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Modeling watershed-scale effectiveness of agricultural best management practices to reduce phosphorus loading

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ABSTRACT

Planners advocate best management practices (BMPs) to reduce loss of sediment and nutrients in agricultural areas. However, the scientific community lacks tools that use readily available data to investigate the relationships between BMPs and their spatial locations and water quality. In rural, humid regions where runoff is associated with saturation-excess processes from variable source areas (VSAs), BMPs are potentially most effective when they are located in areas that produce the majority of the runoff. Thus, two critical elements necessary to predict the water quality impact of BMPs include correct identification of VSAs and accurate predictions of nutrient reduction due to particular BMPs. The objective of this research was to determine the effectiveness of BMPs using the Variable Source Loading Function (VSLF) model, which captures the spatial and temporal evolutions of VSAs in the landscape. Data from a long-term monitoring campaign on a 164-ha farm in the New York City source watersheds in the Catskills Mountains of New York state were used to evaluate the effectiveness of a range of BMPs. The data spanned an 11-year period over which a suite of BMPs, including a nutrient management plan, riparian buffers, filter strips and fencing, was installed to reduce phosphorus (P) loading. Despite its simplicity, VSLF predicted the spatial distribution of runoff producing areas well. Dissolved P reductions were simulated well by using calibrated reduction factors for various BMPs in the VSLF model. Total P losses decreased only after cattle crossings were installed in the creek. The results demonstrated that BMPs, when sited with respect to VSAs, reduce P loss from agricultural watersheds, providing useful information for targeted water quality management.

1. Introduction

Farmers implement best management practices (BMPs) in agricultural watersheds to prevent excessive loss of pollutants from fields and subsequent entry into waterways during storm events. Excessive P levels in streams and waterways impair water quality and ecosystem function in P-limited waters (Sharpley et al., 2004). High P levels have been implicated in problems ranging from bacterial outbreaks (National Research Council, 2000) to algal blooms (Novotny, 2003), which foster eutrophication and prevent the use of water that would otherwise support fisheries, industry, recreation or consumption (Sharpley et al., 2003; Carpenter et al., 1998).

In agricultural watersheds, nonpoint source pollution tends to dominate pollutant loss. As the name suggests, nonpoint source areas can be difficult to characterize due to spatial and temporal heterogeneity. However, many potential sources of nonpoint source pollution have been identified, including storm sewers, urban runoff and drainage ditches, agricultural and silvicultural runoff and mining activities (Freeman, 2000). A number of BMPs have been proposed for urban, industrial, septic systems and agricultural landuses to reduce nutrient levels in lakes and streams. Agricultural sources are responsible for 46% of the sediment, 47% of total P (TP) and 52% of total nitrogen (N) discharged into US waterways, making agricultural runoff a major contributor of pollutants to aquatic systems (Allan, 1995). In many agricultural watersheds, runoff from farm fields is responsible for the majority of P loss, which BMPs may

Abbreviations: Best Management Practices, BMP; Dissolved Phosphorus, DP; Particulate Phosphorus, PP; Total Phosphorus, TP; Variable Source Areas, VSAs.

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help to reduce (Heathwaite et al., 2000; Novotny, 2003). In this study, we focus specifically on the effectiveness of BMPs in reducing nonpoint source P pollution from agricultural areas.

2. Best management practices

Historically, BMPs were designed and implemented to reduce soil erosion (Walter et al., 1979; Clark et al., 1985; Schaller and Bailey, 1983). Beginning in the 1970s, attempts were made to use BMPs to reduce the load of sediment entering waterways, which were assumed to reduce simultaneously other types of pollution such as the dissolved forms of N and P (Walter et al., 1979). However, while these BMPs were effective at reducing sedimentation, they did not prevent dissolved pollutants from running off these fields and polluting water bodies (Walter et al., 1979, 2000, 2003; Novotny, 2003). Presently, soil and nutrient management BMPs that focus on controlling dissolved nutrients are being implemented along with traditional erosion control BMPs in many agricultural watersheds (McKell and Peiretti, 2004). These new BMPs can be Structural or Managerial (e.g., Source Impact) in nature (Novotny, 2003). Structural BMPs either alter hydrologic pathways away from P source areas or decrease the dissolved P (DP) contribution from these areas. Examples of structural BMPs that alter the hydrologic pathways include interceptor drainage ditches or subsurface tile drains that reduce runoff losses, or manure storage ponds that allow farmers to decrease spreading during wet periods thus reducing available P. Source Impact BMPs are used to decrease nutrient export from high runoff producing areas by reducing or redistributing the potential P load to less frequently saturated areas in the watershed. For example, the farmer might follow a Nutrient Management Plan (NMP), which prescribes spreading less manure on frequently saturated areas and spreading more manure in upland or drier areas. Thus, if less manure is spread on areas with a greater propensity to saturate, then when a storm event occurs, these high runoff producing areas will have less manure available for transport and thus carry a lower concentration of DP.

Several studies have reported BMP effectiveness values for reducing P loads (both particulate P (PP) and dissolved forms). The proportion by which a BMP decreases nutrient loading is called a “BMP reduction factor.” These factors are different for various nutrients as well as for various BMPs. Table 1 shows the BMP reduction factors for P from several studies. The literature contains a wide range of factors associated with each BMP, perhaps not surprising since a BMP’s effectiveness can be measured at numerous scales, e.g., plot-, field- or farm-level studies (Lee et al., 2000a; Magette et al., 1989) as well as watershed level studies (Gitau et al., 2004; Inamdar et al., 2001; Brannan et al., 2000). Moreover, because specificity in terms of landscape, topography, landuse, hydrology and meteorology vary, the effectiveness of BMPs likewise varies. The ranges reported in Table 1 reflect studies of BMPs implemented both as a single practice and as a combination of practices.

Despite the variability in BMP effectiveness (Table 1), some trends are consistent. Runoff amounts are usually only slightly affected, if at all, by BMP implementation, especially on shallow soils (Haith and Loehr, 1979). Indeed, Bishop et al. (2005) report no statistically significant changes in streamflow or runoff response after BMP implementation in a Catskill Mountain (NY) watershed. Vegetative filter strips (Magette et al., 1989; Schmitt et al., 1999; Kim et al., 2006) and riparian buffers (Lee et al., 2000a) generally decrease the amount of TP reaching streams and waterways, however it is not readily clear if the reduction is due to biophysical remediation in the filter strip or simply from the decrease in available P due to the reduction in manure spreading in the near-stream area. Conservation tillage can reduce the P reaching the waterways (Laflen and Tabatabai, 1984), especially where snowmelt greatly increases spring runoff (Hansen et al., 2000) by increased soil retention from vegetation. NMPs are effective in reducing DP losses in surface waters (Hamlett and Epp, 1994; Bishop et al., 2005) by prescribing the timing and spatial distribution of nutrient applications to avoid wet periods or areas. Structural practices, such as barnyard improvement, reduce P loss less than when BMPs were implemented on cropland (Brown et al., 1989), but manure lagoons allow specific, favorable spreading schemes resulting in reductions of PP and DP (Brannan et al., 2000).

The large range in BMP effectiveness makes it difficult for watershed managers to target BMPs on farms lacking long-term monitoring data. In these cases, to reduce the variability and obtain better estimates of the effectiveness of BMPs, the potential for runoff and source reduction should be calculated separately for specific locations and then combined in a load estimate, which does not require much more data than is typically available in the planning process. However, consideration should be given to the spatial location of BMPs in the planning process. Phosphorus transport via runoff is influenced by the type of fertilizer or manure, soil type, slope, hydrology and climate among other factors (McDowell et al., 2001). In humid, well-vegetated, topographically steep landscapes with shallow, permeable soils, underlain by a restricting layer, runoff is the dominant mechanism of P transport. In areas characterized by these conditions, runoff originates from saturated areas on the landscape, often called variable source areas (VSAs) (Dunne and Leopold, 1978), that expand and contract seasonally, and are thus difficult to predict (Walter et al., 2000). Few studies have explored BMP effectiveness on landscapes characterized by VSAs at the watershed or basin scale. The objective of this study is to apply a recently developed water quality model that incorporates VSA hydrology to model the changes in dissolved and total P loads from an agricultural watershed in which BMPs have been installed. The research explores the hypothesis that BMPs designed for DP load reduction are effective and do not affect the integrated hydrologic response of the watershed.

3. Models and tools

Several models can be used to simulate the effectiveness of BMPs at the farm or watershed scale, including: the Generalized Watershed Loading Function (GWLF) model (Haith and Shoemaker,

Table 1
Range of BMP reduction factors reported in the literature

BMP	Reduction ranges (%)	References
Buffer	19–99	Gitau et al. (2005, 2004), Novotny (2003), Lee et al. (2000a), Peterjohn and Correll (1984)
NMP	−66–94	Gitau et al. (2005, 2004), Novotny (2003), Osei et al. (2000), Schuman et al. (1973)
Strip cropping	20–93	Gitau et al. (2005, 2004), Novotny (2003), Haith and Loehr (1979)
Crop rotation	30–75	Gitau et al. (2005), Clark et al. (1985), Haith and Loehr (1979)
Filter strips	−56–59	Gitau et al. (2005), Novotny (2003), Eghball et al. (2000), Novotny and Olem (1994), Schmitt et al. (1999)
Culvert crossing	11–38	Flores-Lópes, submitted for publication
Animal waste management	−117–40	Gitau et al. (2005), Brannan et al. (2000), Novotny and Olem (1994), Novotny (2003)
Barnyard management	5–81	Gitau et al. (2005), Brown et al. (1989)

Positive numbers indicate load decreases while negative numbers indicate load increases.

1987), the Soil and Water Assessment Tool (Arnold et al., 1998), the Erosion Productivity Impact Calculator (EPIC) model (Williams et al., 1984), the Hydrologic Simulation Program Fortran (HSPF) model used by the Environmental Protection Agency (Donigian et al., 1984), the Spatially Referenced Regressions on Watershed Attributes (SPARROW) model (Smith et al., 1997), the Soil Moisture Distribution and Routing (SMDR) model (Zollweg et al., 1996; Frankenberger et al., 1999) and the Long-Term Hydrologic Impact Assessment (L-THIA) model (Bhaduri et al., 2000). GWLF uses readily available input data and has been used for predicting nutrient loads (Schneiderman et al., 2002), in evaluation of BMPs (Cryer et al., 2001; Santhi et al., 2003; NYCDEP, 2002) as well as integrated with economic models (Evans et al., 2002) for cost analyses. GWLF is primarily suitable for regional modeling efforts and was used for analysis in the Hudson River watershed (Howarth et al., 2000), the NYC water supply watersheds (Schneiderman et al., 2002), and the Chesapeake Bay watershed (Lee et al., 2000b). Since GWLF is a lumped model, the spatial distribution of runoff and pollution source areas has not been tested, and its use in the delineation of saturated areas is problematic in watersheds characterized by VSA hydrology (Walter et al., 2003; Schneiderman et al., 2007). To address this problem, the Soil Conservation Service-Curve Number (SCS-CN) approach to predict runoff in GWLF was re-conceptualized based on the framework of Steenhuis et al. (1995) to capture VSAs. The new model, termed the Variable Source Loading Function (VSLF) model (Schneiderman et al., 2007) predicts and spatially distributes watershed runoff response to rainfall, using a soil wetness index and landuse classes to define hydrologic response units (HRUs) (Fig. 1), as opposed to the intersection of landuse and soil type in GWLF. In this study, we extend this application to modeling BMPs applied on various parts of a farm-watershed, with respect to the VSAs. As in the original GWLF, VSLF predicts nutrient loads from agricultural watersheds using export coefficients.

3.1. Hydrological model description

We briefly outline the model here, for more information one is referred to Steenhuis et al. (1995) and Schneiderman et al. (2002, 2007). To simulate overland flow, VSLF assumes that all rain falling on unsaturated soil infiltrates while all rain falling on saturated areas becomes runoff. Thus, the rate of runoff generation will be proportional to the fraction of the watershed that is effectively saturated, A_f, which can then be written as (Steenhuis et al., 1995):

$$A_f = \frac{\Delta Q}{\Delta P_e} \tag{1}$$

where ΔQ (m) is saturation-excess runoff depth or, more precisely, the equivalent depth of excess rainfall generated during a time period over the whole watershed area, and ΔP_e (m) is the incremental depth of precipitation during the same time period. In VSLF, saturation-excess runoff, Q in Eq. (1), includes both overland flow where soil is saturated at the surface and rapid subsurface flow due to a perched water table intersecting an upper soil layer where preferential flow exists (Schneiderman et al., 2007). By differentiating the standard SCS-CN equation (USDA-SCS, 1972) with respect to P_e, the fractional contributing area, A_f, for a storm can be written as (Steenhuis et al., 1995):

$$A_f = 1 - \frac{S_e^2}{(P_e + S_e)^2} \tag{2}$$

where S_e (m) is the depth of effective available storage in the watershed, or the available volume of retention in the watershed when runoff begins and varies with antecedent moisture conditions as outlined in Schneiderman et al. (2007). Runoff occurs from areas that have a local storage, σ_e, (mm), less than P_e, according to Eq. (2). Thus, replacing P_e with σ_e results in a relationship for the fraction of the watershed area, A_s, that has a local storage less than

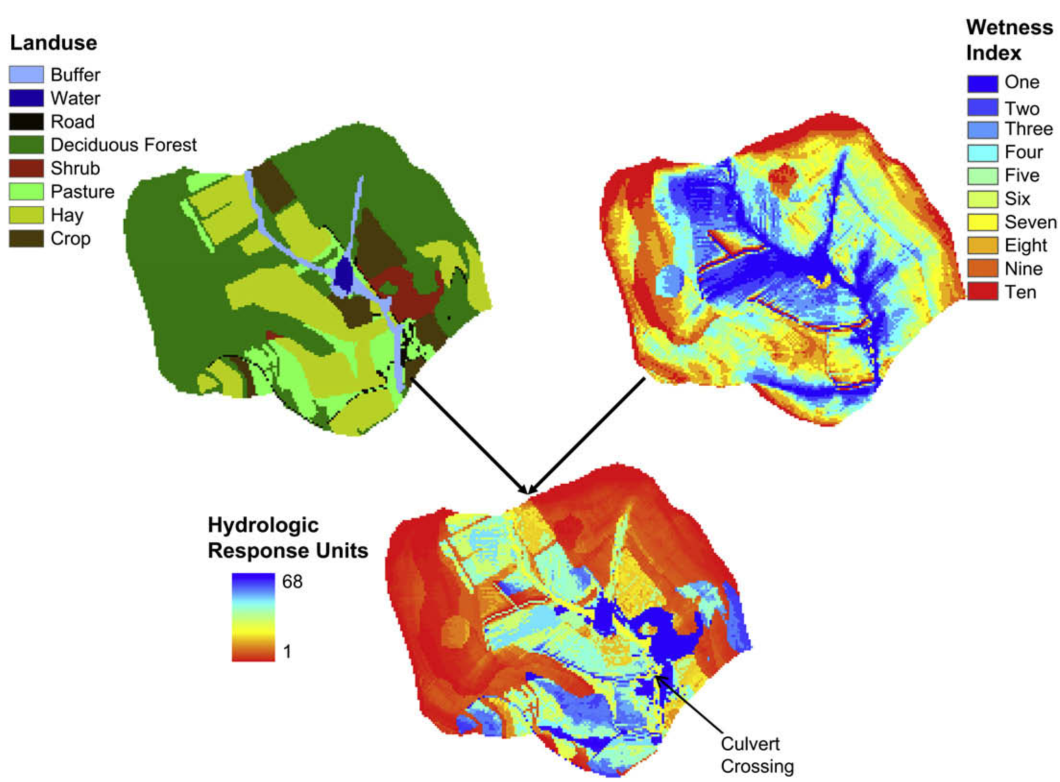

Fig. 1. The top left map shows the landuses on the study watershed. The top right map shows the distribution of wetness indices in the study watershed. The bottom map is a composite showing the hydrologic response units (HRUs) (landuse and wetness index combinations) on the study watershed.

or equal to σ_e for a given overall watershed storage of S_e (Schneiderman et al., 2007):

$$A_s = 1 - \frac{S_e^2}{(\sigma_e + S_e)^2} \tag{3}$$

Solving for σ_e results in the maximum effective local soil moisture storage within any fraction, A_s, of the overall watershed area (Schneiderman et al., 2007):

$$\sigma_e = S_e\left(\sqrt{\frac{1}{(1-A_s)}} - 1\right) \tag{4}$$

For a given storm event with precipitation P, the location of the watershed that saturates first ($A_s = 0$) has local storage $\sigma_e = 0$, and runoff from this location will be $P - I_a$. Successively drier locations retain more precipitation and produce less runoff according to the moisture – area relationship in Eq. (4). Thus, runoff from each area, A_f, is the amount of precipitation in excess of the local soil water deficit (Schneiderman et al., 2007):

$$Q_i = P_e - \sigma_i \quad \text{for } P_e > \sigma_I, \tag{5a}$$

$$Q_i = 0 \quad \text{for } P_e \leq \sigma_I \tag{5b}$$

Once runoff is calculated it is then spatially distributed according to an area-weighted soil topographic index (STI) (Beven and Kirkby, 1979; Sivapalan et al., 1987):

$$\mathrm{STI} = \ln\left(\frac{\alpha}{Kd\tan\beta}\right) \tag{6}$$

where α (m) is upslope contributing area per unit of contour line, K (m d^{-1}) is the soil's hydraulic conductivity, d (m) is the depth of the soil profile above the restricting layer and tan(β) is the topographic slope.

Recession flow (baseflow) was simulated as a linear reservoir with infiltrating water in excess of field capacity as an input and baseflow, $Q_{\mathrm{BF}(t)}$, ($\mathrm{m}^3\,\mathrm{d}^{-1}$) as output (Haith et al., 1992; Schneiderman et al., 2002):

$$Q_{\mathrm{BF}(t)} = R_{s(t)}\left(1 - \exp^{(-\alpha\Delta t)}\right) \tag{7}$$

where $R_{s(t)}$ is the storage in the groundwater reservoir at time t ($\mathrm{m}^3\,\mathrm{d}^{-1}$) and a (d^{-1}) is the recession coefficient, a property of the aquifer, and can be calibrated from the baseflow recession curve (Easton et al., 2007).

Snowmelt occurs when the mean daily temperature is above 0 °C, and is estimated using the U.S. Army Corps of Engineers (1960) temperature index method. An adiabatic lapse rate (0.00636 °C m^{-1}) is used to adjust temperatures based on the elevation, allowing higher, colder areas to retain snowpack longer than lower, warmer areas (Schneiderman et al., 2007; Easton et al., 2007).

Schneiderman et al. (2007) validated VSLF for predicting soil moisture levels by comparing VSLF predictions against those made by a more robust physically based model, SMDR (Gérard-Marchant et al., 2006), and found that, in general, the standard error between the predicted saturated area using VSLF and SMDR varied by less than 5%. Additionally, they compared predicted soil moisture data from several transects in the study watershed (from Frankenberger et al., 1999) and found VSLF to capture the distribution well.

3.2. Phosphorus model description

While original export coefficient based nutrient routines were based on a linear relationship between nutrient loss and surface runoff, Q (Reckhow et al., 1980; Sorrano et al., 1996), NYCDEP (2006) found that using a power function resulted in a significantly better fit with observed data. Thus, in VSLF export coefficients are adjusted by a runoff coefficient, λ (dimensionless), calculated as:

$$\lambda = \left(A + BQ_R^C\right) \tag{8}$$

where A, B and C are calibration parameters invariant across the watershed and thus independent of a particular HRU. The nutrient loss in surface runoff per unit area for a particular HRU, $L_{R,\mathrm{HRU}}$ (kg ha^{-1}), is then estimated as the product of an export coefficient, $\epsilon_{\mu C}$ (kg $\mathrm{ha}^{-1}\,\mathrm{m}^{-1}$), a BMP reduction factor, κ (dimensionless), the runoff coefficient, λ, from Eq. (4) and the depth of surface runoff for the HRU, $Q_{\mathrm{R,HRU}}$ (m), *viz*:

$$L_{R,\mathrm{HRU}} = \epsilon_{\mu,C}\kappa\lambda Q_{R,\mathrm{HRU}} \tag{9}$$

The export coefficient, $\epsilon_{\mu C}$, expresses the amount of nutrient loss per hectare of land per unit amount of runoff and is a function of a particular soil, subscript μ, and crop type, subscript C, but is independent of wetness class. The BMP reduction factor is used for calculating the reduction in the nutrient load for a single or a set of BMPs, compared to zero BMPs installed, and is only a function of the BMPs. These BMP reduction factors are discussed in detail later.

The VSLF model was selected for predicting the impact of BMPs for two main reasons: (1) it uses data that is generally available to planners and (2) it has been shown to simulate runoff production and saturated areas reliably. Here, it is tested on a small watershed in the Catskill Mountains, employing BMP reduction factors found in the literature to simulate both structural and source impact BMPs.

4. Study site

The study site is a 164-ha dairy farm-watershed in the Cannonsville basin in Delaware County, NY. The climatic region is humid continental with an average temperature of degrees 8 °C, and average precipitation of 1120 mm yr^{-1} (NCDC, 2005). The topography is steep with winter snow accumulation and spring snowmelt dominating the hydrology (Bishop et al., 2005). The stream exhibits a rapid response to rainfall with high flow during extended storms and minimal flow during the dry summers. The farm has 102 milking cows and over 70 heifers. Landuse on the farm before BMP implementation included deciduous forest (53.8%), brush/shrub (2.8%), pasture (9.7%), permanent hay (24.4%), cropland (7.5%), barnyard (0.1%), rural road (1.0%) and water (0.7%); after BMP implementation an additional landuse of buffer (3.2%) was added which meant that the other landuses had to be adjusted accordingly (Table 2, Fig. 1). Stream discharge and water quality data were first recorded from 1993 to 1995, before BMPs were implemented on the farm. Water quality monitoring was suspended from 1995 to 1996, when structural BMPs were installed and the nutrient management plan (NMP)

Table 2
Watershed landuse in the pre- and post-BMP periods

Landuse	Total area (ha)	% Landuse	Total area (ha)	% Landuse
	Pre-BMP		Post-BMP	
Buffer	n/a	n/a	5.2	3.2
Water	1.1	0.7	1.1	0.7
Road	1.7	1.0	1.7	1.0
Deciduous forest	88.4	53.8	86.0	52.3
Brush	4.6	2.8	4.5	2.7
Pasture	16	9.7	14.5	8.8
Hay	40.1	24.4	39.1	23.8
Crop	12.3	7.5	12.1	7.4
Barnyard	0.2	0.1	0.2	0.1
Total	164	100.0	164	100.0

was initiated. Exceptions were precision feeding, which started in December 1999 (Cerosaletti et al., 2004) and the additional cattle crossing built in 2001 (labeled in Fig. 1). Water quality monitoring resumed from late 1996 to 2004 (Bishop et al., 2004). The one-farm study watershed drains into the West Branch of the Delaware River and ultimately into the Cannonsville Reservoir which is part of the New York City (NYC) drinking water supply system.

Daily streamflow was measured by a gauge at the watershed outlet. Observed DP concentrations were derived from flow-weighted sampling at the watershed outlet, as described in Bishop et al. (2003, 2005). Dissolved P is defined as molybdate reactive orthophosphate in filtered (45 μm) Kjeldahl digested water samples. Particulate P was computed as the difference between TP and DP. Total P concentrations were determined using sulfuric acid–ammonium digestion with colorimetric assay employing ascorbic acid–molybdate complex method (USEPA, 1979). A detailed description of the sampling protocol, watershed layout and hydrology is given in Bishop et al. (2003, 2005), Gérard-Marchant et al. (2006), Hively et al. (2005, 2006) and Hively (2004).

The agricultural BMPs installed on this farm were distributed across the fields, generally in combination with other management practices such as the NMP. There are 31 agricultural fields managed or owned by the farm within the watershed. On these 31 fields, 26 types of BMPs are distributed, with 80 discrete BMPs across the farm-watershed area. Several fields have more than one BMP on them. NMPs often include a variety of individual BMPs, such as manure spreading and storage, roof water management and animal trail improvements, to name a few.

5. Materials and methods

5.1. Methods

In this paper, we describe a procedure using VSLF to simulate the impact of BMPs in reducing P loads from the study watershed. While the model was run on a daily basis, all analysis was performed on an event basis. An event is defined as a period that lasts from the first day of streamflow hydrograph rise until the beginning of the next event. Thus, it includes both the storm event (runoff) and the return to baseflow after an event.

5.2. Model input data

Weather: During the growing season (e.g., air temperature greater than 1 °C) precipitation was measured in the watershed at 10-min intervals. Daily minimum and maximum temperatures and winter snowfall and precipitation were measured daily at a Northeast Regional Climate Center (NRCC) approximately 15 km southwest of the watershed outlet in Delhi, New York (NCDC, 2005). Temperatures were corrected using an adiabatic lapse rate as described above. Potential evapotranspiration was estimated from climate data using the Penman–Monteith method. All weather data was aggregated to daily values.

Base Maps and Associated Input Data: While VSLF does not explicitly require any distributed data to run, it is needed to compile input data and meaningfully display results. Three primary maps were used: a digital elevation model (DEM); the soil topographic index, STI calculated with Eq. (3); and a landuse map with associated descriptive data. The digital elevation model (DEM) of the basin was obtained from the New York State Department of Environmental Conservation with $10\ \mathrm{m} \times 10\ \mathrm{m}$ horizontal and 0.1 m vertical resolutions. Drainage ditches, stream channel modifications and barnyard improvements not captured by the original DEM were digitized and the DEM modified by adjusting the elevations accordingly (Fig. 1). Digital landuse maps in vector format from 2001 were obtained from the NYCDEP for Delaware County. Broad scale landuse classifications were adapted from the National Land Cover Data center, and can be seen in Fig. 1. We created a STI map using GRASS GIS (U.S. Army CERL, 1997). The associated soils properties for the STI were extracted from the SSURGO database (USDA-NRCS, 2000) and look up tables were linked to the map. For each cell the STI was estimated with Eq. (3). The α and β in Eq. (3) were taken directly from the DEM while K and d in Eq. (3) were extracted from the SSURGO soil database. We lumped the watershed's STI into 10 equal area intervals ranging from 1 to 10, with index class 10 covering the 10% of the watershed area with the lowest STI (*i.e.*, lowest propensity to saturate) and index class 1 containing the 10% of the watershed with the highest STI (*i.e.*, highest propensity to saturate) (Fig. 1). These wetness index classes were intersected with the landuse to create HRUs (Fig. 1).

The STI provides a means of distributing model predictions along a gradient of the propensity of a certain area to saturate and thus contribute to runoff in a storm event. The index also serves as a means to map predictions from VSLF. In VSLF, runoff is created at the wetness index level, but nutrient loss is linked to the intersection of wetness index and landuse, or the HRU. Therefore, while runoff losses from the same wetness index class are the same irrespective of landuse, P loss may vary among landuses within a single wetness index class. On the study site, there are theoretically 80 possible HRUs, determined by the ten soil index classes and eight landuses, though only 68 exist on the farm (Fig. 1).

5.3. Modeling best management practices

In this model application, a single HRU can consist of several non-contiguous areas (e.g., the same HRU can be located in several areas of the watershed provided it contains the same combination of landuse and wetness class). We assumed that when a BMP is implemented on a particular area, the reduction in nutrient load, β, is always the same for structural BMPs independent of the location in the watershed. For managerial BMPs the coefficient for the phosphorus input is altered from one wetness class to another; the reduction factors are dependent on the wetness class but with the constraint that there is no net gain, *i.e.*, the sum is adjusted for amount of land involved. In all cases the nutrient load reduction for a particular HRU was adjusted by the proportion of land, f, over which the BMP was implemented. While this is clearly a simplification, little error is added since the study watershed is relatively small (164 ha) and, in fact, most HRUs are contiguous. However, in applications to larger watersheds with more diverse features, the HRU can be redefined to include more wetness classes, landuses or BMPs.

For one BMP installed on a particular HRU the BMP reduction factor is equal to:

$$k = 1 - f\beta \tag{10}$$

where β is the reduction in nutrient load if the BMP was installed over the whole area and f is the fraction of the landuse over which the BMP was installed in the HRU. For more than one BMP installed on the same HRU, BMP reduction factors, κ, were adjusted according to two methods. With Method I the reduction factor of the set of BMPs is determined by summing the reduction factors of all the BMPs installed in the particular HRU, *viz*:

$$\kappa = 1 - \sum_{i=1}^{N}\left(f_{i,\mathrm{HRU}}\beta_i\right) \tag{11}$$

where the subscript i refers to a single specific BMP and N is the number of BMPs. Method II uses a compounding method derived by Palace et al. (1998). The proportion of a landuse over which the BMP was applied is calculated as in Method I. Then the total load reduction for a BMP on a specific HRU is calculated as the product of the loading reductions for each individual BMPs:

$$\kappa = \prod_{i=1}^{N}\left(1-\left(f_{i,\mathrm{HRU}}\beta_i\right)\right) \tag{12}$$

while it is not entirely clear how BMPs interact in practice, a methodology is needed to quantify their effect, thus Methods I and II were chosen because no standard way to measure the combined effect of different BMPs has proven to be correct, but variations of these methods are being tested. The load of DP or PP in surface runoff for each HRU is calculated with Eq. (9).

Similar to the runoff load, the baseflow load, L_{BF} (kg d^{-1}) is estimated as:

$$L_{\mathrm{BF}} = (1-\kappa_{\mathrm{BF}})C_{\mathrm{BF}}Q_{\mathrm{BF}} \tag{13}$$

where Q_{BF} is baseflow at the watershed outlet, (m^3 d^{-1}), κ_{BF} is the BMP reduction factor for baseflow and C_{BF} is the concentration in the baseflow (kg m^{-3}). The total load in the watershed is calculated as the sum of the load in runoff and baseflow.

In assigning parameter values to the BMP practices, we assumed that all structural BMPs (with the exception of the drainage ditches) such as buffer strips, crop rotation and stream crossings, reduce the P load per unit runoff amount independent of their position in the landscape (*i.e.*, they were assigned all the same κ values). This does not mean, for example, that buffer strips are less effective in decreasing loads in high runoff areas near the river banks. We assumed that they have the same effectiveness per unit runoff amount, and thus areas prone to higher runoff losses (*i.e.*, near-stream areas) will reduce pollutant loads more on an absolute basis. To account for the impact of drainage ditches, the DEM was modified in these areas, changing the distribution of wetness indices in the model, such that more runoff accumulates in the ditches. Management BMPs are designed so that they remove the source of pollutants from the high runoff producing areas (*i.e.*, less manure is spread on lower wetness indices) and thus accordingly increase the pollutant load on the fields with a lower propensity to saturate (*i.e.*, more manure is spread on the higher wetness indices). In this case the "mass balance" was preserved by keeping the average κ constant over the entire agricultural landuse group (pasture, permanent hay and row crop) where the NMPs were implemented. The particular BMP reduction factor used for each HRU landuse is reported in Table 3, and corresponds to the κ values in Eqs. (11)–(13). The values fit within the ranges from the literature

Table 3
Reduction in nutrient loading (β) for Structural and Management BMPs by landuse and soil wetness index, as used in Eqs. (10)–(13)

Landuse	BMP	Reduction factors			
		Soil wetness index			
		Without respect to soil wetness index	One–three	Four–seven	Eight–ten
Buffer	Buffer	n/a	0.80	0.80	0.80
Pasture	NMP	n/a	0.45	0.35	−1.32
	Culvert crossing	n/a	0.36	0.36	0.36
Hay	NMP	n/a	0.60	0.35	−1.07
Crop	NMP	n/a	0.35	0.15	−0.55
	Crop rotation	n/a	0.50	0.50	0.50
	Strip cropping	n/a	0.50	0.50	0.50
	Filter strips	n/a	0.30	0.30	0.30
Barnyard	NMP	0.15	n/a	n/a	n/a
	Animal waste	0.15	n/a	n/a	n/a
	Barnyard runoff	0.15	n/a	n/a	n/a
	Barnyard outlet	0.15	n/a	n/a	n/a

Table 4
Nash-Sutcliffe efficiencies for streamflow, dissolved P (DP) and total P (TP) calibrated to the 1993–1995 pre-BMP period

Calibration to pre-BMP levels	Streamflow	DP	TP
	Efficiency (E)		
1993–1995	0.88	0.87	0.80
1997–1999	0.87	−0.80	0.54
2000–2004	0.80	0.14	0.33
1997–2004	0.85	0.08	0.46

(Table 1). In summary, the reductions varied by wetness index class for Source Impact BMPs but did not vary by wetness index class for Structural BMPs.

The fraction of a landuse over which a BMP is effective corresponds to f in Eqs. (10)–(12). Crop rotation and strip cropping were applied on 50% of the crop landuse, and filter strips were applied on 5% of the crop landuse. The pasture landuse varied in wetness, but the wetness index classes correspond to the near-stream areas, which is where fencing and culvert crossings were implemented (5% of the pasture landuse). Barnyard BMPs, including animal waste management and barnyard runoff control, were assumed to be implemented over 50% of the barnyard.

5.4. Hydrology calibration

The method of Schneiderman et al. (2002) was followed for calibration of the hydrology model. Briefly, the NYCDEP (2004) developed and applied a methodology for calibrating the watershed S_{e} in VSLF against observed runoff estimates from baseflow-separation of daily stream hydrograph data (Arnold and Allen, 1999). Surface runoff was used to calibrate the $\mathrm{CN_{II}}$ values for the VSLF wetness index classes based on 1993–1995 data. A lumped $\mathrm{CN_{II}}$ value was established by determining the basin wide storage (S_{e}) of 14 cm, which we used to distribute the σ_e values in the basin according to the wetness index (Eq. (6)). Subsurface hydrologic flow parameters (recess coefficient and unsaturated leakage coefficient) were calibrated using measured streamflow data for period 1993–1995. To calibrate the recession coefficient, all streamflow recession events were identified as periods with zero precipitation and snowmelt, based on measured daily precipitation and model estimates of daily snowmelt. The streamflow recession coefficient was calculated as the average recession constant during the calibration period (Schneiderman et al., 2002). The unsaturated leakage coefficient, which mainly affects streamflow during low flow periods when soils are below field capacity, was calibrated by an optimization procedure that minimized simulated streamflow error during low flow months (using the baseflow-separated discharge). We did not re-calibrate for discharge during the post-BMP period, but instead this record was used to confirm that BMP practices only minimally affect direct runoff.

5.5. Phosphorus calibration

Sediment P parameters were calibrated using stream chemistry data for the period 1993–1995. Greater detail of the procedure can

Table 5
Mean observed event phosphorus load at the watershed outlet for the pre- and post-BMP periods

Observed phosphorus load	Pre-BMP period (1993–1995)	Post-BMP period (1997–2004)	p-Value*
	kg event^{-1}		
Mean DP load	2.90	1.08	0.00
Mean TP load	6.51	3.45	0.00

*p-Value indicates a statistically significant difference using a t-test at $\alpha \le 0.05$. The p-values here indicate that the means are significantly different.

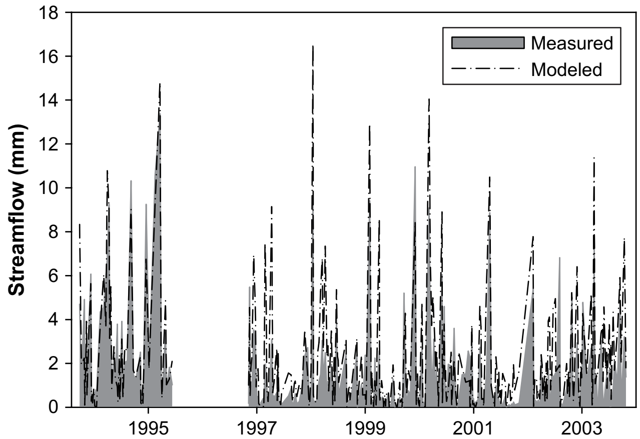

Fig. 2. Observed and modeled event streamflow from 1993 to 2004, calibrated for 1993–1995.

be found in Schneiderman et al. (2002). These parameters were optimized via a multi-variate optimization, where one or more parameters were varied until an optimal parameter value was identified yielding the best fit (minimum sum of squared errors and maximum efficiency (Nash and Sutcliffe, 1970)) between measured and simulated values. The parameters that determine sediment yield magnitude (sediment delivery ratio) and timing (transport capacity power) were optimized in a three-step process. First, the sediment delivery ratio and the transport capacity power were optimized simultaneously to derive the pair of values that minimizes the error in simulated versus measured sediment yield (see Schneiderman et al., 2002). Second, the mean annual erosion, and mean annual transport capacity were estimated using the optimized value for the transport capacity power (described above). Finally, the particulate nutrient parameter (enrichment ratio) was optimized to the measured monthly particulate P concentration.

Dissolved P loads in VSLF are determined by parameters that specify DP concentrations (mean expected concentration) in runoff from different landuses and in groundwater. These parameters were calibrated together with a single multiplicative factor that is applied to all nutrient concentrations. Thus the relationships between concentrations associated with various landuses and groundwater are maintained (concentrations shift up or down with the multiplicative factor). This factor was calibrated to minimize the error in simulated versus measured DP for the pre-BMP period (1993–1995). Once the distribution of expected concentrations was fixed, the runoff concentration parameters A, B and C (Eq. (8)) were dual optimized to the daily and cumulative DP loss. The calibrated values for A, B and C were 0, 0.71 and -0.40, respectively. Finally, the baseflow DP concentration was calibrated to the measured DP level in baseflow during the pre-BMP period. In 2001, a fenced cattle crossing over the stream was installed and the baseflow P levels were recalibrated to account for the addition. The BMP reduction factors were calibrated for the period from 1997 to 1999 by adjustment to optimize the model fit. Reduction factors for the structural and source impact BMPs were constrained within the ranges found in the literature for the years 1997–1999 in the post-BMP period as described above. The model was validated in the post-BMP period from 2000 to 2004. Spatially distributed data were validated with field observations from Gérard-Marchant et al. (2006) and Frankenberger et al. (1999).

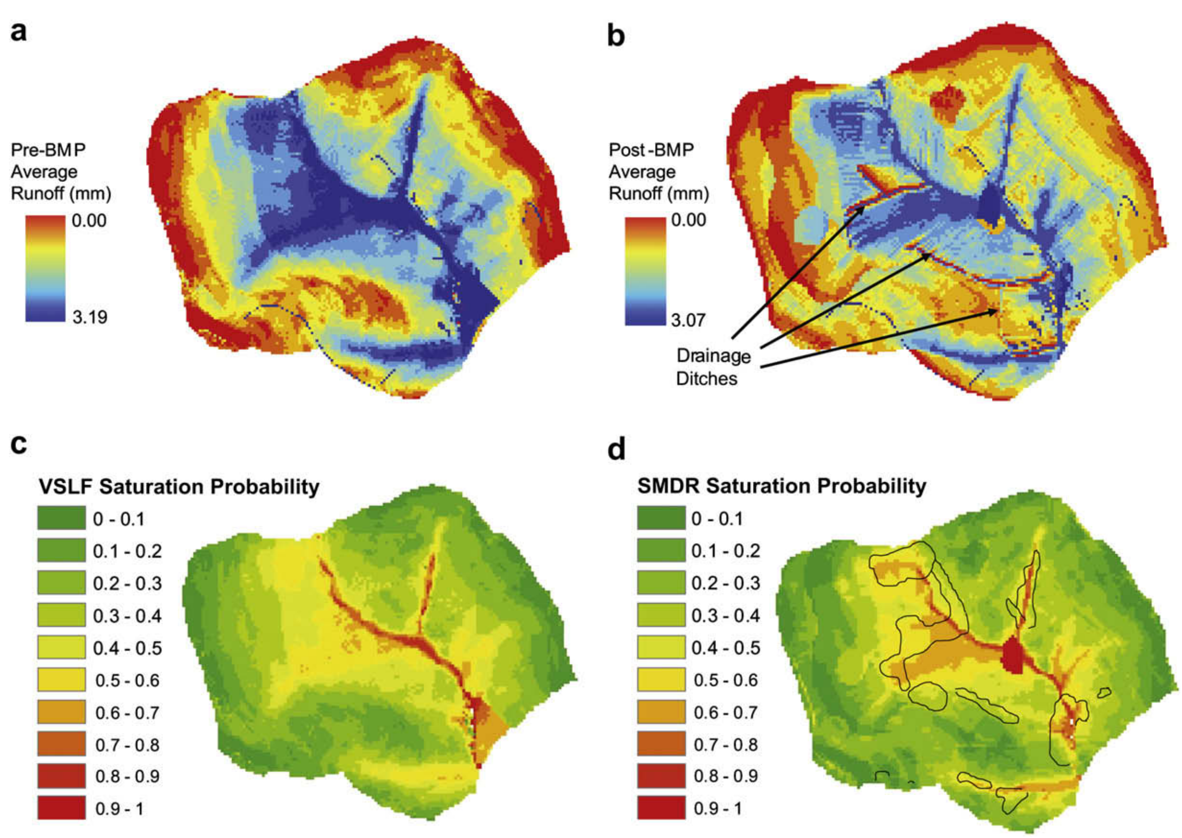

Fig. 3. Average runoff (mm) in the pre-best management practice (BMP) (3a), and post-BMP (3b) periods.3c and 3d compare the probability of saturation in the watershed for VSLF and SMDR, a fully distributed, mechanistic model. Approximately 90% of the watershed area has a difference in the probability of saturation less than 5% (*i.e.*, the standard deviation between the predicted values is less than 5% for 90% of the watershed). Also shown in 3d is the mapped extent of VSAs (from Gérard-Marchant et al., 2006).

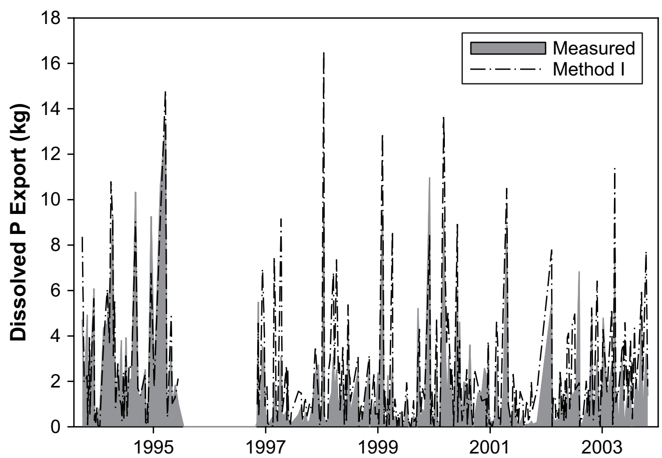

Fig. 4. Observed and modeled event dissolved phosphorus (DP) from 1997 to 2004, calibrated for the 1993–1995 period.

6. Results

6.1. Hydrology

The stream discharge at the outlet did not change with the introduction of BMPs. A multi-variate analysis of covariance (paired watershed study) in this watershed by Bishop et al. (2005) showed no effect on overall water budget due to BMP implementation. The Nash–Sutcliffe efficiency (E) was 0.85 for the pre-BMP, and was 0.85 for the 1997–2004 post-BMP period, using the calibrated parameters of the pre-BMP period (Table 5). The lack of BMP effect on discharge is also evident by visual inspection of the hydrograph (Fig. 2). In model terms this indicates that the overall watershed storage parameter, S_e in Eq. (2), was unchanged by BMP implementation ($S_e = 14$ (cm), which corresponds to a CN = 64.5). However, the distribution of runoff did change with the addition of drainage ditches (and stream channel modifications); runoff losses down gradient from the drainage ditches declined and runoff losses increased where the drainage ditches were installed (compare Fig. 3a and b).

Comparison of the saturated areas and zones of lower wetness indices were very similar to those found in Gérard-Marchant et al. (2006) and Schneiderman et al. (2007) (Fig. 3), who analyzed the same watershed. These studies found similar zones of soil moisture on the farm-watershed using both the SMDR and VSLF models (Fig. 3c and d) and with a transect analysis demonstrated that VSLF

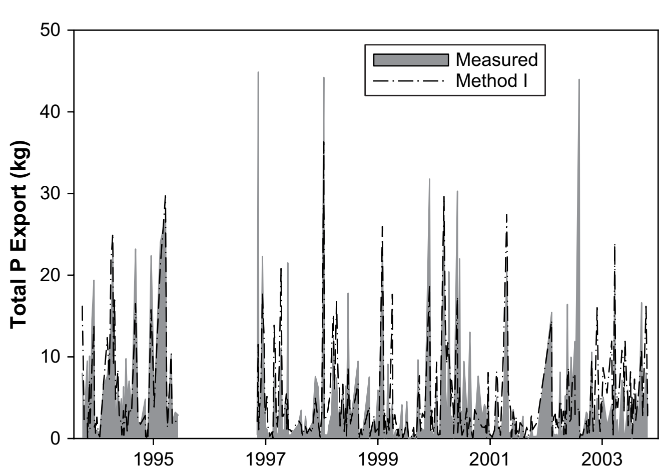

Fig. 5. Observed and modeled event total phosphorus (TP) from 1997 to 2004, calibrated for the 1993–1995 period.

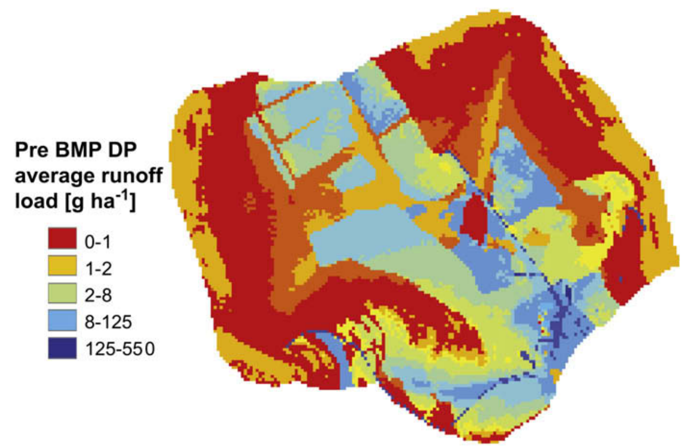

Fig. 6. Average dissolved phosphorus (DP) runoff load (g ha^{-1}) in the pre-best management practice (BMP) period.

adequately captured the spatial evolution of the soil moisture status, an important factor in VSA runoff generation.

6.2. Phosphorus loss

The event based DP and TP loads were generally well predicted (Figs. 4 and 5). The model captured the P loads well for the pre-BMP period with an E of 0.87 and 0.80 for event DP and TP export, respectively (Table 4). Fig. 6 shows the spatial distribution of average DP loss before BMPs were implemented. However, using the extraction coefficients calibrated during the pre-BMP period to model the post-BMP (1997–2004) period resulted in severe over-predictions of DP loads as can be observed in Figs. 4 and 5, and is indicated in Table 4 by the poor DP efficiency in the post-BMP period, $E = 0.08$ (Table 4). The predictions of TP (Fig. 5) were better than the DP prediction up to approximately 2000 with the pre-BMP calibrated extraction coefficients ($E = 0.54$). After the installation of the cattle crossing and fencing the TP concentration dropped significantly except for the large spike in June 2002, which was due to a valve failure in the manure storage lagoon, and thus not captured by the model. During this period the E was 0.14 for DP and 0.33 for TP (Table 4). The mean values for DP and TP were significantly different in the pre- and post-BMP periods, as shown in Table 5 (t-test, $p < 0.01$). The calibrated export coefficients from the pre-BMP period resulted in over-predictions of DP and TP losses in the post-BMP period.

Using the hydrology model and the export coefficients from the pre-BMP period, we calibrated the BMP reduction factors to the observed stream DP load in the 1997–1999 post-BMP period, and validated these results in the 2000–2004 post-BMP period.

Since the hydrology was predicted well and did not change due to BMP implementation, we hypothesize that the BMPs were responsible for the decrease in measured DP at the outlet. We simulated the incorporation of BMPs using reduction factors in the model, as

Table 6
Nash-Sutcliffe efficiencies and percentage difference between model prediction and observed data for the event dissolved phosphorus (DP) calibrated to the 1993–1995 pre-BMP period with BMP reduction factors for phosphorus best management practices (BMPs)

Period	Dissolved P			
	Method I		Method II	
	(E)	% Difference	(E)	% Difference
1997–1999	0.78	0.55	0.75	0.60
2000–2004	0.68	0.81	0.70	0.86
1997–2004	0.71	0.65	0.72	0.69

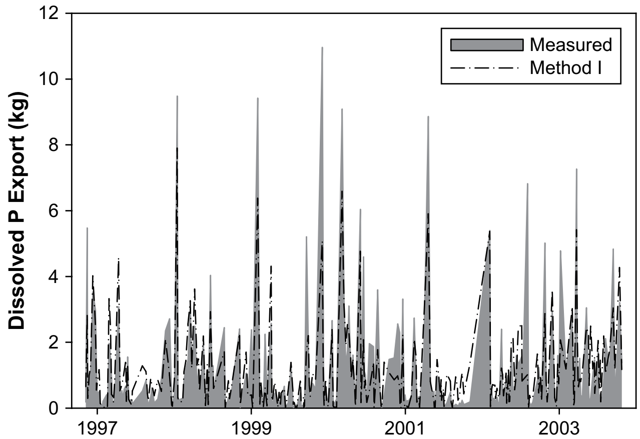

Fig. 7. Observed and calibrated event dissolved phosphorus (DP) from 1997 to 2004 using Method I BMP reduction factors to simulate best management practices (BMPs).

explained above. The BMP reduction factors calibrated for the 1997–1999 period (Table 3) were used for both Methods I and II. The efficiencies for the post-BMP calibration, validation and complete post-BMP time period are in Table 6. Since the results for both Methods I and II are similar we discuss the results individually for both Methods, but plot only results from Method I in Figs. 7 and 8.

For event DP, the E was 0.78 for Method I and 0.75 for Method II for the post-BMP calibration time period (Table 6). For event TP, the E was 0.52 for both Method I and Method II. We validated these predictions during the post-BMP period (2000–2004). The E for event DP was 0.68 for Method I and 0.70 for Method II (Table 6). For TP, the E was 0.36 for both methods. Across the entire post-BMP time period (1997–2004) the event DP E was 0.71 for Method I and 0.72 for Method II (Table 6). For event TP, the E was 0.47 for both methods. Fig. 7 shows observed DP and Method I model predictions, while Fig. 8 shows observed TP and Method I model predictions. The spatial distributions of average runoff DP loss for the two methods are shown in Figs. 9 and 10.

The mean observed event DP export was 1.07 kg event^{-1} while the predicted DP export for Methods I and II was 1.13 kg event^{-1} and 1.19 kg event^{-1}, respectively, neither of which are statistically different (t-test, $p < 0.01$) (Table 7). The mean observed event TP load was 3.42 kg event^{-1} while the predicted TP export for Methods I and II were 3.15 kg event^{-1} and 3.24 event^{-1}, respectively, neither of which are statistically different (t-test, $p < 0.01$) (Table 7).

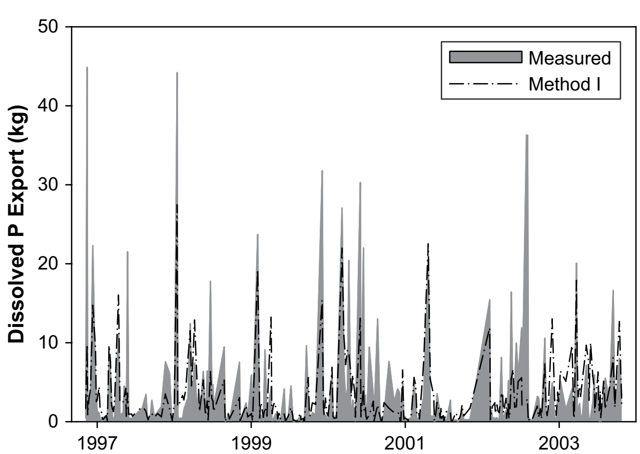

Fig. 8. Observed and calibrated (modeled) event total phosphorus (TP) from 1997 to 2004 using Method I BMP reduction factors to simulate best management practices (BMPs).

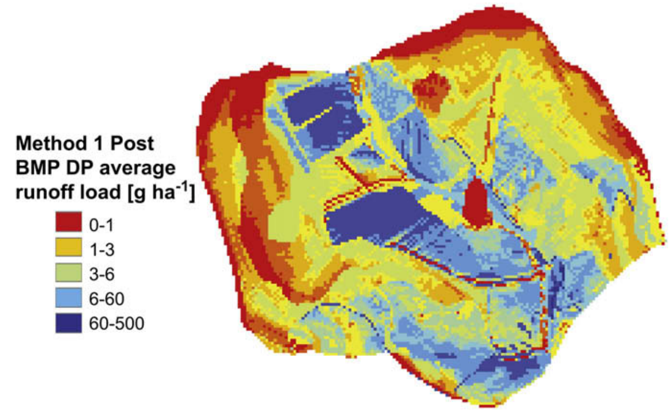

Fig. 9. Average dissolved phosphorus (DP) runoff load (g ha^{-1}) in the post-best management practice (BMP) period using Method I.

7. Discussion

The reported BMP reduction factors available in the literature vary widely for many BMPs, and are non-existent for several. However, using a subset of these BMP reduction factors in both Methods I and II we were able to reproduce the observed P export from the watershed with reasonable results. The prediction accuracies in the post-BMP calibration period (1997–1999) were slightly higher than those in the post-BMP validation period (2000–2004). The spike in DP seen in 2002, which was caused by a valve failure in the manure lagoon that leaked approximately 6.7 kg of DP (Bishop et al., 2005), was not effectively simulated by VSLF. This incident was brought to our attention by the farmer and farm planners (personal communication, July 2006). In terms of the spatial distribution of DP loss, inspection of Figs. 9 and 10 shows Methods I and II to produce very similar predictions. Method II predicts a higher average load of DP in the wetter HRUs. It is possible that specific BMPs applied in the wetter areas are responsible for the very slight difference, specifically the NMP, buffers and the exclusionary fencing.

The distribution of DP before and after BMP implementation is interesting to observe (Fig. 6 versus Figs. 9 and 10; note that the scales are different). The DP load was distributed across the watershed with the implementation of the BMPs, especially the culvert crossings and stream protection measures. Note that in the pre-BMP period the model predicts the highest P source areas to be adjacent to the stream, while during the post-BMP period the near-stream areas are, comparatively, much less of a source area, as more manure is now spread in the upland fields, and the near-stream

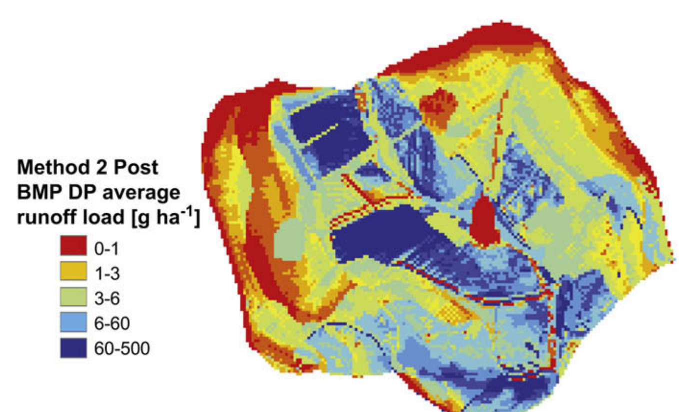

Fig. 10. Average dissolved phosphorus (DP) runoff load (g ha^{-1}) in the post-best management practice (BMP) period using Method II.

Table 7
Observed and simulated event dissolved and total phosphorus loads using BMP reduction factors for the BMPs

Observed P load	Post-BMP period: 1997–2004		p-Value*
	Observed	Simulated	
	kg event^{-1}		
Method I			
Mean DP load	1.07	1.13	0.67
Mean TP load	3.42	3.15	0.39
Method II			
Mean DP load	1.07	1.19	0.62
Mean TP load	3.42	3.24	0.71

*p-Value indicates a statistically significant difference using a t-test at $\alpha \leq 0.05$. The p-values here indicate that the means are not significantly different.

area was converted to buffer. In general, most BMPs do not affect the baseflow P concentration. However, Flores-López et al. (submitted for publication) showed that when cattle crossings are installed, the baseflow P concentration can decrease by as much as 38%, which might explain the dramatic reduction in P levels following the installation of the cattle crossing in 2000 (Fig. 1).

As noted earlier, TP export did not appear to change with the implementation of most of the BMPs. The reason for this might be a time lag between BMP implementation and an observed reduction in TP loss. For instance, reducing manure applications to saturated areas will eventually cause a decrease in soil P levels, which would ultimately reduce both dissolved and sediment P runoff losses. The timing of BMP implementation also affected the results. While most of the BMPs were implemented in the 1995–1997 time period, some critical BMPs were added later, including the exclusionary fencing and animal walkways, which prevent cows from having direct access to the streams. For example, cattle crossings were added in 1998, 2001 and 2002, and thus their full effect might not have been realized until near the end of the monitoring period.

It is interesting to note that, excluding the stream buffers, the majority of the BMPs were implemented on the barnyard and crop landuse classes. These classes together represent a very small proportion of the total area of the farm (approximately 8%), but clearly are major P source areas in the watershed. Finally, the inclusion of the buffer landuse, and the accompanying exclusionary fencing, had a significant influence on the P loading from the farm-watershed. Thus, the modeling effort here has a possible practical result: BMPs targeted to certain landuses and hydrologically active areas can result in large decreases in P loads. Specifically, the near-stream and croplands should be target areas on which to concentrate a few effective BMPs resulting in significant P reduction (this can be seen by comparing the DP loads in Fig. 6 with Figs. 9 and 10).

8. Conclusions

The model results show that if BMPs are designed and implemented on farms in a spatially explicit manner with regard to VSAs, then DP loss can be decreased. In the post-BMP period, the model predictions of DP export were greatly improved by considering the impact of BMPs. This suggests that the implementation of the BMPs in specific areas was responsible for the observed reductions in DP export. Bishop et al. (2005) report event load reductions of 43% for DP and 29% for PP after BMP implementation in this watershed. Here we were able to demonstrate, spatially, where these reductions might have been realized. Further analysis using a combination of VSLF and geo-spatial techniques will allow researchers to advise farm planners on optimal placement of BMPs in VSA watersheds. The model results indicate that the most effective BMPs dissociated pollutant source areas with hydrologically active areas (e.g., VSAs). Specifically, protecting riparian areas (buffer), reducing the spreading of manure during hydrologically active periods (NMP) and excluding livestock from the stream (cattle crossings) resulted in the largest DP reductions.

Greater model precision might be obtainable if BMP reduction factors associated with suites of BMPs could be easily separated, such as is the case with NMPs. This would allow researchers to determine what BMPs are the most effective when placed in combination. The combined effect of several BMPs implemented on a single HRU can only be approximated at this point. Future work should include attempts to refine the methodology to determine how BMPs interact in the field, aiding our analysis of farm-level implementation of BMPs and further allowing us to examine specific policy scenarios of BMP implementation.

The VSLF model can be used to evaluate land management practices in most watersheds where shallow soils overlying an impermeable layer give rise to VSAs and generation of saturation-excess runoff. The model requires data that are generally available in most watersheds and can therefore be used by managers and planners in directing water resource policy throughout the world.

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