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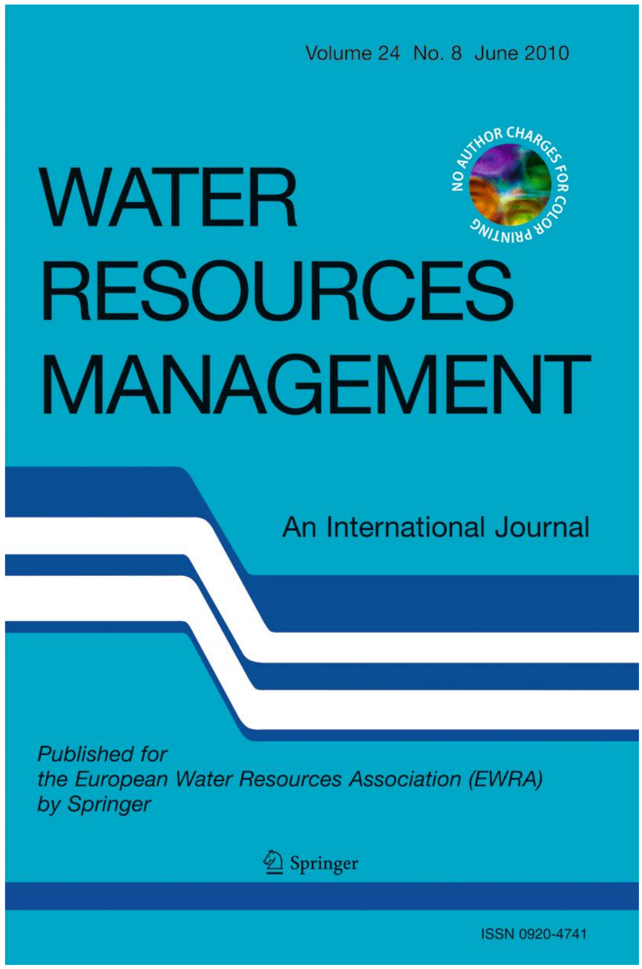

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Spatially Explicit Modeling of Land Use Specific Phosphorus Transport Pathways to Improve TMDL Load Estimates and Implementation Planning

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Abstract Diffuse pollution from urban stormwater and agricultural runoff are among the leading causes of water pollution in the USA. A process-oriented, stakeholder-driven research approach was implemented in the small heterogeneous watershed of St. Albans Bay, Vermont to model the relative load of phosphorus from all sources, including diffuse transport pathways, and compared to goals and assumptions outlined by a Total Maximum Daily Load (TMDL) developed for phosphorus in Lake Champlain. Mass-balance and dynamic landscape simulation models were used to describe the distribution of the average annual phosphorus load to streams (10.57 t/year) in terms of space, time, and transport process. The

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majority of the phosphorus load comes from two subwatersheds dominated by clay soils, Stevens and Jewett Brooks. Dissolved phosphorus in surface runoff from the agricultural landscape, driven by high soil phosphorus concentrations, accounts for 41% of the total load to watershed streams. Direct discharge from farmsteads and stormwater loads, primarily from road sand wash-off, are also significant sources. Results reported in this study could help target watershed interventions both in terms of the types and locations of recommended best management practices (BMPs). The study offers an approach to attaining TMDL diffuse pollution targets in a cost-effective and participatory manner and could be replicated for other TMDL processes around the country.

Keywords Diffuse pollution management · Non-point source pollution · Spatially explicit watershed model · Landscape model · TMDL · Phosphorus · Vermont

1 Introduction

Nutrient pollution accounted for approximately 10% of the water quality impairments identified by the EPA in the USA in 2009. Diffuse (nonpoint source) pollution is the remaining large untreated source of nutrient pollution in many parts of the country. Diffuse pollution is typically not regulated, leaving governmental bodies to implement change through education and incentives which are only sometimes successful at achieving targeted reductions identified through the total maximum daily load (TMDL) process. This is in part because the identification and quantification of diffuse sources and pollutant transport mechanisms have proven difficult at the watershed scale. Furthermore, although maximum pollutant loads can be calculated for a waterbody, there is often considerable uncertainty in quantifying the relative roles of specific phosphorus transport pathways from the landscape. Overcoming this challenge would help watershed managers to better target actions that will lead to cost-effective attainment of TMDL load allocations.

Lake Champlain, like many fresh water lakes, has received excess nutrient runoff for the past 50 years (VTANR and NYDEC 2002) due to changes in agricultural practices and rapid development of open space for residential uses (Hyde et al. 1994). The effect of excess nutrients on the health of Lake Champlain has been most dramatically witnessed in bays such as St. Albans Bay (Fig. 1), which exhibit eutrophic algal blooms every August (Hyde et al. 1994). The Vermont Agency of Natural Resources and the New York Department of Environmental Conservation (DEC) completed a TMDL for phosphorus in Lake Champlain that was approved by the USEPA in 2002 (VTANR and NYDEC 2002). Under this TMDL the St. Albans Bay watershed was allocated a total annual nonpoint source (diffuse) phosphorus load of 4.2 metric tons per year (t/year), requiring an estimated reduction of 33% from diffuse sources. Past efforts to reduce phosphorus loss from the watershed, beginning in the early 1980s, have not resulted in significant reductions of loading to the bay, nor have they yet been successful in reducing nuisance algal blooms during summer months (Meals 1996). This is in part due to the relatively low mixing rate of water in St. Albans Bay with the rest of the lake, but it also suggests that important phosphorus sources or transport pathways may have been overlooked in the past.

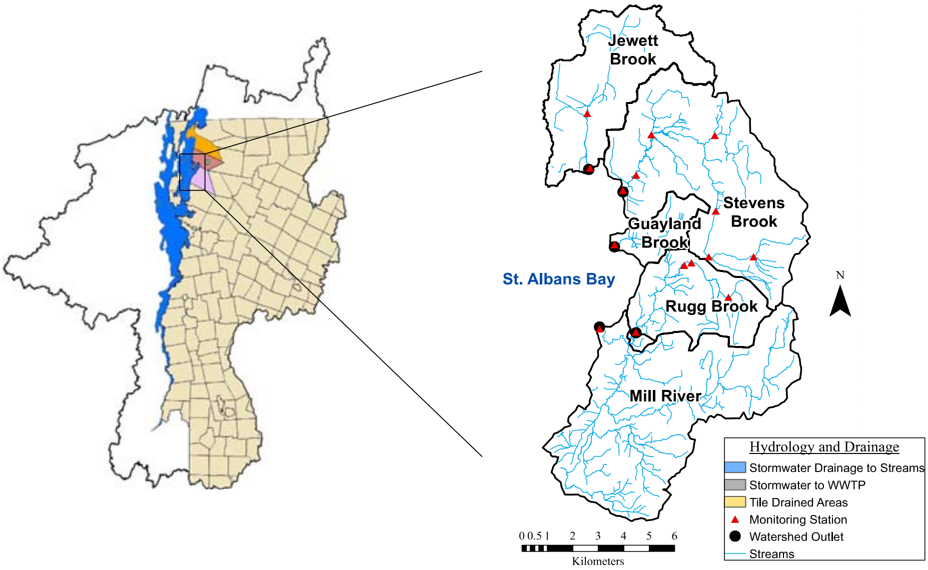


Fig. 1 Lake Champlain Basin and detailed map of St. Albans Bay watershed hydrology

By 1991, St. Albans Bay watershed still had one of the highest phosphorus export rates in the Lake Champlain Basin (VTANR and NYDEC 2002). Quantification of the sources and pathways of phosphorus movement across the landscape would be useful in prioritizing where and how phosphorus reduction efforts in the basin should be focused in the future.

This paper represents one piece of a comprehensive study of the St. Albans Bay watershed in Northern Vermont. The overall study was process-oriented in that it involved a holistic accounting of key watershed processes, identified by stakeholders, at appropriate temporal and spatial scales, requiring integration of mechanistic models and watershed monitoring. Monitoring of water quality in streams in the watershed and hydrologic modeling were used together to calculate the mean annual total phosphorus loads to St. Albans Bay from each of five subwatersheds (Gaddis 2007). Several modeling tools were used to apportion the total load for each subwatershed to specific phosphorus sources and transport pathways. Identifying the relative importance of each phosphorus source and its transport pathway to streams in the watershed is the primary focus of this paper. The results of other components of the study are reported elsewhere and include water quality monitoring (Gaddis 2007), participatory modeling with stakeholders in the watershed (Gaddis et al. 2009), estimation of total phosphorus load to St. Albans Bay during different hydrologic flow regimes (Gaddis 2007), and optimization of phosphorus reduction strategies across the watershed (Gaddis 2007). In this paper, we report on the relative importance of specific phosphorus sources and transport pathways in the St. Albans Bay watershed. These include diffuse sources separated into specific landscape runoff processes (erosion, surface runoff, subsurface runoff, and road sand wash-off), point sources, and semi-diffuse sources such as the impact of waterfowl and farmstead runoff. In addition, we discuss how the results could be used to identify more cost-effective strategies to reduce phosphorus load to the bay in the future.

2 Watershed Description

The 130 km^2 watershed feeding St. Albans Bay is dominated by agriculture (57% of the watershed area), at the same time that the urban and suburban areas (14% of the watershed area) are growing. The watershed is currently home to over 20,000 people primarily in the City of St. Albans and the Town of St. Albans. The upper reaches of the watershed are steep and dominated by deciduous forests while the lower portions are flat and dominated by agriculture. The City of St. Albans is located in the middle of the watershed and is surrounded by suburban development. Other developed areas of the watershed are concentrated along Highway 7, which runs North–South through the middle of the watershed. Between 1850 and 1990 there was an overall shift in the watershed, and throughout Vermont, from small integrated farms to concentrated feeding operations with larger herd sizes. This has led to increased feed and fertilizer imports to the state, resulting in an excess of phosphorus in dairy farming areas (Hyde et al. 1994; Cassell et al. 2002; Magdoff et al. 1997).

The St. Albans Bay watershed is drained by five streams: Jewett Brook, Stevens Brook, Rugg Brook, Guayland Brook, and Mill River. The area draining to Jewett Brook, the northern most stream in the watershed, is dominated almost entirely by agriculture with very little forest and/or suburban area. The headwaters for Rugg and Stevens Brooks originate in the steep forested eastern portion of the watershed. These streams run through suburban areas and the City of St. Albans where they receive stormwater runoff before flowing through predominantly agricultural areas in the low elevation, flat portion of the watershed. Guayland Brook originates in the suburban area downstream of the City of St. Albans and runs through low density residential and agricultural areas. Mill River, the largest and southern most drainage in the watershed, drains agricultural, forest, and low-density residential lands. Mill River joins with Rugg Brook before flowing into St. Albans Bay (Figs. 1 and 2).

The soils in the watershed are primarily loam (varying from silty to stony loams) however a large portion of the lower parts of Jewett and Rugg Brook subwatersheds have clay soils (Fig. 3). These very flat portions of the watershed were once characterized by wetlands and are naturally poorly drained. Agriculture has only been made possible in this part of the watershed through tiled field drains. Surface runoff is substantially reduced in areas that have functioning subsurface drains because soil saturation is rare.

3 Methodology

A distributed landscape model was used to formalize concepts of watershed processes and as such explore the mechanisms and underlying driving forces of phosphorus movement and transport in the landscape. Simpler spreadsheet-based models were used to estimate point source and aggregate diffuse phosphorus loads in the watershed. Together, the landscape model and spreadsheet models were used to

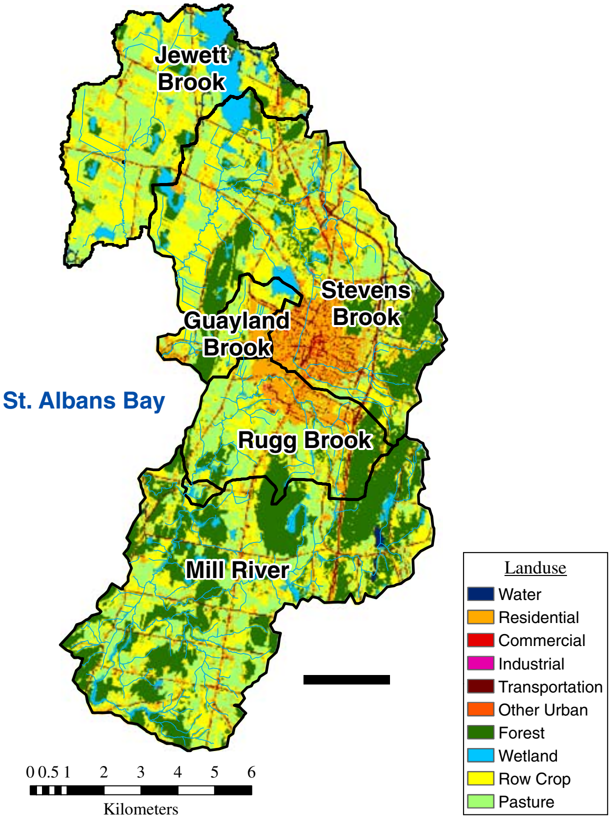


Fig. 2 Map of St. Albans Bay watershed land use

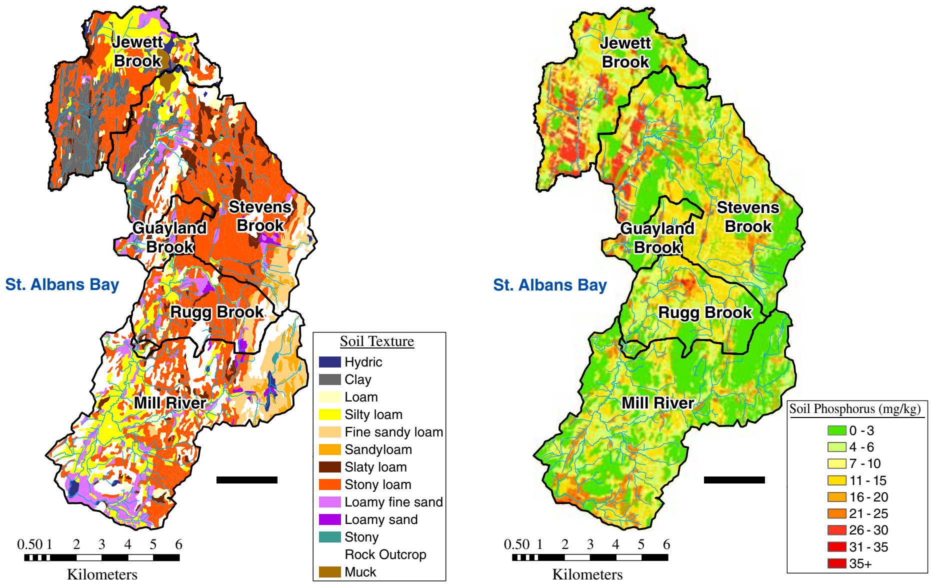


Fig. 3 Map of soil texture and soil phosphorus concentration in the St. Albans Bay watershed

evaluate the relative importance of all sources and transport pathways of phosphorus in the St. Albans Bay watershed.

3.1 Spatially Explicit Landscape Model to Simulate Diffuse Phosphorus Pathways and Sources

A landscape model was developed to simulate the dynamics of phosphorus transport from diffuse sources in the St. Albans Bay watershed. This model was developed as six interconnected modules (Evapotranspiration, Snow hydrology, Hydrology, Plant dynamics, Sediment, and Phosphorus) using Stella systems modeling software (iseesystems.org). A seventh module was used as a central module for input data (time-series and spatial data), conversion rates, and other universal model parameters. The distributed landscape model is partitioned into a spatial grid of square unit cells (30 × 30 m in this application) in which each of the process modules is implemented. Stella modules are linked with time series inputs, spatial inputs, and parameter table files into a complete landscape simulation model using the Spatial Modeling Environment (SME) software (Maxwell and Costanza 1997a, b; Maxwell 1999; Costanza and Voinov 2004). SME automatically converts Stella generated modules into a C++ driver allowing the user to run the modules as one complete process model in each cell of the landscape at a given time step (one day for the St. Albans Bay watershed model application). Horizontal fluxes of water and nutrients are accounted for with cell–cell head differences of surface water and groundwater (Voinov et al. 1999a).

The linkage of ecological and hydrologic modules using SME is described elsewhere (Costanza and Voinov 2004) as the Landscape Modeling Framework (LMF). Feedbacks among the biological, chemical and physical model components are important structural attributes of this framework (Maxwell and Costanza 1995; Maxwell 1999; Voinov et al. 2004). The integration of physical, biological, and social factors into a dynamic model provides a platform for decision makers to explore the dynamics of a watershed system (Madani and Mariño 2009). A simulation run using SME gives a visual representation of the landscape as it evolves over time reflecting changes in hydrology, water quality, and material flows between adjacent cells. The approach differs from other watershed modeling suites in that the user is not limited to pre-designed modules allowing users to alter existing modules and add additional modules with the user-friendly Stella software.

Templates for the hydrology, plant dynamics, and evapotranspiration modules came from the Library of Hydro-Ecological Modules, originally developed for the Patuxent River watershed in Maryland (Voinov et al. 1999a, b). These modules were modified for use in the St. Albans Bay watershed, Vermont, which is smaller, colder, and more heterogeneous than the Patuxent River watershed. New modules, developed specifically for the St. Albans Bay watershed were erosion/sediment transport, phosphorus dynamics, and snow hydrology modules. The full set of equations used in the modules is available on-line at http://www.likbez.com/LHEM.

3.1.1 Hydrology Modules (Hydrology, Snow, and Evapotranspiration)

The hydrology module drives the erosion and phosphorus transport modules since all movement of sediment and phosphorus is assumed to be transported in water.

Water is tracked, in units of meters/day (assumed to be homogenous across a cell), through pooled surface water, flowing water in channels, and two layers of unsaturated water in soil. Water moves within a cell through the processes of runoff, infiltration, percolation, surface and soil evaporation, plant transpiration, and tile drainage. Besides precipitation and snow melt, water also enters some cells through groundwater recharge in tile drained areas or in perennial streams. Water is lost from the cell to groundwater, the dynamics of which were not modeled.

Daily precipitation and snow melt drive the hydrologic model. The former was obtained from the St. Albans NCDC climate station (COOP ID 437032) and the latter is an output from the snow hydrology module described below. Surface runoff is estimated using the curve number method developed by the Natural Resources Conservation Service (USDA 1997) and incorporates simulated Antecedent Moisture Condition (AMC). Interception is calculated using the leaf area index, an output from the plant dynamics module. Soil specific infiltration rates, percolation rates, and porosity were obtained from the STATSGO database available for the watershed (soils.usda.gov/survey/geography/statsgo). Land use specific percent impervious cover was obtained from a technical report, specific to Vermont, which was completed by the Center for Watershed Protection in 1999. Soil evaporation and plant transpiration were calculated using equations from Neitsch (2000).

We assumed that a field was tile drained if it was characterized as poorly drained (hydrologic soil group D) on the US General Soil Map (soils.usda.gov/survey/geography/statsgo), has a slope less than 3%, and is of agricultural land use. These assumptions were confirmed reasonable by agricultural stakeholders in the watershed. Twelve percent of the entire St. Albans Bay watershed (22% of the agricultural area in the watershed) is estimated to be drained. In areas where tile drains are installed on agricultural fields, water in the unsaturated zone, down to the level of the soil's field capacity, is assumed to drain through a tile drain to a channel or ditch at each time step. Tile drains are generally placed at a depth of 0.5 m, which determined the default model depth of the middle layer of the unsaturated zone.

Water movement between cells is accounted for with modified versions of algorithms already developed and described in the literature (Voinov et al. 1999a, b). The only modification made to the algorithms is the separation of the surface water and channel water stocks in the hydrology module such that water is allowed to pond in the cell as surface water for several days while some water, calculated as a function of curve number, is transported to channels and is available for transport downstream. In Stevens Brook, stormwater and wastewater discharge were diverted to outfall locations within the watershed.

A separate evapotranspiration module calculates potential evapotranspiration, using the Penman–Monteith equation (Neitsch 2000; Shuttleworth 1993; Voinov et al. 2004). Leaf surface resistance and aerodynamic resistance are calculated using equations from Shuttleworth (1993). Stomatal leaf conductance, leaf area index, were obtained from Breuer and Frede (2003). The major climatic input for evapotranspiration is daily average wind speed, which was obtained from the St. Albans NCDC climate station.

The snow module simulates snow accumulation, sublimation, and melt in units of meters/day of snow water equivalent (SWE). Snow fall data was obtained from the St. Albans climate station. Snow interception by plants is calculated using leaf area index, an output from the plant dynamics module. Sublimation is modeled as a function of wind speed, air temperature, humidity, slope and aspect. Snow melt is modeled separately for rain on snow and dry periods. The latter is a function of air temperature and snow density, aspect, and slope (Gray and Prowse 1993; Male and Gray 1981). Process-based distributed snow melt models have been found to improve stream flow predictions at the catchment level when compared to temperature-index models (Zeinivand and De Smedt 2009).

3.1.2 Sediment Module

The sediment module simulates overland surface erosion from a cell to channels (intermittent rills or perennial streams). The basis of the sediment module was derived from RUSLE 1.06 (Rendard et al. 1997) which was modified to a daily time scale to calculate potential soil loss, in units of kilogram per day per cell. Climatic input data for the sediment module include maximum storm intensity (mm/h) and daily precipitation, both obtained from the St. Albans NCDC climate station. Soil erosivity factors came from the SSURGO soil database compiled by the NRCS (soildatamart.nrcs.usda.gov). The slope factor is based on percent slope calculated using GIS for each cell (Rendard et al. 1997; McCool et al. 1987). Coefficients for cover crop and land management practices were obtained from NRCS staff in Colchester and St. Albans, Vermont and assumed to be representative of agricultural practices in the watershed. Potential soil loss is modified by overland sediment transport capacity, which is a function of surface runoff (Foster 1982). In this way tile drainage and other runoff mediators are accounted for in calculating actual estimated erosion. The minimum of potential soil loss and overland sediment transport capacity is used as the final overland erosion value.

Road sand applied during winter storms washes off during spring melts and rain storms and contributes a unique source of phosphorus to streams in the area. The City of St. Albans reported that the Public Works Department applies 600 tons of sand per year over the course of an average of 20 snow storms per year. This value was divided by the area of road in the city to derive an application rate of 0.058 kg sand per square meter applied per storm. This estimate is used as a constant in the sediment module, trigged by actual snow events. In this module, a snow event is considered large enough to receive sand if snow fall is greater than 0.025 m/day. Sand wash off is assumed to occur when a rain storm is greater than 30 mm, otherwise, a proportion of the sand is washed off depending on the severity of the storm or melt event.

3.1.3 Phosphorus Module

The phosphorus module captures four distinct diffuse phosphorus transport processes: surface erosion, tile drainage, dissolved phosphorus released from soil, and road sand wash-off. Some of these processes are specific to particular land use types such as tile drainage (agriculture) and road sand wash-off (roads). The phosphorus module incorporates processes that are modeled mechanistically and some that could only be determined empirically. Dissolved and particulate phosphorus are tracked separately in the model. No attempt is made to model in-stream phosphorus processes, though we recognize that this is an important process in the movement of phosphorus through stream channels and final load estimates to St. Albans Bay. All processes are modeled as phosphorus load to streams (g/cell) rather than to the bay.

The initial condition for soil phosphorus was input as a map of soil phosphorus concentrations specific to land use-soil type combinations throughout the watershed (Fig. 3). The map was created based on soil phosphorus tests on agricultural and urban soils in the watershed. Agricultural soil phosphorus data was obtained from the NRCS with the permission of landowners in the watershed. The fields sampled represent 5.2% of the agricultural fields in the watershed, with a range of soil, slope, and crop characteristics. In addition, we collected soil phosphorus samples from over 70 residential, commercial, and park lawns in the city, representing 5.8% of the properties in the city.

The inputs to soil phosphorus are atmospheric deposition, manure application, and fertilizer. The average daily atmospheric deposition of phosphorus was calculated based on the annual AirMON data available from the National Atmospheric Deposition Program (nadp.sws.uiuc.edu/AIRMoN), which measures atmospheric deposition in Underhill, Vermont. The total annual phosphorus load in 2003 was 0.021 kg/ha/year. Agricultural manure and fertilizer application assumptions were obtained from local farmers and confirmed as typical application rates for the area by the local NRCS office. Application rates of fertilizer on the urban landscape were estimated using survey results collected of homeowners in the City of St. Albans (Homziak n.d.). Weatherization of rock is assumed to be a sufficiently slow process as to not be captured in this daily time step model. Phosphorus leaves the soil through plant uptake (Wood et al. 1984), dissolved phosphorus in surface runoff (Sharpley et al. 2002), surface erosion, and tile drainage.

The total dissolved phosphorus (g/day) that leaves the soil in runoff is calculated by multiplying the surface water runoff, from the hydrology module, by dissolved phosphorus concentration in runoff (g/m^3) determined by the concentration of phosphorus in the soil. Soil phosphorus concentrations required to reduce dissolved phosphorus in surface runoff are often significantly less than concentrations maintained for optimal crop growth (Heathwaite et al. 2000) due to the dynamics of soil availability dictated by adsorption and desorption processes (McDowell and Sharpley 2001).

The regression equations linking soil phosphorus to dissolved phosphorus concentration come from Sharpley et al. (2002) and are different for corn and pasture land use types. It was assumed that urban lawns behave in a manner similar to pasture. Since only the water running off of the soil can pick up soil derived phosphorus, this value is then multiplied by the proportion of the cell that is not impervious.

The phosphorus concentration in tile drainage was assumed to be a constant concentration based on monitoring data collected for soil type and land use specific combinations. We sampled eight tile drains in the watershed on different soil type-land use combinations during each season, and validated our findings with tile drain concentrations reported for drainage waters in Quebec in an unpublished manuscript by Enright and Madramootoo in 2004. The phosphorus load from tile drainage was calculated by multiplying the soil type-land use specific phosphorus concentration in tile drainage by the modeled volume of water drained from each cell, as calculated in the hydrology module, during each daily timestep. This method was selected because a mechanistic model could not be validated due to the lack of a strong relationship between soil phosphorus concentration and tile drainage concentrations also reported in Quebec by Enright and Madramootoo in 2004.

Particulate phosphorus removed from soil as erosion is calculated by multiplying the kilograms of soil eroded, calculated in the sediment module, by the concentration of phosphorus in the soil, a parameter tracked in the phosphorus module.

The volume of road sand that washes off of roads and other impervious surfaces, an output of the sediment module, is multiplied by the concentration of phosphorus in the sand which was determined empirically through samples we collected of the road sand applied by the City of St. Albans (0.78 g/kg).

3.1.4 Execution of Complete Landscape Model

The landscape model was run separately, using SME, for each of the five subwatersheds: Stevens Brook, Rugg Brook, Jewett Brook, Mill River, and Guayland Brook. The model was driven with climatic data from 1984 to 1989. These years were selected because they represent a continuous climatic data set for 5 years with both wet and dry years and a variety of climatic patterns.

The model was configured to output results as both time series and spatial maps. Total discharge values at stream monitoring locations in the watershed were output as time series data. Maps were output as raster files at a 30 × 30 m resolution. All map outputs represented cumulative loads over the entire model simulation and were used to calculate total phosphorus movement within the watershed for each land use type. Phosphorus movement was calculated separately for the four transport pathways of phosphorus modeled in the landscape: surface erosion, dissolved phosphorus in surface runoff, tile drainage, and road sand wash-off.

Total phosphorus load to the streams was estimated using a post-processing method (Tim et al. 1992) to account for phosphorus attenuation in the landscape. The closer a cell was to a stream the more phosphorus from that cell is delivered to the stream.

3.2 Point-Source Load to Streams

Wastewater treatment plants are the only regulated point sources of phosphorus in the watershed (VTANR and NYDEC 2002). Although the maximum annual phosphorus load from these sources had been previously estimated and utilized in the Lake Champlain Phosphorus TMDL, a more detailed analysis of this load was necessary to understand the dynamics of combined sewer overflow in the watershed. Phosphorus discharged from the City of St. Albans wastewater treatment plant was calculated using a combination of empirical data and model output from the hydrology and phosphorus modules of the landscape model. Phosphorus in wastewater discharge was estimated on a daily timestep, and summarized over 5-years to calculate an annual average. Separate phosphorus loads were calculated for treated sewage and stormwater overflow.

3.3 Phosphorus Load to Streams from Dairy Farmsteads

Storm runoff from farmsteads where animals are concentrated for feeding and/or milking are considered semi-diffuse sources rather than diffuse sources and were calculated in aggregate for the watershed with help from the NRCS office in St. Albans. There are an estimated 20 to 30 farmsteads in the St. Albans Bay watershed. The specific location of farmsteads and data about their operations is either unavailable or proprietary. Farmstead phosphorus discharge was estimated separately for milk house effluent, manure runoff, and silage leachate. Unless otherwise noted, all of the assumptions in these calculations were derived in consultation with the Franklin County NRCS office in St. Albans.

Phosphorus load from milk house effluent was calculated by multiplying the volume of milk house effluent per day for each farmstead (1,135 to 4,542 l/year), by the average percent in the watershed that is not treated (10%) and the concentration of phosphorus in milk house effluent as described in Krider (1999).

The total phosphorus load originating from manure (t/year) that runs off from barns and barnyards is calculated as the product of the volume of manure deposited on barnyards untreated (1,195 m^3/year), the average concentration of P_2O_5 in manure (1,321 g/m^3 manure), and the percentage of manure deposited on the barnyard that runs off to streams (3%). The volume of manure deposited on barnyards was calculated as the product of the number of animals in barns or barnyards (6,399 animal units), the percentage of time animals spend in the barn (80%) or barnyard (20%), the percentage of manure treated from the yards (70%), and the quantity of manure produced by each animal unit (43.8 l/AU/day). The percent of manure in the barnyard that actually runs off to streams is calculated as an average erosion rate for barnyards in the area, using the sediment module.

Phosphorus contained in silage leachate is calculated as the product of the number of silos in the watershed (10), the quantity of silage in each silo (4,535 t/silo), the concentration of phosphorus in silage (0.08 kg/t silage; NRAES 1993), and the amount of silage leachate that is transported to streams (40%).

3.4 Waterfowl

Although waterfowl do not provide a new source of phosphorus to the watershed, their presence in large numbers during migration periods can mobilize phosphorus from the land (crops and vegetation) to the water. In the case of the St. Albans Bay watershed, waterfowl feed primarily on wetland plants, young crops, and waste grain on fields. Estimates of the number of waterfowl, primarily geese, present in the St. Albans watershed and the length of residence during each season were provided by William Crenshaw, a wildlife biologist with the State of Vermont Fish and Wildlife Department. Multiplying the total number of geese by the days of residence in the watershed gives the number of geese days per season. Geese days estimated for the St. Albans Bay watershed ranged from 4,500 to 9,000 during the summer season (90 days), 2,000–5,000 during the spring migration period (14 days), and 5,000–10,000 during the fall migration period (14 days). Geese days were multiplied by average phosphorus excretion per goose, 0.45 g/day (Post et al. 1998).

4 Model Calibration and Uncertainty

Model uncertainty is usually a major concern when results are delivered to model users such as stakeholders and decision-makers. We have found that by opening up the modeling process itself and by engaging stakeholders in model development and analysis, the issue of uncertainty is reduced, since the group is well aware of all the assumptions that went into the model construction and know that the best knowledge and data have been incorporated. The process creates trust in the model and its output (Gaddis et al. 2009) and very much facilitates discussions about model validity, limitations, and uncertainty.

The Landscape Modeling Framework provides a transparent and flexible platform for model building, application, analysis and interpretation. Because the model was developed in consultation with a group of stakeholders, this transparency was critical to gaining the confidence of stakeholders in the model results as well as in clearly explaining the model assumptions and limitations. The ability to modify the model to incorporate newly identified processes and sources of phosphorus in the watershed (e.g. adding the role of waterfowl) partially compensated for the lack of data available for extensive model calibration and validation.

4.1 Hydrology

Initial calibration efforts focused on calibration of the hydrologic modules with discharge data collected during the 1980s by the Rural Clean Water Program for which discharge and water quality data are available at intervals of several days over 8 years. The years 1984 through 1989 were selected for model calibration using observed climatic data from that time period and a land use and curve number map from 1992. Calibration parameters associated with groundwater recharge, spring runoff, tile drainage, vegetation interception, maximum travel distance down stream (per day), were used to calibrate the model to annual discharge data. However, daily calibration of the hydrologic model was challenging because of suspected errors in either the measured discharge data and/or the climatic data driving the model. None of the subwatershed outputs correlated well with discharge data ($R^2 < 0.23$). Separating the uncertainty resulting from model error versus that associated with data error was difficult and would require gauging the subwatersheds with more exact monitoring equipment (Hession and Storm 2000). Examination of patterns of discharge in the measured data and model output as well as precipitation, indicates several conflicts between the discharge data and the precipitation data. In some cases discharge was an order of magnitude greater than precipitation, according to the measured data and occurred during periods when snow melt was not a factor. However, in other cases records of significant precipitation did not correspond to a runoff response in the discharge data. The St. Albans NCDC climate station sits at the highest point in the watershed and may not reflect climatic conditions throughout. Some improvements in hydrologic calibration were made when climatic data were interpolated between the St. Albans station and the South Hero climatic station, which is more representative of conditions along the bay. Additional discharge data was collected in 2004 and 2005 as part of this project for the purpose of model calibration. Unfortunately, calibration of the hydrologic model with this data was not possible because NCDC stopped data collection at this site during this period.

The hydrology module was applied to small watersheds in Maryland, during a previous study, where climatic and discharge data are more reliable with calibration resulting in a correlation coefficient (R^2) of 0.62 (Gaddis et al. 2006). The snow melt module was the only new component to the hydrology module for the St. Albans Watershed application. This module was calibrated for the pixel cell in which the snow weather station resides in St. Albans to a high level of certainty ($R^2 = 0.82$; Fig. 4).

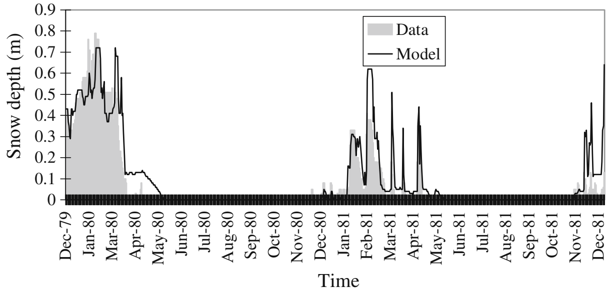


Fig. 4 Calibration of snow depth for St. Albans climate station (1984–1989)

A second method of checking the general reliability of the hydrology modules at an annual scale was comparison of runoff–precipitation ratios from nearby watersheds. Runoff data for the Missisquoi and Lamoille watersheds is available from the US Geologic Survey (waterdata.usgs.gov/nwis/sw). Precipitation data for these watersheds is comprised of an average of several nearby climatic stations in the watersheds available from the National Climatic Data Center (ncdc.noaa.gov). The runoff–precipitation ratio is calculated by dividing the total runoff by the total precipitation in the watershed. Rainfall–runoff ratios were also computed using the simple Thornwaite equation for the St. Albans Bay watershed (Thornthwaite and Mather 1957). Ratios were compared to the output from the landscape model and reported measured discharge for the St. Albans Bay watershed (Table 1). This comparison indicates that the landscape model produces runoff-precipitation ratios that are more similar to ratios computed for nearby watersheds than does the measured data. The hydrologic modules under-predict total runoff from the watershed, which adds an element of conservatism in estimating phosphorus loads, the primary purpose of implementing this model in the St. Albans Bay watershed.

One conclusion from this study was that it is difficult to gather all of the data needed for an appropriate calibration of a detailed process based model, especially in ungaged watersheds. Rykiel (1996) defines model credibility as "a sufficient degree of belief in the validity of a model to justify its use for research and decision making." In this case, the sufficient degree should be evaluated based on the relative risk associated with use of the model results. Since the hydrologic model was primarily used to drive patterns of phosphorus transport, it was assumed that uncertainty in the hydrologic model, which could not be reduced with calibration, are equally distributed across the landscape. The model was used primarily to predict annual loads, as a comparative tool and to communicate to stakeholders the spatial and

Table 1 Comparison of runoff-precipitation ratios for watersheds near St. Albans Bay watershed (1984–1989)

Year	Missisquoi	Lamoille	St. Albans Thornwaite model	St. Albans model	St. Albans measured
1984	63%	58%	51%	47–55%	48–127%
1985	67%	50%	52%	40–50%	20–81%
1986	66%	55%	60%	34–47%	31–158%
1987	59%	48%	62%	36–47%	28–109%
1988	64%	46%	51%	38–53%	19–86%
Average	64%	52%	55%	39–51%	35–103%

temporal timing of phosphorus sources associated with watershed processes. In this way the model is a decision support tool used to compare the relative importance of phosphorus sources and transport pathways, similar to the Spatial Decision Support System (SDSS) developed by Rao and Kumar (2004). The St. Albans Bay Watershed model differs from the SDSS in that it runs on a daily time step and provides the user with information about temporal patterns in the watershed. Stakeholders gain a good understanding of the system and the processes involved which helps to avoid the 'uncertainty deadlock' that often results from black box models. Daily uncertainty in model runs was deemed acceptable and discussed with the stakeholder working group, who plan to use the model results to identify cost-effective implementation projects.

4.2 Phosphorus and Sediment Loads

The watershed model was used to output total phosphorus and sediment movement within each subwatershed as well as total load delivered to streams. The current total phosphorus load to streams in the St. Albans Bay watershed was estimated in a separate part of our research to be 10.6 t/year. The methodology and a discussion of this estimate are available in Gaddis (2007). This value is taken as a starting point for the findings presented in this paper, which focuses on the relative importance of phosphorus sources and transport pathways in the watershed. These values were then converted to area weighted loading rates and compared to other studies in the Lake Champlain basin (Tables 2 and 3).

The total phosphorus load to St. Albans Bay calculated in the Lake Champlain Phosphorus TMDL (7.7 t/year) is slightly higher (11%) than the estimated total phosphorus load to the bay in this study (6.9 t/year; Gaddis 2007). The difference in estimated loads represents model uncertainty in both studies. Our estimate of total phosphorus load to streams in the St. Albans Bay watershed model is 10.6 t/year, which is substantially higher than the load to the bay. The difference between these two loads represents in-stream processing which is not accounted for in the model. The area weighted loading rate of phosphorus to streams in the St. Albans Bay watershed is slightly lower than loading rates modeled for Little Otter Creek watershed, another predominantly agricultural watershed in the southern Lake Champlain Basin (Meals et al. 2008). The area weighted loading rates modeled for cultivated areas in the St. Albans Bay watershed (0.84 kg/ha/year) are higher than

Table 2 Annual area weighted phosphorus loading rates (kg/ha/year) for St. Albans Bay watershed compared to other studies and literature

	St. Albans Bay watershed	Lake Champlain TMDL	Pike River watershed	Little Otter Creek watershed
Phosphorus movement in the landscape	1.20	–	–	–
Phosphorus load to streams	0.84	–	–	1.00
Phosphorus load to bay	0.55	0.61	0.70	–
Reference	This study (Gaddis 2007)	VTANR and NYDEC (2002)	Michaud et al. (2008)	Meals et al. (2008)

Table 3 Annual area weighted phosphorus loading rates (kg/ha/year) for St. Albans Bay land uses compared to other literature

	St. Albans Bay watershed	Lake Champlain TMDL	Pike River watershed	Literature range (kg/ha)
Average of all cultivated sources	0.84	0.42	1.30	0.09–2.66
Corn fields	1.23			
Hay/Pasture fields	0.35			
Developed diffuse sources	0.84	1.50		0.54–1.39
Forest diffuse sources	0.26	0.04		0.09–0.44
Reference	This study (Gaddis 2007)	VTANR and NYDEC (2002)	Michaud et al. (2008)	Hegman et al. (1999)

the loading rate used in the Lake Champlain TMDL (0.42 kg/ha/year) but lower than modeled rates for the Pike River Watershed (1.3 kg/ha/year).

The total sediment load to the bay, estimated with water quality data and modeled hydrologic output, was 970 t/year which equates to approximately 0.1 kg/m^2/year. This value is significantly lower than sediment loading rates in the Pike River watershed to the north of St. Albans. The modeled sediment load to streams for St. Albans Bay watershed was 1,250 t/year. Discrepancies between total sediment estimates to the bay and total sediment estimates to the stream reflect in-stream processes that are captured in the empirical data collected for discharge to the bay but are not captured in the landscape model. It also reflects model uncertainty and error. Comparing sediment load delivered to streams to that delivered to the bay suggests that Jewett, Stevens, and Guayland Brooks contribute sediment load as a result of in-stream erosion whereas Rugg Brook and Mill River accumulate sediment.

5 Results

5.1 Diffuse Sources and Pathways

In total, diffuse sources account for 8.1 t/year or 76% of the total phosphorus load to watershed streams (Table 4). The distribution of phosphorus loss across the watershed is relatively uneven with substantially more phosphorus loss per area from Stevens and Jewett Brook, especially in the City of St. Albans and clay agricultural soils (Fig. 5). The City of St. Albans accounts for a larger percentage (5%) of the load than its area accounts for in the watershed (2%). The remainder of Stevens Brook, also accounts for slightly more diffuse phosphorus (30%) than its relative area (28%). These higher loads are balanced by lower loads coming from Rugg Brook and Mill River. Guayland Brook and Jewett Brook account for the same percentage of the load (4% and 25% respectively) as their areas account for in the watershed (4% and 24% respectively). Corn land use accounts for the majority of diffuse phosphorus to watershed streams especially considering the relatively small percentage of the watershed occupied by this land use type, (Table 5). Developed sources also supply a higher percentage of the total phosphorus diffuse load (19%) than the area occupied by its land use in the watershed. However, hay and pasture

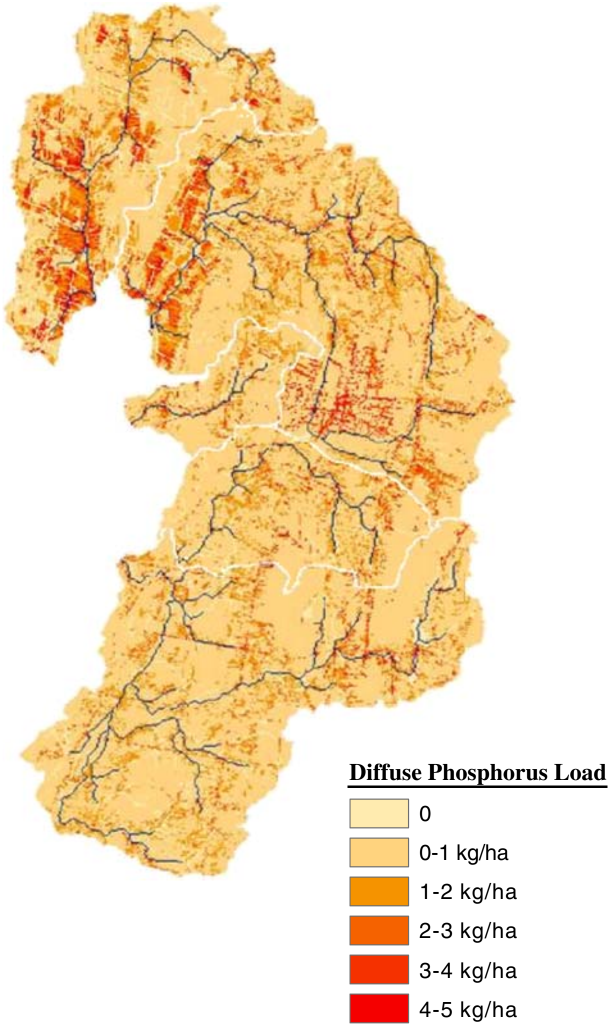
Table 4 Summary of phosphorus loads by transport process or point source

Phosphorus source	Phosphorus load (t/year)	Percent of total to streams
Ag sub-surface drainage (tile)	0.77	7.3%
Ag surface erosion	0.85	8.0%
Ag dissolved surface runoff	4.37	41.3%
Subtotal agriculture	5.99	56.6%
Non-city development road sand	0.98	9.2%
Non-city development dissolved P	0.12	1.1%
Non-city development erosion	0.02	0.19%
Subtotal non-city development	1.12	10.5%
City stormwater road sand	0.28	2.7%
City stormwater dissolved P	0.12	1.1%
City stormwater surface erosion	0.03	0.3%
Subtotal city stormwater	0.43	4.1%
Forest and wetlands	0.51	4.8%
Subtotal forest and wetlands	0.51	4.8%
Geese	0.01	0.1%
Subtotal geese	0.01	0.1%
Subtotal diffuse sources	8.06	76%
Milk house	0.13	1.2%
Manure runoff	0.75	7.1%
Silage leachate	0.70	6.6%
Subtotal farmstead discharge	1.58	14.9%
P in sewage/storm treated	0.84	8.0%
P in sewage/storm overflow	0.07	0.7%
NWCC discharge	0.003	0.03%
Subtotal wastewater	0.91	8.7%
Subtotal point sources	2.49	24%
Total	10.6	100%

lands supply a much smaller percentage of the total load than the area occupied by this land use, suggesting that as land is converted from hay and pasture to corn and/or development, phosphorus losses increase. Both trends are occurring in the watershed. Forest and wetland land uses supply a much smaller proportion of phosphorus than their area, which is expected as these land uses are relatively undisturbed. The forest estimates could be high as they are driven by soil phosphorus concentrations which were not directly sampled but interpolated from minimum values of similar soil types in other land uses in the watershed.

Dissolved phosphorus in surface runoff from agricultural fields accounts for 41% (4.37 t/year) of the total load to streams from the landscape and is the single most important diffuse source of those modeled in the St. Albans Bay watershed (Table 4). Clay soils, in Jewett Brook and Stevens Brook drainages, release more phosphorus in this process per area than other soil types (Fig. 6). This results from the high soil P concentrations found in many clay soils in the watershed compounded by high surface runoff from clay soils.

Fig. 5 Map of total phosphorus load delivered to streams in the St. Albans Bay watershed



The process of surface erosion accounts for 8% (0.85 t/year) of the total phosphorus load to streams. The process is relatively well distributed across the landscape with areas adjacent to streams and with a high slope contributing more phosphorus from surface erosion than other areas (Fig. 6). The erosion output from the spatial model represents only surface erosion of sediment, and the phosphorus attached to it. It does not include in-stream erosion which is an important component of the total watershed load resulting from hydrologic changes such as peak flows associated with stormwater runoff.

Tile drains in agricultural areas represent an important source of phosphorus to surface waters, providing approximately 7% (0.77 t/year) of the total load to streams. The highest tile drain loads come from the clay soils in Stevens Brook and Jewett Brook drainages (Fig. 6), which results from the high concentration of phosphorus in tile drainage water from clay soils compared to other soils.

Table 5 Summary of all phosphorus loads by land use

	Phosphorus load (t/year)	Percent of total P load to streams	Percent land use in watershed
Corn diffuse sources	4.92	46%	22%
Hay/pasture diffuse sources	1.12	11%	35%
Total Ag diffuse sources	6.04	57%	57%
Ag point sources	1.58	15%	57%
Total agricultural	7.62	72%	57%
Developed diffuse sources	1.52	14%	14%
Developed point sources	0.91	9%	14%
Total developed	2.43	23%	14%
Forest diffuse sources	0.45	4%	23%
Other diffuse sources	0.06	1%	6%
Total	10.56	100%	100%

Road sand wash-off represents 12% (1.26 t/year) of the total load of phosphorus to streams in the St. Albans Bay watershed (Fig. 6), which is significant considering it is a source that had not been identified prior to this study and could be a relatively easy source to reduce by the urban community. It is a recognized source in other northern watershed studies (Oberts 1986) and as the St. Albans Bay watershed becomes more developed, this could become a more significant source.

5.2 Semi-Diffuse and Point Sources of Phosphorus

Although the phosphorus problem in the St. Albans Bay watershed is primarily a diffuse problem, there are several processes that cannot be captured with the spatially explicit model. Sources associated with farmstead runoff are semi-diffuse in that they represent concentrated diffuse sources that drain to a specific, though unknown, point along a stream. Waterfowl are also semi-diffuse in that geese concentrate in specific locations in the watershed. Wastewater treatment effluent

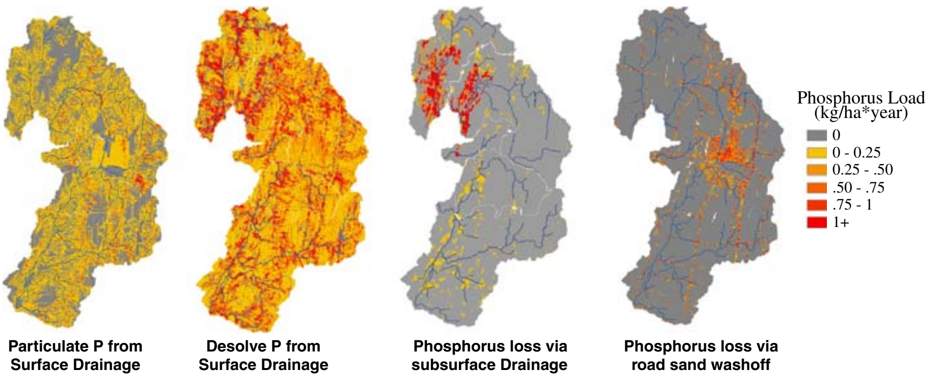


Fig. 6 Maps of diffuse phosphorus load to streams for each phosphorus transport pathway

represents the only regulated point sources in the watershed. Together all of these sources account for 2.5 t/year, or 24% of the total load to watershed streams.

The estimated total phosphorus load to streams from farmstead discharge is 1.58 t/year representing 15% of the total load to streams in the watershed. This load can be further broken down into milk house effluent (0.13 t/year), manure runoff (0.75 t/year), and silage leachate (0.7 t/year).

Geese are responsible, on average, for mobilizing 0.005 to 0.01 metric tons of phosphorus per year throughout the St. Albans Bay watershed, less than 0.1% of the total load to watershed streams.

The wastewater treatment plant discharges an estimated load of 0.91 t/year to Stevens Brook, representing 8.6% of the total watershed load to streams. Of this 0.84 t/year is discharged as treated sewage and 0.07 t/year as wastewater overflow during storms. This load is still well below the load allocation in the Lake Champlain Phosphorus TMDL and permitted discharge for the wastewater treatment plant of 2.8 t/year (VTANR and NYDEC 2002).

A much smaller wastewater treatment plant for the Northwest Regional Corrections Facility also discharges phosphorus to Jewett Brook. Average annual phosphorus load from this facility is estimated to be 0.003 t/year which is approximately ten times less than the allocated load in the Lake Champlain TMDL for this facility (0.028 t/year) but very close to the estimate provided for 2001 in the TMDL (0.004 t/year).

6 Discussion

6.1 Importance of Identifying Phosphorus Transport Pathways to Achieve TMDL Targets

Although diffuse pollution from particular land uses are typically lumped together in TMDL documents, the actual transport processes that make up diffuse pollution may vary across the landscape. For example, phosphorus may leave the landscape through processes of erosion, subsurface drainage, or in a dissolved form in water that runs off the landscape. Interventions required to address each of these processes are very different. Net source reduction of pollution is a key component to any long-term solution to phosphorus problems. Net reduction includes interventions that either reduce the import of phosphorus to the landscape or increase the export of phosphorus in the form of biomass, compost, etc. Process alterations include temporal buffering and stabilization of a pollutant in the landscape. Whereas the former refers to a mechanism by which an intervention might reduce pollution or heavy flows temporarily, the latter refers to sequestration of a pollutant in soil, plant matter, or other stable forms thus reducing the net movement of pollutants in the landscape. Enhanced sinks or capture interventions aim to recapture phosphorus that is moving across the landscape before it reaches water bodies or attenuate it in biomass or soil.

The list of feasible watershed interventions for the processes and sources identified in this study is extensive and ranges from structural changes implemented on a centralized scale, such as stormwater ponds and treatment, to behavioral changes at the level of individual homeowners and farmers, such as eliminating the use of excessive fertilizer in home gardens and lawns. In order to attain TMDL diffuse pollution reduction targets, phosphorus sources and transport processes must be matched with appropriate watershed interventions (Fig. 7).

SOURCE REDUCTION	PROCESS ALTERATION	CAPTURE
------------------------Ag Dissolved Runoff (41%)------------------------>		
Reduce fertilizer (NMP) Convert manure to export product Reduce phosphorus in feed (NMP) Use exclusion	Tillage Timing of fertilizer/manure application (manure storage) Tiling fields	"EAF steel slag barriers for surface runoff P reduction"
------------------------Farmstead Discharge (15%)------------------------>		
Pasture more animals	Manure storage	Farmstead treatment
------------------------Road Sand Washoff (12%)------------------------>		
Reduce use of road sand Road sweeper	Reduce impervious cover "Better back roads"	Sediment traps Centralized detention ponds and treatment
------------------------Wastewater Discharge (8.6%)------------------------>		
Storm sewer separation	Expand wastewater treatment plant	Improve treatment capability (below 0.5 mg/l)
------------------------Ag Surface Erosion (8%)------------------------>		
Reduce fertilizer (NMP) Convert manure to export product Reduce phosphorus in feed (NMP)	Cover crops Tillage Tiling fields	Buffers and/or filter strips
------------------------Subsurface Field Drainage (7.3%)------------------------>		
Reduce Soil P (reduce fertilizer and manure application)	Un-tile	"EAF steel slag barriers for surface runoff P reduction"
------------------------In-stream Erosion (Unknown)------------------------>		
Retain water in landscape (centralized & local retention) Reduce impervious cover Detach roof drains from storm pipes	Armoring stream banks Stream buffering Restore geomorphology	Dredging sediments In-stream detention
------------------------Developed Dissolved Runoff (2.2%)------------------------>		
Reduce commercial and residential fertilizer usage	Local retention (i.e. rain barrels, rain gardens, infiltration trenches) Reduce impervious cover	Centralized detention ponds and treatment
------------------------Developed Surface Erosion (0.5%)------------------------>		
	Local retention (i.e. rain barrels, rain gardens, infiltration trenches) Reduce impervious cover "Better back roads" program	Sediment traps Centralized detention ponds and treatment

Fig. 7 Summary of watershed interventions categorized by process and reduction mechanism

6.2 Opportunities for Innovative Implementation Efforts for the St. Albans Bay Watershed

Several significant sources of phosphorus identified in this study could be addressed through watershed interventions that have not yet been implemented in the St. Albans Bay watershed. Phosphorus in road sand wash off and field drainage water are two important sources in the St. Albans Bay watershed that are not specifically identified in the Lake Champlain Phosphorus TMDL, although they are imbedded in loads associated with agricultural and developed land uses. Road sand applied during winter storms that washes off streets during spring melt and storm events contributes 12% of the phosphorus to streams in the watershed. This could be one of the most cost-effective sources to reduce in the short-term either through improved road sweepers or replacement of sand with other deicing mechanisms. Tile drainage from fields represents 7.3% of the total phosphorus loss to streams and offers a good opportunity for innovative implementation efforts. Drainage water is collected in a series of small pipes that discharge directly to ditches and stream channels. These discharges are effectively small point sources that could receive some sort of pre-treatment prior to discharge.

Dissolved phosphorus leaving agricultural fields through surface runoff represents 41% of the total phosphorus loss to streams in the watershed. Leaching of dissolved phosphorus into surface drainage requires correction of the net imbalance of phosphorus imported to the watershed (Gaddis 2007). In addition, new technologies are being developed to help mitigate the impact of dissolved phosphorus runoff through phosphorus sink enhancement (Drizo et al. 2006).

Phosphorus from farmsteads are an important source of phosphorus that is currently imbedded into the agricultural diffuse phosphorus estimate in the Lake Champlain Phosphorus TMDL. Farmstead discharge is a semi-diffuse source which offers the opportunity for treatment and phosphorus load reduction using existing technologies.

6.3 Spatial Targeting of Watershed Interventions

The majority of the diffuse phosphorus load comes from the Stevens Brook watershed, which accounts for more load than its relative area in the watershed. Much of this imbalance comes from the City of St. Albans. Treatment of the concentrated stormwater load in the city is a priority for achieving TMDL reduction targets for developed land uses. Across the agricultural landscape the highest phosphorus loads come from clay soils that are saturated with phosphorus and are found primarily in the Jewett Brook and lower Stevens Brook subwatersheds. These areas should be the priority for agricultural BMP implementation in the future (Fig. 7).

6.4 Conclusions

The results of this study indicate that the majority of phosphorus loss to streams in the St. Albans Bay watershed is diffuse. Of the diffuse sources, the most important source is dissolved phosphorus in agricultural surface runoff and tile drain water followed by road sand wash-off in the developed landscape. Direct discharge to streams from farmsteads also represents a significant load to streams of which the majority comes from barnyard manure runoff and silage leachate. Addressing these four sources with available and innovative technology and management practices would be the most effective means to move towards achievement of TMDL targets.

This study offers a mechanism to identify the relative importance of specific transport processes of pollutants from the landscape to streams and could inform the types and locations of BMPs recommended for pollutant reduction. The approach has implications for setting realistic TMDL diffuse pollution allocations and could help communities attain targets in a more focused manner by targeting efforts to the transport processes that are most important, the land use and soil types that appear to be particularly problematic, as well as areas of the watershed that are the most impaired. This approach could be replicated for other TMDL processes around the country in designing more effective Watershed Management and Implementation Plans for TMDL attainment.

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