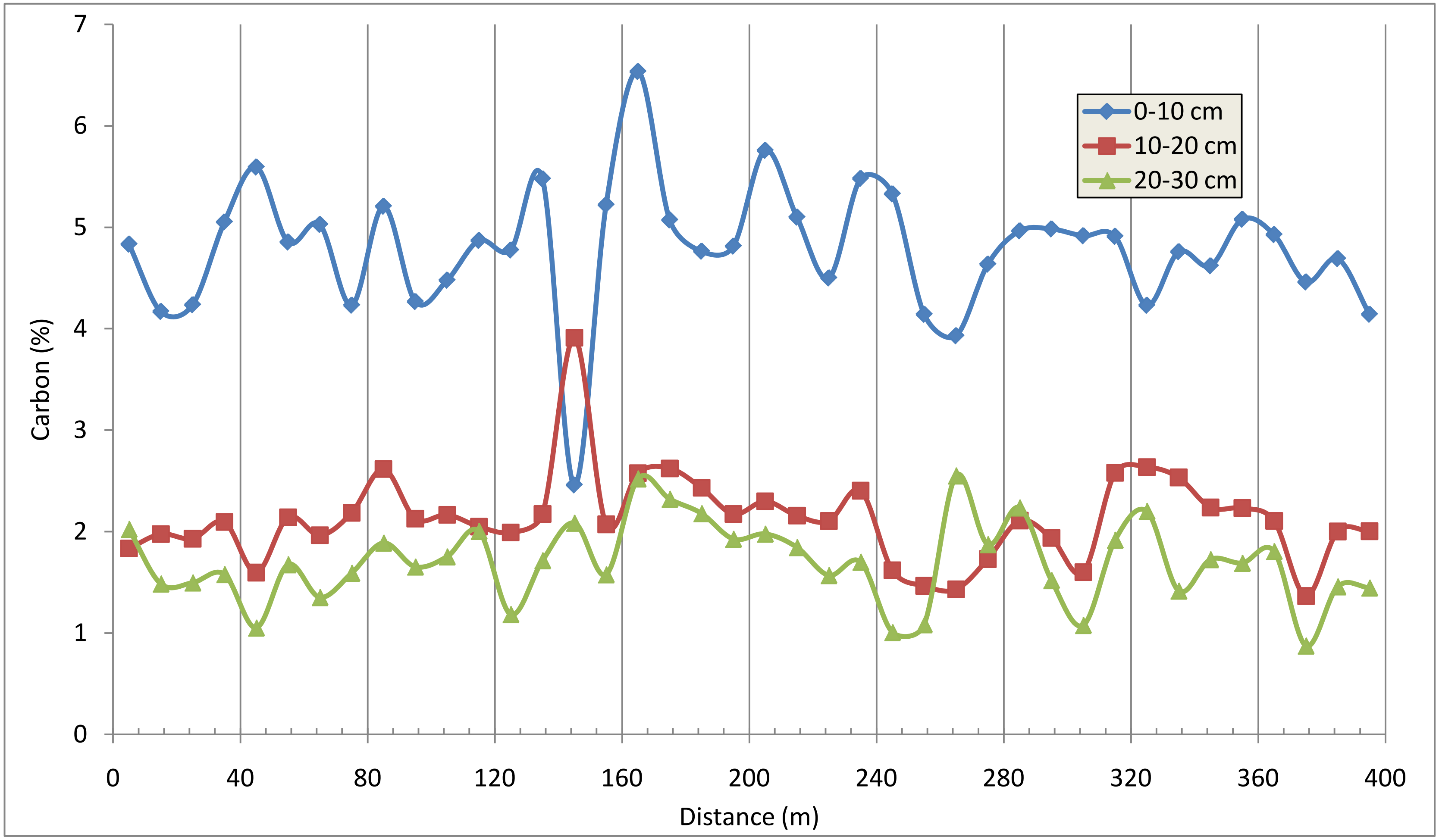


Figure 3.22. Experimental semivariograms for soil C for both treatment years, 0-10 cm depth.

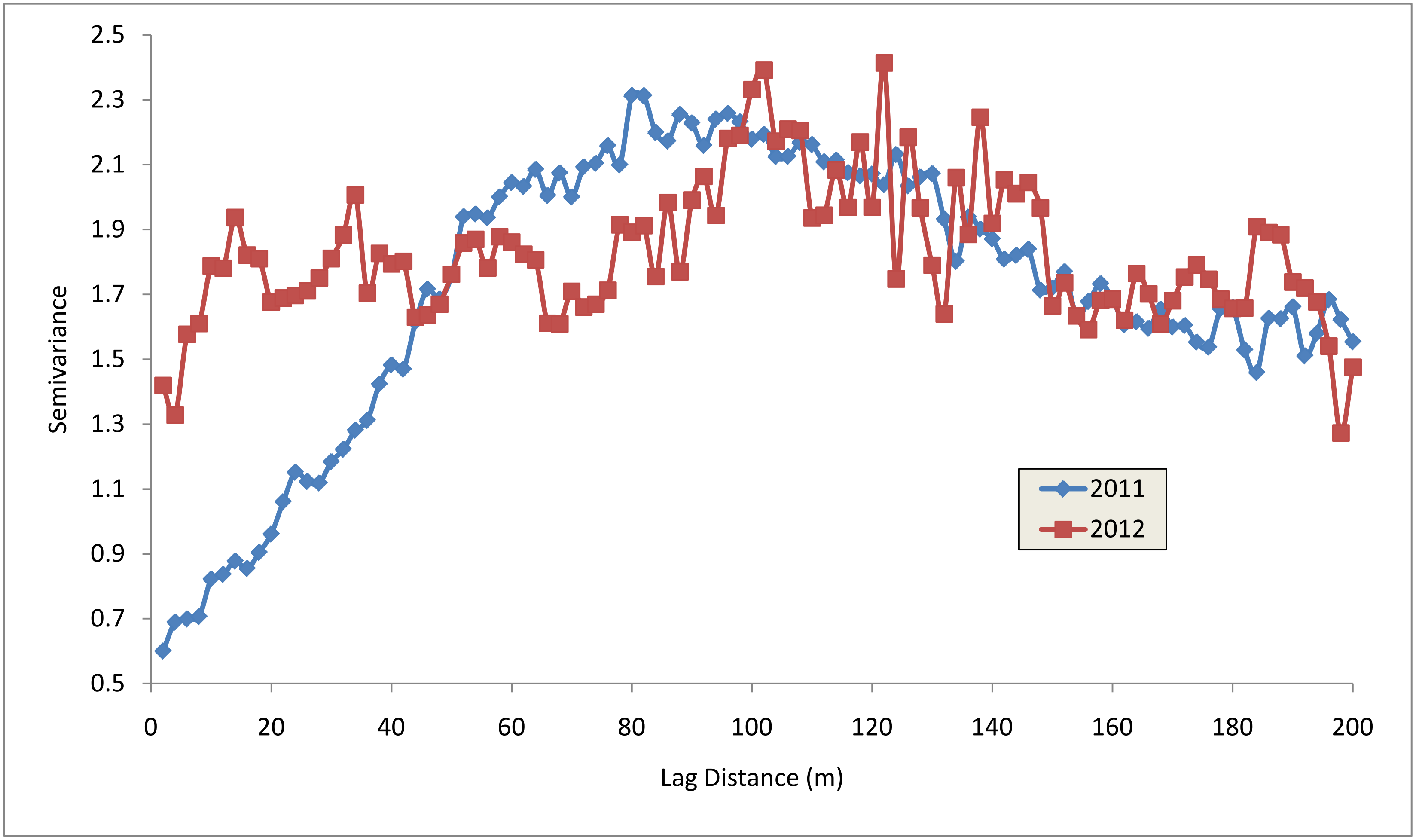


Figure 3.23. Experimental semivariograms for soil C for both treatment years, 10-20 cm depth.

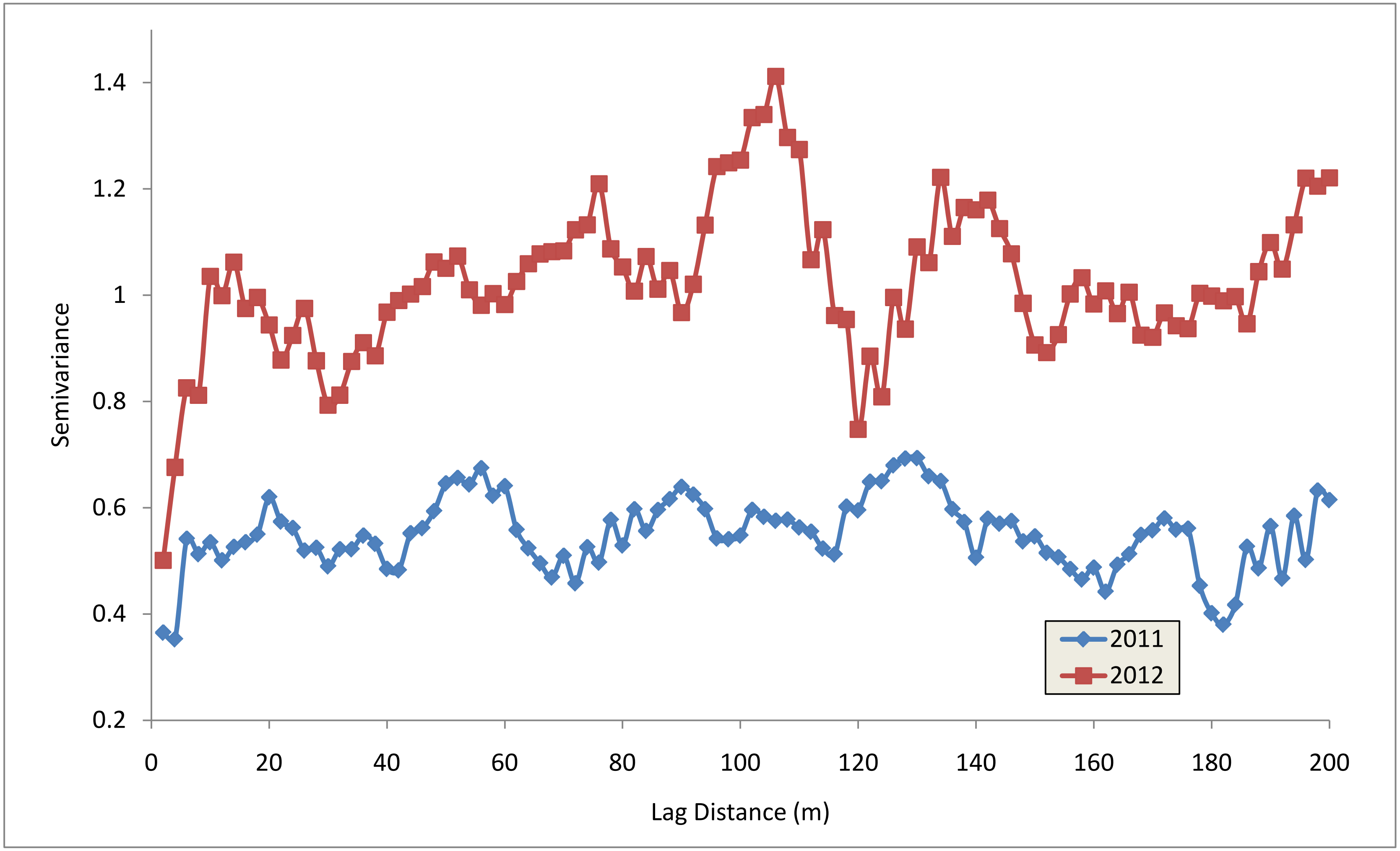
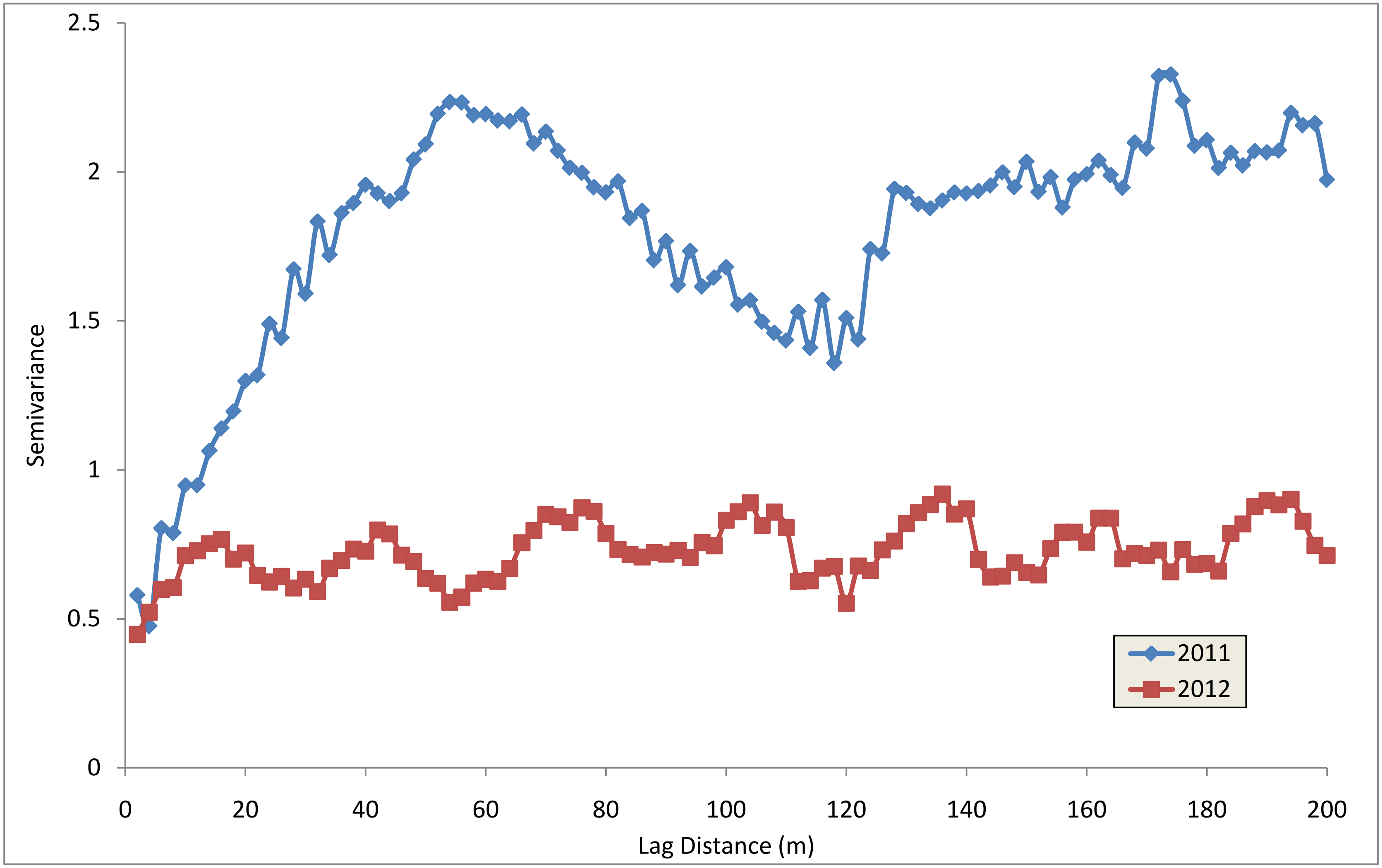


Figure 3.24. Experimental semivariograms for soil C for both treatment years, 20-30 cm depth.



subsurface soil C could be a result of sample contamination from surface-deposited flooding debris.

Spatial variability of soil C at the 0-10 cm depth occurred at a range of approximately 100-m in 2011; in 2012, spatial variability of soil C occurred at ranges of approximately 15-20 m as well as 40-100 m (Figure 3.22). Spatial variability in surface soil C has been found at similar ranges (> 100-m) by other researchers, although their work did not specifically focus on riparian soils (Cambardella et al., 1994; Zeleke and Si, 2005); they attributed soil C variability to large-scale processes such as topography and soil morphology. The smaller scale variability found in 2012 may be evidence of the imposed treatments. At the 10-20 cm depth, spatial variability of soil C occurred at a range of 15-20 m in both sampling years (Figure 3.23). At the 20-30 cm depth, spatial variability of soil C occurred at multiple ranges (15-20 m and 60-m) in 2011 but occurred predominantly at a range of 15-20 m in 2012 (Figure 3.24).

Roots, MWD, and soil C exhibited multiple scales of spatial variability. It was generally observed that these parameters had smaller scales of variability in 2012 than 2011. The occurrence of spatial variability over shorter lag distances in 2012 compared to 2011 suggested that an effect of imposed treatments may be developing over time.

Studies investigating the spatial distribution of riparian buffer soil properties are limited, making it difficult to compare the results of this study to those of other researchers. From a study of Rhode Island streams, Blazejewski et al. (2009) suggested that flooding and deposition play a significant role in riparian buffer subsoil C spatial distribution, especially those associated with alluvial deposits in first- through fourth-order streams. Their study, however, sampled buried A horizons up to 4-m in glacial outwash soils. An Iranian study investigated a 92 km^2 catchment with multiple landforms and found clear spatial patterns in WSA, C, and MWD (Mohammadi and Motaghian, 2011) with close spatial relationships between WSA and MWD. The Iranian study, however, investigated soil properties on a much larger scale and did not focus exclusively on riparian buffers.

Conclusions

This study was performed to assess the influence of vegetation management strategies on root biomass, soil aggregates, hydraulic conductivity, and soil carbon in a riparian buffer. The application of this assessment would be developing riparian buffer management recommendations for land managers. While treatment effects are not strongly supported after two years of implementation, no negative effects to the measured parameters are shown in the data as a result of reduced mowing frequency. Furthermore, the transition from existing grassland vegetation to native grasses using conventional herbicide methods reduced root biomass during the study period, but did not significantly affect soil aggregates or hydraulic conductivity. On the basis of maintaining consistent root biomass, the moderate mow treatment appears to be the best management choice. Although soil aggregate and hydraulic conductivity data do not provide sufficient evidence for a management choice, carbon data indicate that managing riparian areas may impact carbon levels over the short term.

Spatial variability within the study site may be an important factor influencing riparian buffer soil properties. Experimental semivariograms provided evidence of spatial structure in root biomass, soil aggregates, and soil C; these parameters do not occur randomly across the study site. Spatial variability occurred at multiple scales for each parameter. The variability occurred over a shorter lag distance in 2012 than 2011, suggesting an effect of imposed treatments slowly developing over time. This information should be considered in the experiment design of future studies that assess the influences of management strategies on ecosystem properties in this riparian buffer.

The dynamic nature of riparian ecosystems and the natural complexity of soils, coupled with contradictions in the literature regarding land use effects on soil properties, make it difficult to establish concrete relationships for vegetation management influences on riparian buffer soils. Furthermore, considerable changes in soil properties may take long periods to develop, and the treatments established in this study may require additional time to exhibit significant differences. Parameters measured and sampling timing may not have been sensitive enough to detect changes on the temporal microscale. Longer-term study of this riparian buffer is needed to provide additional information for more specific management strategy development.

Chapter Four

Summary of Conclusions

Current Study

This study was developed to: 1) characterize baseline soil physical and chemical properties prior to implementing vegetation management strategies in a riparian buffer; 2) explore and assess spatial processes in a riparian buffer; 3) evaluate the influence of mowing and vegetation management strategies on root biomass, soil aggregate size distribution and stability, hydraulic conductivity, and soil carbon. Landowners and land managers need straightforward maintenance recommendations to maximize riparian buffer function in the agricultural landscape. The results of this study are a step toward these recommendations.

Soil characteristics along the 2-m transect location differed significantly from the soils along the 8-m transect. The utilization of semivariogram analysis to describe soil carbon variability provided explicit information about the range of spatial autocorrelation in this riparian environment. Differences in spatial variation between the two transects indicate that soil properties closer to the water body may be more variable than those further from the stream. Spatial relationships of soil carbon were stronger along the 8-m transect than along the 2-m transect, suggesting that managing for soil carbon farther from the stream would be less intensive than managing closer to the stream because larger land areas could be similarly managed with the same expected outcome. This relationship proposes a choice for riparian buffer managers:

1) retain more land area for production activities but increase the maintenance intensity for a 2-m wide buffer; or 2) reduce land area for production activities and reduce the maintenance intensity for an 8-m wide buffer. In addition, this characterization of riparian buffer soil properties provides insight into the potential nutrient loading of Central Kentucky streams as a result of future sediment loss from sloughing stream banks.

It was difficult to demonstrate the effect of vegetation management strategies on soil properties after only two years of implementation. However, carbon data indicate that managing riparian areas may alter carbon levels over the short term. There were no negative effects such as reduced hydraulic conductivity or loss of aggregate structure or stability evident as a result of reduced mowing frequency or native grass transition. Longer-term study of this riparian buffer is needed to provide additional information for more specific vegetation management strategy development.

Spatial variability within the study site may be an important factor influencing riparian buffer soil properties. Experimental semivariograms provided evidence of spatial structure in root biomass, soil aggregates, and soil C, indicating that these parameters do not occur randomly across the study site. Spatial variability occurred at multiple scales for each parameter. The variability occurred over a shorter lag distance in 2012 than 2011, suggesting an effect of imposed treatments developing over time. This information should be considered in the experiment design of future studies that assess the influences of management strategies on ecosystem properties in this riparian buffer.

It is important to note that extrapolation of the relationships found in this study should be limited to similar riparian environments. The stable streambank and accessible floodplain conditions in this study site likely influenced the results; these findings may not be applicable in watersheds where incised channels and inaccessible floodplains characterize the stream systems.

Future Work

This work, as well as the established experimental site, is a solid building block for future riparian buffer research. Two additional studies have developed as a result of the current study: 1) a project investigating water quality in the stream as a result of imposed treatments; and 2) a study examining denitrification potential within the riparian buffer as a result of imposed treatments.

Future investigations in the established treatment plots of this study could include root biomass, soil aggregate distribution and stability, and infiltration studies along the 8-m transect in addition to the 2-m transect. Coupling root biomass, soil structure, and infiltration data from the 8-m transect with the existing spatial relationships found in this study would provide additional information for landowners to develop management strategies for desired riparian buffer function.

Bulk density was not measured in this study, but should be considered in future research involving mowing in riparian areas. For studies that may examine the 8-m transect, considerations should be made for wheel traffic and clipping deposition because mowing patterns may influence the soil and plant characteristics at this location.

Another consideration for future work should be larger plot size. The experimental semivariograms developed in this study indicated that spatial autocorrelation exists at a greater range than that of the plot width (10-m) established in this study. Therefore, plots could have been larger to aid in the ease of mowing and maintenance and better reflect actual practices employed by land managers.

Hydrologic characteristics of the riparian buffer should be examined in future studies. Monitoring the water table could provide useful information in relation to plant characteristics, such as root growth dynamics as a function of water table fluctuations. The roots in this study were concentrated in the upper 10-cm, which was consistent with other riparian buffer studies. One could surmise this phenomenon is a result of adequate water and no need for roots to mine deeper in the soil profile for water or nutrients. Further research may address water quality implications of superficial root systems, and how superficial root systems may affect soil carbon and subsequently denitrification potential in riparian buffers.

Further analysis of the vegetation communities of this study will be conducted, and may provide insight into the response of plant types to mowing treatments. This type of analysis over a longer period (> 5 years) would provide data comparable to that from other riparian buffer studies establishing native grasses.

Agricultural nonpoint source pollution continues to threaten water resources. The utilization and management of riparian buffers has great potential to reduce water pollution and provide ecosystem services beyond the realm of this study. In addition to longer-term research, educational and policy-driven opportunities are needed to communicate the benefits of riparian buffers as well as motivate land managers to utilize them effectively. Riparian buffers play a key role in agroecosystem functions.

Appendix. Correlation matrix by depth and transect. 2-m x 8-m comparison in bold. *Correlation significant at p<0.05.

		2-m transect					8-m transect				
		0-10 cm	10-20 cm	20-30 cm	30-40 cm	40-50 cm	0-10 cm	10-20 cm	20-30 cm	30-40 cm	40-50 cm
Carbon	**2-m**										
	0-10 cm	1.00									
	10-20 cm	0.21	1.00								
	20-30 cm	0.13	0.54*	1.00							
	30-40 cm	-0.12	-0.14	-0.18	1.00						
	40-50 cm	-0.10	0.18	0.10	0.71*	1.00					
	8-m										
	0-10 cm	**-0.08**	**0.34***	**0.34**	**0.06**	**0.60***	1.00				
	10-20 cm	**-0.12**	**0.22**	**0.15**	**0.56***	**0.45**	0.67*	1.00			
	20-30 cm	**0.20**	**0.33***	**0.28**	**0.27**	**0.06**	0.60*	0.77*	1.00		
	30-40 cm	**-0.02**	**0.05**	**0.25**	**-0.10**	**-0.55***	0.39*	0.49*	0.67*	1.00	
	40-50 cm	**-0.29**	**0.00**	**0.04**	**-0.08**	**-0.08**	-0.14	0.07	-0.13	0.29	1.00
Sand	**2-m**										
	0-10 cm	1.00									
	10-20 cm	0.72*	1.00								
	20-30 cm	0.17	0.51*	1.00							
	30-40 cm	0.26	0.09	0.16	1.00						
	40-50 cm	-0.10	-0.26	-0.13	0.69*	1.00					
	8-m										
	0-10 cm	**-0.07**	**-0.02**	**-0.12**	**0.09**	**0.45***	1.00				
	10-20 cm	**0.30**	**0.33***	**0.10**	**-0.14**	**-0.11**	0.39*	1.00			
	20-30 cm	**-0.09**	**-0.01**	**-0.13**	**-0.23**	**-0.08**	0.33*	0.51*	1.00		
	30-40 cm	**-0.13**	**-0.01**	**-0.12**	**-0.25**	**0.17**	0.27	-0.22	0.34*	1.00	
	40-50 cm	**-0.22**	**-0.04**	**-0.40**	**-0.45***	**-0.57***	-0.05	-0.38	-0.34	0.13	1.00

Appendix. cont. Correlation matrix by depth and transect. 2-m x 8-m comparison in bold. *Correlation significant at p<0.05.

		2-m transect					8-m transect				
		0-10 cm	10-20 cm	20-30 cm	30-40 cm	40-50 cm	0-10 cm	10-20 cm	20-30 cm	30-40 cm	40-50 cm
Silt	**2-m**										
	0-10 cm	1.00									
	10-20 cm	0.64*	1.00								
	20-30 cm	0.20	0.40*	1.00							
	30-40 cm	0.29	0.06	0.28	1.00						
	40-50 cm	-0.24	-0.33	-0.09	0.66*	1.00					
	8-m										
	0-10 cm	**0.11**	**-0.05**	**0.02**	**0.12**	**0.34**	1.00				
	10-20 cm	**0.34***	**0.37***	**0.23**	**-0.30**	**-0.51***	0.28	1.00			
	20-30 cm	**0.07**	**0.12**	**-0.10**	**-0.18**	**-0.37**	0.08	0.57*	1.00		
	30-40 cm	**-0.09**	**-0.11**	**-0.29**	**0.11**	**0.32**	-0.10	-0.49*	0.02	1.00	
	40-50 cm	**-0.08**	**0.18**	**-0.46***	**-0.34**	**-0.41**	-0.03	-0.30	-0.29	0.32	1.00
Clay	**2-m**										
	0-10 cm	1.00									
	10-20 cm	0.38*	1.00								
	20-30 cm	-0.22	-0.08	1.00							
	30-40 cm	-0.15	-0.13	0.27	1.00						
	40-50 cm	-0.15	-0.22	0.00	0.40	1.00					
	8-m										
	0-10 cm	**0.08**	**0.10**	**0.04**	**-0.14**	**-0.47***	1.00				
	10-20 cm	**0.24**	**0.20**	**-0.11**	**-0.55***	**-0.70***	0.38*	1.00			
	20-30 cm	**0.23**	**-0.03**	**0.11**	**-0.15**	**-0.45**	-0.07	0.43*	1.00		
	30-40 cm	**-0.13**	**0.01**	**0.08**	**-0.15**	**0.49**	-0.20	-0.21	0.25	1.00	
	40-50 cm	**0.27**	**-0.28**	**-0.23**	**0.17**	**0.22**	0.02	-0.50*	-0.12	0.09	1.00

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PUBLICATIONS

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Gumbert, A.A. and M.S. Coyne. 2011. Managing riparian buffers to improve soil structural properties. *In* Proceedings North Central Extension-Industry Soil Fertility Conference. Des Moines, Iowa. 27:138-144.

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<u>*Extension*</u>

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SCHOLASTIC AND PROFESSIONAL HONORS

2011 Kentucky Outstanding Graduate Student Award, North Central Extension-Industry Soil Fertility Conference

2011 Robert A. Lauderdale Award for Outstanding Contributions in Water Quality, Kentucky Water Resources Research Institute

2011 Water Quality Award for *Cane Run Watershed Council*, Scott County Conservation District

2010 Lexington-Fayette Urban County Government Environmental Commission Award for *Kentucky Horse Park Stream Vegetation Project*

2008 Lexington-Fayette Urban County Government Environmental Commission Award for *Cane Run Watershed Council Collaborative*

2006 Outstanding Extension Associate Award, Kentucky Association of State Extension Professionals

2005 Southern Region Water Quality System Team Award

2004 American Society of Agronomy Certificate of Excellence, Educational Materials Awards Program for IP-73 *Living Along a Kentucky Stream*

2003 Association of Natural Resources Extension Professionals (ANREP) Silver Award for short publication: IP-71 *Nutrient Management in Kentucky*

2001 Salt Lake City Olympic Committee "Spirit of the Land" Award for *2001: A Water Odyssey*, Kentucky State Fair Educational Exhibit

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