STREAMBANK EROSION AND RISK ASSESSMENT OF CONTAMINANT TRANSPORT IN MISSOURI WATERSHEDS

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ABSTRACT

Agriculture is the leading source of water pollution in the United States. Sediment is documented as one of the top ten leading causes of impairment of waterways, and the occurrence of pesticides in surface and ground waters in agricultural areas of the Midwestern United States is widespread and extensive. As researchers and management specialists look to understand the causes and effects of these water quality issues, the need for continued investigation is apparent. Study sites were established in Crooked and Otter Creek, two claypan watersheds in northeastern Missouri, to determine the effect of stream order, season, and adjacent land use on streambank erosion rates. Erosion rates were consistently higher in winter months than in spring/summer and fall season, however, the significant three-way interaction indicated that the treatments did not respond in the same manner across season. Soil nutrient concentration data showed that forest sites had significantly lower C and N concentrations than other land uses. Streambank erosion rates were applied to the entire study area, and estimates showed that bank sediment accounted for 47-71% of the total in-stream sediment. These results indicate that streambanks are a significant source of sediments in streams of claypan watersheds. A processed-based index model was developed to assess relative landscape vulnerability to hydrologic losses of four commonly used corn herbicides. The model was applied to the Young's Creek watershed in the Central Claypan Region of Missouri. The model applies mathematical functions to assign weights (i.e. degree of risk) on the basis of herbicide, soil, and hydrologic properties relevant to the environmental fate of herbicides. The area-weighted risks were used to evaluate the potential spatial and temporal risk of applying atrazine, diketonitrile (the active metabolite of isoxaflutol), glyphosate, and metolachlor in the Young's Creek watershed. In the case of Young's Creek, the use of metolachlor or isoxaflutole minimizes the overall risk of herbicide transport to surface or ground waters. The information provided by the model can be used to make recommendations regarding the choice of herbicide that will minimize the risk of hydrologic transport for this watershed. By investigating streambank erosion rates as a function of land use, stream order, and season, and by developing an index model for pesticide transport risk, land managers will be able to target the placement of BMPs to areas in a watershed that are most vulnerable to environmental degradation.

CHAPTER 1: LITERATURE REVIEW

INTRODUCTION

Sediment and herbicide contamination have long been recognized as significant environmental concerns in claypan watersheds (Lerch et al., 2008). This thesis includes two studies that represent two separate approaches to watershed management. The first project was a streambank erosion project for which the overall objectives of the study were to investigate the effects of adjacent land use, season, and stream order on streambank erosion rates in two watersheds of the Central Claypan Region of northeastern Missouri. The second study was the development of a risk assessment model for herbicide transport. Using the soil survey geographic database (SSURGO), an index-based risk assessment model to predict relative risks of hydrologic losses of pesticides was applied to a watershed located in the Central Claypan Region, Major Land Resource Area 113 (USDA-NRCS, 2006). The model considers pesticide, soil, and landscape properties that are assigned weights (i.e., degree of risk) through mathematical functions used to describe the relevant processes controlling the transport of pesticides. By investigating streambank erosion rates as a function of land use and by developing an index model for pesticide transport risk, practices for creating and implementing effective management strategies that address water quality issues can be developed and better targeted to vulnerable areas in a watershed.

STREAMBANK EROSION

As a non-point source of pollution, sedimentation is a significant environmental problem. The damage that results from water pollution from sediment costs an estimated $16 billion annually in North America (Wynn and Mostaghimi, 2006). Streambanks contribute substantially to stream sediment (Zaime et al., 2006; Fox et al., 2007; Piercy and Wynn, 2008), contributing as much as 80% in one river system in western Tennessee (Simon et al., 2000). Investigating land use impacts on streambank erosion rates may lead to development of improved management practices or provide the basis for targeting the placement of management practices to mitigate streambank erosion.

Processes Controlling Streambank Erosion

The processes controlling streambank erosion are complex and multifaceted. Factors influencing these processes include the physical processes acting on bank soils, stream characteristics, and management influences. Physical processes consist of the driving and resisting forces of streambank soils and the shear/scour forces acting on streambanks. Stream characterisitics such as discharge, flow velocity, sediment load, and channel morphology modify the effects of physical erosion processes. The effects of vegetation and livestock management also work to change the dynamics controlling erosive forces. Each of these categories (physical processes, stream characteristics, and management practices) should be considered to bring an understanding of the dynamic processes controlling streambank erosion.

Streambank erosion occurs through a combination of mass movement, scour of streambank soil, and bank weathering (Lawler et al., 1999). Bank retreat, and subsequent changes to channel morphology, occurs through undercutting of streambanks by fluvial erosion, mass movement of cohesive upper banks as a result of pore pressure changes, and removal of failed sediment by the stream current. While frequency rates of fluvial erosion are much higher than bank failure rates, both mechanisms contribute nearly equally to bank retreat (Casagli et al., 1999).

Mass failure of bank soil occurs when the driving gravitational forces exceed the resisting frictional forces of the soil composing or resting on the sloped streambank (Easterbrook, 1999a). The stability of any sloped material, including sloped streambank soils, is often measured using the equation $F=F_f/F_g$, where F is the factor of safety, F_f is the sum of resistive forces, and F_g is the sum of driving forces (Easterbrook, 1999a; Fox et al., 2007). A factor of safety greater than 1 indicates a bank is stable, while a value less than 1 indicates instability (Easterbrook, 1999a; Simon and Collision, 2002). Gravitational forces can be divided into two components. Those acting perpendicular to the slope are called normal stresses and those acting parallel to the slope are called shear stresses. The magnitude of driving gravitational forces depends on the mass of the slope material and the slope angle. Consequently, downslope movement of bank soils is determined by the steepness of the slope, friction, and physical properties of the bank soil that influences the elastic, plastic, or fluid characteristics of the soil. In addition to these influences, water content may also contribute to driving forces of bank instability by increasing the weight of bank soils (Easterbrook, 1999a).

Cohesion and friction are resistive forces that help stabilize bank soils. Cohesive strength is the strength of materials or the resistance when normal stress is zero, as on a vertical streambank. Shear strength, a resisting force, is the product of normal stress and the coefficient of friction for bank soil. For cohesive materials, as cohesion increases so does shear strength, which allows silt/clay banks to form vertical slopes while non-cohesive sands cannot form vertical slopes (Easterbrook, 1999a). Cohesion stabilizes bank soils by promoting electrochemical bonding within the soil matrix and increasing surface tension at the air-water interface between saturated and unsaturated zones (Simon and Collision, 2002).

Soil moisture content greatly influences cohesive and frictional forces. In saturated conditions, pore-water pressure reduces normal stress experienced by bank soil particles. As normal stress decreases, internal friction decreases (Easterbrook, 1999a). However, in unsaturated streambanks, matrix suction caused by negative pore pressures produces apparent cohesion and increases shear strength of streambanks (Casagli et al., 1999). Effects of rising and falling stream stage during rain or flood events on streambank stability also occur. Saturation of cut bank material during flood stage can increase fluid-pore pressure if draw-down rates exceed the rates of bank drainage, as is common in cohesive streambanks. As a result, loss of cohesion can lead to bank failure (Easterbrook, 1999a). In sum, gravitational forces and saturated conditions decrease stability while friction and cohesion resist erosion (Fox et al., 2007).

Once bank material erodes, stream flow entrains bank-toe and failed cohesive soil from mass movement, maintaining the steep, but low-profiles of streambanks (Simon et al., 2000). Bank soil is eroded when the shear stress created by stream flow on bank soil particles becomes greater than the critical shear stress (Casagli et al., 1999). This relationship is often modeled using an excess shear stress equation:

$$E_r = Kd(\tau - \tau_c)^a \qquad \text{(Eq.1.1)}$$

where E_r is the erosion rate, Kd is the erodibilty factor of the bank material, τ is the shear stress, τ_c is the critical stress, and *a* is a power term, commonly assumed to be equal to 1 (Fox et al., 2007). Soil bulk density is a significant parameter in determining bank Kd; Kd increases as bulk density increases (Wynn and Mostaghimi, 2006).

Entrainment of bank sediments also depends on the energy of stream water (governed by flow velocity) and the cohesive resistance of bank soils. Entrainment occurs when the gravitational and cohesive forces are overcome by the power of the flowing stream. The relative importance of gravity and cohesion are dependent on bank soil properties. For sands, which are non-cohesive, gravity is the governing force resisting erosion. For banks composed of clay and silt, cohesion between grains is the dominant force resisting erosion. Cohesive forces are stronger than the gravitational force, and therefore, higher flow velocity is required for entrainment of clay and silt banks compared to sand banks (Easterbrook, 1999b). As hydraulic action entrains and transports bank sediment, the particles become a source of abrasion and increase the erosive power of the moving water (Easterbrook, 1999b). Lane et al. (1996) found that scour of alluvial channels is correlated to the rising limb of the hydrograph when there is a positive change in discharge (Q), and deposition occurs during the falling limb of the hydrograph when there is a negative change in Q.

Morphological changes, i.e. rates of deposition and erosion, are a function of existing channel morphology, discharge and velocity, and relative timing of sediment supply waves (Lane et al., 1996). Channel slope, a key feature of channel morphology, affects the rate at which potential energy, and consequently erosive force, is dissipated throughout the channel length, from head to outlet. Headwater streams typically have steeper slopes than streams near the outlet (Easterbrook, 1999b). In graded streams, slope is adjusted in a cycle of dynamic equilibrium to provide the velocity required to transport sediment without net deposition or erosion. As Q increases, flow velocity increases, and causes erosion. As erosion of the stream channel progresses, it works to decrease slope, which in turn reduces velocity until a new equilibrium is established (Easterbrook, 1999b). Changes to channel dimensions (features of channel morphology) also effect erosion. As Q changes, channel depth (d), velocity (v), and width (w) change according to the equation $Q=w * \mathrm{d} * v$ (Easterbrook, 1999b). As stream width increases relative to depth, the flow velocity slows and settling velocity for fine particles may be reached, causing deposition (Nerbonne and Vondracek, 2001). However, widening and deepening processes are limited because they occur as a result of a negative-feedback system with stream power. These processes consume stream energy and subsequently reduce the ability of the stream to further widen and deepen the channel (Easterbrook, 1999b).

In conjunction with slope, bank soils and landscape position govern channel morphology. Upland channels tend to have coarse bed and bank material, while lowland channels and banks are made of finer materials. As a result, the flow velocity controlling alluvial processes of erosion and deposition depends upon the particle size of the eroded bank soils (Whiting et al., 1999). Thus, meandering streams typically form in cohesive soils having high silt and/or clay content. Furthermore, cohesive streambanks lead to wide, deep channels, and lower gradients compared to channels with less cohesive bank materials. Meanders form as turbulent flow deflects water against the bank, eroding a small amount of the bank. The water reflects back toward the opposite bank and material from upstream is deposited, establishing a positive-feedback system and creating a bend in the stream channel. As water comes around the bend it continues in a straight line until it encounters the outside bank. As a result, the outside bank, or cutbank, receives most of the force as flow velocity is highest at this point. As erosion of the cutbank continues, the meander curvature increases, producing conditions for further erosion. Eventually, meanders become so sinuous that two bends intersect, and an oxbow is formed (Easterbrook, 1999b).

Management Practices Affecting Streambank Erosion

The impact of sediment on aquatic habitats and populations and the implications of soil loss on agricultural productivity have been widely recognized by state and federal conservation agencies as major soil and water quality issues in agricultural watersheds (Wilson and Kuhnle, 2006). As a result, the influences of various management practices have been extensively examined. Despite extensive research, consensus on which management practices are best for stabilizing banks and reducing sediment delivery does not exist (Nerbonne and Vondracek, 2001). However, some general principles on the effects of cattle grazing and vegetation on streambank erosion processes have been established.

Cattle impact streambank erosion directly by changing soil structure and indirectly by effecting plant processes. Grazing cattle remove protective plant cover, compact soil, and add nutrients directly to streams (Walker et al., 2009). Consumption of vegetation decreases foliage and root production, and trampling of bank soils increases bulk density and erodibility, decreasing infiltration and percolation, and further inhibiting root production and subsequent foliage growth (Clary and Kinney, 2002). Grazing in riparian zones decreases streambank vegetation and increases slope instability (Nerbonne and Vondracek, 2001). Clary and Kinney (2002) found that heavy grazing in riparian zones caused surface depression, bank retreat, flattening of bank faces as a result of increased bank angle, and a decrease in root biomass. Roots decrease the volume of eroded soil and retard the rate and processes by which it is lost (Piercy and Wynn, 2008). Clary and Kinney (2002) concluded that excessive grazing removes protective vegetation and changes banks to a sloping profile. All of these effects work to destabilize streambanks (Belsky et al., 1999) and increase in-stream sedimentation (Walker et al., 2009).

Streambank vegetation has both hydrologic and mechanical impacts on streambank stability, but its affect can be either stabilizing or destabilizing (Pollen et al., 2007; Simon and Collision, 2002; Easterbrook, 1999a). Plants reinforce soil with their roots and decrease soil moisture content through evapotranspiration and interception. However, plants increase the infiltration capacity of soils and may create conditions of high near-surface moisture content during and after rain events. Additionally, the weight of trees growing on or near streambanks contributes to destabilizing gravitational forces (Pollen, 2007; Simon and Collision, 2002). Mechanically, soil is strong under compression and weak under tension, while roots are weak under compression and strong under tension. As a result, soils held together by roots have enhanced strength. Typically, it is expected that smaller roots contribute more strength per unit area than larger roots (Simon and Collision, 2002). Soil detachment rate decreases with increasing root area ratio. Piercy and Wynn (2008) found that streambanks with switchgrass roots had a reduction in soil volume eroded by scour. However, in a study by Simon and Collison (2002), it was found that the most reinforcement of streambanks came from a small number of large roots rather than a large number of small roots. Surcharge, a function of vegetation weight and the number of plants per unit area, increases normal stress and causes an increase in shear stress due to friction. As such, the effect of surcharge, whether stabilizing or destabilizing, depends on the bank slope. Sometimes surcharge created by trees is destabilizing because of steep bank slopes (Simon and Collison, 2002). However, in unsaturated banks, the stabilizing effects of vegetation usually outweigh the negative impacts of vegetation weight on streambank stability (Easterbrook, 1999a).

There is conflicting data on the relative ability of grasses, shrubs, and trees to prevent streambank erosion. Vegetation characterisitics such as root structure and density, evapotranspiration rates, and plant weight influence streambank stability. While grasses only stabilize material in the upper 50 cm of soil, trees have deeper roots and can stabilize much deeper into the soil profile. Under dry conditions, Simon and Collison (2002) found that trees increased bank stability as a function of hydrological effects, specifically matric suction. Wynn and Mostaghimi (2006) found non-forested stream bends had greater bank erosion than forested meander bends. Therefore, it has been suggested that the greatest bank stabilizing benefits will be achieved by a mix of tree species in combination with grasses along riparian zones (Simon and Collison, 2002).

Importance of Streambank Erosion at the Watershed Scale

Recent findings demonstrate that streambank erosion constitutes a significant proportion of the total erosion at the watershed scale. Many studies investigating streambank erosion rates have reported that bank sediments can supply between 30-80% of in-stream sediments (Amiri-Tokaldany et al., 2003; Lawler et al., 1999; Simon et al., 1996). Recent work in sediment source tracking using rare earth elements and radionuclides have similarly identified channel sediments as the source of 48-80% of suspended sediments during flow events (Wilson and Kuhnle, 2006). However, as the authors note, the radionuclide technique for identifying sediment sources is unable to distinguish between collapsed bank material and re-suspended bed material and in certain cases gully material (Wilson and Kuhnle, 2006). As such, results reported using radionuclide fingerprinting should be interpreted with caution. Because of the difficulty in measuring and estimating the rates and sources of erosion for entire watersheds, there are few studies comparing the contribution of streambank sediment to overland erosion on a mass basis at the watershed scale.

RISK ASSESSMENT MODEL

Pesticides are an essential tool to modern farmers (Flury, 1996; Schulz, 2004). However, the widespread and long-term use of pesticides in agriculture has led to extensive contamination of surface and ground water across the United States (Pereira et al., 1990; Squillace and Thurman, 1992; Kolpin, 1998; Lerch et al., 1998; Battaglin et al., 2003). Losses of pesticides translate into economic loss for farmers and pose potential risks to human and aquatic ecosystem health (Schulz, 2004; Flury, 1996; Cooper, 1993). Herbicide contamination of streams may result in toxic effects to aquatic photosynthetic species, such as algae and macrophytes (Cooper, 1993; Fairchild et al., 1998; Faust et al., 2002). Furthermore, the presence of pesticides in drinking water increases the likelihood of cancer and may result in sub-lethal endocrine disrupting effects, such as lowered fertility and sperm counts in men living in rural areas (Swan et al., 2003) and feminization of male frogs (Hayes et al., 2010).

Transport mechanisms

The fate of a pesticide applied to the soil environment is controlled by a variety of processes, including biodegradation, hydrolysis, photolysis, volatilization, wind erosion, and hydrologic transport. Volatilization and wind erosion transport pesticides into the atmosphere, potentially resulting in deposition many kilometers from the point of application (Majewski and Capel, 1996; McConnell et al., 1998). Among the fate processes, hydrologic transport of pesticides to surface and ground waters has been of particular concern because of its impact on water quality and human and ecological health. The three main hydrologic pathways are leaching, solution runoff, and particle adsorbed runoff (Pionke and Chesters, 1973).

Leaching is the downward movement of water through the soil matrix and into groundwater. A pesticide that is dissolved in percolating rainwater may be leached through the unsaturated zone and into the groundwater. Pesticides can reach groundwater either by moving through the bulk soil matrix, or by movement through preferential flow paths (Flury, 1996; Reichenberger et al., 2007). Movement of pesticides through preferential flow paths is responsible for rapid transfer of pesticides to groundwater. Movement through the bulk soil matrix, however, is a more continuous, slow process, that may take several years or decades for the pesticide to reach groundwater. Therefore, the risk of a pesticide leaching to groundwater is mainly a function of the physical properties of the soil that determine its hydraulic conductivity, and the chemical properties of the pesticide. Weakly sorbing, persistent pesticides, such as atrazine and the diketonitrile metabolite of isoxaflutole, have the highest risk for leaching while strongly sorbing, fast degrading pesticides, such as glyphosate, have the lowest risk for leaching (Pionke and Chesters, 1973; Reichenberger et al., 2007).

Surface runoff transports pesticides in solution and by erosion when the pesticide is adsorbed to soil, also known as particle adsorbed runoff (Poinke and Chesters, 1973). Runoff occurs when the infiltration capacity of the soil is less than the rate of rainfall (Hortonian runoff) or when the soil is completely saturated and cannot hold more water (Reichenberger et al., 2007). High intensity storms typically cause infiltration limited flow, while low intensity storms produce saturation limited flow. Particle adsorbed runoff is a multistep process. First, the pesticide must sorb to the soil particle, then the particle must be detached from the soil surface, either by raindrop impact or by scour from the runoff water, and then be entrained and transported (Reichenberger et al., 2007). Solution runoff is generally considered the more important of the two surface runoff loss pathways, as the amount of sediment produced by a runoff event is generally small in comparison to the volume of water (Wauchope, 1978). However, for strongly sorbing compounds, particle adsorbed runoff can be the main loss pathway (Pionke and Chesters, 1973). Intermediately sorbing compounds tend to be lost in solution runoff, as weakly adsorbing compounds are readily lost through leaching (Reichenberger et al., 2007).

Pesticide Transport Vulnerability

At the watershed scale, the vulnerability of pesticide transport to streams is governed by four factors: 1) pesticide properties, including chemical, physical, and formulation characteristics; 2) intrinsic factors, such as soil, landscape, and hydrologic properties; 3) anthropogenic actions, such as land use and management practices; and 4) climate effects, especially precipitation and temperature (Lerch and Blanchard, 2003). Chemical properties such as soil sorption intensity, acid/base dissociation constants, solubility, volatility, and persistence are important in determining the hydrologic transport pathway(s) for pesticide loss (Lerch and Blanchard, 2003). Letey and Farmer (1974) contend that the most important factor controlling the transport pathway is the adsorption coefficient. Pionke and Chesters (1973) determined that compounds strongly adsorbed to soil particles are lost mainly through erosion of sediment in runoff, rather than in solution. High adsorption coefficients predict that a chemical will be immobile in soil, making it vulnerable to losses through solution and particle adsorbed runoff rather than leaching (Letey and Farmer, 1974).

The ionic character of a pesticide is partly responsible for its adsorption properties. Pesticide electrostatic charge, resulting from differences in electronegativity among covalently bonded atoms in the molecular structure, and acid/base properties govern the intensity of the electrostatic interactions between a pesticide and soil colloids. Cationic compounds, such as paraquat and diquat, strongly bind to soils by cation-exchange to negatively charged organic matter (OM) and clays. Very weakly basic compounds (i.e., pK_a 2-5), such as the s-triazines and triazoles, can become protonated at low pH and adsorb to negatively charged clay and organic surfaces. Acidic compounds, like 2,4-D and dicamba, have carboxylic and hydroxyl functional groups that ionize and produce organic anions. Soil OM and clays are negatively charged at typical soil pH (i.e., pH > 5). Consequently, acidic pesticides do not strongly adsorb to OM or clays by electrostatic binding, but they may bind through other non-ionic mechanisms, such as ligand exchange. The pesticides that are nonionic have a wider range of chemical properties influencing adsorption to soil. Therefore, they may bind to soil through a variety of bonding mechanisms, including hydrophobic bonds, ligand exchange, ion-dipole, H-bonding, and van der Waals forces. In general, the adsorption of nonionic pesticides increases with increasing OM content (Weed and Weber, 1974), suggesting that hydrophobic bonding is the predominant bonding mechanism for non-ionic pesticides (Chiou et al., 1979).

While chemical properties of pesticides are important for determining transport pathways, they must be considered in the context of the soil in which they exist. Intrinsic factors such as pH, and clay and OM content contribute to pesticide behavior in the soil matrix. The specific interactions between pesticides and organic and inorganic soil constituents have been determined in model systems using infrared, Fourier-transformed infrared and mass spectroscopy (Bollag et al., 1992; Martin-Neto, et al., 2001). Using whole soils, binding mechanisms can be inferred by varying the chemistry of extracting solutions to disrupt specific bonding mechanisms and then measuring the extraction recovery of a pesticide from soil as an indicator of the importance of various binding mechanisms (Cheng, 1990; Lerch et al., 1997). Soil pH has often been termed the "master variable" of soil because it influences virtually all chemical reactions within the soil matrix. The interaction of soil pH with the dissociation constants of the soil colloids and pesticides will largely determine the extent of electrostatic interaction between the pesticide and the soil. Soil pH also controls chemical and biological degradation rates of many pesticides. For example, pH is the most important factor determining the rate of atrazine degradation in soil (Houot et al., 2000). Overall, adsorption of pesticides to soil particles reduces their concentration in the soil solution, thereby decreasing pesticide efficacy, reducing bioavailabilty, enhancing chemical degradation, and retarding leaching (Weed and Weber, 1974).

Anthropogenic factors such as type of crop, tillage, and pesticide application rates impact the type and amount of pesticide use within a watershed. While conservation and no-till practices have been encouraged in recent years to prevent soil erosion from fields, no-till crop production can cause increased herbicide losses compared to tilled systems (Flury, 1996; Ghidey et al., 2005; Shipitalo and Ownes, 2006). This is because pesticides that are not incorporated into the soil will sorb less intensely due to less interaction with the soil, resulting in greater losses by solution runoff. Application form affects the vulnerability of pesticides to hydrologic transport based on the placement of the herbicide (soil or foliar application) and the formulation (spray or granular) by affecting the degree to which the pesticide will interact with soil or be exposed to other dissipation processes, such as volatilization or photolysis. Application timing (i.e., pre-plant vs post-plant) will affect the probability of pesticide losses in surface runoff due to its interaction with climate.

For a given field, pesticide losses in surface runoff are mainly a function of the timing of rainfall relative to application (Wauchope, 1978; Lerch and Blanchard, 2003; Ghidey et al., 2005). Rainfall events shortly after application typically transport the highest loads and concentrations of pesticides (Wauchope, 1978; Ghidey et al., 2005). Pesticide losses increase with storm size but concentrations in surface runoff may decrease if extremely high runoff volumes dilute pesticide concentration (White et al., 1967). As scale increases, the risk of stream contamination by pesticides from any one runoff event decreases due to the temporal variation in pesticide application and non-treated areas. At the watershed scale, weakly adsorbing, stable compounds with high use, such as atrazine, have the highest loads in streams (Lerch and Blanchard, 2003). Rainfall events occurring shortly after application will transport weakly adsorbing compounds in solution to a greater degree than strongly adsorbing pesticides. These losses are influenced by application form and generally show an exponential decrease in concentration as a function of time after application (Poinke and Chesters, 1973; Wauchope, 1978; Ghidey et al., 2005). At the watershed scale, the vulnerability to pesticide transport has been shown to be strongly affected by soil properties. Lerch and Blanchard (2003) showed that hydrologic soil group, which indicates the relative runoff potential of a soil, was strongly correlated to herbicide transport across the northern Missouri/southern Iowa region.

Cost-effectiveness, ecological impact, maintenance requirements, and other environmental concerns (e.g., erosion, nutrient transport) should be considered when implementing mitigation efforts for decreasing hydrologic losses of pesticides (Reichenberger et al., 2007). There is a dilemma in that implementing measures to prevent or reduce one problem may increase or create another problem. For example, soil incorporation has been shown to significantly reduce atrazine transport in surface runoff (Ghidey et al., 2005), but more extensive tillage will lead to greater soil erosion. In addition, conservation tillage and no-till systems that preserve soil structure will decrease surface runoff in many well drained soils, but for soils with restrictive layers that reduce percolation, reduced tillage or no-till will not reduce runoff volume, increasing the risk of transport in runoff for soil-applied herbicides that are not incorporated (Ghidey et al., 2005). For well drained soils in which the runoff volume is reduced by conservation tillage systems, the risk of pesticide transport by leaching increases (Flury, 1996). Thus, any proposed management "solution" to reducing pesticide transport must take into consideration these types of trade-offs. The goal must be to find the optimal practice within the context of the cropping system and soil properties that will minimize these trade-offs. In addition, because resources for establishing BMPs are limited, it is all the more imperative that managers have the ability to target vulnerable areas within a field or watershed to establish practices that will produce the greatest effect without exacerbating other environmental concerns.

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CHAPTER 2: STREAMBANK EROSION PROJECT

ABSTRACT

Sedimentation of surface waters in the United States is a significant environmental concern. Investigating land use impacts on streambank erosion rates is intended to lead to the development of improved management practices and provide the basis for targeting the placement of management practices to mitigate this problem. The overall objective of this research was to determine the effect of stream order, adjacent land use, and season on streambank erosion rates. Study sites were established in 2007 and 2008 within Crooked and Otter Creek watersheds, two claypan watersheds located in northeastern Missouri. Detailed site information was recorded, including eroded streambank length, soil descriptions, gullies, debris dams, cattle access areas, and point bars. A factorial experimental design was implemented with four land uses (cropped, forest, pasture, and riparian forest) and three stream orders (1^{st}, 2^{nd}, 3^{rd}). Each treatment was replicated three times for each stream order, except for the cropped 3^{rd} order treatment as only one suitable treatment site could be found. To measure bank erosion/deposition rates erosion pins were installed based on bank height and length at each site. The effect of different seasons was assessed by measuring the length of the exposed pins three times per year (March, July, and November). The carbon and nitrogen content of bank material was also determined. Erosion rates were consistently higher in the winter months than spring/summer and fall seasons; however, the significant three-way interaction indicated that the treatments did not respond in the same manner across

seasons. Soil nutrient concentration data showed that forest sites had significantly lower C and N concentrations that other land uses. Analysis of the sediment loss rates showed the seasonal effect and three-way interactions were highly significant. Streambank erosion rates were applied to the entire study area to estimate the contribution of streambank material to in-stream sediment at the watershed scale. Estimates showed that bank sediment accounted for 47-71% of the total in-stream sediment. These results indicate that steambanks are a significant, if not dominant, source of sediments in streams of claypan watersheds.

INTRODUCTION

Sedimentation is a significant environmental problem, with estimated mitigation costs of $16 billion annually in North America (Wynn and Mostaghimi, 2006), and streambanks contribute substantially to in-stream sediment (Zaimes et al., 2006; Fox et al., 2007; Piercy and Wynn, 2008). Simon et al. (1996) reported that streambank erosion accounted for up to 80% of the in-stream sedimentation in the Forked Deer River system in western Tennessee. Recent work utilizing stable isotopes has shown that streambank erosion can account for the majority of suspended sediment present in streams during high flow conditions (Wilson et al., 2008). The impact of sediment and associated nutrients on aquatic habitats (Shields, 1994) and the implications of soil loss on agricultural productivity have been widely recognized by state and federal conservation agencies as major soil and water quality issues in agricultural watersheds (Wilson and Kuhnle, 2006). As a result, the influences of various management practices have been extensively examined. However, despite extensive research, a general consensus on which management practices are best for stabilizing banks and reducing sediment delivery has not emerged (Nerbonne and Vondracek, 2001). In addition, because of the runoff-prone nature of claypan soils, it has been widely assumed that overland erosion from cropped fields is the primary source of sediment in the streams of claypan watersheds (Jamison et al., 1968). Therefore, no published information has been reported on streambank erosion and its contribution to the sediment and nutrient load in claypan watersheds.

As researchers and management specialists look to understand the causes and effects of these water quality issues, the need to fully understand the sources of in-stream sediment are critical to improving water quality at the watershed scale. Investigating the impacts of adjacent land use on streambank erosion rates is intended to lead to development of improved management practices and provide the basis for targeting the placement of "best management practices" (BMPs) within watersheds to mitigate streambank erosion. The overall objectives of this study were to investigate the effects of adjacent land use, season, and stream order on streambank erosion rates in two watersheds of the Central Claypan Region of northeastern Missouri, and to quantify the total contribution of stream bank material to in-stream sediment and nutrients. The null hypothesis for this research was that streambank erosion would not be a function of stream order or adjacent land use.

MATERIALS AND METHODS

Site Description

Erosion pins were used to measure streambank erosion rates based on adjacent land use, season, and stream order. Study sites were established in Crooked and Otter Creek watersheds, located in northeastern Missouri within the Salt River Basin (Figure 1). The Crooked Creek watershed is 284 km^2 with 56% of the area used for cropland, followed by pasture (26.5%) and forest (14.5%). The Otter Creek watershed encompasses a 271 km^2 area, with 64.6% in cropland, 20.3% in pasture, and 12.6% in forest (Lerch et al. 2008). These watersheds were selected because they are representative of the intensively row-cropped claypan watersheds of Major Land Resource Area 113 (Central Claypan Region) (USDA-NRCS, 2006; Lerch and Blanchard, 2003; Lerch et al., 2008).

Experimental Design

A factorial experimental design was implemented to evaluate the effects of land use, stream order, and season on streambank erosion of 1st, 2nd, and 3rd+ order streams. The 3rd+ category contains a combination of 3rd and 4th order streams as the extent of 3rd order stream reaches was very limited within these watersheds. Land use treatments included crop, forest, pasture, and riparian forest. Forest sites had a minimum 30 m wide stand of trees on each side of the bank, while riparian forest sites had 10-30 m wide tree stands. Erosion pin data were collected three times annually. The three seasons were defined as follows: Season 1-December-March; Season 2-April-July; and Season 3–August-November. Six sets of seasonal measurements, two for each season, were made since 2008. Each treatment was replicated three times with some exceptions for some treatments across seasons. Exceptions were a result of not having the site established at the time measurements began, or in a few cases because of restricted landowner access during measurement periods. Additionally, only one 3rd order crop site was established as two other suitable treatment sites could not be located. These missing values resulted in an unbalanced experiment (see Data Analysis).

Site Set-up

Site selection was based primarily on the presence of a given land use treatment. Site lengths ranged from 300-425 m, with continuous land use on both sides of the stream. Stream order was determined using the National Hydrography Dataset (Dewald and Roth 1998), and in some cases modified after ground truthing. To determine the total eroded bank length, surveys were conducted using hand-held Global Positioning System units (GPS) (Juno ST, Trimble Navigation Ltd, City, State, or Dell X51 with GlobalSat BC-337 Compact Flash GPS Receiver, Dell Computers Inc., Round Rock, TX) and Trac-Mate software (Farm Works Software, Version 12.16, CTN Data Services, Inc., Hamilton, IN). Distances between eroding and non-eroding sections along each bank were recorded. Eroding banks were identified based on criteria from the U.S. Department of Agriculture - Natural Resource Conservation Service (USDA-NRCS, 1998), which were developed for calculating erosion and sediment delivery based on visual inspection of streambanks. This approach has been used in previous streambank erosion studies (Zaimes et al., 2006, Berges, 2009). Banks identified as eroding possessed one or more of the following characteristics: 2/3 of the bank face devoid of vegetative growth or roots; less than 1/3 of bank face protected by roots; overhanging vegetation with eroded undercut face; near vertical slope; and apparent bank failures, such as slumps and slides.

Summary characteristics for each site are listed in Table 1. The site length listed in Table 1 represents the total length of streambank for each site (i.e., twice the stream reach length); the eroded lengths were computed based on the total streambank length.

After surveys were conducted, each 300-425 m reach was sub-divided into four equal length sub-reaches. The total eroded length for each sub-reach was then calculated using the GPS data from the bank surveys. Pin plot placement was randomly assigned within each sub-reach, with pins installed in at least 20% of the eroded length for that sub-reach. Erosion pins made of rolled steel (76.2 cm long and 6.2 mm diameter) were then installed perpendicular to the bank face at each site to measure streambank erosion. In total, there were 3,150 individual erosion pins installed. The pin arrangement for each plot was based on bank height. For banks of 1 m or less, pins were placed approximately at half bank height. For banks greater than 1 m but less than 2 m, pins were placed at 1/3rd and 2/3rd bank height. Three rows of pins were installed for banks over 2 m, with rows at 1/4, 1/2, and 3/4 bank height. Laterally, pins were spaced 2 m apart. An example of pin placement on a 3rd order stream is seen in Figure 2. Pins were inserted horizontally into eroded banks with 10.2 cm of the pin exposed. Pins were then spray painted to aid location of the pins for subsequent measurements. Each pin arrangement was recorded, and a GPS coordinate was obtained at the downstream end of each pin plot as an aid for locating the plots. Methods reported here were similar to those reported by Zaimes et al. (2004, 2006, 2008), Berges (2009), and Willett et al. (2009).

Data Collection

Pin plots were installed beginning in late June through August 2007, with exceptions as previously noted. Each pin was measured for deposition or erosion of bank material three times annually by measuring the length of the exposed pin. Pins that had been completely eroded were recorded as 0.65 m, which was considered reasonable based on the recommendations of past studies (Lawler, 1993). Additionally, due to the highly cohesive nature of these clayey bank soils, the 0.65 m length is believed to be a conservative estimate, as several pins were actually recorded at lengths approaching 0.70 m and were still held in the banks. Where deposition occurred, negative pin lengths were recorded. The inclusion of negative erosion data is seen as the best approach to calculating the actual amount of erosion, as omitting negative readings or replacing negative readings with zeros artificially increases the calculated erosion rate (Couper et al, 2002). A length of -0.012 m was used for pins that were completely buried. Pins that could not be relocated and were without evidence of either erosion or deposition were recorded as missing and replaced with a new pin. Bank area of each pin plot was computed based on height and length measurements. Analysis includes six sets of data, two for each season, from Season 1-2008 to Season 3-2009.

Bulk density and nutrient samples were taken over the course of the summer and fall 2008. Fifty percent, or a minimum of three, of the plots at each site were sampled. Bulk density cores and soil samples were collected for C and N analysis for each identified soil horizon. Soil profile descriptions were recorded for each horizon from which a corresponding bulk density and soil sample was collected. Horizon descriptions included general soil color (which included redoximorphic features), texture, structure, and horizon depth. Complete horizon descriptions are recorded in Appendix A. Bulk density samples were oven-dried at 110ºC for a minimum of 72 hours, brought to room temperature in a desiccator, and weighed. Bulk densities were computed on a depth-weighted basis for each pin plot, and then averaged over the number of plots sampled to obtain a depth-weighted average for each site. Soils collected for C and N analysis were air-dried, sieved to 2 mm, then crushed using a mortar and pestle. Prior to the total C analysis, a representative sub-set of samples was tested for the presence of carbonates using effervescence with HCl as an indicator. None of the tested samples effervesced, and therefore, it was concluded that carbonates were not present in any of the samples. Crushed, air dried samples were then analyzed for C and N content using the LECO TruSpec NC Carbon/Nitrogen Determinator (St. Joseph, MI). Combustion was at 950° C. An infrared radiation cell was used to determine C content and a thermal conductivity cell was used to determine the N content on a percent basis. Moisture content of air dried samples was determined for each sample by drying at 105-110°C for a minimum of 48 hours so that the C and N data could be corrected to a dry weight basis.

Sediment and Nutrient Loss Calculations

Following computation of the net pin length for a given seasonal data set, the mass of eroded or deposited bank sediment was computed based on the average net pin length (m) for an entire pin plot multiplied by the plot area (m^2) and average site bulk density (kg m^{-3}). This mass (kg) was then divided by the pin plot length (m) to give a linear erosion rate (kg m^{-1}) for each pin plot. The average linear erosion rates of the pin plots were then multiplied by the total eroded length (m) of each site (based on the initial GPS surveys described above), giving the total mass (kg) of eroded or deposited sediment for the site. The linear erosion or deposition rate of each site was then computed by dividing the total eroded or deposited mass for a site by the total site length (Table 2.1), giving the linear erosion or deposition rate (kg m^{-1}) for the entire stream reach for each treatment. For a hypothetical example of the streambank erosion rate calculation see Appendix B. To compute C and N loss rates, the concentration data (g kg^{-1}) were multiplied by the linear erosion rate (kg m^{-1}) to obtain the nutrient loss rate (g m^{-1}) for each site.

Data Analysis

Sediment Loss Rates

As discussed above, the experimental design was a three-way factorial with main factors of land use, stream order, and season. The analysis of variance (ANOVA) model for the sediment loss rates included all levels of interaction, and was performed using SAS v9 statistical software (SAS Institute Inc., Cary, NC, USA). The MIXED PROC ANOVA was used with the REPEATED statement included to account for the differences in variability within and between sites resulting from repeated measurements at the same site. Allowance for dependence among the 6 observations (three seasons over 2 years) was accomplished by using an unstructured correlation model. After the ANOVA model was fit (Model 1), the residuals were examined to assess the assumption of normality. Also, because the data are unbalanced, least squared means (LS means) were used to test for pair-wise differences between levels of one factor when the levels of the other two factors were held fixed. LS means are to unbalanced data what arithmetic means are to balanced data. Because p-values were calculated based on LS means, mean values reported will be LS means, and graphs and figures will also be based on LS means, unless otherwise noted. In view of the large number of comparisons, a more stringent $\alpha = 0.01$ was used to determine significance of the pair-wise comparisons rather than the more commonly used α=0.05. For main effects and interactions, $\alpha = 0.05$ was used to determine statistical significance.

Analysis using all of the sediment loss data showed that the residuals, though not highly skewed, had a distribution that may not be normal. Consequently, a second analysis (Model 2) was done using the transformation Y=log(50+erosion rate). Due to the occurrence of net deposition for some sites and seasons (i.e., a negative erosion rate), a value of 50 was used to force all the transformed data to be greater than zero. Analysis of the residuals from this model showed two outliers (standardized residuals greater than 3 in absolute value). When the model was run with these outliers excluded, the resulting residuals were normally distributed. A conservative approach was taken such that main effects, interactions, and pair-wise comparisons were considered significant only if they were significant for both models (Model 1 and 2).

Nutrient Concentrations

The nutrient concentration data were analyzed using a MIXED PROC ANOVA procedure in SAS, and pair-wise comparisons were done using LS means. The data contained no outliers or extreme values. However, because of missing 3rd order crop replicates the data set is unbalanced, requiring the use of LS means rather than arithmetic means. The initial analysis was run including the two-way interaction term (land use by stream order). Once it was determined that the interaction was not significant for either the C or N data, simple main effect models were run. The C and N concentration data are listed by site in Table 2.1.

Nutrient Loss Rates

The same statistical analysis used for the sediment loss rates (discussed above) was used to analyze the C and N loss rates. Since the nutrient loss rate data also contained outliers, two models were run (as described above for the sediment loss data), one with all the data (Model 1) and the other with outliers omitted to meet the assumption of normality (Model 2). Only main effects, interactions, and pair-wise comparisons that were significant in both models (based on $\alpha = 0.05$ as described above) were considered statistically significant.

RESULTS AND DISCUSSION

Sediment Erosion Rates – Main Factors

The arithmetic means for each main effect are graphed below (Fig.2.3-2.5). The mean erosion rates for the land use treatments, averaged across season and stream order, were 34 kg $m^{-1}yr^{-1}$ for crop, 44 kg $m^{-1}yr^{-1}$ for forest, 54 kg $m^{-1}yr^{-1}$ for pasture, and 51 kg m^{-1} yr^{-1} for riparian forest. Mean erosion rates for the stream orders, averaged across season and land use, were 44, 43, and 54 kg $m^{-1}yr^{-1}$ for 1^{st}, 2^{nd}, and 3^{rd} order streams, respectively. Seasonal means, averaged across land use and stream order, were 96 kg m^{-1} for Season 1, 36 kg m^{-1} for Season 2, and 8.5 kg m^{-1} for Season 3. Statistical differences of the LS means for land use, stream order, and season are discussed in the "Statistical Analysis" section. These erosion rates were similar to those found in other studies (McGreal and Gardiner, 1977; Gardiner, 1983; DeWolfe et al., 2004; Zaimes et al., 2006). Zaimes et al. (2006) reported erosion rates of 5-304 kg m^{-1} yr^{-1}in Iowa streams with various riparian land uses. Likewise, DeWolfe et al. (2004) reported erosion rates of 10-447 kg m^{-1} yr^{-1} in Vermont watersheds ranging in size from 108-202 km^2 with grass, pasture, and forest riparian land uses. Average erosions rates for this study, when expressed in m yr^{-1} were also within the range of reported values (0.004-0.387 m yr^{-1}) from erosion pin studies in watersheds of similar size (52 km^2 – 100 km^2) (Gardiner, 1983; McGreal and Gardiner; 1977).

Sediment Erosion Rates – Statistical Analysis

The results of the three-way ANOVA for the sediment erosion rates are summarized in Table 2.2. Season and the three-way interaction between season, land use, and stream order were found to be significant, but land use and stream order effects were not significant. In addition, none of the two-way interactions were found to be significant. Part of the explanation for the significant differences observed was the overwhelming importance of season on streambank erosion rates, which masked the land use and stream order effects within a given season. Also, others have noted that the highly variable nature of streambank erosion data makes it very difficult to discern treatment differences (Belsky, 1999). Furthermore, not only were the erosion measurements highly variable among the treatments, key sources of variation, such as upstream land use and drainage area, cannot be controlled. To sufficiently analyze the three-way interaction, pair-wise comparisons were made by varying one of the effects while the other two effects were fixed (Figs 2.6 and 2.7). When considering the significant interaction between season, stream order, and land use some pair-wise comparisons were significant.

Land Use- Land use was not a significant main effect in the three-way ANOVA, and only one pair-wise comparison was significant. In Season 2, 2^{nd} order crop sites had significantly lower sediment rates (LS mean= -32.2 kg m^{-1}) than 2nd order forest sites (LS mean= 107.3 kg m^{-1}) (Fig. 2.6b). Pair-wise comparisons indicated that this was the only combination of season, stream order, and land use that had a significant land use effect, and this was likely the result of site-specific differences rather than any true land use effect. The lack of significant land use effects was most likely a result of variation among sites in their upstream land uses, total drainage area, drainage network (related to watershed shape), and up-stream channel morphology. Currently, the drainage area, drainage network, and land use data upstream of all sites are being compiled to assess variations in these factors and how they may affect the observed erosion rates. Until these data are compiled, the effect of these factors on streambank erosion as a function of adjacent land use cannot be discussed in any specific manner. However, the variable response of the land use treatments to stream order and season (Figs. 2.6 and 2.7) strongly suggested that these upstream factors had more impact on streambank erosion rates than adjacent land use.

While this study showed non-significant differences between land use treatments, past studies have found significant differences between land uses. Zaimes et al. (2004) found that 2nd order streams flanked by row-crop and pasture fields had higher erosion rates than those with adjacent riparian forest buffers. Other researchers found that the presence of cattle negatively affects the ability of plant roots to hold soil and trampling on and along banks causes destabilization (Belsky et al., 1999). Pasture sites in this study were observed to be more prone to slumping because of cattle access to the streams. Furthermore, pasture sites had few trees, and they were generally not large trees that impart stability to the streambanks.

Stream Order- Stream order was also not a significant main effect. Additionally, there were no significant pair-wise comparisons between stream orders when land use and season were fixed and stream order was varied. The National Hydrography Data Set (Dewald and Roth 1998) uses the Strahler stream order system, which assigns stream order based on tributary hierarchy (Strahler, 1957). While stream order is roughly correlated to the size of the channel, the assignment of a stream order number is mostly independent of other factors controlling stream bank processes such as channel shape, watershed area, sinuosity, etc. Other than their relative place within the hierarchy of the drainage network, streams of the same order often share little in common regarding the upstream characteristics that may control streambank erosion. There may be incidental similarities in channel size and drainage area for like-ordered streams, but those similarities are not guaranteed since assignment of the stream order in the Strahler system is solely based on the channel's location within drainage network hierarchy. Therefore, like-ordered streams may exhibit any number of combinations of the factors that control stream bank processes. From a hydrologic perspective, streams of dissimilar orders may be more alike with respect to the factors controlling streambank erosion than streams of the same order. For example, several of the study sites that were designed as 1st order streams had channel dimensions that were as large as some of the 2nd order study sites. Given the similarity in erosion rates as a function of stream order (Fig. 2.4), it appears that stream order designations have little relationship to streambank erosion processes, and this method of categorizing streams was not a useful basis for predicting streambank erosion rates.

Season- Seasonal effects were found to be highly significant ($p<0.001$). While this result may seem straight forward based on the arithmetic means (Fig. 2.5), the significant three-way interaction among main effects indicates that a more complex interpretation of the effect of season on erosion rates must be considered. While some confusion exists in the literature regarding the interpretation of significant interaction effects in factorial designs (Jaccard, 1998), the interpretation used here was that if a significant interaction between two or more main effects occurred, then the main effect cannot be interpreted unambiguously (Maxwell, 1990). Because a significant interaction effect implies that the effect of one factor is not consistent across all the combinations of the other two factors, it is preferable to interpret the significant main effects only in terms of the significant three-way interaction (Maxwell, 1990). Therefore, to understand the complexity of the season by land use by stream order interaction, the pair-wise comparisons must be considered (Figs. 2.6 and 2.7).

For all significant pair-wise season comparisons, of which there were six, Season 1 was greater than either Season 2 or 3 (Figs 2.6 and 2.7). This supports the general trend seen in Figure 2.5 where the average erosion rate during Season 1 is numerically much larger than Season 2 or 3. However, it should be noted that this trend is only statistically significant for some combinations of land use and stream order. Figures 2.6 and 2.7 illustrate the complexity of the seasonal effect with respect to land use and stream order. Note that while Season 1 often has greater erosion rates than Season 2 and 3, it was not always significantly greater, nor is it numerically greater for every combination. In general though, the pair-wise comparisons support the general trend observed in Figure 2.5.

There is a large literature base supporting the conclusion that season is a significant effect, and the findings of this study were congruent with past studies. Wolman (1959) observed the greatest erosion rates during the winter months (December-March) and lower erosion rates in the summer. Zaimes et al. (2006) report similar results with the largest magnitude erosion occurring in the spring and early summer and little erosion occurring in the fall. Lawler et al. (1999) likewise observed higher rates of erosion in the winter months, commenting that most erosion occurs between December and March. Wolman (1959) attributed some winter erosion to freeze thaw mechanisms but concluded that winter erosion was largely a result of high flow events occurring when the bank soils were already "thoroughly wetted." Flow events in summer months occurred when bank soils were dry and therefore did not produce erosion rates that were as large as those seen in the winter. Even when summer flow events greatly exceeded those occurring in the winter (Fig. 2.8), the resulting erosion was less (Wolman, 1959). Furthermore, a study by Hooke (1979) revealed that while soil moisture was the most important factor controlling streambank erosion, significant erosion only occurred in association with peak discharge. Zaimes et al. (2006) identified precipitation pattern as a major factor contributing to the seasonal effect on bank erosion, finding that most erosion occurred following many medium-sized precipitation events or two large precipitation events that occurred close together. This pattern fits that described by Wolman (1959), where banks become "thoroughly wetted" during the first precipitation event and then were eroded during subsequent precipitation events. Lawler et al. (1999) described winter conditions (i.e., high frequency and large magnitude events, freeze/thaw cycles, high antecedent moisture conditions, and lack of vegetation) as the "optimum combination" for producing large amounts of bank erosion, and the results of this study support that conclusion.

Considering the erosion data in light of the discharge data for Crooked Creek (U.S. Geological Survey, 2009) (Fig. 2.8) reveals an extreme example of the Wolman pattern, where many moderate-sized events in Season 1-2008 and very infrequent low magnitude events during Season 1-2009 produced considerably more erosion than that caused by the frequent medium to high magnitude events that occurred in Seasons 2 and 3 of 2008 and 2009. The high frequency of summer events suggested that wet bank conditions would persist through Seasons 2 and 3 of 2008 and 2009, and the disparity between seasonal erosion rates was therefore not entirely explained by the Wolman pattern. Zaimes et al. (2006) attributed additional seasonal differences in erosion rates to differences in vegetative cover density. Lack of vegetation in the winter leaves banks completely exposed and vulnerable to scour, while lack of evapotranspiration prevents banks from drying. This combination maintains saturated conditions and weak cohesion of bank material, resulting in a high degree of vulnerability to erosion during the winter months (Lawler et al., 1999). In contrast, dense vegetative cover in the summer months protects banks from scour and evapotranspiration dries bank soil, mitigating the erosive effects of even an exceptionally wet year with multiple sequential flow events in Seasons 2 and 3 (Fig. 2.8).

Nutrient Concentrations

The results of the carbon and nitrogen nutrient concentration data analysis indicate that land use was a significant effect. Stream order and the interaction term (land use by stream order) ($p = 0.48$ for C and 0.40 for N) term were not significant for either the C or N concentration data (Table 2.3). Pair-wise comparisons showed that forested sites had a significantly lower C and N content than crop sites at the α =0.01 level. The difference between forest sites and pasture and riparian sites was more marginal, but a definite trend was seen in the data. At α=0.05, forest was significantly lower than pasture for both C and N concentration. Forest sites were significantly lower in C content than riparian sites and lower, but not significantly, than riparian sites for N content (Table 2.4).

Nutrient concentration of streambank materials is controlled in large part by flooding regime as this mechanism controls deposition of alluvial materials. The frequent and flashy flooding that is prevalent throughout the watershed is common among all riparian land uses in these watersheds and the common flooding regime is likely to account for similarities between land uses. The factors working to modify alluvial deposits and therefore the factors that account for differences between land use include local vegetation, nutrient cycling efficiency, and a combination of adjacent land use and proximity to eroding fields. Because the forest treatment was defined as having at least a 30 m wide stand of trees along the streambanks, these sites by design lack proximity to surrounding crop and pasture land. As such, the nutrient inputs from adjacent crop and pasture fields would impact the other land use treatments much more than the forested treatment sites because surface runoff would have a longer and more circuitous route to the streambanks of the forested sites than any other treatment site. In addition, the opportunity for nutrient uptake or N losses by denitrification would be greater for the forested sites because of the extent to which these sites were buffered from the effects of adjacent agricultural land use. This lack of proximity may account in part for the lower N concentration of the forest bank materials. Walker et al. (2009) found that restored riparian zones, which were previously degraded by cattle grazing, had lower N concentrations than grazed sites and were highly efficient at reducing inorganic N contributions to adjacent stream water. It has further been suggested that even subtle disturbances in riparian areas can influence nutrient cycling in a watershed (Bolstad and Swank, 1997; Walker et al., 2009), and the conversion of forested land to row-crop field and pasture in these watersheds over the last 100 years certainly represents major disturbance to the riparian areas of these watersheds.

Nutrient Loss Rates

The ANOVA for the C and N loss rates showed that the season effect, stream order by season, and the three-way interaction were significant (Table 2.5). The significant pair-wise comparisons (where two factors are fixed and the third is varied) showed similar results to that of the sediment loss rates in that when differences existed, Season 1 was greater than Season 2 or 3. Of the eight significant pair-wise comparisons for season, six indicated that Season 1 was greater than either Season 2 or 3. The other two significant pair-wise comparisions showed that 2nd order forest and 1st order pasture sites had greater erosion in Sesaon 2 than in 3. These relationships are congruent with the sediment loss rate data (Fig. 2.5), where Season 1> Season 2> Season 3, which leads to the conclusion that the seasonal effect on soil loss also dominated the nutrient loss rates as well.

Applying Results to Watershed Scale

Overland erosion has historically been the primary focus of erosion control efforts in the Central Claypan Region. Thus, a quintessential goal of this work was to determine the relative importance of streambank erosion to the total in-stream sediment at the watershed scale. The National Hydrography Data Set (Dewald and Roth, 1998) was used to calculate the total length of streams in the watershed by stream order. There are a total of 1,365 km of streambank length in the study area (streambank length= total stream length x 2 banks). Applying the sediment and nutrient erosion rates for each stream order and multiplying by the length of that stream order in the study area gives an estimate of the mass of streambank sediment or nutrients that were deposited in the streams of these watersheds (Table 2.6). The arithmetic mean linear erosion rates for each stream order were used to compute the total streambank contribution to in-stream sediment. Although the estimates of sediment loss rates were not significantly different between stream orders, they were numerically different. Therefore, sediment and nutrient loss rates were based on stream order to provide the best estimate of erosion rates at the watershed scale as opposed to applying a single grand mean for the entire study area. Using this approach, the total amount of sediments from banks was estimated to be 63,000 Mg yr^{-1}, which contained an estimated 730 Mg C yr^{-1} and 60 Mg N yr^{-1}.

When considering the contribution of streambank material as a percent of the total in-stream sediment, a simplifying assumption was that overland erosion and streambank erosion were the only two sources of sediment in the watersheds. Gullies act as a conduit for delivering overland sediment to streams, so their contribution to in-stream sediment is accounted for in the overland erosion estimate. However, watersheds with actively incising and down-cutting streams need to include bed material as a source of sediment. The streams in the Crooked and Otter Creek watersheds are in the degradation and widening phase, or Stage III and Stage IV in channel evolution (Schumm et al., 1984).

However, the material that is being eroded is likely not newly excavated material. Following the clearing of the land for cultivation the stream system would have experienced a large in-flux of overland sediment. This sediment, along with the constant sediment supply from overland and bank sources, protects the bed from degradation as the stream uses all its power to deal with these sources of sediment. Very rarely is the substratum even exposed to be vulnerable to erosion. Additionally, there is so little relief in these watersheds and so much sediment from bank and overland sources that there is little energy available for degrading the bed. Lastly, Mark Twain Lake, which is located at the outlet of the two streams, has effectively raised the base level as the dam is some 15-30 meters high. Thus, as the stream cannot degrade past the level where it meets the lake, the potential for down-cutting is limited, and thus the contribution of stream bed degradation to in-stream sediment is considered negligible. Therefore, overland erosion and streambank erosion were considered to represent the only major sources of sediment in the system.

To estimate overland erosion the tolerable soil loss rate, or the T-value, was used. For the study area, the T-value is estimated to be 7.6 Mg $ha^{-1}yr^{-1}$ (USDA-NRCS, 2000). While the T-value represents the maximum amount of erosion that can occur and still sustain high level of crop productivity, much of the land in the study area exceeds this sustainable threshold. Data from the USDA-NRCS (2000) was used to estimate erosion from crop land that exceeds the T-value. Conservative estimates of rates less than the T-value were used for non-cropped lands. Table 2.7 contains the total and percent area of each land use and the assumed erosion rate applied as a fraction of the T-value for each land use within the watersheds. From these assumptions, total overland erosion was estimated at 507,000 Mg yr^{-1}.

For streambank erosion, it is reasonable to assume that 100% of bank sediments are delivered to the stream. However, to estimate the percentage of overland sediment that reaches the stream, a sediment delivery ratio (SDR) must be applied. SDR is the ratio of annual sediment yield to annual gross erosion (USDA- Soil Conservation Service, 1983), or more broadly, the percentage of sediment that arrives at a point of reference in comparison to the gross erosion that occurred above that point (Roehl, 1962; Duijsings, 1986). SDR is influenced by any number of watershed characteristics including precipitation patterns, channel density, topography, vegetation cover and type, land use, soil properties, and watershed size and shape (USDA- Soil Conservation Service, 1983; Walling, 1983). There are many models that compute SDR based on watershed area alone, and while this may be an over-simplification it is justified as size may be considered a "composite variable" that incorporates many of the individual effects of topographic, climatic, and geologic variables (USDA-Soil Conservation Service, 1983). In general, as drainage area increases the SDR decreases (e.g., Roehl, 1962; Robinson, 1977). However, any estimate of SDR based on drainage area only should be considered in light of the other topographic, hydrologic, geologic, and climatic characteristics of the watershed (USDA-Soil Conservation Service, 1983). Thus, other approaches to computing SDR should be considered (Maner, 1958; USDA-Soil Conservation Service, 1983). To incorporate a range of models available to estimate SDR, two models based on relief and length were also employed (Maner, 1958; Roehl, 1962; USDA-SCS, 1983). Review of existing literature found no approaches specifically designed for estimating

SDR in claypan watersheds. Given that none of the available means for estimating SDR have gained prominence as the preferred method, the approach used here was to apply several models to calculate a range of estimates for the SDR of these watersheds. The models used for calculating the SDR are listed in Table 2.8. Sediment delivery ratio estimates ranged from 2.7 to 13%, with an average of 9.4% ± 4.6% (standard deviation) for a 275 km^2 (100 mi^2) watershed. Using the average SDR combined with the estimated overland erosion rates, resulted in an estimated 48,000 ± 23,000 Mg yr^{-1} of overland sediment reaching the stream channels. Based on the estimated annual streambank erosion rate given above (63,000 Mg yr^{-1}), streambanks accounted for 45-71% (mean of 58%) of the total in-stream sediment in the study area (Fig. 2.9). Using the low end of the overland erosion contribution, the ratio of streambank to overland in-stream sediment was 2:1 while the upper end of the overland erosion contribution indicated a ratio closer to 1:1. Regardless, streambank erosion was shown to account for a large, if not dominant, proportion of the in-stream sediment of these two watersheds. This finding has important implications to the management of erosion in watersheds. Currently, conservation practices in these watersheds emphasize BMPs that address overland erosion and little effort has been focused on controlling streambank erosion.

Management Implications

There are many practices available for mitigating streambank erosion. Traditional engineering approaches include installation of rock rip rap, concrete lined channels, bulkheads, boulder weirs/riffle structures, etc. (Bentrup and Hoag, 1998; Schultz et al., 2004). These structures require minimal maintenance during their lifetime but may be very expensive to repair if they fail. Bioengineering techniques rely on a combination of vegetative and structural components (e.g. erosion control fabrics) to stabilize banks and reduce erosion rates. Bioengineering systems may be costly to establish, but because of their self-sustaining nature can be more cost effective in the long-run (Bentrup and Hoag, 1998). Establishment of riparian forest buffers and grass buffer strips are also effective streambank management strategies. Riparian zones represent the interface between overland and in-channel processes and therefore offer a unique approach that can reduce streambank erosion rates while also mitigating overland sources of sediment (Gregory et al., 1991; Schulz et al. 2004). There is extensive evidence in the literature that also supports exclusion of cattle from riparian zones, either as part of a rotational grazing system, or by fencing them off streambanks (Platts, 1981; Kauffman and Krueger, 1984; Bentrup and Hoag, 1998; Sovell et al., 2000; Belsky et al., 1999; Clary and Kinney, 2002; Walker et al, 2009). Specific management plans are dependent on a variety of aspects such as channel characteristics, land owner goals, and available resources. Government programs such as the Environmental Quality Incentives Program (EQIP) through NRCS offer cost-share options for landowners.

Currently, establishment of streambank stabilization measures are voluntary, and therefore, despite programs and practices available to landowners for streambank erosion mitigation, the riparian zones and streambanks in the study area, and across the Midwest, are largely unmanaged (Lyons et al., 2000). There were no differences in streambank erosion rates between the land use treatments in the study area; however, if riparian zones are actively managed it has potential to produce significant land use effects. Conversely, in light of the highly disturbed landscape (nearly 70% of the study area is in row crops), it may require more extensive management that addresses the hydrology of the entire watershed system; and mitigation efforts that solely address adjacent land use, while ignoring up-stream processes, may not produce significant reductions in streambank erosion in claypan watersheds.

CONCLUSIONS

Of the three main factors assessed in this study, only season showed a significant impact on streambank erosion rates in Crooked and Otter Creek watersheds, while land use and stream order were not significant factors. Differences were not seen between land use treatments because of the variation in up-stream factors that collectively have more impact on streambank erosion than adjacent land use only. Stream order was not a significant factor influencing streambank erosion rates because stream order designation is largely independent of the factors controlling erosion processes. Season was found to be highly significant in controlling streambank erosion, with most erosion occuring in the winter months. However, this seasonal effect must be considered within the context of the significant three-way interaction between land use, stream order, and season. The three-way interaction, as well as the lack of adjacent land use effect, highlights the complex and highly variable nature of streambank erosion processes and brings to light the difficulty in managing streambank erosion. Nutrient concentrations of bank soils were significantly lower in forest sites, which is likely the result of lack of proximity to adjacent crop land as well as accelerated nutrient cycling. The nutrient loss rates had significant effects and interaction terms that were reflective of the sediment loss rates.

Comparing the contribution of bank and overland sediment underscores the need for targeting management efforts to streambank erosion. While overland erosion is by far the major source of gross watershed erosion, this work has demonstrated the importance of streambanks as a major source of in-stream sediment in claypan watersheds. Management plans and funding exist for implementing streambank stability measures if landowners are willing to adopt these practices. Currently, riparian zones in these watersheds are largely unmanaged. In light of the lack of adjacent land use effect, further research is needed to determine if managing riparian zones alone is enough to reduce streambank erosion rates in claypan watersheds. Managers may be required to address larger scale watershed hydrology in order to address all of the processes that control streambank erosion. Controlling in-stream sedimentation in these watersheds may not only require continued efforts to reduce overland erosion, but an additional effort to minimize streambank erosion by targeting BMPs to the riparian areas of the landscape.

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Table 2.1. Summary of streambank erosion site characteristics.

Stream Order, Land Use, Replication #	Total Bank Length (m)	Eroding Length (m)	Bulk Density (Mg m^{-3})	Carbon Concentration (g kg^{-1})	Nitrogen Concentration (g kg^{-1})
1st Crop 1	800	330	1.36	13.2	1.25
1st Crop 2	800	188	1.37	15.3	1.30
1st Crop 3	800	365	1.38	14.8	1.44
1st Forest 1	680	318	1.40	12.2	1.17
1st Forest 2	600	223	1.52	8.18	0.63
1st Forest 3	800	488	1.49	8.92	0.63
1st Pasture 1	800	567	1.40	11.9	0.94
1st Pasture 2	800	186	1.36	15.1	1.30
1st Pasture 3	800	580	1.49	12.0	0.93
1st Riparian 1	800	573	1.43	13.8	1.18
1st Riparian 2	800	689	1.30	15.0	1.30
1st Riparian 3	800	665	1.48	10.1	0.76
2nd Crop 1	800	800	1.26	19.2	1.59
2nd Crop 2	800	459	1.40	12.6	1.11
2nd Crop 3	800	549	1.30	17.8	1.48
2nd Forest 1	800	467	1.63	5.93	0.31
2nd Forest 2	800	321	1.47	10.5	0.99
2nd Forest 3	880	252	1.48	9.80	0.78
2nd Pasture 1	800	780	1.45	12.1	0.98
2nd Pasture 2	950	145	1.37	13.6	1.34
2nd Pasture 3	800	294	1.34	18.9	1.76
2nd Riparian 1	800	186	1.49	10.1	0.80
2nd Riparian 2	800	178	1.48	14.3	1.34
2nd Riparian 3	800	364	1.36	13.1	1.11
3rd Crop	800	423	1.48	11.2	0.93
3rd Forest 1	800	134	1.48	8.73	0.80
3rd Forest 2	800	240	1.38	14.9	1.26
3rd Forest 3	700	340	1.41	11.8	0.99
3rd Pasture 1	800	703	1.42	11.6	0.99
3rd Pasture 2	800	690	1.42	12.8	1.06
3rd Pasture 3	800	463	1.49	11.3	0.95
3rd Riparian 1	800	268	1.52	9.02	0.76
3rd Riparian 2	800	562	1.35	13.6	1.22
3rd Riparian 3	800	562	1.36	16.9	1.45

Table 2.2. Summary of sediment loss rate LS means† and ANOVA statistics‡.

	Season 1	Season 2	Season 3
Stream Order 1	kg m^{-1}		
Crop	22	-8.0	1.5
Forest	44	16	1.9
Pasture	27	33	-3.9
Riparian	3.1	9.7	7.0
Stream Order 2			
Crop	39	-32	-8.7
Forest	140	110	17
Pasture	29	-23	6.0
Riparian	18.1	-4.6	-1.5
Stream Order 3			
Crop	30	-12	-22
Forest	18	6.6	8.9
Pasture	88	-41	-9.2
Riparian	120	0.5	27

		Model 1		Model 2	
ANOVA	df	F	p-value	F	p-value
Land Use (LU)	3	4.2	0.017	2.0	0.151
Stream Order (SO)	2	0.8	0.472	0.9	0.436
Season (Seas)	2	16	<0.001	104	<0.001
LU x SO	6	4.6	0.004	2.3	0.072
LU x Seas	6	1.9	0.136	2.5	0.055
SO x Seas	4	2.4	0.078	3.9	0.016
LU x SO x Seas	12	2.6	0.024	2.8	0.017

†LS Means are from Model 1

‡Only effects with significance in Model 1 and Model 2 were considered significant.

Table 2.3. Summary of nutrient concentration LS means and ANOVA statistics.

Land Use	C	N		
	--------------g kg^{-1} soil-------------			
Crop	15	1.3		
Forest	10	0.84		
Pasture	13	1.1		
Riparian	13	1.1		
	C		N	
ANOVA	F	p-value	F	p-value
Land Use (LU)	4.3	0.013	3.7	0.024
Stream Order (SO)	0.32	0.732	0.18	0.837

Table 2.4. Nutrient concentration pair-wise comparisons of land use LS means.

	C		**N**	
Comparison	Difference g kg^{-1}	p-value	Difference g kg^{-1}	p-value
Crop to forest	4.6	0.002	0.45	0.004
Crop to pasture	1.5	0.285	0.15	0.287
Crop to riparian	1.8	0.202	0.19	0.190
Forest to pasture	-3.1	0.018	-0.30	0.031
Forest to riparian	-2.8	0.031	-0.26	0.056
Pasture to riparian	0.30	0.816	0.04	0.783

Table 2.5. Summary of nutrient loss rate LS means† and ANOVA statistics‡.

	Season 1		Season 2		Season 3	
	Carbon	Nitrogen	Carbon	Nitrogen	Carbon	Nitrogen
Stream Order 1	---------------------------------g m^{-1}---------------------------------					
Crop	450	41	-110	-10	31	2.1
Forest	390	29	190	15	40	3.5
Pasture	580	51	630	54	-1.5	0.66
Riparian	400	40	440	41	170	15
Stream Order 2						
Crop	790	73	-310	-21	-110	-8.3
Forest	1200	92	820	55	140	10
Pasture	600	56	-120	-6.7	120	10
Riparian	320	28	-9.2	-0.10	-8.6	-0.58
Stream Order 3						
Crop	810	81	290	33	-170	-13
Forest	250	23	110	11	110	10
Pasture	1300	110	-290	-21	-75	-6.1
Riparian	1600	140	35	3.3	430	38

Carbon		Model 1		Model 2	
ANOVA	df	F	p-value	F	p-value
Land Use (LU)	3	0.58	0.632	1.1	0.379
Stream Order (SO)	2	0.30	0.742	5.0	0.016
Season (Seas)	6	24	<.001	170	<.001
LU x SO	2	2.3	0.070	7.3	<.001
LU x Seas	6	1.2	0.330	3.0	0.028
SO x Seas	4	2.9	0.044	13	<.001
LU x SO x Seas	12	2.7	0.021	9.9	<.001
Nitrogen					
ANOVA					
Land Use (LU)	3	0.38	0.769	1.0	0.400
Stream Order (SO)	2	0.54	0.590	1.6	0.225
Season (Seas)	6	27	<.001	71	<.001
LU x SO	2	2.0	0.106	3.1	0.023
LU x Seas	6	1.2	0.351	1.8	0.148
SO x Seas	4	3.2	0.034	5.8	0.003
LU x SO x Seas	12	2.8	0.018	3.9	0.003

†LS Means from Model 1.

‡ Only effects with significance in both ANOVA Models were considered significant overall.

Table 2.6. Watershed scale streambank sediment and nutrient calculations.

Stream Order	Bank Length (km)	Erosion Rate (kg $m^{-1}yr^{-1}$)	C Loss Rate (g $m^{-1}yr^{-1}$)	N Loss Rate (g $m^{-1}yr^{-1}$)
1st	704	44	530	43
2nd	323	43	470	37
3rd+	338	54	610	52
Watershed (Mg yr^{-1})		**63,000**	**730**	**60**

Table 2.7. Study area land use and overland erosion rates.

Land-Use	Area (ha)	Percent of Watershed	Erosion Rate (Mg ha^{-1})	Total Mass Eroded (Mg)
Impervious/ Urban	934	1.5	0	0
Total Cropland	39,434	64.8	5.7 to 38	
0.75 T	14,985	24.6	5.7	85,413
1 T	12,958	21.3	7.6	98,480
2 T	5,868	9.6	15.2	89,190
3 T	2,200	3.6	22.8	50,169
4 T	1,222	2.0	30.4	37,162
5 T	2,200	3.6	38.0	83,615
Grassland	12,954	21.3	3.8	49,227
Forest and Wetland	7530	12.4	1.9	13,391
Open Water	482	0.8	0.0	0
Sum	**60,851**	**100**		**507,000**

Table 2.8. Equations for Estimating the Sediment Delivery Ratio in Crooked and Otter Creek Watersheds

Equation	Author	Region	Input	Calculated SDR (%)
log (SDR)= 2.94259+0.82362 log R - 0.854 log L	(Maner 1958)	Red Hills	R= 220 ft L=21,6275 ft	2.7
graph pg 6-11	(USDA- SCS 1983)	6 studies across US	A= 100 mi^2	11.0
log (SDR)= 1.91349-0.33852 log (10* A)	(Roehl 1962)	Southeast Piedmont Region	A= 100 mi^2	13.0
log (SDR)= 2.8875-0.83291- colog R/L	(Roehl 1962)	Southeast Piedmont Region	R/L= 0.001	11.0
Average				9.4
Standard Deviation				4.6

R= relief, average difference between highest and lowest point in watersheds
L= length, average length of the main channels in each watershed from headwaters to outlet
A= average area of the watersheds

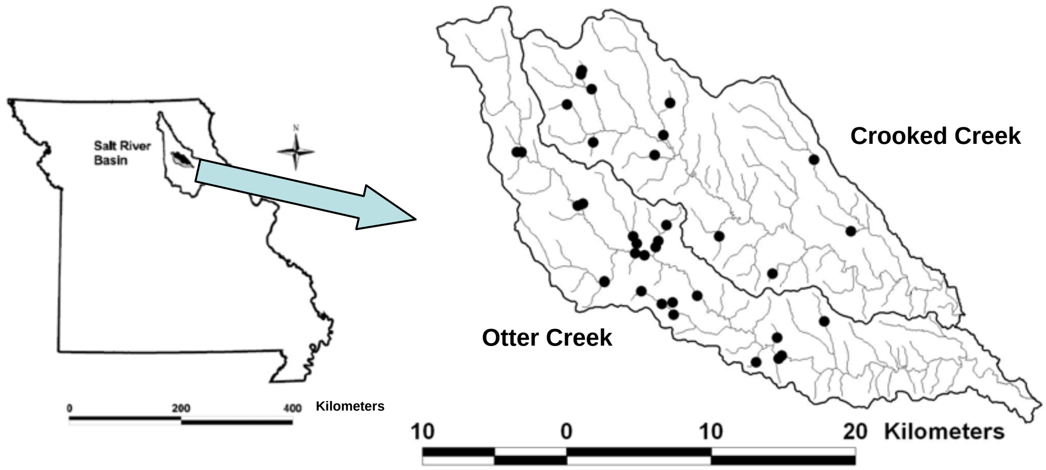

Figure 2.1. Crooked and Otter Creek streambank erosion site locations.

Figure 2.2. Pin arrangement on 3rd order stream. Banks are approximately 2.5m high.

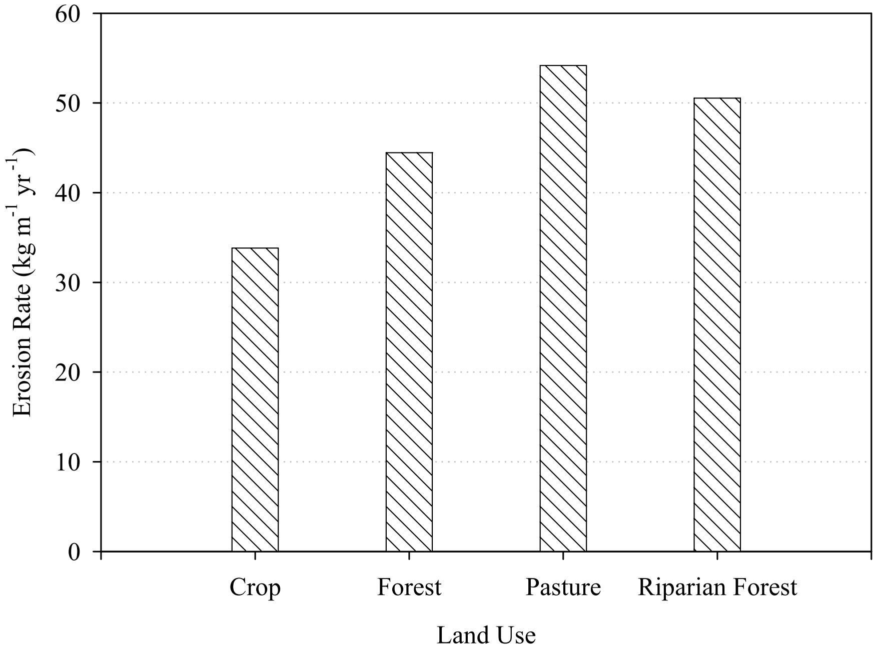

Figure 2.3. Sediment erosion rates for each land use treatment (simple means averaged over season and stream order).

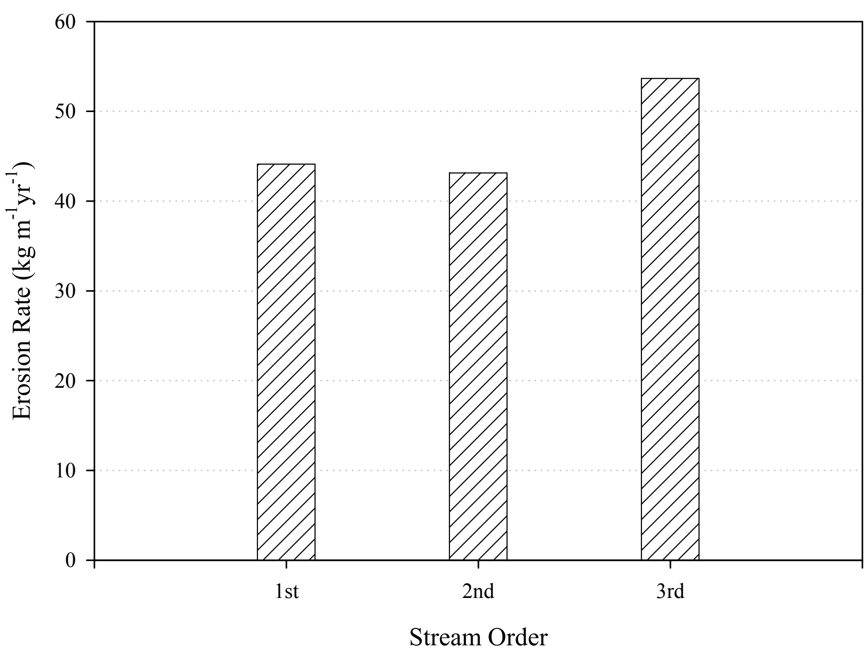

Figure 2.4. Sediment erosion rates for each stream order (simple means averaged over land use and seasons).

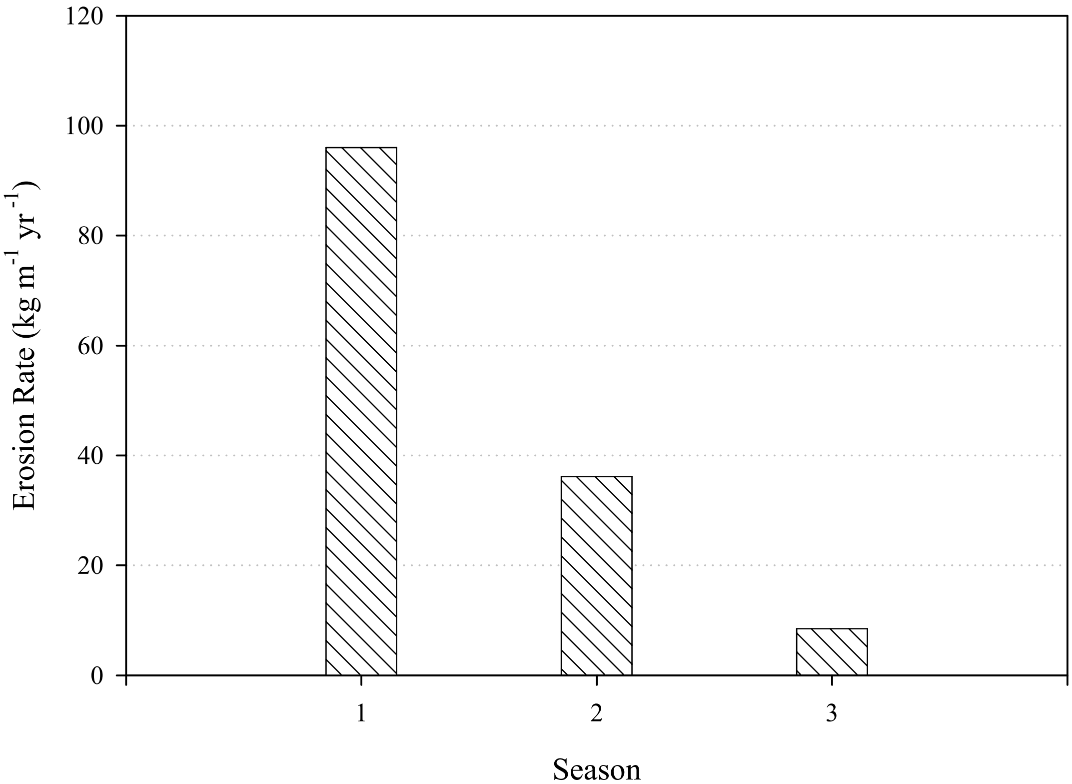

Figure 2.5. Sediment erosion rates for each season (arithmetic means averaged over land use and stream order).

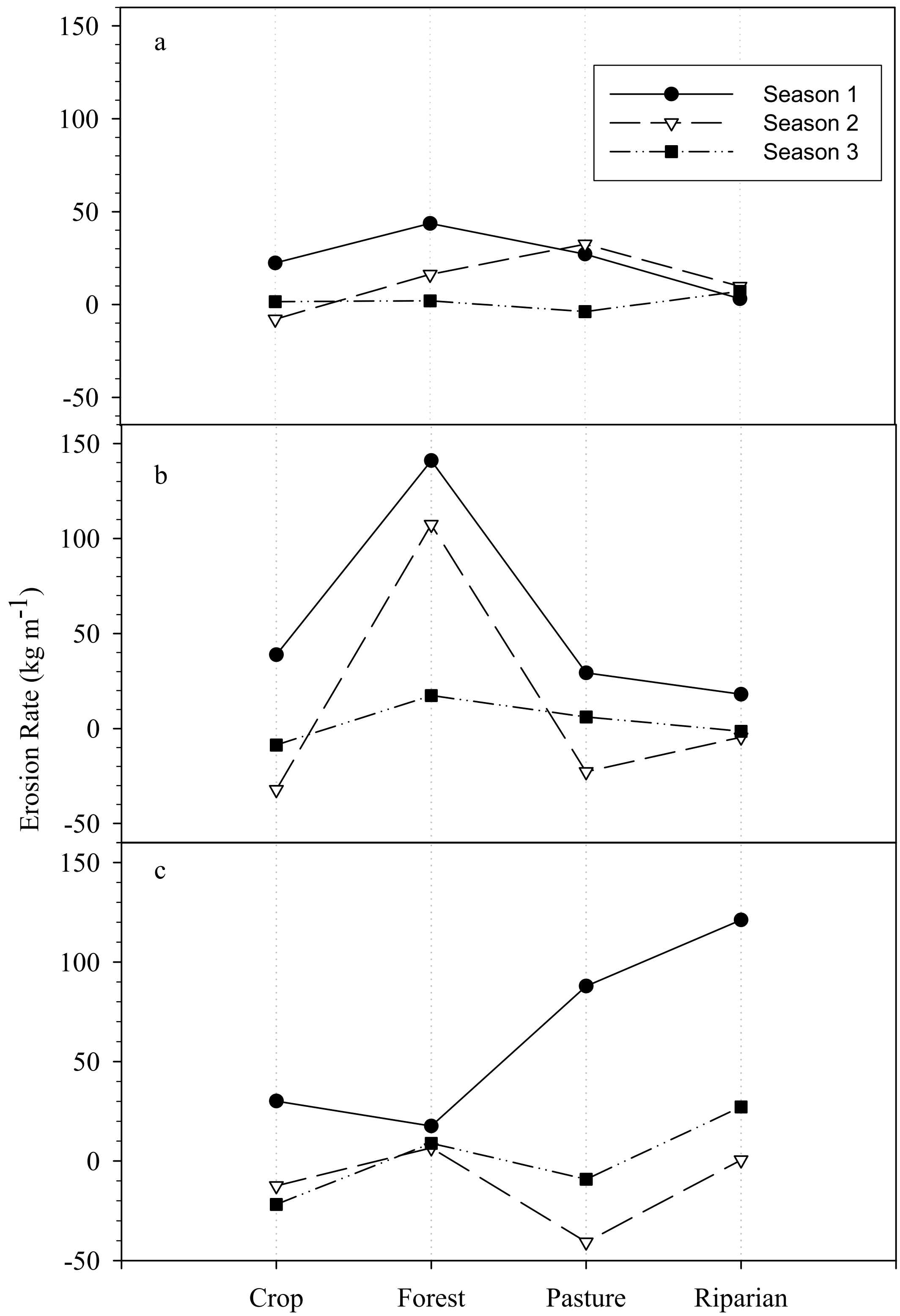

Figure 2.6. Pair-wise comparisons of LS means within land use treatments by a. Stream Order 1, b. Stream Order 2, and c. Stream Order 3.

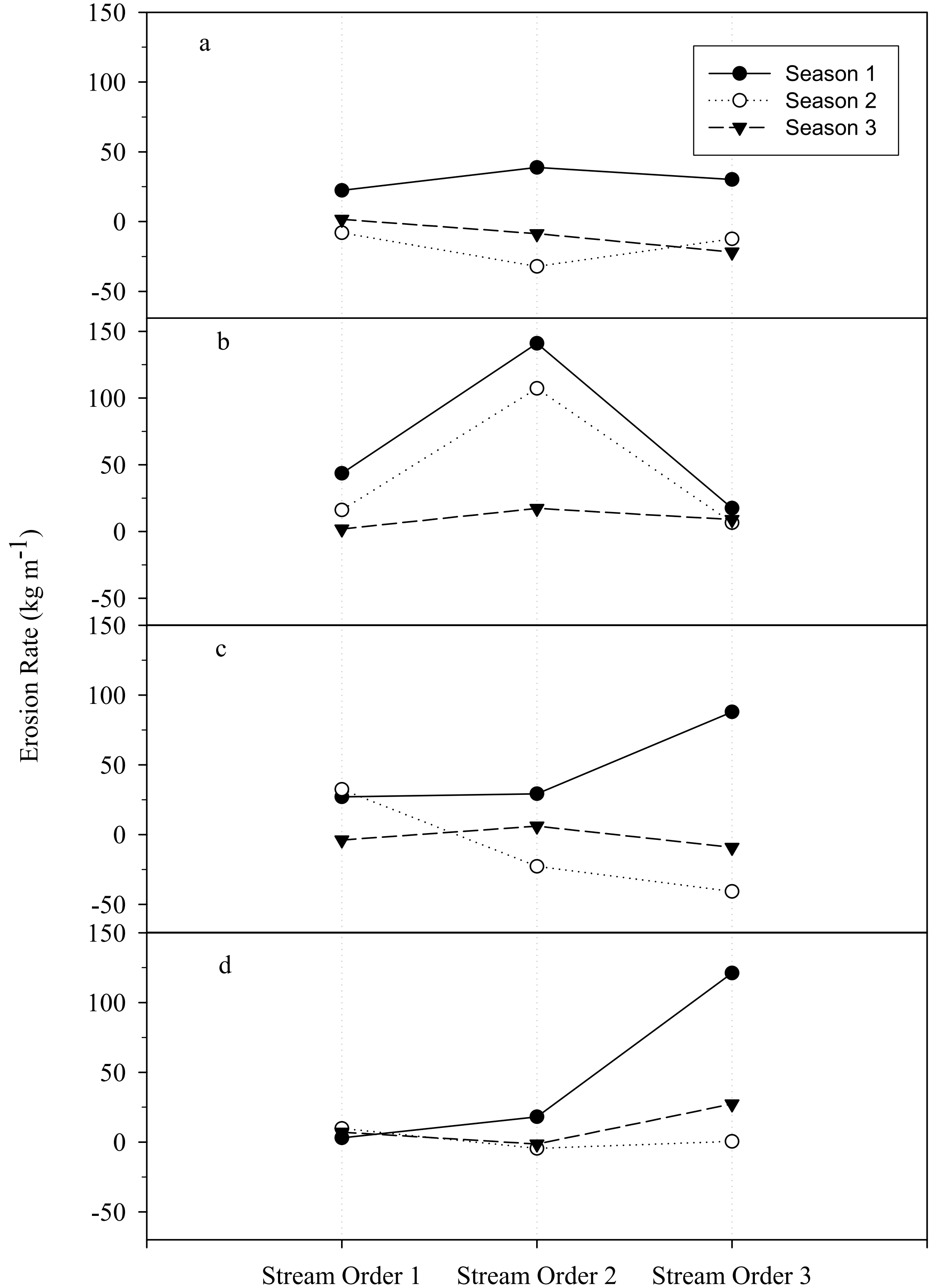

Figure 2.7. Pair-wise comparison of LS means within stream order by a. Crop, b. Forest, c. Pasture, and d. Riparian Forest land use treatments.

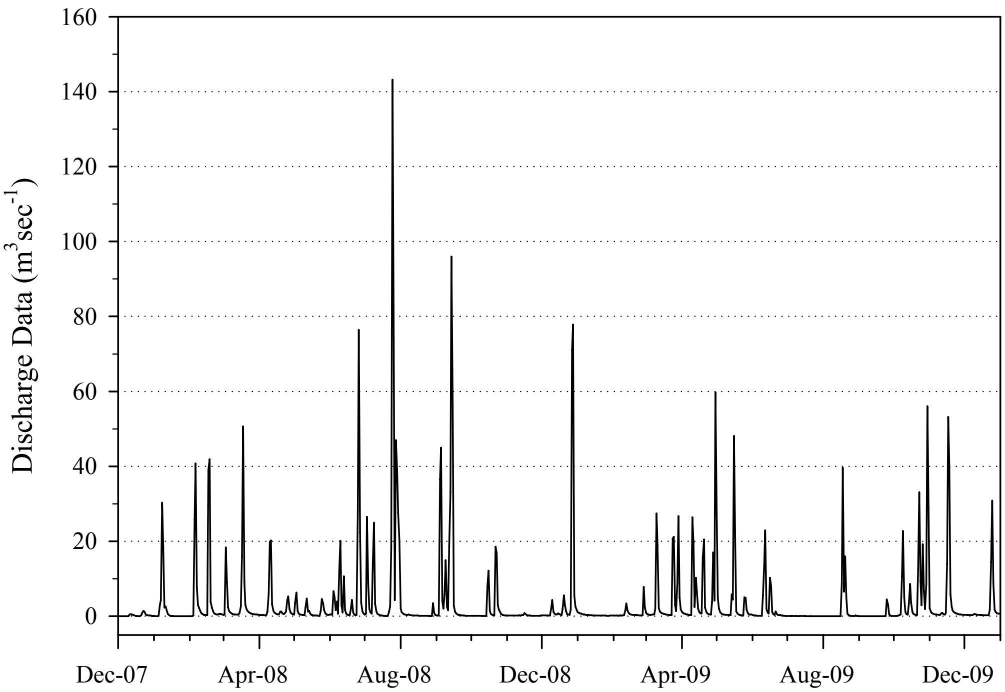

Figure 2.8. Average daily discharge of Crooked Creek from Dec 2007 through Dec 2009.

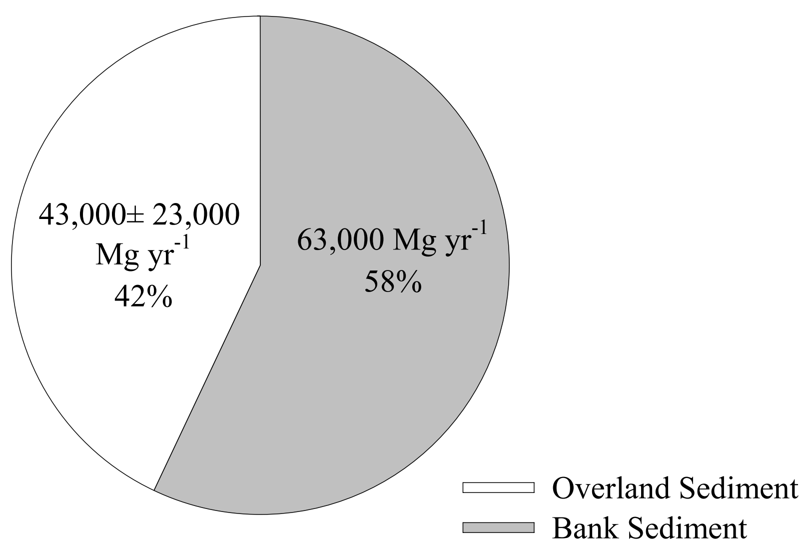

Figure 2.9. Comparison of overland and bank sediement sources of in-stream sediment.

CHAPTER 3: RISK ASSESSMENT MODEL

ABSTRACT

Identification of areas at risk for pesticide loss is a necessary requirement for targeting best management practices (BMPs) to the most vulnerable areas within fields or watersheds. To work toward that goal, a process-based index model was developed to assess relative landscape vulnerability to hydrologic losses of some commonly used corn herbicides. The model applies mathematical functions to assign weights (i.e., degree of risk) on the basis of herbicide, soil, and hydrologic properties relevant to the environmental fate of herbicides. The model uses the NRCS county Soil Survey Geographic database (SSURGO) as the input source of soil and hydrologic data. The risk of herbicide transport is considered for three hydrologic pathways, or scenarios: leaching, solution runoff, and particle adsorbed runoff. The model was used to compute the relative risk of herbicide transport for a given herbicide for each hydrologic pathway, providing the spatial and temporal risk of transport. The model was applied to the Young's Creek watershed in the Central Claypan Region of Missouri, a watershed with known herbicide contamination problems. The risk of herbicide loss for each hydrologic pathway was determined for each of four corn herbicides (atrazine, metolachlor, isoxaflutole, and glyphosate). By computing area-weighted watershed risks the risk for each herbicide and hydrologic pathway scenario combination was determined. The watershed risks were

used to evaluate the potential spatial and temporal risk of applying these herbicides in the Young's Creek watershed. The information provided by the model can be used to make recommendations regarding the choice of herbicides that will minimize the risk of hydrologic transport for this watershed. In the case of Young's Creek, the use of metolachlor or isoxaflutole minimized the overall risk of herbicide transport to surface or ground water.

INTRODUCTION

Pesticide contamination of surface and ground water is extensive across the United States (Pereira et al., 1990; Squillace and Thurman, 1992; Koplin et al., 1998; Lerch et al., 1998, Barbash et al., 2001; Battaglin et al., 2003). In order to identify areas vulnerable to pesticide transport and to support management decisions for reducing off-site movement many computer models have been developed to predict the environmental fate of pesticides. In the past, models that attempted to link landscape effects to pesticide fate often used STATSGO soils data, which limited model application to larger scales and percluded model usefulness for making local management decisions. The availability of county-level Soil Survey Geographic data sets (SSURGO; Soil Survey Staff, 2005), which has a finer resolution and is widely available for most counties across the United States, opened new possibilities for modeling pesticide fate on smaller scales. The USDA-NRCS Windows Pesticide Screen Tool (USDA-NRCS, National Water and Climate Center, 2004) is an index model that assesses risk of pesticide leaching and runoff based on soil, landscape, pesticide, and management components from SSURGO data. However, it is limited by its use of hydrologic soil groups to predict runoff. While hydrologic soil groups have been found to be useful for assessing differences in vulnerability to herbicide transport between watersheds (Lerch and Blanchard, 2003), they are not useful for assessing vulnerability within watersheds because of their independence from landscape position. Process models often are complex, require large amounts of data, and produce output that is difficult to assess objectively. There is a need for decision making tools for landowners and managers that utilize commonly available soil and herbicide data, show sensitivity to landscape position, and produce output that is easily and objectively interpreted. The limitations of the currently available models suggests the need for alternative approaches to modeling the risk of pesticide transport.

Identification of areas at high risk for pesticide loss is a necessary requirement for targeting best management practices (BMPs) to the most vulnerable areas within fields or watersheds. Therefore, an index-based risk assessment model was developed that predicts the relative risks of hydrologic transport of pesticides. The model applies mathematical functions to assign weights (i.e., degree of risk) on the basis of pesticide, soil, and landscape properties relevant to the environmental fate of pesticides. The risk of pesticide transport is considered for three hydrologic pathways: leaching, solution runoff, and particle adsorbed runoff. The objective of the study was to evaluate the risk of herbicide loss through leaching, solution runoff (SRO), and particle adsorbed runoff (ARO) using components of landscape, soil, and herbicide properties for the purpose of targeting conservation practices at the watershed scale. To achieve this objective, the model was applied to Young's Creek watershed, a claypan watershed located in north-central Missouri with known herbicide contamination problems (Lerch and Blanchard, 2003).

MATERIALS AND METHODS

Site Description

Young's Creek watershed is located in north-central Missouri and has an area of 227 km^2, with boundaries crossing through Audrain, Boone, and Monroe Counties (Fig.3.1). Typical slopes range from 1-4% and the soils are characterized by restrictive clay layers, called claypans. Claypan soils possess a slowly permeable, subsoil horizon, with a large and abrupt increase in clay content that is distinguished from overlying material by a sharply defined boundary (Soil Science Society of America, 2001; Lerch et al., 2008). The presence of the argillic horizon impedes downward movement of water, with water often perching over the claypan, which causes an increased vulnerability to runoff (Blanco-Canqui et al., 2002). Mexico-Putnam and Mexico-Leonard-Armstrong are common catena associations found in the watershed, and are characterized by nearly level to gently sloping, loamy over clayey, poorly drained soils (USDA-NRCS, Audrain County MO soil survey, 1995). Another common catena is the Adco-Mexico-Putnam association. The geomorphology model of this association is illustrated in Figure 3.2. The Young's Creek watershed was chosen because it is representative of the intensely row-cropped claypan watersheds of Major Land Resource Area 113 (USDA-NRCS, 2006) which are known to have significant herbicide contamination problems (Lerch and Blanchard, 2003; Lerch et al., 2008).

Herbicide Selection

Four soil-applied corn herbicides, representing four herbicide classes, were selected for study because they are commonly used, or in one case represent a possible alternative, and possess a wide range of sorption intensities and persistence in soil. Atrazine [6-chloro-N^2-ethyl-N^4-(1-methylethyl)-1,3,5-triazine-2,4-diamine] is applied as a pre- and post-emergent herbicide in corn (*Zea mays*) and sorghum (*Sorghum bicolor*) for control of annual broadleaf weeds. Diketonitrile [2-cyano-3-cyclopropyl-1-(2-methylsulfonyl-4-trifluoromethyl-phenyl)-1,3-dione, (DKN)] is the active metabolite of isoxaflutole [5-cyclopropyl-4-(2-methanesulfonyl-4-trifluoromethyl-benzoyl)-isoxazole], and is used on corn as a pre-emergent grass and broad-leaf herbicide (Beltran et al., 2000; 2002). Isoxaflutole has been proposed as an alternative to atrazine for corn production in the midwestern U.S. Metolachlor [2-chloro-*N*-(2-ethyl-6-methylphenyl)-*N*-(2-methoxy-1-methylethyl)acetamide] is a broad-spectrum corn and soybean (*Glycine max*) herbicide that is used extensively throughout the Midwest (Seybold and Mersie, 1996). Glyphosate (N-phosphonomethyl-glycine) is a broad-spectrum, non-selective herbicide primarily used on corn and soybeans that have been engineered with a naturally occurring protein that imparts resistance to glyphosate (Borggaard and Gimsing, 2008). Glyphosate was the most used active ingredient in United States agriculture in 2001, outranking even atrazine, which had been the most heavily used for decades (Kiely et al., 2004). Atrazine and metalochlor usage was also in the top ten of most commonly used active ingredients in agriculture in 2000 and 2001 (Kiely et al., 2004).

Model Inputs and Functions

SSURGO Input Parameters

Model input data were obtained from the SSURGO database and included soil and hydrologic properties. Soil data included profile descriptions, organic matter (OM) content, clay content, pH, the soil erodibility factor (K_w), and saturated hydraulic conductivity (K_{sat}). Hydrologic and landscape data included the depth to the water table, flooding frequency, slope (S), and slope length (LS). The following section lists each parameter with a brief definition and, where applicable, a discussion of the measurement/calculation of the parameter. Parameter names as denoted in SSURGO are included in Appendix C. Most of the information in this section comes directly from the SSURGO metadata (Soil Survey Staff, 2005) and/or Soil Survey Lab Manual (Soil Survey Staff, 1996).

Organic matter is defined as the decomposed plant and animal residues present in the soil. The OM value used in the model is the percent organic mater (%OM) (by weight) present in the fine earth fraction (Soil Survey Staff, 2005). Organic matter is not directly measured, but calculated by multiplying the measured soil organic carbon by the Van Bemmelen factor of 1.724. This assumes OM contains 58% organic carbon (USDA-NRSC, 2007). Percent organic carbon is reported on an oven dry basis (Soil Survey Staff, 1996). Organic matter content is important in determining the sorption phase of a herbicide as well as the rate of biotic degradation. This parameter was used for calculating weights associated with the soil function adsorption weight (Eq. C.1-3), and the dissipation functions biotic degradation (Eq. C.12) and adsorption weight (Eqs. C.13,14).

The total clay content (% clay) parameter is the percent (by weight) of mineral particles less than 0.002 mm in equivalent diameter of the fine earth fraction (particles <2mm) of a soil (Soil Survey Staff, 2005). Clay content is important in determining sorption of herbicides, particularly in low organic content soils. The %clay was used to calculate a soil adsorption clay weight function (Eqs. C.2, 3) whichI was applied in the dissipation adsorption function (Eqs. C.13,14).

The pH parameter is the pH of the soil as measured using the 1:1 soil-water ratio method and is a relative expression of acidity or alkalinity of a soil sample (Soil Survey Staff, 1996). Soil pH has often been termed the "master variable" of soil because it influences virtually all chemical reactions within the soil matrix. In the model, it was used in calculating the biotic degradation (Eq. C.12) and hydrolysis weight (Eq. C.15) functions.

The K_w erodibility constant is one component of calculating the K factor, which is a measure of erodibility, in USLE and RUSLE 2 (Soil Survey Staff, 2005). Erodibility is used to describe the characteristic of some soils being inherently more prone to erosion than others when all other factors are the same. It is the measure of how susceptible a soil is to erosion by runoff and raindrop impact and the rate at which it erodes per unit area (Renard, 1997; USDA-NRSC, 2007). Silt and OM have the most influence on erodibilty (Wischmeirer and Smith, 1978). Soils high in silt have high erodibility. Silts are more erodible than sand because they are smaller and require less energy for detachment and are more erodible than clay because they lack cohesive resistance. Soils high in OM have low erodibility; OM promotes aggregate stability which resists detachment, increases filtration rates, and mitigates runoff (Wischmeirer and Smith, 1978; Renard, 1997). The K_w parameter was used to calculate an erodibility weight in the model (Eq. C.9).

Saturated hydraulic conductivity is the velocity of water moving vertically through a unit area of saturated soil over a unit time under a unit of hydraulic gradient. It is expressed in micrometers per second (Soil Survey Staff, 2005). Ideally, K_{sat} is measured in the field using a compact constant head permeameter, such as an Amoozemeter. However, these measurements are time consuming and require special training to operate the equipment, so K_{sat} is often estimated using data collected for similar type soils (personal communication, Clayton Lee, USDA-NRCS, State Soil Scientist, Missouri). Saturated hydraulic conductivity is enhanced by coarse texture and good structure, as these properties influence the size, shape, and distribution of pores within a soil (Dane and Topp, 2002). The K_{sat} input parameter was used in the index of surface runoff (ISRO) weight function (Eq.B.5).

The water table depth, or depth to free water, parameter is a report of the shallowest water table depth in centimeters that occurs during April through June (Soil Survey Staff, 2005), which is the time most herbicides are applied. This parameter relates to the potential for herbicide leaching to a shallow water table, and was used to calculate a leaching penalty (Eq. C.4).

The flooding frequency maximum of a soil is the annual probability of a flooding event, and is expressed as classed data (None, Rare, Occasional, or Frequent). The dominant flooding frequency class for the map unit is reported based on component percentage of the map unit (Soil Survey Staff, 2005). This parameter is used to directly calculate risk based on the categorical data (Eq.C.4) and is utilized in all three hydrologic scenarios in which risk of transport is directly proportional to the flooding frequency.

Slope gradient is the difference in elevation between two points (Soil Survey Staff, 2005). It is measured using a hand level or clinometer. Slope gradient is reported as a percent change over the total distance between two points (Soil Survey Staff, 1996). For example, if the change in elevation is 3 m between two points that are 150 m apart the S is 2%. The S was used in combination with LS to calculate the estimated LS/Inverse LS (Eqs. C.10, 11) and ISRO weight (Eq.C.5) functions.

Slope length is defined as the horizontal distance between the point where runoff originates and the point at which deposition occurs or runoff water enters a distinct channel (Wischmeirer and Smith, 1978; Soil Survey Staff, 2005). It was used in combination with S to determine the effect of topography on erosion (USDA-NRSC, 2007). This combination is proportional to the velocity of surface runoff, which determines the energy of the runoff for transporting detached soil particles. Slope length is used to calculate the estimated LS weight function.

Herbicide Input Parameters

Herbicide parameters included sorption coefficient (K_d) values, biodegradation half-life ($t_{1/2}$), and vapor pressure (*vp*). The K_d is the ratio of herbicide sorbed to soil particles to the concentration of herbicide remaining in solution at equilibrium (Weber et al., 2000). The K_d is used to calculate the soil organic carbon-water partitioning coefficient (K_{oc}), which varies spatially as a function of the OM content of a soil series. The use of K_{oc} implies that soil OM is the colloid controlling adsorption, which is clearly not the case for glyphosate. However, the use of K_d rather than K_{oc} is problematic in the model as this would require laboratory determination of K_d values for every soil series in a watershed. Such a model requirement is clearly impractical, and therefore, K_{oc} was used since it readily allows for varying the sorption intensity by soil series. The K_{oc} value was used in the dissipation adsorption functions (Eqs. C.13,14). The biodegradation half-life ($t_{1/2}$) for each pesticide is given in days and is used in the biotic degradation function (Eq. C.12). Vapor pressure, expressed as mm Hg at 25°C, was used in the volatilization function (Eq. C.16). There is no single source used for this information, although much of the information is available on pesticide product labels, in the literature, or in the Agricultural Research Service Pesticide Properties Database (USDA-ARS, 2001). The input values used for each of the four herbicides are listed in Table 3.1.

Soil and Hydrologic Functions

The soil and hydrologic functions included in the model compute relative weights (i.e., relative risk) for OM content, clay content, erodibility, depth to free water, flooding frequency, ISRO, and estimated LS. These functions were then combined to assess the overall risk for each soil map unit and hydrologic pathway (i.e., leaching, SRO, and ARO) represented in the study area.

The OM weight function (Eq. C.1) is a sigmoidal function (Figure C.1). At the low end (OM<2%) the impact of OM on pesticide sorption was considered to be minimal. From 0-2% OM, there is little sensitivity to changes in OM. From 2-5%, the line becomes nearly linear with a positive slope. At the high OM contents (OM>5%), the effect of increasing OM also has little effect on sorption as the available sorption sites become saturated (Weed and Weber, 1974 ; Villaverde et al., 2008). The degree of pesticide sorption was a very significant parameter to all three scenarios (leaching, solution runoff, and particle adsorbed runoff) as it influences the amount of pesticide found in the liquid (vulnerable to leaching or solution runoff) or solid phase (vulnerable to particle adsorbed runoff).

Clay weight is a function of %clay and %OM (Eqs. C.2, 3). The highest clay weight was assigned for soils with low OM (i.e., < 1% OM). In these soils, clay is the dominant soil colloid controlling pesticide sorption. Clay becomes less important (smaller assigned weight) as OM increases. This reflects the much greater effect that OM has relative to clay on pesticide sorption intensity (He et al., 2006). The clay weight function was only applied when OM is < 2.1%.

The erodibilty weight (Eq. C.9) was assigned using a simple linear relationship with the K_w factor. The higher the K_w factor the lower the weight and the higher the vulnerability to pesticide loss. This weight was the single most important factor in the ARO scenario.

The depth to free water penalty was only considered for the leaching scenario since this hydrologic pathway is the only one of the three in which groundwater is directly impacted. The depth to free water equation (Eq. C.4) computed the weight as a linear function of the depth to the water table if it is shallower than the depth to the minimum K_{sat}. A shallow water table depth (0 cm) resulted in a larger penalty than a deep water table depth (180 cm) because the proximity of the water table to the surface is directly related to the vulnerability of the groundwater to contamination.

The flooding frequency weight was the only weight that was assigned based on classed data rather than a continuous function. Flooding frequency influences runoff and leaching risk. An area that never floods (Class: No Flood in SSURGO) will not be at risk for runoff or leaching by flooding so it wa assigned a zero weight, but an area with frequent flooding (Class: Frequent) will have higher risk for flooding and was assigned a smaller (i.e. more negative) weight. The flooding frequency classes can be found in Appendix C. The weights assigned to the runoff scenario and the leaching scenario based on flooding frequency reflects the relative influence of this process in each situation. Flooding frequency has a larger impact in the runoff scenario because a field that frequently floods will more likely lose a substantial amount of the applied pesticide directly to the stream. Leaching risk also increases because saturated flow conditions will increase the rate of flow from the surface to the groundwater, and thus, increase the likelihood of pesticide transport to groundwater.

The ISRO weight (Eq. C.5) is a function of K_{sat} and S. As S increases there is less infiltration and more runoff. The equation is in the form of a 1^{st} order kinetic reaction, and shows high sensitivity at low slopes (Figure C.2). The equation was derived from the index to surface runoff classes from the Soil Survey Manual (Soil Survey Staff, 1993 table 3-10). The minimum K_{sat} within the first 100 cm of soil is used to calculate the ISRO weight. ISRO is used in all three transport scenarios. The leaching ISRO (LISRO) weight (Eq. C.6) is the difference between the maximum runoff ISRO weight, 5.5, and the calculated runoff ISRO (RISRO) weight (Eq. C.5). This reflects the fact that a high potential for runoff results in a low potential for leaching and vice versa. The most vulnerable condition for leaching is one in which a soil has a low slope gradient and high K_{sat}.

The estimated LS weight is determined based on the USLE equation, and is adapted from Wischmeier and Smith (1978, page 13, Figure 4). The USLE estimation of

LS does not show sensitivity for low slopes (<15%), which represents the largest proportion of slopes in agricultural watersheds where the model was designed to be applied. The equation for LS was re-worked to gain sensitivity for small slopes, and was made into a continuous function that could be related to the slope of each soil map unit from SSURGO. The estimated LS function was developed using a USLE fact-sheet developed by the Ministry of Agricultural Food and Rural Affairs (Ontario, Canada) and is available at: http://www.omafra.gov.on.ca/english/engineer/facts/00-001.htm#tab3a. First, the estimated LS (y) was plotted against the square root of slope length (x) to linearize the relationship:

$$Estimated\ LS = m * LS^{0.5} + b \qquad \text{(Eq. 3.1)}$$

where, *m*= slope of the function, *LS*= slope length parameter from SSURGO, and *b*= the y-intercept. Next, an equation was developed to relate this slope of the graph to the SSURGO map unit slope:

$$m = -0.1300 + 0.1218 * \exp^{0.1149*Slope}$$

(Eq. 3.2)

where *Slope* is the soil map unitS from SSURGO and *m* is the slope of the function in Eq. 3.1. An additional equation was also established to relate the y-intercept (*b*) from Eq. 3,1 to the SSURGO map unit slope:

(Eq. 3.3)

Inserting the slope (Eq. 3.2) and y-intercept (Eq. 3.3) equations into the original LS equations gives:

$$EstimatedLS =$$
$$\left(-0.1300 + 0.1218 * \exp^{0.1149*Slope}\right) * LS^{0.5} + 0.1127 + \frac{-0.0072 * \left(\exp^{0.1127*Slope} - 1\right)}{0.1127}$$

(Eq. 3.4)

Thus, the estimated LS weight (Eq. C.10) is a function of LS and slope gradient. The inverse LS was used for calculating ARO risk (Eq. C.11). The graph of the estimated LS and inverse LS functions can be found in Appendix C (Fig. C.3).

Dissipation Functions

The dissipation functions in the model all represent time-dependent functions that reduce the risk for herbicide transport, and included biotic degradation, adsorption, hydrolysis, and volatilization. For a list of function equations see Appendix C.

Biotic degradation (Eq. C.12) is a function of both time and soil pH. This reflects of the influence of time and pH on soil microbial activity required for biotic degradation. The time function reflects a first order kinetic relationship between time (days) and the biotic degradation of pesticides. Day 189 is the point at which the time component reaches its maximum weight. In other words, by day 189 the maximum amount of biotic degradation has occurred. This corresponds roughly to a 6 month growing season. Biotic degradation as a function of soil pH is a bell-shaped curve with maximum weight (i.e., lowest risk) at a soil pH of 6.8 (Figure C.4), which is assumed to be the optimal soil pH for pesticide degradation. Any deviation from pH 6.8 (whether a shift to more acidic or more basic conditions) resulted in decreased degradation rates and a lower weight (i.e. higher risk) assignment for biotic degradation.

The dissipation adsorption weight functions combine the sorptive phases (OM and clay) with a time component that assumes the sorption process follows 1st order kinetics (e^{-kt}), where k= 1.609 (Eq. C.14). With such a large k, the reaction proceeds quickly, reaching equilibrium in approximately 3 days. The sum of the function for SRO and ARO is equal to 100% of the applied herbicide mass. This follows the rationale that whatever is not adsorbed onto particles will remain in solution. This is reflected in the functions as solution runoff is simply the difference between 1 and the rate of particle adsorption. Therefore, the risk of pesticide transport in solution (leaching or SRO) was the inverse of its risk for particle adsorbed transport since compounds that adsorb strongly to soil particles will be lost by ARO (Pionke and Chesters, 1973). Higher adsorption coefficients predict that a chemical will be immobile in soil, making it vulnerable to losses through particle adsorbed runoff rather than solution runoff and leaching (Letey and Farmer, 1974).The rapid kinetics of pesticide adsorption to soil partly explains the exponential decrease in edge-of-field herbicide concentrations in runoff (Ghidey et al, 2005).

The hydrolysis weight is a function of time and pH. At neutral pH, there is no hydrolysis. The more that a soil deviates from neutral pH, the greater the pesticide hydrolysis will be (Armstrong et al, 1967). Hydrolysis reactions are abiotic chemical reactions of the pesticide with water. The time function assumes that the rate of hydrolysis follows 1st order kinetics (e^{-kt}), where k=1.10 (Eq. C.15). This relatively high k reflects the fast rate at which hydrolysis reactions occur in soils, and forces completion within 7 days. Although hydrolysis reactions occur quickly (k=1.1), their kinetics are assumed to be slower than that of adsorption (k=1.609). One important note is that this function was only applied to pesticides that are susceptible to hydrolysis based on literature reports. If no literature reports could be found to support the application of the hydrolysis function, then the hydrolysis weight was assigned as zero for that herbicide. For example, metalochlor is a herbicide that does not hydrolyze. Therefore, some knowledge of pesticide properties is therefore required before applying this equation.

The volatilization weight is a combination of the herbicide's vapor pressure (*vp*) and a 1st order kinetic time function (Eq. C.16). Because volatilization losses occur rapidly, the k value is large (1.1), forcing completion of this reaction within 7 days after application. This equation should only be applied to pesticides that are vulnerable to volatilization and therefore, knowledge of the chemical properties of the pesticide under consideration is required.

Assessing Risk

Calculating Risk

The series of equations for soil, hydrologic, and time-dependent dissipation functions were developed for each hydrologic pathway: leaching, SRO, and ARO. SSURGO and herbicide specific input data were used to calculate weights for each function described above. Weights from each function were summed for each soil-herbicide-time combination by soil series (e.g., Adco silt loam, 0-2% slopes). The total weight for each mapping unit was then transformed into a standardized risk (S_R) with a scale of 0 (lowest risk) to 10 (highest risk) using the following equation:

$$S_R = (SS_{wt} - Min_{wt})/(Max_{wt} - Min_{wt})*10 \qquad \text{Eq. 3.5}$$

where SS_{wt} equals computed soil mapping unit weight, Min_{wt} equals the minimum weight for a given hydrologic pathway, and Max_{wt} equals the maximum weight for a given hydrologic pathway. The S_R values allowed for comparisons across all time, herbicide, and soil combinations. To develop maps for each soil-herbicide-time combination, the S_R values were then combined with the spatial distribution of each mapping unit within Young's Creek watershed using ArcView 3.3. An area-weighted risk was computed for each herbicide and time combination by multiplying the S_R for a given soil mapping unit by the fractional area of each mapping unit within the watershed, then summing the risks. The area-weighted risks were then plotted against time and a time-integrated watershed risk (W_R) was then determined by integrating the area under each curve for a given herbicide and hydrologic pathway (Table 3.2).

Day 0

Conceptually, Day 0 can represent either the inherent watershed vulnerability or the moment of herbicide application. Which of these two concepts applies depends upon the type of comparison(s) being made by the user. When comparing across watersheds or between hydrologic pathways before application, Day 0 is representative of the risk based solely on the soil resource, independent of herbicide properties. Therefore, if a land manager needed to choose between two watersheds for targeting conservation funding, comparisons of Day 0 maps for each watershed would be most appropriate for that type of decision making. For the purpose of comparing herbicides within a watershed along a time sequence, Day 0 represents the moment of herbicide application. Day 0 is still independent of herbicide properties because no time has elapsed and no functions specific to herbicide properties have been applied. However, in this context, if Day 0 were considered the inherent watershed vulnerability for a time series comparison, it would lead to the erroneous conclusion that risk decreases as a result of herbicide application since the Day 0 maps generally have the highest risk within a time sequence. However, for rational interpretation of risk over time, Day 0 must be considered the moment of herbicide application when making temporal comparisons within a watershed.

RESULTS AND DISCUSSION

Model Development

The default model originally included the hydrologic functions for K_w, depth to free water, flooding frequency, ISRO, and estimated LS, and the dissipation functions biotic degradation, adsorption, hydrolysis, and volatilization. Application of the default model functions produced maps (Fig. 3.3) that showed risks for herbicide transport that were not consistent with observations from previous research in this area (Lerch et al., 1995; Blanchard and Donald, 1997; Donald et al., 1998). It was evident that the default model over-estimated leaching risk and underestimated risks for SRO and ARO for the claypan soils in Audrain County, Missouri.

In order to correct the default model, it was necessary to develop a series of penalties that accounted for the presence of a restrictive soil layer so that the model output risks would more closely match field research observations. The modifications to the model included two components: 1) identification of the presence of a restrictive clay layer; and 2) development of a penalty function based on the available water holding capacity (AWC) above the restrictive clay layer. In claypan watersheds, argillic horizons represent the most hydrologically restrictive layer and control the movement of water. The restrictive layer criterion developed was based on the presence of argillic horizons (Bt) in the SSURGO profile descriptions. The amount of clay and the thickness of the clay layer are both important factors related to the intensity of a restrictive clay layer. Therefore, a Bt intensity (Bti) factor was developed:

$$\text{Bti} = \Sigma\,(\text{thickness of Bt horizon(s) cm} \times \%\text{clay}) = \text{cm of clay} \qquad \text{(Eq. 3.6)}$$

Since restrictive soil layers should have the lowest K_{sat} values within the profile, a sensitivity analysis using data from Audrain County, MO and Jefferson County, NE was conducted to determine the Bti value that most often identified the soil horizon with the lowest K_{sat}. This analysis showed that a Bti ≥ 35 identified the horizon with the lowest K_{sat} 89% of the time, and therefore, this became the criterion by which restrictive clay layers were identified.

The Bti was then used as a criterion for applying and/or modifying several of the hydrologic functions. For the depth to free water penalty, leaching risk was decreased for soils with a restrictive layer (Bti>=35) above the depth to the water table. In this case, the claypan behaves as an aquitard and thus, it reduces the likelihood of groundwater contamination. For soils meeting the Bti criterion, the ISRO weight was decreased based on the available water holding capacity (AWC) above the restrictive layer. The AWC reflects the risk of capacity limited runoff. Thus, the lower the AWC above the restrictive layer, the greater the risk of herbicide transport by SRO or ARO pathways. In addition, soils with restrictive clay layers are more prone to runoff and erosion than soils with similar slope and slope length without a restrictive clay layer. In other words, restrictive layer soils behave as if they have greater slope and/or slope length than comparable soils without a restrictive layer. Therefore, a severe runoff penalty was developed that increased the calculated LS weight for soils with restrictive layers to reflect their increased risk of SRO and ARO. This penalty was applied via the estimated LS and K_w functions, which alters the risk for both SRO and ARO pathways. These modifications to the model resulted in an overall decrease in leaching risk and increase in SRO and ARO risks (Figures 3.4a, 3.8a, and 3.12a) that were more consistent with previous research in this area.

Hydrologic Pathways

After applying the Bti criterion and associated penalties, all herbicides had a greater risk for ARO and SRO than for leaching in Young's Creek watershed. Relative risk between hydrologic pathways was consistent with field data for the area with ARO > SRO > Leaching (Figs. 3.4-3.15; Table 3.2) (Lerch et al., 1995; Blanchard and Donald, 1997; Lerch and Blanchard, 2003). The combination of the restrictive clay layer and highly erodible surface soils resulted in a high risk for herbicide transport by SRO and ARO in Young's Creek watershed. Under saturated conditions, water is perched above the claypan, creating conditions that favor surface runoff (Blanco-Canqui et al., 2002). The risk of transport by SRO and ARO is proportional to the AWC above the claypan; thus, providing sensitivity for distinguishing between claypan soils, such as the Putnam, Adco, Mexico, and Leonard soil series (see discussion below). Additionally, because the surface soils in the watershed are highly erodible (mostly silt loams), they are easily entrained and transported by surface runoff, leading to a very high risk for ARO (Figs. 3.12-15).

The differences seen over time and between herbicides are a result of the sorption and degradation functions. For example, risk for ARO increases from Day 1 to Day 7 for glyphosate (Fig. 3.14 b and c, Fig.3.19). Glyphosate is a strongly sorbing compound (K_d= 55), and as herbicide sorbs to soil particles within the first few days after application, risk for ARO for strongly sorbing compounds actually increases. After the first few days following application, once adsorption is complete, the biotic degradation process becomes dominant and risk subsequently decreases (Fig. 3.14 c, d, e, f). Landscape sensitivity and distinction between soil series also becomes more apparent with time. One specific example is seen in the leaching risk for DKN (Fig. 3.5). In Day 0 only steeply sloping and alluvial soils are distinguished; by Day 1, differences among claypan soils such as Leonard and Mexico 1-4% are evident (Fig. 3.5 a and b).

Landscape Sensitivity

The model was sensitive to differences in soil properties across the landscape. Alluvial soils have a high risk of leaching and upland soils have a high risk for transport by SRO and ARO. An illustration of a typical hillslope soil association is seen in Figure 3.2. The Putnam, Adco, Mexico, and Leonard series are all claypan soils. The Moniteau is an alluvial soil and lacks a claypan. Specific examples of landscape sensitivity along a soil association are illustrated in Figures 3.16-3.18. In Figure 3.16, the greatest risk of metolachlor loss by leaching was for the alluvial Moniteau soil. This alluvial soil lacks a restrictive clay layer (Bti < 35) and is therefore vulnerable to metolachlor transport through leaching. The soils with claypans (Putnam, Adco, Mexico, and Leonard) have much lower leaching risk because a restrictive clay layer (Bti > 35) prevents the downward movement of water and dissolved herbicides. The depth to the water table is below the minimum K_{sat} and/or the restrictive clay layer for the Adco and Leonard series, which results in lower leaching risk for these soils than the Putnam and Mexico soils. For atrazine loss by SRO, the Mexico and Leonard series had the greatest risk for transport (Fig 3.17). These soils, located on the backslope position, have less AWC than the Adco soils on the shoulder position, so risk slightly increases when moving from the shoulder to the backslope. In the alluvial Moniteau soil, the risk for atrazine loss by SRO sharply decreases compared to upland landscape positions because this soil has a much higher AWC, lacks a restrictive clay layer, and has low slope. The trend for risk of glyphosate loss by ARO (Fig. 3.18) is very similar to that seen for atrazine by SRO, but in this scenario, the sorption intensity among the different soils becomes apparent (also see Figs. 3.14a-c). For example, the risk of glyphosate transport by ARO for the Armstrong loam and the Leonard silt loam soils increased at Days 1 and 7 compared to Day 0 because these two claypan soils have greater OM content and greater adsorption weights in the model compared to the Mexico, Adco, and Putnam series. Thus, the basis for landscape sensitivity between soils resulted from the application of the restrictive clay layer criterion (Bti) and the associated penalties for soils with restrictive clay layers (i.e., AWC and severe runoff) as well as from differences in sorption intensity among soils. Furthermore, the model correctly identified differences between claypan alluvial soils, as well as facilitated distinctions across landscape even for claypan soils of very similar properties that primarily differed in their depth to the claypan or OM content.

Herbicide Comparisons

The model correctly discerned differences among herbicides with a wide range of properties, and was sensitive to differences in K_d and dissipation half-life. The herbicide with the highest sorption intensity (glyphosate) showed the highest risk for transport by ARO and the lowest risk for leaching (Figure 3.19, Table 3.2). Conversely, atrazine and DKN showed highest risks for loss by leaching and SRO. DKN and metolachlor showed the lowest risks for all hydrologic pathways by Day 30 (Figure 3.19), reflecting their shorter half-lives compared to atrazine and glyphosate.

Watershed Risk

Based on the mean W_R in Table 3.2, metolachlor and DKN would be the best overall choices for minimizing the risks of transport in Young's Creek watershed. However, if leaching were the main concern in this watershed, metolachlor would be the best choice because its strong soil sorption leads to decreased transport by leaching compared to the other three herbicides evaluated. If transport by ARO were the main concern, then DKN would be the best choice. Given the runoff prone nature of the soils in the study area, SRO and ARO are the two pathways of greatest concern for Young's Creek watershed. On this basis, DKN would be the best choice as it minimized the overall risk (i.e., average of W_R for SRO and ARO in Table 3.2) for these two pathways compared to the other herbicides. Of course, other considerations will influence the "best" herbicide choice(s), such as application rates, weed control spectrum, and cost. The ability of the model to objectively quantify the risk of hydrologic transport for all three major pathways combined with its sensitivity to soil properties highlights its value as a management tool for minimizing the environmental impact of hesticides.

Model Limitations

While the model offers reasonable sensitivity to pesticide, soil, and landscape properties, it still over-estimated risk at Day 0 for leaching. The model predicts the risk for leaching is moderate to high at Day 0, indicated by the yellow and red hues on the maps in Figures 3.4a-3.7a. The model accurately assigns higher risk to the alluvial areas which have better drained soils and potentially more risk of leaching. However, the risk of leaching on summits and side slopes should be categorized as low risk given the claypan horizon that impedes downward movement of water, and the low levels of herbicides detected in groundwater beneath these upper landscape positions (Blanchard and Donald, 1997).

Time-dependent functions (biotic degradation, adsorption, hydrolysis, and volatilization) under-estimated dissipation. Many dissipation processes take place over a relatively short time span and are complete after a few days (adsorption, hydrolysis, and volatilization) or at most a few weeks (biotic degradation). For the compounds included in this study, at least 4 half-lives had occurred by Day 180, and all hydrologic pathways should converge to very low risk (dark green). However, as seen in the maps of Figures 3.4e-3.15e, risk for ARO loss was still moderately high (red/pink) for some soils by day 180, indicating that the time-dependent functions significantly underestimated herbicide dissipation and/or the soil-hydrologic functions significantly overestimated risk at Day 0.

A weakness present but not apparent in the figures and tables was that the computation of the S_R (Eq. 3.5) caused the model sensitivity to decrease with increased number of comparisons. As the number of soil, watershed, or herbicide choices increases, the range of potential model weights for a given pathway also increases. Thus, the S_R values were scaled over a broader range when more comparisons were included, decreasing the over all sensitivity of the model. This partly explains the over-estimation of risk at Days 100 and 180. Model sensitivity will be greatest when the number of comparisons is limited. One solution would be to limit temporal comparisons to the critical transport period of approximately 60 days after application and limit herbicide comparisons to two compounds. Some would argue that this severely limits the applicability of the model. However, after more than about 60 days, the risk of herbicide transport is known to be very low based on field and watershed studies (e.g., Lerch et al., 1995; Ghidey et al., 2005). Limiting comparisons to this critical period would be of most interest to land managers without diminishing the usefulness of the model. Limiting herbicide comparisons to just two compounds at a time, however, does weaken the utility of the model. As an alternative, including a relative mass-balance or "fraction remaining" approach to the model has potential to address the issue of decreased sensitivity with increased comparisons, and it would properly account for risk with time after application. A relative-mass balance approach would calculate risk in relation to the fraction of the applied pesticide remaining at any time after application; thus, more accurately assessing changes in risk over time. It is referred to as a "relative-mass balance" because it would be calculated as a the relative change in the initial application . The calculation would be based on the difference between 1 (i.e., 100% of applied at Day 0) and the sum of all the dissipation functions (biotic degradation, hydrolysis, and volatilization).

CONCLUSIONS

Development of restrictive clay layer criteria was essential for accurately assessing hydrologic risk of herbicide transport in a claypan watershed. After inclusion of the restrictive clay layer criterion, the predicted risks between hydrologic pathways were more consistent with previous field research, where the risks for SRO and ARO were much higher than for leaching. The model was demonstrated to be sensitive to soil properties, landscape position, and herbicide properties. Sensitivity to soil properties and landscape position was demonstrated by the model's ability to discern differences in risk not only between claypan and alluvial soils, but also among claypan soils. The model was sensitive to differences in K_d and $t_{1/2}$ among the four corn herbicides tested, indicating that it is useful for comparing herbicides with a wide range of chemical properties.

The index model approach allows for watershed-scale quantification of herbicide risk of transport using time-integrated, area-weighted watershed risks. Maximum sensitivity requires that the number of comparisons over time or herbicide choices be limited (i.e. pair-wise comparisons of herbicides, limit time scale to 60 days), or the incorporation of a relative-mass balance approach to application of the time-dependent functions. The developed model can be used to target management practices to the most vulnerable areas within watersheds and as a decision support tool for making environmentally sound herbicide choices. The time-integrated watershed scores can be used for decision-making regarding herbicide choice(s) that would result in the least contamination of water resources for a given transport pathway. Given the flexibility of the model it may be feasible to broaden the application to assess risk for other agrochemicals such as nutrient transport and soil erosion. Additionally, with minor adjustments, field-scale application of the model would also be a practical addition to the current capabilities of the model. A field scale version of the model would allow for assessment of more specific management options, such as limiting application rates or the effects of soil incorporation on risk of hydrologic transport.

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Table 3.1 Herbicide properties.

	K_d (sorption coefficient)	Half life (days)	Vapor Pressure (mm Hg at 25° C)
Atrazine	1.44[†]	36[†]	2.9×10^{-7}[‡]
DKN	0.77[§]	12[¶]	NA
Glyphosate	55[¥]	40[††]	NA
Metolachlor	1.83[‡‡]	27[¥]	3.1×10^{-5}[¥]

[†](Kazemi et al., 2008). [‡](Ahrens, 1994). [§](Bresnaham et al., 2004; Koskinen et al., 2006). [¶](Lin et al., 2003). [¥](Montgomery, 1993). [††](Geisy et al., 2000). [‡‡](Seybold and Mersie, 1996; Weber et al., 2000).

Table 3.2 Time-integrated watershed risk (W_R) for each herbicide and hydrologic pathway.

Herbicide	Leaching	SRO	ARO	Mean
	---------------------------------Score Days	-------------------------------		
Atrazine	634[†]	981	1128	914
DKN	617	964	1022	868
Glyphosate	588	934	1207	910
Metolachlor	587	933	1114	878

[†]**The lower the W_R, the lower the risk.**

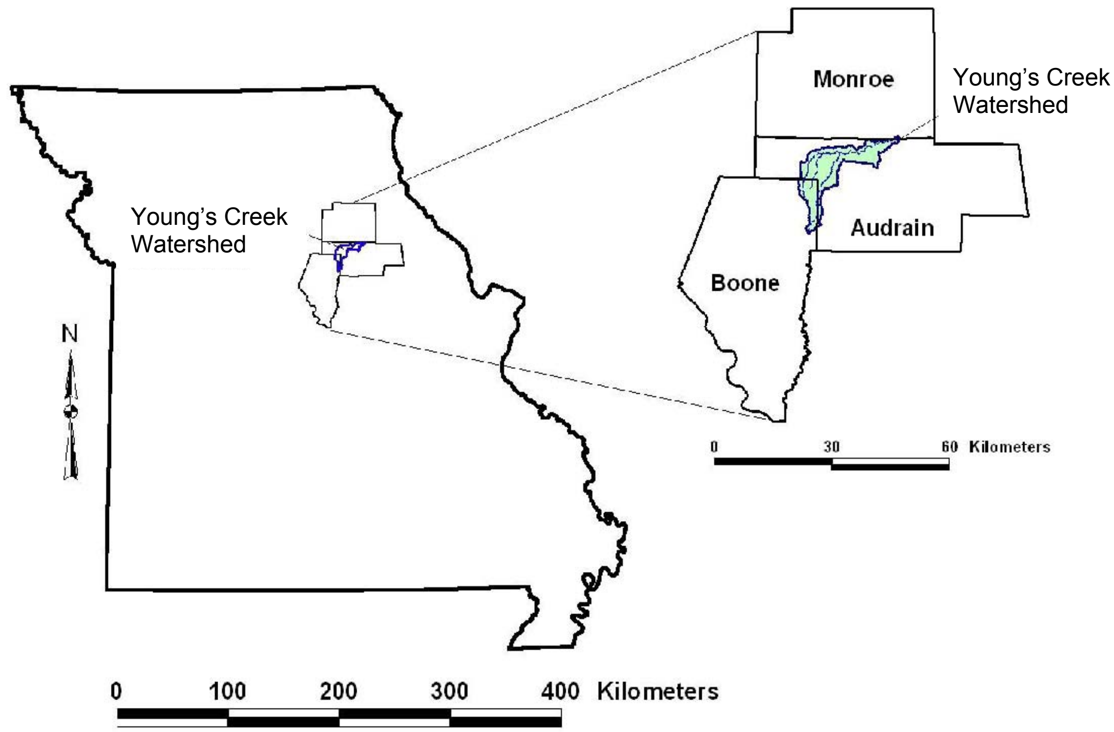

Figure 3.1. Young's Creek Watershed located in north-central Missouri.

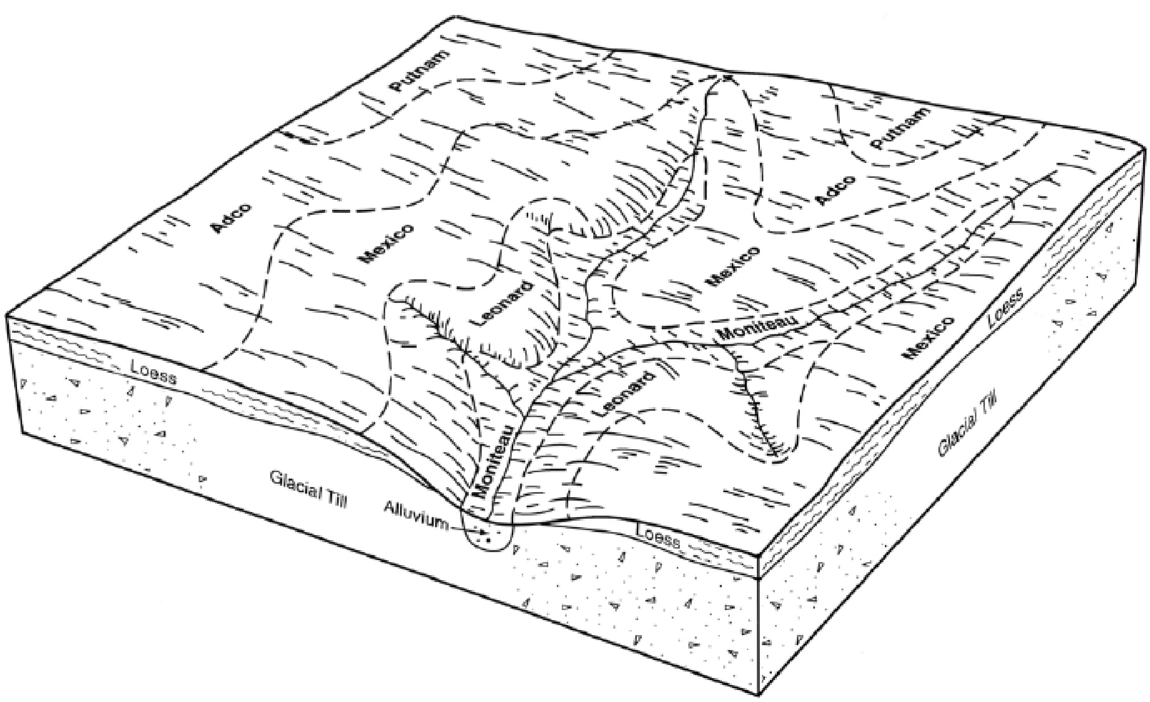

Figure 3.2. Geomorpholgy model of Adco-Mexico-Putnam Association (USDA-NRCS, 2001).

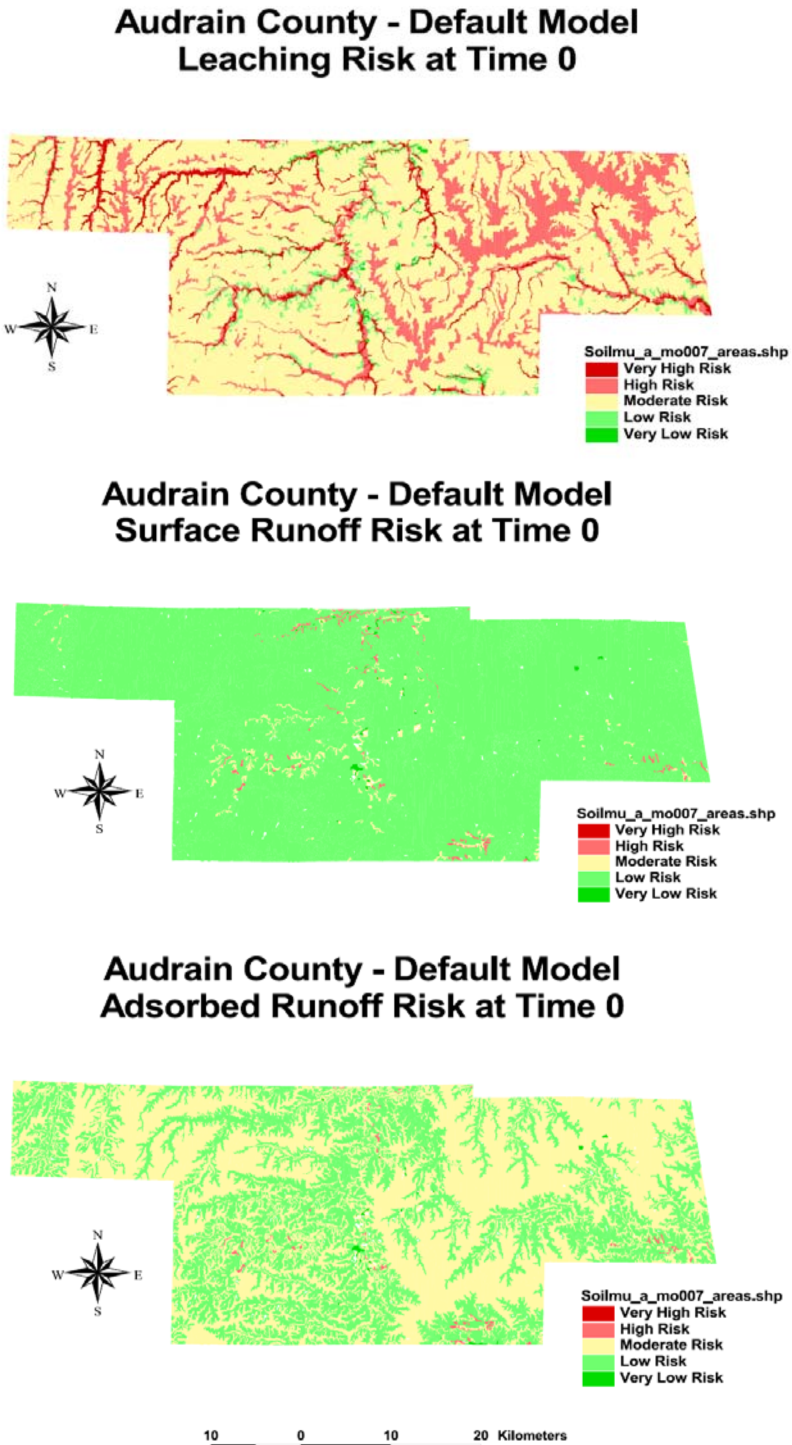

Figure 3.3. Application of the default model functions for the claypan soils of Audrain County, MO showed highest risk for herbicide transport by leaching and lowest risk for surface runoff. This finding was in opposition to field studies in claypan watersheds.

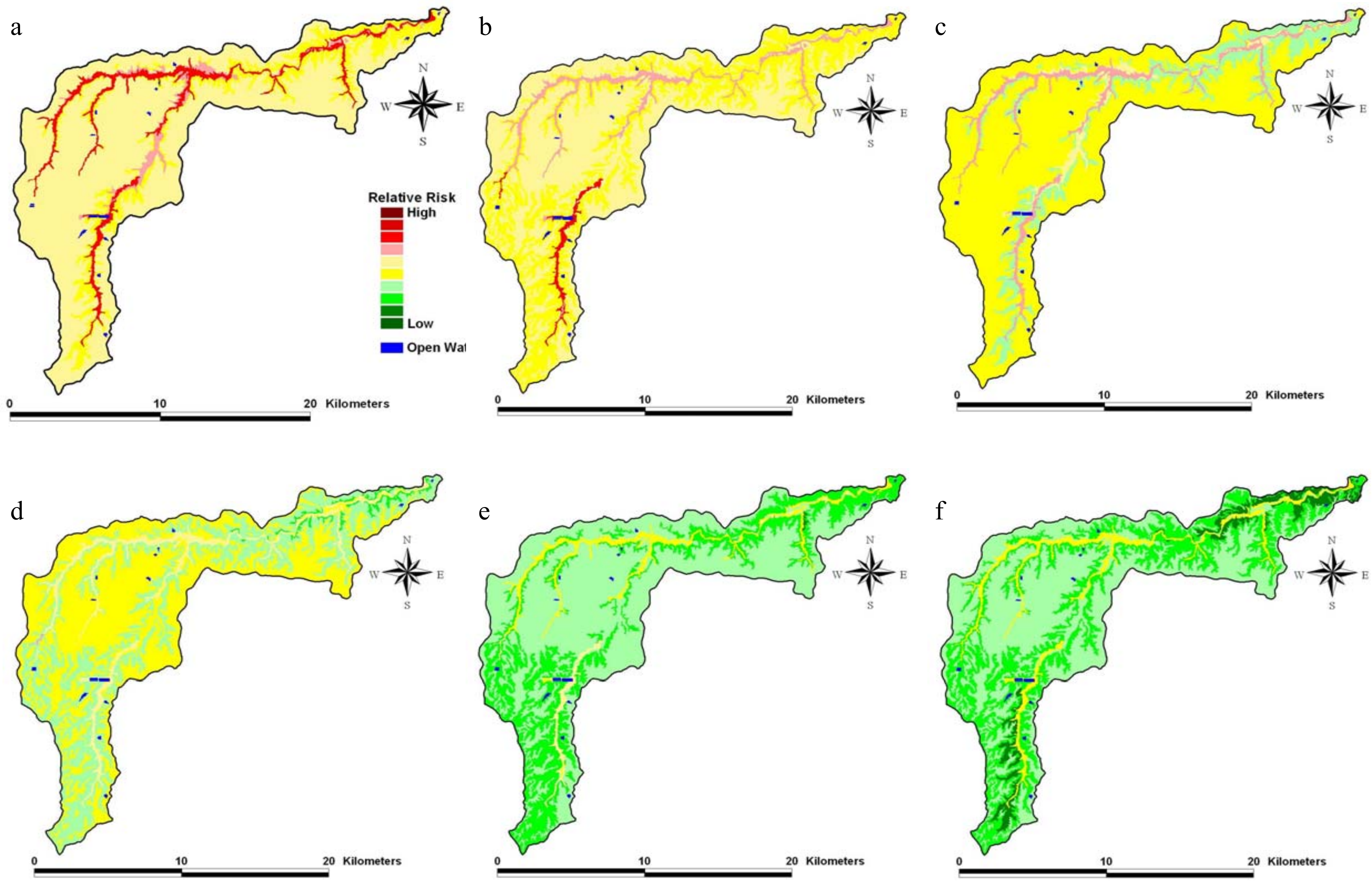

Figure 3.4. Atrazine leaching risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, f. Day 180.

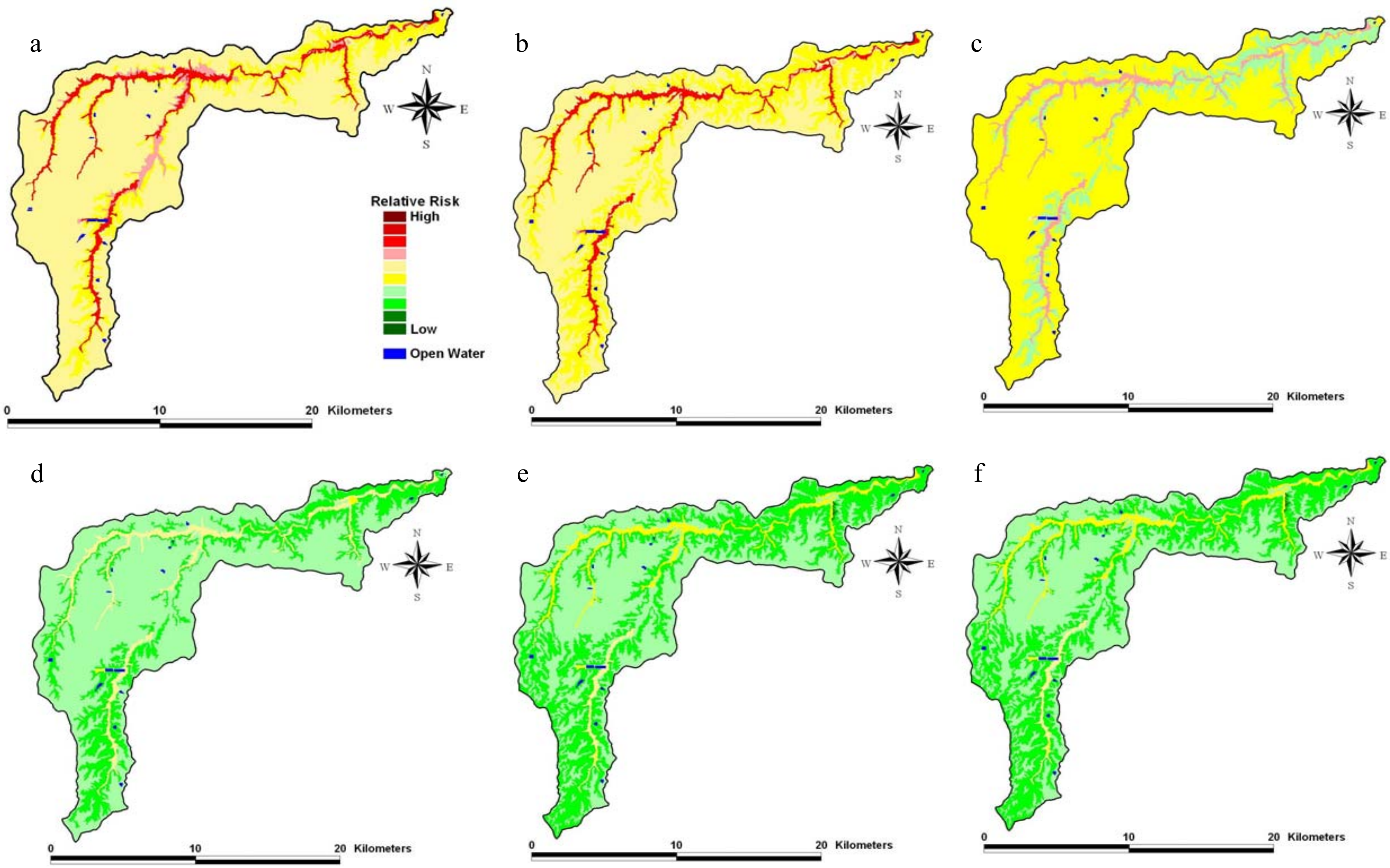

Figure 3.5. DKN leaching risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, f. Day 180.

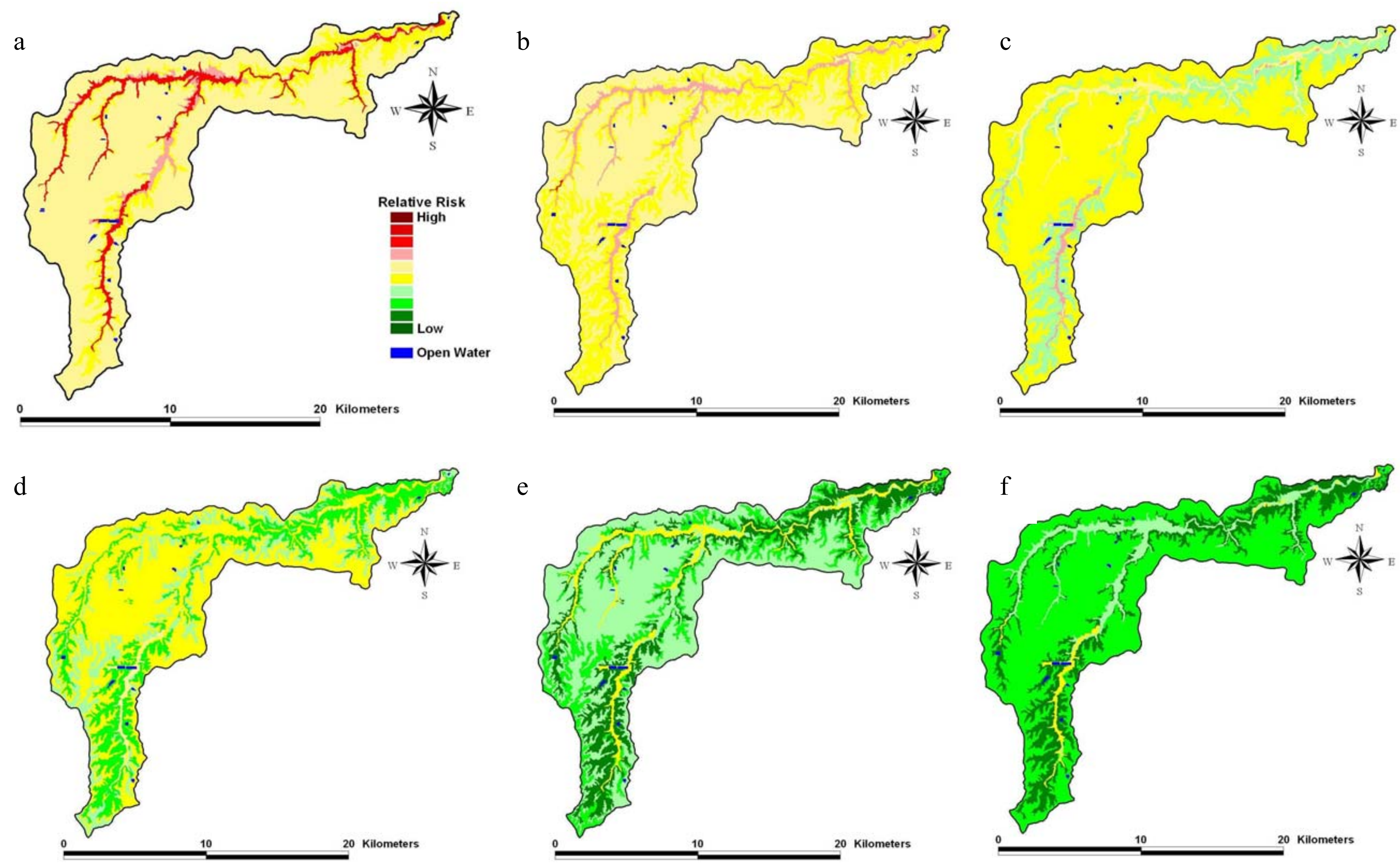

Figure 3.6. Glyphosate leaching risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, f. Day 180.

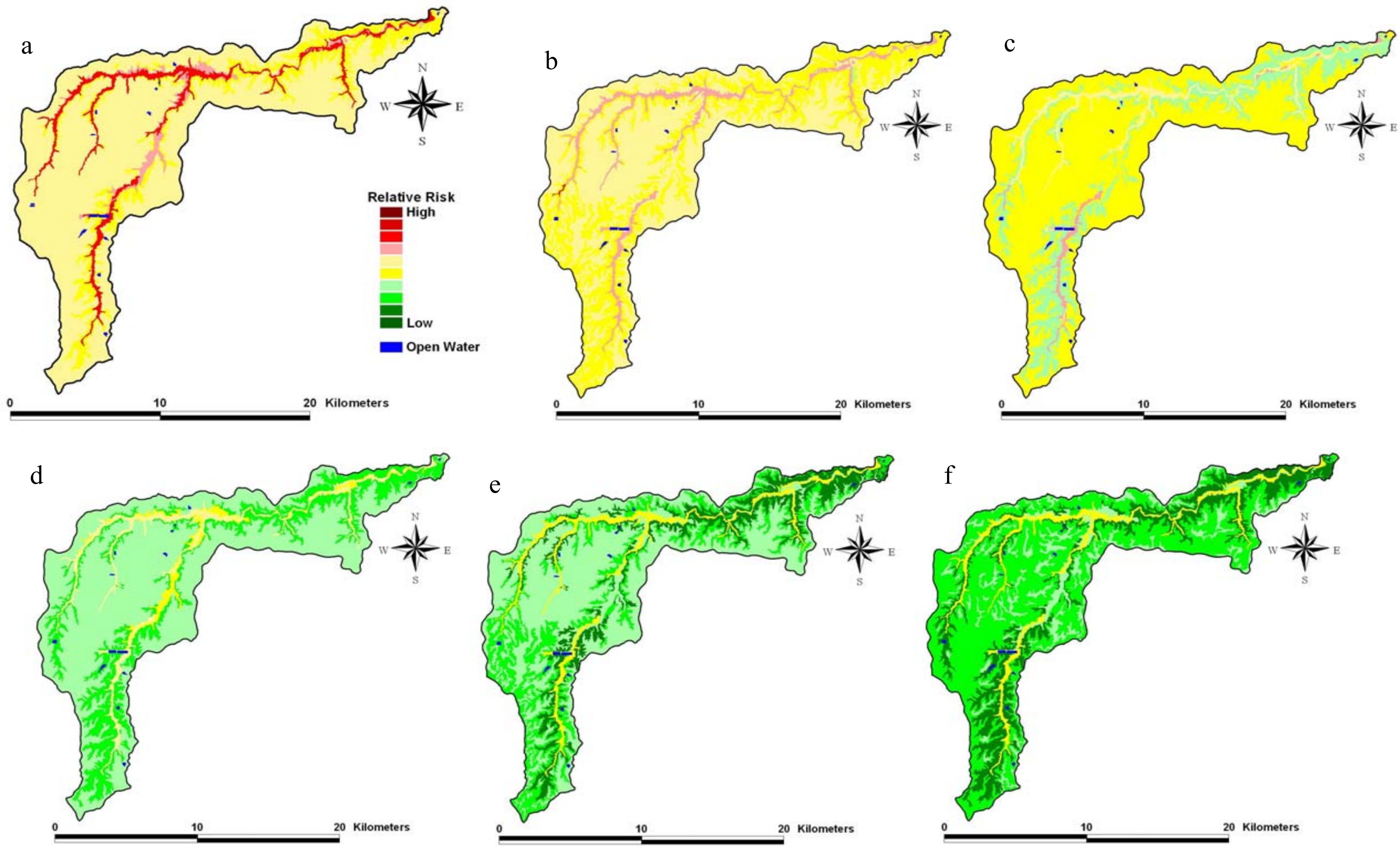

Figure 3.7.Metolachlor leaching risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, f. Day 180.

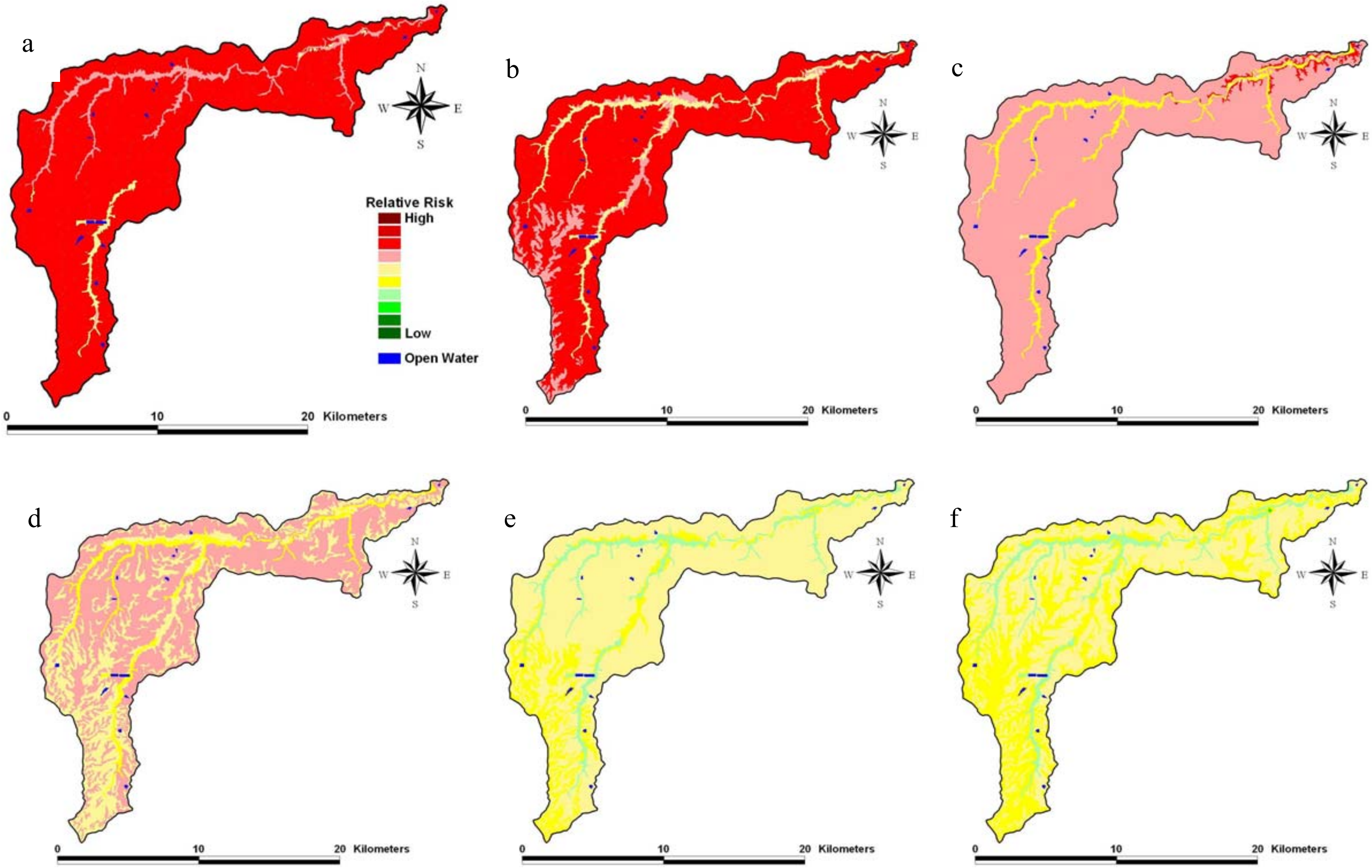

Figure 3.8. Atrazine solution runoff risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, f. Day 180.

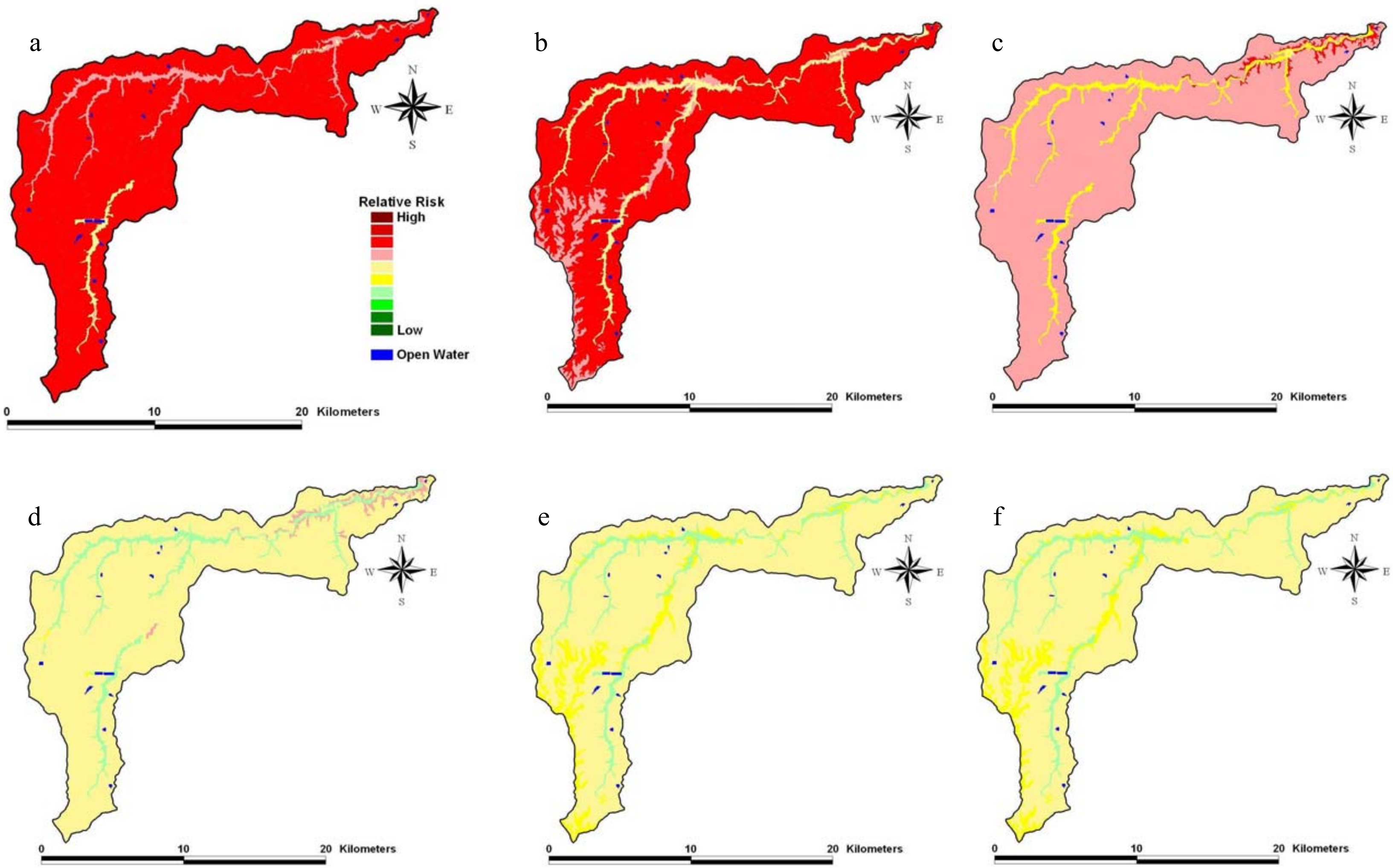

Figure 3.9. DKN solution runoff risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

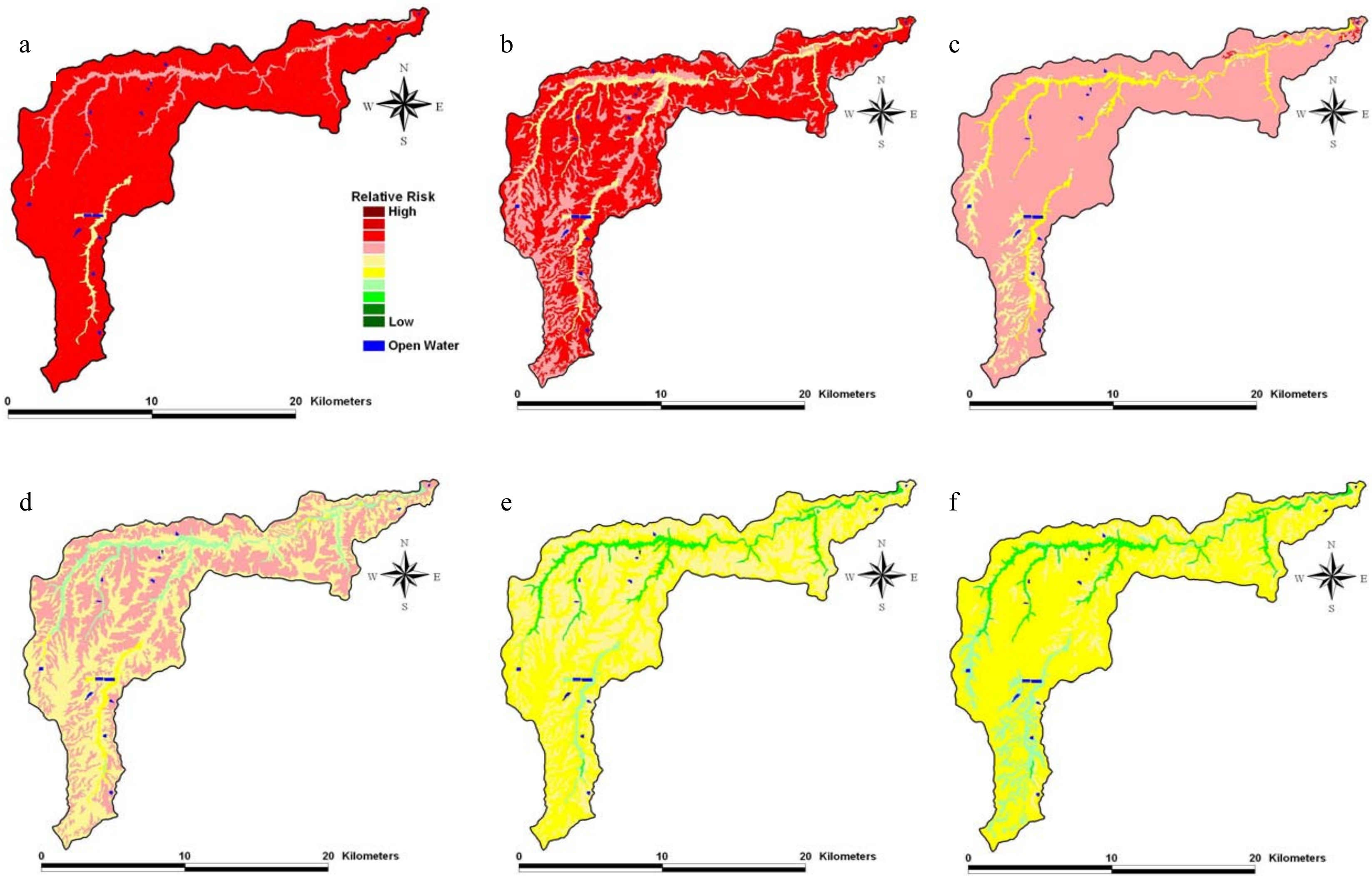

Figure 3.10. Glyphosate solution runoff risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

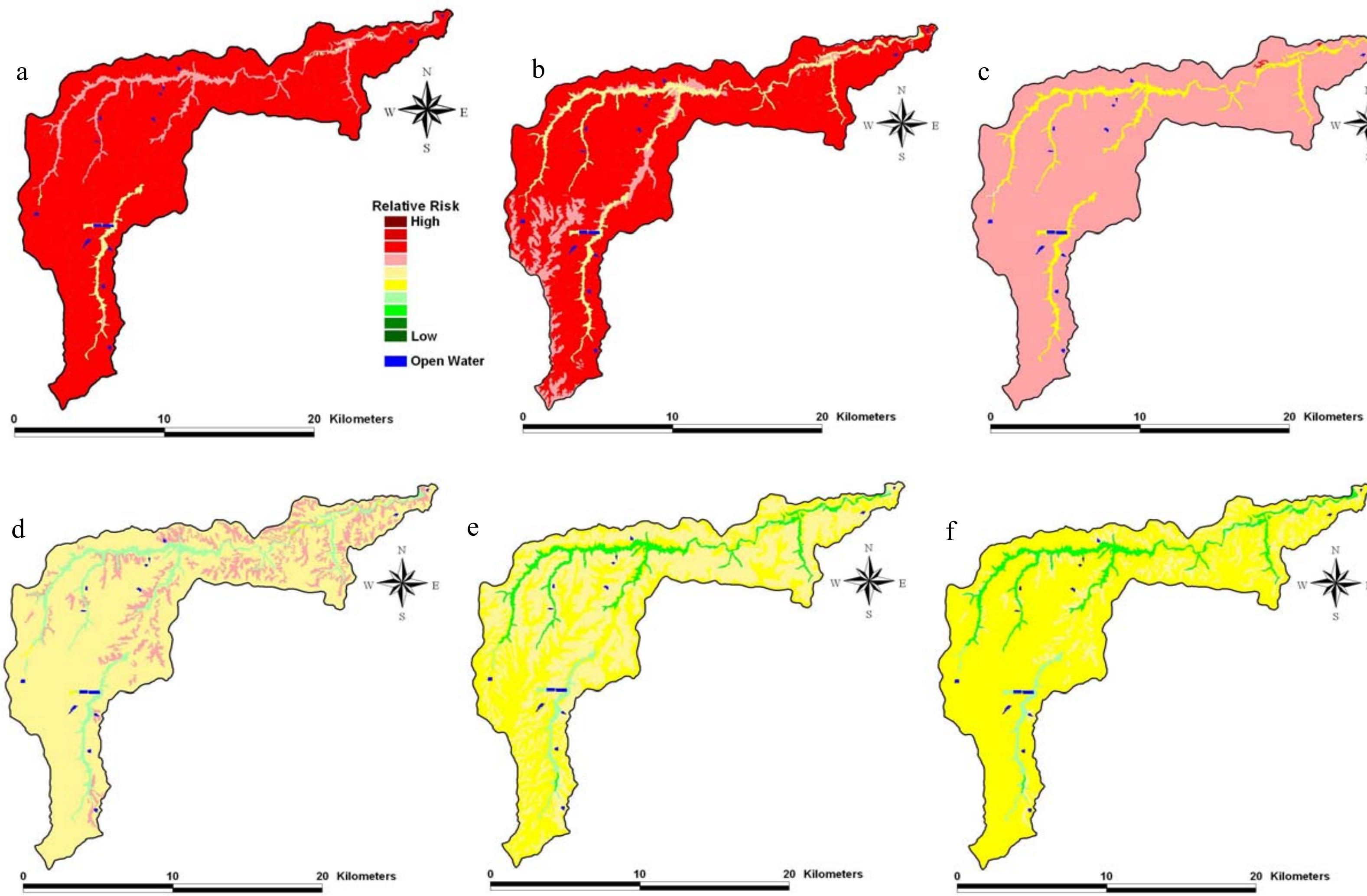

Figure 3.11. Metolachlor solution runoff risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

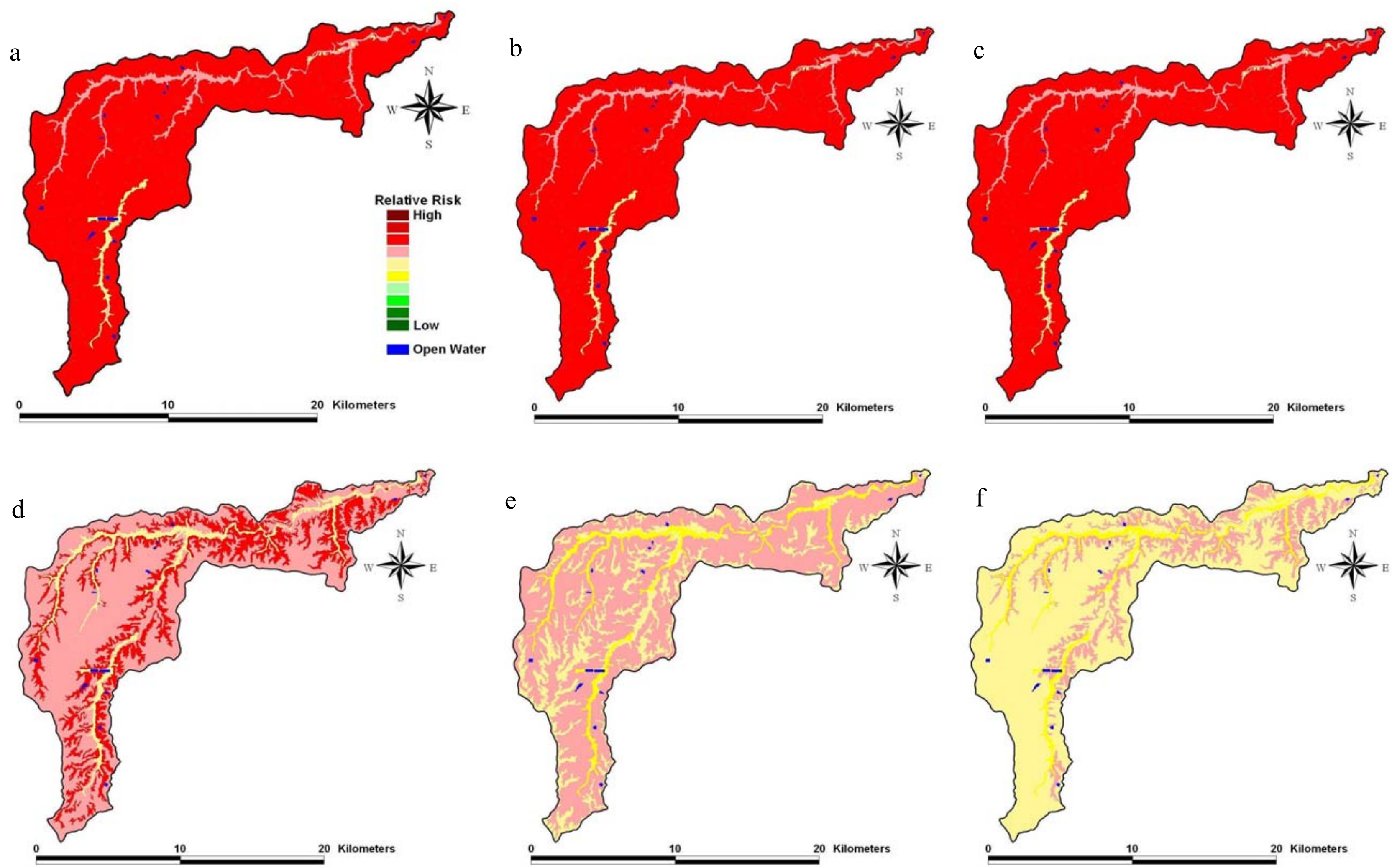

Figure 3.12. Atrazine particle adsorbed runoff risk at a. Day 0, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

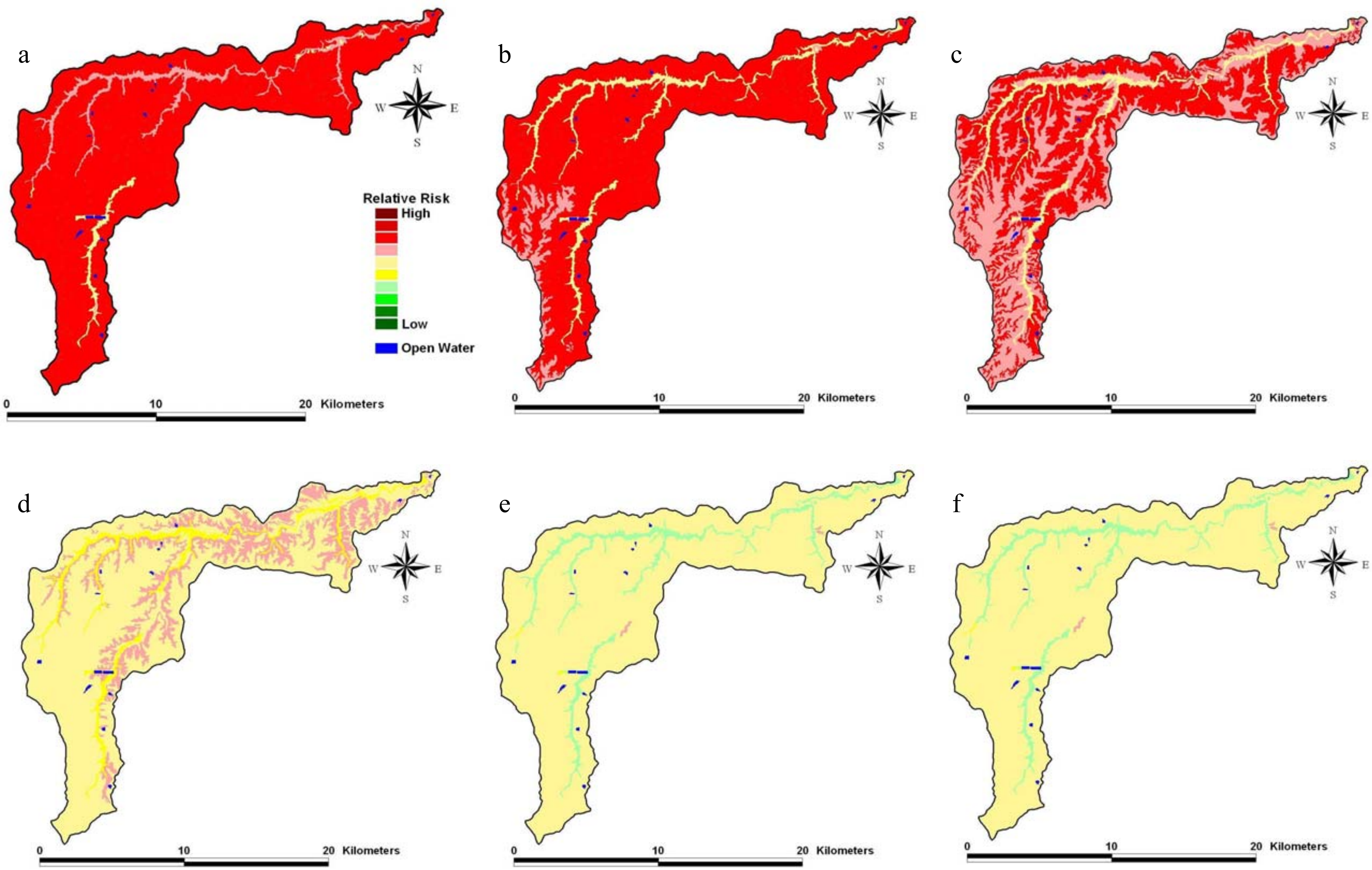

Figure 3.13. DKN particle adsorbed runoff risk at a. Day o, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

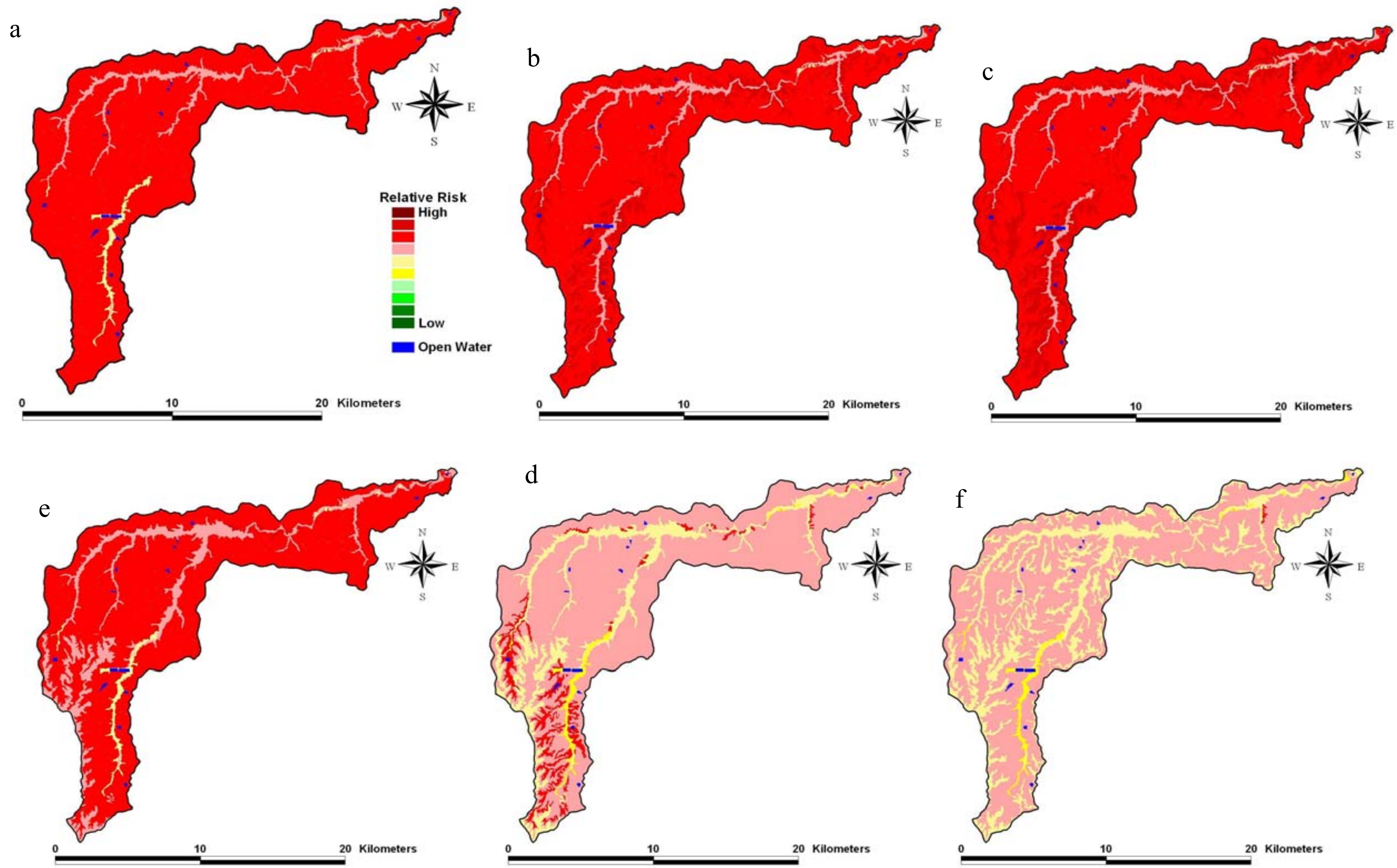

Figure 3.14.Glyphosate particle adsorbed runoff risk at a. Day o, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

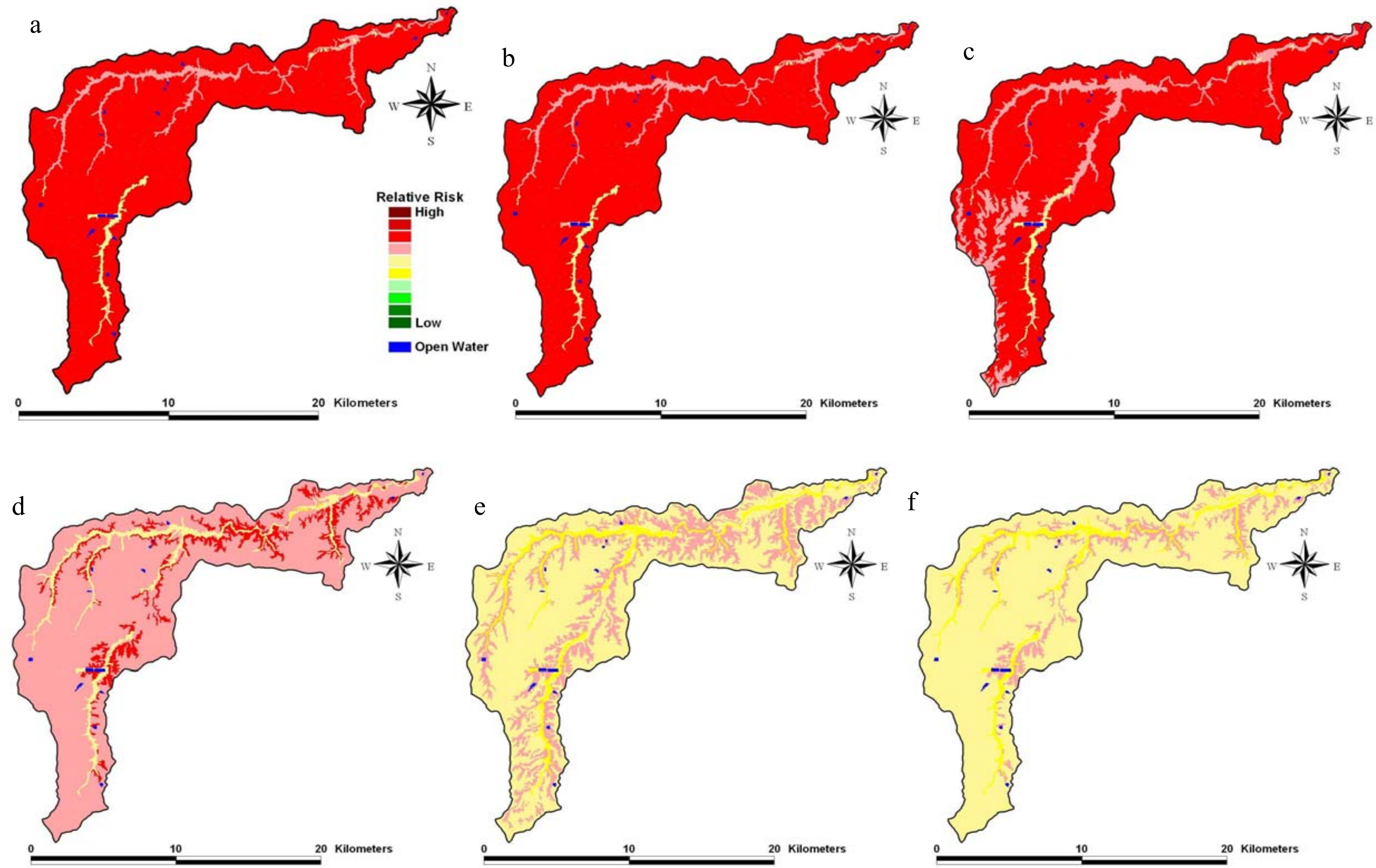

Figure 3.15.Metolachlor particle adsorbed runoff risk at a. Day o, b. Day 1, c. Day 7, d. Day 30, e. Day 100, and f. Day 180.

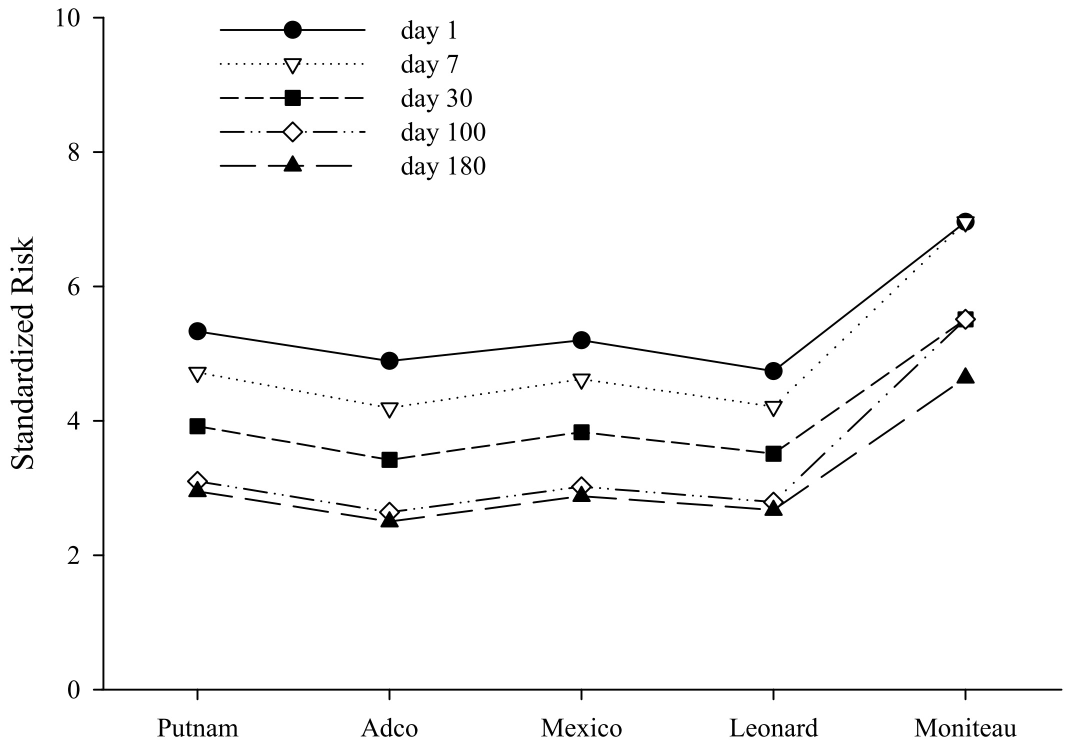

Figure 3.16. Risk of metolachlor loss by leaching over time along the Adco-Mexico-Putnam Association.

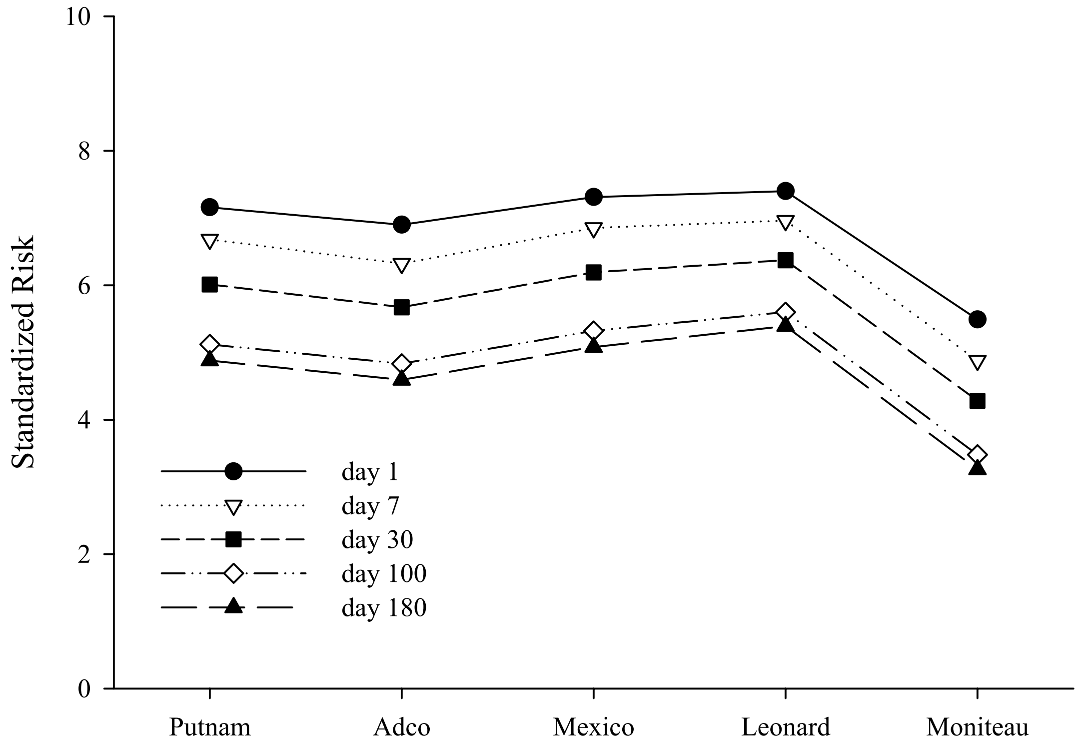

Figure 3.17 Risk of atrazine loss by SRO over time along the Adco-Mexico-Putnam Association.

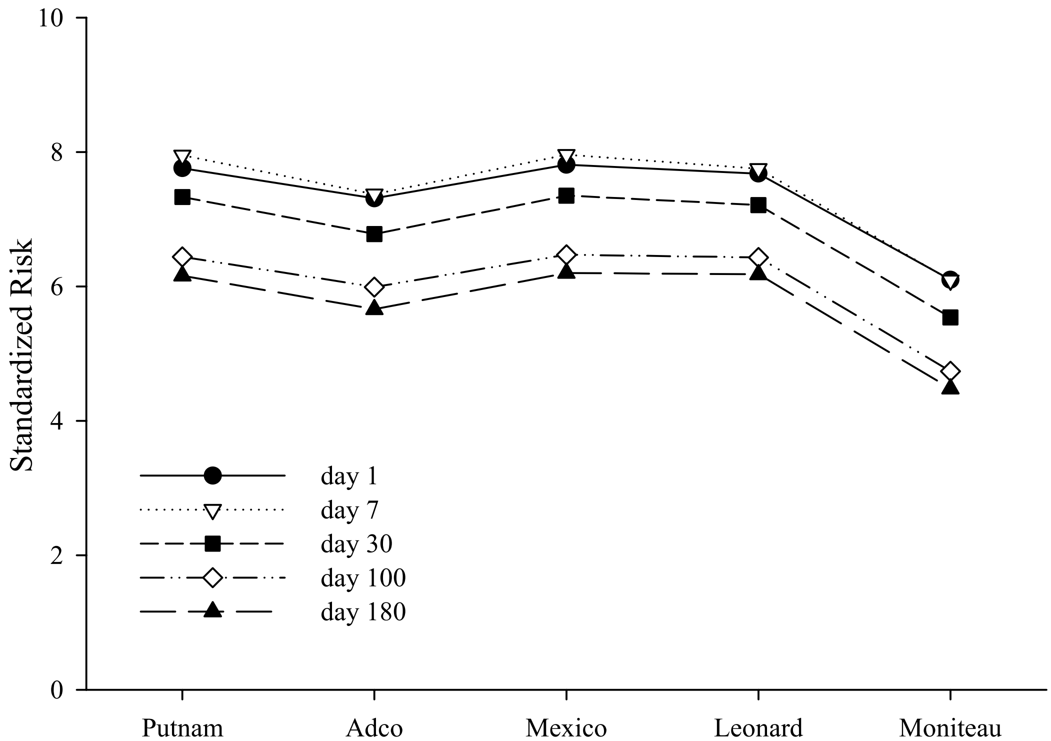

Figure 3.18. Risk of glyphosate loss by ARO over time along the Adco-Mexico-Putnam Association.

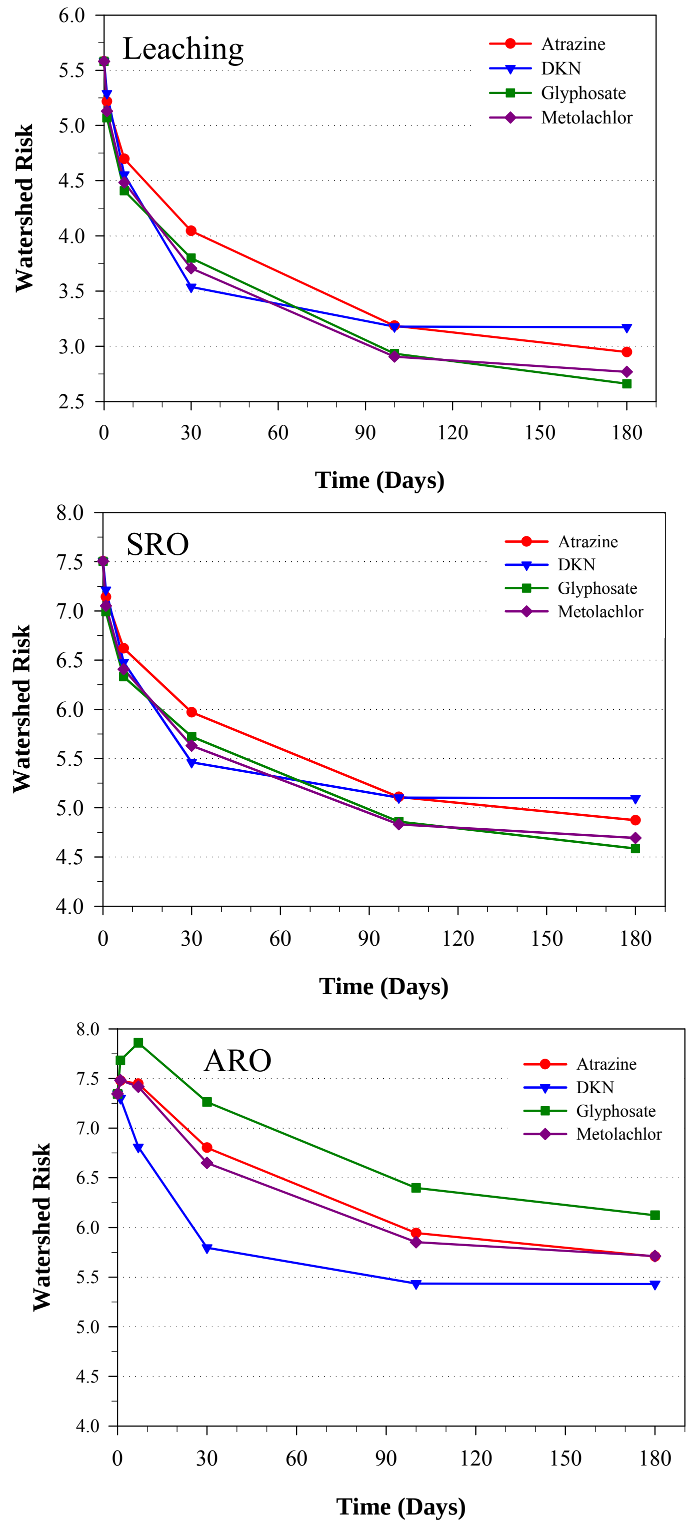

Figure 3.19. Watershed risks over time for atrazine, DKN, glyphosate, and metolachlor for each hydrologic pathway.This data was used to compute the watershed risks (W_R) in Table 3.2.

APPENDIX A

Site Name/ Plot Designation	Horizon depth (cm)	Description
1st order Crop 1		
Plot A2	0-35	Silt loam, brown, granular
	35-75	Sandy clay loam, sub-angular blocky, brown to gray with mottles, Fe concretions
Plot A3	0-55	Silt loam, brown, sub-angular blocky
Plot B2	0-40	Silty clay loam, brown, sub-angular blocky
	40-80	Silty clay loam, brown to gray with red mottles and Fe concretions, angular blocky
1st order Crop 2		
Plot A1	0-35	Silty clay loam, sub-angular blocky, brown
Plot B2	0-40	Silt loam, brown, sub-angular blocky
	40-80	Silt loam, angular blocky, gray-brown with red mottles
Plot B3	0-20	Silt loam, granular to sub-angular blocky, brown to gray with red mottles
	20-65	Clay loam, platy, gray with red mottles
	65-105	Silt loam, brown with gray and red mottles, sub-angular blocky, roots
1st order crop 3		
Plot A1	0-25	Silty clay loam, gray, sub-angular blocky
	25-65	Silty clay, sub-angular blocky, gray
	65-105	Clay, sub-angular blocky, gray with red mottles
Plot A2	0-25	Silty clay loam, granular to sub-angular blocky, brownish gray
	25-65	Clay, sub-angular blocky, light gray
	65-115	Clay, sub-angular blocky, dark gray with red mottles
Plot B1	0-40	Silty clay loam, platy, brown-gray
	40-75	Silty clay, sub-angular blocky, gray
Plot B4	0-20	Silt loam, granular to sub-angular blocky, brown-gray
	20-40	Clay, sub-angular blocky, gray

Site Name/ Plot Designation	Horizon depth (cm)	Description
	40-65	Clay, sub-angular blocky, gray with red mottles
1st Order Forest		
Plot A1	0-30	Silt loam, brown, granular
	30-65	Silt loam, brown gray, platy to sub-angular blocky
	65-125	Silty clay loam, sub-angular blocky, dark brown gray
Plot A3	0-20	Loam, brown, platy
	20-55	Loam, brown, sub-angular blocky
	55-80	Sand loam, very weak sub-angular blocky, brown with red mottles and Mn concretions
Plot B2	0-15	Silt loam, sub-angular blocky/granular, light brown
	15-45	Silt loam, platy/sub-angular blocky, brownish gray
	45-100	Silt loam, dark brownish gray, sub-angular blocky
Plot B4	0-25	Silt loam, granular, brown
	25-65	Silt loam, dark gray with red mottles, sub-angular blocky
1st order Forest 2		
Plot A1	0-20	Silt loam, light brown to brown, weak sub-angular blocky
	20-40	Silty clay loam, reddish brown, sub-angular blocky
	40-65	Silt loam, brown granular
	65-90	Sandy loam, reddish brown to light brown, weak blocky
	90-135	Sandy loam, gray to reddish brown with reddish orange mottles, weak platy
	135->170	Abrupt top boundary, sandy clay loam, weak blocky, light to reddish brown with gray mottles
Plot B2	0-55	Silt loam, brown to reddish brown, granular
	55-100	Silty loam, weak platy, dark gray and brown
	100-125	Clay loam, platy, light brown and medium gray with orange mottles
	125-160	Clay loam, sub-angular blocky, light gray with orange mottles, Mn concretions, dark gray clayey inclusions

Site Name/ Plot Designation	Horizon depth (cm)	Description
Plot B4	0-25	Silt loam, brown, granular
	25-45	Clay loam, small gravel, light gray to reddish brown, weak sub-angular
	45-75	Sandy clay loam, small gravel, light gray to dark gray with reddish orange mottles, few Mn concretions, sub-angular blocky
	75-150	Clay loam, small gravel, gray with reddish orange mottles, platy structure
	150-190	Silty clay loam, light gray with reddish orange mottles, sub-angular blocky, Mn concretions and small gravel
1st Order Forest 3		
Plot A2	0-45	Silt loam, granular to sub-angular blocky, reddish brown
	45-95	Loam, reddish brown to brown, sub-angular blocky
	95-150	Silty clay loam, brown, sub-angular
Plot A3	0-50	Loam, light brown, sub-angular blocky
	50-150	Clay loam, light brown to gray with red mottles, Mn concretions, platy, cemented
	150-250	Clay loam, light brown to gray, intense orange mottles, Mn/Fe concretions, platy
	250-280	Sandy clay loam, light brown with orange mottles, Mn concretions
	280-325	Clay loam, gray to orange, copious Mn coated Fe oxides, cobbles
Plot B1	0-75	Silt loam, granular, brown to reddish brown
	75-100	Loam, sub-angular blocky, reddish brown
	100-120	Silt loam, sub-angular blocky
Plot B2	0-65	Silt loam, brown-gray, sub-angular blocky
	65-130	Silty clay loam, brown-gray with red mottles, weak platy
1st Order Pasture 1		
Plot L2	0-20	Fine sandy loam, platy, grayish brown
	20-65	Loam, sub-angular blocky, dark gray-brown with red mottles
	65-120	Clay, sub-angular blocky, dark gray with red mottles
	120-170	Clay loam, massive, dark gray with red and gray mottles

Site Name/ Plot Designation	Horizon depth (cm)	Description
Plot L5	0-25	Silt loam, sub-angular blocky, brownish-gray
	25-125	Silty clay loam, fine sub-angular blocky to platy, gray with red mottles
	125-180	Clay, massive, dark gray with red mottles
Plot R4	Slump at top horizon 0-20 couldn't sample	
	20-90	Clay loam, sub-angular blocky, dark gray with red mottles
	90-170	Clay, sub-angular blocky, gray with red mottles
	170-220	Sandy loam, massive, brown
Plot R6	0-25	Silt loam, granular, dark brown
	25-110	Silty clay loam, sub-angular blocky, dark gray-brown
	110-195	Silty clay, weak sub-angular blocky, gray
1st Order Pasture 2		
Plot A1	0-25	Silt loam, brown, granular
	25-70	Silty clay loam, brown to gray, subang blocky
	70-100	Silty clay loam, strong angular blocky, gray to brown with red mottles, Mn conc
Plot A2	0-35	Silt loam, brown, granular
	35-75	Silty clay loam, brown, weak platy
	75-100	Silty clay loam, gray with strong reddish mottles
Plot B4	0-25	Silt loam, granular, brown
	25-60	Silty clay loam, brown with orange mottles (ferric) inclusions, sub-angular blocky
	60-100	Silty clay loam, sub-angular blocky, gray-brown with ferric inclusions
Plot B5	0-45	Silt loam, brown, granular
	45-95	Silty clay loam, sub-angular blocky, brown
	95-130	Silt loam, brown to gray, ang blocky
1st Order Pasture		
Plot L3	0-30	Sandy loam, very weak sub-angular blocky, brown
	30-55	Clay loam, weak sub-angular blocky, brown

Site Name/ Plot Designation	Horizon depth (cm)	Description
	55-150	Sandy clay loam, weak sub-angular blocky, light brown to brown
	150->170	(loamy) sand, single grain, mottled, gray to reddish brown
Plot L5	0-20	Silty clay loam, brown to dark brown, weak sub-angular blocky
	20-65	Clay loam, light brown to reddish brown, sub-angular blocky
	65-105	Clay loam, light brown to reddish brown, with gray and red mottles, Mn concretions
	105->130	Sand, light brown to reddish brown, single grain, small gravel (glacial till?)
Plot R1	0-20	Silt loam, weak sub-angular blocky, dark brown
	20-95	Silt loam, weak sub-angular blocky, reddish brown
	95->160	Silty clay loam, sub-angular blocky, gray mottles
Plot R3	0-30	Clay loam, light brown to brown, weak sub-angular blocky
	50-105	Clay loam, gray to dark gray, strong platy structure
	105-165	Silty clay loam, weak sub-angular blocky, light gray to bright red
	165-250	Clay loam, sub-angular blocky to blocky (dark gray) weaker (light gray), light to dark gray with bright red mottles and Mn concretions
	250->270	Sandy loam, no structure- single grain, light gray to bright red, gray and red mottles, a few Mn concretions
1st Order Riparian Forest 1		
Plot L1	0-20	Silty clay loam, granular to sub-angular blocky, grayish-brown
	20-75	Clay, sub-angular blocky, brownish gray with red mottles
	75-115	Clay, massive, dark gray with red mottles
	115-235	Clay, massive, dark gray with red mottles
Plot L4	0-35	Silt loam, brown, granular to platy
	35-80	Silty clay loam, brownish gray, fine sub-angular blocky
	80-120	Silty clay, gray, sub-angular blocky

Site Name/ Plot Designation	Horizon depth (cm)	Description
	120-160	Clay, dark gray with red mottles, massive
Plot R2	0-35	Silt loam, platy to granular, brown
	35-110	Silty clay, sub-angular blocky, dark gray with red mottles
	110-145	Silty clay, sub-angular blocky, dark gray with red mottles
	145-195	Clay, massive, gray with red and gray mottles and Fe/Mn concretions
Plot R3	0-25	Silt loam, brown, granular to platy
	25-110	Silty clay loam, brownish gray, sub-angular blocky
	110-140	Clay, gray with red and gray mottles, massive
1st Order Riparian Forest 2		
Plot L1	0-30	Silt loam, granular to platy, brown
	30-105	Silty clay, sub-angular blocky, brown-gray with mottles
Plot L4	0-25	Silt loam, granular, brown
	25-50	Silt loam, brown-gray, platy/sub-angular blocky
	50-115	Silty clay loam, sub-angular blocky, brown
Plot R0	0-30	Silt loam, platy/granular, brown-gray
	30-100	Silty clay, sub-angular blocky, gray
Plot R1	0-25	Silt loam, granular, brown
	25-85	Silty clay loam, platy/sub-angular blocky, brown-gray
	85-130	Silty clay loam, sub-angular blocky, dark gray
Plot R4	0-35	Silty clay loam, granular, brown
	35-65	Silty clay loam, sub-angular blocky, brown-gray
	65-120	Silty clay, sub-angular blocky, gray with red mottles
1st Order Riparian Forest 3		
Plot L2	0-20	Silt loam, granular, grayish brown
	20-50	Silt loam, platy/sub-angular blocky, brownish gray
	50-100	Loam, weak sub-angular blocky, gray with red mottles
	100-145	Loam, fine sub-angular blocky, grayish

Site Name/ Plot Designation	Horizon depth (cm)	Description
		brown with re mottles
	145-250	Clay loam, massive, reddish brown with gleying along fracture
Plot L3	0-20	Silt loam, granular, brown
	20-45	Silt loam, grayish brown, platy
	45-80	Loam, platy/sub-angular blocky, light gray
	80-115	Clay, sub-angular blocky, brownish gray
	115-220	Clay, weak sub-angular blocky, gray with red mottles
	220-250	Clay loam, massive, reddish brown with gray mottles
Plot R2	0-25	Silt loam ,granular, grayish brown
	25-55	Silt loam, platy, gray with red mottles
	55-95	Clay loam, gray with red mottles, sub-angular blocky
	95-130	Clay, weak sub-angular blocky, gray with red mottles
	130-165	Clay loam, massive, brown with red and gray mottles
Plot R4	0-30	Silt loam, granular, grayish brown
	30-85	Clay, sub-angular blocky, brownish gray
	85-125	Clay, massive, gray with red mottles
Plot R5	0-50	Clay loam, sub-angular blocky, brown with red mottles
	50-85	Clay, sub-angular blocky, gray with red mottles
2nd Order Crop 1		
Plot C1	0-20	Silt loam, granular, brown
	20-90	Silty clay loam, sub-angular blocky, gray-brown
	90-145	Silty clay, gray-brown, sub-angular blocky
Plot C3	0-80	Silty clay loam, sub-angular blocky, gray
	80-165	Silty clay, sub-angular blocky, gray-brown
Plot D1	0-30	Silty clay loam, sub-angular blocky, gray-brown
	30-110	Silty clay loam, sub-angular blocky, gray-brown
	110-200	Silty clay, massive, gray-brown
Plot D3	0-90	Sandy loam, weak sub-angular blocky, light brown

Site Name/ Plot Designation	Horizon depth (cm)	Description
	90-125	Clay, sub-angular blocky, gray
2nd Order Crop 2		
Plot A2	0-55	Loam, granular, brown
	55-90	Silt loam, sub-angular blocky, brown
	90-165	Silt loam, sub-angular, brown
	165-200	Silty clay loam, brown to gray with red mottles, sub-angular blocky
Plot B1	0-60	Silt loam, brown, granular
	60-130	Silt loam, sub-angular blocky, brown to gray
	130-160	Silty clay loam, brown to gray brown, sub-angular blocky
	160-190	Silty clay loam, gray, mottles, sub-angular blocky
Plot B4	0-45	Silt loam, brown, granular to sub-angular blocky
	45-160	Silt loam, light brown to brown, sub-angular blocky
	160-220	Sandy clay loam, blocky, light brown to brown
2nd Order Crop 3		
Plot A1	0-35	Silt loam, brown, granular
	35-75	Silty clay loam, platy, brown
	75-125	Silty clay loam, brown to gray with gray mottles, platy
Plot A3	0-65	Silt loam, brown, granular, high OM content, oak leaf debris
	65-90	Silt loam with sand lenses, brown with light brown sand lenses, angular blocky
	90-105	Silty clay loam, borwn to gray with red and dark gray mottles, angular blocky
Plot B2	0-35	Silty clay loam, sub-angular blocky, reddish brown
	35-110	Silt loam, brown, sub-angular blocky
	110-180	Clay loam, orange-brown with gray mottles, sub-angular blocky
Plot B4	0-45	Silt loam, brown, angular
	45-165	Silty clay loam, weak platy, brown to light brown
	165-180	Silty clay loam, gray with red mottles, sub-angular blocky
2nd Order Forest 1		
Plot A1	0-20	Silt loam, brown to dark brown, weak

Site Name/ Plot Designation	Horizon depth (cm)	Description
		sub-angular blocky
	20-40	Clay loam, reddish brown, weak sub-angular blocky
	40-140	Silty clay loam, weak platy, gray to reddish brown, gray and red mottles, numerous Mn concretions
	140-215	Clay loam, sub-angular blocky, light gray to brownish red, some Mn concretions
Plot A2	0-35	Sandy clay loam, brown, weak sub-angular blocky
	35-45	Silt loam, brown to grayish brown, sub-angular blocky
	45-70	Sandy clay loam, light brown to brown, weak sub-angular blocky
	70-100	Sandy loam, weak sub-angular blocky, light brown to brown
	100-145	Loamy sand, light brown to reddish brown, very weak sub-angular blocky
	145-170	Sandy clay loam, light brown, gray and red mottles, sub-angular blocky to blocky
	170->185	Sand, gravelly, light gray, intense Mn concretions, gray and red mottles, single grain
Plot A4	0-35	Silt loam, brown, weak sub-angular blocky
	35-75	Clay loam, light gray to dark gray with yellowish-orange mottles, sub-angular blocky, Mn concretions
	75-140	Clay loam, light gray to dark gray with pinkish-red and orange mottles, very weak sub-angular blocky
	140-215	Clay, light to dark gray with orange mottles, weak platy to sub-angular blocky
Plot B3	0-20	Silt loam, light brown to brown, weak sub-angular
	20-80	Clay loam, reddish brown to light brown, sub-angular
	80-130	Sandy loam, light gray with red mottles and abundant Mn concretions, sub-angular
	130-150	Sand, light gray with red mottles, very abundant Mn concretions, weak sub-angular

Site Name/ Plot Designation	Horizon depth (cm)	Description
2nd Order Forest 2		
Plot A2	0-80	Silt loam, brown, sub-angular blocky
	80-160	Silty clay loam, light reddish brown to light gray, sub-angular blocky
	160-210	Sandy loam, reddish brown with light gray mottles
Plot A3	0-20	Silt loam, granular, brown
	20-50	Silty clay loam, weak platy, brown to light brown
	50-95	Silty clay loam, sub-angular blocky, light brown to gray
	95-120	Silty clay loam, angular blocky, reddish brown to gray
	120-170	Silty clay loam, angular blocky, gray with orange mottles
Plot B4	0-20	Silt loam, granular and sub-angular blocky, brown
	20-60	Silty clay loam, sub-angular blocky, brown
	60-120	Silty clay loam, angular blocky, brown
	120-165	Clay loam, light brown to gray with orange mottles, sub-angular blocky
2nd Order Forest		
Plot A1	0-30	Silt loam, brown, granular
	30-90	Silt loam, sub-angular blocky to granular, reddish brown
	90-120	Silt clay loam, gray brown, sub-angular blocky
	120-195	Silt clay loam, platy, gray with red mottles
Plot A3	0-30	Silty clay loam, brown, sub-angular blocky
	30-105	Silt loam, light brown to gray-brown, sub-angular blocky
	105-175	Clay loam, sub-angular blocky, light brown to brown
	175-240	Sandy loam, sub-angular blocky, light brown
Plot B2	0-80	Silty clay loam, granular to sub-angular blocky, brown
	80-110	Clay loam, sub-angular blocky, reddish brown to gray brown
	110-185	Silty clay loam, light brown, sub-angular

Site Name/ Plot Designation	Horizon depth (cm)	Description
		blocky
Plot B4	0-45	Loam, light brown to brown, granular
	45-170	Silt loam, sub-angular, reddish brown to dark brown
	170-205	Sandy loam, sub-angular blocky, light brown to dark brown with gray and red mottles
2nd Order Pasture 1		
Plot A1	0-20	Silty clay loam, granular, brown
	20-65	Silty clay loam, angular blocky, brown
	65-130	Silty clay loam, angular blocky, brown
	130-175	Clay loam, angular blocky, brown to gray, many worms
	175-195	Loam, massive, gray with red mottles
Plot A7	0-35	Silt loam, granular to weak platy, brown to light brown
	35-90	Silty clay loam, sub-angular blocky, brown to gray with red mottles
	90-120	Silty clay loam, platy, reddish brown to gray with few red mottles
	120-220	Silty clay loam, platy, gray with red mottles, Mn concretions
Plot B2	0-35	Silt loam, granular/platy, brown
	35-60	Silty clay loam, sub-angular blocky, grayish brown
	60-85	Silty clay loam, sub-angular blocky, grayish brown with brown splotches
	85-265	Sand, pockets of clay, massive, brownish gray
Plot B4	0-30	Silty clay loam, strong platy, brownish gray with red mottles
	30-65	Silty clay, platy/sub-angular blocky, brownish gray
	65-175	Silty clay, massive, gray with red mottles
Plot B8	0-30	Silty clay, sub-angular blocky, gray
	30-125	Silty clay, sub-angular blocky, brown with red and gray mottles
	125-175	Loam, massive, gray with red mottles
2nd Order Pasture 2		
Plot A1	0-40	Silt loam, granular, brown
	40-100	Silty clay loam, platy, brown to gray-brown
	100-150	Silty clay loam, brownish-gray, weak

Site Name/ Plot Designation	Horizon depth (cm)	Description
		platy
	150-205	Silt loam, light brown to gray, massive
Plot A3	0-80	Silt loam, granular, light brown to reddish brown
	80-150	Silt loam, granular, reddish brown
	150-200	Silty clay loam, sub-angular blocky, brown with gray mottles
Plot B3	0-30	Silt loam, granular, light brown to brown
	30-100	Silty clay loam, platy, brown to dark brown
	100-180	Silty clay loam, dark brown to gray with red mottles, sub-angular blocky
2nd Order Pasture 3		
Plot 1L	0-35	Silt loam, granular, brown
	35-100	Silty clay, sub-angular blocky, brownish gray
Plot 4L	0-30	Silt loam, granular/platy, grayish brown
	30-90	Silty clay, sub-angular blocky, grayish brown
	90-110	Loam, massive, gray with red mottles
Plot 3R	0-25	Silt loam, granular, brownish gray
	25-105	Silty clay loam, weak sub-angular blocky, gray
	105-130	Loam, massive, gray with red mottles
2nd Order Riparian Forest 1		*Only sampled 2 of 3 plots*
Plot 1L	0-30	Silt loam, brown, sub-angular blocky
	30-110	Silty clay loam, sub-angular blocky, brown to gray
	110-140	Silty clay loam, angular blocky, brown to gray
Plot 2L	0-30	Silty clay loam, dark brown, granular to sub-angular blocky
	30-80	Clay loam, pea-sized and smaller gravel, Mn concretions, angular blocky, light brown
	80-200	Clay loam, angular blocky, light brown to gray with orange and red mottles-intense color and numerous
2nd Order Riparian Forest 2		
Plot A1	0-30	Silt loam, grayish brown, granular/platy

Site Name/ Plot Designation	Horizon depth (cm)	Description
	30-65	Silt loam, brownish gray, platy
	65-120	Silty clay, sub-angular blocky, gray
	120-180	Clay, gray with red mottles, massive
Plot A3	0-25	Silt loam, granular/platy, grayish brown
	25-70	Silty clay loam, platy, gray
	70-95	Silty clay, sub-angular blocky, gray
	95-135	Silty clay loam, massive, dark gray
Plot B2	0-25	Silt loam, brown, granular/platy
	25-110	Silty clay loam, brownish gray, sub-angular blocky
	110-170	Clay, gray with red mottles
2nd Riparian Forest 3		
Plot A1	0-35	Silt loam, brown, granular
	35-70	Silt loam, brown, sub-angular blocky
	70-90	Loam, gray with orange mottles, sub-angular blocky
Plot A3	0-70	Loam, reddish brown, platy
	70-95	Sandy loam, blocky, gray brown with orange mottles
Plot B2	0-35	Silt loam, brown, sub-angular blocky to platy
	35-75	Silt loam, brown, sub-angular blocky
	75-105	Loam, gray with orange mottles, sub-angular blocky
Plot B4	0-30	Silt loam, reddish brown, granular
	30-80	Silt loam, gray brown, granular to sub-angular blocky
	80-100	Loam, gray brown with orange mottles, sub-angular blocky
3rd Order Crop		
Plot L2	0-40	Silt loam, granular, light brown to brown
	40-150	Silty clay loam, sub-angular blocky, grayish brown
	150-220	Clay loam, sub-angular blocky, gray with strong red mottles
Plot R1	0-40	Silt loam, reddish brown to light brown, granular
	40-100	Silty clay loam, granular, reddish brown
	100-210	Silty clay loam, sub-angular blocky, gray to light brown with orange mottles, few Mn concretions

Site Name/ Plot Designation	Horizon depth (cm)	Description
Plot R2	0-70	Silt loam, granular, light brown to brown
	70-150	Silty clay loam, sub-angular blocky, brown
	150-210	Silty clay loam, reddish brown to gray with orange mottles, sub-angular blocky
3rd Order Forest 1		
Plot A1	0-30	Silty clay loam, reddish brown, granular
	30-75	Silt loam, gray/brown, sub-angular blocky
	75-105	Loam, gray with orange mottles, sub-angular blocky
	105-160	Loam, gray, sub-angular blocky
	160-260	Clay loam, dark gray with orange mottles, angular blocky
	260-290	Sandy clay loam, gray with numerous orange mottles, weak angular blocky
	290-315	Sandy loam, gray, weak sub-angular blocky
Plot A2	0-45	Loam, light brown, sub-angular blocky
	45-90	Silt loam, brown, sub-angular blocky
	90-175	Loam, dull gray to reddish brown, sub-angular blocky
	175-250	Silty clay loam, dark gray with reddish brown mottles, sub-angular blocky
	250-310	Silty clay loam, dark gray with reddish brown mottles, sun-angular blocky
	310-350	Silty clay loam, dark gray with reddish brown mottles, sub-angular blocky
Plot B1	0-25	Sandy clay loam, light brown, granular
	25-85	Silty clay loam, light brown, angular to sub-angular blocky
	85-120	Clay loam, light brown with brownish red mottles, weak angular blocky
	120-205	Clay loam, dark gray with red/orange mottles, strong angular blocky
	205-235	Clay loam, dark to light gray with red/orange mottles, angular blocky
3rd Order Forest 2		
Plot A1	0-25	Silt loam, granular, dark brown
	25-60	Silty clay loam, sub-angular blocky, brown
	60-95	Silty clay loam, light gray to gray, sub-angular blocky

Site Name/ Plot Designation	Horizon depth (cm)	Description
	95-160	Silty clay loam, strong sub-angular blocky, gray with orange-red mottles
Plot B1	0-20	Silt loam, granular, brown
	20-75	Silty clay loam, sub-angular blocky, light brown
	75-175	Silty clay loam, sub-angular blocky, brown to light gray
	175-250	Clay loam, weak blocky, dark gray to gray
Plot B3	0-70	Silty clay loam, brown, sub-angular blocky
	70-105	Silt loam, sub-angular blocky, gray with red mottles
3rd Order Forest 3		
Plot A1	0-35	Silty clay loam, sub-angular blocky, medium gray
	35-65	Silty clay loam, sub-angular blocky, light gray
	65-125	Loam, sub-angular blocky, light brown
Plot A3	0-30	Silt loam, brown, granular to sub-angular blocky
	30-100	Silt loam, brown gray, sub-angular blocky
Plot B1	0-30	Silt loam, sub-angular blocky, brown
	30-60	Silt loam, brownish gray, sub-angular blocky
	60-200	Silt loam, sub-angular blocky, gray
3rd Order Pasture 1		
Plot A1	0-15	Silt loam, granular, light brown
	15-145	Silty clay loam, sub-angular blocky, brown
	145-225	Silt loam, granular to sub-angular blocky, light brown to brown
	225-275	Silty clay loam, sub-angular blocky, reddish brown to gray with orange mottles, Mn concretions
Plot B3	0-20	Silt loam, light brown, sub-angular blocky
	20-50	Silt loam, sub-angular blocky, light brown to brown
	50-140	Silty clay loam, reddish brown to brown, weak sub-angular blocky
	140-160	Silty clay loam, weal platy, brown

Site Name/ Plot Designation	Horizon depth (cm)	Description
	160-200	Silty clay loam, angular blocky, brown to dark brown with gray and orange mottles, Mn concretions
Plot B6	0-15	Silt loam, granular, light brown to brown
	15-60	Silty clay loam, granular, reddish brown to brown
	60-90	Silty clay loam, brown to dark brown, sub-angular blocky
	90-160	Silty clay loam, light gray with orange mottles, weak platy
	160-210	Silty clay loam, sub-angular blocky, brown to dark brown
	210-230	Clay loam, sub-angular blocky, medium gray with orange mottles, Mn concretions
3rd Order Pasture 2		
Plot C1	0-50	Loam, light brown gray, sub-angular blocky
	50-140	Loam, medium brown gray, sub-angular blocky with sand lenses
	140-190	Clay loam, brown gray with gray mottles, massive
Plot C2	0-25	Silt loam, dark brownish gray with some red mottles, weak sub-angular blocky
	25-65	Silty clay loam, dark gray with red mottles, sub-angular blocky
	65-175	Clay, gray with red mottles, massive
Plot D1	0-20	Silt loam, brown gray, weak platy to sub-angular blocky
	20-120	Silty clay loam, brown gray, massive
Plot D3	0-25	Loam, dark brown gray, weak sub-angular blocky
	25-150	Loam, brown gray, massive
3rd Order Pasture 3		
Plot A3	0-25	Sandy loam, no structure
	25-41	Silt loam
	41-70	Clay loam
	70-115	Silty clay loam, reddish brown, very weak sub-angular blocky
	115->200 -below 200 bank saturated	Clay loam, grayish brown, sub-angular blocky
Plot B2	0-25	Silt loam

Site Name/ Plot Designation	Horizon depth (cm)	Description
	25-50	Sandy clay loam
	50-85	Clay loam, platy
	85-155	Silty clay loam
	155-185	Sandy clay loam, weak platy structure
	185->260	Highly mottled, gray and red mottles, sub-angular blocky
Plot B4	0-27	Silt loam
	27-100	Silty clay loam
	100-180	Silty clay loam
3rd Order Riparian Forest 1		
Plot A2	0-60	Silt loam, brown, sub-angular blocky
	60-120	Silty clay loam, angular blocky, light brown to brown
	120-180	Silt loam, gray-brown with orange-red mottles, massive
Plot A3	0-35	Silt loam, granular, brown
	35-125	Silt loam, light brown, sub-angular blocky
	125-240	Silt loam, platy to sub-angular blocky, light brown to gray
Plot B3	0-30	Silt loam, granular, brown-gray
	30-65	Silt loam, platy, dark gray
	65-115	Silty clay loam, sub-angular blocky, dark gray with red and brown mottles
	115-185	Silty clay loam, massive, gray with brown mottles
	185-345	Loam, massive, brown with gray and red mottles
3rd Order Riparian Forest 2		
Plot 2L	0-30	Silty clay loam, weak platy, brown to reddish brown
	30-90	Silty clay loam, weak sub-angular blocky, brown to reddish brown
	90-125	Silty clay loam, platy, reddish brown to gray with orange-red mottles
	125-195	Silty clay loam, sub-angular blocky, light gray to gray with orange-red mottles, Mn concretions
Plot 4L	0-35	Silt loam, granular, brown to gray
	35-100	Silty clay loam, sub-angular blocky, reddish brown to brown

Site Name/ Plot Designation	Horizon depth (cm)	Description
	100-190 Saturated below 150	Silty clay loam, reddish brown to gray, sub-angular
Plot 4R	0-25	Silty clay loam, light brown to brown, granular
	35-85	Silty clay loam, reddish brown to grayish brown, granular
	85-100	Silty clay loam, platy reddish brown to grayish
3rd Order Riparian Forest 3		
Plot A1	0-25	Loam, brown, granular
	25-105	Silty clay loam, brown, sub-angular blocky
	105-170	Silt loam, brown, sub-angular blocky
Plot A4	0-90	Silt loam, brown, granular to sub-angular blocky
	90-150	Silty clay loam, brown, sub-angular blocky
Plot B2	0-50	Silt loam, brown, granular
	50-145	Silty clay loam, brown to dark brown, sub-angular blocky
	145-190	Silt loam, brown to gray-brown, sub-angular blocky
Plot B4	0-55	Silty clay loam, brown, sub-angular blocky
	55-150	Silt loam, brown to dark brown, platy

APPENDIX B

Hypothetical Example of Streambank Erosion Calculation

Assumptions:

Site length= 100 m (bank length is 2x100= 200 m)

Bulk Density= 1500 kg m^{-3}

Total eroded length= 60 m

2 erosion pin plots, A and B, with dimensions: length= 6 m, height= 1.5 m

All pins begin with 10.2 cm exposed in the beginning of a measuring period.

At the end of the measuring period, the length of each pin is recorded as follows:

Plot A

0	6.6	20.2
11	10.2	missing

Plot B

totally eroded	19.2	4.2
30.7	17.5	9.2

Taking the difference between the current pin length and the previous pin length (10.2 cm) gives:

Plot A

-10.2†	-3.6	10
0.8	0	missing

Plot B

65‡	9	-6
20.5	7.3	-1

†Note: a negative pin length indicates deposition. ‡Assumed length for totally eroded pins.

The average pin length for each pin plot is then calculated:

Plot A

average pin length= -0.6 cm= -0.006 m

Plot B

average pin length= 15.8 cm= 0.158 m

This value is then multiplied by the area of the pin plot and the bulk density for the site:

Plot A

$-0.006 \text{ m} \times 9 \text{ m}^2 \times 1500 \text{ kg m}^{-3} = -81 \text{ kg}$

Plot B

$0.158 \text{ m} \times 9 \text{ m}^2 \times 1500 \text{ kg m}^{-3} = 2133 \text{ kg}$

which gives the mass of eroded sediment for each pin plot.

Next, the mass is divided by the length of the pin plot:

Plot A

$-81 \text{ kg} / 6 \text{ m} = -13.5 \text{ kg m}^{-1}$

Plot B

$2133 \text{ kg} / 6 \text{ m} = 355.5 \text{ kg m}^{-1}$

which gives the linear erosion rate on a pin plot basis.

Taking the average linear erosion rate from the pin plots and multiplying by the total eroded length of the site:

average site erosion= $171 \text{ kg m}^{-1} \times 60 \text{ m} = 10{,}260 \text{ kg}$

which is the total mass of eroded sediment for an entire site.

Finally, to calculate a linear erosion rate on a stream reach basis, the total mass of the eroded sediment was divided by the total bank length:

$10{,}260 \text{ kg} / 200 \text{ m} = 51.3 \text{ kg m}^{-1}$

which is the rate of eroded sediment per meter of stream bank.

APPENDIX C

Soil and Hydrologic Functions

Adsorption Functions

<u>Organic Matter (OM) Weight</u> (Eq. C.1)

$$WeightOM = -0.1450 + \frac{8.2901}{1 + e^{-\left(\frac{OM\% - 2.7500}{0.6827}\right)}}$$

Where %OM is the representative organic matter content on a percent basis (om_r) from SSURGO.

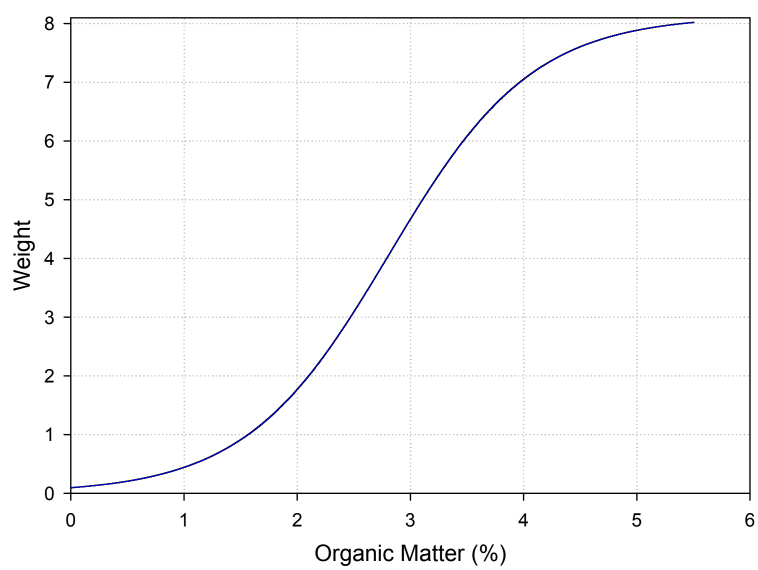

Figure C.1 Sigmoidal Organic Matter Weight Function

<u>Clay Content Weight</u>

Solution Runoff (SRO)/Leaching Clay Weight (Eq. C.2)

$$WeightClay = \%Clay * (0.0329 - 0.0072 * \%OM + 0.0041 * \%OM^2 - 0.0039 * \%OM^3)$$

ParticleAdsorbed Runoff (ARO) Clay Weight (Eq. C.3)

$$WeightClay = \%Clay * -(0.0329 - 0.0072 * \%OM + 0.0041 * \%OM^2 - 0.0039 * \%OM^3)$$

Where %clay is the representative clay content value (claytotal_r) from SSURGO, and %OM was previously defined.

Hydrologic Functions

Depth to Free Water (Free H_2O) Penalty

If the depth to the water table is less than the depth to the minimum K_{sat} layer, or if the depth to the minimum K_{sat} layer is less than the depth to the water table and there is a restrictive layer (Bti>35) which indicates a perched water table, then the following penalty is applied to the leaching pathway:

$$FreeH_2OPenalty\text{-}= (0.0167 * \text{depth to free } H_2O)-3 \qquad \text{(Eq. C.4)}$$

where depth to free H_2O is the minimum depth to the water table in April-June parameter (wtdepaprjunmin) from SSURGO.

Flooding Frequency Weight

Based on the flooding frequency maximum parameter (flodfreqmax) from SSURGO. Uses classed data.

Flooding Frequency Class	Weight
None	0
Rare	-1
Occasional	-2
Frequent	-3

Index of Surface Runoff (ISRO) Weight

Runoff (R) ISRO (Eq. C.5)

$$WeightRISRO = 2 + \left(1.037 * \exp^{-0.7934*Slope} + 2.6819 * \exp^{-0.0617*Slope} + 0.435 * \ln(\min K_{sat})\right)$$

Where slope is the representative slope gradient value (slope_r) from SSURGO, K_{sat} is the representative saturated hydraulic conductivity (ksat_r) from SSURGO, and min K_{sar} is the minimum Ksat occurring in the top 1m of the soil profile.

Leaching (L) ISRO

$$Weight_{LISRO} = 5 - Weight_{RISRO} \qquad \text{(Eq. C.6)}$$

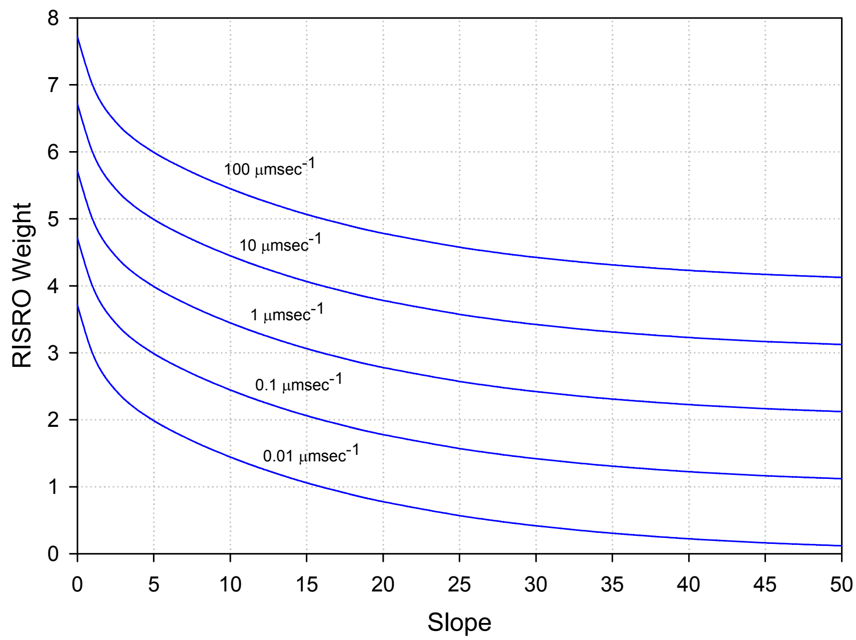

Figure C.2 Runoff Index to Surface Runoff (RISRO) Weight at various K_{sat} values as a function of slope.

Available Water Holding Capacity (AWC) Penalty

If the map unit met the restrictive layer criteria (Bti>35) then AWC above the minimum K_{sat} layer was calculated and the following function was used to assess a penalty:

$$AWCPenalty = (AWC * (2/5)) - 10 \qquad \text{(Eq. C.7)}$$

where AWC is the available water storage (aws025wta, aws050wta, aws0100wta) SSURGO input parameter.

Severe Runoff Penalty

A penalty applied to soils where Bti>35.

$$SevereRunoffPenalty = 3 + LSWeight \quad \text{(Eq. C.8)}$$

Landscape and Erodibility Functions

Erodibility Weight (Eq. C.9)

$$WeightErod = 0.1613 - 8.065 * K_{wf}$$

Where K_{wf} is the K_w erodibility constant (Kwfact) from SSURGO.

LS Weight (Eq. C.10)

$$LSWeight = \left(-0.1300 + 0.1218 * \exp^{0.1149*slope}\right) * LS^{0.5} + 0.1127 + \frac{-0.0072 * \left(\exp^{0.1127*slope} - 1\right)}{0.1127}$$

Where slope was previously defined, and LS is the representative USLE slope length (slopelenusle_r) from SSURGO.

Inverse LS Weight

$$InverseLSWeight = LSWeight - 5.5 \quad \text{(Eq. C.11)}$$

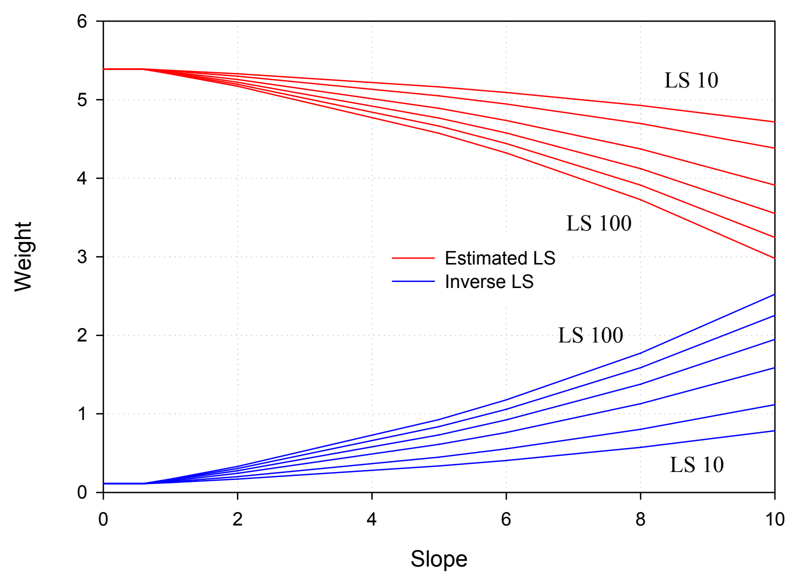

Figure C.3 Estimated LS and Inverse LS Weight varied over Slope and LS.

Dissipation Functions

Biotic Degradation Weight (Eq. C.12)

$$WeightBioDeg = 3.0 * e^{-0.5\left(\frac{pH-6.8}{0.69}\right)^2} * \left(0.2003 + \frac{0.8121}{1+\exp^{-\left(\frac{OM\%-2.8095}{0.6321}\right)}}\right) * \left(1-\exp^{\frac{-0.6931*t}{t1/2}}\right)$$

Where pH is the representative pH value (pH1to1H20_r) from SSURGO, %OM was previously defined, t is time in days, and $t_{1/2}$ is the biodegradation half-life in days.

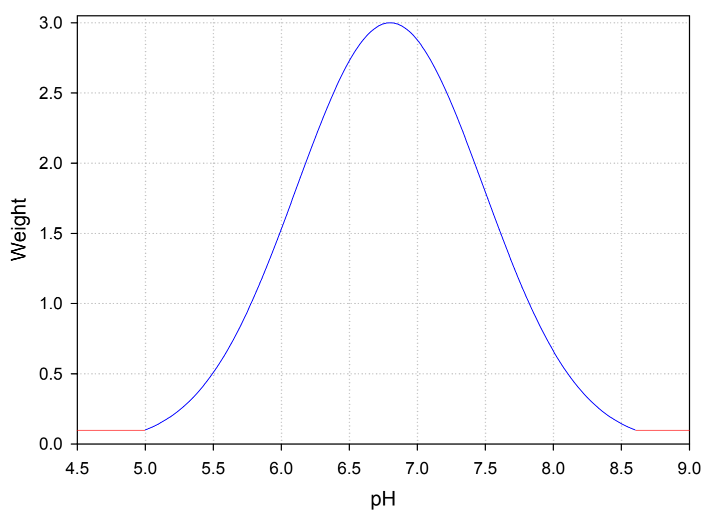

Figure C.4 pH component of biotic degradation weight function. Biotic degradation peaks at pH 6.8

Adsorption Weight

SRO/Leaching (Eq. C.13)

$$WeightAdsorp = 2.5 * (OMwt + Claywt) * \left(1 + e^{(-0.6931*Day_n)}\right) * \left(\frac{1.0010}{1+\left(\frac{Koc}{100}\right)^{-2.0424}}\right)$$

ARO (Eq. C.14)

$$WeightAdsorp = 2.5 * -(OMwt + Claywt) * \left(1 - e^{(-1.6094*t)}\right) * \left(1 - \left(\frac{1}{1+\left(\frac{Koc}{100}\right)^{2.048}}\right)\right)$$

Where *OMwt* and *Claywt* are the organic matter weight and the clay weight as previously defined, *t* is time in days, and K_{oc} is the octanol-water partitioning coefficient calculated using K_d.

<u>Hydrolysis Weight</u>

$$WeightHydrolysis = \left[4.44 + \frac{-4.44}{1 + \left(\frac{pH - 7.0}{1.38}\right)^2}\right] * \left(1 - \exp^{(-0.794 * t)}\right) \qquad \text{(Eq. C.15)}$$

Where all variables were previously defined.

<u>Volatilization Weight</u>

$$WeightVolatil = (0.0724 \ln vp + 1.3334)\left(1 - \exp^{(-1.1 * t)}\right) \qquad \text{(Eq. C.16)}$$

Where *vp* is the herbicide vapor pressure in mmHg at 25ºC, and *t* was previously defined.