Assessing Critical Source Areas in Watersheds for Conservation Buffer Planning and Riparian Restoration

Zeyuan Qiu

Abstract A science-based geographic information system (GIS) approach is presented to target critical source areas in watersheds for conservation buffer placement. Critical source areas are the intersection of hydrologically sensitive areas and pollutant source areas in watersheds. Hydrologically sensitive areas are areas that actively generate runoff in the watershed and are derived using a modified topographic index approach based on variable source area hydrology. Pollutant source areas are the areas in watersheds that are actively and intensively used for such activities as agricultural production. The method is applied to the Neshanic River watershed in Hunterdon County, New Jersey. The capacity of the topographic index in predicting the spatial pattern of runoff generation and the runoff contribution to stream flow in the watershed is evaluated. A simple cost-effectiveness assessment is conducted to compare the conservation buffer placement scenario based on this GIS method to conventional riparian buffer scenarios for placing conservation buffers in agricultural lands in the watershed. The results show that the topographic index reasonably predicts the runoff generation in the watershed. The GIS-based conservation buffer scenario appears to be more cost-effective than the conventional riparian buffer scenarios.

Keywords Conservation buffers · Variable area source hydrology · Geographic information systems · Hydrologically sensitive areas · Critical source areas · Topographic index

Z. Qiu (✉)
Department of Chemistry and Environmental Science, New Jersey Institute of Technology, Newark, NJ 07102, USA
e-mail: zeyuan.qiu@njit.edu

Introduction

A large amount of water pollutants, such as nitrogen, phosphorus and sediments, are sourced from a relatively small hydrologically sensitive portion or critical source areas (CSAs) in a watershed (Dickinson and others 1990; Pionke and others 2000). Targeting CSAs in landscapes for implementing best management practices (BMPs) has long been recognized as an effective and efficient way to control nonpoint source pollution and to improve environmental quality (Maas and others 1985; Duda and Johnson 1985; Fox and others 1990; Gburek and others 2002). In recent years, there has been renewed interest in the targeting approach under the banner of precision conservation because of the advances in information technologies, such as geographic information systems (GIS), global positioning systems, remote sensing, environmental modeling and software, and the increased availability of spatial data for topography, soils, hydrology, land use/land cover and others (Berry and others 2003). Polyakov and others (2005) presented a comprehensive review on such BMPs as precision conservation buffer planning.

Conservation buffers have been used as a common practice to control diffuse water pollution, repair impaired streams and restore the ecosystem functions of a damaged watershed. Practical questions concerning where the conservation buffer should be placed and how wide the buffers should be are common. The most common practice is to define the riparian areas for buffer placement and buffer width by stream proximity (i.e., a fixed distance from the stream). Reconstruction and/or preservation of riparian buffers along stream banks are often blended into the local land use planning process by creating and implementing riparian buffer ordinances to control diffuse water pollution. The width of the stream buffer varies from 6.1 to

61 m (20–200 feet) in the buffer ordinances throughout the United States (Heraty 1993). The width chosen by a jurisdiction usually depends on the sensitivity and characteristics of the resource being protected and the political environment in the community.

Although the fixed-width riparian buffer placement strategy discussed above is simple, straightforward and easy to implement and enforce, it does not necessarily achieve full protection of streams (Bren 1998). Some studies show that conservation buffers are sometimes ineffective in removing diffuse water pollutants, such as sediment, nutrients and chemicals, in watersheds (Daniels and Gilliam 1996; Dosskey and others 2002). Ineffectiveness is related to the improper placement and size of buffers (Dosskey and others 2002; McKergow and others 2003). Although a 4.6- to 9.1-m (15- to 30-ft) wide buffer could be effective in trapping sediments and removing nitrogen, phosphorus and other pollutants in the short term (Dillaha and others 1989; Magette and others 1989), much wider buffers are required for long-term sediment retention (Cooper and others 1986; Lowrance and others 1986).

There were several alternative conservation buffer strategies that considered the topographic, soil and land use conditions within and around the buffers in watersheds and linked the location and size of buffers to the upland contributing areas. Bren (2000) and Tomer and others (2003) used the hydrologic and pollutant loading in delineating areas for buffers. Areas where the computed hydrologic loads exceed a threshold level were prioritized for buffer placement. Dosskey and others (2006) developed a method that uses soil survey attributes to guide the placement of conservation buffers. Xiang (1993) developed a GIS procedure that used topographic and soil conditions within buffers to derive the width for functioning buffers along streams in North Carolina. Herron and Hairsine (1998) developed two equations that defined the proportion of the hillslope that needed to be set aside as a riparian zone to stop overland runoff from reaching the stream under both the storage-limiting (corresponding to the saturation-excess runoff mechanism) and the infiltration-limiting (corresponding to infiltration-excess runoff mechanism) conditions, respectively. Their equations included climate, soil and landscape convergence factors. A riparian zone width not exceeding 20% of the total hillslope length was proposed as a practical management option after examining various hillslope, riparian zone and climate conditions in Australia.

The variable source area (VSA) hydrology had been closely linked to the delineation of CSAs for managing diffuse agricultural runoff (Johnes and Heathwaite 1997; Heathwaite and others 2000). According to the VSA hydrology, runoff is generated from saturated areas in landscapes where soil saturation capacity is exceeded and

the runoff generation is controlled by the development, expansion and contraction of these saturated areas (Hewlett and Hibbert 1967). Qiu (2003) explicitly emphasized the role of the VSA hydrology in planning conservation buffers. He suggested that the pattern of VSAs in landscapes provided a scientific basis for defining the location and size of conservation buffers, which is applicable from field level to watershed and regional levels. Installing conservation buffers within the VSAs could potentially prevent the formation of concentrated runoff flow and achieve both environmental and cost effectiveness. However, Qiu (2003) did not present a practical VSA modeling procedure that implemented the VSA-based buffer placement strategy for buffer planning and restoration.

The objectives of this study are to apply a topographic index approach to evaluate VSAs and to develop a practical GIS-based procedure that identifies CSAs in watersheds for conservation buffer placement using readily available spatial information. The procedure is illustrated by an application in the Neshanic River watershed in Hunterdon County, New Jersey that identifies agricultural lands in the watershed for conservation buffer placement and riparian restoration. The study will also validate the topographic index through its application in predicting stream flow and its two components (i.e., baseflow and runoff) in the watershed, and finally compare the cost-effectiveness of the GIS-based conservation buffer scenarios with the conventional riparian buffer scenarios in the watershed. Cost-effectiveness is a comprehensive economic framework to evaluate environmental policy and conservation management. It has gained widespread acceptance as a method for evaluating alternative policies and best management practices for controlling nonpoint source pollution (Ribaudo and others 1999). In this application, a simple measurement on cost-effectiveness, as used by Prato and Shi (1990) and Qiu (2003), will be developed to compare different conservation buffer placement scenarios.

Methods

Deriving Topographic Index

The topographic index is the basis for the rainfall-runoff model, TOPography based hydrological MODEL (TOPMODEL), which simulates the pattern of surface runoff contributing areas following the VSA hydrology. The most common form of the topographic index (λ) is:

$$\lambda = \ln\left(\frac{\alpha}{\tan\beta}\right), \tag{1}$$

where α is the upslope contributing area per unit contour length to a given point in a watershed in meters and β is the

local surface slope angle in decimal. The index measures the propensity of a given point in watershed to become saturated and act as a source area for surface runoff (Beven and Kirkby 1979). The index is often called a wetness index (Moore and others 1991). In practice, a watershed is divided into small grids and the index is measured at each grid. The index is often derived from a digital elevation model (DEM) using terrain analysis in GIS (Tarboton 1997). The higher the index, the more likely the grid is saturated during a storm event. The index was also used in other VSA-based models, such as TOPOG (O'Loughlin 1986) and WET (Moore and others 1993).

Bren (2000) and Tomer and others (2003) implicitly and explicitly used the index in their conservation buffer planning approaches, respectively. Bren (2000) used two slope convergence parameters (i.e., specific area and slope index) as surrogate measures of hydrologic loading. The specific area and slope indices used by Bren (2000) were very similar to α and $\tan(\beta)$, respectively. Tomer and others (2003) used a wetness index and an empirical erosion index to measure the hydrologic and sediment loading for buffer planning. The wetness index was the same as this index and the empirical erosion index was a function of α and $\tan(\beta)$.

Because of its simplicity and the ability to visualize the spatial predictions, TOPMODEL has been widely adopted to simulate hydrologic processes with various modifications and extensions. As noted by Beven (1997), TOPMODEL is not a single model, but rather a set of conceptual tools that can be used to simulate hydrological processes in a relatively simple way, particularly the dynamics of surface or subsurface contributing areas. The original TOPMODEL implicitly assumed a water table below the entire watershed, which did not conceptually apply to watersheds hydrologically controlled by shallow interflow of perched groundwater. Walter and others (2002) further re-conceptualized the TOPMODEL by using soil moisture deficit instead of water table depth as the state variable to make it applicable to shallow, interflow-driven watersheds. Consequently, the topographic index that predicts the spatial distribution of runoff contributing areas can be approximated as follows:

$$\lambda = \ln\left(\frac{\alpha}{\tan(\beta)K_sD}\right) = \ln\left(\frac{\alpha}{\tan(\beta)}\right) - \ln(K_sD), \qquad (2)$$

where D is the local soil depth in meters to the fragipan, bedrock or other type of restrictive layer, K_s is the mean saturated hydraulic conductivity of the soil profile in meters per day above the restrictive layer, and other variables are defined as before. The redefined topographic index is similar to the one defined by Ambroise and others (1996). Equation 2 has two components: the wetness index, $\ln\left(\frac{\alpha}{\tan(\beta)}\right)$, and the soil water storage index, $\ln(K_sD)$. Unlike Eq. 1, Eq. 2 considers the impacts on the index of soil

water storage capacity (K_sD) above the restrictive layer. In general, the deeper the topsoil (D) and the higher the saturated hydraulic conductivity (K_s), the lower the value of λ, which implies a lower likelihood of generating surface runoff. In reality, there are several topsoil layers with different K_s values above a restrictive layer or bedrock. In this case, K_s is defined as

$$K_s = \frac{d}{\sum_1^n (d_i/k_i)}, \qquad (3)$$

where d is total thickness of the soil above the restrictive layer; d_i is the thickness of layer i, k_i is the saturated hydraulic conductivity of layer i (Freeze and Cherry 1979).

Evaluating the Topographic Index

Since it was first introduced, the wetness index had been subject to intensive investigation on its ability to predict the spatial pattern of runoff contributing areas in the hydrology literature. The predicted spatial patterns were compared to the measured soil moisture pattern (Grayson and Western 1998; Grayson and others 2002), the shallow groundwater table (Moore and Thompson 1996; Seibert and others 1997) and the observed wetland occurrence (Rodhe and Seibert 1999; Merot and others 2003). However, the validation results were mixed. While some reasonable match was reported, many found the predicted and observed spatial patterns did not always match well. The mismatch was not entirely unexpected because the soil moisture, shallow groundwater table and wetland formation were subject to many micro and macro natural and man-made conditions such as soil, geology, climate and land use in addition to the topographic condition in watersheds. Many variants of the wetness index were developed by incorporating those factors and conditions to therefore improve their capacity to predict the spatial pattern of runoff contributing areas.

The revised topographic index used here represented one of many variants of the wetness index. It had been extensively evaluated in terms of its ability to predict the spatial pattern of runoff contributing areas, and to predict stream flow in the Catskill Mountains of New York. Lyon and others (2004) and Agnew and others (2006) compared the predicted runoff potential pattern by Eq. 2 to the field monitored moisture pattern and to the predicted runoff pattern given by a fully distributed Soil Moisture Routing (SMR) model in the Town Brook watershed in the region. SMR is a mechanistic physical-based, spatially distributed model that captures the VSA hydrology of humid, temperate regions like the Northeastern United States that have highly permeable, sloping soils over a restricting layer (Frankenberger and others 1999). They concluded the topographic index given in Eq. 2 is a good indicator of the

risk of runoff, especially in the Northeastern United States. Schneiderman and others (2007) developed a variable source loading function (VSLF) model that modified the soil conservation curve number (SCS-CN) rainfall-runoff method by incorporating the index. Their model was precise and accurate in predicting runoff, stream flow and water quality in Cannonsville watershed in the region. de Alwis and others (2007) derived the normalized difference water index from medium resolution Landsat 7 ETM+ images to delineate the saturated area using an unsupervised classification method and found that the delineated pattern of saturated areas matched well the predictions made using the revised topographic index in the Town Brook watershed.

In this application, the VSLF model developed by Schneiderman and others (2007) is used to evaluate how well the derived topographic index can be incorporated to predict the stream flow and its runoff and baseflow components in the study area. Following Schneiderman and others (2007), the watershed is divided into 10 different saturation classes based on the spatial pattern of the topographic index derived using Eq. 2. The resulting runoff is predicted by

$$Q = \sum_{i=1}^{10} q_i \Delta A_{s,i}, \qquad (4)$$

where i is the index of saturation class, $\Delta A_{s,i}$ is the fraction of the watershed in the saturation class i and q_i is the per unit runoff depth for saturation class i predicted by the traditional SCS-CN rainfall-runoff method based on land use and soil conditions within the saturation class.

Determining Critical Source Areas (CSAs)

In this application, the topographic index given in Eq. 2 is used to determine CSAs for conservation buffer placement and riparian restoration in a watershed. The general procedure for determining CSAs consists of three steps. In step 1, Eq. 2 was used to derive the topographic index and identify the spatial patterns of VSAs from a DEM and a soil database. GIS software, such as ArcGIS, was used to process these data. When applied to a watershed, Eq. 2 generates a topographic index for every grid of the watershed (the grid size is depended upon the resolution of the DEM) that indicated the propensity or probability of generating overland runoff of each grid.

It is always challenging for land and resource managers to use probability-based VSAs patterns to make management decisions. They tend to prefer well-defined, static HSAs for targeting BMPs, such as conservation buffers. Therefore, step 2 delineates HSAs from the predicted VSA pattern in step 1. HSAs are defined as the parts of VSAs

that most actively contribute to generation of overland runoff in landscapes. Various subjective and objective criteria can be used to delineate the HSA pattern. For example, the HSAs should be delineated to control the impacts of a specific storm event (Lyon and others 2006). Given natural conditions, such as topography, soil, land use/cover and hydrology, in landscapes, a series of VSA patterns corresponding to dynamic rainfall events can be identified in the watershed (Gérard-Marchant and others 2003; Qiu 2003). Agnew and others (2006) suggested using the average saturation probability to delineate the HSAs. One can also simply apply the practical option suggested by Herron and Hairsine (1998), which is to target 20% of the watershed area with the highest topographic index to prevent the concentrated overland flow from entering the stream and therefore achieve the potential effectiveness of the buffers. Other criteria include the local preferences, feasibility, and funding availability. The most practical way for delineating the HSAs is to choose a threshold level of the topographic index and to classify the areas that have higher topographic indices than the threshold level as HSAs. Several threshold levels can be experimented to derive the HSAs. The subjective and objective criteria discussed can be then applied to select the most proper threshold level in the study area.

CSAs are areas where pollution sources and transport coincide (Walter and others 2000; Heathwaite and others 2005; Agnew and others 2006). In step 3, CSAs can be determined as the intersection of the identified HSAs in step 2 and a set of pollutant source areas in landscapes. The pollutant source areas are generally the areas in landscapes that have been actively generating pollutants because of intensive land use activities such as agricultural and silvicultural production, residential development, and industrial and commercial uses. Walter and others (2001) presented an example of determining CSAs based on HSAs in an agricultural field. For an agricultural field in the Catskill Mountain region of New York State in which dairy manure is spread, CSAs were the intersection of the HSAs and the manure spread areas in the field. At a large landscape such as a watershed, the pollutant source areas can be interpreted from an existing land use map. In particular, CSAs for conservation buffer placement and riparian restoration are a subset of HSAs identified in step 2 that have intensive land use activities, such as agricultural production and urban development, which have the high potential to degrade stream water quality. It should be noted that land use activities have the impacts not only on sources, but also on patterns of VSAs and HSAs. Examples are roads, stormwater infrastructures, tramlines, tracks and field drainage systems (Tomer and others 2003; Heathwaite and others 2005). The impacts of such land use activities on the VSAs and HSAs patterns are ignored in this application,

but offer opportunities for future research. The identified CSAs provide the basis for targeting BMPs, such as conservation buffers.

The Study Area

The general GIS procedure of determining CSAs for conservation buffer placement and riparian restoration was applied to the 8,029-ha (31-mi^2) Neshanic River watershed in the Raritan River Basin located in Hunterdon County, New Jersey. The Neshanic River is a tributary to the South Branch of the Raritan River, which drains to the Atlantic Ocean. The Neshanic River was listed in the New Jersey 2008 Integrated Water Quality Monitoring and Assessment Report for impaired aquatic life, and nonpoint source pollution in bacteria, phosphorus, and total suspended solids (NJDEP 2008). The Neshanic River had either the highest concentrations of constituents or the highest frequency of not meeting water quality standards for 13 of the 17 constituents and was considered to be one of the worst water bodies in terms of overall water quality in the Raritan River Basin (Reiser 2004). Like many other parts in New Jersey, this watershed had been experiencing rapid suburbanization during the last two decades. Based on the land use/cover database compiled by the New Jersey Department of Environmental Protection (NJDEP), the percentage of the urban land in the watershed had increased from 17.4% in 1986 to 30.7% in 2002. The increases in urban land primarily came from agricultural land in the watershed. While other land uses were relatively steady, the agricultural lands in the watershed had decreased from 55% in 1986 to about 40% in 2002. While much of the water quality problems were attributed to the rapid urban development, agriculture was still an important source of water pollution in the watershed. The Neshanic River watershed had the highest percentage of agricultural lands among all watersheds in the Raritan River Basin. Therefore, controlling agricultural nonpoint source runoff is important for achieving the overall water quality goals in this watershed.

The Neshanic River watershed was recognized as one of the priority watersheds to implement agricultural best management practices including conservation buffers to improve water quality because of its relatively poor water quality and high percentage of agricultural lands (NJWSA 2002). Although there are many federal, state and local programs that fund conservation buffer placement and riparian restoration efforts in impaired watersheds, the $100 million New Jersey Conservation Reserve Enhancement Program (CREP) is still the primary funding mechanism for implementing conservation buffers on agricultural lands in New Jersey. New Jersey CREP covers 100% of the implementation costs of installing buffers and offers land rental payments to the enrolled landowners for 15 years. New Jersey CREP supports four types of conservation buffer practices on agricultural lands: grass waterways (CP8A); contour grass strips (CP15A); filter strips (CP21); and riparian buffer (CP22). So far, landowners had requested and received support for only three practices (i.e., CP8A, CP21 and CP22). Identifying CSAs for placing conservation buffers is expected to significantly improve the efficiency of CREP and the water quality in the watershed.

Data Analyses

Deriving Topographic Index

Two spatial datasets were used to derive the topographic index using Eq. 2 in the watershed: a digital elevation model (DEM), and a soil survey database. The 10-m resolution DEM was downloaded from NJDEP website. The NRCS Soil Survey Geographic (SSURGO) soil database for Hunterdon County was downloaded from the NRCS Soil Data Mart website.

The DEM was first processed using the open source ArcGIS extension Terrain Analysis Using Digital Elevation Models (TauDEM) by Tarboton (2005) and the ArcGIS Raster Calculator to obtain a wetness index grid, $\ln\left(\frac{\alpha}{\tan(\beta)}\right)$. There was a problem in calculating the wetness index when the local surface slope angle, β, is zero. To ensure the full coverage of the topographic index, we assumed a minimum surface slope of 0.0001 for all grids with surface slopes less than 0.0001.

The soil depth (D) and the mean saturated hydraulic conductivity (K_s) were extracted for each soil type from the Hunterdon County SSURGO soil database. There were two types of data associated with the county soil database: an ArcGIS shapefile showing the spatial distribution of all soil types in the county and a Microsoft Access database containing the attributes of those soil types. The Access database was first inquired to determine there was any restrictive layer such as fragipan or bedrock under each soil in the *corestrictions* table in the database. Then, the layer thickness and the associated K_s in the different soil layers of each soil were extracted from the *chorizon* table in the database. For a soil type that had a restrictive layer, only the layer thickness and the associated K_s of the soil layers above the restrictive layer were extracted. The soil depth D was calculated as the sum of the soil layer thickness and the geometric mean of the saturated hydraulic conductivity, K_s, in different soil layers was calculated using Eq. 3. The product (K_s*D) for each soil finally was calculated and linked to the soil spatial distribution GIS layer through the

name of the soil mapping unit *MUSYM*. The soil layer was converted into a soil water storage index raster layer based on the value of $\ln(K_s D)$. The converted raster layer had the same resolution as and was in alignment with the wetness index raster layer. About 149 ha or 1.87% of the watershed was classified as water, rough broken land, shale, quarry, and pits, sand and gravel for which there was no attribute value on K_s in the soil database. Since these soil classifications did not have any soil water storage capacity, the grids with these soil classifications were assigned a soil water storage index of −3, which was about 1.2 lower than the highest calculated soil water storage index of −1.8 in the watershed. Although the index of −3 was arbitrary, it was not expected to have a significant impact on the resulting topographic index because 90% of those grids were water, which generally had high wetness indices.

The Raster Calculator in ArcGIS was used to manipulate the raster layers for the wetness index from the DEM and the soil water storage index from the SSURGO soil data and to calculate the topographic index based on Eq. 2. Figure 1 presented the spatial distribution of the derived topographic index for the watershed, which ranged from 1 to 28. The darker color indicated a higher topographic index. Figure 2 gave the area distributions of the wetness index and the topographic index after rounding the index value of each grid to the nearest integer. The wetness index ranged from 3 to 25 and had a bell-shaped distribution with a longer right tail. The areas with an index value of 7, 8, and 9 accounted for about 26, 31 and 18% of the total area of the watershed, respectively. The calculated soil water storage index ranged from 2 to −3, but mainly had values

of 0, −1, and −2. The areas with a soil water storage index of 0, −1, and −2 comprised 45, 25, and 21% of the total area of the watershed, respectively. The added impacts of the soil water storage index on the wetness index resulted in an almost perfect bell-shaped distribution in the topographic index. Most areas in the watershed had a topographic index of 6, 7, 8, 9 or 10. The areas corresponding to these indices comprised 12, 19, 20, 17 and 12% of the total area of the watershed, respectively.

The higher the topographic index, the higher the likelihood of saturation and runoff. During a storm event, runoff would first be distributed among the grids with the higher topographic indices and then to the grids with the lower indices (Lyon and others 2004). Figure 2 also indicated the proportion of the areas of saturation in a watershed versus the measured topographic indices in the Neshanic River watershed. The proportion of the areas of saturation corresponding to a specific topographic index is measured as the accumulated area distributions of all greater topographic indices. For example, as indicated in Fig. 2, the proportion of runoff contributing areas is about 13.7% when all grids with the topographic index greater than or equal to 11 are saturated and 25% with the topographic index greater than or equal to 10. The curve presented in Fig. 2 can be theoretically connected to the probability of saturation based on more detailed hydrological modeling and measured climate data and provides a basis for defining HSAs (Agnew and others 2006).

VSLF Modeling

The VSLF model was applied to a 6,493-ha portion of the watershed that drains to the USGS Reaville stream flow gage station at the intersection of Reaville Road and the Neshanic River as shown in Fig. 1. Following Schneiderman and others (2007), the watershed was divided into 10 saturation classes with equal areas based on the topographic index. The 2002 land use/cover data developed by NJDEP was overlapped with the derived saturation class layer to identify the land use distribution in each saturation class. The land use/cover data were compiled from the aerial photographs taken in the spring of 2002 and were downloaded from NJDEP website. NJDEP land use/cover data used a modified Anderson classification system. The land uses in this watershed were classified into six broad land use categories including agriculture, barren, forest, urban, water and wetlands, and 50 subcategories that were further distinguished using a 4-digital land use classification codes. The NJDEP land use/cover data also contained the information on the impervious rate for each land use polygon. The impervious surface rate was used to separate the total area of each land use category in each saturation class into two parts: pervious and impervious areas. Based
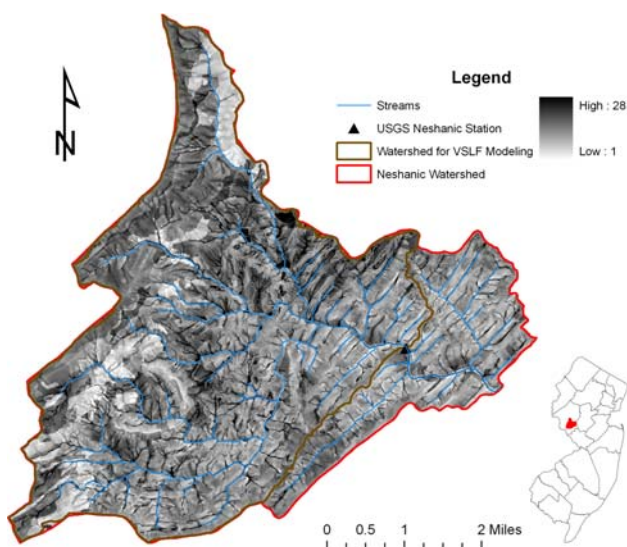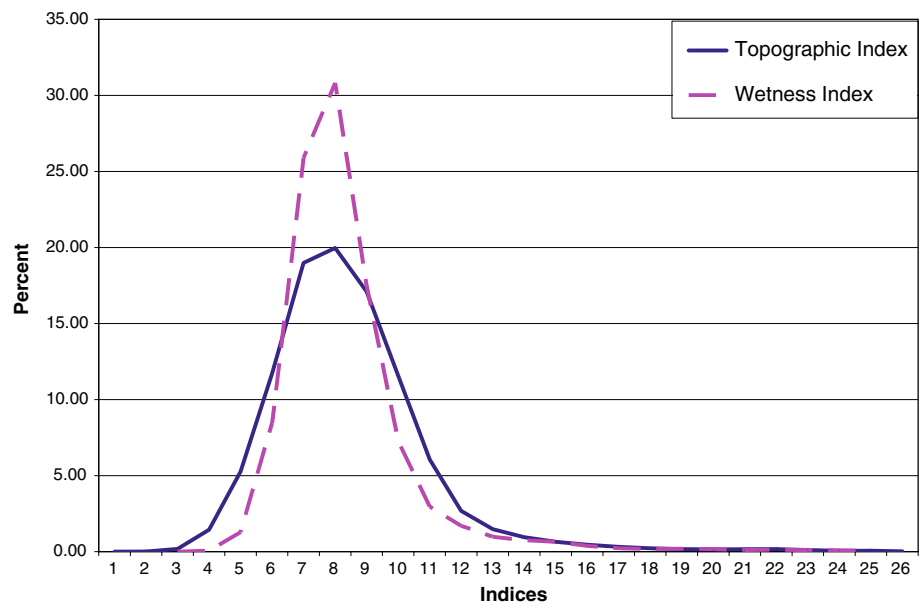

Fig. 1 Spatial distribution of the derived topographic index in Neshanic River watershed, Hunterdon County, New Jersey

Fig. 2 Area distribution of the derived topographic index and the wetness index in Neshanic River watershed
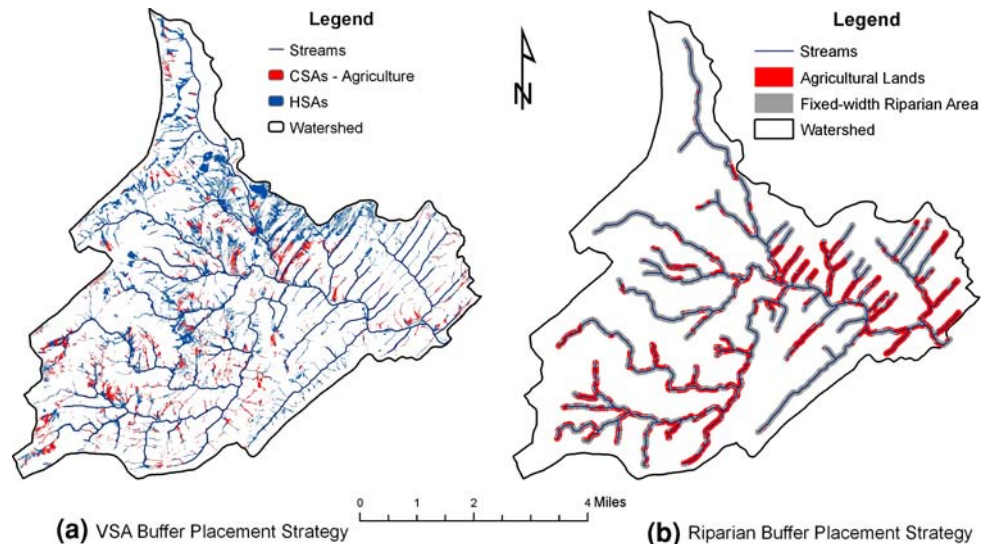

Applying Conservation Buffer Planning Strategies

The HSAs were defined as areas where the topographic index was greater than a specific threshold level. Corresponding to each threshold level, the spatial distribution of the HSAs could be identified from Fig. 1 and the aggregate area distribution in the watershed from Fig. 2. Several sets of HSAs could be produced by using several different threshold levels. The HSAs information could then be presented to a local watershed stakeholder group to evaluate and select the most proper set of HSAs. In this application, the areas with a topographic index of 11 or higher were considered to be HSAs, which is about 1,069 ha, i.e., 13.7% of the watershed. Although the topographic index 11 was arbitrarily selected as the threshold level for illustration purpose, it was considered to be reasonable. The 13.7% was considered to be more practical to achieve in the watershed (1998).

The identified HSAs were overlaid with a 2002 land use/cover layer to identify the CSAs for conservation buffer placement in the watershed. In the application, CSAs were the agricultural lands within the identified HSAs. The spatial distribution of the identified HSAs and CSAs in the watershed is presented in Fig. 3a. The distribution clearly indicated that the majority of the HSAs and CSAs were located in the upland areas outside the immediate riparian areas of the existing streams.

Figure 3 presents the distribution of the agricultural land for targeting conservation buffer under the two conservation buffer placement strategies. As shown in Fig. 3a, the VSA-based conservation buffer strategy was to place conservation

on its NJDEP 4 digital land use classification codes, the pervious areas in each saturation class were reclassified into the following land use categories required by VSLF model: forest deciduous; coniferous forest; mixed forest; brushland; cropland and pasture; residential; commercial and industrial, roads; wetlands; and water. The impervious areas in all saturation classes were lumped into the following land use categories: barnyard, residential impervious, commercial and industrial impervious, and roads, for the watershed. For example, the impervious areas from all agricultural lands and residential land uses were lumped into barnyard and residential impervious, respectively. The NJDEP land use classification did not distinguish the specific uses of agricultural lands. In the application, the total agricultural lands in each saturation class were further divided into pasture, permanent hay, and cropland based on the following percentages, 30, 20 and 50%, respectively. Those percentages were based on the field observation data obtained through two rounds of agricultural land use inventories throughout the watershed during the period 2007–2008. The precipitation, maximum and minimum temperature during 1960–2006, solar radiation and relative humidity during 1960–2004 in the Flemington weather station, which was just outside of the watershed boundary, were collected for the modeling. The stream flow observed at the gage station during 1960–2006 was also compiled into the model. Although there were long-term weather and stream flow data, VSLF model was calibrated for the period 1992–2000. The calibrated model was then used to evaluate its capacity to predict the stream flow during the period 2001–2005.

Fig. 3 Spatial distribution of agricultural lands for conservation buffer placement under two buffer placement strategies. **a** VSA buffer placement strategy. **b** Riparian buffer placement strategy

(a) VSA Buffer Placement Strategy

(b) Riparian Buffer Placement Strategy

buffers in the identified CSAs, i.e., the agricultural lands located within the identified HSAs. It should be realized that the distinction among the different types of agricultural lands should be made when actually placing conservation buffers. This assumption was made for simplicity due to the data limitation. Figure 3b shows the conventional riparian conservation buffer strategy that placed conservation buffers in the agricultural lands within the riparian areas of the streams. This application considered two scenarios under the riparian buffer strategy: establishing riparian buffers in all agricultural lands within the 54-m, for which the riparian areas were equivalent to the HSAs in the VSA strategy in size; and 30.5-m, which was commonly adopted by local communities for non-trout streams, of riparian corridors of the streams in the watershed.

Assessing Cost-Effectiveness

This application compared the cost-effectiveness of different scenarios for placing agricultural lands into conservation buffers supported by New Jersey CREP based on two buffer placement strategies discussed above. The cost-effectiveness of those scenarios was compared for achieving particular environmental benefits, such as improvements in water quality. The environmental benefits of different scenarios were approximated by the controlled runoff potential, which is measured by the average value of the topographic indices in these agricultural lands converted into conservation buffers. The costs for establishing and maintaining buffers were based on the average costs of the buffer installation and maintenance costs in the agricultural lands enrolled in the New Jersey CREP. For agricultural lands enrolled in New Jersey CREP, USDA, NJ Department of Agriculture and other agencies offered a one-time sign-up incentive, 100% of the installation costs, and 15 years of buffer maintenance

costs and soil rental costs. The soil rental costs were the governmental payment to the land owners for the agricultural lands enrolled in the CREP. The annual soil rental rate was calculated based on the soil types of the converted agricultural lands and the associated soil rental rate published by the USDA Farm Service Agency. The soil types of those agricultural lands were compiled from the NRCS soil SSURGO database for Hunterdon County. The program costs were based on the actual average costs of the existing enrolled CREP lands in New Jersey during the period 2004–2007. The average installation costs of the three buffer practices supported by the program were quite different. CP8A (grass waterway) was the most expensive practice with an average installation costs of $40,402 per ha in New Jersey. CP21 (filter strips) was the cheapest buffer practice with an average installation costs of $974 per ha. Average installation costs for CP22 (riparian buffers) was $3,430 per ha. The average maintenance costs were $9.88 per ha for CP8A and CP21 and $14.83 per ha for CP22. The cost-effectiveness was measured by the average controlled runoff potential divided by the average program costs. The cost-effectiveness ratios were used to compare the cost-effectiveness of the riparian buffer strategy to the VSA-based conservation buffer strategy.

Results

Topographic Index Evaluation

The topographic index was evaluated two ways. First, a field visit to the HSAs in watershed as indicated by the high topographic index was conducted with a group of local watershed specialists and soil conservationists in the morning of April 29, 2008 after a light storm in the evening

Fig. 4 A snapshot of the HSAs distribution in Neshanic River watershed
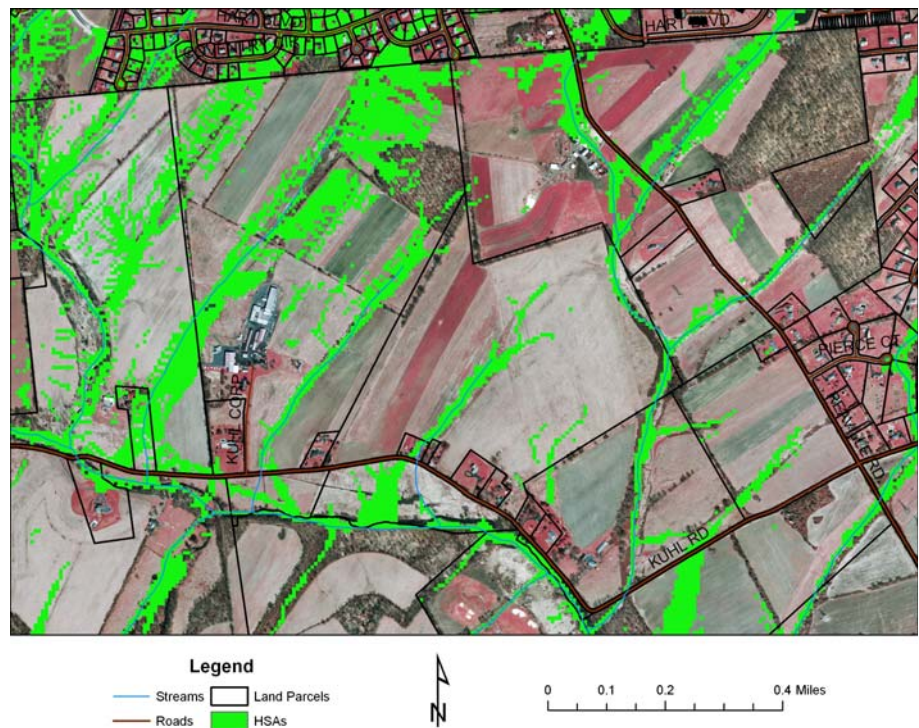
of April 28, 2008. Figure 4 presented a snapshot of the HSAs overlaid with the boundaries of land parcels and the 2002 aerial photography. As shown in Fig. 4, while some HSAs were located along the streams, some were in the middle of the fields. The field visit was to evaluate whether such pattern held especially after the light storm. It was quite convincing that the modified topographic index model gave good indication on runoff generation. Where the predicted HSAs were located, there were clear evidences of accumulated water and/or historical water problems, such as tile drainage pipes or ditches in the particular parts of fields.

Second, the topographic index was incorporated into the VSLF model to predict the stream flow and its baseflow and runoff components in a major portion of the watershed. Following Schneiderman and others (2008), seven parameters that significantly affected watershed hydrology were calibrated using the streamflow observed at the USGS Reaville gage station in the Neshanic River during the period 1992–2000. The original and calibrated values of the seven parameters were presented in Table 1. The Nash-Sutcliffe coefficient measures the goodness-of-fit between model predictions and measured data on the [0, 1] interval. The closer the coefficient is to 1, the better the fit. The coefficients of the calibrated model were 0.18, 0.47 and 0.66 for the daily stream flow, baseflow and runoff and 0.72, 0.73 and 0.70 for monthly stream flow, baseflow and runoff during the calibration period, respectively. The calibrated model was used to predict the stream flow, baseflow and

runoff for the period 2001–2005, which was then verified by the measured stream flow and the derived baseflow and runoff from the measured stream flow in the period. The Nash-Sutcliffe coefficients during the 2001–2005 verification period were 0.41, 0.54 and 0.32 for the daily stream flow, baseflow and runoff and 0.65, 0.60 and 0.48 for monthly stream flow, baseflow and runoff, respectively. Those measurements indicated the VSLF model that incorporates the topographic index reasonably predicts the stream flow in the watershed.

Cost-Effectiveness Assessment

The cost-effectiveness assessment assumed that agricultural lands were enrolled in conservation buffers following the New Jersey CREP. It was assumed that CP22 were installed in the agricultural lands in both the 30.5- and 54-m riparian buffer scenarios under the conventional riparian buffer strategy. The buffer practices in agricultural lands under the VSA strategy would be different. As presented in Fig. 3a, the CSAs for conservation buffers, i.e. the agricultural lands in the HSAs were located not only within the immediate riparian areas of the streams, but also the upland beyond the riparian area. The VSA strategy would assume that CP22 were placed in the agricultural lands in the riparian areas (which is about one-third of the total agricultural lands) while CP21 and/or CP8A were installed in the remaining agricultural lands in the upland areas. Two VSA scenarios were evaluated depending on the conservation buffer

Table 1 Original and calibrated values of seven VSLF model parameters

VSLF parameter	Affected output variable	Original value	Calibrated value
Precipitation correction factor	Stream flow	0.951625	0.934834
SMIN factor	Runoff per (day dormant)	0.170942	0.191209
SMAX factor	Runoff per (day growing)	1.343430	0.952385
Runoff recess coefficient	Runoff	0.412917	0.561485
Melt factor	Stream flow	0.409741	0.979910
Recess coefficient	Base flow	0.061885	0.112644
Bypass coefficient	Stream flow (during lowflow)	0.059946	0.025477

Table 2 Cost-effectiveness of three buffer restoration strategies in Neshanic River watershed, New Jersey

	Units	Riparian buffer strategy[a]		VSA buffer strategy	
		100-foot riparian	177-foot riparian	VSA Scenario I[b]	VSA Scenario II[c]
Riparian or hydrologically sensitive areas	ha	610.34	1074.14	1069.44	1069.44
Agricultural lands	ha	130.06	285.20	280.07	280.07
Average runoff potentials controlled[d]	per ha	1011.50	970.45	1236.68	1236.68
Annual soil rental rate	$/ha	95.30	93.97	98.90	98.90
Signing incentive payments	$/ha	247.11	247.11	247.11	247.11
Installation costs	$/ha	3429.82	3429.82	1792.33	5735.14
Annual maintenance	$/ha	14.83	14.83	11.53	11.53
Average program costs[e]	$/ha	5328.87	5308.84	3695.86	7638.66
Total program costs[f]	$million	0.69	1.51	1.04	2.14
Cost effectiveness[g]	per $	0.19	0.18	0.33	0.16
Cost-effectiveness ratio[h]		0.57	0.55	1.00	0.48

[a] Agricultural lands within the riparian areas are placed into riparian forest buffer (CP22)

[b] Agricultural lands beyond the riparian areas are placed into filter strips

[c] For the agricultural lands beyond the riparian areas, 85% is placed into CP21 and 15% into grassed waterway (CP8A)

[d] 100 times the average runoff potential controlled per grid (i.e. 100 m^2)

[e] 15 times (annual average soil rental rate + annual average maintenance cost) + average signing incentive payment + average installation cost

[f] Average program costs times the total area of agricultural lands

[g] Average runoff potential controlled per hectare divided by the average program cost

[h] Cost-effectiveness divided by 0.33, which is the cost-effectiveness of the VSA Scenario I

specifications in the agricultural lands in the upland areas. VSA Scenario 1 assumed all agricultural lands in the upland areas were placed into CP21 (filter strips). VSA Scenario II assumed 85% of these agricultural lands in the upland areas would be placed into filter strips (CP21) and 15% into grassed waterways (CP8A).

Table 2 presented the total HSAs considered, the acreage of the agricultural lands that were converted to buffers, the average runoff potential, annual soil rental rate, signing incentive payments, installation and annual maintenance costs, average and total NJCREP program costs, cost-effectiveness and cost-effectiveness ratio for the two riparian buffers and two VSA conservation buffer scenarios. The total acreages of the HSAs evaluated were 610, 1,074 and 1,069 ha under the 30.5- and 54-m riparian buffer and VSA conservation buffer scenarios, respectively. The corresponding agricultural lands for conservation buffer

placement were 130, 285 and 280 ha, respectively. The average runoff potentials on those agricultural lands were 1,012 units, 970 units and 1,237 units per ha, which were converted from the average runoff potentials per grid calculated from the topographic index layer. The annual soil rental rates in those agricultural lands were about $95 per ha under the riparian buffer strategy and $99 per ha under the VSA buffer strategy.

Average and total NJCREP program costs were also presented in Table 2. The average costs were $5,329 per ha and $5,309 per ha for the 30.5- and 54-m riparian buffer scenarios and $3,696 per acre and $7,639 per acre for the VSA conservation buffer Scenario I and II, respectively. The corresponding total program costs were $0.69 and $1.51 million for the 30.5- and 54-m riparian buffer scenarios, respectively, and $1.04 and $2.14 million for the VSA conservation buffer Scenario I and II, respectively.

The cost-effectiveness for these four Scenarios was 0.19, 0.18, 0.33 and 0.16 units of runoff potential controlled for every New Jersey CREP program dollar spent, respectively. VSA Scenario I achieved the highest cost-effectiveness of the four cases considered. Other scenarios achieved only half of the cost-effectiveness of the VSA Scenario I. The causes of the low cost-effectiveness were quite different for those scenarios. For the two riparian buffer scenarios, the lower cost-effectiveness was explained primarily by the lower runoff potential in the targeted agricultural lands. Including substantial amounts of expensive conservation buffer practice (CP8A) in the upland areas caused the lower cost-effectiveness for the VSA Scenario II. The CP8A installation costs in New Jersey appeared to be much higher than in other states because there are limited acres enrolled in CP8A in New Jersey CREP and the engineering structure at the end of the grassed waterway that disperses water to streams was very expensive and was considered to be the part of the installation costs. If an installation costs of $9,884 per ha in CP8A, which was more comparative to other states, was used in the analysis, the cost-effectiveness of VSA Scenario II were 0.27 units of runoff potential controlled for every CREP program dollar spent.

Discussion and Conclusion

The science-based GIS procedure proposed here for implementing the VSA-based conservation buffer strategy is a powerful screening tool for identifying the potential sites for the placement of conservation buffers and riparian restoration in watersheds. Contrary to the conventional wisdom that buffers should always be in the riparian areas of existing stream corridors, the patterns of HSAs and CSAs in the Neshanic River watershed show that many upland areas should be targeted for conservation buffer placement because of their active role in generating runoff. The cost-effectiveness assessment clearly showed the economic advantages of the VSA-based buffer placement scenario (VSA Scenario I). However, the economic advantage of the VSA buffer strategy does not warrant targeting conservation buffers to CSAs alone. Placing expensive buffer practices, such as grassed waterways (CP8A), in the targeted CSAs could significantly decrease the cost-effectiveness of the VSA buffer scenario (VSA Scenario II).

The proposed strategy not only has a strong theoretical foundation in VSA hydrology, but also extends several empirical conservation buffer placement strategies discussed in literature. The modified topographic index model is the core of the VSA-based conservation buffer placement strategy. The assessment showed the derived topographic index not only provided reasonable predictions of the spatial pattern of runoff generation, but also helped improve the prediction of runoff amounts in the watershed.

Since there are obvious economic and environmental benefits to target CSAs in watersheds for conservation buffer placement, government programs, such as the New Jersey CREP, should encourage farmers to install and construct conservation buffers in CSAs by providing higher financial and other incentives than on non-CSA lands. Because of the profound impacts of runoff on environmental quality, the spatial pattern of CSAs may provide a basis for prioritizing areas of a watershed for other conservation efforts, in addition to conservation buffer placement. Such efforts include conservation easements, open space and farmland preservation that target CSAs for achieving higher environmental benefits. Various land use planning tools and ordinances can be used to protect and preserve CSAs from urban development. Since the HSAs and CSAs can be spatially presented at high resolution, they can be valuable information for designing on-site BMPs that mitigate the negative environmental impacts of runoff.

The cost-effectiveness assessment presented here was preliminary. This assessment only used the runoff potential controlled by conservation buffers as a proxy for the water quality benefits achieved by conservation buffers. Different conservation buffer practices might achieve different water quality impacts. Future research could incorporate site-specific water quality effects of various buffer practices in a more comprehensive cost-effective assessment framework to optimally allocate buffer practices to various locations in a watershed, thereby achieving the protection of streams and water quality in a cost-effective manner.

References

Agnew LJ, Walter MT, Lembo A, Gérard-Marchant P, Steenhuis TS (2006) Identifying hydrologically sensitive areas: bridging

science and application. Journal of Environmental Management 78:64–76

Ambroise B, Beven K, Freer J (1996) Toward a generalization of the TOPMODEL concepts: topographic indices of hydrological similarity. Water Resource Research 32(7):2135–2145

Berry JK, Delgado JA, Khosla R, Pierce FJ (2003) Precision conservation for environmental sustainability. Journal of Soil Water Conservation 58(6):332–339

Beven K (1997) TOPMODEL: a critique. Hydrological Processes 11(9):1069–1085

Beven KJ, Kirkby MJ (1979) A physically based, variable contributing area model of basin hydrology. Hydrological Science Bulletin 24(1/3):43–69

Bren LJ (1998) The geometry of a constant buffer-loading design method for humid watersheds. Forest Ecology and Management 110(1–3):113–125

Bren LJ (2000) A case study in the use of threshold measures of hydrologic loading in the design of stream buffer strips. Forest Ecology and Management 132(2–3):243–257

Cooper JR, Gilliam JW, Daniels RB, Robarge WP (1986) Riparian areas as filters for agricultural sediment. Soil Science Society of American Journal 60(2):416–420

Daniels RB, Gilliam JW (1996) Sediment and chemical load reduction by grass and riparian filters. Soil Science Society of American Journal 60(1):246–251

de Alwis DA, Easton ZM, Dahlke HE, Philpot WD, Steenhuis TS (2007) Unsupervised classification of saturated areas using a time series of remotely sensed images. Hydrology and Earth System Sciences 11:1609–1620

Dickinson WT, Rudra RP, Wall GJ (1990) Targeting remedial measures to control non-point source pollution. American Water Research Bulletin 26(3):499–507

Dillaha TA, Reneau RB, Mostaghimi S, Lee D (1989) Vegetative filter strips for agricultural nonpoint source pollution control. Transaction of the American Society of Agricultural Engineers 32:491–496

Dosskey MG, Helmers MJ, Eisenhauer DE, Franti TG, Hoagland KD (2002) Assessment of concentrated flow through riparian buffers. Journal of Soil and Water Conservation 57(6):336–343

Dosskey MG, Helmers MJ, Eisenhauer DE (2006) An approach for using soil surveys to guide the placement of water quality buffers. Journal of Soil and Water Conservation 61(6):344–354

Duda AM, Johnson RJ (1985) Cost-effective targeting of agricultural nonpoint-source pollution controls. Journal of Soil & Water Conservation 40(1):108–111

Fox G, Umali G, Dickinson WT (1990) An economic analysis of targeting soil conservation measures with respect to off-site water quality. Canadian Journal of Agricultural Economics 43(1):105–118

Frankenberger JR, Brooks ES, Walter MT, Walter MF, Steenhuis TS (1999) A GIS-based variable source area hydrology model. Hydrological Processes 13:805–822

Freeze RA, Cherry JA (1979) Groundwater. Prentice-Hall, Englewood Cliffs, 604 pp

Gburek WJ, Drungil CC, Srinivasan MS, Needelman BA, Woodward DE (2002) Variable-source-area controls on phosphorus transport: bridging the gap between research and design. Journal of Soil and Water Conservation 57(6):534–543

Gérard-Marchant P, Hively WD, Steenhuis T (2003) Distributed modeling of total dissolved phosphorus loading on a small farm watershed. In: Pfeffer MJ, Van Abs DJ, Brooks KN (eds) Watershed management for water supply systems, AWRA's 2003 international congress. American Water Resources Association, Middleburg, Virginia, TPS-03-2, CD-ROM

Grayson RB, Western AW (1998) Towards areal estimation of soil water content from point measurements: time and space stability of mean response. Journal of Hydrology 207(1–2):68–82

Grayson RB, Blöschl G, Western AW, McMahon TA (2002) Advances in the use of observed spatial patterns of catchment hydrological response. Advances in Water Resources 25(8–12):1313–1334

Heathwaite L, Sharpley A, Gburek W (2000) A conceptual approach for integrating phosphorus and nitrogen management at watershed scales. Journal of Environmental Quality 29(1):158–166

Heathwaite L, Quinn PF, Hewett C (2005) Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. Journal of Hydrology 304(1–4):446–461

Heraty M (1993) Riparian buffer programs: a guide to developing and implementing a riparian buffer program as an urban best management practice. Metropolitan Washington Council of Governments. USEPA Office of Wetlands, Oceans and Watersheds, Washington, DC

Herron NF, Hairsine PB (1998) A scheme for evaluating the effectiveness of riparian zones in reducing overland flow to streams. Australian Journal of Soil Research 36(4):683–698

Hewlett JD, Hibbert AR (1967) Factors affecting the response of small watersheds to precipitation in humid regions. In: Sopper WE, Lull HW (eds) Forest hydrology. Pergamon Press, Oxford, pp 275–290

Johnes PJ, Heathwaite L (1997) Modelling the impact of land use change on water quality in agricultural catchments. Hydrological Processes 11(3):269–286

Lowrance RR, Sharpe JK, Sheridan JM (1986) Long-term sediment deposition in the riparian zone of a coastal plain watershed. Journal of Soil Water Conservation 41:266–271

Lyon SW, Gérard-Marcant P, Walter MT, Steenhuis TS (2004) Using a topographic index to distribute variable source area runoff predicted with the SCS-curve number equation. Hydrological Processes 18(15):2757–2771

Lyon SW, McHale MR, Walter MT, Steenhuis TS (2006) The impact of runoff generation mechanisms on the location of critical source areas. Journal of the American Water Resources Association 42(3):793–804

Maas RP, Smolen MD, Dressing SA (1985) Selecting critical areas for nonpoint source pollution control. Journal of Soil Water Conservation 40(1):68–71

Magette WL, Brinsfield RB, Palmer RE, Wood JD (1989) Nutrient and sediment removal by vegetated filter strips. The Transaction of the American Society of Agricultural Engineers 32(2):663–667

McKergow LA, Weaver DM, Prosser IP, Grayson RB, Reed AE (2003) Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. Journal of Hydrology 270:253–272

Merot P, Squividant H, Aurousseau P, Hefting M, Burt T, Maitre V, Kruk M, Butturini A, Thenail C, Viaud V (2003) Testing a climato-topographic index for predicting wetlands distribution along an European climate gradient. Ecological Modeling 163(1–2):51–71

Moore ID, Thompson JC (1996) Are water table variations in a shallow forest soil consistent with the TOPMODEL concept? Water Resource Research 32(3):663–669

Moore ID, Grayson RB, Ladson AR (1991) Digital terrain modelling: a review of hydrological, geomorphological, and biological applications. Hydrological Processes 5(1):3–30

Moore ID, Norton TW, Williams JE (1993) Modeling environmental heterogeneity in forested landscapes. Journal of Hydrology 150:717–747

New Jersey Department of Environmental Protection (NJDEP) (2008) The 2008 New Jersey integrated water quality monitoring and assessment report (Draft). Water Monitoring and Standards, Bureau of Water Quality Standards and Assessment, NJDEP.

http://www.state.nj.us/dep/wms/bwqsa/draft_2008_integrated_report.pdf

New Jersey Water Supply Authority (NJWSA) (2002) Raritan Basin watershed management plan. The Watershed Protection Unit, NJWSA. http://www.raritanbasin.org/

O'Loughlin EM (1986) Prediction of surface saturation zones in natural catchments by topographic analysis. Water Resource Research 22:794–804

Pionke HB, Gburek WJ, Sharpley AN (2000) Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. Ecological Engineering 14:325–335

Polyakov V, Fares A, Ryder MH (2005) Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: a review. Environmental Reviews 13(3):129–144

Prato T, Shi H (1990) A comparison of erosion and water pollution control strategies for an agricultural watershed. Water Resources Research 26:199–205

Qiu Z (2003) A VSA-based strategy for placing conservation buffers in agricultural watersheds. Environmental Management 32(3):299–311

Reiser RG (2004) Evaluation of streamflow, water quality and permitted and nonpermitted loads and yields in the Raritan River Basin, New Jersey, water years 1991–98. Water-resources investigations report 03-4207. U.S. Geological Survey, West Trenton

Ribaudo MO, Horan RD, Smith ME (1999) Economics of water quality protection from nonpoint sources: theory and practices. Agricultural economics report no. 782. p 106. Resource Economics Division, Economic Research Service, U.S. Department of Agriculture

Rodhe A, Seibert J (1999) Wetland occurrence in relation to topography: a test of topographic indices as moisture indicators. Agricultural and Forest Meteorology 98:325–340

Schneiderman EM, Steenhuis TS, Thongs DJ, Easton ZM, Zion MS, Neal AL, Mendoza GF, Walter MT (2007) Incorporating variable source area hydrology into curve number based watershed loading functions. Hydrological Processes 21:3410–3430

Schneiderman EM, Pierson D, Zion MS (2008) Instruction for using the GWLFVSA model with the Vensim model reader. New York City Water Supply Modeling Group, New York City Department of Environmental Protection, Kingston

Seibert J, Bishop KH, Nyberg L (1997) A test of TOPMODEL's ability to predict spatially distributed groundwater levels. Hydrological Processes 11(9):1131–1144

Tarboton DG (1997) A new method for the determination of flow directions and upslope areas in grid digital elevation models. Water Resources Research 33(2):309–319

Tarboton DG (2005) Terrain analysis using digital elevation models (TauDEM). http://hydrology.neng.usu.edu/taudem/

Tomer MD, James DE, Isenhart TM (2003) Optimizing the placement of riparian practices in a watershed using terrain analysis. Journal of Soil and Water Conservation 58(4):198–206

Walter MT, Walter MF, Brooks ES, Steenhuis TS, Boll J, Weiler K (2000) Hydrologically sensitive areas: variable source area hydrology implications for water quality risk assessment. Journal of Soil and Water Conservation 55(3):277–284

Walter MT, Brooks ES, Walter MF, Steenhuis TS, Scott CA, Boll J (2001) Evaluation of soluble phosphorus loading from manure-applied fields under various spreading strategies. Journal of Soil and Water Conservation 56(4):329–335

Walter MT, Steenhuis TS, Mehta VK, Thongs D, Zion M, Schneiderman E (2002) A refined conceptualization of TOPMODEL for shallow-subsurface flows. Hydrological Processes 16(10):2041–2046

Xiang WN (1993) A GIS method for riparian water quality buffer generation. International Journal of Geographical Information Systems 7(1):57–70