Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: A review

Viktor Polyakov, Ali Fares, and Micah H. Ryder

Abstract: Numerous studies have shown the effectiveness of riparian buffers in reducing sediment, pathogen, and nutrient loads into surface and groundwater in agricultural catchments. Reported retention rates of sediment, N, and P were as high as 97%, 85%, and 84%, respectively. Often, however, riparian buffers fail to perform their protective functions due to low adaptability of their designs to local settings. This is caused by our inadequate understanding of the conditions under which riparian buffers perform the best at field scale. Therefore, a precision oriented approach based on thorough analysis of spatially variable characteristics of landscape has to be undertaken in riparian buffer construction. Such an approach has a potential to improve the protective qualities and the economic viability of the riparian buffers. This paper gives an overview of the current level of research on riparian buffers and discusses the importance of spatial variability of local conditions on their performance. It presents the approaches for precision buffer design and its practical implementation and highlights the directions for future development of precision conservation.

Key words: riparian buffer, vegetative filter, water quality, precision conservation.

Résumé : De nombreuses études ont démontré l'efficacité des tampons riverains pour réduire les problèmes de sédimentation, de maladies et d'eutrophisation dans les eaux de surface et souterraines, par captage en milieux agricoles. On rapporte des taux de rétention des sédiments et du N et P aussi élevés que 96 %, 85 % et 84 %, respectivement. Cependant, il arrive souvent que les tampons riverains n'exercent pas leurs fonctions protectrices, vu la faible adaptabilité de leurs concepts aux conditions locales. Ceci provient d'une compréhension inadéquate des conditions sous lesquelles ces tapons riverains fonctionnent le mieux, à l'échelle du terrain. Conséquemment, il faut utiliser une approche orientée sur la précision, basée sur des analyses complètes des caractéristiques des variables du paysage, avant d'entreprendre la construction de tels tampons riverains. Cette approche pourrait améliorer les qualités de protection et économiques des tampons riverains. Les auteurs présentent une revue de l'intensité actuelle de la recherche sur les tampons riverains, et discutent l'importance de la variabilité spatiale et des conditions locales qui conditionnent leurs performances. On présente les approches nécessaires pour concevoir et mettre en

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place des tampons de précision, et on suggère les directions à prendre pour continuer le développement de la conservation de précision.

Mots clés : tampon riverain, filtre végétal, qualité de l'eau, conservation de précision.

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Introduction

Modern agricultural practices have contributed significantly to nonpoint source pollution. Pollutants leaving agricultural catchments in surface and groundwater have caused severe declines in the water quality of associated streams, rivers, and lakes. Farming and ranching have allowed an excess of nutrients, sediment, and chemicals to runoff or to leach from farms and pastures into nearby streams and groundwater (Vought et al. 1995). To control and mitigate the impact of modern agriculture on the surrounding environment, conservation management practices have been developed and studied. Among these practices are riparian buffers that began to be systematically established in the United States during the 1960s (Calhoun 1988). Riparian buffer is defined as an area of permanent vegetation adjacent to a water body or wetland managed for the purpose of removing pollutants from runoff or groundwater (Muscutt et al. 1993). In the scientific literature, this term is often used interchangeably with the terms vegetative filters or vegetative buffer. In this paper, we preserved the original terminology when referring to published studies.

Riparian buffers influence the aquatic communities by altering incoming surface and groundwater (Osborne and Kovacic 1993). A wide variety of ecological and economical values provided by riparian buffers include, but are not limited to: trapping sediment, nutrient, and pesticides in runoff; stabilizing stream banks; storing storm water; providing habitat for variety of organisms; and serving recreational and aesthetical purposes (Gregory et al. 1991). Buffers can be used in the form of filter strips along drainage ditches, bordering concentrated flow pathways or on steep terrain within agricultural fields. Buffers also represent an important management component of grazing practices. The major mechanisms that enable riparian buffers to reduce incoming soluble pollutants in runoff are physical retention, plant uptake, dilution, and chemical transformation (Osborne and Kovacic 1993).

With recent advances in precision agriculture, precision conservation has started to draw more attention (Wallace 1994). Precision conservation is defined as a set of technologies directed to implement conservation practices that take into account spatial and temporal variability across natural or agricultural systems (Berry et al. 2003). These authors argued that the precision approach to conservation is vital to maintaining agricultural sustainability in the future (Berry et al. 2003). Although studies demonstrating riparian buffer effectiveness have led to widespread use in management (Baird et al. 2000), more work is needed to optimize buffer size and placement (Lee et al. 2004).

Riparian buffers can be examined in three dimensions (Lee et al. 2004): longitudinal (along the stream or protected area), vertical (the structure of root zone and canopy), and transverse (perpendicular to the stream). Until now, most of the research on buffer performance has been focused on the transverse dimension with the aim to maximize buffer ability to intercept pollutants, while the aspects of longitudinal dimension received less attention (Weller et al. 1998). Parameters such as precipitation, flow convergence, infiltration rate, water storage capacity, slope, and vegetative cover may vary considerably along a stream and will have great effect on riparian buffer performance (Herron and Hairsine 1998).

There are two approaches for designing riparian buffers. The first one is the fixed width approach, when a minimum width is defined according to regional conditions and government agencies recommendations (Lee et al. 2004). This approach is easily implemented since it requires minimum planning. However, it is based on either the empirical relationships between buffer width and desired percent of pollutant reduction (Dukes et al. 2002; Hook 2003) or could be even purely arbitrary (Phillips 1989). The second approach is a precision riparian buffer or variable buffer. Precision riparian buffer is a spatially variable riparian buffer designed to achieve specific water conservation goals of reduction of nonpoint source pollutants via optimizing its characteristics with respect to runoff contributing area, slope, soil type, land use, and climate in that particular location. The meaning of precision buffer becomes apparent when one considers it as an integrated part of the watershed rather than a stand-alone protective area. According to Lee et al. (2004), a critical objective of riparian management is to address the spatial variations in physical processes taking place during water flow through the buffer. These processes were reflected in a number of models (Abu-Zreig et al. 2001; Flanagan et al. 1989; Munoz-Carpena et al. 1999).

Although it is often difficult to implement precision riparian buffers in practice due to substantial input data requirements and need for trained personnel, the use of GPS (Global Positioning System) and GIS (Geographic Information System) techniques could aid in the solution (Baker et al. 2001; Berry et al. 2003; Xiang 1993). Transition to precision conservation is also necessary due to apparent economical benefits of such approaches (Sparovek et al. 2002; Stonehouse 1999). Level of protection that a conservation practice offers must be balanced with the cost of its implementation, thus creating an incentive for the end users. The economical viability analysis may be part of conservation planning, especially in areas where land is expensive or scarce.

The objectives of this review are (1) to give an overview of the current level of research on riparian buffers and the importance of spatial variability of local conditions on their performance, (2) discuss the approaches for precision buffer design and its practical implementation, and (3) highlight the need for future development of precision conservation.

Variability of protective properties of riparian buffers

Sediment retention

Riparian buffers have been shown to significantly reduce sediment loading in surface runoff from agricultural catchments with various pollutant sources (Arora et al. 2003; Srivastava et al. 1996; Young et al. 1980). Buffers remove sediment from the overland flow by decreasing its velocity and allowing particles to settle. Electrostatic forces acting on the surface of vegetation may also retain fine soil particles (Stocking 1994). Increased water infiltration into the soil profile within buffer zones aids in sediment interception by decreasing the amount of runoff. The primary factors that effect sediment entrapment rate are vegetation density and spacing, initial soil water content, saturated hydraulic conductivity, and sediment characteristics (particle size, fall velocity, and aggregate density) (Munoz-Carpena et al. 1999).

Cooper et al. (1992) estimated that 90% of the sediment leaving fields in a North Carolina coastal plain watershed remained in the wooded riparian zone. Sheridan et al. (1999), when studying the impact of forest management practices within the riparian zone, reported sediment trapping efficiencies of 67%–90% across three different management schemes (clearcut, thinned, and untouched). Sediment trapping efficiency can be defined as the capacity of a buffer to retain a fraction of sediment from the incoming runoff. Trapping efficiency is affected by various factors such as buffer width, vegetation type, slope gradient and length, flow convergence, source area, and pollutant concentration (Herron and Hairsine 1998; Hook 2003).

Research has shown that the first 3–6 m of a riparian buffer plays a dominant role in sediment removal (Daniels and Gilliam 1996; Robinson et al. 1996). For example, Gharabaghi et al. (2000) reported that the first 2.5 m of grass buffer retained 50% of sediments and an additional 25%–45% was retained in the following 2.5 m. In a study comparing the performance of bromegrass filter strips on various slopes, Robinson et al. (1996) found that sediment concentrations from the 7% and 13% slope plots decreased by 70%–80% within the first 3 m of the buffer. Dillaha et al. (1989) and Magette et al. (1989) reported sediment removal efficiencies of 70%–80% for 4.6 m and 84%–91% for 9.1 m wide grass filter strips. In an investigation of the mitigation potential of grass filters of manure runoff using simulated rainfall, Lim et al. (1998) reported increasing sediment removal of 70%, 90%, and 98% for 6, 12, and 18 m wide grass buffer strips, respectively. Despite this ample research, there is still a lack of a comprehensive relationship between buffer width and its sediment removal potential. Case studies are still the primary source of information for buffer width comparisons and planning.

Attempts have been made to use the ratio of sediment contributing area to the area of the buffer (area–buffer ratio) as a predictor for trapping efficiency. For example, Tingle et al. (1998), studying the effectiveness of fescue vegetative filter strips, demonstrated a direct relationship between area–buffer ratio ranging between 0.02 and 0.18 and sediment trapping efficiency ranging between 85% and 96%. This approach performed well when used for the same watershed or adjacent watersheds with similar conditions. However, the same approach failed when used by Dosskey et al. (2002) to analyze 20 field studies from various climatic and soil conditions. Analysis of several case studies with similar area–buffer ratio (0.02–0.03) and on similar slopes (2%–4%) yielded a range of trapping efficiencies between 45% (Daniels and Gilliam 1996) and 85% (Tingle et al. 1998). Apparently, site-specific differences played an important role in trapping efficiency. In addition, Dosskey et al. (2002) found that the VFSMOD model (Abu-Zreig et al. 2001), which utilizes area–buffer ratio in its routine, severely underestimated trapping efficiency when compared to field data. Barfield et al. (1998) attributed variation in trapping efficiency to variation in runoff loading, arguing that high flow tends to inundate the buffer, resulting in reduced trapping efficiency irrespective of buffer size, while low input loads result in very high trapping efficiency. In hilly or incised terrain, flow pathways within the field may allow runoff to concentrate prior to entering the buffer causing inundation and making buffers ineffective (Dillaha et al. 1989).

Sediment trapping potential of riparian buffers is also related to the sediment particle size; thus, the effectiveness of buffers is reduced as sediment size decreases (Lee et al. 2000). These authors concluded that more than 95% of the aggregates larger than 40 μm in diameter could be captured in the first 5 m of the buffer. This suggests that trapping efficiency depends on soil type from which the sediment was produced and rainfall energy as a primary force of aggregate dispersion. Although it is possible to relate riparian buffer efficiency to a single parameter (such as width) in a specific situation, spatial variability of field conditions would require an integrated solution, where a subset of parameters is used (Berry et al. 2003).

Phosphorus

Phosphorus (P) is one of the primary surface runoff pollutants from agricultural fields and watersheds, and the one that contributes significantly to eutrophication and other environmental and health problems. Phosphorus from fertilizers and manure is commonly adsorbed by soil particles and organic matter in the runoff entering filter strips. Therefore, its removal from runoff is closely associated with the retention of suspended sediment. In addition, P can be removed by becoming occluded in soil organic matter, through mineral deposition, precipitation, or plant uptake. The movement of soluble P within buffer strips depends largely on plant uptake potentials, soil chemical and physical properties, and subsurface flow paths (Abu-Zreig et al. 2003).

Phosphorus removal rates from surface runoff leaving riparian buffers can be as high as 93% (Lim et al. 1998), however the majority of studies conducted under a variety of vegetation types and widths of buffers reported rates of 60%–90% (Line et al. 2000; Young et al. 1980). The removal mechanisms responsible depend on the form of P entering the buffer. The soluble form is likely to be infiltrated and subsequently consumed by the plants, diluted and (or) transformed. However, riparian buffers may become saturated and allow large concentrations of soluble P to bypass them. Daniels and Gilliam (1996) found that although on average 50% of total P (TP) load was retained by a fescue filter strips buffer, 80% of soluble P frequently passed through. In a study by McKergow et al. (2003), a riparian buffer had little effect on total P concentrations or loads in runoff over a 10 year period after buffer installation. However, investigators found a significant change in P form, from sediment-bound to filterable reactive P as runoff passed through the filter. These results indicated that the buffers trapped sediment bound P while allowing soluble P to bypass them. Buffer saturation from heavy storms or large nutrient input concentrations may impact soluble P removal efficiencies. Soil texture, physical characteristics including hydraulic conductivities, and subsurface flow paths can potentially impact the rate of movement of soluble nutrients through buffer strips. In a study of six Wisconsin watersheds with contrasting riparian buffer attributes, Reed and Carpenter (2002) found that the variability in P yield in the adjacent stream closely correlated with the buffer size, percent wetland cover of the watershed, riparian continuity, and stream sinuosity. These authors particularly emphasized such spatially variable characteristics as continuity and uniformity of riparian buffers as moderators of P flow from upland agricultural lands into streams.

Nitrogen

Nitrogen (N) removal by riparian buffers depends on complex interaction of spatially variable components of a riparian system such as plants, microbial communities, soil properties, and hydrological site attributes. Nitrogen retention relies on three major mechanisms: plant uptake, microbial immobilization, and bacterial denitrification (Martin et al. 1999).

In an early study examining filter effects on feedlot runoff, Young et al. (1980) reported mean total N reductions of 83% and nitrate reductions of similar rates for sorghum and oat cropped filters in Central Minnesota. Kuusemets et al. (2001) reported reduction in total N of 78%–84% in a complex riparian zone consisting of wet meadow and grey alder (*Alnus incana*) stand. Pinay et al. (1993) found that a 30 m forested riparian buffer was capable of removing all nitrate from the incoming groundwater. However, the ability of riparian zones to mitigate nitrogen loading is often questionable. For example, in a study by Dillaha et al. (1989), soluble N levels in runoff leaving buffer plots showed an increase over the levels in runoff entering the plots. McKergow et al. (2003) examined total N levels entering stream waters in Western Australia before and after riparian buffers were established. During the 10 year study (6 before and 4 after buffer installation) annual total N exports from the catchment remained relatively constant, between 1 and 3 kg ha^{-1} a^{-1}.

Such controversial results can be explained when one considers riparian buffer in three-dimensional perspective in relation to its N retention potential (Martin et al. 1999; Pinay et al. 1993). Indeed, most studies investigating groundwater nitrogen use a grid sampling approach, neglecting the vertical variability of hydrological and biological processes in soil. Often major pathways of nitrogen transport lay in deeper horizons. Agricultural nutrients can be transported to a depth of up to 16 m below the surface (Geyer et al. 1992). Measurements made in surface water or shallow subsurface flow may not reflect the greater picture of nitrogen dynamics in a riparian zone.

Researchers have paid great attention to the discharge of nitrates into water bodies (Haycock and Pinay 1993; Lowrance et al. 1997; Willems et al. 1997). There is a consensus that riparian zones are effective in removing nitrates from shallow subsurface water (Hill 1996). Considerable environmental effects such as eutrophication in lakes (Zakova et al. 1993) and severe health risks (Sotomayor and Rice 1996) stimulate this interest. Nitrogen losses in groundwater associated with denitrification are commonly named as the dominant removal mechanism, but supporting experimental data are scarce. Spruill (2000), studying a small basin, found that out of the 95% reduction of N in groundwater, caused by a riparian buffer, 65%–70% was due to denitrification. Denitrification in soil primarily occurs within carbon rich regions (Martin et al. 1999) or is localized around sites with high soil organic carbon content (Addy et al. 1999). This causes variability of the process in the macro and micro scales. In addition to carbon, spatial variability of denitrification can be impacted by groundwater flow rate, incoming nitrate concentrations, and temperature. Willems et al. (1997) described this relationship using a linear combination of the previously discussed parameters.

There are conflicting reports on the ability of various plants to remove nitrate from runoff. Osborne and Kovacic (1993) demonstrated that, on an annual basis, grass riparian buffers were less effective in removing nitrate than forested buffers. However, this conclusion was only applied to shallow groundwater. Schnabel et al. (1997), comparing grass and wooded riparian sites, observed denitrification rates three times greater on grass sites than those on wooded ones. Corley et al. (1999) examining the effect of type and height of grassy riparian vegetation, found that there was no consistent difference among the species in the removal of N and P. In a similar study using tree species, Borin and Bigon (2002) found that tree size showed no significant effect on the reduction of the nitrate loading into streams. Irrespective of the chemical and biological mechanisms involved, it appears that the interaction of subsurface hydrology and active rooting depth could be a decisive factor in the denitrification efficiency. Hence, the choice of species for precision buffers must reflect the nature of subsurface flow in a particular location.

Geyer et al. (1992), in a watershed experiment in Washington, found that higher denitrification potential existed in locations with a shallow groundwater table and impending stratigraphic layers. These areas were found at the bottom of slopes, thus indicating that there was a topographical control over nitrate removal. Hill (1996) suggested that riparian zones are least effective where groundwater has limited interaction with vegetation, i.e., nutrient transport occurs over the surface or through deep soil horizons. Gold et al. (2001), examining the variability of groundwater nitrate removal in riparian zones in the Northeastern United States, attributed it to site characteristics, that are readily available from soil survey and topographical maps. Gold et al. (2001) argued that spatially variable attributes, such as hydric soil status, geomorphology, and landscape controls, must be translated into data bases and used for riparian buffer planning. Targeting high-value riparian locations and tailoring buffer design accordingly can significantly improve nitrogen interception.

Landscape attributes controlling buffer potential

Relief characteristics

Topographical factors affecting the efficiency of riparian buffers must be incorporated into integrated watershed conservation schemes; however, the landscape units controlling storm runoff generation are poorly understood (McGlynn and McDonnell 2003). Riparian areas differ from other topographical elements by large contributing areas draining into them, relatively shallow groundwater table, and relatively low slope gradients (Fried et al. 2000). Riparian buffers remove contaminants most effectively if the incoming flow is not canalized but distributed across a wide area.

There are several topographic characteristics of landscape pertaining to riparian buffers. Convergence factor is the ratio of the active area of the riparian buffer to its total area (Herron and Hairsine 1998), whereas the active area is the portion of buffer through which runoff flow from the upslope occurs. Consequently, low convergence values would represent dissected relief, gully heads, and steep valley floors, where values close to one refer to planar conditions. Convergent areas have the highest hydrologic loading. Other parameters defining convergence are specific area and slope index. Specific area is the ratio of the watershed area divided by the length of the riparian zone receiving runoff. Dividing specific area by slope gradient yields slope index.

Terrain analysis can be successfully used to predict surface and groundwater flow, especially on a large scale, where applications of physically based models are too complex. For example, in TOPMODEL (Beven 1997), topographic index distribution is derived from digital elevation data. Other terrain models use digital elevation data to estimate runoff (Band 1989), identify areas of soil saturation (Oloughlin 1981), and define water table depths in catchments.

The distribution of runoff among the streams of different orders requires special attention. Leopold et al. (1992) found that approximately 20% of runoff enters directly into first order streams, while the remaining 80% enters the drainage system through second and third order tributaries. A similar situation was described by Wondzell and Swanson (1996) in a 6400-ha watershed in Oregon, where a sharp decrease in relative direct drainage area with increase in stream order was observed. McGlynn and Seibert (2003) studied a 280-ha watershed in New Zealand and found that runoff from 85% of its total area was directed through only 28% of the riparian buffers along catchment streams.

The slope inside the buffer zone was found to be a good predictor of riparian buffer trapping efficiency (Jin and Romkens 2001) with these two parameters being inversely related. However, the same authors found that, at slopes greater than 6%, filter strips failed to retain sediment. Abu-Zreig (2001), when validating the VFSMOD model using a range of controlling factors, found that buffer slope had a significant effect on water flow and sediment transport.

Infiltration

Water infiltration in a watershed determines the excess precipitation available for runoff. Within riparian buffers, infiltration controls the decrease in incoming runoff velocity, and as a result effects sediment retention and the downward movement of soluble pollutants into the subsurface. Bharati et al. (2002), while comparing a riparian buffer, cultivated fields and grazed pasture on a loamy Midwestern soil, observed a five-fold increase in infiltration rate within the riparian buffer over cultivated fields and grazed pasture within 6 years after buffer construction. Similar results were obtained by Lee et al. (2000) in a study involving the establishment of vegetative buffers composed of various combinations of grasses, shrubs, and trees. Cooper et al. (1995) in an experiment near a stream in Taupo, New Zealand, compared soils within a converted grass buffer, a native shrub, and a grazed pasture. The grassed buffer had extremely high hydraulic conductivities in comparison to the grazed pasture, indicating a higher infiltration capacity.

The importance of accounting for infiltration rates in riparian buffer design has been recognized as the only mechanism of pollutant removal from runoff (Misra et al. 1996). Infiltration rates were used in modeling (Srivastava et al. 1998) to validate an event-based nutrient transport prediction tool. In addition, Bharati et al. (2002) employed an infiltration index in calculations to establish multi-species buffers in the Midwestern region capitalizing on the fact that infiltration was five times greater under buffers than under cultivated field and pasture. Herron and Hairsine (1998), when assessing riparian zones in Australia, incorporated infiltration rate into a set of equations relating buffer effectiveness to various hydrological parameters. They expressed buffer width as a proportion of slope length directly linking it to the ability of soil to infiltrate runoff. Infiltration rates are often highly variable in riparian areas due to combined effects of vegetation, soil properties, management, and topography (Herron and Hairsine 1998). Thus, spatial variability of infiltration rates needs to be properly addressed in precision buffering practice by incorporating this parameter into various landscape indexes used for buffer delineation.

Subsurface flow and storage capacity

Nutrient and pollutant movement in the subsurface of vegetative buffers is dependent on the hydraulic characteristics of the underlying soil. Therefore, the effectiveness of a buffer may be subverted if subsurface flow occurs below the rooting zone, in areas of preferential flow, or in soils with rapid infiltration. Soil layers may direct, impede, or retard the movement of water through the buffer. Groundwater flow paths will determine the pollutant residence time, any interactions within the rooting zone, exposure to surface horizons rich in organic carbon and microbial activity, and subsurface leaching.

Water flow dynamics in riparian buffer systems are highly variable. The presence of soil layers with different hydraulic conductivities may dictate water flow in both the horizontal and vertical directions. McCarty and Angier (2001) studied preferential flow pathways in riparian wetlands and found that continuous layers of high conductivity alluvial material created strong anisotropic soil structures that affected water movement and associated denitrification rates. They concluded that strong spatial variability in hydraulic conductivities would limit the ability of a buffer to mitigate incoming soluble agricultural pollutants. Preferential flow paths allowed nutrient plumes to bypass the root zone of a riparian buffer in a study conducted on the Georgian coastal plain (Vellidis et al. 2001). The preferential flow paths were determined to be due to old drainage ditches that ran through the riparian area and were later filled-in prior to the buffer installation. Preferential flow pathways in conjuncture with a strong spatial distribution of biological activities, such as denitrification, might limit the effectiveness of riparian wetlands in mitigating soluble agricultural pollutants.

These findings show the importance that subsurface and near stream hydrology can have on nutrient and pollutant transport and subsequent removal. To ensure buffer effectiveness, site selection must take into account past land disturbances, alluvial deposition, restricting soil layers, preferential flow paths or any other feature that control horizontal and lateral subsurface movement, and the residence times of water and solutes.

Storage capacity of soil in the riparian zone is an important factor controlling how much runoff can be intercepted during a single rainfall event. The depth of soil, as well its porosity, has the major influence on storage capacity (Band et al. 1993; Herron and Hairsine 1998). Inundation and overflow of the riparian buffer may occur if the storage capacity of its soil is limited either due to antecedent moisture or a shallow water table. Potential storage areas within watersheds are often localized and include valley floors and alluvial fans (Herron and Wilson 2001). Successful attempts to include a soil water storage limiting equation into assessments of riparian buffer effectiveness have been made (Herron and Hairsine 1998). To maximize storage capacity, efforts to reduce antecedent moisture through increased evapotranspiration seems to be a viable strategy. This would require careful selection of riparian species linked with the use of precision agriculture techniques.

Approaches to precision riparian buffer delineation

The advances in precision agriculture technology (Wallace 1994) highlight the necessity for similar approaches to be undertaken in watershed conservation. Berry et al. (2003) defined precision conservation as a set of techniques and procedures directed to implement conservation management practices that take into account spatial and temporal variability across landscape. The need to account for spatial variability of soil erosion potential in conservation has been widely recognized (Desmet and Govers 1995; Le Bissonnais et al. 2002; Weller et al. 1998). By combining hydrological modeling, erosion prediction technologies, remote sensing, and geographic information systems, precision conservation can have a key impact on future soil and water conservation and global environmental sustainability (Berry et al. 2003).

Today, it is common to design buffers of uniform width around a protected area (Lowrance et al. 2001). However, flow concentration, which often occurs, reduces the efficiency of a uniform buffer (Dosskey et al. 2002) resulting in some areas along the stream being under protected and other areas overprotected. A wide range of reported buffer efficiencies to remove sediment (Dillaha et al. 1989) or nutrient (Osborne and Kovacic 1993) makes it difficult to devise a simple practical recommendation for establishing a riparian buffer that would perform well in a wide range of conditions. Weller et al. (1998) argued that riparian buffers designed with constant width in many cases will probably not meet management goals. The authors modeled buffer performance on four different landscapes using a simple first-order transfer function on a gridded area, where every cell transmits a fixed fraction of material it receives and all buffer cells have identical retention capabilities. The authors concluded that variable width buffers were more efficient.

Buffer designs are generally based on the assumption that riparian areas receive runoff in a uniform sheet flow under which the maximum buffer efficiency is observed. However, a survey of four farms in Nebraska (Dosskey et al. 2002) showed that only 9%–18% of the total buffer area actually contacted runoff water. These authors estimated that, although the area of riparian buffers could potentially remove 41%–99% of sediment from runoff under uniform flow, the actual removal rate was 15%–43%. Tomer et al. (2003), when analyzing a 20 000-ha Iowa watershed with undulating terrain using a 30 m elevation grid, found that 23% of riparian zone cells did not receive any runoff during rainfall, 57% had contributing area of less than 0.4 ha, and 6% received runoff from more than 10 ha. While targeting critical areas to apply conservation management is the key to achieving water quality improvement (Maas et al. 1985; Tomer et al. 2003), there is a lack of quantitative methods, which enable evaluation of field runoff patterns and their impacts on buffer effectiveness (Dosskey et al. 2002).

One of the early attempts to optimize conservation practices was made by Maas et al. (1985) who established a number of qualitative criteria for selecting critical areas for nonpoint source pollution control and combining them into land resource and water resource perspective. The specific criteria were type of water resources impairment, erosion rates, manure sources, fertilizer rates and timing, pathogen sources magnitude, distance to watercourse, distance to impaired water resource, and present conservation status.

Variable width buffers fit well into a broader concept of precision conservation management zones formulated by Berry et al. (2003). This concept relies on differentiation between gross buffer area and effective area. The latter being the portion of a buffer that actually contacts field runoff. Dosskey et al. (2002), in a case study on four Nebraska farms, using the effective area approach estimated that optimizing buffer placement to the most critical areas may improve sediment retention three to four-fold.

The use of various topographic indexes has proved to be a useful tool in precision buffer delineation. Bren (1998) proposed to use area index, a proportion of upslope contributing area (*A*) to buffer length (*l*) along a section of a stream, to determine optimum buffer width on a 6600-ha watershed in Victoria, Australia. The area index is expressed as

[1] $$a = A/l$$

The author, assuming that water flows orthogonally to contour lines, identified elementary runoff cells and used a simple algorithm to calculate the area ratios. It was estimated that in order to include 10% of the watershed in a uniform buffer, a 16.5 m wide strip was required. However, when area index approach was used the buffer zone varied between 5 and 200 m. In such cases wider buffers were located near stream sources, otherwise under-protected, while narrow buffers were placed in divergent areas near stream channels.

The surface flow is driven by the effective hydraulic gradient equal to the surface slope (*a*), which is reflected in slope index (*s*) defined as

[2] $$s = a/\tan\alpha$$

The use of area and slope indexes to delineate protective buffers resulted in considerable buffer width variability with some stream sections receiving no buffering because of a lack of inflow, while gully and stream head buffers were becoming extremely wide (Bren 2000). In some cases protective buffers had no perennial stream associated with them, which takes buffer function beyond the generally excepted domain. For example, an area of convergence under the described approach may require a discrete zone of protection spatially detached from the rest of buffer area (Bren 2000). Despite a number of simplifications, such as steady state solution to a dynamic problem of overland flow and an assumption of surface homogeneity, the use of area and slope indexes appears to be a rigorous approach.

The ideas used by Bren (1998) were further developed by Tomer et al. (2003) to create planning maps at a watershed scale and identify sites for wetland and riparian practices. The authors used the wetness (*W*) index, which accounted for slope factor and contributing area and was defined as

[3] $$W = \ln(\mathrm{AS}/\tan\alpha)$$

where AS is the upslope area contributing to unit cell ($m^2\ m^{-1}$). The wetness index is built on the concept similar to those of slope index and is intended to identify buffer cells with large contributing areas and low slopes, which are most suitable for buffer establishment. Using wetness index on Iowa watershed, Tomer et. al (2003) showed that nearly half of riparian zones had $W > 8$, indicating the area where surface flow would be generated during and after storms and allowing for considerable buffer optimization.

Barling et al. (1994) developed a quasi-dynamic wetness index that improves on the steady state wetness index by incorporating upslope area, topography, and time of concentration. Fried et al. (2000)

used both static and quasi-dynamic approaches to model variable width buffers in a 27 000-ha Michigan watershed with rolling topography. The authors used four models to assess the sensitivity of buffer delineation decision to different types of wetness index and flow routing algorithms. The use of wetness index for buffer delineation seems to be especially appropriate if one considers that areas dominated by saturated soils are the most critical for nitrate removal (Gold et al. 2001). Fried et al. (2000) demonstrated that the buffers created using a static wetness index were more evenly distributed along the streams, while those created using dynamic models were much more consolidated. Although the dynamic models performed better from a water quality standpoint, the static approach seemed to be more practical.

Using the concept that the wetness index and stream power provide a basis for locating likely sites for ephemeral gullies formation, Fried et al. (2000) and Moore et al. (1988) demonstrated the need to identify hot spots where sediment from potentially erodible source areas might be delivered to the stream channels. Addressing these concerns Tomer et al. (2003) introduced the empirically based erosion index (E) to identify erosion-prone areas adjacent to streams

$$[4] \qquad E = \left(\frac{\mathrm{AS}}{22.13}\right)^{0.4}\left(\frac{\sin\alpha}{0.0896}\right)^{1.3}$$

The use of erosion index on a watershed with undulating topography (Tomer et al. 2003) showed that erosion sensitive areas along the stream bank were relatively small in size and uniformly distributed along the channel length. Conceptually, the erosion index is similar to LS factor in RUSLE (Renard et al. 1997), however the erosion index uses contributing area rather than slope length in its expression. The use of erosion index is similar to a more general approach to conservation planning proposed by Berry et al. (2003), which is based on combining slope and flow concentration maps to derive erosion potential maps.

Despite the abundance of various landscape hot spots indicators, simply following an optimization algorithm could make buffers too dissected for practical implementation. For example, area index of contiguous stream sections on an Australian watershed (Bren 2000) differed one order of magnitude or more, which resulted in extremely variable buffer width. Often the economical constraints preclude the installation of protective areas in close agreement with the optimal model. The right balance between the level of optimization and the feasibility of implementation presents the greatest challenge in precision buffer design.

Economic benefits of precision riparian buffers

An estimate of riparian buffer efficiencies is not complete without consideration of its economic feasibility. Usually, riparian buffer is evaluated with respect to its ability to abate agricultural nonpoint source pollution (Fox et al. 1995; Qiu and Prato 2001; Rein 1999). The gross economic value of riparian buffers is estimated by comparing total watershed net returns with and without riparian buffers. The net economic value is the value of improved water quality, including mitigation of the downstream environmental impacts, under a given conservation practice minus the on-farm costs of its adoption or the opportunity cost (Qiu and Prato 1998).

Cost and benefit analysis of riparian buffers in order to justify allocation of funds for ecosystem maintenance has been attempted in a number of studies (Stonehouse 1999). The costs usually include foregone agricultural benefits and strip establishment; meanwhile, the benefits include soil loss reduction, reduction of the need for herbicides, groundwater recharge and associated pumping cost reduction, establishment of beneficial insect and animal habitat, reduction of harbor dredging, mosquito abatement, drinking water quality improvement, recreational value, and flood control (Rein 1999).

Using cost benefit approach, Qiu and Prato (1998) showed that riparian buffers usually have positive net economic value. Lant and Roberts (1990), using a survey, investigated the recreational and intrinsic values that residents place upon local streams, rivers, and reservoirs in river basins in Iowa and Illinois.

The authors found that the environmental services provided by riparian buffers may exceed the value of agricultural products on a per-acre basis.

Different sections of riparian buffers have uneven efficiencies in terms of pollution abatement within a watershed (Sparovek et al. 2002). Modeling conducted by Yang and Weersink (2004) on an Ontario watershed showed that cost effectiveness of conservation buffers could be increased if the width of the buffer strip was allowed to vary by location taking into account slope steepness and other parameters. These results have important implications for practical decision-making and must encourage further research in precision conservation. In addition, the extension of buffer strips from visible streams to runoff channels may have a positive effect on their economic value (Yang and Weersink 2004). Targeting the low-cost, high environmental benefit locations (Babcock et al. 1996) may enhance the cost effectiveness of riparian buffers, particularly where the spatial heterogeneity of the watersheds is high. In addition, it can be corroborated from previous research (Bren 2000; Tomer et al. 2003) that precision buffers are more area effective (occupy less space while providing the same pollution mitigation benefits) than traditional buffers.

Despite the overall positive net economic value of riparian buffers, producers often lack the incentive for their implementation, because the prices of agricultural commodities produced in a given area do not include the value of water quality improvement (Qiu and Prato 2001). Under these circumstances non-market incentives need to play a role. In the United States concern about the impact of agriculture on the environment has led to the establishment of a number of state-wide and national mitigation programs, many of which support practices to improve water quality. Precision riparian buffers fit well into the general scope of activities funded under these programs. In addition, emphasis on the optimal use of land under precision conservation may stimulate greater interest and involvement from farmers and land owners.

Conservation Reserve Program (CRP) encourages land users to convert highly erodible cropland or environmentally sensitive land to vegetative cover, such as wildlife habitat plantings, native grasses, filter strips, or riparian buffers on a cost sharing basis. Environmental Quality Incentives Program (EQIP) provides assistance, including incentive payments and cost sharing, to establish conservation practices such as manure management systems, pest management, erosion control, and others to help farmers meet water quality objectives. The Wetlands Reserve Program (WRP) is a voluntary program to restore wetlands. Participating landowners can establish conservation measures, which may include filter strips and riparian buffers. In exchange for establishing a permanent easement, the landowner receives payment for the agricultural value of the land and the restoration costs. Wildlife Habitat Incentives Program (WHIP) is a United States Department of Agriculture program that provides financial incentives to develop habitat for fish and wildlife on private lands. Cost-share assistance is provided for the initial implementation of a wide range of wildlife habitat development practices including riparian zones. The Stewardship Incentive Program (SIP) is designed to stimulate improved management of private forest land on a cost sharing basis. Activities supported by this program include establishment, maintenance and restoration of shelterbelts, windbreaks and riparian zones.

Conclusions

Existing research has proven the effectiveness of vegetative buffers in the reduction of sediment and nutrient movement from field to streams at the scale of individual transects. Sediment retention ratios between 45% (Daniels and Gilliam 1996) and 98% have been reported (Lim et al. 1998). A number of studies conducted under a variety of vegetation types and buffer widths report phosphorus removal rates from surface runoff between 60% and 90% (Line et al. 2000; Young et al. 1980). Despite this ample research, there is still a lack of understanding of buffer effectiveness at the landscape scale. Case studies are still the primary source of information for protective buffer planning and assessment. The variability of landscape appears to be the major factor that defines riparian buffer efficiency. Surveys show that usually only a small fraction (9% to 18%) of the total buffer area actually contacted runoff water (Dosskey et al. 2002), which may reduce its trapping efficiency from the potential 41% to 99% to the actual 15% to 43%. Other reasons for the discrepancy between the potential and the actual efficiency include spatial variability of contributing areas adjacent to specific buffer sections, erosion rates, soil properties, nonpoint pollution sources, distance to watercourse, topography, etc. As various planning and research tools became more widely available, technical advances such as GIS, GPS, greater instrumentation and computational capabilities make our approach to precision conservation more flexible. Given the general tendency of development of agriculture towards precision-oriented methods it is logical that precision conservation should follow the same path. The use of various topographic indicators such as area, slope, wetness, and erosion indexes has proven to be a useful tool in precision buffer delineation. Efficient management of natural resources is needed if we intend to meet the challenges of sustainable agriculture. Precision buffer planning should start by defining potential pollutants of great concern for a given area, after which the buffers must be designed to counteract the specific pollutant, making it a goal-oriented rather than a standardized approach. Today there is still a lack of quantitative methods, capable of evaluating field runoff patterns and their impacts on riparian buffer effectiveness. Development of new indexes to identify hot spots from where sediment and pollutants might be delivered to the stream channels is needed. The consideration of economic viability should become a driver and the major advantage of precision conservation over traditional approaches. Development of climate, region, and site-specific recommendations for determining buffer placement and vegetation composition will help ensure that the optimal environmental benefits are achieved.

References

Abu-Zreig, M. 2001. Factors affecting sediment trapping in vegetated filter strips: simulation study using VFSMOD. Hydrol. Process. **15**: 1477–1488.

Abu-Zreig, M., Rudra, R.P., and Whiteley, H.R. 2001. Validation of a vegetated filter strip model (VFSMOD). Hydrol. Process. **15**: 729–742.

Abu-Zreig, M., Rudra, R.P., Whiteley, H.R., Lalonde, M.N., and Kaushik, N.K. 2003. Phosphorus removal in vegetated filter strips. J. Environ. Qual. **32**: 613–619.

Addy, K.L., Gold, A.J., Groffman, P.M., and Jacinthe, P.A. 1999. Ground water nitrate removal in subsoil of forested and mowed riparian buffer zones. J. Environ. Qual. **28**: 962–970.

Arora, K., Mickelson, S.K., and Baker, J.L. 2003. Effectiveness of vegetated buffer strips in reducing pesticide transport in simulated runoff. Trans. ASAE, **46**: 635–644.

Babcock, B.A., Lakshminarayan, P.G., Wu, J., and Zilberman, D. 1996. The economics of public fund for environmental amenities: A study of CRP contracts. Am. J. Agric. Econ. **78**: 961–971.

Baird, J.H., Basta, N.T., Huhnke, R.L., Johnson, G.V., Payton, M.E., Storm, D.E., Wilson, C.A., Smolen, M.D., Martin, D.L., and Cole, J.T. 2000. Best management practices to reduce pesticide and nutrient runoff from turf. *In* Fate and management of turfgrass chemicals. Vol. 743. *Edited by* J.M. Clark and M.P. Kenna. American Chemical Society, Washington, D.C. pp. 268–293.

Baker, M.E., Wiley, M.J., and Seelbach, P.W. 2001. GIS-based hydrologic modeling of riparian areas: Implications for stream water quality. J. Am. Water Resour. Assoc. **37**: 1615–1628.

Band, L.E. 1989. A terrain-based watershed information-system. Hydrol. Process. **3**: 151–162.

Band, L.E., Patterson, P., Nemani, R., and Running, S.W. 1993. Forest ecosystem processes at the watershed scale — incorporating hillslope hydrology. Agric. Forest Meteorol. **63**: 93–126.

Barfield, B.J., Blevins, R.L., Fogle, A.W., Madison, C.E., Inamdar, S., Carey, D.I., and Evangelou, V.P. 1998. Water quality impacts of natural filter strips in karst areas. Trans. ASAE, **41**: 371–381.

Barling, R.D., Moore, I.D., and Grayson, R.B. 1994. A quasi-dynamic wetness index for characterizing the spatial distribution of zones of surface saturation and soil-water content. Water Resour. Res. **30**: 1029–1044.

Berry, J.K., Delgado, J.A., Khosla, R., and Pierce, F.J. 2003. Precision conservation for environmental sustainability. J. Soil Water Conserv. **58**: 332–339.

Beven, K. 1997. TOPMODEL: A critique. Hydrol. Process. **11**: 1069–1085.

Bharati, L., Lee, K.H., Isenhart, T.M., and Schultz, R.C. 2002. Soil-water infiltration under crops, pasture, and established riparian buffer in Midwestern USA. Agrofor. Syst. **56**: 249–257.

Borin, M., and Bigon, E. 2002. Abatement of NO_3-N concentration in agricultural waters by narrow buffer strips. Environ. Pollut. **117**: 165–168.

Bren, L.J. 1998. The geometry of a constant buffer-loading design method for humid watersheds. For. Ecol. Manag. **110**: 113–125.

Bren, L.J. 2000. A case study in the use of threshold measures of hydrologic loading in the design of stream buffer strips. For. Ecol. Manag. **132**: 243–257.

Calhoun, J.M. 1988. Riparian management practices of the Department of Natural Resources. *In* Streamside management: riparian wildlife and forestry interaction. *Edited by* K. Raedeke. Contribution # 59. Institute of Forest Resources, University of Washington, Seattle, Wash. pp. 207–211.

Cooper, A.B., Smith, C.M., and Bottcher, A.B. 1992. Predicting runoff of water, sediment, and nutrients from a New Zealand grazed pasture using CREAMS. Trans. ASAE, **35**: 105–112.

Cooper, A.B., Smith, C.M., and Smith, M.J. 1995. Effects of riparian set-aside on soil characteristics in an agricultural landscape — implications for nutrient transport and retention. Agric. Ecosyst. Environ. **55**: 61–67.

Corley, C.J., Frasier, G.W., Trlica, M.J., Smith, F.M., and Taylor, E.M. 1999. Technical Note: Nitrogen and phosphorus in runoff from two montane riparian communities. J. Range Manage. **52**: 600–605.

Daniels, R.B., and Gilliam, J.W. 1996. Sediment and chemical load reduction by grass and riparian filters. Soil Sci. Soc. Am. J. **60**: 246–251.

Desmet, P.J.J., and Govers, G. 1995. GIS-based simulation of erosion and deposition patterns in an agricultural landscape — a comparison of model results with soil map information. Catena, **25**: 389–401.

Dillaha, T.A., Reneau, R.B., Mostaghimi, S., and Lee, D. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. Trans. ASAE, **32**: 513–519.

Dosskey, M.G., Helmers, M.J., Eisenhauer, D.E., Franti, T.G., and Hoagland, K.D. 2002. Assessment of concentrated flow through riparian buffers. J. Soil Water Conserv. **57**: 336–343.

Dukes, M.D., Evans, R.O., Gilliam, J.W., and Kunickis, S.H. 2002. Effect of riparian buffer width and vegetation type on shallow groundwater quality in the Middle Coastal Plain of North Carolina. Trans. ASAE, **45**: 327–336.

Flanagan, D.C., Foster, G.R., Neibling, W.H., and Burt, J.P. 1989. Simplified equations for filter strip design. Trans. ASAE, **32**: 2001–2007.

Fox, G., Umali, G., and Dickinson, T. 1995. An economic analysis of targeting soil conservation measures with respect to off-site water quality. Can. J. Agric. Econ. **43**: 105–108.

Fried, J.S., Brown, D.G., Zweifler, M.O., and Gold, M.A. 2000. Mapping contributing areas for stormwater discharge to streams using terrain analysis. *In* Terrain analysis: principles and applications. *Edited by* J.P. Wilson and J.C. Gallant. John Wiley, New York. pp. 183–203.

Geyer, D.J., Keller, C.K., Smith, J.L., and Johnstone, D.L. 1992. Subsurface fate of nitrate as a function of depth and landscape position in Missouri Flat Creek watershed, USA. J. Contam. Hydrol. **11**: 127–147.

Gharabaghi, B., Rudra, R.P., Whiteley, H.R., and Dickinson, T. 2000. Sediment removal efficiency of vegetative filter strips. Annu. Res. Rep. Guelph Turfgrass Institute, Guelph, Ont.

Gold, A.J., Groffman, P.M., Addy, K., Kellogg, D.Q., Stolt, M., and Rosenblatt, A.E. 2001. Landscape attributes as controls on groundwater nitrate removal capacity of riparian zones. J. Am. Water Resour. Assoc. **37**: 1457–1464.

Gregory, S.V., Swanson, F.J., McKee, W.A., and Cummins, K.W. 1991. An ecosystem perspective of riparian zones. Bioscience, **41**: 540–551.

Haycock, N.E., and Pinay, G. 1993. Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. J. Environ. Qual. **22**: 273–278.

Herron, N.F., and Hairsine, P.B. 1998. A scheme for evaluating the effectiveness of riparian zones in reducing overland flow to streams. Aust. J. Soil Res. **36**: 683–698.

Herron, N., and Wilson, C. 2001. A water balance approach to assessing the hydrologic buffering potential of an alluvial fan. Water Resour. Res. **37**: 341–351.

Hill, A.R. 1996. Nitrate removal in stream riparian zones. J. Environ. Qual. **25**: 743–755.

Hook, P.B. 2003. Sediment retention in rangeland riparian buffers. J. Environ. Qual. **32**: 1130–1137.

Jin, C.X., and Romkens, M.J.M. 2001. Experimental studies of factors in determining sediment trapping in

vegetative filter strips. Trans. ASAE, **44**: 277–288.
Kuusemets, V., Mander, U., Lohmus, K., and Ivask, M. 2001. Nitrogen and phosphorus variation in shallow groundwater and assimilation in plants in complex riparian buffer zones. Water Sci. Technol. **44**: 615–622.
Lant, C.L., and Roberts, R.S. 1990. Greenbelts in cornbelt: riparian wetlands, intrinsic values and market failure. Environ. Plann. **22**: 1375–1388.
Le Bissonnais, Y., Montier, C., Jamagne, M., Daroussin, J., and King, D. 2002. Mapping erosion risk for cultivated soil in France. Catena, **46**: 207–220.
Lee, K.H., Isenhart, T.M., Schultz, R.C., and Mickelson, S.K. 2000. Multispecies riparian buffers trap sediment and nutrients during rainfall simulations. J. Environ. Qual. **29**: 1200–1205.
Lee, P., Smyth, C., and Boutin, S. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. J. Environ. Manag. **70**: 165–180.
Leopold, L.B., Wolman, M.G., and Miller, J.P. 1992. Fluvial processes in geomorphology. Dover, Mineola, New York.
Lim, T.T., Edwards, D.R., Workman, S.R., Larson, B.T., and Dunn, L. 1998. Vegetated filter strip removal of cattle manure constituents in runoff. Trans. ASAE, **41**: 1375–1381.
Line, D.E., Harman, W.A., Jennings, G.D., Thompson, E.J., and Osmond, D.L. 2000. Nonpoint-source pollutant load reductions associated with livestock exclusion. J. Environ. Qual. **29**: 1882–1890.
Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucas, W., and Todd, A.H. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environ. Manag. **21**: 687–712.
Lowrance, R., Williams, R.G., Inamdar, S.R., Bosch, D.D., and Sheridan, J.M. 2001. Evaluation of coastal plain conservation buffers using the riparian ecosystem management model. J. Am. Water Resour. Assoc. **37**: 1445–1455.
Maas, R.P., Smolen, M.D., and Dressing, S.A. 1985. Selecting critical areas for nonpoint source pollution control. J. Soil Water Conserv. **40**: 68–71.
Magette, W.L., Brinsfield, R.B., Palmer, R.E., and Wood, J.D. 1989. Nutrient and sediment removal by vegetated filter strips. Trans. ASAE, **32**: 663–667.
Martin, T.L., Kaushik, N.K., Trevors, J.T., and Whiteley, H.R. 1999. Review: Denitrification in temperate climate riparian zones. Water Air Soil Pollut. **111**: 171–186.
McCarty, G., and Angier, J. 2001. Impact of preferential flow pathways on ability of riparian wetlands to mitigate agricultural pollution. *In* Preferential Flow, Water Movement and Chemical Transport in the Environment, Proceedings of the 2nd International Symposium, Honolulu, Hawaii, 3–5 January 2001. *Edited by* D.D. Bosch and K.W. King. American Society of Agricultural Engineers, St. Joseph, Mich. pp. 53–56.
McGlynn, B.L., and McDonnell, J.J. 2003. Quantifying the relative contributions of riparian and hillslope zones to catchment runoff. Water Resour. Res. **39**(11), 1310. DOI: 10.1029/2003WR002091.
McGlynn, B.L., and Seibert, J. 2003. Distributed assessment of contributing area and riparian buffering along stream networks. Water Resour. Res. **39**(4), 1082. DOI: 10.1029/2002WR001521.
McKergow, L.A., Weaver, D.M., Prosser, I.P., Grayson, R.B., and Reed, A.E.G. 2003. Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. J. Hydrol. **270**: 253–272.
Misra, A.K., Baker, J.L., Mickelson, S.K., and Shang, H. 1996. Contributing area and concentration effects on herbicide removal by vegetative buffer strips. Trans. ASAE, **39**: 2105–2111.
Moore, I.D., Burch, G.J., and Mackenzie, D.H. 1988. Topographic effects on the distribution of surface soil-water and the location of ephemeral gullies. Trans. ASAE, **31**: 1098–1107.
Munoz-Carpena, R., Parsons, J.E., and Gilliam, J.W. 1999. Modeling hydrology and sediment transport in vegetative filter strips. J. Hydrol. **214**: 111–129.
Muscutt, A.D., Harris, G.L., Bailey, S.W., and Davies, D.B. 1993. Buffer zones to improve water quality — a review of their potential use in UK agriculture. Agric. Ecosyst. Environ. **45**: 59–77.
Oloughlin, E.M. 1981. Saturation regions in catchments and their relations to soil and topographic properties. J. Hydrol. **53**: 229–246.
Osborne, L.L., and Kovacic, D.A. 1993. Riparian vegetated buffer strips in water quality restoration and stream management. Freshw. Biol. **29**: 243–258.
Phillips, J.D. 1989. Evaluating estuarine shoreline buffer zonez for nonpoint source pollution control. *In*

Coastal Zone '89, Proceedings of the 6th Symposium on Coastal and Ocean Management, Charleston, South Carolina, 11–14 July 1989. *Edited by* O.T. Magoon. American Society of Civil Engineers, New York. pp. 399–411.

Pinay, G., Roques, L., and Fabre, A. 1993. Spatial and temporal patterns of denitrification in a riparian forest. J. Appl. Ecol. **30**: 581–591.

Qiu, Z.Y., and Prato, T. 1998. Economic evaluation of riparian buffers in an agricultural watershed. J. Am. Water Resour. Assoc. **34**: 877–890.

Qiu, Z.Y., and Prato, T. 2001. Physical determinants of economic value of riparian buffers in an agricultural watershed. J. Am. Water Resour. Assoc. **37**: 295–303.

Reed, T., and Carpenter, S.R. 2002. Comparisons of P-yield, riparian buffer strips, and land cover in six agricultural watersheds. Ecosystems, **5**: 568–577.

Rein, F.A. 1999. An economic analysis of vegetative buffer strip implementation — case study: Elkhorn Slough, Monterey Bay, California. Coast. Manag. **27**: 377–390.

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K., and Yoder, D.C. 1997. Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). US Department of Agriculture, Washington, D.C.

Robinson, C.A., Ghaffarzadeh, M., and Cruse, R.M. 1996. Vegetative filter strip effects on sediment concentration in cropland runoff. J. Soil Water Conserv. **51**: 227–230.

Schnabel, R.R., Shaffer, J.A., Stout, W.L., and Cornish, L.F. 1997. Denitrification distributions in four valley and ridge riparian ecosystems. Environ. Manag. **21**: 283–290.

Sheridan, J.M., Lowrance, R., and Bosch, D.D. 1999. Management effects on runoff and sediment transport in riparian forest buffers. Trans. ASAE, **42**: 55–64.

Sotomayor, D., and Rice, C.W. 1996. Denitrification in soil profiles beneath grassland and cultivated soils. Soil Sci. Soc. Am. J. **60**: 1822–1828.

Sparovek, G., Ranieri, S.B.L., Gassner, A., De Maria, I.C., Schnug, E., dos Santos, R.F., and Joubert, A. 2002. A conceptual framework for the definition of the optimal width of riparian forests. Agric. Ecosyst. Environ. **90**: 169–175.

Spruill, T.B. 2000. Statistical evaluation of effects of riparian buffers on nitrate and ground water quality. J. Environ. Qual. **29**: 1523–1538.

Srivastava, P., Edwards, D.R., Daniel, T.C., Moore, P.A., and Costello, T.A. 1996. Performance of vegetative filter strips with varying pollutant source and filter strip lengths. Trans. ASAE, **39**: 2231–2239.

Srivastava, P., Costello, T.A., Edwards, D.R., and Ferguson, J.A. 1998. Validating a vegetative filter strip performance model. Trans. ASAE, **41**: 89–95.

Stocking, M.A. 1994. Assessing vegetative cover and management effects. *In* Soil erosion research methods. *Edited by* R. Lal. Soil and Water Conservation Society, Ankeny, Iowa. pp. 211–234.

Stonehouse, D.P. 1999. Economic evaluation of on-farm conservation practices in the Great Lakes region of North America. Environmetrics, **10**: 505–520.

Tingle, C.H., Shaw, D.R., Boyette, M., and Murphy, G.P. 1998. Metolachlor and metribuzin losses in runoff as affected by width of vegetative filter strips. Weed Sci. **46**: 475–479.

Tomer, M.D., James, D.E., and Isenhart, T.M. 2003. Optimizing the placement of riparian practices in a watershed using terrain analysis. J. Soil Water Conserv. **58**: 198–206.

Vellidis, G., Lowrance, R., Hubbard, R., and Gay, P. 2001. Preferential flow caused by past disturbance in a restored riparian wetland. *In* Preferential Flow, Water Movement and Chemical Transport in the Environment, Proceedings of the 2nd International Symposium, Honolulu, Hawaii, 3–5 January 2001. *Edited by* D.D. Bosch and K.W. King. American Society of Agricultural Engineers, St. Joseph, Mich. pp. 61–64.

Vought, L.B.M., Pinay, G., Fuglsang, A., and Ruffinoni, C. 1995. Structure and function of buffer strips from a water quality perspective in agricultural landscapes. Landsc. Urban Plann. **31**: 323–331.

Wallace, A. 1994. High precision agriculture is an excellent tool for conservation of natural resources. Commun. Soil Sci. Plant Anal. **25**: 45–49.

Weller, D.E., Jordan, T.E., and Correll, D.L. 1998. Heuristic models for material discharge from landscapes with riparian buffers. Ecol. Appl. **8**: 1156–1169.

Willems, H.P.L., Rotelli, M.D., Berry, D.F., Smith, E.P., Reneau, R.B., and Mostaghimi, S. 1997. Nitrate removal in riparian wetland soils: effects of flow rate, temperature, nitrate concentration and soil depth.

Water Res. **31**: 841–849.

Wondzell, S.M., and Swanson, F.J. 1996. Seasonal and storm dynamics of the hyporheic zone of a 4th-order mountain stream. 1. Hydrologic processes. J. North Am. Benthol. Soc. **15**: 3–19.

Xiang, W.N. 1993. Application of a GIS-based stream buffer generation model to environmental policy evaluation. Environ. Manag. **17**: 817–827.

Yang, W.H., and Weersink, A. 2004. Cost-effective targeting of riparian buffers. Can. J. Agric. Econ. **52**: 17–34.

Young, R.A., Huntrods, T., and Anderson, W. 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. J. Environ. Qual. **9**: 483–487.

Zakova, Z., Berankova, D., Kockova, E., and Kriz, P. 1993. Influence of diffuse pollution on the eutrophication and water quality of reservoirs in the Morava river basin. Water Sci. Technol. **28**: 79–90.