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Editorial

Using competitive grants to address water resource challenges at the local level: The Conesus Lake Watershed project

Index words: Water quality, Agriculture, USDA, Management practices, Lakes

The Cooperative State Research, Education, and Extension Service (CSREES) of the U.S. Department of Agriculture (USDA) was created in 1994 and is the leading source of extramural funding for agricultural research, education, and extension projects and programs (http://www.csrees.usda.gov/). The CSREES National Water Program offers two competitive grant programs: the National Research Initiative (NRI) Water and Watersheds Program and the National Integrated Water Quality Program (NIWQP). Funding from these two programs is used to create and disseminate knowledge that insures a safe and reliable source of water of the appropriate quality and to meet the needs of:

- Food, fiber, and energy production;
- Human health, use, and economic growth; and
- Maintenance and protection of natural environmental systems and ecosystem functions.

CSREES' unique niche is conducting research, education, and extension programs to protect and improve water resources in agricultural, rural, and urbanizing watersheds including forest lands, rangelands, and croplands — agricultural working lands. Funded projects lead to science-based decision making and management practices that improve the quality of the USA's surface water and groundwater resources in these watersheds. Technology and product development to promote improved water resource management also is encouraged through Small Business Innovation Research programs sponsored by CSREES.

The CSREES National Water Program unites social, economic, and environmental concerns with research and education devoted to "scaling-up" the cumulative effects of site-specific actions on rangelands, forests, agricultural lands, and rural communities. CSREES works through a national network of water resource projects to engage stakeholders in the watershed management process, which includes watershed scale planning and implementation, resulting in changing attitudes and behaviors that reduce contamination throughout watersheds and consequently improve water quality to downstream systems while increasing the scientific and public understanding that human well-being is inextricably linked to the sustainable use and management of ecosystems. Activities of all land uses within the Great Lakes' watershed impact the water quality of downstream water bodies. Water resources in the Great Lakes region are part of complex natural systems that are increasingly viewed as being at risk due to a variety of chemical and biological pollutants. Point and nonpoint sources of pollution in a watershed contribute nutrients, bacteria, and chemical contaminants that are potential health threats to drinking water and ultimately human health. Identifying a water quality problem in receiving waters is often the first step in the watershed management process. Effective watershed management results in a reduction of contaminants within watersheds and in the improvement of water quality and is an on-going process that must be flexible enough to adapt to the unique characteristics of different watersheds as well as to the changing circumstances within a single watershed.

Water quality monitoring is also critical to ensure that watershed management strategies are improving water quality. Research within the National Water Program is improving sampling design and watershed monitoring approaches, enhancing monitoring methods and techniques, and examining data credibility. Research, such as the CSREES' competitively funded research on the Conesus Lake watershed, improves the understanding of local water resources and helps communities make informed decisions that improve water quality. With people in the Great Lakes region depending on a safe water supply for both drinking and recreation, it is critical to protect these waters by conducting quality research, disseminating results, and educating citizens concerning the wise use and management of these waters.

Watershed management recognizes that the water quality of our streams, lakes, and estuaries results from the interaction of upstream features. Because of the loss of soil from the land and the effects this may be having on downstream environments, local agricultural agencies with participation of local farmers joined with scientists to take an integrated and collaborative approach to focus attention on the Conesus Lake watershed to foster their sense of stewardship and to assist and coordinate collaboration between academic researchers, governing bodies, and the agricultural community. This special edition represents an integrated approach to conduct hypothesis-based research at the watershed level that has tested the impact of Best Management Practices (BMPs) on mitigation of nonpoint sources of nutrient and soil loss. The goal of the Conesus Lake Watershed project was to demonstrate, through the experimental watershed approach, that implementation of BMPs in agriculturally dominated watersheds will preserve soil and reduce nutrient loss from a series of sub-watersheds while having positive impacts on the downstream lake community, especially the nearshore of a large lake.

The goal of the USDA National Water Program is to protect and improve the quality of water resources throughout the United States and its territories and has identified Watershed Management as a theme on which to focus these efforts. This special issue highlights a critical interest of the CSREES Water Program — restoration of aquatic and estuarine ecosystems impacted by agriculture. From 2005 through 2007, the NIWQP included a priority to fund watershed-scale projects

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that addressed "hydrologic, geomorphic, and ecologic conditions necessary to restore the structure and function of aquatic or estuarine ecosystems impacted by agricultural water use or agricultural nonpoint source pollution." These projects combined research, education, and extension activities to address impacts of agriculture on water resources at the watershed scale. Successful projects were required to involve stakeholders in project management — seamlessly integrating discovery through research with dissemination through outreach and extension. Projects also offered new opportunities for students — focusing education on holistic approaches to watershed restoration. Overall, CSREES supported 11 competitively reviewed watershed-scale projects nationwide aimed at identifying and restoring impacts of agriculture on water resources. The papers in this special issue are products of CSREES' funding in support of this restoration focus.

Using the small watershed approach, the "Conesus Lake Watershed Project" directly targets the identification and resolution of agriculturally related degradation of water quality to address issues of concern to the region and the nation. At Conesus Lake, these included the following:

- Evaluate the efficacy of currently recommended management practices and strategies to improve water quality;
- Assess the sources of water quality impairment in targeted watersheds;
- Evaluate the impact of land management practices on downstream bacteria, algae and macrophytes;
- Develop and recommend options for continued improvement of water quality in targeted watersheds;
- Evaluate the relative cost and benefits associated with cleanup from all responsible sectors; and
- Develop and validate cost-effective means to reduce the contribution of agriculture to the development of algal blooms, including harmful algal blooms.

Lastly, this issue highlights the importance of integrating research, education, and extension in solving watershed problems. Research papers in this issue describe complex linkages between physical and biological processes that control nearshore eutrophication in deepwater lakes. Papers highlighting the Extension component of the work demonstrate the need to establish partnerships with local landowners and build trust among landowners and the project team. Finally, the importance of education is exemplified by young professionals completing undergraduate (7) and graduate (4) degrees participating as co-authors on these papers. Their professional development intricately links research and extension aspects of the project. Through extension and education, the research completed demonstrates to the farming community the utility and effectiveness of the implemented BMPs. The research allows regional policy makers and managers to develop strategies not only for improving land usage in watersheds but also for improving water quality and decreasing abundance of nuisance plant species in downstream ecosystems. As such, the results of this project are a logical step, a catalyst, and a mechanism for the farming community to be proactive in watershed issues.

Special Editor's Note: The Cooperative State Research, Education, and Extension Service (CSREES) will transition to the National Institute for Food and Agriculture (NIFA) by 1 October 2009.

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Nonpoint source reduction to the nearshore zone via watershed management practices: Nutrient fluxes, fate, transport and biotic responses — Background and objectives

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ABSTRACT

Studies that evaluate the linkages between watershed improvement through Best Management Practices (BMPs) and downstream outcomes are few. Water quality of coastal waters is often impacted by soil and nutrient loss from watersheds in agriculture. Mitigation of these impacts is of concern in the Great Lakes, the Finger Lakes Region of New York State, and generally in water bodies of North America. In this issue, we report on hypothesis-based research at the watershed level evaluating the impact of BMPs on mitigation of nonpoint sources of nutrient and soil loss to streams and the nearshore zone of a lake. Specifically, we hypothesize not only reductions in nutrient and soil losses from watersheds but also a resultant decrease in metaphyton (filamentous algae), coliform bacteria, and macrophyte populations in the nearshore at stream mouths draining sub-watersheds where BMPs were introduced. Small experimental sub-watersheds, predominantly in agriculture (>70%), were selected to ensure that effects on downstream systems would not be confounded by other land use practices often observed in large watershed approaches. In this introductory paper, we provide background information on Conesus Lake, its watershed, and the Conesus Lake watershed project, a large multi-disciplinary study evaluating agricultural management practices. The series of papers in this volume consider the effect of BMPs designed to control nonpoint sources on water chemistry, metaphyton, macrophytes, and microbial populations in the coastal zone of a lake. Ultimately, this volume expands the basic understanding of the ability of BMPs to control nonpoint source pollution while contributing toward the goal of improving water quality of downstream systems including streams, embayments, and the nearshore of large lakes.

Introduction

Lake Ontario coastal waters are a valuable resource for drinking water and industrial usage, recreational boating, fishing and swimming, tourism, and wastewater processing and are a key asset in the economies of upstate New York and Ontario Province. Article 14 of the New York Ocean and Great Lakes Ecosystem Conservation Act 2006 states "… coastal ecosystems are critical to the state's environmental and economic security and integral to the state's high quality of life and culture." Yet in Lake Ontario, many of the bays, rivers and drowned river mouths, as well as the coastal zone of the south shore of Lake Ontario, are suffering from high turbidity, sedimentation, nutrient enrichment, and algal blooms that are often associated with agricultural land use (Makarewicz and Howell, 2007). Sediment loads, nutrient concentrations, and Cyanobacteria appear to be higher in the streams and embayments and at shoreside sites compared to offshore sites west of the Genesee River. Phosphorus (P) levels often exceed the New York State Department of Environmental Conservation (NYSDEC) Ambient Water Quality Guideline for P while in the Province of Ontario, total phosphorus (TP) levels do not generally exceed the Provincial Water Quality Objective. In the coastal zone there are many locations, such as embayments, river mouths, and locations near the shoreline, where TP will periodically, if not frequently, exceed the NYSDEC ambient guideline of 10 or 20 μg/L (T. Howell, Personal Communication, Ontario Ministry of the Environment).

Whether it be Lake Ontario or the Finger Lakes region of New York State (Fig. 1), public beaches are often closed or posted due to elevated levels of fecal pollution indicators and poor water quality. Elevated levels of fecal indicators may result from factors other than strictly poor water quality in a conventional sense (e.g., losses from farming operations, beach sediments, gulls). In Lake Ontario, water quality of the coastal zone is generally poorer than water from the offshore zone (Makarewicz and Howell, 2007). Structure and function of the littoral zone, whether it be a Finger Lake or a Great Lake, are complex, variable, and influenced by the proximity of the shoreline, localized sources of meso-scale variability (e.g., tributaries, land use in the watershed, embayments, geology, effluent pipes), and variations in the current regime (wind direction, upwellings, etc.), as

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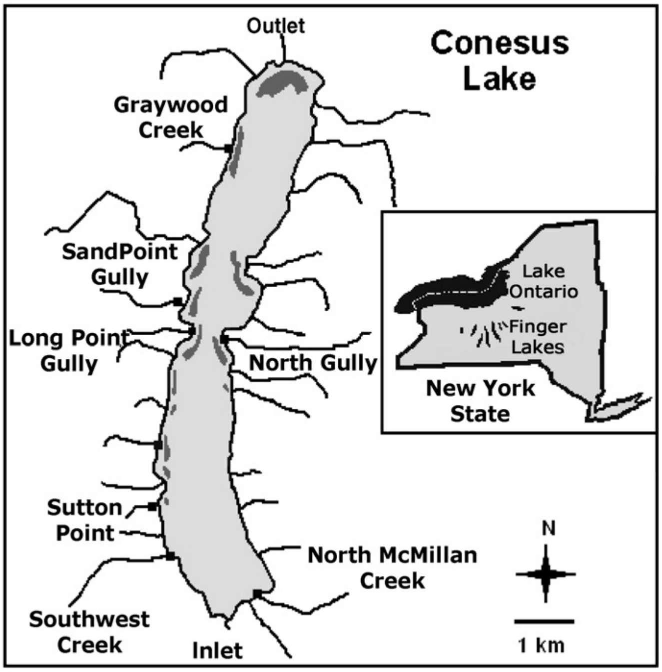

Fig. 1. Macrophyte beds in Conesus Lake (N 42° 46.784′, W 77° 43.068′), NY, USA. Squares represent stream discharge monitoring sites. Irregular areas near streams are GPS identified macrophyte beds. Adapted from D'Aiuto et al. (2006). Mean depth = 11.5 m, Maximum depth = 20.2 m (Forest et al., 1978).

well as by land use and nonpoint source loading. Nonpoint source impacts of agricultural practice on water quality of lakes via tributary-delivered loads of pollutants and the potential role of management practices to reduce impacts are the primary concerns of this volume.

Since the 1970s, environmental management at the watershed scale has emerged as a promising procedure to deal with water quality problems in general (Hawkings and Geering, 1989) and in particular with those associated with agricultural land use (Staver et al., 1989; Whitelaw and Solbe, 1989). Water pollution from areas of intense livestock production is caused by deficiencies in the management of animal manure (e.g., lack of appropriate storage facilities, no treatment of feedlot runoff, application problems) and milk house wastewater (e.g., direct discharge into water courses) and in inadequate fertilization processes caused by excessive or untimely field applications of manure or fertilizers in general (Cooper and Lipe, 1992; Clausen et al., 1992). Jones et al. (2004) have suggested that nonpoint sources (cropland) account for 60 to 70% of the variation in nutrient chemistry in Missouri reservoirs; that is, there is a positive correlation with increasing TP and total nitrogen with increasing cropland in a watershed. Similarly, Makarewicz et al. (2007) and D'Aiuto et al. (2006) have suggested that a positive correlation existed between percent cover of nearshore metaphyton (filamentous) with the amount of agricultural land use and that macrophyte standing crop is positively correlated with TP loading.

The demonstrated losses of soil and nutrients from agricultural activities in the watershed and the probable effects on downstream communities are serious issues locally, regionally, and nationally. Herein lies the growing dilemma for governmental leaders in agricultural areas: their most important economic industry (agriculture) may also be the cause of environmental degradation which heightens public concern. For farmers, this is further exacerbated by the high profile increase of governmental regulation on agricultural operations [e.g., Total Daily Maximum Loads (TDML), Concentrated Animal Feeding Operations (CAFO)]. The agricultural industry needs scientific evidence that farmers are capable of being part of the solution, not just part of the problem.

In P-limited lakes, theory states that nutrient reduction from impacted watersheds will improve water quality by reducing limiting nutrients thus decreasing phytoplankton, metaphyton, and macrophyte populations (Osgood, 1999). Best Management Practices (BMPs), if properly implemented, should lead to a reduction of nutrient and soil loss from the watershed and to nutrient declines in the water column of the lake. In Lake Erie for example, massive reductions in point loads of P led to significant improvements in water quality, reductions in phytoplankton populations, and reductions in nuisance algal species (Makarewicz and Bertram, 1991).

In reference to watersheds and lakes, the "Phosphorus Paradigm" is that P reduction will reduce algae and macrophyte populations and will improve water clarity and quality. However, the evidence for a connection between nutrient reductions by BMPs and reductions of nuisance plant growth is not as strong as it is generally believed (Osgood, 1999). Most examples of successful restoration of lakes suffering from overloads of P represent "point" source reduction or control: that is, sewage effluent that has been treated or diverted (Moss et al., 1997). Dealing with nonpoint sources has proven to be more difficult. Phosphorus in runoff usually comes from multiple sources such as storm sewers, fertilized lawns, or agricultural fields. In addition, the treatment or management practice is normally a low-tech approach resulting only in a partial reduction in P compared to much higher rates of removal for point source loading. As a result, it is not possible to totally eliminate the limiting nutrient P. Unlike point sources which technically could be turned off or reduced to a regulatory standard, nonpoint sources can only be turned down and with great difficulty because of the large spatial component.

The assumption of the "Phosphorus Paradigm" that we can obtain the same results from reduction of nonpoint loading as point source loading has to be critically reviewed (Osgood, 1999). The proposed BMPs being tested in this study selectively target nutrient and erosion management such as animal waste handling (lagoons, timing of manure spreading) and erosion prevention (terracing, buffer strips). Clearly, several studies have shown the "local" effectiveness of BMP implementation in reducing nutrient loss, soil loss, and bacteria counts (Cook et al., 1996; Meals 1996, 2001; Gilliam, 1995). For example, with livestock operations a >90% reduction in P loss to immediate downstream locations was achieved by capturing and storing manure in a lagoon (Gilliam, 1995). Phosphorus export from cornfields was 1500% higher where manure was spread in the winter rather than in the spring (Meals, 1996). It follows that seasonal timing of manure spreading is critical to reducing P losses and other nutrients, such as nitrate, from a watershed to downstream systems.

However, from a watershed perspective, local effects did not always translate to ecosystem or watershed-wide reductions in nutrients and soil loss (Meals, 1996). In fact, BMPs introduced in small portions of watersheds in the Finger Lakes within the Lake Ontario watershed have not effectively demonstrated a connection between nutrient and erosion reduction and visual or measurable nearshore, offshore, or lake-wide reductions of nutrients, metaphyton, or aquatic plant populations (Bosch et al., 2001). This inability to demonstrate mitigation of stresses caused by nutrient enhancement in downstream systems is the result of confounding factors inherent in nonexperimental evaluations and/or the use of very large watersheds where a single manipulation (i.e., a BMP) of a small area will not provide a large enough reduction in nutrients to affect significant change in downstream nutrient concentration, nutrient loading, and metaphyton and macrophyte population size. In the Finger Lakes Region of New York, the lack of demonstrable evidence of the success of selected BMPs on reducing nuisance species of lake plants has inhibited the ability of managers and planners to convince the agricultural community to voluntarily initiate expensive changes through BMPs (P. Kanouse, Personal Communication, Livingston County Soil and Water Conservation District). In some cases within the Finger Lakes Region, individual farmers fail to recognize that some

Table 1
Nutrient and soil loss from selected Conesus Lake watersheds during events in autumn, 2000.

Watershed	Percent agriculture	Watershed area (ha)	Event loading data (g/ha/day, TSS = kg/ha/d)				
			NO_3	TP	SRP	TSS	TKN
Graywood Gully	74	33.8	1,202	34.7	30.7	1,651	3.7
Sand Point Gully	83	325	54.9	5.4	3.2	1,310	4.1
Cottonwood Gully	75	76	180	9.1	3.8	3,749	3.3
Sutton Point Gully	76	62.2	43.5	1.0	0.7	77.0	3.1
Long Point Gully	86	622.5	590	11.1	6.4	1,807	21.4
North McMillan Creek	12	2045	9.5	0.43	N.D.	426	5.64

NO_3 = nitrate, TP = total phosphorus, SRP = soluble reactive phosphorus, TSS = total suspended solids, and TKN = total Kjeldahl nitrogen.

agricultural practices are leading to major losses of soil and nutrients from their fields.

Ecosystem experiments are critically needed to improve environmental management practices and policies (Carpenter, 1998). Ecosystem experiments are a powerful tool for evaluating and predicting impacts of environmental change (Carpenter et al., 1995). The small watershed approach allows evaluation of land use management techniques. The assumption is that as water passes through the terrestrial ecosystem, it may be altered in chemical composition by management practices that reflect or can be attributed to known biological and geochemical processes (Carpenter et al., 1995). Such large-scale experiments have significant advantages over small artificial systems (e.g., lysimeters, plot analysis) (Carpenter, 1996; Carpenter et al., 1995). The fundamental problem with learning from small-scale experiments is that results must be translated across scales to draw conclusions about ecosystems (Carpenter, 1998) and can lead to spurious results (Schindler, 1998). Even interpolation from small watershed experiments, such as employed in the Conesus Lake Watershed Study, across diverse landscapes is a challenge. The scale of ecosystem experimentation is especially useful in environmental management because it is very difficult to convince managers or other stakeholders to change policies using complex arguments and extrapolations (Lee, 1993).

Here, background information is provided on Conesus Lake, its watershed, and a multi-disciplinary project conducting longer-term (1 Sep 2002 to 31 Aug 2007) hypothesis-based research at the watershed level designed to evaluate management practices associated with nonpoint sources of nutrient and soil pollution to downstream systems. Research at Conesus Lake had demonstrated that loss of nutrients and soils from some watersheds in agriculture is high compared to other areas with less agriculture (Table 1). Furthermore, the location and abundance of large macrophyte beds and metaphyton at the mouths of creeks in this Finger Lake appear to be a function of high nutrient loads from watersheds associated with agricultural land use (Figs. 1 and 2, D'Aiuto et al., 2006).

Our general hypothesis is that BMPs [see Herendeen and Glazier (2009) for a complete description of BMPs] implemented in experimental watersheds will lead to reductions in nutrient loads, metaphyton growth and abundance, fecal pollution as inferred from indicator bacteria, and reductions in the size and biomass of macrophyte beds in the nearshore near the mouths of streams draining managed watersheds. The extensive localized growth of metaphyton and macrophyte beds at the mouth of streams draining agriculturally dominated sub-watersheds offers an opportunity to experimentally test the effects of nutrient and soil BMPs implemented in small sub-watersheds on biota at the base of affected watersheds rather than on phytoplankton populations in the offshore of the lake. Since management plans were introduced into only 5 of 18 sub-watersheds, the effects from BMPs implemented on a few small sub-watersheds within the catchment are not expected to affect the phytoplankton of the offshore community. Before improvements are observed in the offshore, management plans would need to be introduced to other sub-watersheds to overcome the cumulative impacts from the 15 non-managed sub-watersheds.

The study site

Conesus Lake and its watershed

The Conesus Lake watershed is located in a broad glacial valley and is part of the Genesee River basin that drains into Lake Ontario. Approximately 9800 people live in the seven municipalities within the 180.5-km^2 watershed of Conesus Lake. Conesus Lake was one of the first lakes in New York State to have a perimeter sewer system, which was completed in 1972 with 26 pumping stations and a processing treatment plant that releases treated effluent into a stream draining away from the lake. The general climatic conditions can be described as humid continental with warm dry summers and cold snowy winters. Average yearly precipitation is approximately 80.5 cm. Conesus Lake is fed by 18 tributaries and a number of smaller streams and rivulets (Forest et al., 1978). The terrain in the watershed is characterized by gentle slopes at the northern outlet and southern inlet areas. Steep hilly slopes characterize the flanks and southern portion of the watershed. Elevation ranges from 249 m above sea level at the lakeside to about 549.9 m above sea level at the southern edge of the basin along the divide between the headwaters of the Conesus Inlet and South McMillan Creek (Forest et al., 1978). From the middle third of the lake to the southern end of the watershed, the lake and valley are flanked by steep slopes exceeding 45%.

The soils of the Conesus Lake watershed are mostly derived from locally-occurring shale and sandstone bedrock material that has been reworked by glacial action (Bloomfield, 1978). Towards the north of the watershed, limestone materials transported by the glaciers from the central NY limestone belt influence the soil. This influence is less as one moves south, and in general, soils are more agriculturally productive to the north of the watershed compared with the south (Stout, 1970). The soils vary widely in other

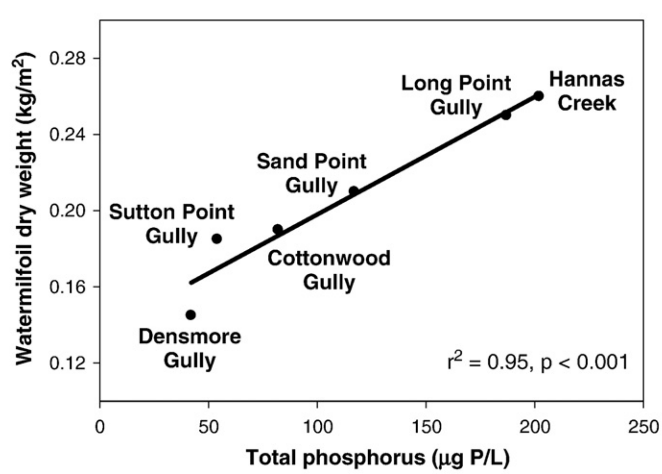

Fig. 2. Relationship between average stream event total phosphorus concentration and stream mouth Eurasian watermilfoil biomass in Conesus Lake, fall 2000.

properties of significance to land use management and water quality impacts. Many of the soils are highly susceptible to erosion, presenting the risk of sediment or sediment-borne nonpoint source pollution. Other soils are poorly drained, which make them likely to be important surface runoff generation areas. They are also risk zones for generation of nonpoint source pollution. Where soils of poor drainage exist, drainage tiles have often been installed (see Noll and Magee, 2009). Overall, the soils of this watershed present a diverse and complicated mosaic of management imperatives — they prescribe land use decisions at the field scale.

In 1999 about half of the entire land use within the Conesus Lake watershed was and continues to be in agriculture. Much of the agriculture (>70%) is concentrated in the western sub-watersheds of the lake (Fig. 3, SOCL, 2001). The deep, glacially-derived limestone soils that dominate the watershed are productive and support field crops (mostly corn), an occasional vineyard, and livestock. The vast majority of the livestock operations are in dairy operations while field crops are mostly for livestock feed. The lake is also a source of drinking water to ~15000 residents inside and outside of the watershed boundaries. As a recreational attraction, the lake is heavily used for swimming, boating, fishing, and aesthetic enjoyment. Economically, lake-based tourism is substantial but agriculture is predominant (SOCL, 2001).

Streams

In the Conesus Lake catchment, sub-watersheds with a large percentage of land in agricultural practices are losing large amounts of soil and nutrients (soluble P, nitrate, and organic nitrogen), especially during hydrometeorological events (Makarewicz et al., 1999, 2001, 2007; SOCL, 2001). Segment analysis, a process of subdividing a stream into segments and performing water analyses above and below the segment (Makarewicz and Lewis, 1999), demonstrated that the nonpoint sources are of agricultural origin in several sub-watersheds (Makarewicz, 1992, 1993, 1994). Average concentrations of TP during baseline flow in six streams draining agriculturally dominated catchments (60 to 80% of the land use) ranged from 41.9 to 245.3 μg P/L in 2000 (Makarewicz et al., 2001). In contrast, summer epilimnetic lake TP concentrations ranged from 15.0 to 38.3 μg P/L. During hydrometeorological events, levels of tributary TP ranged a magnitude higher (258 to 1313 μg P/L) than during nonevents as materials were washed off the landscape and carried downstream (Makarewicz et al., 2007). The mass loss from the watershed, that is, the loading into the lake from the watershed of nitrate (up to 1800 g N/ha/day) and TP (up to 34 g P/ha/day) during hydrometeorological events, was high (Makarewicz et al., 2001). A concentration gradient, high to low, existed from the tributary to the nearshore zone and on into the offshore region of the lake (Makarewicz et al., 2007). The distribution of nutrient-enriched water from streams into the nearshore of the lake is determined by the nature of the entry plumes of streams and lake circulation (Trexler et al., 2006). Li et al. (2007) have documented the flow regimes in and around two creeks in Conesus Lake. This three-dimensional, macrophyte-drag hydrodynamic model demonstrates that nutrient-laden stream water is focused as a result of currents, local bathymetry, and prevailing winds into a macrophyte bed where metaphyton are prevalent. The plume development at stream mouths in Conesus Lake during storm events is site-dependent and may either be current or wind driven. The nutrient-laden stream water draining sub-watersheds predominantly in agriculture has a major impact on nearshore metaphyton biomass and production (D'Aiuto et al., 2006, Makarewicz et al., 2007) and probably on macrophytes (Fig. 2) and cumulatively on the offshore region of the lake. Also, the stream water draining the sub-watersheds carried microbial populations into the lake. Somarelli et al. (2007) assessed the

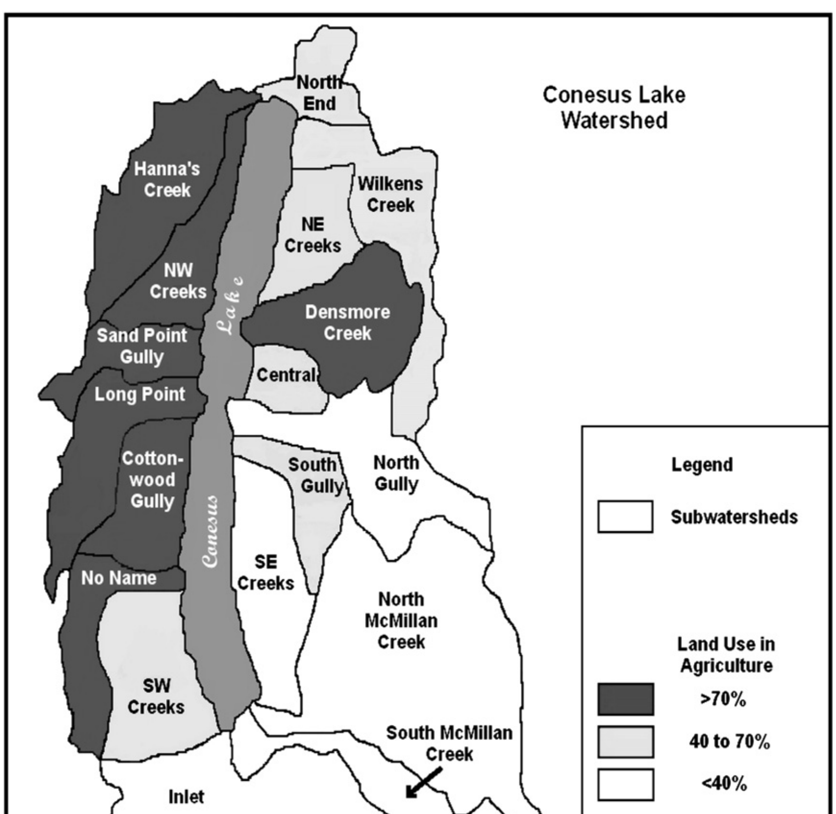

Fig. 3. Percentage of land in the Conesus Lake watershed in agriculture.

sources of bacteria contamination to streams within the Conesus Lake watershed using Rep-PCR. Surprisingly, geese were the dominant source of each of the sub-watersheds (44.7–73.7% of the total sources), followed by cows (10.5%–21.1%), deer (10.5–18.4%), humans (5.3–12.9%), and unidentified sources (0.0–11.8%) during the 2 years sampled.

The nearshore

In recent years, shoreline biomass of metaphyton and aquatic macrophytes has increased (Bosch et al., 2000, 2001). Field spatial analysis using GPS technology discovered that metaphyton and macrophyte [especially Eurasian milfoil (*Myriophyllum spicatum*)] beds are strongly associated with the mouth of the streams draining many of the sub-watersheds (Fig. 1, Bosch et al., 2000, 2001). In fact, a direct positive relationship exists between P and macrophyte biomass with higher concentrations of P in water draining the watershed being associated with greater macrophyte biomass (Fig. 2). In a series of field microcosm experiments, a 130 to 200% increase in growth of metaphyton occurred in water from six watersheds heavy in agriculture, compared to metaphyton grown in lake water (D'Aiuto, 2004; D'Aiuto et al., 2006). This major increase in metaphyton growth is attributed to elevated levels of dissolved P (~35 μg P/L) from watersheds dominated by agriculture compared to lake levels of 4.0 μg P/L. Metaphyton, which are the responsive algal type to nutrient inputs (Havens et al., 1999), are used as one indicator of BMP-induced change in nutrient retention in our watershed manipulations (Makarewicz et al., 2007).

The offshore

Despite the construction of a 28.6-km perimeter sanitary sewer completed in the early 1970s, Conesus Lake water quality has degraded in recent years (Makarewicz, 2001; SOCL, 2001; Makarewicz et al., 1991, 1999, 2001) due to "top-down" and "bottom-up" mechanisms. Phytoplankton populations have doubled from 2–4 μg/L in the 70 s to 7–8 μg/L of chlorophyll in the 90s due in part to the accidental introduction of alewives (*Alosa pseudoharengus*) that eliminated large herbivorous populations of zooplankton (Makarewicz, 2001). *Daphnia pulex*, which are effective grazers of smaller phytoplankton, were removed by the invasive alewives leading to major increases of phytoplankton in the 20- to 70-μm range. Simultaneously, watersheds with a large percentage of land in agricultural practices are losing large amounts of soil and nutrients (Makarewicz et al., 1999, 2001, 2007; SOCL, 2001) that are believed to stimulate the growth of pelagic phytoplankton communities (SOCL, 2001). Water clarity has decreased from secchi disk depths of 6–7 m in the 60 s to less than 3 m in the 90 s. Lake-wide manifestations of this degradation include higher than normal turbidity, blooms of algae in the open lake, and potentially harmful levels of fecal pollution as inferred from *Escherichia coli* and other coliform bacteria (SOCL, 2001). The cumulative flux of nutrients into the nearshore and offshore from the 18 creeks that drain the watershed, especially those from agriculturally dominated sub-watersheds, and the introduction of an invasive species, the alewive, appear to have had a negative impact on the trophic status of Conesus Lake. A bathymetric map of Conesus Lake may be found in Forest et al. (1978).

The Experiment

The use of experimental watersheds has been a successful method of accounting for environmental variability (Bishop et al., 2005). As annual variation and long-term trends in weather and nutrient export occur naturally in undisturbed watersheds (e.g., Esterby, 1996; Moog and Whiting, 2002), these effects must be separable from BMP effects when performing an evaluation of treatments (Bishop et al., 2005). At Conesus Lake, small sub-watersheds (Table 1) were chosen because they were predominantly in agriculture (over 70%, Fig. 3), are farmed by only one or two landowners, and thus impacts from other potential changes in land use are minimized. The five sub-watersheds targeted for nutrient and soil BMPs were Graywood Gully (74% in agriculture), Sand Point Gully (83%), Cottonwood Gully (75%), Long Point Gully (86%), and Sutton Point Gully (76%). Management practices were not introduced at North McMillan Creek (12% in agriculture) or North Gully (46%). Details on how watersheds were selected and management practices introduced are discussed in Herendeen and Glazier (2009). Monitoring of streams draining the watershed occurred for 5 years. Based on our previous work (Makarewicz et al., 1991, 1999; Makarewicz, 1992, 1993, 1994, 2001), we knew that these watersheds are event responsive; that is, over 80% of the soil and nutrient loss occurs in less than six or seven major precipitation events each year. Precipitation events, as runoff, sweep soil and nutrients from the landscape into drainage tiles and streams. Thus stream monitoring included instantaneous measurements of event and nonevent discharge and analyte chemistry (Makarewicz et al., 2009).

The overall objectives of the project were:

Objective 1: Implement a series of BMPs on individual farms in selected sub-watersheds with a goal of evaluating the impact of management plans on maintaining nutrients and soil on the landscape and of reducing losses of soil and nutrients to the downstream aquatic systems. The papers by Herendeen and Glazier (2009), Moran and Woods (2009), Makarewicz et al. (2009), Lewis and Makarewicz (2009), Zollweg and Makarewicz (2009), Noll and Magee (2009), and Noll et al. (2009) in this issue are directed at this objective.

Objective 2: Evaluate the effectiveness of agricultural management practices and strategies on bacteria, metaphyton, and macrophyte populations in the nearshore of Conesus Lake. Specifically, we hypothesize not only reductions in nutrient and soil concentrations and loading but also a decrease in metaphyton, bacteria, and macrophyte populations in the nearshore zone near stream mouths draining managed watersheds. The papers by Bosch et al. (2009a,b), Shuskey et al. (2009), and Simon and Makarewicz (2009a,b) in this issue address this objective. In related papers, Somarelli et al. (2007) identified wildlife as a major source of *Escherichia coli* in these agriculturally dominated watersheds, and Li et al. (2007) documented the flow regimes in the nearshore near two creeks draining from managed sub-watersheds of Conesus Lake.

Objective 3: Determine what physical and agricultural processes control/effect the biogeochemical cycling of nutrients through a watershed over time. Papers on winter manuring practices (Lewis and Makarewicz, 2009), on the impact of the "built environment" on extending a watershed beyond its topographic divide (Noll et al., 2009) on the fractionation of soil and sediments transported along a continuum from agricultural fields to nearshore lake sediments (Noll and Magee, 2009), and on the effect of antecedent hydrologic/meteorologic conditions on methodology for evaluating BMPs (Zollweg and Makarewicz, 2009) focus on this objective.

Objective 4: Experimentally evaluate the mode of nutrient uptake of the Eurasian milfoil. The hypothesis that Eurasian milfoil uptake of nutrients via leaves may provide a competitive advantage to this invasive species is tested in a series of experiments by Shuskey et al. (2009).

Objective 5: Develop a rigorous, physically-based GIS model of hydrology and nonpoint source pollution to analyze watershed processes and to evaluate management practices. The paper by Zollweg and Makarewicz (2009) deals with aspects of this issue.

Objective 6: Develop and implement an extension/education program that will demonstrate the link between watershed-specific agricultural management practices and water quality improvement, that will educate the agricultural community, and that will assist policymakers and managers in developing optimal strategies for water quality improvement. The papers by Herendeen and Glazier (2009) and Moran and Woods (2009) in this issue and by Glazier (2006), and Jacobs (2006a,b) in other sources comment on this final objective.

The series of manuscripts presented in this special edition should be of interest to people in the fields of watershed, soil, and aquatic science and restoration and conservation biology as well as to those in extension and planning who provide technical support to the agriculture community. In summary, a series of cultural and structural best management plans were voluntary and successfully implemented in several watersheds of Conesus Lake. Dramatic decreases in P, nitrogen, and soil loss from managed watersheds were realized. The greater the number of BMPs implemented, the greater the reduction of nutrient and soil loss to downstream systems. As a result of the BMPs introduced on the watersheds, significant reductions in metaphyton, macrophyte and microbial populations were observed in the nearshore of Conesus Lake.

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Comprehensive watershed planning in New York State: The Conesus Lake example

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ABSTRACT

Conesus Lake and its agricultural watershed exemplify many of the challenges of effectively managing land and water resources for multiple uses. One of the smaller of New York's Finger Lakes, Conesus Lake is eutrophic with abundant growth of rooted aquatic plants and algae. The trophic condition of the lake has been relatively stable for decades as measured by phosphorus concentrations and hypolimnetic dissolved oxygen depletion rates. However, recent changes to the lake ecosystem have led to diminished water clarity, higher density of macrophytes in shallower water, and the proliferation of nearshore macroalgae. These water quality changes caused serious concern regarding the ultimate causes of the degradation; community conflict was evident between the upland agricultural producers and shoreline residents. With funding from the New York State Department of State, Livingston County Planning Department led a successful effort to engage the community in a science-based assessment of the interrelationship of land use and lake water quality. The planning process was designed to develop consensus that the lake is a shared resource and that effective solutions require changes by many stakeholders. The outcome of this effort was the Conesus Lake Watershed Management Plan. An intermunicipal Conesus Lake Watershed Council was formed to implement the plan's recommendations. The Council has been active for 5 years and has weathered two local elections with full community support. Although many factors contribute to the success of this watershed planning initiative, the importance of a science-based foundation and ongoing efforts to build political consensus cannot be overstated.

Introduction

Watershed planning in New York State requires navigating a complex web of agencies and stakeholders with regulatory and programmatic responsibilities for water resources management (Finger Lakes-Lake Ontario Watershed Protection Alliance, 2000). Over the last three decades, top-down regulation has been an effective tool for reducing point sources of pollution. Nonpoint sources of pollution present a different set of challenges; among these challenges is the need to directly incorporate the human dimension. With guidance from the federal Environmental Protection Agency, New York and other states are adopting collaborative, community-based approaches to watershed planning (USEPA, 2008).

Land use decisions are largely under municipal control in New York, a "home rule" state, and watersheds of large lakes may encompass several counties and scores of municipalities. Decisions made by landowners and municipal officials ultimately determine the success of watershed management efforts; these decisions need to reflect both the science of lake management and the values of watershed communities. Consequently, effective watershed management programs couple strong science with extensive community involvement. Community respect for the scientists is essential; this outcome is, in turn, dependent on the scientists' ability to listen well and communicate complex information in an effective manner.

Any watershed project begins with understanding the nature of the system to be managed. This is largely a scientific issue, defining the components of the system and understanding the processes by which they interact. The natural and built environments in the watersheds characterize existing conditions. Managing a watershed to achieve a desired future condition requires a discussion of the values and perceived needs of the stakeholders: What qualities of the system matter? What attributes do they want to optimize? To what uses is the system to be put? Finally, a watershed management plan requires the imposition of constraints on the desires of the stakeholders: What are the scientific and economic limits to achieving the goals? In an era of limited resources for implementation, which actions take highest priority?

Many of the issues facing the Conesus Lake watershed are common to the Great Lakes region: residential development outside of sewered areas, intensification of agricultural production in response to shifting economics, diminishing community cohesion, and pressure on local

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government to control property taxes. At the same time, water quality and aquatic habitat are affected by invasive species, changes in hydrology, and rising water temperatures.

Here we describe the watershed planning process for Conesus Lake, one of the smaller NY Finger Lakes. The Finger Lakes region of central New York is characterized by a dozen lakes whose watersheds include a variety of valuable natural resources, including water, fish and wildlife habitat, wetlands, and forest. The lakes and their watersheds are used extensively for agriculture, recreation, and tourism, resulting in a firm link between resource protection and the regional economy.

The planning process began in 1999 and resulted in the Conesus Lake Watershed Management Plan in 2003 (Livingston County 2003). The watershed management plan required a scientific foundation to determine the relative importance of nonpoint sources of sediment and phosphorus to the lake. With a successful history of collaboration between the academic community and local government, developers of the watershed management plan were able to guide the research agenda towards issues of nutrient loading, algal abundance, and macrophyte proliferation. The place-based research initiatives focused on identifying the linkages between the landscape and the lake. In turn, this documentation helped build community support for recommended actions and fostered support for local elected officials to commit the resources in tax dollars and staff time for implementation.

Similar to a community's Comprehensive Plan, a watershed management plan provides guidance on potential effective actions but is not a legally binding document. Implementation of the recommendations typically requires legislative actions such as modifications to zoning or subdivision regulations, adoption of local laws related to stormwater management, entering into intermunicipal agreements, and budgetary allocations. Other recommendations may require executive or administrative actions for their implementation. For example, municipal officials may direct the efforts of the local work force towards efforts such as street sweeping or road ditching that affect the nature of stormwater runoff. Voluntary actions by private landowners, both residents and agricultural producers, to minimize the potential for transport of pollutants into surface water and groundwater are essential. The challenge is to develop and communicate a shared vision for how the collective actions of the community will help protect and restore the lake.

Community understanding of the relative importance of the underlying causes of degradation, grass-roots support for the investment of public funds, public willingness to examine the effects of actions on their own properties, and unwavering commitment from elected officials are all necessary for the plan to move from recommendations to actions. On these metrics, the Conesus Lake watershed planning effort has been a success. Several factors contributing to the success are discussed in this article.

Water quality conditions and perceived impairments

Conesus Lake is a eutrophic lake; total phosphorus concentrations average 20–25 μg/L in the upper waters (Makarewicz, 2009). Notably, the trophic state of Conesus Lake has been relatively stable for decades as measured by total phosphorus concentrations and dissolved oxygen profiles (NYSDEC 2001, Forest et al. 1978). In 1910 limnologists reported on the clear water and abundant macrophyte growth in Conesus Lake (Birge and Juday 1914). The lake is the westernmost and one of the smallest of the Finger Lakes, surface area 13.5 km^2, mean depth 11.6 m, and maximum depth 20 m. The littoral area encompasses approximately 20% of the lake surface area.

Changes in the food web in recent decades have resulted in diminished water clarity and a shift in macrophyte distribution to shallower waters. As described in Makarewicz (2009), the introduction of the alewife in the 1970s and resultant loss of larger *Daphnia*, an efficient grazer of phytoplankton, have led to higher chlorophyll-*a* concentrations in the lake water. This food web interaction is consistent with the trophic cascade theory of Brooks and Dodson (1965). Zebra mussel larvae were first detected in 1992. By 1994 juveniles were abundant; adult zebra mussels have been widely distributed since 1997 (Bosch et al., 2009).

Issues of increasing concern are the proliferation of native and exotic species of macrophytes in shallow waters, algal mats, elevated bacterial levels, and cyanobacterial blooms with the potential for harmful exudates. Similar to the Great Lakes, nearshore areas of Conesus Lake have increasing levels of filamentous green algae (*Zygnema* sp., *Spirogyra* sp., and others). This is attributed to dynamic ecosystem changes in nutrient flux and water clarity. Zebra mussels, warming water temperatures, lower lake levels, and inputs of biologically available phosphorus and nitrogen are considered to potentially contribute to the increased abundance of nearshore macroalgae (Harris 2004, Pillsbury et al. 2002). Of these potential causes, phosphorus and nitrogen inputs are the most amenable to local control.

Declines in water quality diminish recreational use of the lake and are of great concern to the community. Complaints of beaches awash with decaying vegetation, weed-choked bays, and fears of declining property values, conditions reminiscent of the south shore of Lake Ontario, began to dominate meetings of the Conesus Lake Association and the local legislative bodies. Shoreline residents placed the blame for the degradation on the upland agricultural producers, citing intensification of production in the dairy industry. In turn, upland residents pointed to the construction and development activities around the shoreline, as watershed population increased and seasonal camps were converted to year-round residences. There was agreement that Conesus Lake is a regionally important natural resource, aesthetic asset, tourist destination, and water source and that “something” needed to be done.

Nature of the watershed

The watershed of Conesus Lake extends over approximately 181 km^2 and encompasses portions of seven municipalities within Livingston County, New York. The watershed, home to approximately 10,000 people, forms part of the Genesee River basin. The outlet of Conesus Lake flows north to join the Genesee River and ultimately reaches Lake Ontario. Land use in the watershed is dominated by agriculture (approximately 42%) and residential parcels (36%) according to the Conesus Lake Watershed Characterization Report (Livingston County 2002).

Residential properties surround the majority of the lake shoreline and are served by a perimeter sanitary sewer. Development in the watershed has exhibited a steady increasing trend. In 1974 the population of the watershed was estimated at 5900 and grew to 9800 in 1990, a 72% increase. A more modest increase was evident in 2000, with the population estimated at 10,000. Much of the development between 1974 and 1990 occurred close to Conesus Lake; there are 1750 residential dwelling units within 1000 ft of the shoreline (Livingston County 2002).

Very few shoreline building lots remain. As developers and potential homeowners seek to build new homes with a view of Conesus Lake, the upland areas of the lake valley are increasingly targeted for development. With the development of upland areas comes a series of issues that affect water quality. Hillsides cleared of vegetation during construction are subject to erosion. There are long-term consequences as well; as hillsides are developed with homes and roads, the amount of impervious surfaces increases, which impedes infiltration of precipitation into the ground. Runoff becomes an issue because precipitation is funneled down the steep hillsides directly into the lake (Livingston County 2002). Local laws requiring stormwater management and erosion and sedimentation controls during construction have been enacted. In addition, a Conesus Lake Watershed Inspector position was created within the Livingston County Department of Health to provide education, investigate complaints, and enforce codes.

Planning process

The process of developing a management plan for Conesus Lake began in 1999; the New York State Department of State, Division of Coastal Resources, awarded three grants totaling $192,000 through the Environmental Protection Fund— Local Waterfront Revitalization Program to the Town of Livonia to finance the planning effort. These grant funds were matched by local contributions and in-kind services. The Livingston County Planning Department served as overall project manager. To better guide future efforts aimed at protecting and improving water quality, the Division of Coastal Resources encourages the completion of intermunicipal watershed plans as a means of establishing a consensus on priority actions needed to protect or improve water quality. Within the Finger Lakes, the Department of State has awarded grants from the Environmental Protection Fund for preparation of such plans for Cayuga, Canandaigua, and Conesus Lakes.

Two standing committees and two subcommittees guided the planning. The Policy Committee, comprised of elected officials in watershed towns and villages, was responsible for major decisions. Their active participation ensured that the municipal officials were kept abreast of developments and thus able to communicate effectively with their constituents. The Planning Committee was responsible for the technical work and for advising on technical aspects of policy decisions. By design, the Planning Committee included representatives of agencies with lake-related responsibilities. This Committee included representatives of the following: Livingston County Planning Department, Livingston County Department of Health, Town of Livonia, New York State Department of Environmental Conservation (NYSDEC), Livingston County Soil and Water Conservation District, Conesus Lake Association, Livingston County Water and Sewer Authority, and SUNY Geneseo. One subcommittee of the Planning Committee was responsible for public education and outreach. The second was responsible for engaging the agricultural community and included both local producers and professional staff from the agricultural support agencies.

The first element in the planning process was to document the current water quality and ecological conditions of Conesus Lake and its watershed. This effort was designed to address the question "Where are we?" Historical and contemporaneous data were compiled and analyzed. *State of Conesus Lake: Watershed Characterization Report* was completed in May 2002 and is available at http://www.co.livingston.state.ny.us/plan_lake-rpt.htm. Specific areas of concern affecting the attainment of the desired use of the system were identified. These areas of concern included issues such as sedimentation, nutrient enrichment, bacterial contamination, and pesticide levels that threaten the long-term health of the lake and its desirable uses as a drinking water supply and recreational resource. Data gaps, areas where more data and information were needed to assess conditions and identify specific sources, were identified. As described in the Introduction, local scientists and academics were key participants in formulating and testing hypotheses and filling data gaps.

In addition to defining the current conditions of the natural and built environments, the *State of Conesus Lake: Watershed Characterization Report* (SOCL 2001) evaluated the nature of the programmatic environment. The multitude of activities underway by local, state, and federal agencies, local government, the private sector, and individuals to protect and improve this unique and treasured resource were catalogued. Local laws affecting the potential for nonpoint sources of pollution to reach surface waters were reviewed.

The second element of the planning process was to determine "Where are we going?" Trends in water quality and aquatic habitat, land use, agricultural production and the status of best management practices, and the regulatory environment were reviewed, along with broader metrics of population, demographics, employment, and wealth. The network of participants drawn together to complete the planning effort included local experts, community stakeholders, and local experts on a range of natural resource and socio-economic conditions. The organizational structure for the planning process is illustrated in Fig. 1.

The question of "Where do we want to be," the third element of the planning process, was a conversation with the community that began with a visioning session in June 1999 and continues to this day. The Policy Committee adopted a vision statement for the watershed management plan as well as for the lake and its watershed: "To preserve, enhance, and restore the health, natural beauty, and rural character of Conesus Lake and its watershed for this and future generations." Specific metrics and issues associated with this vision are summarized in Fig. 2. Conesus Lake is an important asset of Livingston County; the lake serves as the primary water supply for some 15,000 residents and is a regional recreational resource for fishing and boating. The lake's impact on property taxes and tourism income is well documented. There is broad agreement on the need to protect and improve water quality and habitat conditions in order to maintain use of the lake for water supply and recreational uses. An active lake association promotes educational efforts and continues to advocate for effective controls on point and nonpoint sources of pollution.

Finally, the planning process culminated in a detailed analysis of "How do we get there?" This fourth element of the planning process is a blueprint of actors and actions. Completed in January 2003, the Conesus Lake Watershed Management Plan (Livingston County 2003) presents priority actions, strategies for implementation, and metrics to measure and report progress. The plan is posted at http://www.co.livingston.state.ny.us/plan_clwmp.htm. Priority actions address both sources of pollution and impaired uses: (1) reducing sediment and nutrient loss from agricultural lands, (2) finding effective in-lake measures to reduce abundance of algae and weeds, (3) improving water and wastewater infrastructure, (4) reducing

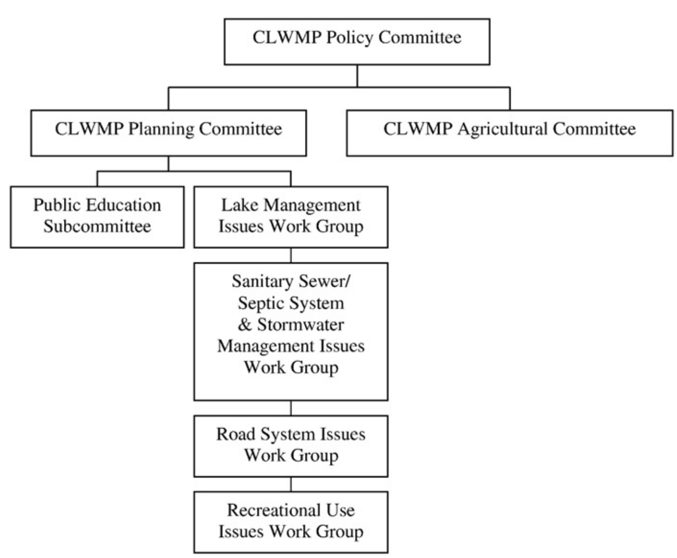

Fig. 1. Organizational structure adopted for the Conesus Lake Watershed Management Plan (CLWMP).

VISION

"TO DESIGN A MANAGEMENT PLAN THAT PRESERVES, RESTORES, AND ENHANCES THE HEALTH, NATURAL BEAUTY AND RURAL CHARACTER OF CONESUS LAKE AND ITS WATERSHED."

What is the objective of the Conesus Lake Watershed Management Plan?

To ensure the sustainability of designated uses for Conesus Lake and its continued role as a positive influence on the social and economic well being of watershed communities.

What are the goals for water quality of the lake?

To improve water quality conditions in Conesus Lake to ensure its continued use as a water supply and make it more attractive for water contact recreation.

How does the plan view ecosystem management?

The plan includes actions designed to restore Conesus Lake to a diverse ecosystem composed primarily of native species of plants and animals.

What part does the community plan in the plan?

For the plan to succeed, it must promote cooperation of all stakeholders at the local level to develop a comprehensive approach that seeks to build collaboration and balance diverse concerns.

How does the plan address the specific concerns of agriculture?

The plan recognizes the value of high quality agricultural practices in meeting its goals. The plan seeks to promote the viability of agriculture and best management practices in land use.

How does the plan affect residents of the watershed?

The plan seeks to provide necessary services to all watershed residents while preserving the natural beauty and rural character of the countryside.

Fig. 2. Summary of the community's vision and goals for Conesus Lake and its watershed.

sediment and salt runoff from roadways and ditches, and (5) managing recreational uses.

Focus on agriculture

The *State of Conesus Lake: Watershed Characterization Report* documented the significance of sediment and nutrient losses from the agricultural activities in the watershed. The demonstrated losses of soil and nutrients from agricultural activities in the watershed and the probable effects on downstream communities are serious issues locally, regionally, and nationally. Herein lies the growing dilemma for governmental leaders in agricultural areas: that their most important economic industry (agriculture) may also be the cause of environmental degradation which heightens public concern. For farmers, this is further exacerbated by the increasing regulation of agricultural operations.

The Conesus Lake Watershed Group (CLWG) was formed in 1999. The majority of the CLWG represent the agricultural community. Members include the Natural Resources Conservation Service (NRCS), Livingston County Soil and Water Conservation District, the Farm Service Agency (FSA), the Conesus Lake Watershed Inspector, Cornell Cooperative Extension, Livingston County Health Department, Livingston County Planning Department, and the State University of New York Colleges at Brockport and Geneseo. In 2001 the CLWG worked together to develop a research agenda reflecting the needs of the agricultural community, the lake community, and resource managers to address demonstrated nonpoint source watershed problems resulting from agricultural practices. The CLWG collaborated with academic researchers and extension professionals to implement the USDA-CSREES funded research program, made connections with the agricultural community, and helped identify farmers willing to implement BMPs. The investigations described in this special volume were completed to advance the CLWG research agenda.

Institutional structure for implementing the Watershed Management Plan

Supporting the public perception that the plan itself was not the outcome was the focus on a framework for implementation. A Conesus Lake Watershed Council was formed to oversee implementation of the recommended actions in the Watershed Management Plan. The Watershed Council was created by an Intermunicipal Agreement between Livingston County and the watershed municipalities. By signing the Intermunicipal Agreement, each town and village committed to financially supporting the Watershed Council. The annual payment varies by town and village and is based on a multifaceted funding formula. Funds are used to help offset the costs of the watershed inspection program and a part-time contractor serving as the Watershed Manager. The Livingston County Planning and Health Departments provide staff and technical support to the

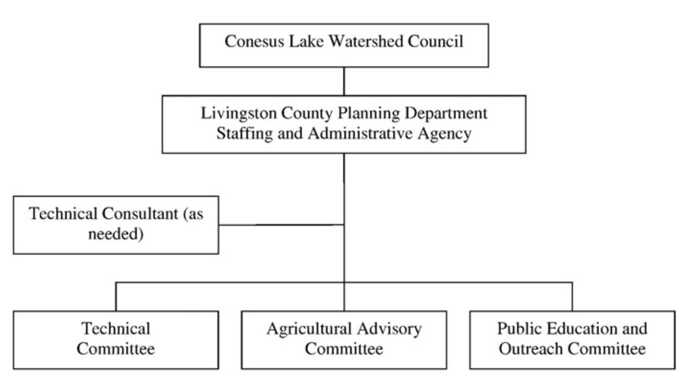

Fig. 3. Organizational structure for implementation of the Conesus Lake Watershed Management Plan.

Watershed Council. The organizational structure for implementation is illustrated in Fig. 3.

The members of the Watershed Council are the elected supervisors and mayor of the towns and village in the watershed, the mayors of the two villages using Conesus Lake as a drinking water supply, and the Chairman of the Livingston County Board of Supervisors, or their designated representatives. Nonvoting members on the Watershed Council include the Livingston County Planning Department, the Livingston County Department of Health, the Conesus Lake Association, the Livingston County Farm Bureau, and the Livingston County Water and Sewer Authority.

Discussion

The Conesus Lake Watershed Council has now been in place for 5 years. The watershed community and stakeholders remain engaged in the process and committed to rehabilitating the community asset. We offer the following observations as a guide to other communities planning for watersheds that transcend municipal boundaries.

The watershed plans serve an important function by establishing a consensus among state and local government on future actions needed to protect water quality. With completion of these plans, the intermunicipal organizations continue this collaborative effort to focus on implementation. As a result, limited resources can more effectively be targeted to projects which flow from a deliberative regional planning process. The local governments within the region recognize the link between protection of water quality and natural resources and the region's tourism economy. The plan's role as a consensus between local governments and State agencies is key, since the use of State grants is an important source of implementation.

Strong leadership is essential. Stakeholder involvement is necessary but not sufficient to complete a community-based collaborative effort to define problems and define effective solutions. An individual or agency must be the “first among equals” and commit the time and resources needed to keep the effort focused and on schedule. While responsibility was centralized in the Planning Department, information was not. Frequent communications, formal and informal, with elected officials and other stakeholders were a priority for the Planning Department. The focus on communication and consensus-building served to mitigate the potential conflict between the shoreline residents and the agricultural community and kept the elected officials involved and committed.

The Conesus Lake example illustrates another critical element: ongoing implementation. As the Watershed Management Plan was being developed, it became evident that sedimentation from construction and other land disturbance was a significant cause of water quality impairment. The Livingston County Planning Department drafted a model local law with stringent stormwater management controls to reduce erosion and sedimentation. Three of the four towns bordering Conesus Lake adopted the local law for application throughout the towns. The fourth town already had a comparably protective local law in place.

When it became clear that agricultural land use was also a significant factor affecting nutrient and sediment loss from the landscape and that the losses were degrading water quality and habitat conditions, the CLWG began to engage local partners in demonstration projects. The theme of ongoing implementation was used in public outreach efforts throughout the planning process, to reassure concerned residents that the end product of the investment was not a study.

The analysis and recommendations in the watershed management plan are based on site-specific monitoring and scientific investigations. Fortunately for the Conesus Lake watershed, the SUNY investigators brought years of experience with the lake and its watershed to the process. We consistently heard comments reflecting the local pride in the SUNY colleges and confidence in the integrity and skills of the research team. The science must be strong; the local perception of the scientists must be strong as well. For a decade prior to the watershed management planning initiative, Livingston County allocated modest amounts of state funds originating from the Finger Lakes-Lake Ontario Watershed Protection Alliance to support efforts of the local research community. This investment in monitoring and assessment of the lake and watershed allowed the research community to become involved as a stakeholder and build local awareness of their projects. As findings of the CSREES project became available, the scientific basis for the recommended actions became increasingly stronger, as did the support of elected officials and the watershed community.

Adaptive management, remaining open to new information, was another feature of the Conesus Lake planning experience. Managing any natural resource is subject to uncertainty, and effective watershed management plans will specify an iterative process that sequences through four stages: planning, implementation, measuring effectiveness, and adaptation. Changes in technology and advances in our understanding of lake processes will likely continue, and a mechanism to respond to new information is essential.

Finally, the commitment to communication has been a key factor in keeping the community engaged in the long process of lake rehabilitation. The researchers and managers share a commitment to explaining their progress in terms that are accessible to all. The Watershed Council publishes an annual Conesus Lake Report Card, detailing progress on the specific recommendations of the watershed management plan and reporting on activities over the past year. This simple tool has been quite effective in communicating progress and engendering public support. Documenting efforts, both successes and failures, informs the public and prevents the misperception that the plan is gathering dust.

The Conesus Lake Watershed Management Plan is an example of effective collaboration between the academic community, municipal officials, and the public to address these important questions regarding the desired future of a treasured resource. Ultimately, effective watershed management requires far more than a narrow focus on water quality. A planning process that directly engages residents in a long conversation about what they value and how to restore and protect these resources is essential.

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Agricultural best management practices for Conesus Lake: The role of extension and soil/water conservation districts

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ABSTRACT

Small sub-watersheds of the Conesus Lake catchment were the site of a project evaluating the ability of agricultural Best Management Practices (BMPs) to maintain soil and nutrients on the landscape and to reduce the impact of agriculture on downstream aquatic systems. Local agricultural agencies, with participation of local farmers, joined with scientists to focus attention on watershed issues, to develop and foster a sense of stewardship among the farming community, and to assist and coordinate collaboration among academic researchers, governing bodies, and the agricultural community. Cornell Cooperative Extension served as a liaison and as a resource and assisted in the development and implementation of voluntary BMPs in selected sub-watersheds of Conesus Lake. We discuss our approach to working with producers, the selection of watersheds for management, and our decision-making process for implementation of BMPs. Decisions to establish traditional structural and nonstructural management practices on sub-watersheds of Conesus Lake were based on field assessments, soil testing, the Phosphorus Index, and the software package Cornell Cropware. For example, the use of soil testing and the Cornell Cropware software allowed the cooperating farms to apply fertilizer only as needed for optimum crop production. Farmers achieved cost savings because previous plans had not given enough credit to soil reserves, manure, and sod crop nutrients. Low-cost voluntary practices based on well-established agricultural management practices have been combined with cost-shared (structural) practices in Conesus Lake watersheds to mitigate the impact of agricultural runoff on water quality and to improve cost efficiency of agricultural operations.

Introduction

Conesus Lake, one of the smaller Finger Lakes in western New York, is used for recreation and fishing and is a source of municipal water for five local communities. The shoreline area is densely populated with residences, primarily year-round homes. The upstream area is a mixture of agricultural land and mixed deciduous hardwood forests encompassing an area of 16,714 ha. In 1999 about half of the entire land use within the Conesus Lake watershed was and continues to be in agriculture. Much of the agriculture (>70%) is concentrated in the western sub-watersheds of the lake (Fig. 1, Table 1; SOCL, 2001). The deep, well-drained, glacially derived limestone soils that dominate the watershed are productive and support field crops (field corn, forages, winter wheat, soybeans), vegetable crops (dry beans, sweet corn), and a small acreage of vineyards. Dairy farms are the major animal agriculture with a few livestock and horse farms. More information on Conesus Lake and its surrounding watershed may be found in this issue (Makarewicz, 2009).

Stakeholders believed that the growth of shoreline aquatic macrophytes and algae was increasing in Conesus Lake. Agricultural runoff is widely known to be a significant contributor to algal blooms in aquatic habitats (e.g., McDowell et al., 2004, Beman et al., 2005). In Conesus Lake, studies have shown that plant biomass was especially prominent in areas where streams draining primarily agricultural watersheds entered the lake (D'Aiuto et al., 2006, Makarewicz et al., 2007). Consequently there was a general consensus that excessive plant growth in Conesus Lake was caused, at least in part, by repetitive agricultural runoff and nutrient loss from these fields (SOCL 2001, Bosch et al., 2009a, b; Moran and Woods, 2009). A purpose of the Conesus Lake Watershed Project was to implement management practices, so-called Best Management Practices (BMPs), and to determine if there were reductions in nutrients entering the lake from that farmland.

BMPs are actions, behaviors, or techniques that reduce pollution and the amount of runoff flowing into waterways and that cover a wide range of practices on the land. Non-structural BMPs include such practices that minimize site disturbance through sound planning and design and include cropping sequence, soil testing, fertilization rates, tillage practices, etc. For example, on dairy or livestock farms, soil testing is combined with manure analysis to develop manure application plans to make best use of recyclable manure nutrients. Structural BMPs include construction of manure lagoons, terraces, buffer strips, sediment control basins, etc. Contour strip cropping, where alternate strips of land are planted to row crops or sod crops

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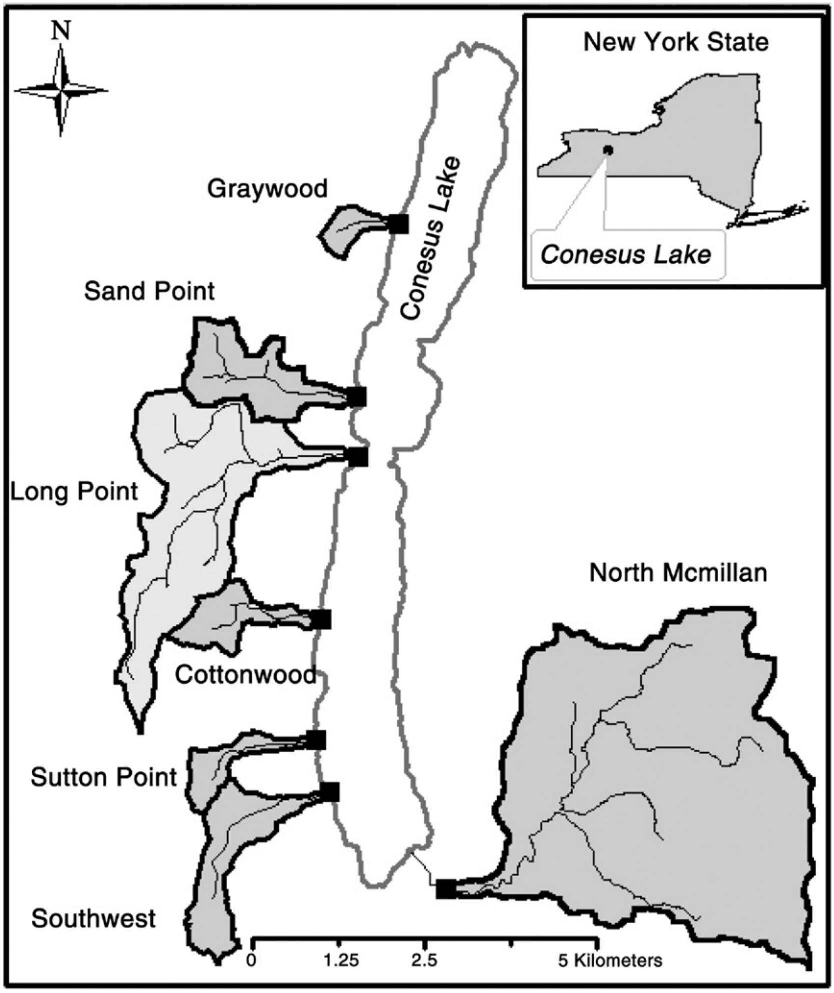

Fig. 1. Conesus Lake and sub-watersheds selected for the BMP-watershed manipulation study. Other sub-watersheds are not shown. Square symbols are the location of stream sampling sites discussed in Makarewicz et al. (2009).

perpendicular to the predominant slope, is an example of another structural BMP. By shortening the lengths of slope where water can flow freely (tilled strips), runoff and kinetic energy of water are reduced, and soil erosion is minimized. The recommended BMPs were all proven technologies, designed to meet or exceed Natural Resources Conservation Service (NCRCS) standards. This project was not an effort to evaluate new experimental management practices but to evaluate the impact of accepted NCRCS practices on downstream aquatic systems at the watershed level.

Cornell Cooperative Extension (CCE) connects the research knowledge of academic institutions to individuals, communities, families, and other agencies in New York to enhance their economic well-being and quality of life. Extension's part in this project had several important goals and involved multi-agency cooperation. The North West New York (NWNY) Team of Cornell Cooperative Extension served as the liaison between the cooperating farmers and academic researchers at The College at Brockport, SUNY Geneseo, and Rochester Institute of Technology. With technical assistance from the Livingston County Soil and Water Conservation District (SWCD), CCE planned and encouraged implementation of a series of best management agricultural practices. CCE also maintained their traditional effort of outreach to the agricultural community. Here we discuss the approach to working with producers and the various BMPs implemented on all study watersheds but focus on the results from the Maxwell Farm in the Graywood Gully sub-watershed.

Approach and methods

Voluntary farmer participation

Staff from the NWNY Team visited potential farm cooperators to explain the CSREES-USDA (Cooperative State Research Education Extension Service) funded program and to encourage voluntary participation. The contentious atmosphere that existed between the Conesus Lake Association (CLA) and the agricultural community was an issue prior to the start of the project. As expected, the farmers had questions relative to the use of their farm information, media publicity, and relationships with the CLA.

The Cornell Cooperative Extension Team and the Livingston County SWCD staff assured the farmers that no mandatory practices would be required, and there would be no retroactive enforcement of previous runoff problems if identified by the CSREES-USDA sponsored research. It was explained that there would be no release of personal information relative to individual farms without written permission. Also, the farm operators were skeptical of having researchers on their property. All agreed that the NWNY Team would be the intermediary between the BMP farms and academic researchers. When researchers needed to know more about specific farm practices, they would direct their questions to the NWNY staff. As a result of these discussions and agreements, the farmers in the three BMP watersheds agreed to participate.

Table 1
Watershed area and mean annual daily discharge (2002 to 2007) (from Makarewicz et al., 2009) for six sub-watersheds of Conesus Lake.

	Watershed area (ha)	Percent of watershed in agriculture	Mean daily discharge (m^3/ha/day)	Management plans implemented
Graywood Gully	38.1	74	41 (29[a])	Winter manure spreading eliminated, manure management, fertilizer use reduced, subsurface drainage tiles, grass filter strips, contour tillage, roof water separation, fall tillage curtailed, cover crops added, cattle fenced out of stream
Cottonwood Gully	98.8	75	22	Gully plugs, 28% reduction in croplands, cover crops added, fall tillage curtailed, zone tillage
Long Point Gully	587.9	86	13	37% reduction in croplands, removal of cows, barn no longer in use, elimination of winter spreading, manure injection rather than spreading
Sand Point Gully	188.0	83	12	9.5% of area converted to rotational grazing, fenced cattle from streams, gully plugs and tiles one month prior to project start, crop rotation whereby soybeans increased by 50%
Sutton Point Gully	67.5	76	26	60% conversion of cropland to alfalfa, gully plugs installed in last year of project
North McMillan Creek	1778.2	12	17	No management practices

The Maxwell, Barber and Gray Farms are located in Graywood, Cottonwood, and the Long Point Gully sub-watersheds, respectively.
[a] Discharge weighted by extended watershed of Graywood Gully.

Selection of watersheds

The research team in consultation with Livingston County SWCD chose six sub-watersheds and their associated farm(s) of the Conesus Lake watershed (Fig. 1). In general, there was one farm operation that controlled nearly all of the crop acreage being farmed in the study watersheds. Items considered while selecting watersheds were:

- amount and type of agriculture (farms with animals versus crop farms),
- enthusiasm of the farmer to cooperate with implementation of BMPs,
- willingness to allow researchers on their farm,
- previous known losses of nutrient and soils from the watersheds, and
- development of aquatic macrophyte and algae biomass in the nearshore area of the stream mouths that drain the watersheds.

Site evaluations/field assessments

Livingston County SWCD technicians evaluated the farms for structural and/or land management practices that would improve nutrient and sediment runoff. They assessed the need for enhanced surface or subsurface drainage, strip cropping, water and sediment control basins, filter strips, roof water separation, farmstead runoff water management, manure storage structures, feed storage facility runoff containment or treatment, milking center waste management, and fencing cattle out of streams.

The NWNY Team agronomist met with the farmers to discuss soil testing, whole farm nutrient management planning (including manure applications), and implementation of BMPs. A whole farm plan was developed for the Maxwell dairy farm operating in the Graywood Gully watershed. This was the only study watershed where structural BMPs were implemented by this project with a dairy farm physically located in the watershed. A whole farm plan was developed in the Long Point watershed by a private consulting firm as part of the Concentrated Animal Feeding Operation (CAFO) requirement for Dairy Knoll Farms. The operators of Dairy Knoll Farms leased the cropland in this watershed when the dairy farmer in Long Point watershed ceased milking cows and went out of business. This ended the animal presence on the farm, except for some pasture situations. The land became part of the Dairy Knolls Farms CAFO plan. The Cottonwood watershed land is operated by Barber Farms, a crop farm with no animals and no animal manure as part of their system. No whole farm plan was developed, but structural changes were implemented and farm cultural practices were changed.

All fields where structural BMPs were introduced (Cottonwood Gully, Graywood Gully, and Sand Point Gully) were assessed by the NWNY Team staff for slope gradient and length plus the presence of hydrologically sensitive and concentrated flow areas. Soil types were mapped based on detailed soil survey information provided by Livingston County SWCD and National Resource Conservation Service (NRCS). All measurements were made using USDA standardized methods. Field data were used in the Revised Uniform Soil Loss Equation (RUSLE, http://www.ars.usda.gov/Research/docs.htm?docid=5971) to determine potential soil loss. In fields where structural BMPs were not implemented by this project (Sutton Point Gully, Long Point Gully, and North McMillan Creek), but cultural BMPs may have been introduced by the producer, no assessment of soil type, slope, etc. was completed by the NWNY team due to cost issues. For these watersheds, the intent was to allow the producer to operate in a normal manner without benefit of advice from the NWNY Team. North McMillan was designated the “reference” watershed and no cultural nor structural practices were implemented. However, sampling of water was undertaken at the base of each watershed for the duration of the study.

Soil testing

The farmers in some watersheds agreed to have a complete set of soil samples taken on their fields as a baseline for fertilizer and manure management recommendations. The NWNY staff completed soil sample and field assessments in 2003 and 2004. Additional soil samples were taken in 2005 and 2007. Farm maps noting hydrologically sensitive areas, concentrated flow areas, and intended crop rotations were delineated.

Available soil nutrients were extracted from the soil samples using Morgan's solution, a sodium acetate/acetic acid solution, buffered at pH 4.8. Activated carbon was added to the extraction to aid in the removal of organic matter and to help decolorize the extraction solution. After vigorous shaking for 15 min at 180 rpm, the extraction slurry was filtered through a fine-porosity filter paper. The clear filtrate was analyzed for K, Ca, Mg, Fe, Al, Mn, and Zn on a Jyobin Yvon manufactured inductively coupled plasma (ICP) spectrophotometer. The plant available NO_3-N and PO_4-P were measured using an Alpkem Automated rapid flow analyzer (CNAL 2007, Method 1030; Morgan, 1941).

Soil samples were collected from fields in three sub-watersheds (Maxwell Farm, Graywood Gully; Barber Farms, Cottonwood Gully; Dairy Knoll Farms, Long Point Gully). At least 10 replicate subsamples were taken at plow depth from each field using a standard soil probe. Samples were dried and composited for each field, and approximately

400 g of soil for each field was submitted to the Cornell Nutrient Analysis Laboratory (CNAL, 2007). Organic matter analysis followed Storer (1984), while pH, water content, and lime requirements followed Black et al. (1965), Eckert and Sims (1995), Gardner (1965), and Grewling and Peech (1960).

At the Dairy Knoll Farms in Long Point Gully, soil samples were taken by a private consulting firm and analyzed by a private laboratory. These results are not part of this summary. The whole farm nutrient plan and management recommendations for this CAFO operation were similar to those used by the Cornell Nutrient Management Planning Program on the Maxwell Farm.

The NWNY Team agronomist met with farmers to discuss soil test results and to assist them in developing practical fertilizer and manure plans for their farm. This included crediting other nutrient sources on the farm plus soil reserves as indicated by soil testing. Previously, fertilizer recommendations had been made primarily by fertilizer vendors. Explaining the value of nutrients from sod crops and manure plus soil reserves was not part of the fertilizer vendors practice. Other soil testing services do not include credits (nitrogen, phosphorus, and potassium nutrients being recycled) from manure and/or sod crops as part of their recommendations. In all cases, the farmers achieved cost savings because previous practices had not given enough credit to soil reserves, manure, and sod crop nutrients.

On the Maxwell Farm in the Graywood Gully sub-watershed, soil test and manure analysis data were entered into the "Cornell Nutrient Management Planning Cropware" (Cornell Cropware) program (http://nmsp.css.cornell.edu/software/cropware.asp). This software developed by Cornell University Department of Crop and Soil Science allows input of manure analysis, quantity of manure produced, sod crops, and other sources of nutrients to develop individual field recommendations and manure spreading schedules. For example, sod crops, particularly legume-grass forage mixes, are typically planted and harvested for 3 to 5 years before being rotated to a crop such as corn. Rotating these sod fields allows bacterial decomposition of roots and crop residue. This biological process will potentially supply from 111 to 168 kg N/ha as nutrients to the succeeding crop. Residues that have greater than 50% legume generally supply greater than 168 kg N/ha (Ketterings et al., 2003a).

The Maxwell Farm also participated in a fertilizer research and demonstration plot involving phosphorus fertilizer rates at planting time. This was conducted in cooperation with Cornell University staff as part of a statewide study on reducing phosphorus applications on corn at planting time. This study also confirmed the value of following recommendations based on Cornell Soil Testing protocol (Ketterings et al., 2003b). Another study by Cornell staff members in cooperation with the NWNY Team, but not done in a BMP watershed, confirmed the value of credits for nitrogen following sod crops to replace nitrogen fertilizer (Ketterings et al., 2003a). Such studies were part of the discussions with farmers to reduce the use of purchased fertilizers on the BMP farms.

The NY Phosphorus Runoff Index (PI) portion of the Cornell Cropware program was used to assist producers and planners to identify fields or portions of fields that were at highest risk of contributing phosphorus (P) to lakes and streams (Czymmek et al., 2003). The New York PI is not a measure of actual P loss but rather an indicator of potential loss. A high or very high PI score is a warning to further examine the causes. Fields with high or very high site vulnerability were managed so as to reduce P additions to the soil in the form of fertilizer or manure. The PI assigns two values to the field, one for potential loss of dissolved P in groundwater and one for potential loss of particulate P attached to soil particles that might be eroded from the surface. This results in a "source score" and a "transport score" that are multiplied together to give a PI score. The advantage of this system is that it includes soil test P values along with physical properties of the site being evaluated. Details are available in the publication "Cornell University Cooperative Extension Agronomy Fact Sheet 10" (http://nmsp.css.cornell.edu/publications/pindex.asp). This system is part of the software package in "Cornell Cropware" that was used to develop the whole farm nutrient management plan at Maxwell Farm.

Structural practices

The Livingston County SWCD and NWNY staff worked with farmers to find funding sources to assist with the cost of structural changes and BMP implementation. Funding for management practices came from the Natural Resources Conservation Service of USDA, SWCD grants from New York State, and the USDA-CSREES grant that funded the research parts of this project. Participation in implementing the upstream practices by farmers was voluntary and also involved their own investments in BMPs.

Examples of structural practices implemented in this study included: water and sediment control basins (gully plugs that retard and manage water movement on slopes), a manure storage facility, grass waterways, subsurface drainage, grass filter strips, fencing systems, and roof water collection systems. All structural practices were designed to Natural Resource Conservation Service (NRCS) practice standards that are described in the USDA NRCS National Handbook of Conservation Practices (www.nrcs.usda.gov/). A manure storage facility (NRCS Standard NY312-1) allowed for the elimination of winter spreading and the maximization of nutrient recycling through crop production (Dairy Knoll Farm, Long Point Gully). Grass filter strips (NRCS Standard 393a) were designed and installed to collect runoff from feed storage facilities and/or farmstead driveways (Maxwell Farm, Graywood Gully). Roof water management (NRCS Standard 558) involved structures to collect clean roof runoff and to discharge it safely to streams or road ditches so it does not become contaminated with farmstead wastewater (Graywood Gully). Water and sediment control basins (gully plugs) on long slopes (Barber Farm, Cottonwood Gully) were designed by the Livingston County SWCD to retain soil and collect runoff water allowing water to discharge slowly through a standpipe and subsurface drainage tubing. Fencing systems (NRCS Standard 382) that keep pastured livestock out of streams was another structural practice implemented (Graywood Gully and Long Point Gully). This practice requires a method of providing water to pastured livestock at a location away from streams or intermittent waterways. Grass filter strips, subsurface drainage, strip cropping, and grass waterways are other examples that were used in the Conesus project.

Cultural (nonstructural) management practices

Managing crop rotations to make maximum use of recyclable nutrients and to minimize soil erosion was another strategy employed. Crop residue recyclable nutrients were considered where legume crops were part of the crop rotation system. The Cornell Nutrient Management Planning System (CNMP) combines all these elements into a comprehensive whole farm plan developed for the Maxwell Farm in the Graywood Gully watershed. Modified nutrient management plans were developed for the other farms involved in BMP watersheds. Also, highly erodible fields were identified as part of the farm planning process. Here row crop planting was limited to 1 year out of 6 using conventional mold board plowing. If reduced tillage systems were used, row crops could be planted in a tighter rotation. None of the farms in the BMP watersheds was routinely using cover crops. Cover crops became part of the plan to be used where corn was harvested as silage.

Results

The use of the soil test data in conjunction with the Cornell Cropware software led to a recommendation that cooperating farms

Table 2
Soil nitrate and phosphorus (kg/ha) for the Maxwell Farm.

Field	Ha	2002		2005		2007	
		NO_3	P	NO_3	P	NO_3	P
5	1.8	83	543	ND	ND	34	152
6	6.9	114	543	95	38	18	24
7A	2.0	252	55	22	48	ND	ND
7B	2.0	252	55	24	43	17	35
7C	2.0	252	55	17	26	ND	ND
7D	2.0	252	55	20	32	18	25
7E	2.0	252	55	20	22	46	19
8	4.8	26	22	36	19	12	32
9	3.2	20	16	57	13	44	46
10	1.6	32	7	46	12	12	18

Location of fields is shown in Fig. 2. ND = no data.

apply fertilizer only as needed for optimum crop production. In general, fertilizer use was reduced and is now based on sampling plus credits for nutrients recycled from manure and crop residues. This system credits the nitrogen, phosphorus, and potassium recycled from manure applications plus nitrogen recycled from cover crops and legume sod crops. The reduction in fertilizer use has varied considerably. On farms where manure was applied, the reduction has been over 50% for both nitrogen and phosphorus.

We focus on the Maxwell Farm in the Graywood Gully sub-watershed as an example of the results obtained to develop management practices in the various sub-watersheds. Fields 5 and 6 at Graywood Gully had high P, K, Ca, and Mg in 2002 prior to management (Tables 2 and 3, Fig. 2). Phosphorus Index (PI) calculations indicated that these two fields in the Graywood Gully watershed had excessive values based on erodibility and high soil test P values (Table 2, Ketterings et al., 2005). Fertilizer and manure applications were curtailed and the fields were planted to grass pasture and grass waterway.

The Maxwell Farm in the Graywood Gully watershed was the first to reduce fertilizer applications for crop production. Prior to the 2003 crop season, the standard application of P for planting corn was band application at planting of 52 kg P (as mono-ammonium phosphate)/ha. This rate was reduced to 29 kg/ha (Ristow et al., 2007). By the 2004 cropping season, applications rates were adjusted for every field (Fig. 2) based on recommendations from the Cornell Cropware program. The maximum P rate on any cornfield was 29 kg/ha with lower rates for fields with high soil test P values. Hydrologically sensitive and concentrated flow areas were delineated as part of the PI assessments. Manure spreading was not allowed within 30.5 m of such areas. By 2007, soil test P levels were reduced (range 72 to 95%) in high P fields (Fields 5 and 6, Table 2).

Nitrogen fertilizer application rates were also adjusted based on Cornell Cropware recommendations and soil testing. At Graywood Gully for example, fields 5, 6, and 7 had high nitrate levels (Table 2, Fig. 2). Management recommendations considered the nitrogen recycled from manure applications plus the nitrogen released from biological decomposition of legume crops and/or cover crops. Fields planted in

Table 3
Soil pH, organic matter (OM) and cations (kg/ha) for the Maxwell Farm.

Field	pH	K	Mg	Ca	Al	Fe	Mn	Zn	OM
5	7.2	1406	1277	6686	8	9	36	10	8
6	7.2	1406	482	3674	18	4	43	4	2
7A	7.1	241	510	4166	10	1	29	3	2
7B	7.1	241	510	4166	10	1	29	3	2
7C	7.1	241	510	4166	10	1	29	3	2
7D	7.1	241	510	4166	10	1	29	3	2
7E	7.1	241	510	4166	10	1	29	3	2
8	7.3	196	521	3898	11	2	36	2	3
9	7.3	151	470	3987	12	1	21	1	2
10	6.8	134	487	3282	15	1	29	0	3

Location of fields is shown in Fig. 2.

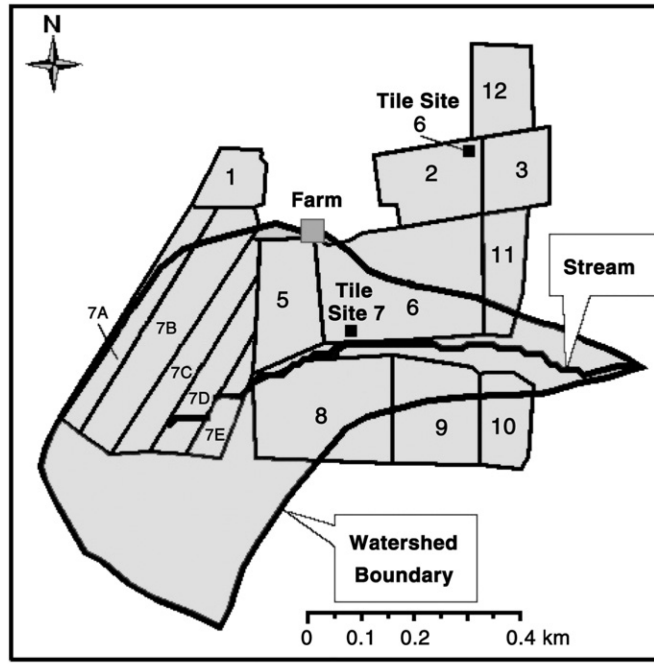

Fig. 2. Map showing various fields (numbers) within the Graywood Gully watershed where soil tests were performed.

corn after alfalfa-grass sods had only 26 kg/ha of nitrogen fertilizer applied as a starter at planting time (Lawrence et al., 2008). Other fields had rates based on manure applications and years from a sod crop in the rotation. The maximum application rate of nitrogen was 95 kg/ha where no manure was applied, and it was at least 2 years from a sod crop.

A manure management plan was implemented on the Maxwell Farm in the Graywood Gully watershed. This is the only farm that had close control of its management practice implementations to allow for farm data collection and monitoring of changes. The plan allowed for manure to be applied on fields with the greatest needs based on soil tests but prevented manure applications at times and in places where there was a high risk of runoff. For example, winter manure spreading was eliminated in Graywood Gully by hauling manure to land in another watershed that met acceptable standards for winter application. In spring and/or summer, manure was applied to fields in the Graywood Gully sub-watershed as prescribed by the whole farm plan. Modifications in farm fertilizer and manure practices were fully implemented at the Maxwell Farm by 2004 and were successful.

Progressively significant decreases in nitrate concentrations in stream water exiting the Graywood Gully sub-watershed were observed from 2002 to 2007 (Makarewicz et al., 2009). By 2007, soil nitrate levels in high nitrate fields in this sub-watershed were reduced as much as 92% (Table 2) further suggesting that the reduction in soil nitrate levels was due to the implementation of fertilizer and manure management practices rather than to changes in weather and soil conditions. Runoff erosion has also been reduced by use of land management plans, water and sediment control basins, strip cropping, and subsurface drainage (Makarewicz et al., 2009).

Management practices implemented

Based on the initial results of the Phosphorus Runoff Index, soil testing, manure analysis, and the Cropware Management software, both structural and cultural management practices were implemented in several watersheds. The cooperating farmers accepted and voluntarily implemented the recommended BMPs. Various structural and cultural management practices were implemented in the six sub-watersheds of Conesus Lake. The number of BMPs implemented from watershed to watershed varied (Table 1). At Graywood Gully for example, “Whole Farm Planning” and a multitude of BMPs were introduced while no BMPs were introduced at North McMillan Creek, a mostly forested reference watershed (Makarewicz, 2009).

A description of BMPs introduced to each watershed follows.

Graywood Gully (Maxwell Farm)*:* At Graywood Gully, major BMPs were introduced initially and minor BMPs over the next 2 years. This small watershed (38.1 ha) contains one farm with about 100 dairy cows plus an equal number of calves and heifers. It was the focus of much of our efforts as “Whole Farm Planning” was introduced and both structural and nonstructural BMPs were implemented. Corn, alfalfa, grass, and wheat are produced on a total of about 200 ha of land (Fig. 2).

Structural practices in the Graywood Gully watershed included subsurface drainage, installation of strip cropping, roof water separation, and grass filter strips. Over 7000 m of subsurface drainage was installed on the farm in 2004 to improve infiltration and to manage runoff. Approximately 350 m^2 of grass filter strip was established in a concentrated flow area in 2006 to reduce erosion.

A grass filter strip of approximately 550 m^2 was improved to retain runoff from a commodity (silage) storage area at Maxwell Farm in the watershed. A small barnyard area on the same dairy farm had been used as a nighttime exercise lot and became a source of barnyard manure runoff. It was replanted to sod and used only when the soil was dry.

Roof water separation was installed where needed. This allowed for the clean water to stay clean and to be safely discharged away from the barnyard areas.

Contour tillage strips were established in the watershed on approximately 13 ha, and alternate strips were planted to soil conserving crops. Grass filter strips were established adjacent to the Graywood Gully creek and on Field 10 (Fig. 2) in a highly erodible concentrated flow area and along the south side of Field 6. Fall tillage was eliminated except for planting of winter wheat. Cover crops were planted on fields following corn silage harvest in Graywood (Maxwell Farm) starting at the end of the 2004 crop season.

Cattle were fenced out of the Graywood Gully stream. Instead, a standpipe and watering source were installed in a heifer pasture area in an adjacent watershed with subsurface drainage to a road ditch. Heifers no longer drink from a wetland area in the pasture.

Manure management practices were fully implemented on the Maxwell Farm in this watershed by 2004. The Cornell Cropware program (CNMP Maxwell Farm Plan) generated manure spreading schedules and rates of application. This resulted in the elimination of winter spreading in this hydrologically sensitive, steep-sloped watershed. For this farm, manure was transported to additional property outside the watershed that met the guidelines for suitable winter application. However, there was one exception when winter spreading on the Graywood watershed occurred for a few days due to extreme weather (sub-freezing temperatures and significant snowfall) and concern for operator safety on the highway. The impact of this short burst of manure application on stream water quality is well documented in Lewis and Makarewicz (2009).

Long Point Gully (Dairy Knoll Farm, Gray Farm): In 2003 the one dairy operation in Long Point watershed ceased operations. The issues of milking center waste, roof water separation, and open barnyards were no longer a problem. The land was leased to Dairy Knoll Farms who operate a large dairy *located* outside the watershed. The largest structural practice implemented was a dairy manure storage structure at Dairy Knoll Farms. This allowed them to eliminate winter spreading and to use manure injection or incorporation rather than surface application. These practices were implemented in spring 2004. This farm has a Concentrated Animal Feeding Operation (CAFO) plan that was developed by a private consultant group. Land in the Long Point watershed is operated according to the requirements in that plan. Practices required in that plan are similar to those used in the Graywood (Maxwell Farm) watershed.

Cottonwood Gully (Barber Farm): Previous work had determined that soil loss from this small watershed (98.8 ha) was high and conservatively estimated at 130 tons (metric)/year in the 1990s (Makarewicz et al., 2001). Row crop (corn, red kidney and soybean) production dominates in this watershed. Initially, BMPs implemented were limited to two major efforts: the construction of water and sediment control basins (gully plugs) with associated subsurface drainage plus strip cropping designed to retard soil movement on the watershed. After BMP implementation, a 28% reduction of acreage in row crop production occurred with conversion to long-term vegetative type crop (alfalfa-grass hay), starting in the fall of 2006. Other BMPs subsequently implemented included cover crops planted on fields following corn and bean harvest. Fall tillage was eliminated except for planting of winter wheat. In 2007 this farm adopted another practice, the use of zone tillage on a limited acreage. This minimum tillage system replaces moldboard plowing and reduces soil erosion potential.

Sand Point Gully (Gray and Hainsworth Farms): Dairy heifers had access to this stream along a relatively steep portion of this watershed. The Livingston SWCD developed a rotational grazing plan that included fencing so strips could be rotated and water troughs could be provided for the cattle. Cattle were fenced out of the creek starting in May of 2003 (WY 2). Two gully plugs and tiles were also installed in a small portion of the watershed in November of 2002 prior to the beginning of this project. Cropland rotation, a cultural management practice, resulted in an increase of soybeans in this watershed from zero in 2002 to 33.0 ha in 2003 to 83.7 ha by 2007, almost 45% of the watershed.

Sutton Point Gully (White and Lombardo Farms): No physical infrastructure improvements were implemented in this watershed until 2007 (WY 5) when gully plugs were added at the end of the project. However, a significant and increasing portion of the watershed has been in alfalfa/grass production since WY 1 (37% in WY 2 to 60.3% in WY 5).

North McMillan Creek: Only 12% of the watershed is currently in agriculture and over 77% in vacant land (abandoned land including agricultural parcels in early forest succession) and in single family use (SOCL 2001). No management practices were implemented in this “reference” watershed.

Discussion

The role of Cornell Cooperative Extension and farm agencies, such as Soil and Water Conservation Districts, USDA Farm Service Agency (FSA), and Natural Resources Conservation Service, is critical for a voluntary process. The BMPs implemented were all the result of voluntary efforts of farm cooperators. Cooperative Extension has the mission of outreach to farmers with information for improved decision-making. Ongoing informal and continuous education programs have resulted in a level of trust between farmers and extension educators. Locally based service agencies, such as SWCD and FSA, know the rural landscape and had access to funds to assist farmers.

A total of six watersheds were selected for this study (Fig. 1). Since a requirement of the researchers was to work on small watersheds to improve the signal to noise ratio, most of our BMP watersheds were less than 600 ha in size (Table 1) and often had only one or two major farm operators. Based on the field assessments, soil data, and in consultation with the farmer, a series of management plans were designed and implemented voluntarily. Fertilizer use was reduced based on soil testing data plus credits from recycled nutrients. All farmers achieved cost savings because previous practices had not given enough credit to soil reserves, manure, and sod crop nutrients. At Graywood Gully, a savings of over $3000 was realized in the first year of using the Cornell Cropware recommendations (Jacobs, 2005).

Unfortunately, it was beyond the goals of this project to fully evaluate the economics of all the farm practices.

Farming is a business where decisions are made by individuals who have major impacts on the local landscape, and practices are repetitive on an annual basis. Crops are produced during the growing season then harvested, stored, and marketed in the "non-crop" season or year-round as in dairy farming. This puts farmers into the mode of doing what has been successful for them in the past. An example is fertilizer applications. If the senior partner on a farm was applying 330 kg/ha of a 5-10-5 analysis fertilizer 50 years ago, the farm operator today may still be applying 330 kg/ha of a 6-24-24 analysis fertilizer. Fifty years ago, there were likely no more than 20 dairy cows on the farm, and they spent half the year on grass pasture. Today, with improved feeding and breeding, that same farm may have 100 dairy cows producing more manure per cow and having a greater nutrient (especially phosphorus) density. The result is high soil test phosphorus and potassium values, high PI scores, and increased runoff potential. To make significant changes, the farmer has to significantly change past practices. He has to tell the local fertilizer supplier that only one-third of the fertilizer used in the previous year is needed. That is not easy to do when the farmer perceives the risk of reduced yields and when the supplier is likely a friend in the community.

When extension educators teach the use of BMPs and when local service agencies provide technical assistance and financial incentives, farmers are much more likely to make changes. Most farmers still have the attitude, "If it's not broke, don't fix it." They need the education and awareness that it is broken if it contributes to sediment and phosphorus runoff that promotes weed growth in the lake. The "starter phosphorus" and "sod nutrient credit" studies done by Cornell and extension staff on local farms demonstrated that yields could be maintained while reducing purchased fertilizer inputs. Changes are implemented on the land at the local level, one farmer at a time.

Another simple example is cover crops. The farmer has to invest out-of-pocket cash to buy the seed and spend time and fuel to plant cover crops. It is something that was never done before in the Conesus watershed, and it had to be done at a time when fall harvest was demanding all of his labor. It was a major shift to allocate time and resources to plant a crop that will not generate a commodity to sell. But once the practice was started, it has continued.

In summary, the cooperating farmers accepted and voluntarily implemented recommended BMPs. Various structural and cultural management practices were implemented in the BMP watersheds of Conesus Lake. The number of BMPs implemented from watershed to watershed varied (Table 1). At Graywood Gully for example, "Whole Farm Planning" was introduced and a multitude of BMPs were implemented, while at North McMillan Creek no BMPs were introduced. The cultural and management changes implemented in these watersheds were generally done on a phase-in basis and based on scientific data. Farmers reduced fertilizer applications a little the first year and more the second year. Eliminating winter spreading on high PI or hydrologically sensitive fields required planning and increased fuel costs. It is now an accepted practice in the BMP watersheds. Runoff and erosion have been reduced by use of land management plans and subsurface drainage. Starting in 2007, highly erodible fields were planted to row crops using limited tillage (zone till) systems instead of using conventional mold board plowing on the Cottonwood watershed.

Most importantly, the impact of BMPs on downstream aquatic systems has been substantial. In general, significant reductions in total phosphorus , soluble reactive phosphorus, nitrate, total Kjeldahl nitrogen, and total suspended solids concentration and flux occurred by the second year and third year of implementation of BMPs (Makarewicz et al., 2009). At Graywood Gully for example, where Whole Farm Planning was practiced and a myriad of structural and cultural BMPs were introduced, the greatest percent reduction (average = 55.8%, range 47% to 65%) and the largest number of significant reductions in analytes (4 out of 5) were observed. Where fields were left fallow or planted in a vegetative type crop (alfalfa), reductions, especially in nitrate, were observed. Where structural implementation occurred, reductions in total fractions (dissolved + particulates) were particularly evident (Makarewicz et al., 2009). In general, both structural and cultural BMPs were observed to have profound effects on nutrient and soil loss; that is, soil and nutrients were retained on the land and not lost to downstream aquatic systems.

Interestingly, reductions in nutrients and soil delivered to downstream systems have had an effect on metaphyton, macrophytes, and microbial communities in the nearshore area of Conesus Lake where streams enter the lake. Comparisons of Pre-BMP (2–3 years) to the Post-BMP (4 years) periods at Cottonwood Gully, Graywood Gully, and Sand Point Gully (sites receiving the most extensive BMPs) revealed that algal cover was statistically lower than baseline in 8 of 11 sample years (72.7%) (Bosch et al., 2009a). Similarly, in macrophyte beds downstream from managed sub-watersheds, quadrat biomass decreased by 30–50% within 1 or 2 years of BMPs implementation and was statistically lower than Pre-BMP values in 7 of 11 sample years (Bosch et al., 2009b). In the three macrophyte beds where minimal or no BMPs were introduced, biomass was statistically indistinguishable from Pre-BMP values in 12 experimental sample years. Also, a major decrease in microbial levels in nonevent Graywood Gully stream water was evident after management practices were implemented. *Escherichia coli* levels in here dropped more than 10 fold to levels significantly below the 235 cfu/100 mL EPA bathing beach standard, while the yearly maximum for *Enterococcus* dropped by a factor 2.5 (Simon and Makarewicz, 2009a, b).

Clearly, BMPs implemented in the Conesus Lake watersheds have had major impacts within the nearshore of Conesus Lake. As Makarewicz et al. (2009) note, the utility and effectiveness of the implemented BMPs should allow regional policy makers and managers to develop strategies for improving watershed land usage while improving downstream water quality in the embayments, nearshore, and open waters of large lakes.

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The impact of agricultural best management practices on downstream systems: Soil loss and nutrient chemistry and flux to Conesus Lake, New York, USA

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ABSTRACT

Six small, predominantly agricultural (>70%) watersheds in the Conesus Lake catchment of New York State, USA, were selected to test the impact of Best Management Practices (BMPs) on mitigation of nonpoint nutrient sources and soil loss from farms to downstream aquatic systems. Over a 5-year period, intensive stream water monitoring and analysis of covariance provided estimates of marginal means of concentration and loading for each year weighted by covariate discharge. Significant reductions in total phosphorus, soluble reactive phosphorus, nitrate, total Kjeldahl nitrogen, and total suspended solids concentration and flux occurred by the second year and third year of implementation. At Graywood Gully, where Whole Farm Planning was practiced and a myriad of structural and cultural BMPs were introduced, we observed the greatest percent reduction (average = 55.8%) and the largest number of significant reductions in analytes (4 out of 5). Both structural and cultural BMPs were observed to have profound effects on nutrient and soil losses. Where fields were left fallow or planted in a vegetative type crop, reductions, especially in nitrate, were observed. Where structural implementation occurred, reductions in total fractions were particularly evident. Where both were applied, major reductions in nutrients and soil occurred. After 5 years of management, nonevent and event concentrations of total suspended solids in streams draining agricultural watersheds were not significantly different from those in a relatively "pristine/reference" watershed. This was not the case for nutrients.

Introduction

Best Management Practices (BMPs) are structural or cultural operational and maintenance practices intended to prevent or reduce the movement of sediment, nutrients, pesticides, and other pollutants from land to surface or groundwater: that is, to prevent nonpoint source pollution. Watershed management programs often cover several levels of geographic scales ranging from the large regional (e.g., Chesapeake Bay, Mississippi River basin) to small local projects representing a field or a single farm (Bishop et al., 2005). In general, BMPs are compatible with the traditional, voluntary approach to resource management practiced in the United States. Nevertheless, it was recognized in the 1990s that this management approach had failed to produce significant national reductions in nonpoint source pollution (Logan, 1990) and had not been widely accepted by the agricultural community, especially in the absence of cost sharing or a clear economic advantage or benefit of the practice (Logan, 1990; Napier et al., 1986). Even though nutrient export to downstream systems from agriculture crop production is believed to be high (e.g., Dillon and Kirchner, 1975; Hill, 1978; Neill, 1989; Correll et al., 1992), implementation of management plans has not been as effective as desired for several reasons (Beegle et al., 2000). Livestock systems, which utilize pastureland for grazing animals and cropland for disposal of manure waste, represent an agricultural activity in which effectiveness of BMPs was not adequately demonstrated (Brannan et al., 2000). Part of the problem was that local implementation of BMPs did not always translate to observable ecosystem or watershed-wide reductions in nutrients and soil loss (Meals, 1996). In fact, BMPs introduced in small portions of watersheds in the Finger Lakes/Lake Ontario region have not effectively demonstrated a connection between nutrient and erosion reduction and visual or measurable lake-wide reductions of nutrients, metaphyton, or aquatic plant populations (Bosch et al., 2001).

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Assessment of BMPs has ranged from subfield experiments (e.g., plots) to large basin-scale monitoring with various sampling and statistical designs. On the subfield level, monitoring studies have demonstrated that agricultural BMPs are effective at decreasing sediment and phosphorus (P) loss (e.g., Murray, 2001; Robillard and Walter, 1984; Udawatta et al., 2002), but relating these results to improvements in water quality has been difficult (Bishop et al., 2005; Meals, 1996; Schindler, 1998). In basin-wide studies, assessments of BMP impacts include monitoring of stream water quality before and after implementation of management practices (e.g., Inamdar et al., 2001; Udawatta et al., 2002) and the use of watershed models based on empirical data (e.g., Benaman, 2002; Gitau et al., 2003). The inability to demonstrate mitigation of stress in downstream systems, however, is the result of confounding factors inherent in nonexperimental evaluations and/or issues of areal size of the BMP relative to the watershed size (Makarewicz, 2009). For example, a single manipulation (i.e., a BMP) in a small area of a large watershed may not provide a large enough reduction in nutrients to affect demonstrable change in downstream nutrient concentration, nutrient loading, and metaphyton and macrophyte population size. Research directed at evaluating BMP impacts on a small watershed scale (e.g., Bishop et al., 2005; Gallichand et al., 1998; Galeone, 1999; Shirmohamadi et al., 1997) is minimal despite the watershed approach being recognized as a method of accounting for environmental variability (Bishop et al., 2005; Carpenter, 1998; USEPA, 1993). The small watershed approach provides an area that is large enough to capture nutrient and soil transport processes and dilution effects (Gburek et al., 2000) yet small enough to focus on nutrient loading from a single farm and BMPs adopted at that scale (Bishop et al., 2005).

Conesus Lake is one of the smaller Finger Lakes in western New York, is used for recreation and fishing, and is a source of municipal water for five local communities. The shoreline area is densely populated with residences, primarily year-round homes. The upstream area is a mixture of agricultural land and mixed deciduous hardwood forests encompassing an area of 16,714 ha. In 1999 about half of the entire land use within the Conesus Lake watershed was and continues to be in agriculture. Much of the agriculture (>70%) is concentrated in the western sub-watersheds of the lake (Fig. 1; SOCL, 2001). The deep, well-drained, glacially derived limestone soils that dominate the watershed are productive and support field crops (field corn, forages, winter wheat, soybeans), vegetable crops (dry beans, sweet corn), and a small acreage of vineyards. Dairy farms are the major animal agriculture with a few livestock and horse farms. More information on Conesus Lake and its surrounding watershed may be found in this issue (Makarewicz, 2009).

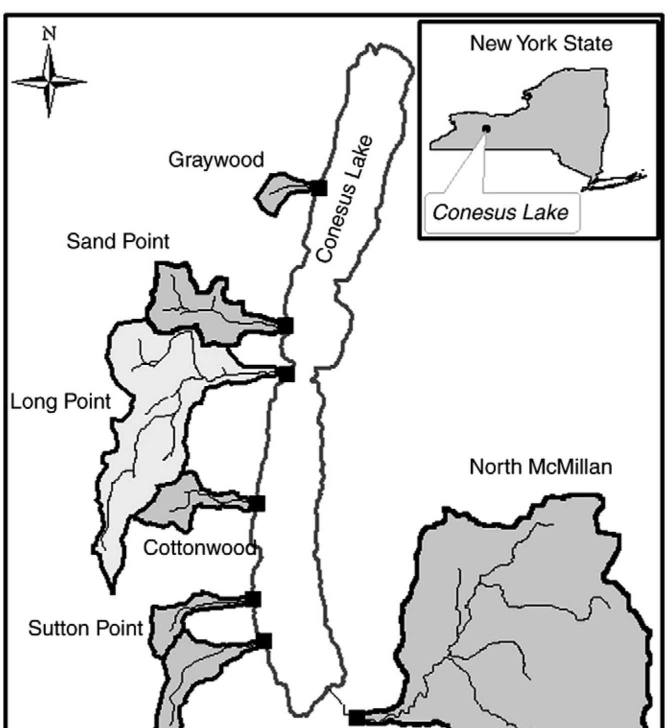

Fig. 1. Conesus Lake and sub-watersheds monitored over the study period. Squares represent sampling sites at the base of the watershed.

The Conesus Lake Watershed Project (Makarewicz, 2009) describes research that quantitatively evaluates and characterizes the effectiveness of various agricultural BMPs in minimizing the local impact of stream loading on water quality. The overall goal of this project was to implement a series of BMPs on individual farms in selected experimental sub-watersheds with an objective of retaining soil and nutrients within the watershed and simultaneously reducing the loss of nutrients and soil to downstream systems. Here we evaluate the effectiveness of agricultural management practices and strategies in improving the water quality of streams by evaluating nutrient and soil loads and their associated concentrations on six watersheds. Specifically, we hypothesize not only reductions in nutrient and soil concentrations and loading from streams draining managed agricultural sites but also reductions in nutrients in soil and in groundwater leaving the watershed.

Methods

Management practices

Within the Conesus Lake watershed, dairy and row crop farms were the focus of Best Management Practices (BMPs) designed to retain soil, nitrate, organic nitrogen (N), and P. Structural BMPs include construction of manure lagoons, terraces, buffer strips, and sediment control basins, while nonstructural or cultural BMPs include such practices that minimize site disturbance through sound planning and design and include cropping sequence, soil testing, fertilization rates, and tillage practices. Structural BMPs (e.g., filter strips) modify the transport of the pollutant to waterways while cultural BMPs (e.g., timing of manure spreading, fertilization rates) are designed to minimize pollutant inputs to waterways through land management practices (Herendeen and Glazier, 2009; Agouridis et al., 2005, Logan, 1990). Structural and cultural BMPs implemented in this study included nutrient management (manure and fertilizer), construction of water and sediment control basins (gully plugs), crop rotation and rotational grazing, removal of acreage from crop production, and in general, improved infrastructure and practices associated with Whole Farm Planning (Table 1).

BMP implementation generally began in the spring of the first year of the study, 9 months after initiation of stream sampling (Herendeen and Glazier, 2009). Structural BMPs were generally in place by the end of the first year, while other practices were introduced in the second year of the study. No management practices were introduced to the North McMillan watershed, our reference watershed. Details on selection of watersheds and implementation of BMPs may be found in Herendeen and Glazier (2009). Acreage of land in various crops was provided by the N.Y. Farm Bureau Office in Livingston County, NY.

Stream discharge

A Water Year (WY) was defined as the period from 1 September to 31 August of the following year. For example, Water Year 1 (WY 1) extended from 1 Sep 2002 to 31 Aug 2003; Water Year 2 (WY 2) extended from 1 Sep 2003 to 31 Aug 2004, etc. A total of 5 water years of daily discharge data was collected starting on 1 Sep 2002 and

Table 1
Watershed area, mean annual daily discharge and daily weighted discharge for seven sub-watersheds of Conesus Lake.

	Cottonwood Gully	Graywood Gully	Sand Point Gully	Sutton Point Gully	Long Point Gully	Southwest Creek[b]	North McMillan Creek	Annual Total for 7 watersheds	Rainfall (cm)
Major management practice implemented	Gully plugs, 28% reduction in cropland, 80% conversion to alfalfa	Whole farm planning (# of BMPs)	Cattle fenced from streams, rotational grazing pens (9.5% of land)	Conversion of 60% cropland to alfalfa	Removal of 37% of crops and cows	Manure pit installed	Reference watershed no BMPs		
Watershed area (ha)	98.8	38.1	188.0	67.5	587.9	176.4	1778.2	2985	
Mean annual discharge (m^3/d)									
(Water Year 1)	1487	629	2324	566	4673	4223	28,790	42,692	87.3
(Water Year 2)	2274	2878	1739	2865	8631	2940	38,796	60,123	110.3
(Water Year 3)	2857	2316	4320	2451	12,708	5214	34,330	64,196	117.8
(Water Year 4)	1556	767	1370	1157	6987	4309	17,434	33,580	90.1
(Water Year 5)	2897	1261	1192	1749	5695	2621	30,430	45,845	95.8
Mean daily discharge (m^3)	2214	1571	2189	1758	7739	3861	29,969	49,287	101.4
Mean annual discharge (m^3/ha/d)	22	41 (29)[a]	12	26	13	22	17	16.5	

Precipitation data were measured in the Graywood Gully watershed. A "Water Year" extends from 1 September to 31 August.
[a] Discharge weighted by extended watershed of Graywood Gully.
[b] Water year is displaced 3 months later compared to other watersheds. Chemistry data for this watershed are presented elsewhere.

ending on 31 Aug 2007. During this period, the stage of six Conesus Lake tributaries (Fig. 1) (Graywood Gully, Long Point Gully, Sand Point Gully, Cottonwood Gully, Sutton Point Gully, and North McMillan Creek) was monitored continuously with a differential pressure transducer (Isco 720) attached to an ISCO recording flow meter (Model 4220 or 6712) equipped with automatic samplers. Southwest Gully was not monitored continuously and is not reported on here. Submerged probe sensors were attached to the cement or metal culverts or bridges at the base of each watershed. In open-channel discharge monitoring, fixed culverts/bases substantially reduce error associated with shifts in stream basin morphometry (Rantz 1982; Chow 1964). Each monitoring station was maintained and calibrated weekly. Streambed movement was verified monthly but was not observed. In addition, discharge measurements were verified annually by remeasuring cross-sectional areas of streams beds, pipes, or cement bridge beds at different stage heights. The velocity–area method (Rantz, 1982) and a calibrated Gurley current meter (Chow 1964) were used to construct rating curves for all streams. During time periods where stage was not available due to probe damage, extreme hydrometeorological events, vandalism, power failure, etc., we estimated stage by using manual readings taken during calibration visits or by using regressions developed from other creek monitoring stations in the Conesus Lake watershed (Rantz, 1982). For example, during a portion of the winter of 2005 when ice dams affected stream depth and discharge at Graywood Gully, we determined discharge here by using a discharge regression from the nearest study tributary, Sand Point Gully ($r^2 = 0.67$). Similarly, North McMillan Creek and Long Point Gully had abnormally high discharge during the winter of 2004 due to a cracked probe and vandalism. Consequently, hourly discharge here was determined by linear regression ($r^2 = 0.74$ and 0.98, respectively) based on hourly discharge from Cottonwood Gully.

Soil analysis

Cornell Cooperative Extension personnel collected soils from several fields in the Graywood Gully and Cottonwood watersheds in 2002, 2005, and 2007 (Herendeen and Glazier, 2009). At least 10 replicate soil samples were taken at plow depth from each field using a soil probe. Soil analyses were conducted by the Cornell Nutrient Analysis Laboratory (CNAL, 2007). Bio-available soil nutrients were extracted from soils with Morgan's solution (sodium acetate and acetic acid) buffered at pH 4.8. Activated carbon was added during extraction to aid in organic matter removal and to help decolorize the extraction solution. After 15 min of vigorous shaking at 180 rpm, the extraction slurry was filtered through a fine-porosity filter paper (Whatman #2 or equivalent). Bio-available NO_3–N and PO_4–P were measured using an Alpkem Automated rapid flow analyzer (CNAL Method 1030; Morgan, 1941).

Water sampling and chemistry

Weekly grab stream water samples were taken and analyzed for total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate + nitrite ($NO_3 + NO_2$), total Kjeldahl nitrogen (TKN), and total

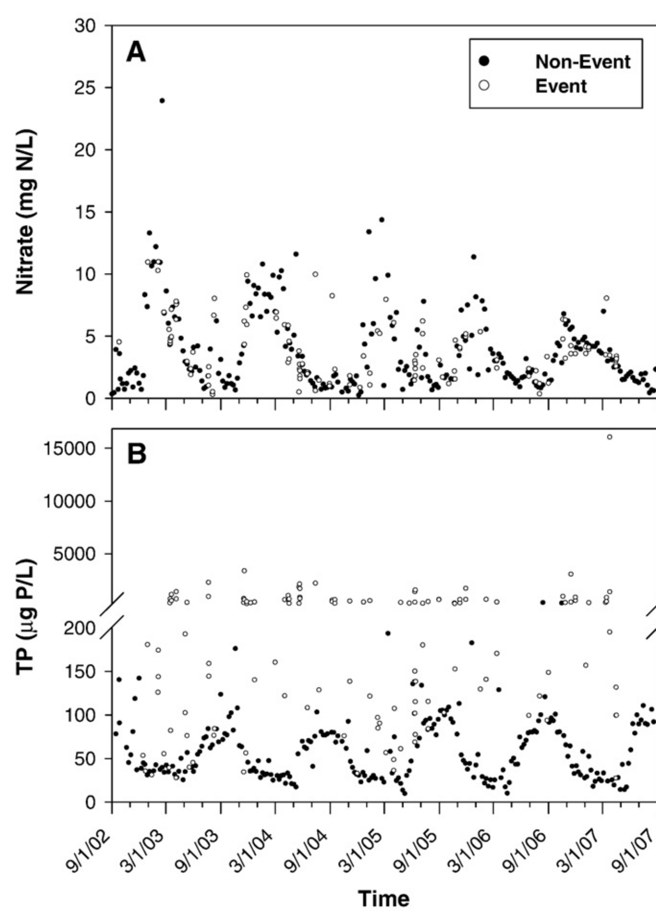

Fig. 2. (a) Time trend in nitrate (Cottonwood Gully) and (b) in total phosphorus concentration (Sand Point Gully), WY 1 to WY 5. A Water Year (WY) is defined as the period from 1 Sep to 31 Aug of the following year.

Table 2
Rating curves of study streams draining into Conesus Lake.

Location	Rating curve equation (Y = discharge in m^3/s, x = stream level in mm)	
Graywood Gully	$Y = 0.000007x^2 - 0.0001756x$	$r^2 = 0.85$
Sand Point Gully	$Y = 0.000007779x^2 + 0.000111669x + 0.000126538$	$r^2 = 0.82$
Long Point Gully	$Y = 0.000030x^2 + 0.00019456x$	$r^2 = 0.86$
Cottonwood Gully	$Y = 0.000006985x^2 - 0.000141730x + 0.000033697$	$r^2 = 0.85$
Sutton Point Gully	$Y = 0.00000000094x^4 - 0.00000010790x^3 + 0.00000947831x^2 - 0.00012311640x$	$r^2 = 0.96$
Southwest Creek	$Y = 0.0088171x^2 + 0.6207933x - 0.0043616$	$r^2 = 0.94$
North McMillan Creek	$Y = 0.000019492x^2 - 0.001387173x + 0.005397970$	$r^2 = 0.82$

suspended solids (TSS) for all watersheds. A hydrometeorological event sample was defined as a rise in the creek level of at least 2.54 cm in 30 min. If this occurred, an ISCO refrigerated sampler (4 to 5 °C) began taking hourly samples until the event had ended. Hourly samples representing rising and falling limbs of the event hydrograph were composited into two water samples. In addition, grab samples of water from two tiles draining a managed (Site 6) and an unmanaged field (Site 7) were taken (Fig. 2, in Herendeen and Glazier, 2009) in and near the Graywood Gully sub-watershed from April through July 2006 (WY 4).

Water samples were taken, preserved, and analyzed using standard methodologies (USEPA, 1979; APHA, 1999). Samples were analyzed for TP (APHA Method 4500-P-F), TKN (USEPA Method 351.2), $NO_3 + NO_2$ (APHA Method 4500-NO_3-F), and TSS (APHA Method 2540D). Except for TSS, all analyses were performed on a Technicon AutoAnalyser II. Method Detection limits were as follows: SRP (0.48 μg P/L), TP (0.38 μg P/L), $NO_3 + NO_2$ (0.005 mg N/L), TKN (0.15 μg N/L), and TSS (0.2 mg/L). Sample water for dissolved nutrient analysis (SRP, $NO_2 + NO_3$) was filtered immediately on site with 0.45-μm MCI Magna Nylon 66 membrane filters and held at 4 °C until analysis the following day.

All water samples were analyzed at the Water Chemistry Laboratory at The College at Brockport, State University of New York (NELAC — EPA Lab Code # NY01449) within 24 h of collection. In general, this program includes biannual proficiency audits, annual inspections and documentation of all samples, reagents and equipment under good laboratory practices. All quality control (QC) measures are assessed and evaluated on an on-going basis. As required by NELAC and New York's ELAP certification process, method blanks, duplicate samples, laboratory control samples, and matrix spikes are performed at a frequency of one per batch of 20 or fewer samples. Field blanks (events and nonevents) are routinely collected and analyzed. Analytical data generated with QC samples that fall within prescribed acceptance limits indicate that the test method was in control. For example, QC limits for laboratory control samples and matrix spikes are based on the historical mean recovery plus or minus three standard deviations. QC limits for duplicate samples are based on the historical mean relative percent difference plus or minus three standard deviations. Data generated with QC samples that fall outside QC limits indicate that the test method was out of control. These data are considered suspect and the corresponding samples are reanalyzed. As part of the NELAC certification, the lab participates semi-annually in proficiency testing program (blind audits) for each category of ELAP approval. If the lab fails the proficiency audit for an analyte, the lab director is required to identify the source and correct the problem to the certification agency.

Calculation of nutrient load

Daily, weekly, monthly, and annual losses of nutrient and soil from each watershed were calculated from continuous discharge measurements and from water chemistry samples collected for each watershed during events and nonevents. During nonevent periods, hourly discharge was summarized into a daily discharge and multiplied by that period's analyte concentration. For non-detect samples, a zero was entered into the calculation; for samples with a recognizable absorbance peak but below the level of quantitation, a concentration of 1/2 the detection limit was used to calculate loss from the watershed. Only eight non-detect samples were measured in the entire study. By summing hourly discharge independently for rising and falling limbs and multiplying by the respective analyte concentration of the composite rising and falling limb of the hydrograph, we calculated the event loss of an analyte via stream drainage from a watershed. The end of an event was defined as the point in time where the descending limb leveled off.

Statistics

A Kolmogorov–Smirnov *D* test revealed that even with natural log (ln) transformations, the data were generally not normally distributed. Moderate violations of parametric assumptions have little or no effect on substantive conclusions from Analysis of Variance (Zar, 1999). Even so, the ln transformed data were used to ensure near normal distribution (Zar, 1999). Natural log transformation is a common practice in hydrologic and watershed studies (Cohn, 1995; USEPA, 1997). Analysis of Covariance (ANCOVA) was used to test for temporal trends in nutrient loading or concentration with stream discharge as the covariate and loading/concentration × sampling year as the interaction term. A significant interaction term indicated that the slope of the loading/nutrient concentration-stream discharge

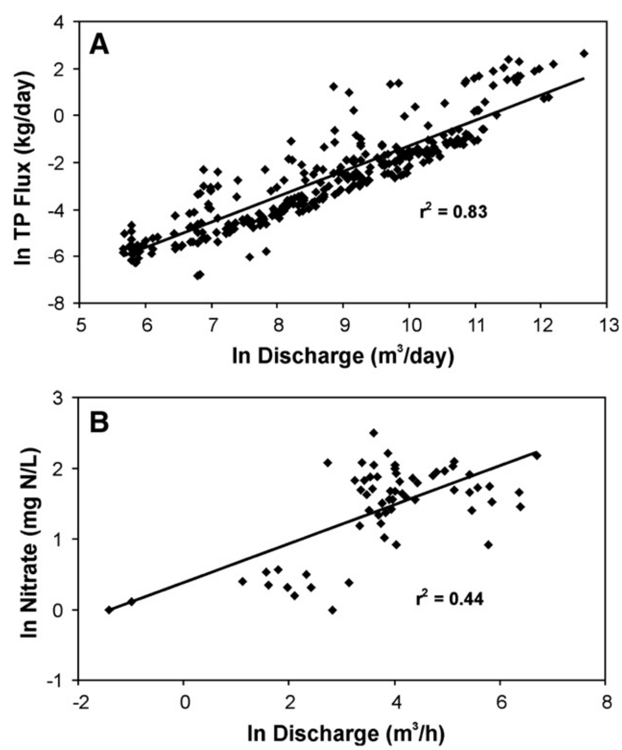

Fig. 3. (a) Total phosphorus (North McMillan Creek) and (b) nitrate (Graywood Gully) concentration versus discharge in WY 5.

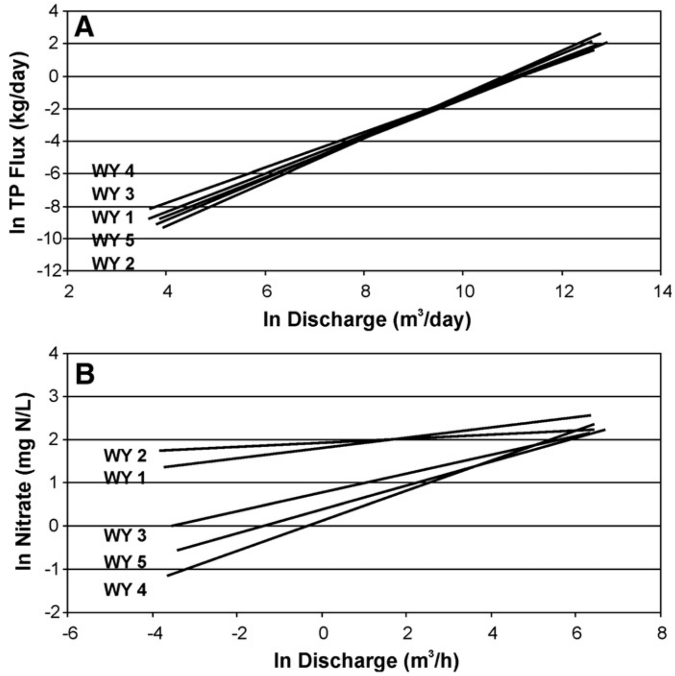

Fig. 4. (a) Total phosphorus flux (North McMillan Creek) and (b) nitrate concentration (Graywood Gully) versus discharge from WY 1 to WY 5. A Water Year (WY) is defined as the period from 1 Sep to 31 Aug of the following year.

regression line was dependent on sampling year. Regression slopes were compared using a pair-wise *t*-test in which significance levels were corrected using the Bonferroni procedure. Regression line elevations were also analyzed for significant differences using a Bonferroni test of estimated marginal means for each sampling year. The Bonferroni procedure offers an adjustment for multiple comparisons and is considered a conservative procedure for post-hoc analysis (Norleans, 2001). Marginal means are analyte concentration means after adjustment for the discharge covariate. The statistical analysis assumed that watersheds were similar in physical, hydrological, and soil properties and that meteorological variations among the closely spaced watersheds were negligible during the study.

Results

Discharge

Correlation between stream stage height and discharge for experimental and reference watersheds was generally very good with coefficient of determination values (r^2) exceeding 0.82 (range 0.82–0.96; Table 2). As expected, considerable variability in precipitation occurred over the 5-year study period. Annual discharge reflected annual precipitation and was highest in WY 3 (Sep 2004–Aug 2005) and lowest in WY 4 (Sep 2005–Aug 2006, Table 1). Overall, greatest discharge occurred in the largest watershed (North McMillan Creek, mean daily = 29,969 m^3) and the lowest in the smallest watershed (Graywood Gully, 1571 m^3). However, areal weighted discharge (m^3/ha) was within the same order of magnitude (13 to 43 m^3/ha), with Graywood Gully having the highest discharge. If weighted by external sources outside the traditional topographical watershed definition, the weighted average dropped to 31 m^3/ha (discussed further in Noll and Magee, 2009).

Analyte concentration and flux

A time-trend analysis indicated that some seasonal patterns existed. For example, in all watersheds, stream $NO_3 + NO_2$ concentration decreased from spring to summer and then increased in early winter (e.g., Fig. 2a). A similar annual cycle of TP and SRP concentrations was observed with a peak in late summer (e.g., Sand Point Gully; Fig. 2b). Because of seasonal cycles, simple plots of time trends do not offer a rigorous statistical approach for analysis. Also, nutrient and TSS concentrations and fluxes are controlled by both analyte mass and water volume (e.g., Figs. 3a and b), two factors which can vary independently. Thus concentrations and fluxes can increase or decrease in response to mass loading and/or dilution. To account for differences in annual discharge in each watershed (Table 1), we utilized the marginal means of the discharge adjusted analyte (TP, SRP, TSS, TKN, and $NO_3 + NO_2$) concentration and flux from the ANCOVA analysis to compare the slopes and elevations of each yearly regression line. We provide two complete examples of this analysis including TP flux of North McMillan Creek and changes in $NO_3 + NO_2$ concentration of Graywood Gully. For all other parameters and creeks, only statistical data are provided for evaluation of elevations of marginal means of flux and concentration.

North McMillan Creek

Temporal trends in TP flux in North McMillan Creek were evaluated by considering regression slope of TP flux versus stream discharge for each collection year using ANCOVA with discharge as the covariate (Fig. 4a). Pairwise *t*-test comparisons of ANCOVA regression slopes for each collection year indicated that the WY 5 regression slope was not significantly different ($df = 1,4$; $P > 0.05$; Table 3) from those of all previous water years (1, 2, 3, and 4) (Fig. 4a).

Utilizing marginal discharge means of adjusted TP flux (from ANCOVA analysis), we compared the difference in elevations of each regression line (Table 4). Discharge adjusted marginal mean TP flux remained approximately the same from WY 1 to WY 5 (0.12, 0.15, 0.14, 0.16, 0.15 kg P/D, respectively). A post-hoc Bonferroni test (Table 4)

Table 3
ANCOVA table and slopes of regression lines for respective years for the relationship between discharge and total phosphorus flux, North McMillan Creek.

	df	*F*	*P*
Water Year	4	7.308	0.000
Discharge	1	5019.680	0.000
Year*Discharge	4	6.981	0.000

		Pairwise comparisons				
WY		1 ($n = 365$)	2 ($n = 366$)	3 ($n = 365$)	4 ($n = 365$)	5 ($n = 334$)
	Slope	1.20	1.35	1.19	1.08	1.29
1			0.277	0.225	0.123	0.132
2				1.000	1.000	1.000
3					1.000	1.000
4						1.000
5						

P values of pair-wise comparisons of the slopes of the regression for each water year (WY) are provided from a Bonferroni multiple comparison test. Probability values <0.05 indicate a significant difference. General linear model in SPSS version 14.0 (SPSS Inc.).

Table 4
Probability values from a comparison (Bonferroni Test) of inter-year elevations of regression lines from Water Years (WYs) 1 to 5 utilizing marginal means of ANCOVA for the relationship between discharge and total phosphorus flux, North McMillan Creek.

WY	1	2	3	4	5
1		0.277	0.225	0.123	0.132
2			1.000	1.000	1.000
3				1.000	1.000
4					1.000
5					

revealed that WY 5 regression line elevation was not significantly different from all other years ($P>0.05$). The marginal means of the discharge versus TP flux regression lines did not appear to change over time (Table 7).

Graywood Gully

Similar to North McMillan Creek, temporal trends in NO_3+NO_2 concentration of Graywood Gully were evaluated by considering the slope of the regression line of NO_3+NO_2 concentration versus stream discharge for each collection year using ANCOVA with discharge as the covariate. Pairwise *t*-test comparisons of ANCOVA regression slopes for each collection year indicated that WY 5 regression slope was not significantly different ($df=1,4$; $P>0.05$; Table 5) from WYs 3 and 4 but was significantly different from WYs 1 and 2 ($df=1,4$; $P<0.001$). Similarly, WYs 3 and 4 were significantly higher than WYs 1 and 2 (Table 5), while WYs 1 and 2 were not significantly different. Over the study, NO_3+NO_2 concentrations decreased faster at lower discharge rates than at higher ones (Fig. 4b).

According to the statistical analyses, discharge adjusted mean NO_3+NO_2 concentration decreased from WY 1 to WY 5 (10.02, 7.77, 5.25, 3.71, and 4.20). A Bonferroni test (Table 6) revealed that WY 5 regression line elevation was significantly lower than in WYs 1 and 2 ($df=1,4$; $P<0.001$) but not in WYs 3 and 4 ($df=1,4$; $P>0.72$). Similarly, marginal means were significantly lower in WYs 3 and 4 than in WY 2 (Table 6). The marginal means of NO_3+NO_2 concentration decreased over time (Table 8). In general, the overall results from the six different watersheds are presented as the difference in the elevations of each regression line utilizing the marginal means of the discharge-adjusted analyte concentration and flux from the ANCOVA analysis (post-hoc Bonferroni) (Tables 7–12).

North McMillan Creek

The post-hoc analyses indicated that elevation of the marginal means of concentration and generally the flux of TP, TKN, NO_3+NO_2, SRP, and TSS in WY 1 were not significantly lower ($P>0.05$) than in WYs 2, 3, 4 and 5 (Table 7). There was no significant difference in the marginal means of flux for TKN and SRP among WYs 1, 2, 3 and 4. Only in WY 5 were marginal means of TKN and SRP significantly lower than in all previous years (Table 7). In general, the marginal means of analyte concentration and flux at North McMillan Creek did not decrease during the study (Table 7).

Graywood Gully

Unlike North McMillan Creek, significant decreases in the marginal means of the flux and concentration of TKN, NO_3+NO_2, TP, and TSS were generally observed from WY 1 to WY 5 (Table 8). There was some analyte to analyte variation in the time from initiation of a BMP to a significant response and in the level or intensity of a response. Significant decreases in NO_3+NO_2 and TKN concentration were observed by WY 3 while significant decreases in TP and TSS were observed a year later in WY 4. Significant decreases in flux mimicked changes observed in concentration but often preceded significant changes in concentration (e.g., TP, TSS, NO_3+NO_2) by a year (Table 8). This difference in the year of significant reduction between concentration and flux was a result of different degrees of statistical freedom between flux and concentration data. For concentration, the number of chemistry samples ranged from ~50 to 90 annually, while flux data were based on daily discharge (~300–400 measurements). No significant decrease was observed in SRP concentration or flux over the study period. Assuming that TP was dominated by the particulate fraction, the general trend from WYs 1 to 5 was a reduction in dissolved and particulate fraction concentration and flux from the watershed. Reductions in analyte concentration ranged from 47% to 65% from 2003 to 2007 (TP: 47%, TKN: 54%, NO_3+NO_2: 58%, TSS: 65%) (Table 13).

Long Point Gully

Inspections of the data revealed that significant ($P<0.05$) reductions in NO_3+NO_2, TP, SRP, and perhaps TKN concentration and flux were generally observed when compared to the initial WY 1 (Table 9). These decreases corresponded with increases in land taken out of crop production (e.g., Fig. 5). Although there was a significant decrease in TKN concentration and flux from WY 1 to WY 3 and from WY 1 to WY 5, the slight increase in TKN from 527 μg/L in WY 3 to 602 μg/L in WY 4 was large enough to make the 2006 value not significantly different from WY 1. Significant reductions in TSS concentration or TSS flux were not observed. Reductions of 42% and 53% in NO_3+NO_2 and SRP concentration were observed while the total fractions, TP and TKN, decreased 36% and 24%, respectively, from WY 1 to WY 5 (Table 13).

Cottonwood Gully

A general reduction in NO_3+NO_2, TSS, and perhaps TKN was observed over the study period. The post-hoc analysis indicated that the elevation of the marginal means of concentration and the flux was

Table 5
ANCOVA table and slopes of regression lines for respective years for the relationship between discharge and nitrate concentration, Graywood Gully.

	df	*F*	*P*
Water Year	4	9.703	0.000
Discharge	1	34.680	0.000
Water Year*Discharge	4	3.085	0.016

WY		Pairwise comparisons				
		1 ($n=74$)	2 ($n=93$)	3 ($n=87$)	4 ($n=67$)	5 ($n=62$)
	Slope	0.119	0.049	0.218	0.348	0.437
1			0.962	0.000	0.000	0.000
2				0.005	0.000	0.000
3					0.940	0.907
4						1.000
5						

P values of pair-wise comparisons of the slopes of the regression for each Water Year (WY) are provided from a Bonferroni multiple comparison test. Probability values <0.05 indicate a significant difference. General linear model in SPSS version 14.0 (SPSS Inc.).

Table 6
Probability values from a comparison (Bonferroni Test) of inter-year elevations of regression lines from WYs 1 to 5 utilizing marginal means of ANCOVA for the relationship between discharge and nitrate concentration, Graywood Gully.

WY	1	2	3	4	5
1		0.521	0.000	0.000	0.000
2			0.023	0.000	0.000
3				0.154	0.726
4					0.995
5					

significantly lower ($P<0.05$) compared to WY 1 for NO_3 flux (WYs 2 to 5), NO_3+NO_2 concentration (WYs 2 to 5), TKN flux (WYs 4 and 5), TKN concentration (WY 5), and TSS flux and concentration (WYs 4 and 5) (Table 10). There were no significant changes ($P>0.05$) in concentration and flux of SRP and TP over the study period (Table 10). The significant reduction (71%) in TSS concentration was one of the largest observed of any watershed studied (Table 13). Starting 2 years prior to the study, large portions of the cropland were rotated into a long-term vegetative type crop (alfalfa–grass hay). By WY 1, over 80% of the cropland was in alfalfa (Fig. 6). By WY 5, this land was still high in alfalfa (over 50%).

Sand Point Gully

The marginal mean concentrations of TKN, TP, SRP and TSS, and flux of TP and SRP were not significantly different over WYs 1 to 5 (Table 11). Marginal means of NO_3+NO_2 concentration and flux in WYs 2, 3, 4, however, were significantly lower than in WY 1. Unlike other watersheds where significant or insignificant reductions in concentration were generally mirrored in the results for the flux analysis, this was not true at Sand Point. Here, TSS and TKN flux were significantly lower in WY 1 even though TSS and TKN concentration showed no significant change. Reductions in TSS and TKN concentration were observed, but variability was high, thus reducing significance.

Sutton Point Gully

The post-hoc analysis indicated that the elevation of the marginal means of TP and SRP concentration and flux was not significantly lower ($P>0.05$) from WY 2 to WY 5 compared to WY 1. Marginal means of TSS concentration and flux in WYs 4 and 5, however, were significantly lower ($P<0.05$) than in WYs 1, 2 and 3 (Table 12). Nitrate concentration and flux were significantly lower than in WYs 2 to 5 when compared to WY 1. Also, TKN concentration and flux in WY 5 were significantly lower than in all other study years (Table 12).

Soils and tile drains

Soil analysis from Graywood Gully indicated that major decreases in available NO_3 and SRP were observed in some fields during this study. In particular, soil NO_3 levels decreased by 80% to 93% within 2 years in fields H-7A through H-7E, only by 17% in field H-6 in 2 years, but by 84% in 5 years (Fig. 7a). Other fields (H8–10) did not show such dramatic drops. Small declines in SRP were observed in fields H-7A through H-7E (mean = 52%), no decreases in H-8 to H-10, and major drops in fields H-5 and H-6 (72 to 96%) (Fig. 7b). Soluble reactive phosphorus concentrations of groundwater draining field tiles in managed areas were lower (mean ± S.E.: 5.81 ± 3.4 μg P/L) than those draining unmanaged areas (219 ± 162 μg P/L, Fig. 8).

Discussion

Mitigation of soil and nutrient losses from agricultural land continues to be a concern within watersheds of the United States and indeed worldwide. There are a number of reasons for this concern. First, depletion of agricultural soil is counterproductive to good farming practices and crop productivity. And perhaps more importantly, over fertilization and concomitant nutrient loss to downstream aquatic ecosystems may produce undesirable effects including increased numbers of bacteria, algae, and macrophytes (Somarelli et al., 2007; Makarewicz et al., 2007; D'Aiuto et al., 2006; Jamieson et al., 2003; Inamdar et al., 2002), increased siltation, decreased aesthetics — in general, a deterioration in both surface and groundwater quality downstream (Gallichand et al., 1998) resulting in cultural eutrophication of lakes and streams (McDowell et al., 2004, Carpenter et al., 1998). In large lakes (e.g., Great Lakes), cultural eutrophication like this is often manifested in pollution of coastal zones, river mouths, and embayments (Makarewicz and Howell, 2007; Makarewicz, 2000).

Since the 1970s, environmental management of watersheds, particularly those in agricultural settings, has emerged as a promising tool to deal with the water quality problems noted above (Hawkings and Geering 1989; Staver et al., 1989; Whitelaw and Solbe, 1989). Water pollution from intense row crop production is related to erosion of soils, to nutrient fertilization as an insurance to maintain

Table 7
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the North McMillan Creek watershed, Conesus Lake, New York.

North McMillan Creek	Water Years				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	0.31 ± 0.07	0.18 ± 0.04	0.23 ± 0.05	0.27 ± 0.07	0.20 ± 0.05
Total Kjeldahl nitrogen (μg N/L)	331 ± 61	343 ± 71	367 ± 81	410 ± 93	330 ± 72
Total phosphorus (μg P/L)	12.9 ± 3.7	18.2 ± 5.3	14.9 ± 4.7	27.8 ± 9.1	20.7 ± 6.5
Soluble reactive phosphorus (μg P/L)	4.33 ± 0.75	4.11 ± 0.73	4.88 ± 0.92	4.60 ± 0.90	3.69 ± 0.69
Total suspended solids (mg/L)	2.06 ± 1.05	3.41 ± 1.79	2.78 ± 1.57	8.14 ± 4.86	4.02 ± 2.26
Number of chemistry samples (n)	83	82	69	72	70
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	3.21 ± 0.34	2.07 ± 0.21[a]	2.44 ± 0.24[a]	2.74 ± 0.29	2.18 ± 0.23[a]
Total Kjeldahl nitrogen (kg N/d)	3.36 ± 0.27[b]	3.48 ± 0.28[b]	4.29 ± 0.34[b]	3.47 ± 0.29[b]	2.82 ± 0.24[a]
Total phosphorus (kg P/d)	0.12 ± 0.013	0.15 ± 0.017	0.14 ± 0.015	0.16 ± 0.018	0.15 ± 0.018
Soluble reactive phosphorus (kg P/d)	0.048 ± 0.004[b]	0.047 ± 0.004[b]	0.054 ± 0.004[b]	0.043 ± 0.003[b]	0.037 ± 0.003[a]
Total suspended solids (kg/d)	15.00 ± 3.01	26.00 ± 5.31	28.65 ± 5.77	28.62 ± 6.11	21.54 ± 4.56
Number of flux estimates (n)	365	365	365	365	365

Values are the marginal means (± S.E.) of the concentration (mg/L or μg/L) or flux (kg/d) calculated from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included. Non-detectable values for chemical analytes and TSS and days with no flow are not included.

[a] The mean difference is significantly lower at the 0.05 level (Post hoc Bonferroni) from WY 1.

[b] The mean difference among WY 1, WY 2, WY 3 and WY 4 is not significantly different ($P>0.05$).

Table 8
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the Graywood Gully watershed, Conesus Lake, New York.

Graywood Gully	Water Year				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	10.02 ± 2.03	7.77 ± 1.39	5.25 ± 0.97[a]	3.71 ± 0.78[a]	4.20 ± 0.90[a]
Total Kjeldahl nitrogen (μg N/L)	1124 ± 233	873 ± 160	706 ± 133[a]	531 ± 114[a]	514 ± 113[a]
Total phosphorus (μg P/L)	352.1 ± 73.9	343.7 ± 53.5	240.0 ± 46.1	219.8 ± 48.1[a]	186.6 ± 41.8[a]
Soluble reactive phosphorus (μg P/L)	137 ± 28	103 ± 18	109 ± 20	97 ± 20	104 ± 22
Total suspended solids (mg/L)	21.9 ± 10.8	30.3 ± 13.1	15.4 ± 6.9	5.9 ± 3.1[a]	7.7 ± 4.0[a]
Number of chemistry samples (n)	74	93	87	67	62
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	9.32 ± 0.86	6.99 ± 0.56[a]	3.91 ± 0.34[a]	2.68 ± 0.22[a]	2.94 ± 0.24[a]
Total Kjeldahl nitrogen (kg N/d)	0.57 ± 0.05	0.48 ± 0.04[a]	0.42 ± 0.03[a]	0.32 ± 0.02[a]	0.29 ± 0.02[a]
Total phosphorus (kg P/d)	0.19 ± 0.015	0.17 ± 0.013	0.13 ± 0.010[a]	0.12 ± 0.009[a]	0.10 ± 0.007[a]
Soluble reactive phosphorus (kg P/d)	0.097 ± 0.009	0.073 ± 0.006	0.076 ± 0.007	0.061 ± 0.005	0.061 ± 0.005
Total suspended solids (kg/d)	6.92 ± 1.30	10.29 ± 1.75	4.79 ± 0.84	2.04 ± 0.34[a]	3.25 ± 0.54[a]
Number of flux estimates (n)	300	343	340	360	356

Values are the marginal means (± S.E.) of the concentration (mg/L or μg/L) or flux (kg/d) calculated from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included. Non-detectable values for analytes and TSS and days with no flow are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post-hoc Bonferroni) from WY 1.

production yields rather than nutrient deficiencies of the soil, to untimely tillage and fertilization practices, to the lack of buffer or filter strips along water passages, etc. In New York, water pollution from dairy farming is often associated with high P levels in soils receiving long-term manure application, while recently applied manure can produce high concentrations of total dissolved P in overland flow (Kleinman et al., 2002; Hively et al., 2005); that is, “Critical Source Areas” do exist (McDowell et al., 2004). Soil, P, and N losses from a watershed are variable and may be largely limited to small areas that affect downstream systems. For example, intense livestock production in a limited area may have several problems (e.g., lack of appropriate manure storage facilities, no treatment of feedlot runoff, release of untreated milk house wastewater, and excessive or untimely field applications of manure) (Cooper and Lipe, 1992; Clausen et al., 1992). “Critical Source Areas” can be recognized and allow the focusing of management efforts. This can be accomplished through a technical evaluation of soil types, of production rates and P and N contents of manure, of farming practices, of hydrological sensitive areas and slope of the land, etc. This was the approach taken in implementing management practices in the Conesus Lake Study.

In general, nutrient export from cropland to downstream systems is several times higher than from grass and forest (McDowell et al., 2004; Beaulac and Reckhow, 1982; Frink, 1991; Jones et al., 2004). For example, overland flow of TDP from a manured barnyard in New York was 11.6 mg/L compared to ~0.1 mg/L from a deciduous forest (Hively et al., 2005). Recent research has shown that within the Conesus Lake watershed, agriculture dominated sub-watersheds lose larger amounts of soil and nutrients than those with a lesser percentage of land in farming (D'Aiuto et al., 2006; Makarewicz et al., 2007; Herendeen and Glazier, 2009; SOCL, 2001). This was confirmed by our study. For example, at North McMillan Creek where 12% of the watershed is in farmland, marginal mean concentrations of all analytes were low compared to those in Graywood Gully where agriculture dominated land use (>70%) (Tables 7 and 8).

In the Graywood Gully watershed where row crops and dairy farming were present, application of a full spectrum of management practices [fertilizer reduction, cover crops, contour strips, reduction in fall and winter manure spreading, various grass filters for runoff from bunker storage of silage and milk house wastes, cows and heifers fenced from the creek and pond (Herendeen and Glazier, 2009; Jacobs, 2006)] resulted in significant reductions in the flux and concentration ranging from 47 to 65% from WY 1 to WY 5 in four out of five analytes monitored (TP, NO_3+NO_2, TKN, TSS); that is, significantly more soil and nutrients were maintained on the watershed (Table 13). The time from BMP implementation to a significant impact varied with the analyte: NO_3+NO_2 in 1 year, TKN in 2 years, TSS and TP in 3 years. For

Table 9
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for Long Point Gully watershed, Conesus Lake, New York.

Long Point Gully	Water Year				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	5.82 ± 1.33	4.73 ± 1.07[a]	2.83 ± 0.60[a]	2.96 ± 0.70[a]	3.35 ± 0.83[a]
Total Kjeldahl nitrogen (μg N/L)	737 ± 115	632 ± 97	527 ± 76[a]	602 ± 97	558 ± 93
Total phosphorus (μg P/L)	97.7 ± 24.1	82.4 ± 20.1	58.5 ± 13.3[a]	50.0 ± 12.8[a]	62.1 ± 16.5[a]
Soluble reactive phosphorus (μg P/L)	47.7 ± 13.8	26.1 ± 7.5[a]	25.8 ± 6.9[a]	20.7 ± 6.2[a]	21.6 ± 6.7[a]
Total suspended solids (mg/L)	4.8 ± 2.3	9.8 ± 4.7	4.3 ± 1.9	3.3 ± 1.7	3.3 ± 1.7
Number of chemistry samples (n)	52	53	62	48	45
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	19.79 ± 2.08	15.50 ± 1.47[a]	8.31 ± 0.76[a]	8.84 ± 0.81[a]	9.81 ± 0.98[a]
Total Kjeldahl nitrogen (kg N/d)	1.95 ± 0.13	1.69 ± 0.10	1.55 ± 0.09[a]	1.80 ± 0.11	1.63 ± 0.10[a]
Total phosphorus (kg P/d)	0.25 ± 0.03	0.20 ± 0.02[a]	0.17 ± 0.01[a]	0.14 ± 0.01[a]	0.16 ± 0.02[a]
Soluble reactive phosphorus (kg P/d)	0.131 ± 0.017	0.077 ± 0.009[a]	0.078 ± 0.009[a]	0.057 ± 0.006[a]	0.058 ± 0.007[a]
Total suspended solids (kg/d)	6.89 ± 1.34	19.71 ± 3.50	12.57 ± 2.17	9.43 ± 1.66	7.67 ± 1.45
Number of flux estimates (n)	246	302	320	310	269

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post hoc Bonferroni) from WY 1.

Table 10
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the Cottonwood Gully watershed, Conesus Lake, New York.

Cottonwood Gully	Water Years				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	3.70 ± 0.63	3.01 ± 0.49	2.16 ± 0.38[a]	2.51 ± 0.44[a]	2.51 ± 0.44[a]
Total Kjeldahl nitrogen (μg N/L)	687 ± 114	699 ± 112	587 ± 101	556 ± 95	469 ± 81[a]
Total phosphorus (μg P/L)	90.2 ± 16.5	104.0 ± 18.5	94.8 ± 18.2	79.7 ± 15.0	79.8 ± 15.3
Soluble reactive phosphorus (μg P/L)	34.5 ± 5.5	25.2 ± 3.9	41.0 ± 6.8	27.7 ± 4.5	35.8 ± 6.0
Total suspended solids (mg/L)	9.5 ± 4.7	12.2 ± 5.9	8.1 ± 4.3	3.0 ± 1.5[a]	2.8 ± 1.5[a]
Number of chemistry samples (n)	79	80	69	71	70
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	4.44 ± 0.34	3.48 ± 0.26[a]	2.25 ± 0.17[a]	2.81 ± 0.21	2.56 ± 0.19
Total Kjeldahl nitrogen (kg N/d)	0.61 ± 0.04	0.55 ± 0.04	0.58 ± 0.04	0.50 ± 0.03[a]	0.45 ± 0.03
Total phosphorus (kg P/d)	0.08 ± 0.005	0.08 ± 0.005	0.09 ± 0.006	0.07 ± 0.004	0.08 ± 0.005
Soluble reactive phosphorus (kg P/d)	0.039 ± 0.002	0.037 ± 0.002	0.046 ± 0.003	0.035 ± 0.002	0.045 ± 0.003
Total suspended solids (kg/d)	4.64 ± 0.92	5.03 ± 0.96	5.72 ± 1.09	1.53 ± 0.29[a]	1.78 ± 0.34[a]
Number of flux estimates (n)	365	366	365	358	365

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post-hoc Bonferroni) from WY 1.

SRP, however, there was no impact even after 4 years. This variability was due, in part, to the differing times at which BMPs were implemented during the study.

Although reductions of 24% in stream SRP concentration (137 ± 28 to 104 ± 22 μg P/L) were observed from WY 1 to WY 5, the difference was not statistically significant (Table 13). Yet soil analysis indicated that available SRP in soils decreased in ~4 years, especially in areas where rates of P fertilization were decreased (Fig. 7b, Herendeen and Glazier, 2009). Similarly, SRP concentrations in groundwater from tiles draining managed Graywood Gully fields had lower concentrations of SRP than in adjacent non-managed fields (Fig. 8). Using a completely different analytical approach, the Thornthwaite–Mather soil moisture status model, Zollweg and Makarewicz (2009) demonstrated a 55% reduction in SRP concentration from events in the Graywood watershed.

The lack of statistical significance in the reduction of SRP is related to at least two factors: unexpected application of manure to fields and external inputs from outside the topographic watershed boundary. Lewis and Makarewicz (2009) demonstrated that in the winter of WY 3, unexpected manure operations caused significantly elevated levels of P in stream water draining Graywood — likely affecting our annual analysis. Similarly, Noll et al. (2009) demonstrated that during storm events, flow high in P from outside of the topographic watershed boundary of Graywood Gully affected our estimates of stream P, perhaps prohibiting a statistical significant decrease in our annual calculation. Work in other small watersheds including this study (see Long Point Gully) has demonstrated that significant reductions in SRP loss are possible from the Conesus Lake watershed. Also, Bishop et al. (2005) reported reductions in TP and SRP fluxes of 43 and 29%, respectively, from dairy farms in the watershed of the Cannonsville Reservoir in New York State, while BMPs were not effective in reducing SRP in three watersheds in Owl Run in Virginia (Brannan et al., 2000).

Dairy cattle were removed from the Long Point Gully watershed in WY 1, and a 37% reduction (76.7 ha) in crop acreage occurred by WY 2. Here major reductions in $NO_3 + NO_2$ (42%), TP (36%), and SRP (53%) concentrations were observed within a year (WY 2) of removal of cropland from production (Table 13, Fig. 5). As expected, removing land from crop production reduced nonpoint nutrient sources and led to major reductions of nutrients from the watershed.

In Cottonwood Gully where row crops predominate, BMPs were limited to two: construction of three water and sediment control

Table 11
Descriptive statistical data collected from 1 Sep to 31 Aug 2007 for the Sand Point Gully watershed, Conesus Lake, New York.

Sand Point Gully	Water Years				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	2.09 ± 0.44	1.53 ± 0.32[a,b]	1.21 ± 0.24[a,b]	1.25 ± 0.28[a,b]	1.18 ± 0.26[a,b]
Total Kjeldahl nitrogen (μg N/L)	718 ± 120	765 ± 126	545 ± 86	640 ± 115	631 ± 114
Total phosphorus (μg P/L)	79.9 ± 19.9	103.1 ± 25.5	75.7 ± 17.8	94.0 ± 25.2	95.3 ± 25.6
Soluble reactive phosphorus (μg P/L)	28.07 ± 5.8	28.0 ± 5.6	33.8 ± 6.5	25.3 ± 5.5	25.7 ± 5.6
Total suspended solids (mg/L)	8.8 ± 4.1	14.7 ± 6.7	6.6 ± 2.9	8.6 ± 4.3	7.9 ± 4.0
Number of chemistry samples (n)	76	77	90	66	66
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	2.03 ± 0.19	1.53 ± 0.13[a]	0.99 ± 0.09[a,c]	1.31 ± 0.11[a,c]	1.15 ± 0.10[a,c]
Total Kjeldahl nitrogen (kg N/d)	0.66 ± 0.04	0.54 ± 0.03[a]	0.45 ± 0.03[a,c]	0.51 ± 0.03[a,c]	0.46 ± 0.03[a,c]
Total phosphorus (kg P/d)	0.07 ± 0.006	0.07 ± 0.006	0.06 ± 0.005	0.06 ± 0.005	0.06 ± 0.005
Soluble reactive phosphorus (kg P/d)	0.026 ± 0.002	0.026 ± 0.002	0.035 ± 0.003	0.023 ± 0.002	0.024 ± 0.002
Total suspended solids (kg/d)	6.63 ± 1.19	6.39 ± 1.06	3.11 ± 0.54[a,c]	3.58 ± 0.60[a,c]	2.64 ± 0.44[a,c]
Number of flux estimates (n)	328	365	365	365	365

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post hoc Bonferroni) from WY 1.
[b] The mean difference among years WY 2, WY 3, WY 4 and WY 5 is not significantly different (P>0.05).
[c] The mean difference among years WY 3, WY 4 and WY 5 is not significantly different (P>0.05).

Table 12
Descriptive statistical data collected from 1 Sep 2002 to 31 Aug 2007 for the Sutton Point Gully watershed, Conesus Lake, New York.

Sutton Point Gully	Water Year				
	1	2	3	4	5
Concentration (mean ± 95% C.I.)					
Nitrate (mg N/L)	2.8 ± 0.5	1.9 ± 0.3[a]	1.3 ± 0.2[a]	1.3 ± 0.2[a]	1.7 ± 0.3[a]
Total Kjeldahl nitrogen (μg N/L)	428 ± 84	371 ± 61	381 ± 64	332 ± 59	286 ± 52[a]
Total phosphorus (μg P/L)	39.2 ± 7.6	40.2 ± 6.6	45.5 ± 7.5	38.4 ± 6.8	39.7 ± 7.1
Soluble reactive phosphorus (μg P/L)	20.2 ± 3.7	21.0 ± 3.2	23.8 ± 3.7	21.5 ± 3.6	22.6 ± 3.8
Total suspended solids (mg/L)	5.0 ± 2.3	3.0 ± 1.2	2.9 ± 1.2	1.8 ± 0.8[a]	1.4 ± 0.6[a]
Number of chemistry samples (n)	50	63	62	54	53
Flux (mean ± 95% C.I.)					
Nitrate (kg N/d)	1.43 ± 0.12	1.12 ± 0.08[a]	0.70 ± 0.05[a]	0.75 ± 0.05[a]	0.97 ± 0.07[a]
Total Kjeldahl nitrogen (kg N/d)	0.20 ± 0.02	0.20 ± 0.01	0.20 ± 0.01	0.18 ± 0.01	0.15 ± 0.01[a]
Total phosphorus (kg P/d)	0.018 ± 0.001	0.022 ± 0.001	0.025 ± 0.002	0.021 ± 0.001	0.021 + 0.001
Soluble reactive phosphorus (kg P/d)	0.011 ± 0.001	0.012 ± 0.001	0.014 ± 0.0001	0.012 ± 0.001	0.013 ± 0.001
Total suspended solids (kg/d)	1.75 ± 0.20	1.45 ± 0.21	1.70 ± 0.25	0.94 ± 0.14[a]	0.61 ± 0.09[a]
Number of flux estimates (n)	299	366	337	357	332

Marginal mean values are the concentration (mg/L) or flux (kg/d) adjusted for each sampling year from the Analysis of Covariance (ANCOVA) of concentration versus discharge or flux versus discharge with the covariate discharge. Values of zero discharge are not included.
[a] The mean difference is significantly lower at the 0.05 level (Post-hoc Bonferroni) from WY 1.

basins (gully plugs) and strip cropping designed to retain soils. Previous to BMP introduction in this small watershed (98.8 ha), soil loss was high and conservatively estimated in the 1990s at 130 tons (metric) per year (Makarewicz et al., 2001). As in Graywood Gully, significant impacts from management practices were observed in the second year (WY 3) after introduction of BMPs (WY 2). Unlike Graywood Gully, retention of soil and nutrients was recorded for only three of five analytes (TKN, TSS and NO_3+NO_2). With the exception of TSS (71% reduction), the magnitude of reduction was low relative to Graywood Gully [e.g., NO_3+NO_2 concentration: 32% (Cottonwood) versus 58% (Graywood)]. With regard to TKN, concentration reduction was not significant until WY 5 (4 years post BMP) when a 28% reduction of acreage in crop production occurred. While construction of gully plugs undoubtedly reduced soil loss, conversion of portions (up to 80% the watershed) to a long-term vegetative type crop (alfalfa–grass hay) likely contributed to a reduction in stream NO_3+NO_2 levels. Lysimeter studies indicated that NO_3+NO_2 concentrations were much less in groundwater from fields cropped to a forage legume, such as alfalfa, than from fields with corn (Owens 1990). Also, N fertilizers are not generally used for alfalfa. Therefore, the reduction in N fertilizer use and establishment of alfalfa likely contributed to the reduction in stream NO_3+NO_2 observed.

Sheffield et al. (1997) and Owens et al. (1996) observed major reductions in total fractions (TSS: 90%, total N: 54%, TP: 81%) but not in dissolved fractions when water troughs and cattle were fenced out of streams. At Sand Point Gully rotational grazing pens and water troughs were installed, and cattle were fenced out of the creek starting in May of WY 2. Two gully plugs and tiles were also installed in a small portion of the watershed in November 2002 prior to the beginning of this project. We did not expect a large impact of management practices here, especially since the major management area, rotational grazing and the "gully plugs" accounted for less than 9.5% of the entire watershed. Also, manure-spreading operations continued in large portions of the watershed throughout the study (P. Kanouse, Personal Communication, Livingston County Soil and Water Conservation District), which theoretically could cause elevated levels of NO_3+NO_2 and TP. In spite of these expectations, a significant 44% reduction in NO_3+NO_2 concentration was observed (Table 13) by WY 3 with no further significant changes over the study period. A reduction in other analytes was not observed.

The reduction in Sand Point Gully water concentrations of NO_3+NO_2 was unexpected because no structural management practices were implemented that would account for the decrease in NO_3+NO_2. If anything, the continued manure spreading would suggest the potential for elevated levels of NO_3+NO_2. Since only NO_3+NO_2 decreased and not TP, SRP, TKN or TSS, cropland rotation was considered a possible cause. Fallow land, wheat, and the alfalfa grass mix were converted to soybean production acreage starting in WY 1 at Sand Point Gully. Production of soybeans increased from zero, 1 year prior to this study beginning, to 33.0 ha in WY 2 to 83.7 ha by WY 5, almost 45% of the watershed. However, soybeans are not suited to high uptake of NO_3 (K. Czymmek, Personal Communication, Cornell University), and a fairly strong negative correlation existed between acres of soybeans planted and NO_3+NO_2 ($r^2=0.55$). Thus soybean uptake of NO_3 should be fairly low and not the likely cause of the reduction observed in stream water.

However, the plow down of sod (spring of WY 1) would be expected to release a significant amount of NO_3. Estimates of 224 to 336 kg of N/ha per acre are typically lost from sod plow down in New York State (K. Czymmek, Personal Communication, Cornell University). Also, wheat planted in fall of WY 1 would have been fertilized in spring of WY 1, and this fertilizer could be a source of NO_3 to groundwater resulting in the elevated levels observed in WY 1. What may have happened is that WY 1 stream NO_3+NO_2 concentrations were elevated due to a practice that occurred only in the first year of our study. After this ended, a slow but steady decrease in NO_3+NO_2

Table 13
Percent change in marginal mean concentration from Water Years 1 to 5.

Watershed	Management practice implemented	NO_3	TKN	TP	SRP	TSS
North McMillan	No management practices	−35	0	+60	−15	+95
Sutton Point	60% Conversion of cropland to alfalfa	−39*	−33*	+1	+12	−72*
Graywood	Whole farm planning (myriad of BMPs)	−58*	−54*	−47*	−24	−65*
Cottonwood	Gully plugs, 28% reduction in croplands	−32[a]*	−32*	−12	+4	−71*
Sand Point	9.5% Converted to rotational grazing, Fenced cattle from streams	−44*	−12	+19	−8	−10
Long Point	37% Reduction in croplands, removal of cows	−42*	−24*	−36*	−53*	−31

A star (*) indicates a significant decrease.
[a] Significant decrease from Water Years 2 to 5.

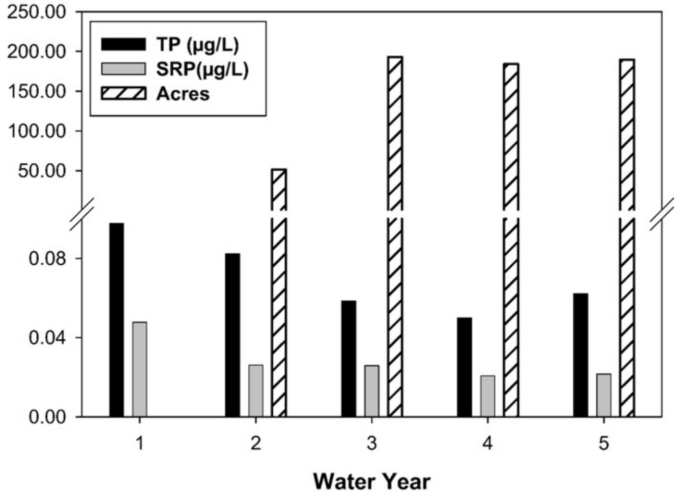

Fig. 5. Time trend in soluble reactive phosphorus (SRP), total phosphorus (TP) and acres of land taken out of crop production in the Long Point Gully watershed. 1 acre = 0.405 ha.

occurred with time. Since no other analyte (TP, SRP, TKN, and TSS) was affected, the response of NO_3+NO_2 was likely due to crop land rotation — a cultural management practice. Typically, when fallow or alfalfa fields are plowed and moved into production, corn is the recommended rotation crop to utilize NO_3 lost to groundwater as a result of plowing.

Significant reductions in TKN and TSS flux, but not concentration, occurred by the second year and third year of the study in the Sand Point Gully watershed. Since there was general agreement in statistical significance between data that considered changes in stream concentration and stream flux in the other five watersheds examined, we were surprised at this difference. Other researchers have discussed such inconsistencies and have noted that expectations of results can be confounded by catastrophic loading events (Meals 2001), unexpected and unknown changes in cropping practices (Boesch et al., 2001), and natural interannual variability (Longabucco and Rafferty, 1998). Any of these factors could explain the differences observed.

Significant reductions in NO_3+NO_2 (39%), TSS (72%), and TKN (33%) occurred at Sutton Point (Table 13) within 1, 3, and 4 years, respectively, after WY 1. No physical infrastructure improvements were implemented in this watershed until WY 5 when gully plugs were added. However, a significant and increasing portion of the watershed has been in alfalfa/grass production since WY 1 (37% in WY 2 to 60.3% in WY 5). As in Cottonwood Gully, the conversion of portions of this watershed to a long-term vegetative type crop (alfalfa–grass hay), a cultural BMP, would indicate that no N fertilizer was added to these fields (N. Herendeen, Personal Communication, Cornell Cooperative Extension). Also during this period, manure slurry was not added to fields (P. Kanouse, Personal Communication,

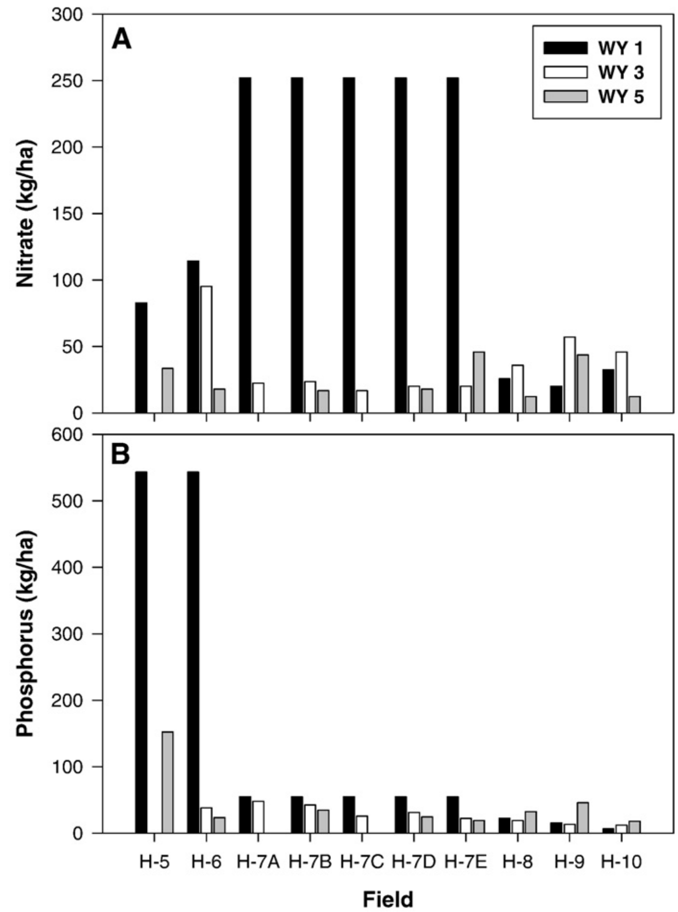

Fig. 7. (a) Nitrate and (b) phosphorus in soils from Graywood Gully. Field names refer to locations within the Graywood Gully watershed in Fig. 2 of Herendeen and Glazier (2009).

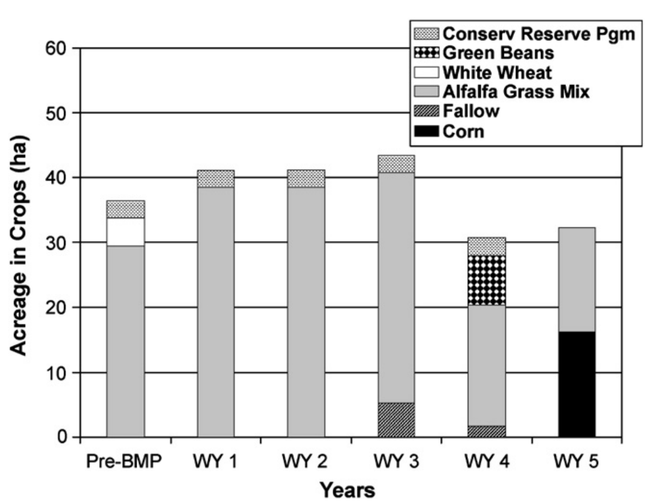

Fig. 6. Acreage of various crops in the Cottonwood Gully watershed over time.

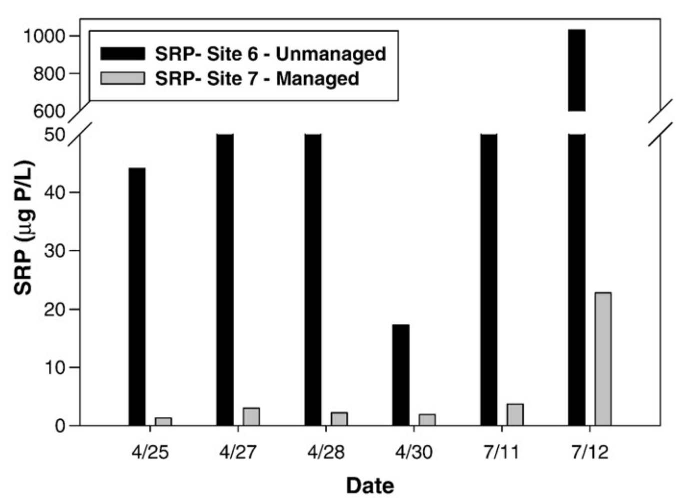

Fig. 8. Concentration of soluble reactive phosphorus in water draining from tiles at Sites 6 and 7 in and near the Graywood Gully watershed (see Fig. 2 of Herendeen and Glazier 2009).

Table 14
Annual average nonevent and event (in parentheses) concentrations of nitrate and total suspended solids.

	North McMillan	Graywood Gully	Cottonwood Gully	Long Point	Sutton Point	Sand Point
Nitrate (mg/L)						
WY 1	0.38 (0.62)	15.53 (11.34)	4.43 (5.26)	8.01 (6.75)	2.83 (3.98)	3.02 (4.56)
WY 2	0.26 (0.29)	11.83 (9.61)	4.71 (3.68)	5.19 (6.02)	2.31 (2.05)	1.87 (1.96)
WY 3	0.27 (0.40)	8.41 (6.47)	3.34 (3.32)	4.28 (4.46)	1.68 (1.99)	1.45 (3.31)
WY 4	0.25 (0.43)	4.53 (4.92)	3.05 (2.68)	3.47 (3.68)	1.47 (1.96)	1.37 (3.02)
WY 5	0.26 (0.30)	4.72 (5.60)	3.17 (3.91)	3.72 (4.50)	1.87 (2.04)	1.31 (1.33)
Total suspended solids (mg/L)						
WY 1	2.9 (267)	16.7 (282)	9.7 (183)	2.9 (54)	7.4 (154)	6.7 (247)
WY 2	11.5 (268)	11.6 (1,110)	7.8 (297)	11.5 (49)	4.4 (49)	6.5 (369)
WY 3	7.9 (113)	6.7 (499)	5.7 (257)	7.9 (26)	4.1 (26)	5.1 (154)
WY 4	3.8 (123)	3.4 (289)	1.5 (125)	3.8 (23)	2.0 (23)	7.1 (161)
WY 5	2.6 (265)	5.3 (318)	2.0 (200)	2.6 (24)	2.7 (25)	2.1 (511)

WY = Water Year.

Livingston County Soil and Water District). Both practices, reduction in manure spreading and the establishment of increasing acreage of a vegetative crop, likely led to the observed decrease in NO_3+NO_2 and TKN to the downstream system.

Widely variable lengths in response times to BMPs are reported in the literature. Generally, smaller watersheds show water quality improvements in less time than larger ones (Gallichand et al., 1998). In the St. Albans watershed (1384 ha), Vermont, USA, no significant change in concentration or flux was observed in tributary streams 10 years after implementation of BMPs (Clausen et al. 1992). Coffey et al. (1992) suggested response times of 6 to 15 years depending on the size of the catchment, but quicker responses to BMPs have been reported. In the Belair River watershed of Quebec Province, Canada (529.4 ha), for example, where river pollution was due to intense livestock production, reductions in TP and SRP were observed 2 years post BMPs (Gallichand et al., 1998). Within the Conesus Lake watershed, significant changes were generally observed within 1 to 2 years after BMP implementation and were still observable 3 to 4 years later. Since data were evaluated on an annual basis, responses in less than 1 year were not observed. However, one exception was the Graywood Gully watershed where positive effects were evident a few weeks after cessation of certain manure practices (Lewis and Makarewicz, 2009).

Within the Conesus Lake study, both event and nonevent decreases were generally observed after BMPs were introduced. For example, both event and nonevent NO_3+NO_2 concentrations decreased as a result of management practices in Cottonwood and Graywood Gullies (Figs. 2a and 4b). In event and nonevents, dissolved fractions such as NO_3+NO_2 and SRP decreased in Cottonwood, Graywood, Sutton, Sand, and Long Point Gullies. At best, a weak trend was observed for total fractions. For example, during nonevents TSS decreased in both Long Point and Cottonwood Gullies while no trend was observed during events. This was somewhat surprising especially at Cottonwood Gully and suggested that gully plugs were more effective during small rain events than large ones. A similar result was observed at Garfoot and Brewery Creeks in Wisconsin (Graczyk et al. 2003). Here, after providing stream bank protection and fencing cattle from streams, a significant decrease in suspended solids occurred during base-flows (i.e., nonevents) but not during events. In the Belair River watershed, Gallichand et al. (1998) observed a 10-fold decrease in TP concentration during peak flow snowmelt runoff in March and April but no significant changes in median annual concentrations after implementation of BMPs.

As noted above, interpretation of field results may be confounded by catastrophic loading events. Nevertheless, by using a small watershed approach, experimental signal to noise (other stresses on the watershed) ratio is likely reduced (Makarewicz, 2009), thereby facilitating evaluation of BMPs on downstream water quality. In general, where implementation impacted downstream water quality, significant reductions in TP, SRP, NO_3+NO_2, TKN and TSS concentration and flux occurred by the second year and third year of implementation. The implementation of structural (e.g., gully plugs) and cultural (e.g., modification of manure practices) BMPs had significant effects in preventing soil loss. In Graywood Gully, where Whole Farm Planning was practiced and a myriad of structural and cultural BMPs were introduced on a dairy farm, the greatest percent reduction (average = 55.8%, range 47% to 65%) and the largest number of significant reductions in analytes (4 out of 5) were observed. At Long Point Gully where 37% of the cropland was taken out of production and dairy cows were removed, concentration reductions were significant, averaging 39% in 4 out of 5 analytes. One of the largest impacts was a 71% reduction in soil loss to downstream systems that resulted from the implementation of both a structural (installation of three gully plugs) and a cultural [long-term vegetative type (crop, alfalfa–grass hay) as associated buffers and strip crops] management practice in the Cottonwood watershed. Removal of nutrient sources also had a major impact on losses from these watersheds.

The importance of proper rotation of crops, a cultural BMP, is also evident. Maintenance of cover crops, such as alfalfa grass mix, can also lead to major reductions in nutrient loss from these managed systems. At Sutton Point no physical infrastructure improvements were implemented. However, the progressive conversion of large portions of cropland in this watershed to a long-term vegetative type crop (60.3% as alfalfa–grass hay by WY 5) was related to major reductions in TSS (72%), TKN (33%), and NO_3+NO_2 (39%) concentrations. Similarly, the proper choice of a production crop after long-term maintenance of a vegetative-type plant is important. For example,

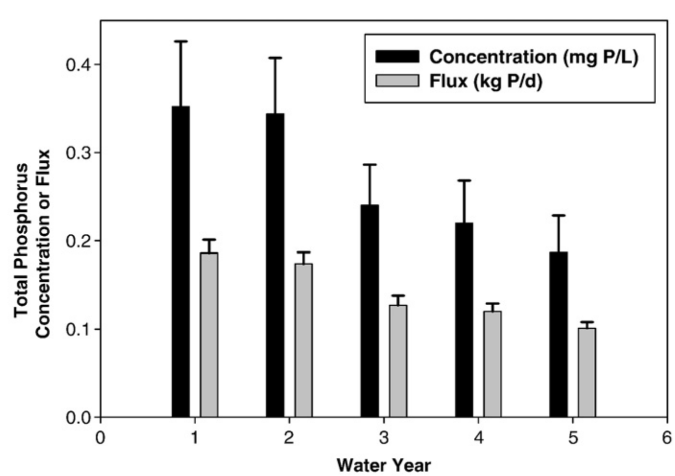

Fig. 9. Mean annual concentration and flux of total phosphorus in Graywood Gully. A Water Year (WY) is defined as the period from 1 Sep to 31 Aug of the following year. Values are the marginal mean ± S.E.

after plow down of sod/alfalfa in Sand Point Gully, a crop such as corn, which readily takes up NO_3, would have been a better choice to rotate over alfalfa and sod and reduce losses to downstream systems. Without this rotation, NO_3+NO_2 levels appeared to be elevated in the first year of this study at Sand Point Gully. Where rotational grazing pens and fencing out of cattle from streams were implemented (Sand Point Gully), no significant effect on downstream systems occurred, as only a small amount of acreage was affected by the BMP.

Both structural and cultural BMPs were observed to have profound effects on nutrient and soil loss. Where fields were left fallow or planted in a vegetative type crop (alfalfa), reductions especially in NO_3+NO_2 were observed. Where structural implementation occurred, reductions in total fractions, most likely particulate fractions, were particularly evident. Where both were applied, major reductions in nutrients and soil occurred. Taking significant portions of the watersheds out of crop production or by removing dairy cows had a similar effect; nutrients and soil were maintained on the watershed and significant reductions in nutrient and soil loads and concentrations to downstream systems were evident. In fact, significant decreases of TP and SRP concentrations occurred only in the two watersheds where considerable effort went into managing manure (Graywood Gully) and where dairy cows were removed (Long Point Gully) (Table 13).

Although significant reductions in nutrient and soil loss to streams were observed in the managed watersheds, the question does arise as to how much more soil and nutrients can be maintained on the watershed and not be lost to the downstream system. In order to evaluate the effectiveness of BMP implementation, reductions in nutrient and soil loss in managed sites in the watershed were compared to the relatively 'pristine' North McMillan Creek watershed. This watershed was deemed pristine because with only 12% of the land in agriculture, nutrients and soil concentrations in stream water draining the watershed were low (Table 14). After 5 years of management, nonevent and event concentrations of TSS in WY 5 in streams draining watersheds dominated by agriculture were not significantly different (ANOVA, $df=6{,}100$; $P>0.05$) from the "pristine" North McMillan Creek (Table 14). This result suggests that management for soil loss can be very effective in a relatively short period of time and that the reductions are comparable to our "pristine/reference" watershed of North McMillan Creek.

This was not the case for nutrients. Although significant reductions in nutrient levels in managed agricultural systems were noted 5 years post BMP implementation, event and nonevent nutrient concentrations in agricultural watersheds were still significantly different (ANOVA, $df=1{,}6{,}100$; $P<0.05$) than those in North McMillan Creek (Table 14). Post hoc analysis (Bonferroni) indicated that during events all watersheds, except Sand Point Gully, were significantly higher in NO_3+NO_2 than North McMillan. Hysteresis is the concept that there is a lag effect between an action and an effect. We are likely observing such an effect with nutrients. Annual marginal mean concentrations in streams still appear to be decreasing 5 years after implementation of management practices (e.g., Fig. 9). Just how long it could take before a new equilibrium would be reached between groundwater, soil, and stream chemistry is not known or suggested by our data. Clearly, it would take longer than 5 years.

Interestingly, reductions in nutrients and soil delivered to downstream systems have had an effect on metaphyton, macrophytes, and microbial communities in the nearshore area of Conesus Lake. Comparisons of Pre-BMP (2–3 years) to the Post-BMP (4 years) periods at Cottonwood Gully, Graywood Gully, and Sand Point Gully (sites receiving the most extensive BMPs) revealed that algal cover was statistically lower than baseline in 8 of 11 sample years (72.7%) (Bosch et al., 2009a). In sites downstream from sub-watersheds that were not extensively managed, percent cover of filamentous algae was lower than Pre-BMP levels in only 3 of 12 sample years (25%). Where major reductions in percent cover occurred, strong positive relationships existed with summer flux of NO_3+NO_2 and SRP to the nearshore of Conesus Lake; that is, reductions in N and P resulted in reductions of metaphyton populations.

Similarly, in macrophyte beds downstream from managed sub-watersheds, quadrat biomass decreased by 30–50% within 1 or 2 years of BMPs implementation and was statistically lower than Pre-BMP values in 7 of 11 sample years (Bosch et al., 2009b). In the three macrophyte beds where minimal or no BMPs were introduced, biomass was statistically indistinguishable from Pre-BMP values in 12 experimental sample years. Lastly, microbial populations declined in the nearshore below managed watersheds. For example, over the 5-year study period, a major decrease in bacterial levels in nonevent Graywood Gully stream water was evident after management practices were implemented. *Escherichia coli* levels dropped more than 10 fold to levels significantly below the 235 cfu/100 mL EPA bathing beach standard while the yearly maximum for *Enterococcus* dropped by a factor 2.5 (Simon and Makarewicz, 2009a,b).

Clearly, BMPs implemented in the Conesus Lake watersheds have had major impacts within the nearshore of Conesus Lake. The utility and effectiveness of the implemented BMPs should allow regional policy makers and managers to develop strategies for improving watershed land usage while improving downstream water quality in the embayments, nearshore, and open waters of large lakes. But, as Moran and Woods (2009) suggest, effective watershed management requires far more than a narrow focus on water quality. The importance of a strong science foundation coupled with an ongoing effort to build political consensus cannot be overstated. It is essential that watershed residents be engaged in a dialog where they can communicate to politicians how to restore and protect the resources they value.

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Detecting effects of Best Management Practices on rain events generating nonpoint source pollution in agricultural watersheds using a physically-based stratagem

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ABSTRACT

Nonpoint source pollution (NPSP) is the export to receiving waters of nutrients originating from diffuse sources. This research documents a methodology for confirming reductions in NPSP resulting from implementation of agricultural Best Management Practices (BMPs). It employs that methodology to confirm the success of BMPs implemented in Graywood Gully, a study sub-watershed that drains into Conesus Lake, NY. Evaluating the effects of BMPs in agricultural watersheds is often complicated by significant temporal variability in weather and hydrologic conditions. In many cases NPSP demonstrates much greater variability in response to antecedent hydrologic/meteorologic conditions than to commonly implemented BMPs. In essence, weather variability can mask the beneficial effects of the BMPs. By using the Thornthwaite–Mather procedure to model soil moisture status in addition to event rainfall total, it is possible to remove the major sources of weather/hydrologic-related variability, essentially reducing the number of experimental variables to the BMP itself. Application of this method to the Graywood sub-watershed reveals that BMPs can greatly reduce export of NPSP generated pollutants to receiving waters. Estimates of NPSP reductions range from 53% for soluble reactive phosphorus to 89% for nitrate.

Introduction

Agencies that are mandated to protect aquatic environments need to understand the effects of land management programs on these environments. Specifically, the quantification of effects of land management programs on water quality can guide planning of conservation strategies and the distribution of environmental protection funds. Conservation and land management strategies are generally referred to as Best Management Practices (BMPs). The EPA (2004) further defines BMPs as "*Schedules of activities, prohibitions of practices, maintenance procedures, and other management practices to prevent or reduce the discharge of pollutants to waters of the United States. BMPs also include treatment requirements, operating procedures, and practices to control plant site runoff, spillage or leaks, sludge or waste disposal, or drainage from raw material storage*."

Specifically quantifying the actual effectiveness of agricultural BMPs is generally made via water quality monitoring in the streams draining study watersheds (Spooner and Line 1993). The water quality is measured during storm events both before and after BMPs are implemented to determine the effect of the land use changes and manipulations. For a set of BMPs to be considered as having a positive effect on water quality, one must measure less nonpoint source pollution (NPSP) after than before implementation. Although simple in concept, this measurement of effective BMPs can be complicated due to significant variability in weather and hydrologic conditions over the study period. The majority of NPSP is generated by a relatively small number of large and/or intense rainfall events. A year during which, by chance, there are no major storms would *appear* to demonstrate great overall success of management. Conversely, a year during which several very large events occurred would *appear* to be a failure, even if the BMPs actually reduced potential export by a significant percentage. In many cases NPSP demonstrates much greater variability in response to antecedent hydrologic conditions than to commonly implemented BMPs (Zollweg et al. 1995). This variability can mask improvements achieved by the BMPs (Longabucco and Rafferty 1998). Conversely, low rainfall amounts and intensities during a study period could translate to reduced delivery of NPSP and exaggerate the efficacy of BMPs. This situation demands a methodology that removes weather variability and "levels the playing field."

In the humid Northeast, most storm runoff is believed to derive from saturation excess, i.e., via precipitation on, or water flowing out of, saturated areas in the landscape (Meals et al. 2006, Dunne and Black 1970). The most significant aspects of the hydrologic system governing runoff generation and consequent NPSP generation are the antecedent moisture condition, quantified as the amount of moisture in the soil profile at the beginning of a storm event, and the rainfall

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amount (Zollweg et al. 1996). The evaluation methodology used in this paper is based upon these hydrologic principles.

The method is applied to Graywood Gully, a study sub-watershed that drains into Conesus Lake, NY. The overall Conesus Lake Watershed Study (Makarewicz 2009) is intended to evaluate the impact of agricultural BMPs on maintaining nutrients and soil on the landscape and reducing losses of soil and nutrients to downstream aquatic systems. In this volume, Makarewicz et al. (2009) also provide an analysis of the effectiveness of BMPs by using a traditional small watershed approach via water quality measures [flux (kg/ha/d) and concentration (mg/L)] weighted by stream discharge (events and baseflow) and ANCOVA. Since storm-runoff events represent as much as 60 to 80% of the flux of nutrients and sediments to downstream systems in the Conesus Lake watershed (Makarewicz et al. 1999), we also took the second approach detailed in this paper to evaluate BMPs by analyzing storm-runoff events. We show that by removing a major source of hydrologic-related variability we essentially reduce the number of experimental variables to one (the BMPs) and show that it is possible to make valid Pre-BMPs versus Post-BMPs comparisons of NPSP-generating events to quantify the benefits derived from these practices.

Methods

As part of the Conesus Lake Watershed Study (Makarewicz 2009), a number of BMPs were implemented in the sub-watershed drained by Graywood Gully (Herendeen and Glazier 2009). Graywood Gully drains a small sub-watershed in Livingston County, NY, and enters Conesus Lake (see Fig. 1 in Herendeen and Glazier 2009). The size of the Graywood sub-watershed is 38 ha, approximately 70% of which is in active agriculture as a single-operator 100-cow dairy operation (Herendeen and Glazier 2009, Jacobs 2006). The most significant NPSP problem in Graywood is phosphorus and nitrogen export to Conesus Lake. We measured and analyzed soluble reactive phosphorus (SRP), total phosphorus (TP), total Kjeldahl nitrogen (TKN), and nitrate (NO_3) as indicators of NPSP. This small watershed approach is advantageous because it evaluates an area that is large enough to capture landscape transport processes and dilution effects (Gburek et al. 2000), yet small enough to focus on NPSP loading from a single farm and the BMPs adopted at that scale. The major BMPs implemented were the reduction of fertilization rates and the virtual elimination of fall and winter spreading of manure on hydrologically sensitive areas (HSAs) (Herendeen and Glazier 2009). HSAs are those areas subject to disproportionate likelihood of generating surface runoff (Walter et al. 2000). HSAs for Graywood were selected based upon the general principles, such as stream proximity and slope, established in the literature (Walter et al. 2000). Other BMPs implemented for Graywood included management of barnyard runoff,

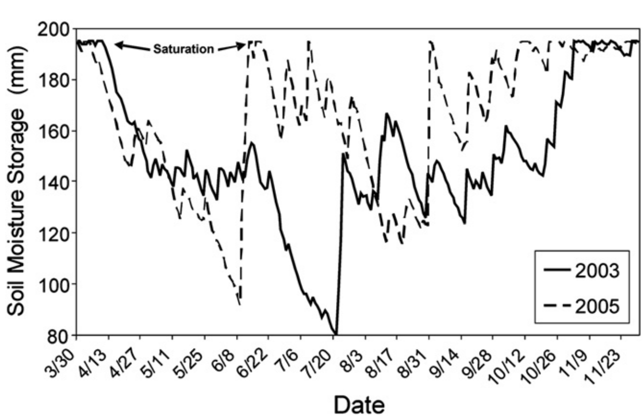

Fig. 1. Modeled soil moisture in Graywood sub-watershed (2003, 2005) showing wide interannual variations between 2003 and 2005.

improved materials handling, planting cover crops, contour strips, various grass filters for runoff from bunker storage of silage and milk house wastes, and fencing streams to isolate animals from the creek and pond. Additional discussion of the BMPs implemented is found in this volume in Herendeen and Glazier (2009).

Implementation of BMPs in this watershed was started in late 2003. It must be noted that implementation of BMPs does not lead to immediate reductions in NPSP. In the case of phosphorus, for example, years may pass before the effects of BMPs become detectable by water quality measurements (Boesch et al. 2001) due to accumulation of phosphorus in soils and stream sediments. Also, practices that involve planning and management procedures take time to become fully and effectively established. For purposes of this study, therefore, the years 2002 and 2003 are defined as the Pre-BMPs period, the year 2004 is a transition period, and the years 2005 and 2006 are defined as Post-BMPs period.

During the entire study period (2002–2006) stream discharge and water quality (SRP, TP, TKN and NO_3) were measured (see Makarewicz et al. 2009). Daily rainfall data used for the soil moisture balance model was collected by means of a standard nonrecording rain gage located within the sub-watershed. A trained weather observer collected data daily. Rainfall data was supplemented by 15-minute interval recording rain gage data from the nearby NWS Cooperative weather station at Dansville, NY.

The stream hydrograph and the rainfall record were examined to identify and isolate storm-runoff events. Storm events are defined as periods of rainfall that led to a discernible response in the hydrograph. Runoff and NPSP amounts were determined by separating the measured amounts from the extended baseline/baseflow amounts that were evident just before the rise of the hydrograph. Rainfall events that produced no identifiable increase in stream flow were not examined in this study. Most of these were very minor storms that probably produced no actual runoff and were not examined with regard to generation of NPSP. In other cases it was impossible to separate the runoff hydrograph from high baseflow hydrographs. This occurred most commonly during snowmelt events or during the interval of very large flows due to sudden release of soil water when the soil frost layer melted in the spring. Also, precipitation events recorded during December through March (winter) were not examined because they mostly were snow events, and the times of actual hydrologic impact (when the snow melts) were delayed and difficult to determine. A total of 94 defined storm events were identified during the study period. Of these, 41 occurred during the Pre-BMPs interval, 20 occurred during the transition interval, and 33 after BMPs were established. Examination of the rainfall record reveals that the three intervals were quite different weather-wise. During the Pre-BMP time frame (September–November 2002 and April–November 2003) there was a total of only 3 storms greater than 35 mm. As the transition phase (April–November 2004) unfolded and positive impacts on NPSP were expected to start, the watershed experienced 7 storms of 35 mm or more. During the final interval (April–November 2005 and April–October 2006) 11 of these larger storms occurred, including the three biggest of the whole study — 85.8 mm, 92.7 mm, and 177.3 mm. The discharge (m^3), TP (kg), NO_3 (kg), TKN (kg), and SRP (kg) loadings recorded at the stream outlet were totaled for each event as well.

The soil antecedent moisture condition was computed using the Thornthwaite–Mather methodology [developed by Thornthwaite and Mather (1955) and further refined and applied by Steenhuis and Van der Molen (1986) and Zollweg (1994)]. This modeling methodology has been applied successfully in numerous studies of soil moisture and its effect on hydrologic response (e.g., Bugmann and Cramer 1998, Feddema 1998, Frankenberger et al. 1999, Varni and Usunoff 1999, Leathers et al. 2000, Boll et al. 2001, and Zollweg et al. 1996). This technique uses the parameter APWL (accumulated potential water loss), a created parameter designed to model the response of soil moisture to evaporative forcing. Soils at field capacity will tend to have actual evapotranspiration (ET) equal to the potential ET, while soils at wilting point will have actual ET much less than potential as a consequence of increased soil moisture tension. APWL is used to parameterize the continuous function relating soil moisture, potential ET, and actual ET. The actual soil moisture (SW) is related to APWL by Eq. (1):

$$SW = AWC \exp\left(\frac{APWL}{AWC}\right) \tag{1}$$

where:

AWC = available water capacity (soil moisture content at field capacity minus soil moisture content at wilting point)
SW = soil moisture
APWL = accumulated potential water loss.

The estimates of field capacity and wilting point used to calculate AWC are based on the soil texture, which in turn is determined from the county soil survey. All variables used in the Thornthwaite–Mather procedure have consistent units of length, in this analysis we use mm. Calculations to determine SW and APWL were performed for a daily time step using daily precipitation (*P*) and potential evapotranspiration (PET) data as follows (from Mehta et al. 2006):

Type of day	SW	APWL
Soil is drying $\Delta P<0$	$= AWC \exp\left(\frac{APWL_t}{AWC}\right)$	$= APWL_{t-1} + \Delta P$
Soil is wetting $\Delta P>0$ but $SW_{t-1}+\Delta P \leq AWC$	$= SW_{t-1} + \Delta P$	$= AWC \ln\left(\frac{SW_{t-1}}{AWC}\right)$
Soil is wetting and above capacity $\Delta P>0$ but $SW_{t-1}+\Delta P > AWC$	$= AWC$	$= 0$

where:

AWC = available water capacity (soil moisture content at field capacity minus soil moisture content at wilting point)
SW = soil moisture
APWL = accumulated potential water loss
ΔP = net precipitation; $P -$ PET
P = precipitation
PET = potential evapotranspiration.

Potential evapotranspiration data were derived from daily pan evaporation data collected at the Vegetable Crops weather station at the New York State Agricultural Experiment Station at Geneva, New York. This is NOAA benchmark weather station #3031840, located at 42°52.6′ N, 77°01.9′ W. For days in which there was missing data, other meteorological data collected at the same site (temperature and solar radiation) was applied to the Hargreaves equation (Hargreaves 1975) to generate an estimate of PET.

Results

Fig. 1 illustrates the daily water content of the soil in the watershed as modeled by the Thornthwaite–Mather methodology for 2003, a Pre-BMPs year, and 2005, a Post-BMPs year. These two years are illustrative of the basic problem this paper's methodology is intended to address — that the weather does not cooperate with one's experimental schedule. Clearly, there are large variations in soil moisture both seasonally and interannually. Note especially the difference in soil moisture between mid-July 2003 and the same time of year in 2005. Around 12 July 2005, with soil moisture near capacity, almost any rainstorm would produce surface runoff and NPSP. Contrast this with 12 July 2003 when a hypothetical storm producing at least 6 cm of precipitation, a relatively rare event, would be required to produce significant surface runoff (Zollweg et al. 1996). Fig. 2 of daily stream discharge further illustrates how the weather varied during the study and indicates the complexity of sorting out the effects of BMPs. In 2002–2003 (Pre-BMPs) there were relatively few significant events and a moderate number of runoff events resulted. In 2004 (transition) with more numerous storm events, the runoff events were larger and more frequent, leading one to wonder if the BMPs were failing or being overwhelmed by hydrology. During 2005–2006 (Post-BMPs), there were even more frequent and intense storms, but they led to only moderate runoff events. This suggests that BMPs, once established, may be functioning. It might suggest (conversely) that there were fortuitous conjunctions of dry antecedent moisture conditions with larger storm events. Clearly, there is a need for a methodology to analyze the measured NPSP, compensating in some way for the many and large differences in antecedent moisture conditions and rainfall that occurred over the 5 years of the study. This methodology is developed below.

Runoff is produced by an interaction of rainfall and the soil. If there is a lot of soil moisture storage available, then rain will tend to enter the soil rather than become runoff. If there is little storage available, then the rain will be forced to travel overland (Zollweg et al. 1996). For each rain event, a single parameter that combines both soil moisture storage available at the start of the event *and* the rainfall amount is the soil water deficit (SWD) after the rainfall event. This is equal to:

$$SWD = AWC - SW - P$$

where:

AWC = available water capacity (soil moisture content at field capacity minus soil moisture content at wilting point)
SW = soil moisture
P = precipitation.

Fig. 3 shows the relationship between SWD and the actual runoff amounts measured. The r^2 value is 0.59, indicating that much of the variability in runoff generated is explained by the soil moisture conditions and rainfall amount. This is especially notable, considering that seasonal variation in vegetative cover and storm rainfall intensity were not considered in this analysis. The importance of soil moisture is demonstrated by the fact that the r^2 value using rainfall alone as a predictor is only 0.38. Additionally, this graph/analysis lumps Pre-BMPs, transition, and Post-BMPs conditions. Separating Pre- and Post-BMP data leads to r^2 values of 0.75 and 0.76, respectively. We can now proceed with some confidence that SWD can be used as a leveling parameter among a mass of storm events with numerous combinations of rainfall and antecedent moisture conditions. Analysis can proceed under the assertion that events with equal SWD have equivalent potential for generating runoff and NPSP.

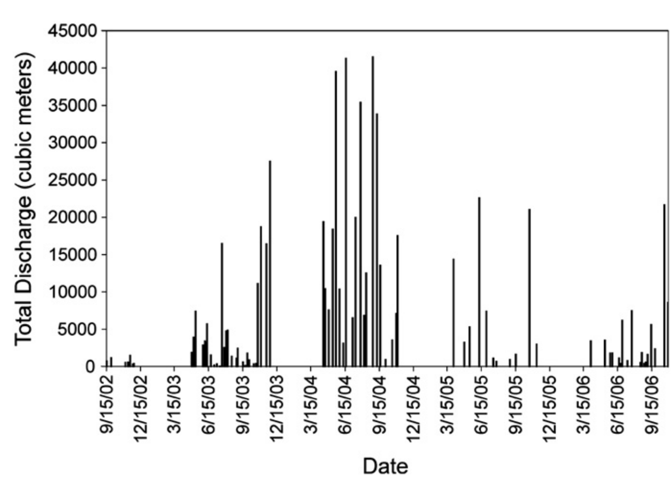

Fig. 2. Daily runoff amounts measured during the study. Although rain events were larger and more frequent after 2004, the stream response was less.

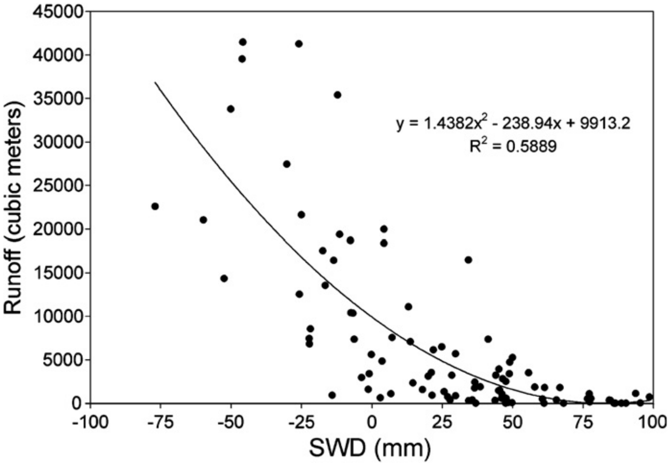

Fig. 3. Runoff amount versus SWD (soil water deficit remaining after event). SWD is an effective modeling parameter for event runoff.

Fig. 4A–D shows the export of nutrients with respect to SWD. In each case, filled dots are used to represent Post-BMPs data; hollow dots represent Pre-BMPs data.

For each data group, regressions can be used to generate a response function that quantifies the relationship between the SWD parameter and the export of NPSP. For each nutrient, the entire data set is modeled by a pair of functions. The first is a zero-value, zero-slope line for the data range where there is essentially no export of NPSP. The second curve is a power function that shows the actual response of NPSP export to storms. Every single storm event is a unique situation possessing a unique combination of initial soil moisture, rainfall amount, land cover, and rainfall intensity. The regression process allows one to generate a single predictor of NPSP (based upon the SWD parameter) for all of the events occurring during each time period (Pre-BMPs and Post-BMPs). In essence, the SWD parameter quantifies how big or impactful any particular storm is. One expects that smaller SWD values represent storms producing larger hydrologic/NPSP responses (Fig. 3). Our assertion is that storms with the same SWD will generate the same amount of runoff or NPSP. If BMPs are successful, the modeled response for a certain SWD should be smaller after implementation than before; that is, the same storm would produce less pollution.

Table 1a
Modeled NPSP responses (kg exported) for an event with SWD = −25.

Parameter	Pre-BMP (kg)	Post-BMP(kg)	Reduction (%)
NO_3^-	429 ($r^2r^2 = 0.75$)	45.9 ($r^2r^2 = 0.83$)	89
TP	41.1 ($r^2r^2 = 0.45$)	4.79 ($r^2r^2 = 0.71$)	88
TKN	83.1 ($r^2r^2 = 0.65$)	12.0 ($r^2r^2 = 0.71$)	86
SRP	3.91 ($r^2r^2 = 0.70$)	1.82 ($r^2r^2 = 0.52$)	53
Number of events	41	33	

The Pre-BMPs period = 2002 and 2003; Post-BMPs period = 2005 and 2006. NO_3 = nitrate, TP = total phosphorus, TKN = total Kjeldahl nitrogen, SRP = soluble reactive phosphorus.

Fig. 4A–D clearly shows that for equivalent storms, NPSP is significantly reduced during the Post-BMPs period compared with the Pre-BMPs period. This improvement can be quantified by using a design/test storm as the input to the regression model. We chose to use the 10th largest storm that occurred during the entire 6-year study. This is an event of a magnitude that one would expect to have

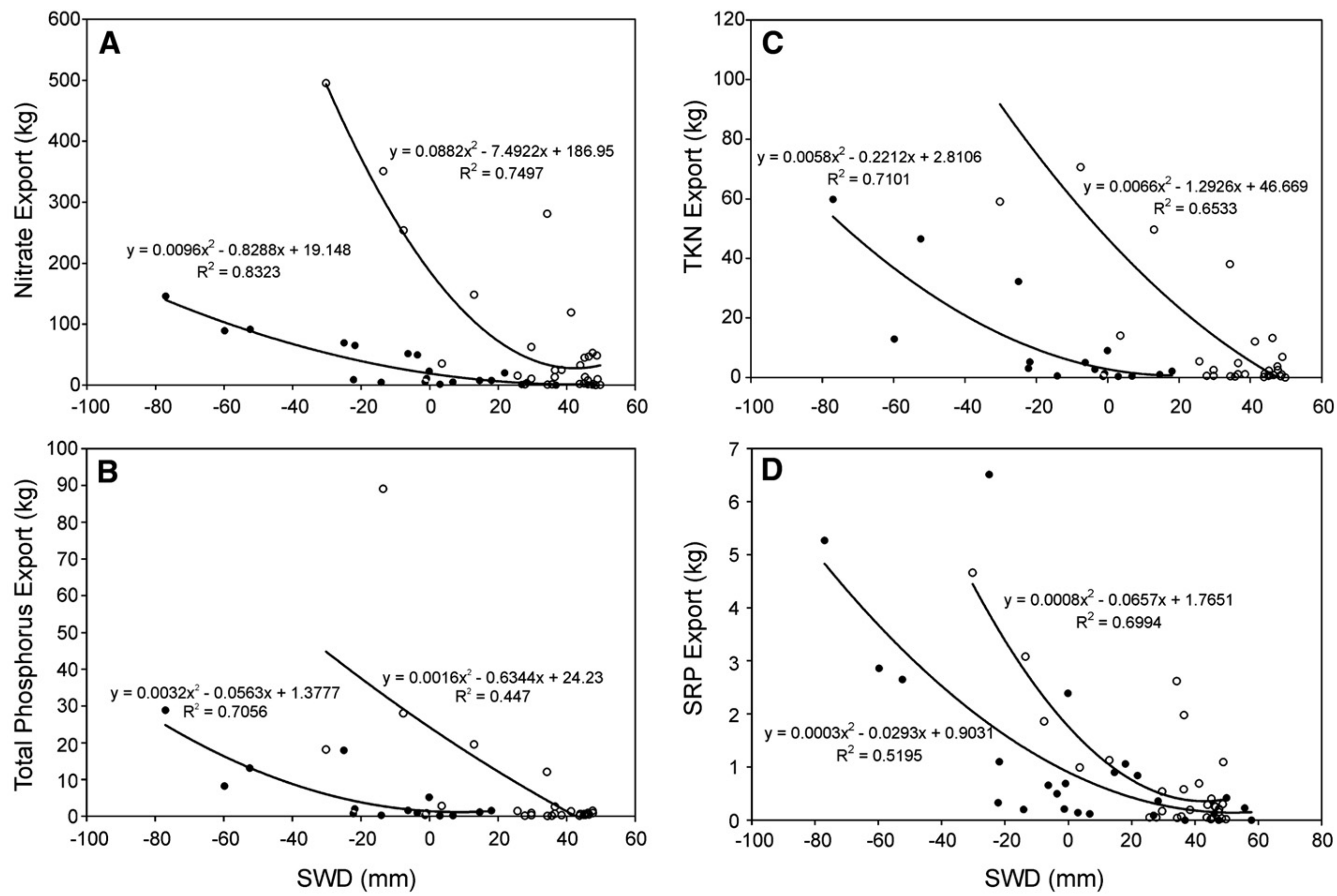

Fig. 4. Observed data for (A) NO_3 export, (B) total phosphorus export, (C) total Kjeldahl nitrogen export, and (D) total soluble reactive phosphorus export, versus SWD (soil water deficit) parameter. Hollow dots represent values in the Pre-BMP period. Filled dots are values in the Post-BMP period.

Table 1b
Modeled NPSP responses (slope of response curve) for an event with SWD = −25.

Parameter	Pre-BMP (kg per unit of SWD)	Post-BMP (kg per unit of SWD)	Reduction
NO_3^-	9.70	1.07	89%
TP	0.675	0.136	80%
TKN	1.46	0.366	75%
SRP	0.0857	0.0368	57%
Number of events	41	33	

The Pre-BMPs period = 2002 and 2003; Post-BMPs period = 2005 and 2006. NO_3 = nitrate, TP = total phosphorus, TKN = total Kjeldahl nitrogen, SRP = soluble reactive phosphorus.

very significant NPSP implications and is the size that BMPs are expected to ameliorate. The tenth largest SWD is −25, and this value can be entered into the regression models to predict the NPSP for that event, for the four nutrients, and for Pre- and Post-BMPs conditions. Results of this modeling exercise are shown in Table 1a. Reductions in nutrients and soil loss from the watershed to the stream range from 53% for SRP to 89% for NO_3. The r^2 value for each curve is included to show that SWD is a good predictor of NPSP response. Table 1b shows the slope of the response curve at the point represented by the test/design storm. Physically, this represents the incremental response of NPSP to additional storm event size. It indicates the expected performance of the BMPs as storms become bigger and bigger. The slope of the NO_3 curve shows an 89% reduction, suggesting that BMPs continue to work well for NO_3 even as storms become bigger. The lower reductions for TP and TKN suggest that the effects of BMPs are strongest at more moderate storms and begin to weaken slightly for really large events.

Discussion

In our study at Graywood Gully, reductions in NPSP occurred within a 5-year period of implementation of BMPs as determined by the event analysis procedure used in this study. The NPSP response indicated a reduction in nutrients as high as 89%, certainly indicating a very significant success of the BMPs, and as low as 53% for SRP. The lower reduction for SRP can be explained by phosphorus dynamics. It is generally acknowledged that phosphorus accumulates in soils and sediments, remaining available to runoff for years, if not decades. Phosphorus export (especially in soluble form) from a watershed in any given year may be the result of farming practices of previous years (Hively et al. 2006). The widely variable lengths in response times or lack of response to management practices reported in the literature are undoubtedly related to length and intensity of previous farming practices, size of the watershed, as well as implementation strategies of BMPs. For example, phosphorus responses to BMPs occurred within 2 years in the Belair River watershed (Gallichand et al. 1998), 6 to 15 years in catchments of various sizes (Coffey et al. 1992), and no change in concentration or flux after 10 years in the St. Albans watershed (Clausen et al. 1992).

Exports of TKN, NO_3 and TP showed large reductions (86–89%) as a result of the BMPs. Although TP showed large estimated reductions, it also had a weaker predictability than NO_3 and TKN. This is also expected because TP, (the bulk of which is sediment borne) is exported only for erosive events that have significant rainfall and runoff energy. The analysis presented does not account for rainfall intensity (impact energy) nor vegetative cover (reduces impact energy and overland flow velocities) so it would not be expected to strongly predict TP loss. Also, as previously discussed, phosphorus accumulates in and only slowly dissipates from soil and stream sediments, so improvement/transition in total phosphorus exports is less clearly defined.

The Graywood Gully sub-watershed study provided an ideal opportunity to evaluate the effectiveness of the Thornthwaite–Mather-based procedure as a tool to evaluate BMPs. The timing and magnitude of BMP impacts on Graywood Gully determined by this procedure are in agreement with empirical monitoring of nutrients and soil loss that utilize ANCOVA techniques with discharge as the covariate to evaluate impacts of BMPs (Makarewicz et al. in press). Confirmation of modeling approaches is important as such approaches provide the opportunity to evaluate implementations of BMPs based on antecedent water levels in soils (e.g., Meals et al. 2006). The observed reductions in NPSP clearly indicate that reasonable, cost-efficient BMPs have the potential to dramatically reduce exports of agricultural pollutants to our aquatic environment. The methodology developed and applied in this paper is clearly effective in analyzing the export of runoff and nonpoint source pollutants with respect to the complete hydrologic setting, i.e., soil moisture conditions at the start of the rainfall event and the total amount of precipitation during the event. This allows the generation of response functions that enable one to detect changes in response due to the implementation of BMPs.

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Winter application of manure on an agricultural watershed and its impact on downstream nutrient fluxes

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ABSTRACT

Whole Farm Planning was instituted and monitored over a 5-year period within the Graywood Gully sub-watershed of Conesus Lake, NY (USA). An array of agricultural Best Management Practices (BMPs) (strip cropping, fertilizer reduction, tiling, manure disposal practices, etc.) were simultaneously introduced to determine the impact of a concentrated management effort on nutrient and soil loss from one watershed within the Conesus Lake catchment. During the study period, significant decreases in winter concentrations of dissolved and particulate fractions, including total phosphorus (TP), soluble reactive phosphorus (SRP), total Kjeldahl nitrogen (TKN), and nitrate (NO_3) but not total suspended solids (TSS), were observed. These decreases may or may not be attributed to cessation of manuring practices. Three years into the study, an opportunity existed to test the responsiveness of the watershed to the curtailment of a single BMP — winter manure application to fields. We field-tested the hypothesis that a change in winter manure applications would impact dissolved and particulate fractions in stream water draining this watershed. We found that the water quality of Graywood Gully is very responsive to winter manure application on environmentally sensitive portions of the sub-watershed. With the short-term resumption of manure application, TP, SRP, TKN, and NO_3 concentrations rose dramatically in stream water; elevated phosphorus concentrations persisted over a 5-week period. Total suspended solids, however, were not elevated after short-term manure application. Factors that affected these results were slope of the land, application of manure over snow and during a snowfall, warm air and soil temperatures, and possibly tile drainage of snowmelt water. Managers of agricultural systems must recognize that phosphorus losses from the watershed during the nongrowing season may detrimentally affect nuisance population of algae in lakes during the summer.

Introduction

Whole Farm Planning is a process that enables a farm to balance farm profitability, community stability, and environmental vitality. It is a decision-making and evaluation model that helps farmers integrate the dynamic relationships of the economic, social, and ecological consequences of management decisions in relation to their contribution to an individual farmer's goals. The management of manure application to agricultural fields is a common practice in North America, Europe, and developing countries and is often an important consideration in any whole farm planning scenario in the United States. Manure is an agricultural by-product that can be environmentally and cost effectively returned to the land to enhance soil productivity, increase soil organic matter, and increase infiltration rates; applied manure can sometimes form a protective layer on the soil surface to reduce runoff (McDowell et al. 2004, Gilley and Risse 2000, Smith et al. 2007). There are also negative aspects to manure application. For example, manure applied in excess can pollute adjacent waterways and infiltrate into groundwater (Zebarth et al. 1996). As a result, agricultural specialists have established recommendations on the timing and field application for manure that often target environmentally sensitive areas and application during the winter season. For example, manure should not be spread on steep slopes nor in areas near water bodies during the fall and winter; manure can be applied in the winter only if adequate storage is not available on the farm; manure application on snow or frozen ground should be avoided; and winter application of manure is the least desirable from both a nutrient utilization and pollution standpoint (Beegle et al. 2000, Maryland Department of Agriculture 2004, Frankenberger et al. 2003).

The Conesus Lake Watershed Study (Makarewicz 2009), a long-term project using the small watershed approach, evaluated the impact of agricultural Best Management Practices (BMPs) on maintaining nutrients and soil on the landscape and on reducing losses of soil and nutrients to downstream aquatic systems. In Graywood Gully, one of several sub-watersheds studied within the Conesus Lake catchment, Noll and Magee (2009) identified not only the importance of critical source areas in the watershed but also the effect of the built

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environment, including drain systems and road ditches, on changing critical hydrological pathways. Also, significant changes in phosphorus (P) fractionation occurred during erosion, transport, and deposition of the particulate or sediment phase (soil) from the watershed to the nearshore sediments. Aluminum and organic matter associated P, for example, dominated in field soils while calcium associated P dominated in nearshore lake sediments (Noll et al. 2009). Zollweg and Makarewicz (2009), using the Thornthwaite-Mather soil moisture status model, demonstrated a reduction in nutrients and soil loss from the watershed to downstream systems as high as 91% and as low as 55% for soluble reactive phosphorus (SRP), indicating a significant success of the BMPs in the Graywood Gully watershed. Empirically, Makarewicz et al. (2009) investigated the effectiveness of agricultural BMPs by evaluating long-term annual changes in seasonal flux (kg/ha/d) and concentration (µg/L) of nutrients and soil in stream discharge water from the managed watershed.

The work by Makarewicz et al. (2009) focused on annual changes and did not consider seasonal aspects of BMPs. Here, we field-tested at the watershed scale the hypothesis that a change in winter manure applications would impact dissolved and particulate fractions in stream water draining the Graywood Gully sub-watershed of Conesus Lake. Evaluation of long- and short-term trends was accomplished by resuming winter manure applications on environmentally sensitive areas of the sub-watershed after 2 years of cessation of fall and winter manure application to frozen, snow-covered fields. After 5 days of manure application, manure spreading on fields in environmentally sensitive areas was once again stopped.

Site description

Graywood Gully, a small New York State sub-watershed (~38 ha) in the northwest portion of the Conesus Lake watershed within the Lake Ontario catchment, drains an upland area dominated by a single 100-cow dairy-farming operation (Jacobs 2005, Fig. 1). Graywood Gully contributes a disproportionate amount of soil and nutrients to Conesus Lake when compared to the other sub-watersheds in the catchment (Makarewicz et al. 2001, 2002; also see Table 1 in Makarewicz 2009). The stream flows eastward through a deeply incised gully (8% grade), often over fractured bedrock, and is surrounded by a small wooded area before reaching the developed margin surrounding Conesus Lake. Soils in the watershed are dominated by the Conesus silt loam, a somewhat poorly drained soil developed on limestone with the water table commonly within 15 cm to 45 cm of the surface, and the Lansing silt loam, a moderately well-drained soil developed on shale with the water table within 45 cm to 60 cm of the surface (USDA NRCS 2008). Further description of Conesus Lake and its watershed, crops planted, tile drains within the watershed, and the implemented BMPs is given in Makarewicz (2009), Noll and Magee (2009), and Herendeen and Glazier (2009). Winters are best described as severe with heavy snows, frozen ground, and long periods of freezing air temperatures interspersed by unpredictable periods of warming and thawing. An overview of the soils, geology, and climate of the Conesus Lake area is given in Forest et al. (1978) and SOCL (2001).

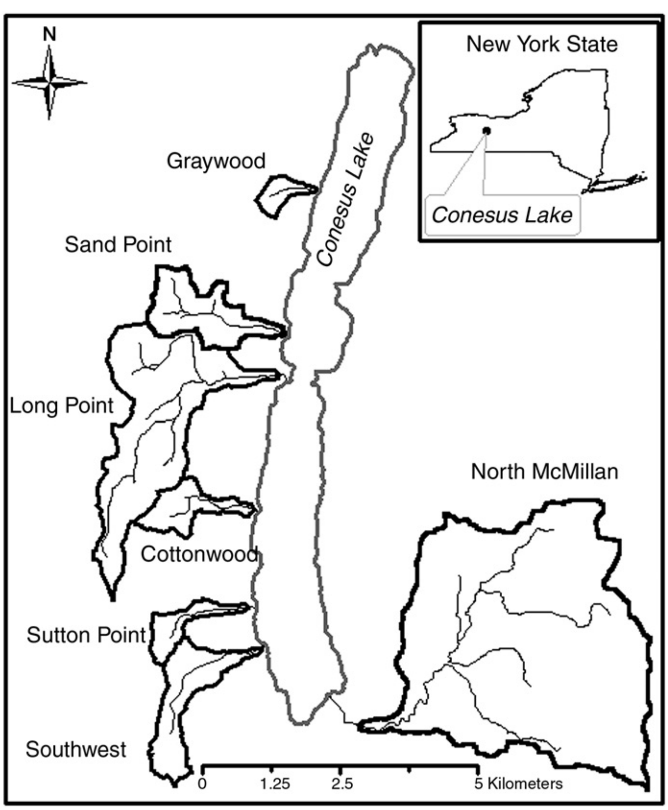

Fig. 1. Map of the sub-watershed of Conesus Lake. There are eighteen sub-watersheds to Conesus Lake. Only the seven sub-watersheds that were part of the Conesus Lake Watershed study are shown.

Table 1
Percent reduction of winter (21 December to 20 March) Graywood Gully marginal mean concentrations of total phosphorus (TP), soluble reactive phosphorus (SRP), total Kjeldahl nitrogen (TKN), nitrate, and total suspended solids (TSS) relative to winter 2002–03 under event plus nonevent (A), nonevent (B) and event (C) stream conditions.

A. Events plus nonevents

Year	2002–03	2003–04	2004–05	2005–06	2006–07
# of samples	21	14	18	15	13
TP	–	59.4[a]	37.6[a]	66.0[a]	68.7[a]
SRP	–	68.0[a]	37.9	74.9[a]	74.5[a]
TKN	–	50.0[a]	54.8[a]	57.9[a]	69.8[a]
Nitrate	–	24.1	43.7[a]	67.1[a]	69.5[a]
TSS	–	39.8	35.3	79.7[a]	64.4

B. Nonevents only

Year	2002–03	2003–04	2004–05	2005–06	2006–07
# of samples	11	13	13	13	12
TP	–	46.5[a]	26.0	53.4[a]	55.7[a]
SRP	–	58.8[a]	42.9	75.2[a]	73.9[a]
TKN	–	31.1	47.2	35.7	54.8[a]
Nitrate	–	47.5[a]	56.2[a]	78.6[a]	78.9[a]
TSS	–	−55.1	−28.0	39.6	−21.2

C. Events only

Year	2002–03	2003–04	2004–05	2005–06	2006–07
# of samples	10	1	5	2	1
TP	–	50.7	−13.7	59.7	79.7
SRP	–	98.2[a]	14.7	59.9	77.3
TKN	–	−16.1	−10.2	59.7	58.3
Nitrate	–	13.5	60.8[a]	45.2	77.3
TSS	–	−523.4	−468.0	45.5	57.8

[a] Significant difference from 2002–03. A negative sign represents a percent increase in parameter concentration.

Methods

Graywood Gully was studied for a 5-year period starting on 1 September 2002. As a result of Whole Farm Planning (Herendeen and Glazier 2009, Risse et al. 2005, Janke 2000) and soil sample nutrient analyses (Herendeen and Glazier 2009), a number of BMPs that addressed the identified water quality problems were implemented in the Graywood Gully sub-watershed of Conesus Lake, NY (Herendeen and Glazier 2009) with guidance from Cornell Cooperative Extension and the Livingston County (NY) Soil and Water Conservation District. One of the BMPs that was implemented dealt with timing of manure spreading. Spreading of manure on the surface of the land was common during the fall and winter in the Finger Lakes region, including Graywood Gully. During the winter of 2002–03, winter spreading of manure in hydrologically sensitive areas and in highly erodable land planted in corn was halted (Herendeen and Glazier 2009, Jacobs 2005). In the winter of 2005–06, traditional broadcast manure application (not a slurry) to fields in Graywood Gully was resumed from 21 to 25 January 2005. After the winter of 2005–06, winter spreading of manure was once again halted.

During the study period, Graywood Gully's stage height was monitored continuously for five annual cycles with a differential pressure transducer (ISCO 720) attached to an ISCO continuously recording flow meter (Model 6700) equipped with an automatic sampler (Makarewicz et al. 2009). Water samples were taken using two different methodologies: weekly manual grab samples and automated hydrometeorological event samples (Makarewicz et al. 2009). A hydrometeorological event was defined as a rise in the creek level of at least 2.54 cm in 30 min. Event and nonevent samples were analyzed for total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate + nitrite (NO_3), total Kjeldahl nitrogen (TKN), and total suspended solids (TSS) by standard methods (APHA 1999, Makarewicz et al. 2009). Replicate analysis, blind audits, and spiked additions are part of the QA/QC of our NELAC certified lab. Here we confine our analysis to concentration data from the winter seasons (21 December to 20 March) of the 5-year study period. Analysis of the entire data set may be found in Makarewicz et al. (2009). Weather data (daily temperature, precipitation) were obtained from the National Weather Service, Rochester, NY. Rochester is located 50 km north of Conesus Lake.

An Analysis of Covariance (ANCOVA) was performed on transformed (natural log) data to test for temporal trends in nutrient concentration with stream discharge as the covariate and concentration times sampling period as the interaction term. Slopes of each regression line were compared using a pairwise *t*-test of all possible pairs, in which the significance levels were corrected using the Bonferroni procedure. Regression line elevations were also analyzed for significant differences using a Bonferroni test of the estimated marginal means for the winter sampling period. The Bonferroni procedure offers an adjustment for multiple comparisons and is considered a conservative procedure for post-hoc analysis (Norleans 2001). The marginal means are the means for the analyte concentration after they have been adjusted for the covariate of discharge. Further information and an example of the ANCOVA may be found in Makarewicz et al. (2009).

Results

Analyses of Covariance (ANCOVA) were completed for the winter concentration of each analyte (TP, SRP, TKN, NO_3, and TSS). For example, temporal trends in winter nonevent TP concentration of Graywood Gully water were evaluated by considering the slope of the regression line of TP concentration versus stream discharge for each winter period using ANCOVA with discharge as the covariate. Pairwise *t*-test comparisons of the slopes of the ANCOVA regression lines for each winter period indicated that the slopes of the 2003–04, 2005–06, and 2006–07 regression lines were significantly different ($df = 1,4$; $P < 0.05$) from the regression lines of 2002–03. The slope of 2004–05 was not significantly different from the 2002–03 regression line. We also compared the difference in the elevations of each regression line by utilizing the marginal means of the discharge adjusted TP concentration from the ANCOVA analysis. The discharge adjusted nonevent mean TP concentration significantly (post-hoc Bonferroni test) decreased from 2002–03 to 2003–04, 2005–06, and 2006–07 (Fig. 2a) but not from 2002–03 to 2004–05 ($df = 1,4$; $P = 0.68$, Fig. 2a). A similar result occurred for nonevent SRP concentrations (Fig. 2b). The generally significant downward trend in nonevent winter TP and SRP marginal mean concentration was interrupted by a significant increase in 2004–05 ($df = 1,4$; $p > 0.05$, Figs. 2a, b).

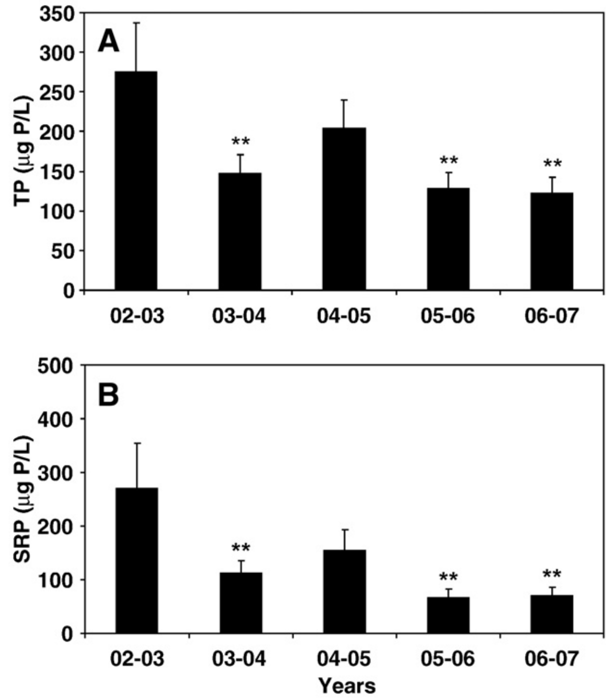

Fig. 2. Winter (21 December to 20 March) marginal mean concentration (±S.E.) of nonevent total phosphorus (TP) (A) and soluble reactive phosphorus (SRP) (B). **Significantly different from 2002–03 ($p < 0.05$, Bonferroni post-hoc test).

Considering the average of the data (nonevents plus events), SRP concentrations were not significantly different in winter 2004–05 from winter 2002–03 (Fig. 3a). This was not the case for TP (Fig. 3b). In general, considering the event/nonevent data for TKN and NO_3, the post-hoc analyses of the ANCOVA indicated that the elevation of the marginal mean winter concentration of TKN and NO_3 decreased significantly ($df = 1,4$; $p < 0.05$) from 2002–03 to 2006–07 (Figs. 3c, d). This was not true for TSS where no significant ($df = 1,4$; $p > 0.05$) decrease was observed from 2002–03 to 2006–07; however, TSS was significantly lower during 2005–06 than 2002–03 ($df = 1,4$; $p < 0.05$) (Fig. 3e).

The percent decrease in the marginal mean concentration of various analytes was surprisingly similar. We expected more variability from year to year as observed in the annual evaluations of impact of BMPs (Makarewicz et al. 2009). Where significant reductions occurred, percent reduction in marginal mean concentration (event plus nonevents) after 4 years of BMP implementation was: 68.7% for TP, 69.5% for NO_3, 69.8% for TKN, and 74.5% for SRP (Table 1a). Total suspended solids had a similar percent reduction (64.4%) as the other analytes but was not significantly different from 2002–03 to 2006–07 ($df = 1,4$; $p > 0.05$) due to the high variability within the data (Table 1a). All nonevent analytes, except TSS, had significant decreases from 2002–03 to 2006–07 (Table 1b), mimicking the results for the combined event/nonevent samples. During events, there were no significant changes ($df = 1,4$; $p > 0.05$) in TP and SRP concentrations and all other analytes in stream water (Table 1c). In general, where significant reductions of analytes occurred during the winter, they occurred during nonevent-flow regimes rather than during events.

Reductions of some analyte concentrations occurred quickly, reaching near maximum reductions within a year after the initiation of BMPs. Total phosphorus, SRP, and TKN were reduced in stream water by 50% or more by the first winter (2003–04) after BMP implementation (Table 1a, $df = 1,4$; $p < 0.05$) followed by a much slower decrease in concentrations until 2006–07. For example, there was a 50% decrease in TKN by 2003–04, 1 year after implementation of BMPs. This decrease slowly increased to a maximum of 69.8% by

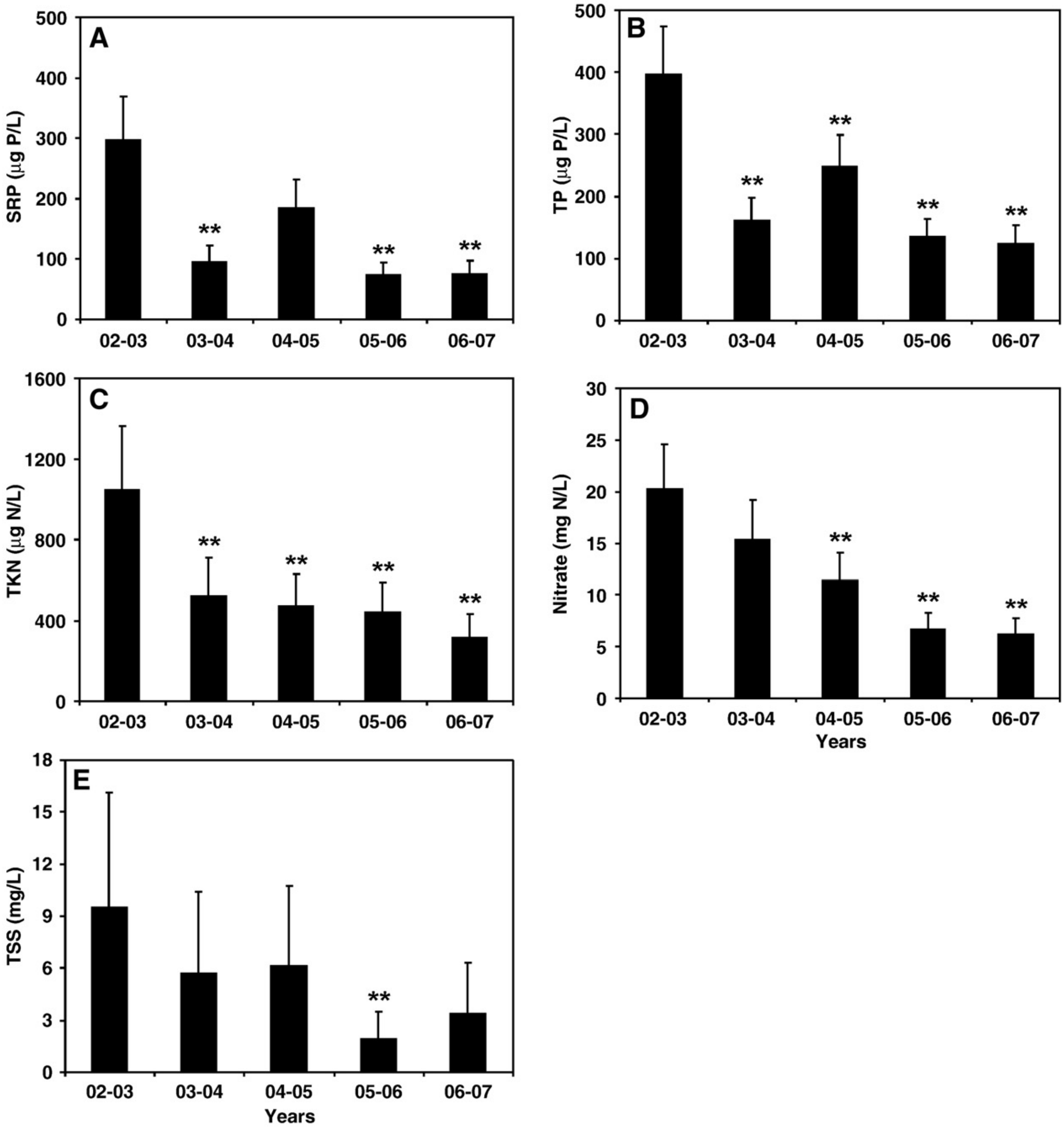

Fig. 3. Average event and nonevent winter marginal mean concentration (± S.E.) of soluble reactive phosphorus (SRP) (A), total phosphorus (TP) (B), total Kjeldahl nitrogen (TKN) (C), Nitrate (NO_3) (D), and total suspended solids (TSS) (E). **Significantly different from 2002–03 (p<0.05, Bonferroni post-hoc test).

2006–07 (Table 1a). The trends for NO_3 followed a different pattern, however. One year after the implementation, a 24.1% decrease in NO_3 was observed with an additional ~20% decrease per year to ~67% in 2005–06 with a small but insignificant decrease to 2006–07 (Table 1a). In fact, it was not until 2004–05 when NO_3 concentrations were significantly reduced (43.7% reduction, df=1,4; p<0.05) (Table 1a, Fig. 3d).

When comparing the winter yearly trends among all variables, significant decreases in TP, TKN, and NO_3 occurred with time (Figs. 3b, c, d). This was not the case for SRP. The decrease in SRP concentration was interrupted by a significant increase in the average winter SRP concentration in 2004–05 from previous years (Figs. 2b, 3a). Focusing on the winter of 2004–05, elevated levels of TKN, TSS, TP, SRP, and NO_3 were observed on 4 January 2005 prior to resumption of winter application of manure (Fig. 4). In the case of NO_3, concentrations were also high the previous week (27 December 2004) and the following week (10 January 2005). Maximum air temperatures for the 4 days preceding this peak were 13.3, 9.4, 9.8, and 2.8 °C (average above 0 °C); after this peak, maximum temperatures were generally below 0 °C with the average air temperature also below 0 °C (Fig. 5a). Through early January, creek discharge varied little (Fig. 5b).

Concentrations of TP, TKN, and NO_3 increased in the stream within 4 days of the application of manure (21 to 25 January 2005; Figs. 4a, c, d). An increase in SRP concentrations was observed but was delayed by a week (Fig. 4b). Nitrate remained elevated in the Graywood Gully stream for the rest of the winter; TKN, SRP and TP decreased for the next 7 weeks (Fig. 4). Discharge peaked for 1 day (18 January 2005), 3 days prior to the application of manure (Fig. 5b), and remained relatively similar to the end of February 2006. Average (Fig. 5a) and maximum air temperatures were below

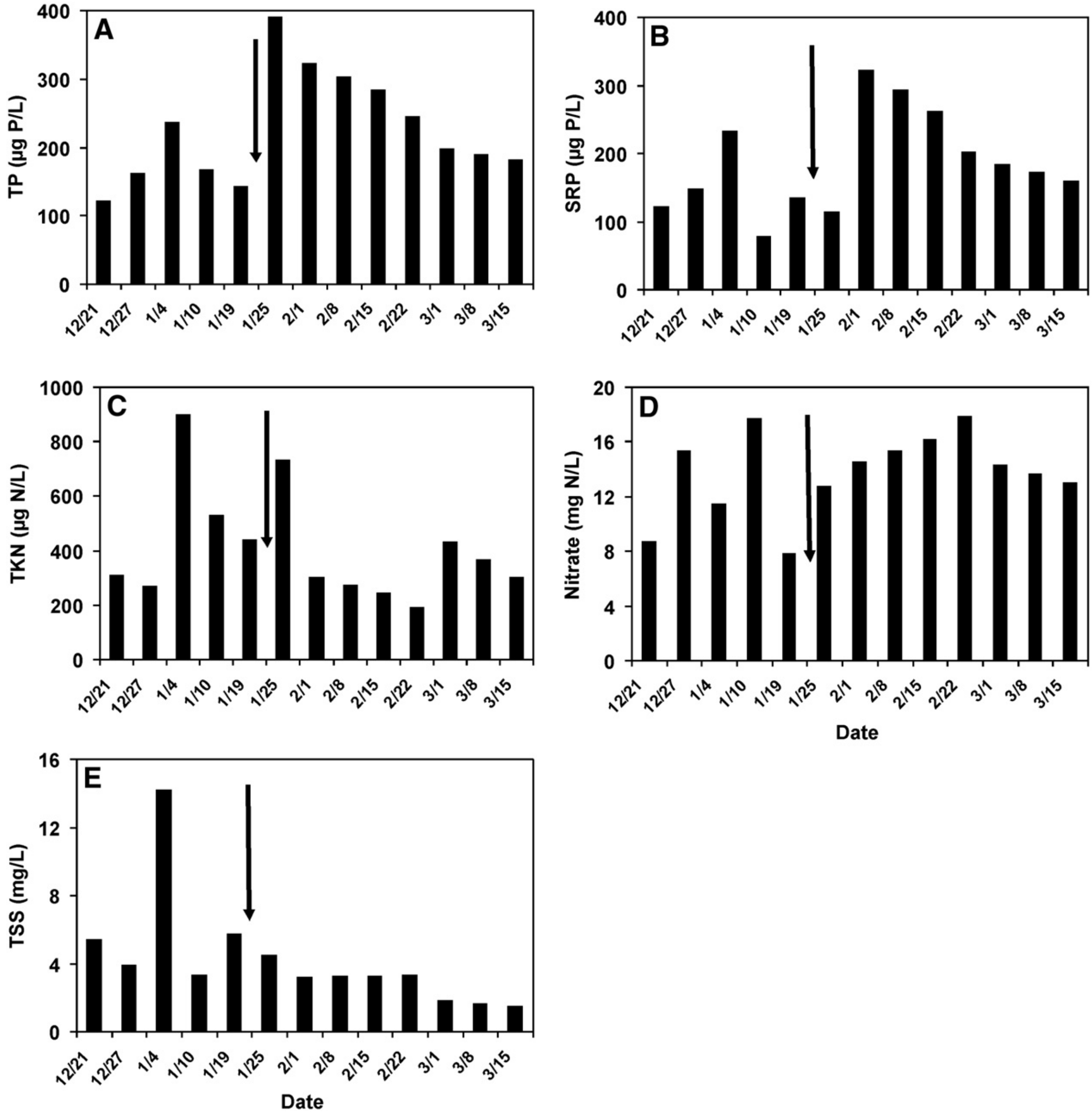

Fig. 4. Time trend of concentrations (event/nonevent) of total phosphorus (TP) (A), soluble reactive phosphorus (SRP) (B), total Kjeldahl nitrogen (TKN) (C), nitrate (NO_3) (D), and total suspended solids (TSS) (E) during the winter of 2004–05. The arrows indicate the period of manure application.

freezing for the entire period of manure application and for several weeks after application.

Discussion

Slope, application of manure over snow and during a snowfall, warm air and soil temperatures played a major role in the loss of P, nitrogen (N), and soil to downstream systems at the Graywood Gully watershed. During the winter of 2004–05 for a period of ~5 days, 2 years after the initial implementation of the BMP (curtailing winter manure applications), manure was again applied to the Graywood Gully sub-watershed. This resumption of winter manure application after a period of below freezing air temperatures (−14.4 °C) offered an opportunity to further examine the sub-watershed's response to the short-term reversal of an established management practice. Following the reintroduction of manure application, TP and SRP, albeit with a 1-week lag, increased dramatically and then slowly decreased for approximately 5 weeks back to previously observed concentrations (Figs. 4a, b). Except for the first date after application of manure, the increase in the amount of P being lost from the sub-watershed was due to the dissolved fraction of P rather than to the particulate fraction as SRP represented 91.7% of the TP concentration. Since stream discharge did not change over the 4-week period after manure application, and average and maximum air temperatures (Fig. 5) were well below freezing at least 3 weeks after the introduction of manure, the increase in concentration of nutrients was not associated with increased air temperatures nor increased flow from the watershed. However, during the manure application period, a steady snow fall occurred (over 10 cm). Also, just a few weeks before the application of manure, ground temperatures were still relatively warm due to air temperatures exceeding 10 °C. We hypothesize that the increase in SRP, TP, NO_3, and TKN, but not TSS, observed in stream water was from snow melt and perhaps from drainage tiles located in portions of this sub-watershed.

Similar to our results, Meals (1996) demonstrated that winter manure application drastically increased the export of P (SRP up to 1500%, TP 11%) from agricultural watersheds in Vermont. Repeated

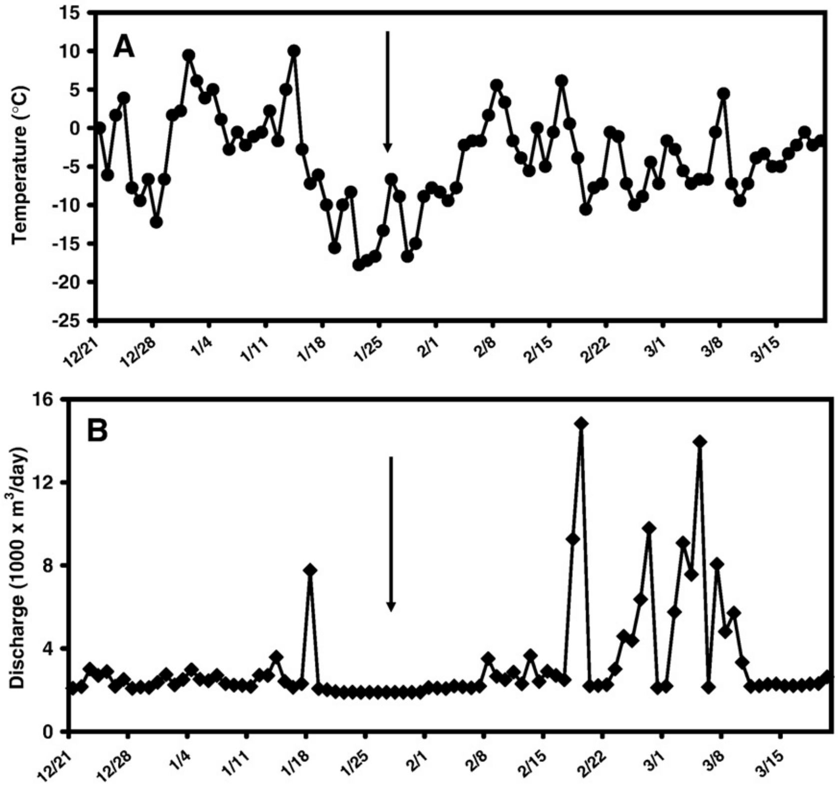

Fig. 5. Average temperatures (A) and daily discharge (B) during the winter of 2004–05 at Rochester, NY. The arrows indicate the period of manure application.

freezing and thawing significantly increased "water extractable P" from catch crop biomass and resulted in significantly elevated concentrations of dissolved P in runoff (Bechmann et al. 2005). Also, large losses of *N* in runoff occurred after the spreading of manure on frozen ground (Midgley and Dunklee 1945). Excessive nutrient losses were observed when manure spreading coincided with warming or active thawing periods (Klausner et al. 1976). Philips et al. (1981) and Lorimor and Melvin (1996) concluded that winter spreading resulted in considerably higher concentrations of N and/or P in runoff.

Others have shown differently. Hensler et al. (1970) observed that runoff losses from manure applied to frozen ground were variable. During one winter when it rained after manure application, significant losses of P and N were observed. In another year where little precipitation occurred, minimal nutrient losses were observed in runoff. In plot experiments, Converse et al. (1976) observed no significant difference in nutrient losses in various seasons of the year, including the winter. Young and Holt (1977) observed that winter-applied manure decreased runoff and soil and nutrient loss, apparently due to the mulching capabilities of solid manure. Many factors affect the quantity of nutrients that are lost from agricultural fields. These include amount and timing of precipitation and thawing (frozen ground), soil conditions, volume of runoff, slope and proximity to surface water, air temperature, snow cover, soil temperature, and soil permeability (Midgley and Dunklee 1945, Converse et al. 1976, Philips et al. 1981, Hensler et al. 1970, Steenhuis et al. 1981, Bechmann et al. 2005).

Our data also demonstrate that losses of nutrients and soil may occur during the winter even if manure is not applied. The peaks in early January of stream concentrations of TP, SRP, TSS, and possibly NO_3 were not associated with recent manure application but with air temperatures that were well above freezing for a 4-day period prior to 4 January 2005. Prior to January, manure had not been applied to the watershed since early autumn. The "January thaw" did not lead to an increase in discharge but did lead to a thawing of the soils. Application of manure to these fields in early January was not possible because of the muddy, water soaked ground.

In general, the agricultural BMP of removing winter manure spreading on environmentally sensitive areas over the 5-year study period at the Graywood Gully sub-watershed had an immediate impact on winter nutrient concentrations in the water draining off this watershed. Ultimately, a >68% decrease was observed 4 years after BMP implementation of all monitored analytes, except TSS (Table 1a). For example, there was a significant decrease in winter SRP concentrations for 2003–04, 2005–06, and 2006–07, but not 2004–05, from 2002–03. Clearly, the short-term resumption of winter manure spreading in 2004–05 lead directly to elevated analyte stream concentrations observed in the winter of 2004–05 and likely affected the interpretation of the annual SRP data by Makarewicz et al. (2009).

Unlike in Makarewicz et al. (2009), where a significant decrease in annual SRP was not observed in Graywood Gully, there was a significant decrease in winter SRP concentrations for 2003–04, 2004–05, and 2006–07 from 2002–03. Along with the decrease in SRP in drain tiles and soils (Makarewicz et al. 2009), the significant winter decrease in SRP in the stream lends support to the hypothesis that annual SRP concentrations were trending downward due to the management practices implemented. In fact, if the five sampling dates associated with the winter 2004–05 manure applications to the environmentally sensitive areas of the Graywood sub-watershed were removed from the data set, concentrations would be significantly lower (ANCOVA, $df = 1,4$; $p < 0.05$) than the initial year of 2002–03.

Decreased winter loading of nutrients to lakes is of importance. The agricultural community often assumes that winter losses of nutrients to downstream systems are not important as overabundance of algae or macrophytes is not directly observed in the lake during the winter. Thus, winter disposal of manure may occur because the environmental effects are not easily observed, often lake-cottage dwellers are not in residence, and the winter manure recommendations are not strictly followed. However, nonpoint inputs of P during the winter from manure operations can and will affect summer chlorophyll levels in a lake. Summer chlorophyll levels are a function of the lake's retention time and of P supplied from external sources seasonally. The winter concentration of TP in the water column of a lake is an excellent predictor of summer algae levels and serves as an index of the P pool available to phytoplankton during the following growing season in the Finger Lakes of New York State (Oglesby and Schaffner 1978). Managers of agricultural systems need to recognize that P losses from the watershed during the nongrowing season may detrimentally affect nuisance population of algae in lakes during the summer.

In summary, the water quality of Graywood Gully is very responsive to winter manure application on environmentally sensitive portions of the sub-watershed. Over the 5-year study period, significant decreases in winter nonevent but not event TP, SRP, TKN and NO_3 concentrations occurred within a year of the initial curtailment of manure application (BMP). A decrease in TSS concentrations was observed only in 2005–06 but not in 2006–07. The reversal of this downward trend for P was just as responsive, as the reinstatement of only a few days of winter manure application resulted in a significant increase in winter nonevent concentrations of SRP. That is, the short-term application of manure to a snow-covered landscape resulted in immediate losses of dissolved fractions including SRP and NO_3. Total Kjeldahl nitrogen, which represents ammonia and organic N, was also lost from the watershed for ~1 week after the application of manure, while the particulate fraction TSS did not increase to the downstream system. Management practices directed at improving the quality of water from agricultural watersheds that apply animal waste should consider application timing, topography, and soil and weather conditions to environmentally sensitive areas of a watershed. Deviations from the manure management plan can and do have short- and long-term effects on P concentrations in the water leaving a watershed. A comprehensive management plan should have a contingency for unforeseen circumstances, such as weather events, which may include short- to mid-term storage capabilities for manure (Edwards et al. 1997) so that the farmer can avoid any deviations from the BMP. Lastly, BMP-induced reductions of seasonal P and N losses from various Conesus Lake sub-watersheds (Makarewicz et al. 2009) have led to significant reductions in metaphyton and macrophyte populations in the nearshore zone (Bosch et al. 2009a, 2009b) and to a reduction in fecal indicators in streams coincident with the cessation of manure application (Simon and Makarewicz 2009).

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Quantification of phosphorus sources to a small watershed: A case study of Graywood Gully, Conesus Lake, NY

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ABSTRACT

Phosphorus sources within the Graywood Gully watershed impact water quality within the stream and receiving waters of Conesus Lake, New York. A mass balance approach was instructive in demonstrating the semi-quantitative impact of nonpoint and point nutrient sources on downstream aquatic systems and provided a mechanism to assist in targeting and prioritizing structural best management practices (BMPs) for agricultural areas. The identification and quantification of these sources reveal substantial sources coming from outside the topographic watershed boundary due to the overprint of the built environment on natural surface runoff pathways. The analysis of water sources and phosphorus loading indicated the importance of critical source areas in the watershed and adjacent areas and the effect of the built environment, including drain systems and road ditches, on changing critical hydrological pathways. The impact of BMPs within the watershed was masked by the external contributions from the "extended" watershed, adding over 40% of the total P load to Conesus Lake. This result suggests that the lack of significant decrease in dissolved phosphorus observed in the heavily managed Graywood sub-watershed is a result of not considering the "extended" watershed.

Introduction

Phosphorus export to local waterways has been a topic of concern over recent years (e.g., Sharpley et al., 1994; Heathwaite et al., 1996; Johnes and Hodgkinson, 1998; Haygarth and Jarvis, 1999). As various regulations have limited point source (e.g., discharge pipes) contributions of phosphorus (P), much of the attention has shifted to nonpoint sources (e.g., high soil P from long-term manuring). With this has come increased concern over the export of P from agricultural lands. As we endeavor to implement Best Management Practices (BMPs) (e.g., Makarewicz et al., 2009) to alleviate the impact of P on aquatic systems, it is necessary to better quantify both point and nonpoint source contributions at the watershed scale.

Gburek et al. (2001) and Heathwaite et al. (2000) developed the concept of Critical Source Areas (CSA) to explain the spatial variation in the risk of P export from agricultural land. Using the CSA concept, high risk is defined where high hydrological connectivity (transport) coincides with high P inputs (as fertilizers, animal manures or biosolids) and/or high soil P concentration. The risk will be lower where high connectivity coincides with a low P source or low connectivity coincides with a high P source. Haygarth and Jarvis (1999) and Haygarth et al. (2000) coined the term 'incidental' losses to account for situations where high source and high transport risks coincide during short periods of time. The term is used in particular to describe circumstances when the application of manures, fertilizers, or biosolids coincides with conditions favoring fast discharges and direct discharges to watercourses. An example would be the occurrence of a high intensity rainfall event which increases the opportunity for a faster mobilization of P into surface runoff waters.

In addition to overland flow transport of P to local streams, the contribution of drainage systems (e.g., tile drain and roof drain systems) has been found to contribute to P loading to streams. Heathwaite et al. (2006) described conditions by which high P fluxes were seen from a subsurface drainage system. Incidental events are described where wet conditions coincide with biosolids application and result in a loss of P via subsurface pathways. Soluble and fine particulate P may be transported via macropores to subsurface drainage systems. The observed events persisted for a maximum of approximately 2 weeks until the specific source of P susceptible to transport was diminished. The actual source of P is not important, as animal waste or mineral fertilizers may produce similar results given the correct set of conditions (Heathwaite et al., 2006).

The impact of the built environment on P export is an additional factor of concern. As Gburek et al. (2001) and Heathwaite et al. (2000) contended, areas of increased connectivity increase the risk of P export. In some cases, the connectivity factor is increased by the anthropogenic overprint on the natural landscape. This overprint may

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be due to agricultural activity (e.g., plowing) or development (e.g., roads and impervious surfaces). Carluer and De Marsily (2004) described the results of modeling the impact of anthropogenic features on a rural watershed. The 5 km^2 Kervidy catchment was modeled using TOPOG and ANTHROPOG, and it was determined that roads and ditches had a significant impact on the hydrologic response of the watershed. Moussa et al. (2001) showed similar results using the MHYDAS model. These studies highlight the importance of anthropogenic structures such as roads, ditches and tile drains on hydrologic response.

The Conesus Lake Watershed Study (Makarewicz, 2009), a long-term study using the small watershed approach, evaluated the impact of agricultural BMPs on maintaining nutrients and soil on the landscape and reducing losses of soil and nutrients to downstream aquatic systems. Makarewicz et al. (2009) investigated the effectiveness of agricultural BMPs by evaluating long-term changes in flux (kg/ha/d) and concentration (μg/L) of nutrients and soil in stream discharge water from the managed watershed. This study assumed that hydrologic inputs to the Graywood Gully watershed system were defined by the topographic divide of the catchment. Here, we take a mass balance approach to identify sources of P that enter the stream from both within the watershed and outside the surface topography watershed divide due to anthropogenic structures (i.e., culverts, tile drains, etc.) during a hydrometeorologic event. To better understand the sources of P, their pathways to the stream, and the efficacy of implemented BMPs, we evaluated the relative contributions of various point sources to the total water balance and phosphorus flux to Graywood Gully and Conesus Lake. Mass balance and mixing calculations with direct measurement of point sources allowed an evaluation of the actual contribution of nonpoint sources, primarily from the agricultural land-use, within the watershed to the total discharge to Conesus Lake. By understanding these “extra watershed” sources, we are better able to evaluate the impacts of BMPs implemented on agricultural land within the Graywood Gully watershed. Results of a smaller-scale study, such as this one, have direct implications for large lakes.

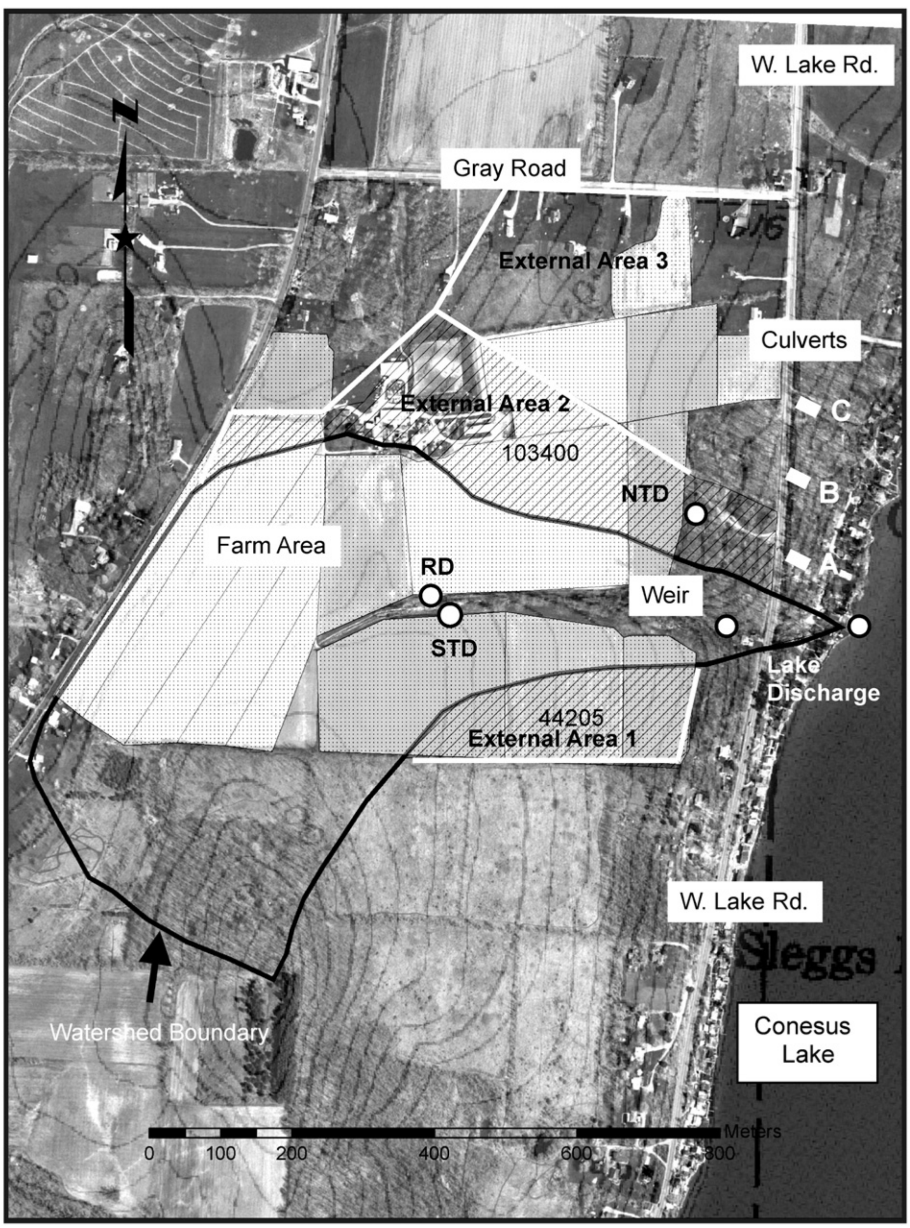

Fig. 1. Map of the Graywood Gully watershed and adjacent areas showing the location of major features and sampling points. NTD and STD: north and south tile drainage, respectively. RD: roof drainage.

Site description

Graywood Gully is a small watershed (~38 ha) in the northwest portion of Conesus Lake within the Lake Ontario watershed (see Fig. 1 in Makarewicz, 2009) and drains an upland area that is dominated by a single dairy-farming operation (Fig. 1). The stream flows eastward through a deeply incised gully (8% grade) surrounded by a small wooded area (~6.5 ha), finally reaching the developed margin surrounding Conesus Lake. In many reaches, the stream flows on fractured bedrock. Graywood Gully empties into Conesus Lake near the center of a large macrophyte bed (Fig. 1) described by Bosch et al. (2009). A further description of Conesus Lake and its watershed and the implemented BMPs are given in Makarewicz (2009) and Herendeen and Glazier (2009). An overview of the soils and geology of the Conesus Lake area are given in Forest et al. (1978). Soils in the watershed are dominated by the Conesus silt loam, a gently sloping somewhat poorly drained soil developed on limestone with the water table commonly within 15 to 45 cm of the surface, and the Lansing silt loam, a gently sloping moderately well drained soil developed on shale with the water table within 45 to 60 cm of the surface (USDA NRCS, 2008).

Within the Graywood Gully watershed, several point sources exist that contribute water to the stream (Figs. 1, 2 in Herendeen and Glazier, 2009). These include a tile drainage system and a roof drain system from the farm that discharges directly to the stream near its headwaters. In addition, a road ditch along Lake Road empties into the stream approximately 100 m upstream from the Graywood Gully Creek discharge point and into Conesus Lake. Fig. 1 provides a view of the watershed with the locations of different point sources and water sampling locations. The south tile drain (STD) system collects water from cropped fields within the watershed and External Area 1 and discharges to the stream approximately 100 m from its origin. The drained fields are currently under management with BMPs limiting manure spreading in this area due to existing high P status of the soils (Herendeen and Glazier, 2009). Some of the area drained by the STD system, however, is outside the mapped watershed boundaries (Fig. 1, External Area 1). Depending on seasonal rainfall amounts, locally elevated groundwater levels create persistent discharge to the stream from this source.

The north tile drain (NTD) system collects water from cropped fields, most of which are also outside of the topographic watershed boundary including fields in External Area 2 and some extending into External Area 3 (Fig. 1) and are not restricted from manure spreading. The NTD system discharges to the wooded area east of the farm. This area is outside the topographic watershed boundary, but the discharge reaches the road ditch along West Lake Road during rain events. A small culvert (Culvert A, Fig. 1) under Lake Road exists at the point where the NTD system discharge reaches the road ditch. A roof drain system takes water from the farm buildings and discharges to the Graywood Gully approximately 80 m from its origin. The farm buildings are outside the topographic watershed divides.

To the north of the Graywood Gully watershed, several small rivulets parallel the Graywood Gully stream but are not in the watershed as defined by surface topography. Field observations show that some of the road ditch water, mixed with the NTD system discharge, is diverted through Culvert A and enters Conesus Lake, north of the mouth of Graywood Gully, outside the watershed. Much of the water, however, continues flowing along the road ditch and enters Graywood Gully Creek at their intersection. The road ditch accumulates runoff for an approximately 0.75-km stretch from External Areas 2 and 3, extending north to Gray Road. Along this stretch of road, a total of three culverts (Culverts A–C, Fig. 1) divert water under West Lake Road and into Conesus Lake at points outside the watershed but in close proximity to the large macrophyte bed associated with Graywood Gully. Depending on the intensity of a storm event, varying amounts of water accumulate and ultimately reach Graywood Gully Creek. The area of the watershed from along West Lake Road to Conesus Lake is residential with a combination of year-round and summer homes, comprising ~0.9 ha. These homes are serviced by a perimeter sewer system around the lake.

Methods

A review of 3 years of flow data for Graywood Gully Creek (Makarewicz et al., 2009) indicated that while baseflow discharge was low, storm events may produce rapid runoff events with high discharges. A trapezoidal or Cippoletti type weir (L = 61 cm) was selected to optimize flow measurements during rain events in Graywood Gully; some rain events may have exceeded the capacity of a V-notch weir (Fig. 1). The weir was constructed of concrete blocks and plywood with an aluminum knife edge. The first row of concrete blocks was placed in a trench approximately 10 to 15 cm deep until bedrock was encountered. Plastic sheeting was placed on the upstream side of the weir, covered the face of the weir and extended upstream for approximately 7 m. The plastic sheeting covers the base of the stream and banks to the height of the weir creating a pool. While the pool created behind the weir was filling, no flow was observed bypassing the weir. Discharge at the south tile drain and at the roof drain was calculated by measuring the volume of water in large buckets over a known time-interval timed with a digital stop

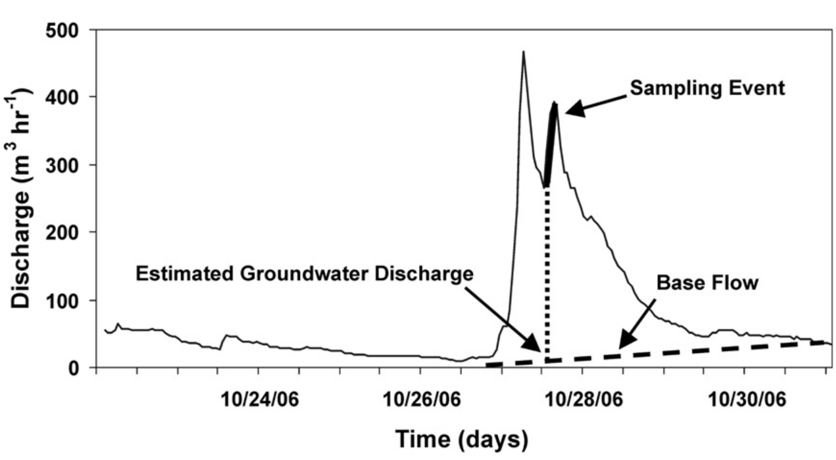

Fig. 2. Hydrograph for the mouth of Graywood Gully showing hydrologic response of the stream to the synoptic rain event of 27–28 Oct 2006. The highlighted portion of the hydrograph indicates the time over which discrete samples were collected. Base flow during the event was estimated using a constant slope method with the value selected being at the beginning of the sampling event. Groundwater discharge to the stream was estimated from the base flow at the beginning of the sampling event. Tick marks on the *x*-axis indicate noon (1200 h) on the date shown.

watch. Base flow, or groundwater discharge, to the stream was estimated using the hydrograph for the watershed (Fig. 2) that showed flow immediately before, during and after the storm event. The contribution of base flow during the storm event was estimated by using the constant slope method and by estimating base flow during the time of sampling.

Water sampling occurred between 0930 and 1045 on the morning of 28 October 2006. A synoptic rain event began in the evening of 27 October with persistent rain occurring through the night and continuing through the day of 28 October. A weather spotter in the watershed measured the total rainfall for the event (3.15 cm). Water samples were collected in 500-mL acid washed polypropylene bottles from each of six locations within the watershed and from a tile drain system discharge point that drains fields north of the watershed boundary. Samples were kept on ice and refrigerated at 4 °C until analyzed for total phosphorus (TP). Unfiltered water samples were digested within 48 h with persulfate and sulfuric acid prior to analysis (Method 4500-P B5, APHA, 1998). Analysis of P was completed using the ascorbic acid method (Murphy and Riley, 1962) on a Beckman model DU640 spectrophotometer. Discharge monitoring at the mouth of Graywood Gully Creek was continuously monitored (Makarewicz et al., 2009) and used to determine the intensity of the hydrometeorological event. Discharge was also determined at the south tile drain and roof drain by direct measure of the volume of discharge over time and at the weir by determination of flow rate through the fixed geometry profile.

A Geographical Information System (GIS) was constructed for the Graywood Gully watershed using ArcGIS version 9.3. Aerial photographs were acquired from the NYS GIS Clearinghouse, and other data was obtained from the Cornell University Geospatial Information Repository (CUGIR). Additional site specific data was created for the use of the project. Aerial photographs (e.g., Fig. 1) were used to delineate the land use around the watershed, and a topographic map of the area was used to determine watershed boundaries. Visual inspection of the aerial photographs and important features, such as tile drains, the constructed weir, and individual farm fields, were mapped in the watershed. Using the topographic map and aerial photos, we mapped the adjacent watersheds that feed the rivulets to the north of the Graywood Gully watershed. These areas outside the watershed may be contributing water to the Graywood Gully stream discharge under certain conditions and represent a potential variable boundary of the watershed during a rain event due to increased anthropogenic connectivity.

Results

The 6 days prior to the 12-hour rain event (2.84 cm) of 27–28 October 2006 were overcast with small amounts (0.58 cm) of measurable precipitation. The small sub-watersheds of the Conesus Lake catchment are typically “flashy”, responding very quickly to a rain event (Fig. 2). The initiation of the event, as measured by the hydrologic response of the stream, began at 2200 on 27 October and ended at approximately 0400 on 30 October.

GIS analysis indicated that during nonevent flow, water came from within the watershed mapped by surface topography. However, when a rain event occurred, water was added to the stream through a series of road ditches from outside of the topographic watershed, making the contributing area much larger. That is, water was channeled through ditches to the Graywood Gully stream to a point slightly above the sampling site of Makarewicz et al. (2009). Surface topography indicated that the total area of the Graywood Gully watershed was 38.1 ha. The extra area of the watershed that contributed event water through road ditches was 14.8 ha. Considering the larger watershed to include roadside ditches, the total amount of agricultural land within the Graywood Gully increased from 28.2 to 30.8 ha and included areas where manure management and other BMPs were not implemented.

Mass balance

Upland flow

Initial calculations partitioned water flow to the various sources leaving the upland agriculture-dominated portion of the watershed as follows:

$$\begin{aligned}\text{Weir discharge}(\mathrm{Q}-\text{weir}) = {} & (\mathrm{Q}-\text{base flow}) + (\mathrm{Q}-\text{tile drain}) \\ & + (\mathrm{Q}-\text{roof drain}) \\ & + (\mathrm{Q}-\text{suface runoff}).\end{aligned}$$

The estimated base flow for the entire watershed was approximately 350 L/min. When this value was evenly distributed over the length of the stream, base flow discharge to Graywood Gully for the stream reach above the weir was calculated to be 217 L/min. Surface runoff was calculated to be the difference of Q-weir minus the sum of the other sources. Surface runoff represented ~76.4% of the flow, and the remaining sources in order of contribution were: tile drain (10.8%), base flow (7.6%), and roof drain (5.1%) of the water passing the weir at the time of sampling (Table 1).

Upland phosphorus flux

Loading (L) or flux of P was determined by using a mixing equation where water concentration (C) multiplied by discharge (Q) provided an estimate of loading for each source. Phosphorus concentrations at the various sources (weir, tile drain, roof drain, and at a rivulet near the head that drains a field maintained in alfalfa in the headwaters of the Graywood Gully creek) varied considerably (Table 1). For example, P concentrations ranged from 80 μg/L (tile drain) to a high of 6115 μg/L (roof drain). The concentration of groundwater discharge to the base flow cannot be directly measured but was assumed to be similar to the concentration of groundwater discharged at the south tile drain. The calculated P loading using field measurements underestimates the concentration of P at the weir (Table 1). To correct the concentration at the weir, we assumed the P concentration of surface runoff to be an unknown. Recalculation gave a calculated surface runoff concentration (C-surface runoff) of 515 μg/L, higher than the measured value of 385 μg/L (Table 1). This may be a more realistic value as the single surface runoff sample was taken from a field that was in alfalfa at the time of sampling. Fields that had been in corn and soybean during that season were bare at the time of our field sampling and may contribute more P to overland flow.

Multiplying concentration by flow for each source provided the load (L) contribution of each water source and the load leaving the agricultural portion of the watershed at the weir (Table 1). Surface runoff represented the largest contributor of P (54.8%) to the downstream system, considerably less than its contribution to the water balance of 76.4% (Table 1). The roof drain contributed 43.1% of the P load, which was ~8.5 times greater than its relative contribution to the water balance. The relative contributions of P from base flow and the south tile drain were much less at 0.9% and 1.2%, respectively (Table 1). The latter suggests that P transport through

Table 1
Results of discrete sampling for discharge, total phosphorus (TP) concentration, and phosphorus (P) flux for the upland area of Graywood Gully.

Source	Discharge (L/min)	Concentration (μg/L)	Flux (mg P/min)
Weir	2839 (100)	719	2042 (100)
Baseline	217 (7.6)	85	18.4 (0.9)
Tile drain	308 (10.8)	80	24.6 (1.2)
Roof drain	144 (5.1)	6,115	881 (43.1)
Surface runoff	2170 (76.4)	385 [515]	1118 (54.8)

Values in parentheses are percent of total. Surface runoff discharge is by difference. The value of surface runoff concentration in brackets is the calculated value, determined by difference, for the average TP concentration of surface runoff for the upland flow area.

the soil was limited on this day and was likely reflective of typical results as indicated by data from preliminary sampling on three separate occasions.

Flow to the lake

The area of the watershed below the Cippoletti weir at Graywood Gully (0.9 ha or 2.4% of the total watershed) consists of houses along small portions of Conesus Lake, West Lake Road and Grayshores Road (Fig. 1). West Lake Road roughly parallels the shoreline, decreasing in elevation from north to south along a 0.8-km stretch before intersecting Graywood Gully. Along this route, most of which is outside the watershed boundaries, the road intersects three intermittent rivulets. Each rivulet has a small culvert passing beneath West Lake Road, but the ditch along the west side of the road may divert some of the discharge southward. The road ditch empties through a culvert directly into Graywood Gully as it passes beneath West Lake Road. Discharge to the lake (Q-lake) may then be quantified as the sum of water discharging from the agricultural area of the watershed through the weir plus additions to the stream downstream of the weir. The area below the weir generally slopes toward the lake, rather than the stream. During our study, this area provided direct drainage to the lake. It was assumed that surface runoff to the stream from the segment below the weir to the stream was insignificant. Groundwater infiltration was estimated for the area above the weir. Thus the major contributors to discharge to the lake (Q-lake) were flows from the weir (Q-weir) and base flow (Q-base flow) and discharge from the West Lake Road culvert (Q-road ditch):

$$\text{Discharge to lake}(\mathrm{Q-lake}) = (\mathrm{Q-weir}) + (\mathrm{Q-road\ ditch}) + (\mathrm{Q-base\ flow}).$$

Measured values for Q-lake and Q-weir and the estimated value for Q-base flows were used to estimate Q-road ditch, as it was not possible to directly measure this component. Discharge from the upland area (Q-weir) within the topographic watershed was the highest, contributing 86.5% of the flow with ditch (Q-road ditch) and base flows (Q-base flow) being 9.5% and 4.0%, respectively (Table 2).

Water samples were collected at the weir and road ditch and at the discharge point to Conesus Lake. Total phosphorus concentrations ranged from 85 μg/L for base flow to 719 μg/L at the weir. Phosphorus loads were calculated in the same way as for the area above the weir. The sum of the three primary sources underestimates loading to the lake (L-lake) by only 4 mg $\min^{-1}$ or 0.1% of the total load. The relative contribution to lake loading (L-lake) from the agricultural portion of the watershed (L-weir) was highest at 96% (Table 2). The remaining 4% of the load came from the road ditch (3.4%) and from the base flow within the lower portion of the watershed (0.5%). Although lake load from the road ditches (L-road ditch) was a small percentage contributor, discharge from the road ditch came primarily from outside of the topographic watershed boundary.

Discussion

The Graywood Gully watershed is defined by the surface topography. During rain events, however, areas external to the delineated watershed may contribute to the discharge measured at the stream mouth. Anthropogenic factors, such as storm water management features, can change the nature of water movement across a landscape (Carluer and De Marsily, 2004). In this case, several features, such as road, road ditches and tile drains, impact the natural flow of water, effectively extending the boundaries of the watershed beyond those defined by surface topography. This extended area of contribution of water is the result of modifications to natural drainage from the built environment and results in a variable contributing area depending on the magnitude of the individual storm event.

Three major factors contribute to the extended contributing area: (1) tile drainage of agricultural fields, (2) barn roof drainage, and (3) a storm water ditch along West Lake Road. Applying the concept of critical source areas developed by Gburek et al. (1996) and Heathwaite et al. (2000) to this case reveals unsuspected sources of P loading. The implemented BMPs are shown to be effective in reducing P export from the areas under management (Makarewicz et al., 2009). As part of the BMPs, manure spreading was moved away from high risk areas in Graywood Gully to areas outside the watershed that appear at first to have low risk as defined by their low hydrological connectivity under natural conditions. The overprint of the built environment, however, effectively changes the connectivity factor, thus increasing the P export risk. Whether or not this may be considered an incidental loss (Haygarth and Jarvis, 1999; Haygarth et al., 2000) is debatable given the variability of the contributing area to the stream, although in this case the contributing sources are well defined.

Tile drainage is used in two areas on the farm. In both cases, some of the fields drained are external to the managed studied watershed. Fields to the west and south of Graywood Gully (Fig. 1, External Area 1) are drained by the STD system with discharge entering the stream near its headwater. These fields, limited in area, are part of fields actively managed under BMPs that reduce manure and P fertilizer input. As a result of the implemented BMPs and the current soil P concentrations (Herendeen and Glazier, 2009), P loading from the south tile drain does not impact water quality in Graywood Gully. The fields to the north (Fig. 1, External Areas 2 and 3) are not under the current BMP plan and continue to receive manure. Discharge from the north tile drain, mixed with road ditch water, enters Graywood Gully along West Lake Road. With higher TP concentrations, this discharge may have negative impacts on water quality in Graywood Gully and ultimately Conesus Lake.

The roof drain system installed for farm buildings also contributes large P loads to Graywood Gully. Although roof drains contribute only 5.1% of the flow leaving the water at the weir, the roof drain contributed 43.2% of the P loading during this one rain event. Although the exact source of the P to the roof drain is not known, the sources may come from accumulated dust, bird droppings, or discharge of high P content cleaning detergents used in milkhouse operations. Further investigation and elimination of this source would have a large impact on the resulting P load to Conesus Lake. Recalculation of the mass balance calculation indicates that if the roof contributions were eliminated, P loading at the weir would be reduced from 2041 mg $\min^{-1}$ to 1160 mg $\min^{-1}$ or a concentration of 408 μg L^{-1}. Given that the roof drain system is not part of the watershed and is not included in the implemented BMPs, the apparent P loading to Graywood Gully from active farming operations within the watershed is reduced by approximately 43% in this case study during one large event.

Fields to the north of the Graywood Gully watershed and east of the barn area are partially drained by the NTD system and by overland flow that discharges to the road ditch outside of the watershed. These fields outside the watershed boundary are not currently managed under the BMPs implemented within the watershed. As a result, the fields still receive manure applications.

Table 2
Results of discrete sampling for discharge, total phosphorus concentration, and phosphorus (P) flux for the flow to lake area of Graywood Gully.

Source	Discharge (L/min)	Concentration (μg/L)	Flux (mg P/min)
Lake	3283 (100)	648	2127 (100)
Weir	2839 (86.5)	719	2041 (96.1)
Road ditch	311 (9.5)	230	71.5 (3.4)
Base flow	133 (4.0)	85	10.6 (0.5)

Values in parentheses are percent of total. All discharge and concentration values were measured in the field.

The continued applications result in high soil P concentrations and resulting high concentrations of P in the north tile drain discharge. During rain events and given sufficient soil moisture, the north tile drain discharge flows overland in a small rivulet until intercepted by the ditch on the west side of West Lake Road. Despite a culvert under West Lake Road, observations of the road drainage ditch indicate that water flows down the entire length of the West Lake Road ditch during major rain events (Fig. 1) and past three small culverts (A–C), ultimately reaching the Graywood Gully at the intersection of the road ditch with the stream. A mass balance approach for this one rain event suggests that a reduction in P loading to Conesus Lake may be achieved if road ditch water P concentrations were reduced, possibly by extending BMPs to the area drained by the NTD system. Given a reduction in the concentration of road ditch water from 230 μg/L to a value similar to groundwater discharge (80 μg/L), a further decrease in P loading to Conesus Lake would be achieved to a value of 1194 mg/min. Management of the discharge from this one tile drain could result in a total decreased P loading of 50% from this one tile drainage to the nearshore of Conesus Lake, an area in the nearshore plagued by large macrophytes and metaphyton biomass (Makarewicz et al., 2007; Bosch et al., 2009a,b).

In the heavily managed Graywood Gully, Makarewicz et al. (2009) did observe reductions in dissolved phosphorus concentrations, but they were not statistically significant. The magnitude of the P loss from this one roof drain and from the road drainage ditch, both external to the surface topography-defined watershed of Graywood Gully, suggests a likely cause for this lack of significance. That is, the impact of best management practices within the watershed was masked by the external contributions from the "extended" watershed, adding over 40% of the total P load to Conesus Lake.

Since only one event is sampled, we recognize that broader interpretation is limited, as the data do not reflect all possible conditions. However, the simple mass balance approach for one rain event is instructive in demonstrating the semi-quantitative impact of nonpoint and point sources on downstream aquatic systems. As such, it also provides a mechanism to assist in targeting and prioritizing structural BMPs for agricultural areas. Nutrient reductions due to the implementation of BMPs in watersheds (Makarewicz et al., 2009) are better understood when the various sources of P being contributed to the system are better defined. For example, the contribution from the roof drain and tile drains demonstrates the importance of these sources and provides direction on the implementation of new BMPs while showing the contribution of P to Conesus Lake from surface runoff within the managed watershed to be lower than previously believed. The importance of each of the apparent small P loadings was also identified. As Makarewicz et al. (2009) have suggested, it will be the cumulative implementation of many small BMPs to the watershed that will result in a major reduction of nonpoint sources of nutrients to lakes.

The impact of an "extended" watershed caused by anthropogenic factors, such as tile drains, roads and road ditches, may have had some impact on conclusions from other work (Makarewicz et al., 2009) where the issue of extended watersheds larger than those defined by surface topography was not recognized. This appears to be the case for dissolved phosphorus where significant decreases were not observed by Makarewicz et al. (2009), but not for nitrate, total phosphorus, total suspended solids or total Kjeldahl nitrogen where significant reductions were observed over a 5-year period (Makarewicz et al., 2009).

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Phosphorus fractionation in soil and sediments along a continuum from agricultural fields to nearshore lake sediments: Potential ecological impacts

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ABSTRACT

The movement of phosphorus (P) from agricultural fields to streams and deposition in the nearshore of the lake presents a continuum of related physical and chemical properties that act to partition P into different physico-chemical fractions. We investigated changes in soil and sediment P fractionation as material was eroded from predominantly agricultural fields, transported via stream sediments, and deposited in a nearshore lake environment. Total phosphorus content of the soils and sediment decreased from field soils with an average concentration of 553.81 mg P kg^{-1} to 202.28 mg P kg^{-1} in stream sediments to 67.47 mg P kg^{-1} in lake sediments. Significant changes in P fractionation occurred during erosion, transport, and deposition of the particulate or sediment phase. The fractionation of P within the soils and sediments changed significantly from aluminum and organic matter associated P dominant in field soils to calcium associated P dominant in nearshore lake sediments. Various physical and chemical processes appear to be responsible for these transformations which impact the mobility and bioavailability of P. A significant amount of P was lost from field soils as they were transported and deposited. This P has either become available to biota or deposited in deeper portions of the lake system. Ultimately, the impact of P export on the nearshore lake environment may be influenced by the changes in P fractionation that occurred during transport and deposition and by the influence of macrophytes on the biogeochemical cycling of P in the sediment.

Introduction

The export of phosphorus (P) from agricultural and other land use areas is widely recognized as a contributing factor to eutrophication of surface water bodies. Numerous studies have investigated the contribution of point and diffuse sources of P from source areas to stream channels (e.g., Sharpley et al., 1994; Heathwaite et al., 1996; Johnes and Hodgkinson, 1998; Haygarth and Jarvis, 1999). The impact of excess P on the freshwater aquatic systems has also been widely studied in its effects on biological growth and accumulation in stream and lake sediments (e.g., Bostrom et al., 1988; Hilton et al., 2006). Conesus Lake, Livingston County, New York, is no different; macrophytes and metaphyton in the nearshore zone are impacted by excessive P (Makarewicz et al., 2007, 2009) and perhaps nitrate (Bosch et al., 2009a,b).

While streams have often been viewed as simply a transfer mechanism for P to lakes, recent work has investigated processes that occur during transport. It is important to recognize mechanisms that transform P within different physico-chemical fractions within the stream channel (Melack, 1995; Evans and Johnes, 2004; Evans et al., 2004). House (2003) discussed the various processes that may transform P including sorption, coprecipitation, and redox reactions. Furthermore, it is important to recognize the interaction of soluble reactive phosphorus (SRP) with stream sediments. Numerous recent investigations (e.g., Jarvie et al., 2006; Ryan et al., 2007) identify the ability of stream sediments to act as both sinks and sources for SRP within the stream. The role of stream sediments as either sink or source is dependent on the SRP concentration in the stream water and may change both temporally and spatially within a watershed. In comparing rural versus urbanized watersheds, Owens and Walling (2002) found that particulate phosphorus (PP) increased in stream sediments receiving point sources of SRP. Furthermore, they suggested that up to 20% of the PP in stream sediment is likely to be easily bioavailable as inorganic P phases dominate. These mechanisms may also be active in lake sediments as well.

Physical processes are also important during P transport within streams. It is often found that PP as measured by the difference in total extractable phosphorus (TP) and total dissolved phosphorus (TDP) is the dominant phase in which P is transported through the system. The transport of PP may be the most significant mechanism for P transport to lakes and may play a role within stream processes. Studies on the transport of P by suspended sediment (Walling et al., 2001; Evans et al., 2004) identified suspended fine silt and clay and organic matter fractions as having the highest TP. Both spatial and temporal variations

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in TP and P retention mechanisms by the solid phase were observed. Phosphorus retention within the stream system is typically dominated by calcite co-precipitation within bed sediment and physical trapping of sediment by reduction of flow velocity.

Within lake sediments, various mechanisms controlling P fractionation that are similar to those in stream sediments occur. Within these sediments, two zones are easily identified, an oxidized zone extending a few millimeters to a few centimeters below the sediment–water interface and a reduced zone below. These sediments may act as both sinks and sources for P cycling with a large fraction of the inorganic P in surface sediments in equilibrium with the water column (Golterman 1995). The cycling of P is most prevalent in stratified lakes with anoxic hypolimnion, but studies have shown a significant cycling of P from oxic sediments (Bostrom et al., 1989; Jensen and Andersen, 1992; Rydin and Brunberg, 1998) where nearshore sediments are likely to be. Rydin (2000) determined that P mobilization occurs under both oxic and anoxic conditions and that exchangeable and Fe-bound P are generally mobile. Organic associated P is about 60% mobile, with greater mobility in anoxic sediments. Phosphorus associated with Al and Ca is immobile and may be considered permanently bound.

Biota may also play a role in P cycling in lake sediments. Jensen and Andersen (1992) determined that P release from aerobic sediments may deplete the Fe-bound P despite Fe remaining in the solid phase. The release process involves a complex relationship between nitrate concentrations and microbial activity in the sediment. High nitrate concentrations act not only to increase the thickness of the surface oxidized layer but also to stimulate microbial activity. A seasonal effect that increases P retention in the sediment during the winter with subsequent release during the following late summer and autumn is observed. Rooted macrophytes may also play a role in P cycling from nearshore lake sediments by altering the sediment biogeochemistry. Wigand et al. (1997) suggested that macrophyte species composition may alter the pore water and solid phase P composition of sediment. Kisand and Noges (2003) observed distinct differences in P release from sediments between two shallow eutrophic lakes. The plankton-dominated lake had a two-fold increase in P release during anoxic periods over oxic conditions, but no difference between oxic and anoxic release patterns was seen in a macrophyte-dominated lake.

Here, we investigate changes in soil and sediment P concentration and fractionation as material was eroded from predominantly agricultural fields, transported through a hard-water stream, and deposited in a nearshore lake environment. The study investigated the influence of various physical and chemical processes that alter the concentration and form of P in the sediment along the field, stream, and nearshore lake continuum and evaluated their impact on biogeochemical cycling of P, growth of nuisance macrophyte species, and implications for lake management with respect to nutrient status of Conesus Lake. The combination of P concentration in operationally defined soil and sediment fractions with the relative percentage distribution between fractions is used to elucidate the biogeochemical cycling process that occurs and the size of P pools that might become available for plant uptake.

Methods

Site description

The Graywood Gully sub-watershed (38 ha) drains an upland area that is dominated by a single dairy-farming operation (Fig. 1) with fields typically planted in corn and alfalfa. Soils in the watershed are dominated by the Conesus silt loam, a gently sloping somewhat poorly drained soil developed on limestone with the water table commonly within 15 cm to 45 cm of the surface, and the Lansing silt loam, a gently sloping moderately well-drained soil developed on shale with the water table within 45 cm to 60 cm of

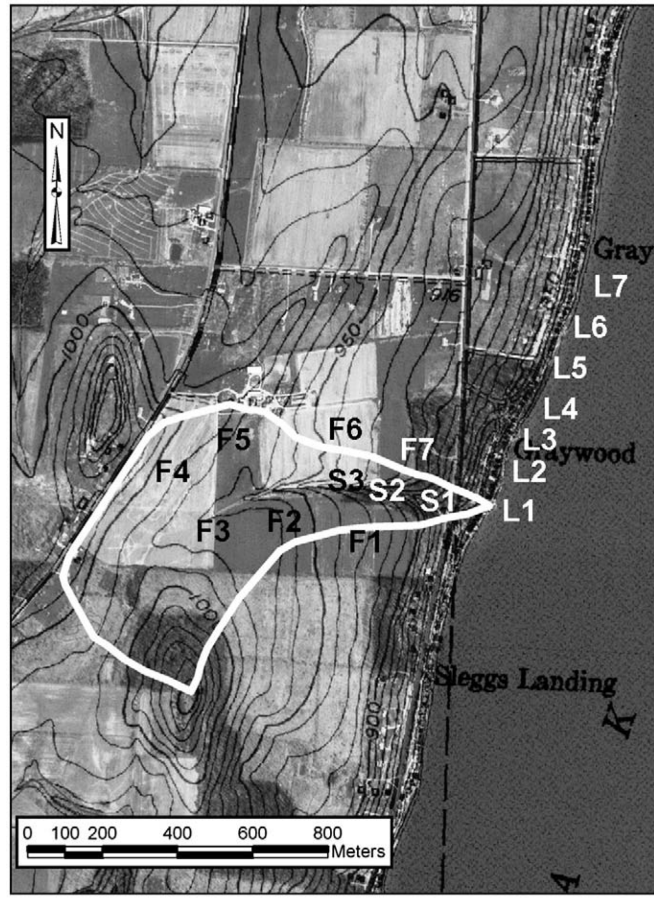

Fig. 1. Sample locations in Graywood Gully and Conesus Lake, NY. F = field soil sample locations; S = stream sediment samples; L = nearshore lake sediments samples. Due to scale, the location of the nearshore lake sediment samples is approximate.

the surface (USDA NRCS, 2007). Soil developed on the Conesus silt loam was found to have high P index values (Herendeen and Glazier, 2009). The stream flows eastward through a deeply incised gully in a small wooded area, finally reaching the developed margin surrounding Conesus Lake. In most places, the stream flows on fractured bedrock. The stream empties into Conesus Lake at the southern extent of a large macrophyte bed. A detailed description of the watershed and the implemented Best Management Practices (BMPs) is given in Makarewicz et al. (2009) and Herendeen and Glazier (2009).

Soil and sediment sampling

Soil samples were collected from six managed fields (F1 through F6) and one sample (F7) from the wooded area east of the farm (Fig. 1) in 2006. In each case, ten randomly located samples were collected from the top 2.5 cm of soil with a stainless steel scoop and composited to produce a single sample. Each composited sample was passed through a 2-mm sieve, homogenized, placed in a plastic container, and stored at field moisture at 4 °C in plastic bags until analyzed.

Stream sediment samples, S1 through S3, were collected from approximately the top 2.5 cm of sediment from three locations in Graywood Gully and placed in acid-washed glass jars (Fig. 1). Sample S1 was collected from sediment that had accumulated behind a weir installed in July 2006. Samples S2 and S3 were collected from natural, shallow pools within the stream channel that had accumulated fine sediments.

Nearshore lake sediment samples were collected from nine locations (L1 through L9, Fig. 1) along a transect running parallel to the shoreline beginning at the mouth of Graywood Gully and moving northward in the general direction of the longshore drift (Li et al., 2007). Samples from approximately the top 2.5 cm of sediment were collected at a water depth of approximately 1.5 m and placed in acid-washed glass jars. In some cases, the sampling depth was less due to the presence of zebra mussels (*Dreissena* spp.) and a hard substrate. In the laboratory, excess water was drained from the nearshore and stream sediment samples and then passed through a 2-mm sieve and stored in glass jars at 4 °C until analyzed.

Chemical analyses

All soil and sediment samples were analyzed for moisture content (MC), organic matter content by loss on ignition (OM-LOI), and P fractionation by sequential extraction. Moisture content was determined on all samples prior to other analyses to determine dry weight of subsamples. The MC was determined by oven drying in ceramic crucibles at 105 °C for a minimum of 24 h or until constant weight and transferred to a muffle furnace to determine OM-LOI by dry oxidation at 360 °C for 16 h (Xie et al., 2000).

Phosphorus fractionation was completed on all samples using the method of Psenner et al. (1988) with minor modifications. The procedure used 1 M NaCl to extract pore fluid and easily exchangeable P. Phosphorus associated with reducible phases was extracted using two extractions of bicarbonate-buffered 0.11 M sodium dithionate with the extracts combined to form a single sample. Aluminum associated P and organic P were extracted using 1 M NaOH with subsequent analysis of the extracts for SRP and TP. Aluminum associated P was determined by the results of the SRP analysis, and organic P was the difference between TP and SRP. Calcium associated P was determined by extraction with 1 M HCl. In all cases, samples were rinsed with 0.1 M NaCl following each extraction to remove entrained fluid and to prevent readsorption of soluble P to the solid phases. The TP concentration of a sample was determined by summation of each fraction. Analysis of all extracts was completed using the ascorbic acid method (Murphy and Riley, 1962) with standards developed for each

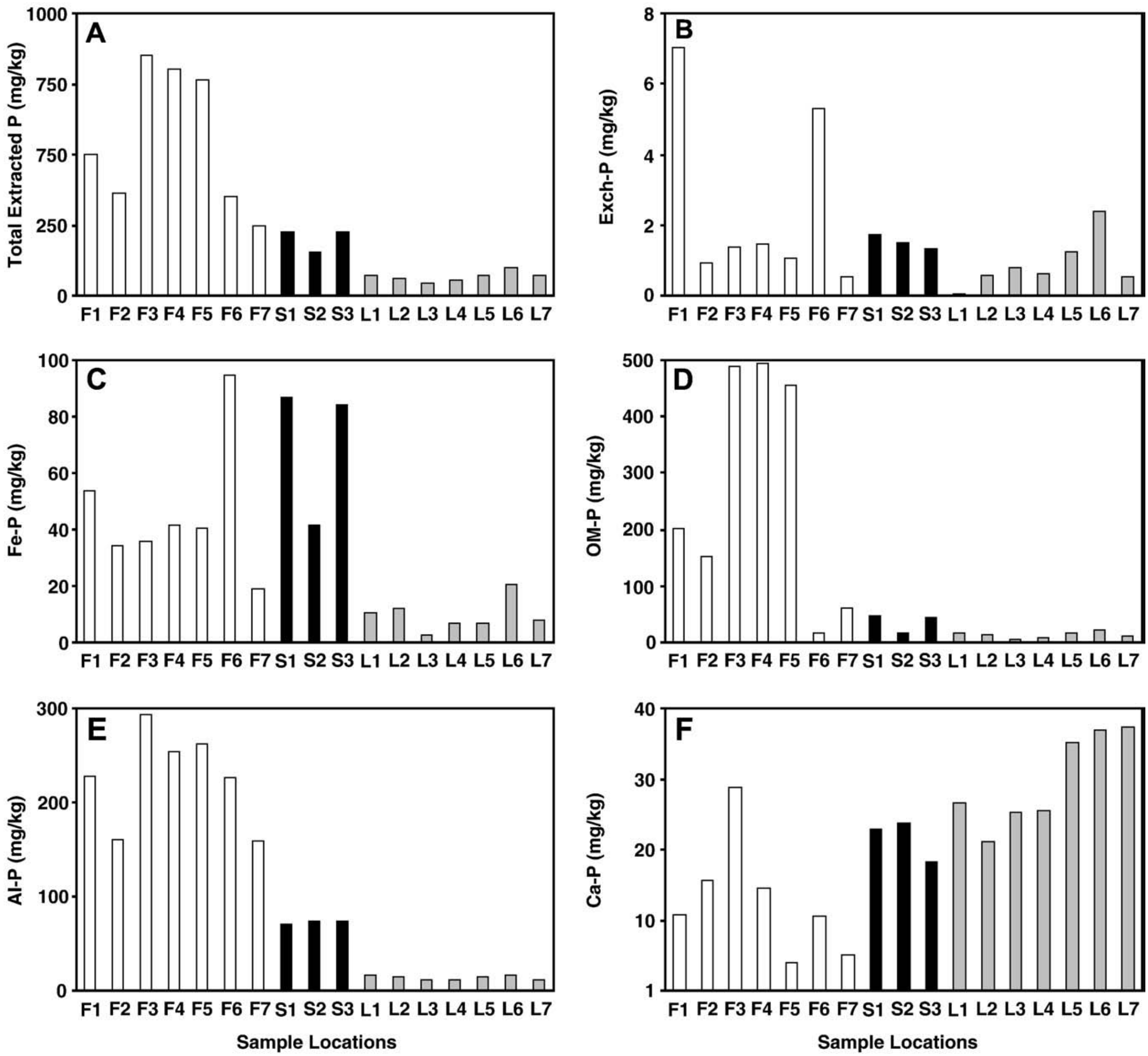

Fig. 2. Concentration of phosphorus in each fraction extracted from field soil (F; white bars) and stream (S; black bars) and nearshore (L; gray bars) sediment samples: (A) Total extracted phosphorus, (B) Exchangeable phosphorus, (C) Iron bound phosphorus, (D) Organically bound phosphorus, (E) Aluminum bound phosphorus, and (F) Calcium bound phosphorus. Total extractable phosphorus (TP) is the sum of each discrete fraction.

background electrolyte solution on a Beckman model DU640 spectrophotometer. For analysis purposes, these fractions represented operationally defined P fractions within the soil or sediment samples.

Statistical analyses

Statistical analyses were completed using the Mann–Whitney test, a nonparametric test, in Minitab 15. Samples were segregated into three populations based on similar environment: field soils ($n=7$), stream sediments ($n=3$), and nearshore lake sediments ($n=7$). Significance was defined as a probability (P) of <0.05.

Results

Results are presented in most cases for both the concentration of P within a fraction or as the percentage of the P in that fraction relative to the total P in the sample. For studies using sequential extraction procedures to determine element fractionation patterns, it is typical to present the data only as the fractions percentage of the total, as comparison of actual concentrations is difficult when comparing samples from different locations. In this case, however, we are concerned not only with the processes that transfer P from one fraction to another but also with the solubilization and removal of P from the solid phase and the amount of P that might be available for plant uptake. Therefore, we have presented both concentration and relative percent data to better elucidate the biogeochemical cycling of P within the system studied.

Extractable TP ranged from 244.93 to 848.98 mg P kg^{-1} (mean = 553.81 mg P kg^{-1}) in soil, 156.36 to 227.58 mg P kg^{-1} (mean = 202.28 mg P kg^{-1}) in stream sediment, and 45.16 to 98.56 mg P kg^{-1} (mean = 67.47 mg P kg^{-1}) in lake sediment (Fig. 2A, Tables 1 and 2). A Mann–Whitney test indicated a significant difference in TP between field soil and stream sediment, between field soil and nearshore sediment, and between stream and nearshore sediments (Fig. 3). The statistically significant drop in TP as material is transported from fields dominated by agriculture to nearshore sediments indicates either a removal of a major portion of the sediment or solubilization of P from the sediment to the water column.

Individual P fractions also indicate trends from field soils to stream and nearshore sediments. The most biologically available fraction (Exch-P) was generally observed with the lowest concentration (range: 0.03 to 7.01 mg P kg^{-1}) (Table 1). The mean Exch-P in field soil, stream sediments, and lake sediments was 2.52, 1.52, and 0.88 mg P kg^{-U1} (Table 2), respectively. A downward trend is suggested from field to stream to nearshore sediment (Fig. 2B) but is not statistically significant. When viewed as a percentage of TP (Fig. 4A), the trend is reversed with Exch-P making up a greater percentage of lake sediment TP than field soil or stream sediment.

The Fe-P fraction, or P extracted under reducing conditions, is relatively stable under oxidizing conditions, such as those likely encountered in the shallow sediment sample depth used here, with bioavailability less than Exch-P. However, high organic matter content may create locally reduced conditions at times. Under reducing conditions, Fe oxyhydroxides will become soluble releasing any associated P to the aqueous phase. The Fe-P fraction in field soils and stream sediment was similar with average values of 45.54 and 70.80 mg P kg^{-1} (Table 2, Fig. 2C). Nearshore lake sediments were significantly lower with an average concentration of 9.63 mg P kg^{-1}. As a percentage of the extractable TP (Fig. 4B), stream sediment was the highest with Fe-P constituting 35.00% of the extractable P on average and significantly higher than either the soil or lake sediment. The stream sediment possibly represents the most oxidizing environment of the three.

The OM-P fraction constitutes P associated with organic phases in the soil or sediment. This P fraction will slowly become bioavailable as the organic matter undergoes natural decomposition. The field soils and stream sediments had similar organic matter content (mean OM-LOI = 3.59% and 3.30%, respectively) while the nearshore sediments were significantly lower in organic matter (mean OM-LOI = 1.05%). The organic matter content had some impact on the concentration of OM-P (Fig. 2D, Table 1) with field soils having the highest mean, although quite variable, concentration (mean = 267.09 mg P kg^{-1}).

Table 1
Total extractable phosphorus (TP), aluminum bound phosphorus (Al-P), organically bound phosphorus (OM-P), calcium bound phosphorus (Ca-P), iron bound phosphorus (Fe-P), exchangeable phosphorus (Exch-P) and organic matter loss on ignition (OM-LOI) for field soils (F), streams (S) and nearshore lake sediments (L) of Graywood Gully and Conesus Lake.

Site	Exch-P	Fe-P	Al-P	OM-P	Ca-P	TP	OM-LOI (%)
F1	7.01	53.48	227.12	200.99	10.85	499.45	2.80
F2	0.91	34.03	160.47	152.12	15.54	363.07	3.62
F3	1.38	35.86	292.79	490.25	28.69	848.98	2.18
F4	1.48	41.37	253.29	493.78	14.53	804.45	2.14
F5	1.04	40.42	262.50	456.07	3.95	763.98	3.74
F6	5.28	94.84	225.77	15.31	10.63	351.83	3.09
F7	0.52	18.75	159.57	61.12	4.96	244.93	7.53
S1	1.73	86.64	70.07	46.31	22.84	227.58	3.84
S2	1.49	41.41	74.32	15.47	23.68	156.36	2.20
S3	1.33	84.35	73.95	45.07	18.21	222.91	3.87
L1	0.03	10.53	16.09	15.89	26.57	69.11	1.34
L2	0.56	11.99	14.13	14.17	21.11	61.95	1.94
L3	0.81	2.83	11.24	5.07	25.21	45.16	0.44
L4	0.64	6.79	11.76	9.37	25.50	54.06	0.95
L5	1.23	6.67	14.29	15.75	35.22	73.14	0.81
L6	2.38	20.71	16.44	22.06	36.97	98.56	0.97
L7	0.51	7.90	12.08	12.37	37.46	70.32	0.90

Total extractable P is the sum of all extractable fractions. Phosphorus fractions are reported as mg P kg^{-1}.

Table 2
Average phosphorus fraction (mg P kg^{-1}) and fraction percent (%) of the total extractable phosphorus (TP) of aluminum bound phosphorus (Al-P), organically bound phosphorus (OM-P), calcium bound phosphorus (Ca-P), iron bound phosphorus (Fe-P) and exchangeable phosphorus (Exch-P) for field soils, streams, and nearshore lake sediments of Graywood Gully and Conesus Lake.

	Ca-P	Exch-P	Al-P	Fe-P	OM-P	TP
(mg P kg^{-1})						
Field soil	12.74	2.52	225.93	45.54	267.09	553.82
Stream	21.58	1.52	72.78	70.80	35.61	202.39
Nearshore	29.72	0.88	13.72	9.63	13.52	67.47
(% Composition)						
Field Soil	2.3	0.5	40.8	8.2	48.2	100.0
Stream	10.7	0.8	36.0	35.0	17.6	100.0
Nearshore	44.0	1.3	20.3	14.3	20.0	100.0

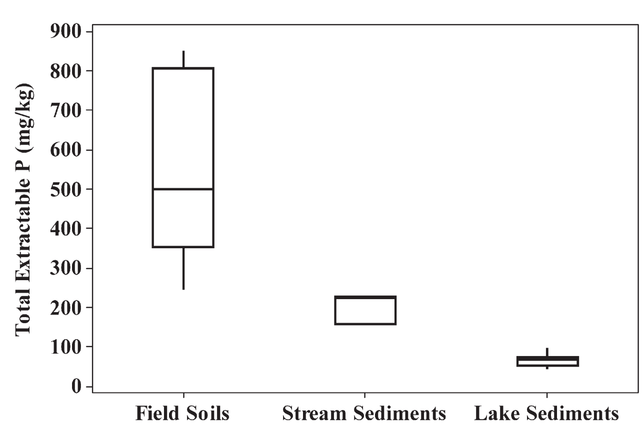

Fig. 3. Boxplot showing the median, first and third quartile, and range of total extractable phosphorus (TP) for field soils, stream sediment, and nearshore lake sediment.

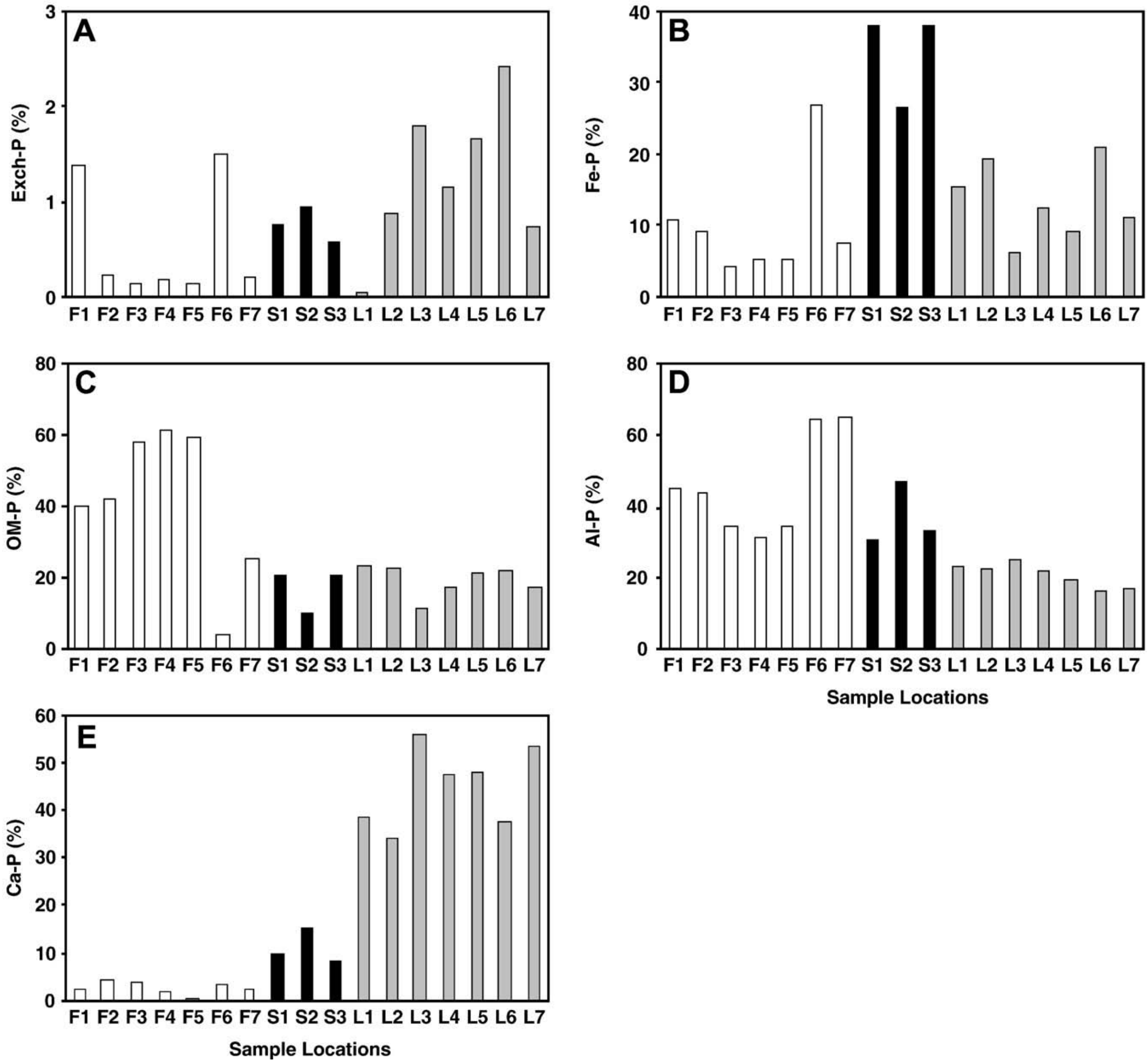

Fig. 4. Concentration of phosphorus in each fraction in field soil (F; white bars) and stream (S; black bars) and nearshore (L; gray bars) sediment samples given as a percent of the sum of all extracted P fractions: (A) Exchangeable phosphorus, (B) Iron bound phosphorus, (C) Organically bound phosphorus, (D) Aluminum bound phosphorus, and (E) Calcium bound phosphorus.

Stream and lake sediments had lower average concentrations of 35.62 and 13.52 mg P kg^{-1}, respectively, than field soils. Due to the highly variable soil concentrations likely due to management practices implemented during the study period (Herendeen and Glazier, 2009), and one sample being a forest soil, field soils were only significantly different from lake sediments. The same pattern was observed as a percentage of extractable TP basis (Fig. 4C).

The Al-P fraction is P associated with aluminosilicate clay minerals or possibly as discrete aluminum phosphate phases. This P fraction is not readily bioavailable as phosphate tends to form strong bonds with Al. The concentration of Al-P differed significantly (Fig. 2E) from field soils (mean = 225.93 mg P kg^{-1}) to stream and nearshore sediments (mean = 72.78 mg P kg^{-1} and 13.72 mg P kg^{-1}). As percent of TP (Fig. 4D), a similar trend was observed, but the difference between field soils and stream sediments was not significant (P>0.05). This was likely due to TP following a similarly decreasing trend.

The Ca-P fraction is P associated with calcium carbonate or as discrete calcium phosphate phases. This is the most stable and least bioavailable form of P in the sediment barring significant decreases in

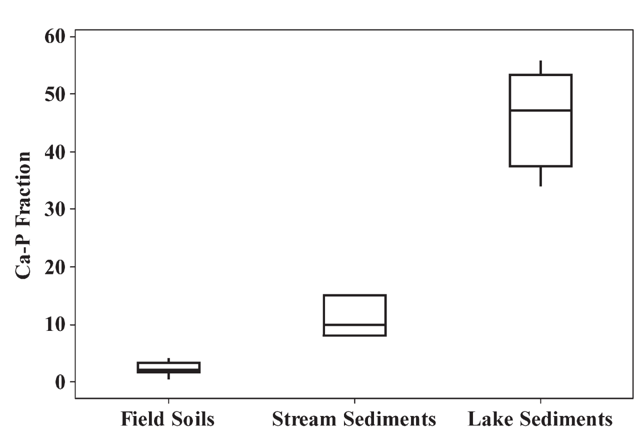

Fig. 5. Boxplot showing the median, first and third quartile, and range of values of the Ca-P fraction as a percent of total extractable phosphorus for field soils, stream sediment, and nearshore lake sediment.

pH. Although a trend along the field/stream/nearshore continuum was observed in the concentration of P in the Ca-P fraction, only the field soil and lake sediment were found to be significantly different. Average concentrations for field soils, stream and lake sediments were 12.74 mg P kg^{-1}, 21.58 mg P kg^{-1}, and 29.72 mg P kg^{-1}, respectively (Table 1, Fig. 2). As percent of TP, a significant trend was observed with the Ca-P percent increasing through the continuum from field soils to lake sediments (field soils 2.3%, stream sediment 10.7%, and nearshore lake sediment 44.0%, Fig. 5).

Discussion

This case study shows the influence of biogeochemical cycling and fractionation processes that occur during the export of P from agricultural soils, through a hard-water stream system with final deposition in a nearshore lake environment. Furthermore, the impact of the biogeochemical cycling of P on availability as a nutrient for plant growth in the nearshore environment may be significant when considering future nutrient management BMPs.

As soil/sediment was transported from the field environment to the aquatic habitat within the watershed, TP decreased from soil in an agricultural field (553.8 mg P kg^{-1}) to stream sediment (202.4 mg P kg^{-1}) to nearshore lake sediment (67.47 mg P kg^{-1}) (Table 2). While the TP concentration is of concern, P fractionation may also play a more important role in determining the impact of P on the system as some fractions are available for biotic uptake while others sequester P in relatively immobile phases. Major differences in the composition of the P fractions were observed along the field soil, stream sediment, and nearshore continuum. The Al-P and OM-P fractions were dominant in the field soils representing 89% of the TP but not in the stream and nearshore lake sediments. In the aquatic environments, the dominance of OM-P and Al-P fractions decreased (stream = 53.6%, nearshore 40.3%) while Fe-P became more important, especially in the stream environment (35.0% of TP, Table 2). Unlike the field environment where Ca-P represented only 2.3% of the TP, the influence of the hard-water environment became apparent as Ca-P became the dominant P fraction (44.0% of TP) in the nearshore environment. Fig. 6 summarizes major changes in P fractionation that occurred during the erosion, transport, and deposition. These changes may be explained considering physical and chemical processes that occurred in each specific environment within the continuum, where processes altered both the total concentration of P and the P fractions that dominated in each environment.

Field soils were dominated by organic bound P which was significantly higher in concentration than either stream or lake sediments. Considering the long history of manure application within the watershed (Herendeen and Glazier, 2009), these results are not surprising. The downward trend in OM-P from field soil to sediments was likely due to organic particles remaining suspended (Walling et al., 2001; Evans et al., 2004) and slow decomposition of organic matter. This is supported by the reduction in OM-LOI observed between field soils and lake sediments. The persistence of OM-P within the nearshore sediments was likely due to annual contribution from the macrophyte beds. This creates a slow cycling of P in the nearshore sediment–water system.

Aluminum bound P is a dominant phase within the soil as would be expected due to the reactivity of phosphate with aluminosilicate clays. During transport, the Al-P fraction is found to decrease. Considering the Al-P phase to be relatively stable chemically (Rydin, 2000), the reduction in both Al-P concentration and percent of TP between field soil and lake sediments suggests a physical process by which fine silt and clay eroding from the watershed soils is sorted, remained in suspension (Walling et al., 2001; Evans et al., 2004), and is likely deposited in deeper portions of the lake. However, the influence of a large macrophyte bed just north of the stream outlet (Bosch et al., 2009b) is to seasonally act to trap fine silt and clay particles (Li et al., 2007). If reduced growth of the macrophyte beds continued (Bosch et al., 2009b), the ability of the beds to trap P-rich fine particles will be reduced as well and reinforces the decline in growth of macrophytes.

Co-precipitation of P with calcite or the precipitation of discrete Ca phosphate phases may occur in a hard-water system (Walling et al., 2001; House, 2003; Evans et al., 2004). Similarly in the Graywood Gully system, the Ca-P phase increased significantly once soil particles were introduced to the hard-water stream and lake system. This mechanism is effective at limiting the bioavailability of P barring significant changes in pH. The observed trend suggests that reactions resulting in the sequestration of P in the Ca-P phase begin within the

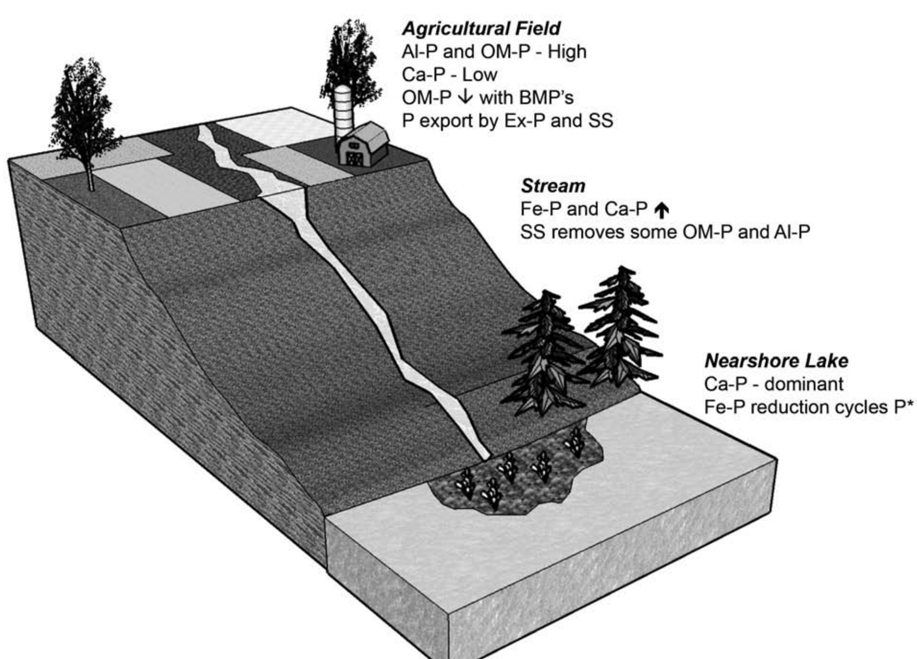

Fig. 6. Schematic diagram of the field soil-stream sediment-nearshore lake sediment continuum indicating the major changes in P content and fractionation within each subsystem.

stream, accelerating once sediment is deposited in the nearshore environment. Assuming that the Ca-P fraction is the most stable in the nearshore environment and that a continuation in reduced loadings of suspended sediment and its associated TP load continues as a result of management practices within the watershed (Makarewicz et al., 2009), a decrease in the availability of P for macrophyte growth should occur with time as other bioavailable fractions decrease and reinforce the decline in macrophyte growth.

Iron bound P was observed to increase in the stream environment. Soils rich in organic matter from manuring may maintain some Fe in reduced form as the organic material mineralizes, locally consuming oxygen (e.g., Bartlett, 1998). Once eroded and introduced into the stream system, the oxygen-rich surface water would quickly oxidize any reduced Fe. The Fe oxyhydroxides that may form would scavenge SRP from the water column. Once deposited in the nearshore lake sediments, a complex set of biogeochemical reactions controls cycling and bioavailability of P in the sediment, with the influenced biota possibly playing a significant role (Wigand et al., 1997; Kisand and Noges, 2003). The nearshore environment may retain the ability to supply both soluble forms of P to the water column, given the persistence of Fe-bound P in the sediment. Bosch et al. (2009a) have observed a decrease in metaphyton growth coincident at Graywood Gully with decreased P flux to the lake. This suggests that while Fe-P may be solubilized from the sediment, its contribution alone to the water column is not sufficient to support metaphyton growth. Despite reductions in phosphorus from the watershed within 2 years of establishing BMPs (Lewis and Makarewicz, 2009; Makarewicz et al., 2009), macrophyte biomass responded with a reduction in biomass but at a much slower rate than metaphyton (Bosch et al., 2009b). This lag in response to reductions in limiting nutrients from the watershed may be related to the Fe-bound P fraction contributing P to rooted macrophyte growth, delaying a response to the implemented BMPs. Significant reductions in macrophyte growth during the last year of the study (Bosch et al., 2009b) may be the result of the decreased suspended sediment input from Graywood Gully to the lake (Makarewicz et al., 2009). Li et al. (2007) indicated that the macrophyte beds will sufficiently decrease water velocities enough to allow fine sediment deposition. As with the Al-P phase, continued fine sediment deposition effectively fertilizes the macrophyte beds with P rich material.

Overall, a reduction in TP was observed through the field soil to lake sediment continuum. As the various mechanisms that influence the transport and fractionation of P within soils and sediments are considered, it is apparent that both physical and chemical processes played a role in P export and bioavailability. A significant amount of P was lost from field soils as they were transported and deposited, as is indicated from results of this study and from the long-term monitoring completed by Makarewicz et al. (2009). This P has either become available to biota or deposited in deeper portions of the lake system. Ultimately, the impact of P export on the nearshore lake environment may be influenced by the changes in P fractionation that occurred during transport and deposition and by the influence of biota on the biogeochemical cycling of P in the sediment. In addition, the implementation of BMPs within the watershed (Makarewicz et al., 2009; Herendeen and Glazier, 2009) is likely to play a role in shifting the fractionation patterns even further, as less P is introduced into the agricultural soils and as P is cycled within the nearshore system.

With BMP implementation in Graywood Gully, a significant decrease in nutrients and suspended solids delivered to the downstream system has occurred (Makarewicz et al., 2009) which has resulted in a decrease in metaphyton biomass (Bosch et al., 2009a) and bacteria (Simon and Makarewicz, 2009) and eventually a significant decrease in macrophyte biomass (Bosch et al., 2009b). The resultant decrease in rooted macrophytes may play a pivotal ecological role in the nearshore zone by accelerating the decrease in total macrophyte biomass. As the beds increase in size and biomass, they tend to reduce the velocity of the stream water entering the nearshore (Li et al., 2007) allowing P-laden suspended particles to drop out prematurely and in essence fertilize the beds. Conversely, as the macrophyte beds decrease in size and biomass, a smaller percentage of P-laden suspended sediment carried by stream water is deposited within the macrophytes, further decreasing the potentially plant available P. Combined with the likely reduction in the organic fraction of P from the watershed due to implementation of BMPs (Makarewicz et al., 2009), these two factors may serve to accelerate the decrease in macrophyte loss within the Graywood Gully system with time.

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Impacts of manure management practices on stream microbial loading into Conesus Lake, NY

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ABSTRACT

The microbiology of stream water has a seasonal component that results from both biogeochemical and anthropogenic processes. Analysis of nonevent conditions in streams entering Conesus Lake, NY (USA), indicated that total coliform, *Escherichia coli*, and *Enterococcus* spp. levels peak in the summer in all streams, independent of the agricultural use in the stream sub-watershed. Prior to implementation of management practices, *E. coli* in water draining Graywood Gully, a sub-watershed with 74% of the land in agriculture, reached as high as 2806 CFU/100 mL, exceeding the 235 CFU/100 mL EPA Designated Bathing Beach Standard (EPA-DBBS). In contrast, North McMillan Creek, a sub-watershed with <13% of its land in agriculture, had *E. coli* maxima generally near or below the EPA-DBBS. Graywood Gully at times had a higher microbial loading than North McMillan Creek, a sub-watershed 48 times larger in surface area. Over a 5-year study period, there was a major decrease in bacterial loading during nonevent conditions at Graywood Gully, especially after manure management practices were implemented, while bacterial loading was constant or increased in streams draining three other sub-watersheds. *E. coli* levels dropped more than 10 fold to levels well below the EPA-DBBS while the yearly maximum for *Enterococcus* dropped by a factor 2.5. Similarly, exceedency curves for both *E. coli* and *Enterococcus* also showed improvement since there were fewer days during which minimum standards were exceeded. Even so, Graywood Gully at times continued to be a major contributor of *E. coli* to Conesus Lake. If wildlife represents a significant source of indicator bacteria to Graywood Gully as has been reported, stream remediation, management efforts and compliance criteria will need to be adjusted accordingly.

Introduction

Bacterial levels exceeding Federal, State and Provincial standards occur at beaches throughout the Great Lakes basin (Great Lakes Commission, 2005). Nearshore, river, and embayment recreational water quality is often impaired because of microbiological contamination, and beaches are closed out of concern for public health. Sources of microbiological contamination to the Great Lakes and lakes in general are many, including combined or sanitary sewer overflows, unsewered residential and commercial areas, failing private household and commercial septic systems, fecal coliforms from animal/pet fecal waste, and wildlife waste (Great Lakes Commission, 2005). Another source is agricultural runoff, especially manure. Manure is an agricultural by-product that is usually returned to the land to enhance soil productivity, increase soil organic matter, and increase infiltration rates (Gilly and Risse, 2000; McDowell et al., 2004; Smith et al., 2007). However, if improperly applied or applied in excess, manure contaminants can pollute adjacent waterways and infiltrate into groundwater (Zebarth et al., 1996).

Conesus Lake, one of the Finger Lakes of New York State, has microbial problems similar in many ways to the Great Lakes. Levels of indicators of microbial pollution are at times well above the levels (SOCL, 2001) recommended by the EPA for bathing or even casual contact with the water (USEPA, 2000). Besides microbial problems, this eutrophic lake (Forest et al., 1978; Makarewicz, 2009) has nuisance algae, invasive aquatic weeds, and large populations of zebra mussels. Lake stakeholders have concerns about the water quality at local beaches and at the shoreline cottages where residents swim and children play in the shallows (SOCL, 2001). In addition, Conesus Lake has a New York State Department of Environmental Conservation (DEC) Classification of AA, serves about 20,000 local residents both as a recreational resource and as a source of drinking water, and is a focal point for regional tourism (NYSDEC, 2006).

Water enters the lake from the surrounding sub-watersheds throughout the year as nonevent (baseline) flow and during ~13 to 15 annual events that are caused by significant rainfall or snowmelt conditions. During events, massive amounts of water and materials, including fecal pollution, are transferred to the lake in a short period of time (Simon and Makarewicz, 2009). Fecal pollution enters the lake

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from several sources, including surrounding farms, some of which house large numbers of cattle. A perimeter sewer system collects waste from homes surrounding the lake, and leaks from this system are a possible source of pollution (SOCL, 2001). Nearly one-thousand homes are set back from the main road and from the perimeter sewer system and have free-standing septic systems that with age and improper management can also act as a microbial source. Finally, there is a large wildlife population in the area ranging from deer to birds such as ducks and geese. Each of these species introduces fecal material that can make its way into streams draining sub-watersheds (Somarelli et al., 2007).

In an effort to improve the quality of the water entering the lake from the sub-watersheds, we have worked closely with farmers on nutrient and animal waste management (Herendeen and Glazier, 2009) to reduce the levels of fecal contaminants leaving farms and being transported to the lake. We view the Conesus Lake catchment system as an excellent surrogate system for a Great Lakes watershed. The lake's catchment has multiple small sub-watersheds within a few kilometers of each other for convenient sampling, are primarily in agriculture, and are owned or operated by one or two farms that allow some control of land use (Makarewicz et al., 2009). In addition, the large number of small watersheds allowed evaluation of the effects of different agricultural management systems on the loads of nutrients and fecal pollution in the streams that drain into the lake. Because of the steep-sided slopes of the sub-watersheds, water transit times were short, and changes in conditions are rapidly reflected in the water quality draining the sub-watershed.

Here we evaluate the seasonal and spatial abundance of microbial populations during hydrometeorological nonevent periods in four streams draining sub-watersheds mostly in agriculture. Nonevent periods are characterized by hydrologic and material export conditions that differ significantly from that of storm flows (Pionke et al., 1999). From a potential pathogen perspective, nonevent flow represents the conditions in the stream (probably more than 300 days a year) to which humans are actually exposed and for which there are State and National exposure limits for “indicator bacteria” (USEPA, 1986; NYSDEC, 2006). Our goal was to test the hypothesis that elevated levels of bacteria during nonevent flows were due to poor manure practices, and finally to determine the extent to which manure management could reduce microbial loading into downstream aquatic systems.

Methods

Implementation of BMPs related to microbiology

Six sites were chosen as study sub-watersheds based on several criteria (Makarewicz et al., 2009). Here we focus not only on the Graywood Gully sub-watershed but also provide comparative data on three other sub-watersheds: Long Point Gully, Sutton Point Gully, and North McMillan Creek (Fig. 1). The Graywood and Long Point sub-watersheds had resident populations of dairy cows, while North McMillan Creek is primarily a forested watershed and portions of Sutton Point Gully are in row crops. Graywood Gully is one of the smallest catchments (38 ha) in the Conesus Lake watershed. Land use is mostly in agriculture (74%), consisting of a single dairy-farm operation with approximately 100 head of cattle and row crops including corn and beans. Starting in the fall of 2003, “Whole Farm Planning” has been instituted at Graywood Gully, and a myriad of structural and cultural Best Management Practices (BMPs) aimed at controlling nutrient and animal waste pollution were implemented based on soil testing, evaluation of the P index, and common agricultural management practices (Herendeen and Glazier, 2009). Changes implemented were designed to improve both the nutrient and microbial characteristics of the runoff from the dairy farm to Conesus Lake, the ultimate recipient of the runoff. At Graywood Gully, many of the BMPs controlled water movement from the farm, kept cows out of streams, and limited the spreading of manure. The BMPs included: the installation of 20,000 subsurface drainage construction tiles (6250 m); the addition of a standpipe and watering source for a heifer pasture area; the fencing of cattle to prevent them from entering the stream; roof water separation allowing for the clean water to stay clean and be safely discharged away from the contaminated barnyard areas; and finally, the elimination of winter but not spring, summer and early fall spreading of manure (Herendeen and Glazier, 2009).

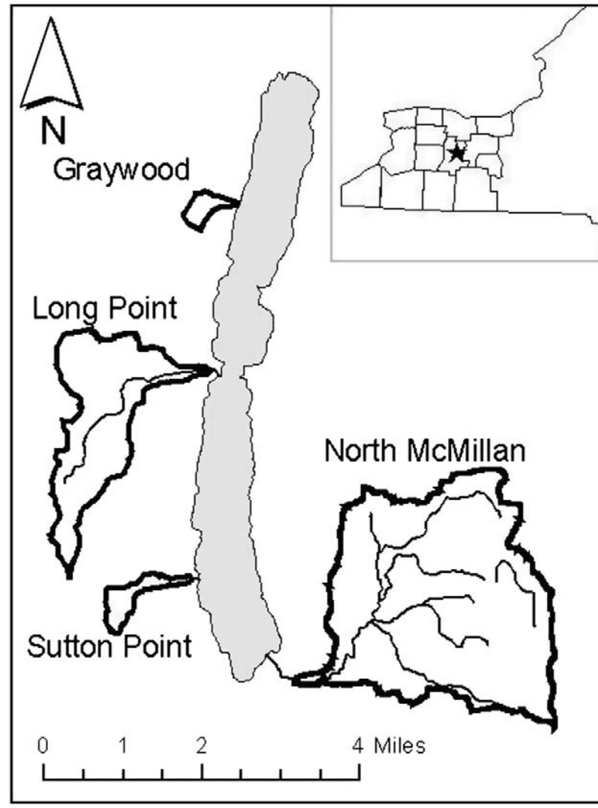

Fig. 1. Conesus Lake sub-watersheds used in this study and their location in Western New York.

At Long Point Gully, the one dairy in this sub-watershed ceased operations and dairy cattle were removed from the sub-watershed in 2003. Additionally, a 37% reduction (76.7 ha) in crop acreage occurred by 2004, although manure spreading continued on the land through 2007. No physical infrastructure improvements were implemented in this watershed until 2007 when gully plugs were added at the end of the project. At Sutton Point Gully a significant and increasing portion of the sub-watershed has been in alfalfa/grass production since 2002 (37% in 2003 to 60.3% in 2006). This indicated that manure slurry was not added to the sub-watershed during the study period. At North McMillan Creek only 12% of the sub-watershed was in agriculture and over 77% was in vacant land, in abandoned land that included agricultural parcels in early forest succession, or in single family use (SOCL, 2001). No management practices were implemented in this sub-watershed in this study.

Stream sampling

All streams were monitored continuously for five annual cycles with a differential pressure transducer (ISCO 720) attached to an ISCO continuously recording flow meter (Model 6700) equipped with an automatic sampler from 1 Sep 2002 to 31 Aug 2007 (Makarewicz et al., 2009). As defined in an accompanying study by Makarewicz et al. 2009, a Water Year (WY) is the period from 1 Sep to 31 Aug of the following year. For example, WY 1 extended from 1 Sep 2002 to 31 Aug 2003, WY 2 extended from 1 Sep 2003 to 31 Aug 2004, etc. Water samples were taken using two different methodologies: weekly manual grab samples and automated hydrometeorological event samples (Makarewicz et al., 2009). A total of 5 water years of daily discharge data was collected on all creeks starting on 1 Sep 2002 and completed on 31 Aug 2007 (Makarewicz et al., 2009). Most of the time there was low flow in the stream — defined as a nonevent period. Hydrometeorological events were associated with dramatic changes in stream water level and were defined as a rise in the creek level of 2.54 cm in 30 min. After reaching a discharge peak, the end of an event was defined by a leveling off of the descending limb of the stream hydrograph (Makarewicz et al., 2009).

Stream temperature was measured *in situ* (Fisher Traceable Thermometer) weekly during nonevent conditions. Simultaneously, water samples were taken for both turbidity and microbial analysis within an hour of each other. Water was transported to the laboratory at SUNY Geneseo and processed for microbial measurement within 6 h. Turbidity was determined using a portable turbidimeter (Orbeco-Hellig Model 966).

The microbial quality of water was measured using surrogates for fecal pollution. New York State uses fecal coliform count levels as the standard for recreational waters and bathing beaches (Public Health Law §225, Chapter 1 State Sanitary Code Subpart 6-2). However, the EPA guidelines for water quality recommend the measurement of *Escherichia coli* and *Enterococcus* levels since they provide the best correlation to the presence of water-related human gastrointestinal disease (USEPA, 1999, 2000). In this study, both *E. coli* and *Enterococcus* levels were routinely measured in all samples. Additionally, the general microbial composition of stream waters was measured by total coliform and total heterotrophic counts (APHA, 1999).

Since most stream water flowing into Conesus Lake have low turbidity, samples were analyzed for total coliform and *E. coli* (CFU/100 mL) using a membrane filter (MF) method employing m-ColiBlue24 (Millipore®) medium (Grant, 1997). *Enterococcus* levels (CFU/100 mL) were determined using a MF method by placing filtered dilutions of water samples on m-Enterococcus Agar (Difco 0746) (Kaneko et al., 1989). Total heterotrophic bacteria were measured following growth on R2A medium (Difco 1826) at 25 °C for 48 h. A method of "spot plating" was used to facilitate these measurements, as multiple 20-μL samples were spotted on R2A agar plates, and micro-colonies were observed under a dissecting microscope after 48 h. This method allows for quantitation of bacterial numbers with many fewer plates and in a shorter time than when sample dilutions are merely spread on plates of growth medium.

Data analysis

Comparison of the temperature differentials between streams assumed that the differences in temperature on any given day were random allowing Chi-square analysis. Only nonevent microbial data are presented here. Event data are presented in Simon and Makarewicz (2009). Microbial data were analyzed directly without transformation. To dampen the effect of extremes typically associated with intra-sample and analytical variability of microbial populations, comparisons were made using monthly geometric means as recommended by the EPA (USEPA, 1999). Most microbial counts showed seasonal periodicity and peaked between July and September or October (see below). Therefore, microbial analysis was done on a calendar year basis because a water year basis would combine numbers from half a seasonal peak in 1 year with numbers from half a seasonal peak of the following year. Linear regression analyses were carried out to examine the trends in bacterial loading over time and to determine relationship between loading of suspended solids and bacteria. (SigmaStat 3.5, SYSTAT Software Inc.).

Microbial "exceedency" curves were constructed for *E. coli* and *Enterococcus*. Exceedency curves allow for the evaluation of the percentage of time in any given period that microbial levels exceed a particular value. In such a curve, the comparison is as a percentage of time, and each microbial measurement is assumed to be held for a period of time between pre- and post sample, in this study usually 7 days. Because events may only last for a short period of time, only nonevent data were employed in determining an exceedency curves. To develop such a curve, the bacterial levels were arranged from highest to lowest, and then each value was multiplied by the time period in days that it represented. Time weighted averages were not necessary because the microbial sampling was done at regular weekly intervals. A calculation was done to determine what fraction of the total time (1 year) each level represented, and then *E. coli* and *Enterococcus* levels (CFU/100 mL) were plotted against the percent of the time that any particular microbial level was exceeded. For a complete discussion of exceedency curves, see the National Center for Water Quality Research (NCWQR, Richards and Baker, 1993).

Results

Stream hydrology

Seasonal weather in western New York has large fluctuations in temperature and rainfall (snow) that dominate stream hydrology. Nonevent flow was generally lower in the summer and higher in the spring (Fig. 2a). Streams have liquid water even during the coldest winter months with flow occurring under the ice. Fig. 2b gives the monthly nonevent flow in Graywood Gully as a fraction of the total flow and shows that the 4-year average of nonevent flow provided 77% of the total yearly water load. This was quite variable over time, reflecting wet years versus dry years. In some months 90% of the water

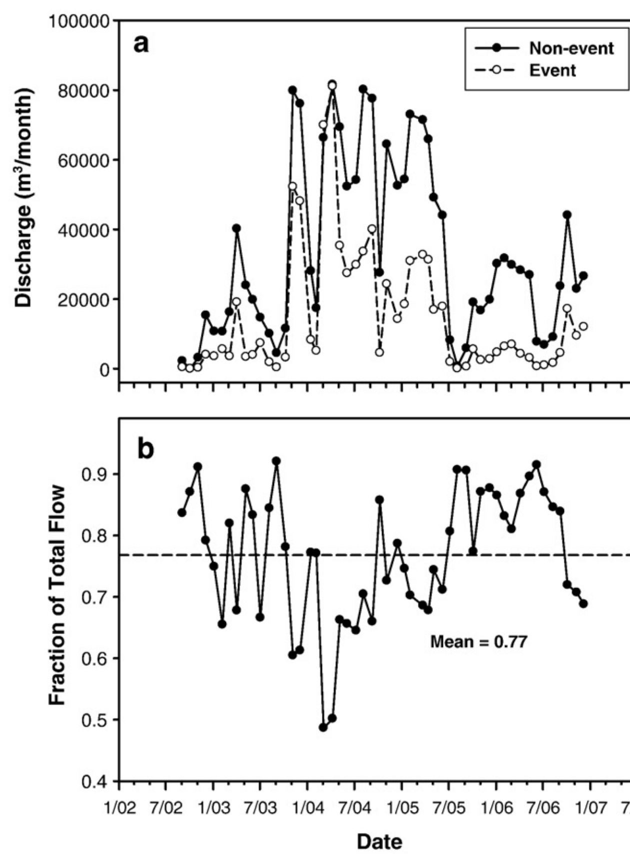

Fig. 2. Graywood Gully stream flow showing (a) nonevent and event discharge in m^3 per month and (b) monthly nonevent flow as a fraction of total flow.

flow was nonevent flow. In a small sub-watershed like Graywood Gully, stream flow may cease in years during limited rainfall as in 2002. However, 2004 was a very wet year and event discharges were significant relative to nonevent flow.

High variability in event versus nonevent discharge within a sub-watershed and between sub-watersheds was evident. In Graywood Gully, areal weighted nonevent discharge (m^3/ha) runoff was greater in all 5 water years than event discharge. In Sutton Point Gully, areal event runoff was higher in 1 of 5 years; in both Long Point Gully and North McMillan Creek it was higher in 2 of 5 years (Table 1). Even when and where event flow was the dominant yearly stream contribution to Conesus Lake, it was no more than 1.7 times as much water as the nonevent flow during the rest of the year. As Makarewicz et al. (2009) have suggested, areal weighted discharge can be much higher in Graywood Gully than in other study streams. For example, in WY 2 Graywood Gully had almost 3.6 times the discharge per ha than did North McMillan Creek. Especially in wet years, external sources outside of the traditional topographical definition of sub-watershed were likely impacting this measurement (Noll and Magee, 2009).

As expected, the seasonal variability in stream temperatures was typical of temperate regions with maxima in the summer and minima in the winter (Fig. 3a). However, a Chi-square test of the temperature differentials between various sub-watersheds indicated that Graywood Gully was significantly warmer than both Sutton Point Gully ($P<0.001$, d$f=164$) and Long Point Gully ($P=0.013$, d$f=131$) but not North McMillan Creek ($P=0.755$, d$f=164$) (Figs. 3b–d).

Microbiology

The *E. coli*, *Enterococcus*, and total coliform levels as indicated by the geometric monthly mean concentrations in the nonevent flow of all streams were seasonal; bacteria were generally present in highest numbers from June to September, with peak amounts in the month of August (Fig. 4). At the beginning of this study prior to the initiation of BMPs, Graywood Gully was a major source of microbial pollution. Abundances of *E. coli* in stream water during nonevents reached as high as 2806 CFU/100 mL during 2003 when management practices concerning the application of manure on snow were not in place. No other sub-watersheds experienced such high *E. coli* levels during 2003 (Table 2, Fig. 5). Long Point Gully, which still received manure application on fields after the closing of a dairy operation, had high and variable *E. coli* levels throughout the study period. North McMillan Creek, the reference sub-watershed with the lowest amount of land in agriculture (<13%), had maximum *E. coli* levels generally near or below the EPA's Designated Bathing Beach Standard (Table 2), while Sutton Point Gully, a sub-watershed in agriculture (76%) but with no dairy farms or manure application activities, had maximum *E. coli* levels only slightly over the EPA Beach criteria.

A small winter peak in Long Point Gully in 2003–2004 was also observed. No such peaks were found in Sutton Point Gully, and a small peak was found in North McMillan Creek. The large July 2005 peaks were associated with significantly elevated discharge rates at this time. With a few exceptions, heterotrophic bacterial levels did not change during the year and were often higher in the winter than they were in the summer (Fig. 4).

Over a 4-calendar year period, a major decrease in bacterial levels in nonevent Graywood Gully stream water was observed. *E. coli* levels in Graywood Gully dropped more than 10 fold to levels significantly below the 235 CFU/100 mL EPA Bathing Beach Standard (Table 2) while the yearly maximum for *Enterococcus* dropped by a factor 2.5. The box plot of monthly (June to September) range of values for *E. coli* and *Enterococcus* decreased over time (Fig. 5a). The decreases in the median values of *E. coli* ($r^2=0.823$) and *Enterococcus* ($r^2=0.546$) (Fig. 5b) in Graywood Gully contrasted with little to no change over the same time period for Long Point Gully, Sutton Point Gully, and North McMillan Creek. Graywood Gully was the only stream where the peak median for total coliforms dropped ($r^2=0.982$), whereas all other streams had increases in total coliform levels. These trends persisted in 2007.

Exceedency curves (Fig. 6a) also demonstrated a decrease in the levels of *E. coli* in Graywood Gully over the study period. In 2003, 33% of the samples taken were above the EPA Standard for infrequent contact, whereas by 2006 that number dropped to 20%. Additionally, in 2003 only 43% of the yearly samples met the EPA Beach Standard for *E. coli* (USEPA, 1999); this increased to 62% by 2006. The improvement in Graywood Gully, which began in 2003, was also seen in the decrease in samples that had the highest levels of *E. coli*. In 2003, 12% of the time the samples were above 10,000 CFU/100 mL, but by 2006

Table 1
Yearly nonevent and event discharge from Graywood Gully, Sutton Point Gully, Long Point Gully, and North McMillan Creek.

Year	Area (ha)	Watershed	Nonevent discharge (m^3)	Event discharge (m^3)	Ratio event/base	Nonevent discharge (m^3/ha)	Event discharge (m^3/ha)
WY 1	38	Graywood Gully	121,998	107,685	0.88	3210	2834
	68	Sutton Point Gully	148,360	58,188	0.39	2182	856
	588	Long Point Gully	477,296	1,228,372	2.57	812	2089
	1778	N. McMillan Creek	3,701,677	6,806,584	1.84	2082	3828
WY 2		Graywood Gully	616,429	436,795	0.71	16,222	11,495
		Sutton Point Gully	505,322	543,390	1.08	7431	7991
		Long Point Gully	2,919,546	239,498	0.08	4965	407
		N. McMillan Creek	6,605,063	7,594,317	1.15	3715	4271
WY 3		Graywood Gully	668,751	176,740	0.26	17,599	4651
		Sutton Point Gully	652,338	242,108	0.37	9593	3560
		Long Point Gully	2,133,648	2,504,593	1.17	3629	4260
		N. McMillan Creek	6,324,154	6,206,475	0.98	3557	3491
WY 4		Graywood Gully	214,823	65,058	0.30	5653	1712
		Sutton Point Gully	386,652	35,829	0.09	5686	527
		Long Point Gully	1,348,078	1,202,166	0.89	2293	2045
		N. McMillan Creek	4,068,563	2,294,813	0.56	2288	1291
WY 5		Graywood Gully	336,007	124,205	0.37	8842	3269
		Sutton Point Gully	553,187	85,312	0.15	8135	1255
		Long Point Gully	1,659,679	419,169	0.25	2823	713
		N. McMillan Creek	7,657,713	3,463,875	0.45	4307	1948

A water year (WY) was defined as the period from 1 September to 31 August of the following year.

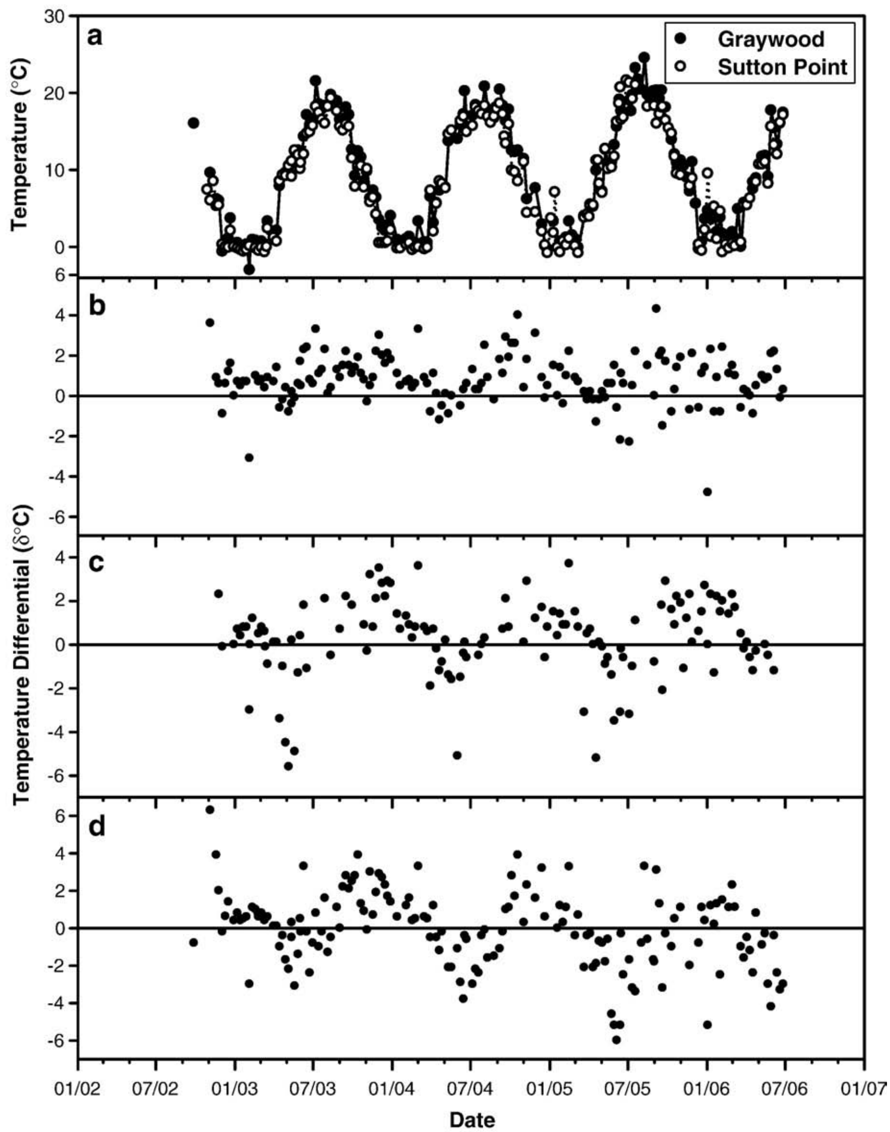

Fig. 3. Stream temperatures (°C) and temperature differentials (δ°C) in Graywood Gully and other sub-watersheds. (a) Weekly stream temperature in Graywood Gully and Sutton Point Gully temperature. Temperature differences between individual weekly samples in Graywood Gully and (b) Sutton Point Gully, (c) Long Point Gully, and (d) North McMillan Creek.

values this high occurred <4% of the time. *E. coli* in 2005 was an anomaly in that a few very high levels occurred. The majority of *Enterococcus* samples (<60%) remained at levels above the EPA recommendation for infrequent contact of 576 CFU/100 mL (USEPA, 1999) (Fig. 6b).

Microbial loading is similar in concept to nutrient loading into a lake. Rather than simply considering the microbial abundance per unit volume, we multiplied discharge of stream water times microbial concentration to obtain loading. Since WY trends in flow were used in these calculations, total microbial loading trends are presented as a function of water years. The loading of *E. coli*, *Enterococcus,* and total coliforms varied over the year because of the seasonality of microbial growth and stream flow. Higher stream flows sometimes occurred in late fall and early spring, times when indicator bacterial levels were low; but during wet years, summer rains often coincided with peak microbial numbers. On an areal (number per hectare) basis, Graywood Gully at times delivered *E. coli* to Conesus Lake that were more than an order of magnitude higher than the other three study streams (Table 3).

The relationship between stream microbiology from June to September (the time of peak indicator microbial levels) and turbidity was examined. In all cases, as turbidity increased, stream microbial numbers (*E. coli*, *Enterococcus*, total coliforms, and total heterotrophic bacteria) increased (r^2 from 0.262 to 0.576, Fig. 7).

Discussion

Both nonevent and event discharge contributed to water entering the lake. While it is impressive that up to 40% of the total yearly volume of water discharged into Conesus Lake originated from ~13 to 15 events (Makarewicz et al., 2009), nonevent flow did represent a substantial amount of water to the lake. In Graywood Gully, nonevent runoff exceeded event runoff in all years between WY 1 and WY 5 and accounted for 53% to 79% of the total discharge.

Analysis of the microbial characteristics of Conesus Lake sub-watersheds indicated that for nonevent conditions, total coliform, *E. coli*, and *Enterococcus* levels peaked in the summer in all streams, whether or not livestock were present; that is, the periodicity observed was independent of the particular agricultural use in the stream sub-watershed. For example, North McMillan Creek had only 10.2% agricultural land use and no livestock but had a similar

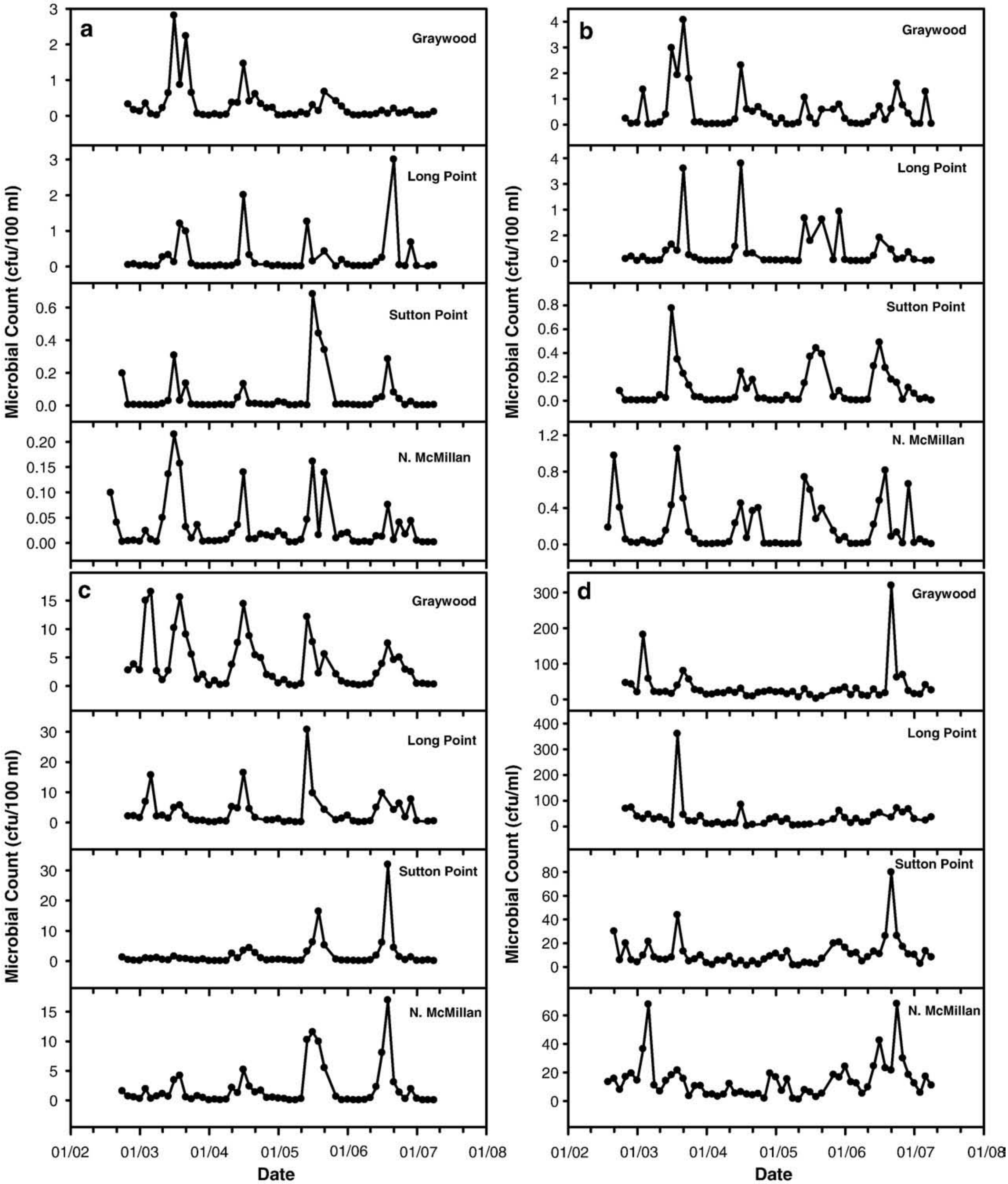

Fig. 4. The geometric mean of monthly microbial counts in Graywood Gully, Long Point Gully, Sutton Point Gully, and North McMillan Creek. (a) *Escherichia coli*, (b) *Enterococcus,* and (c) total coliforms are given in units of CFU/100 mL. Total heterotrophic bacteria (d) are in CFU/mL.

seasonal periodicity for *E. coli* and *Enterococcus* as Graywood Gully, a sub-watershed with 100 head of dairy cattle. *E. coli* derived from wildlife may be a major contributor to the normal stream microbial components and may help define this periodicity. Microbial source typing using Rep-PCR indicated that as many as half of the *E. coli* found in Conesus Lake streams were from wildlife sources (Somarelli et al., 2007). However, autochthonous soil microbial communities may include a component of *E. coli*, which is distinguishable genetically from the communities found in the gut of most common wild animals such as geese and deer (Byappanahalli et al., 2006; Ishii et al., 2006). These naturalized *E. coli* from Great Lakes watersheds also showed seasonal variability (Ishii et al., 2006) and thus may contribute to the overall *E. coli* levels found in streams.

Particulate material (total suspended solids, TSS) and microbial numbers rose dramatically during stream events (Richards et al., 2001; Makarewicz et al., 2009; Simon and Makarewicz, 2009). This was expected, as large volumes of moving water often transfer solids from land to stream as well as resuspend stream sediments, a known source of *E. coli* (Stephenson and Rychert, 1982; Jamieson et al., 2003, 2005). However, during nonevents there were both particulate matter and

Table 2
Yearly calendar maxima (CFU/100 mL) of *Escherichia coli* and *Enterococcus* (*Enter.*) in Graywood Gully, Long Point Gully, Sutton Point Gully, and North McMillan Creek.

Year	Graywood Gully		Long Point Gully		Sutton Point Gully		N. McMillan Creek	
	E. coli	*Enter.*	*E. coli*	*Enter.*	*E. coli*	*Enter.*	*E. coli*	*Enter.*
Y 1 — 2003	2806	4059	1200	3589	306	775	214	1048
Y 2 — 2004	1454	2298	2002	3788	131	234	140	449
Y 3 — 2005	668	1046	1257	1661	680	440	161	738
Y 4 — 2006	196	1592	3000	908	283	488	75	801

EPA standards for "Recreational Fresh Water", "Designated Bathing Beach", and "Infrequent Body Contact" are as follows: *E. coli*: 126, 235, and 576 CFU/100 mL, respectively; *Enterococcus*: 33, 62, and 151 CFU/100 mL, respectively (USEPA, 1999).

bacteria in Conesus Lake streams, and bacterial levels were positively correlated with water turbidity (Fig. 7). Meals (1989) also reported a significant correlation for bacteria and TSS in the LaPlatte Reservoir, Vermont. While the relationship between bacteria and particulates could be due to similar mechanisms being responsible for bringing them into suspension, it may simply be a reflection of the observation that bacteria in nature are generally bound to particulate matter and have higher metabolic rates when in this condition (Crump et al., 1999; Luef et al., 2007).

There can be little doubt that improvements in the microbial quality of the water in Graywood Gully occurred following the application of BMPs for manure management in the sub-watershed. The geometric mean monthly levels of *E. coli* and *Enterococcus* decreased over the study. The biggest drops occurred in the first years of the study, perhaps not unexpectedly, coming after changes in the patterns of manure spreading. By the end of the study, *E. coli* levels in Graywood Gully were below the "Designated Beach Area" standard of 235 CFU/100 mL set by the EPA. Although *Enterococcus* levels have decreased, they were still much higher than recommended (Table 2) (USEPA, 1999). Meals (1989) observed a significant decrease in fecal coliforms and *Streptococcus* in Vermont agricultural watersheds after changes in management practices, but to a lesser degree than observed here. Inamdar et al. (2002) observed a slight decrease in fecal coliforms and a larger decrease in fecal *Streptococcus* during a 10-year study on BMPs for manure management in the Piedmont region of Virginia.

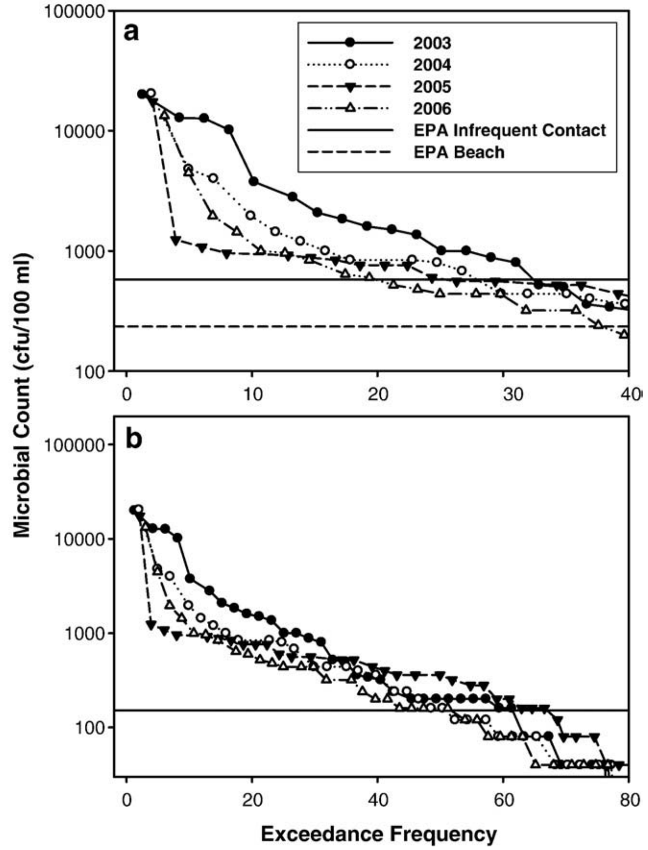

Fig. 6. Graywood Gully exceedency curves for (a) *Escherichia coli* and (b) *Enterococcus* for 2003 to 2006. EPA *E. coli* standards for "Infrequent Body Contact" and "Designated Bathing Beach" are 576 and 235 CFU/100 mL, while *Enterococcus* "Infrequent Body Contact" is 151 CFU/100 mL (USEPA, 1999).

Total coliform levels, a broader measure of the presence of Gram-negative microbial communities, have decreased in Graywood Gully waters but increased in the Long Point, Sutton Point, and North McMillan streams. There was no clear reason why the total coliform

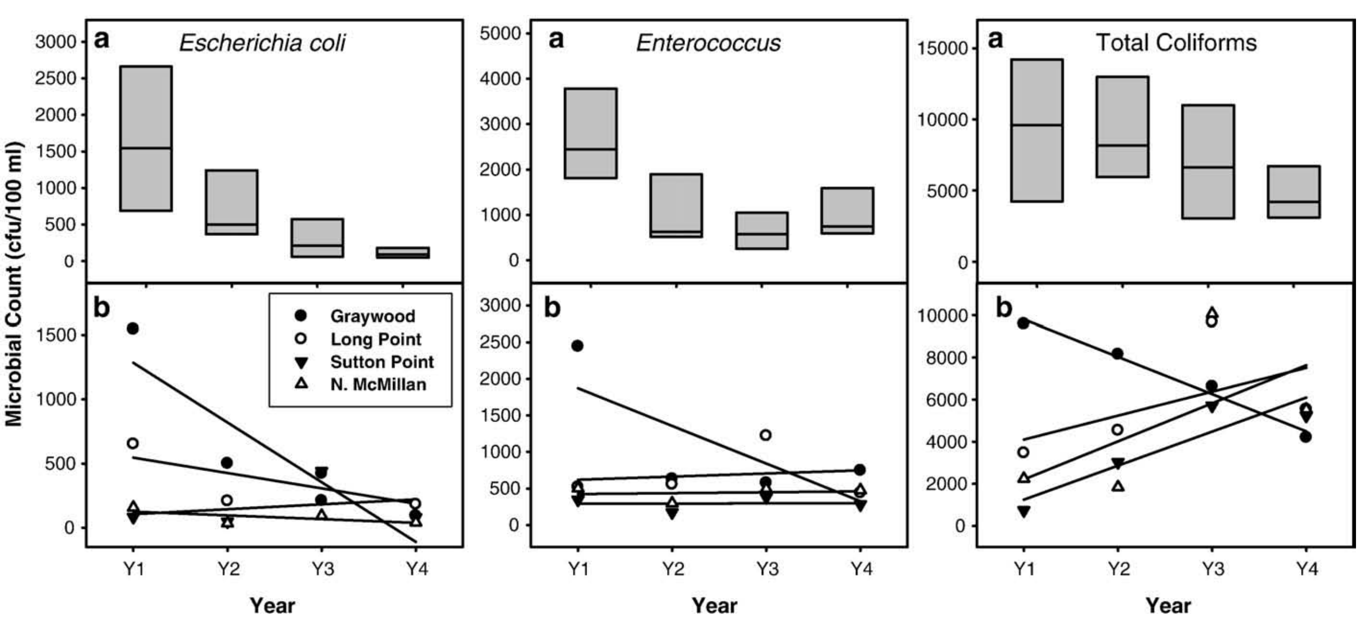

Fig. 5. Graywood Gully *Escherichia coli*, *Enterococcus*, and total coliform levels 2003 to 2006. (a) Range and median for the peak months of bacterial abundance, June to September. (b) Annual median values for Graywood Gully, Long Point Gully, Sutton Point Gully, and North McMillan Creek.

Table 3
Average *Escherichia coli* loading (CFU/month) and areal weighted *Escherichia coli* loading (CFU/ha/month) from Graywood Gully, Long Point Gully, Sutton Point Gully, and North McMillan Creek.

		Graywood Gully	Long Point Gully	Sutton Point Gully	North McMillan Creek
WY1	Average CFU loading/mo	4.35E+10	1.10E+11	1.33E+09	1.47E+11
	Loading ratio to Graywood	1.00	2.52	0.03	3.37
	Average CFU/mo/ha	1.14E+09	1.87E+08	1.96E+07	8.25E+07
	Areal load ratio to Graywood	1.00	0.16	0.02	0.07
WY2	Average CFU loading/mo	1.30E+11	6.73E+10	6.79E+09	9.14E+10
	Loading ratio to Graywood	1.00	0.52	0.05	0.70
	Average CFU/mo/ha	3.44E+09	1.14E+08	9.99E+07	5.14E+07
	Areal load ratio to Graywood	1.00	0.03	0.03	0.01
WY3	Average CFU loading/mo	8.03E+10	5.32E+10	1.15E+10	8.27E+10
	Loading ratio to Graywood	1.00	0.66	0.14	1.03
	Average CFU/mo/ha	2.11E+09	9.05E+07	1.70E+08	4.65E+07
	Areal load ratio to Graywood	1.00	0.04	0.08	0.02
WY4	Average CFU loading/mo	1.67E+10	4.13E+10	8.05E+09	4.20E+10
	Loading ratio to Graywood	1.00	2.47	0.48	2.51
	Average CFU/mo/ha	4.41E+08	7.02E+07	4.34E+09	2.36E+07
	Areal load ratio to Graywood	1.00	0.16	9.85	0.05
WY5	Average CFU loading/mo	1.83E+10	4.76E+10	1.18E+08	5.83E+10
	Loading ratio to Graywood	1.00	2.60	0.01	3.18
	Average CFU/mo/ha	4.82E+08	8.09E+08	6.38E+07	3.28E+07
	Areal load ratio to Graywood	1.00	1.68	0.13	0.07

The average values have been used to compare monthly loadings and monthly areal loadings between Graywood Gully and the other streams. mo = month. ha = hectare.

level should rise over the 4-year period of monitoring. Levels of heterotrophic bacteria remained unchanged in all sub-watersheds. This might be expected because the numbers of total bacteria in stream water could be 3 to 4 orders of magnitude greater than the numbers of those bacterial species that were measured to assess water quality. Also, soil and plant detritus were present all year and may be a large and variable natural source of heterotrophic bacteria (Fierer and Jackson, 2006; Fierer et al., 2007).

Exceedency curves provide the opportunity to examine all stream water samples in a given year and to detect trends, especially in those samples that are at the extremes of water quality. Such curves for both *E. coli* and *Enterococcus* (Fig. 6) showed that the water in Graywood Gully had improved. While there were times during a year when water quality was above that set by the EPA for infrequent human contact, the numbers of those times have decreased steadily as the effects of implemented BMPs in the Graywood Gully sub-

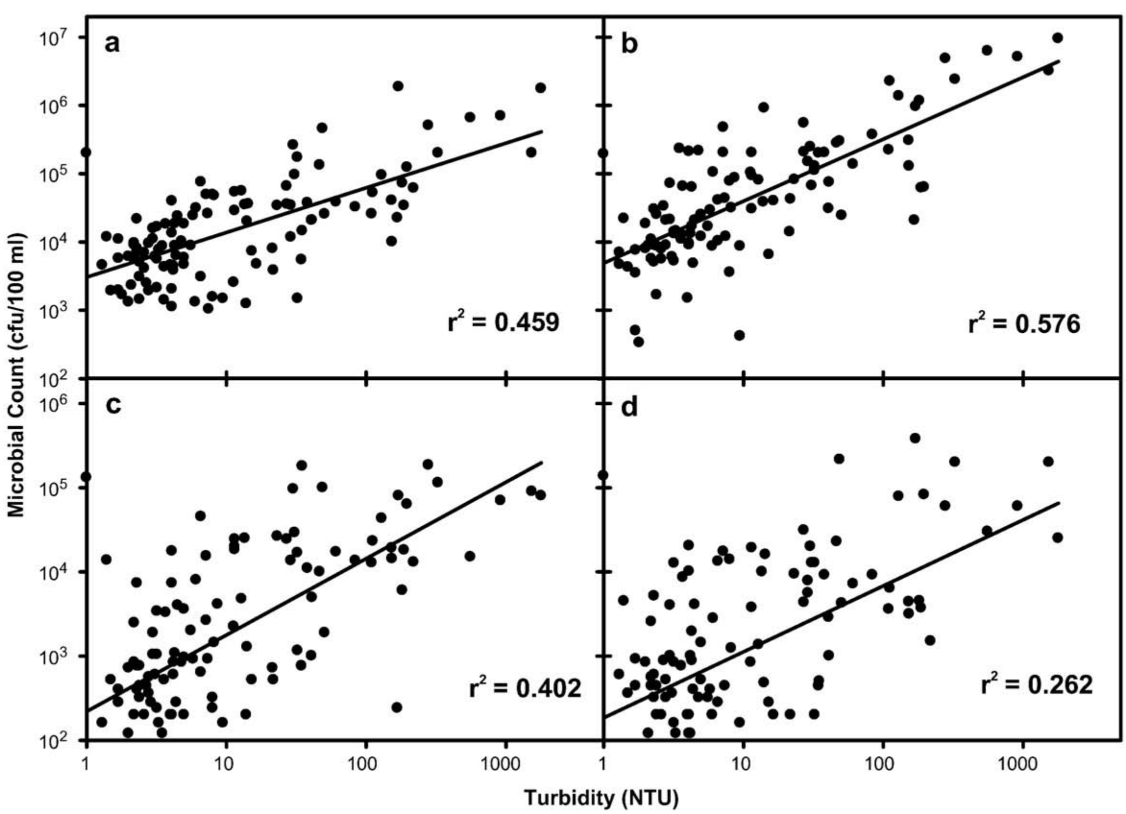

Fig. 7. Microbial levels versus water turbidity in Graywood Gully. (a) Total coliforms, (b) heterotrophic bacteria, (c) *Enterococcus*, and (d) *Escherichia coli*. Because of the seasonality of bacterial abundance, only water samples from June to September were used in this analysis.

watershed became evident. The distribution of the elevated indicator levels during any given year was scattered, so that monthly geometric averaging corrects for these high levels (USEPA, 1999). The source of these elevated levels was not immediately obvious, but it is known that there were variations in stream flow and elevated stream flows, just below those that might define an event. Also, it was not possible to rule out the role of wildlife just upstream prior to the time of sampling. In spite of extraordinary efforts to prevent fecal pollution on a stream with a single farm at Graywood Gully, the exceedency curves demonstrated that EPA infrequent contact standards in nonevent flow were exceeded 20% of the year for *E. coli* and 50% of the year for *Enterococcus*. To meet the EPA standards, either non-agricultural sources of fecal pollution, such as that from wildlife (Somarelli et al., 2007), will have to be addressed or that there will need to be a change in expectations about the acceptable level of stream microbiology standards at bathing beaches. Similar conclusions regarding compliance with current water quality standards have been raised previously (Inamdar et al., 2002; Jamieson et al., 2003). If wildlife represents a major source of indicator bacteria, stream remediation and management efforts and compliance criteria should be adjusted accordingly. Dealing with wildlife contributions will require new approaches and considerations.

Graywood Gully at times had higher total monthly microbial loading than North McMillan Creek, a sub-watershed that is 48 times larger in surface area (Fig. 3). This is a remarkable result and hints at how land use does impact microbial populations that are leaving a sub-watershed. Long Point Gully, a sub-watershed 15.5 times the area of Graywood Gully, had a higher *E. coli* total loading in WY 1 than Graywood Gully. By WY 2 and WY 3, Long Point Gully loading per hectare was >20 fold less than Graywood Gully due to the closing of the dairy operation in that watershed. Loading of *E. coli* from Graywood Gully was 50–100 times greater than from Sutton Point Gully, an agricultural sub-watershed that did not house animals. Changes in farming practice were most likely the cause of the increase in Long Point Gully *E. coli* output in WY 5 and demonstrated the value of utilizing stream bacterial abundance and bacterial loading as a tool to evaluate farm practices (Kay et al., 2007).

There is ample evidence that livestock operations and manure application can elevate fecal coliform and fecal *Streptococcus* abundance in runoff from agricultural lands (Kunkle, 1970; Doran et al., 1981; Baxter-Potter and Gilliland, 1988; Niemi and Niemi, 1991). In this study we applied BMPs to a small sub-watershed in Conesus Lake and asked whether the manipulations reduced microbial loading in comparison to similar small agricultural watersheds as well as to a larger heavily forested sub-watershed. Is this experimental approach reasonable for microbial studies? At first glance the answer would appear to be no because there may be major differences in the physical and biological conditions in the sub-watersheds. For example, fluctuations in physical conditions such as temperature, light, rainfall, etc. may be different due to microclimates. Seasonal variations in critical nutrients that also serve as substrates for microbes (i.e., nitrate, total phosphorous, suspended solids) differ between sub-watersheds (Makarewicz et al., 2009). Sodium levels may be different (Makarewicz, Personal Communication, The College at Brockport) as a result of local differences in deicing salt usage during the winter (Kaushal et al., 2005; Kelly et al., 2009). Even adjacent sub-watersheds may have significantly different water temperatures and discharge when corrected for area, which indicated there were additional sources of groundwater input not present in the other sub-watersheds (Table 1). These may result from differences in rainfall patterns as well as variations in aspect (topography), soil, vegetation, land use, cultivation pattern, etc. There were also seasonal land use differences between sub-watersheds, as farmers rotated crops, changed locations for manure spreading, and varied chemical application of fertilizer and pesticides in their fields (Herendeen and Glazier, 2009). These differences must translate into differences in both the chemical and microbial processes that take place in the sub-watershed and in the stream itself and likely account for some of the variability seen in the bacterial levels between sub-watersheds (Fig. 5). Even in a single, relatively small lake, there is complexity and variability among its sub-watersheds. This requires that the efficacy of BMPs on microbial quality be evaluated at the sub-watershed level (Makarewicz, 2009; Inamdar et al., 2002; Jamieson et al., 2003). In this study we used this particular approach and demonstrated that the application of BMPs in the Graywood Gully sub-watershed led to major reductions in the delivery of microbial populations to downstream aquatic systems in contrast to trends in three other sub-watersheds.

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Storm water events in a small agricultural watershed: Characterization and evaluation of improvements in stream water microbiology following implementation of Best Management Practices

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ABSTRACT

Both storm water event and nonevent flow contributed to the annual discharge from Graywood Gully, a small sub-watershed of Conesus Lake, New York USA, whose land use is 74% agriculture. While events contributed significant amounts of water in short periods of time, nonevents accounted for the majority of water on a yearly basis and could have flow rates matching those that occurred during events. Event storm water was elevated in materials associated with particulates such as total suspended solids, total Kjeldahl nitrogen, and total phosphorus. Water from high flow nonevents was elevated in soluble components such as sodium, nitrate, and soluble reactive phosphorus. As a result, events contributed the majority of particulates to the yearly loading from Graywood Gully whereas nonevents contributed the majority of soluble materials. The levels of total coliforms, *Escherichia coli*, *Enterococcus*, and total heterotrophic bacteria were elevated in storm water relative to nonevent flow, indicating that they acted as particulates. The median level of *E. coli* in nonevents was 200 CFU/100 mL whereas the median level during events was 3660 CFU/100 mL. Consequently, storm events accounted for 92% of all *E. coli* loading from Graywood Gully. Best Management Practices (BMPs) resulted in the mean, median, maximum and minimum levels of event-driven *E. coli* loading from Graywood Gully to decrease 10 fold over a 5-year period. The implementation of BMPs in the Graywood Gully watershed has improved the microbiology of event waters and consequently decreased the role that the watershed plays as a contributor of microbial pollution to Conesus Lake.

Introduction

There are few more dramatic environmental occurrences in the Great Lakes basin than storm water events caused by extended periods of heavy rain and/or snowmelt. These events can transport large amounts of water and material into the lakes in a short period of time and are often associated with disruptive flooding. In the urban setting, storm water can transport significant amounts of heavy metals (Davis et al., 2001; Jartun et al., 2008), polycyclic aromatic hydrocarbons (PAHs) (Hoffman et al., 1985), and polychlorinated biphenyls (PCBs) (Hwang and Foster, 2008). Urban storm water runoff is also elevated in microbial pollution (Salmore et al., 2006) with sources such as pet waste and raccoons (Ram et al., 2007). In the rural setting, agricultural storm runoff may contain large amounts of soil due to erosion (Borah et al., 2003). In addition, runoff may contribute cow manure (McFarland and Hauck, 1999), wildlife fecal material, and agricultural pesticides (Smith et al., 2006).

The growth of cities and suburbs has increased the need to manage rainwater runoff and to understand the capacity and behavior of combined storm water–sewer systems. Thus, significant community efforts in modeling of urban storm water events have been undertaken (e.g., Brezonik and Stadelmann, 2002). In contrast, planning and management for storm water flow in an agricultural watershed is the realm of the individual farmer and regional county agent who are often more concerned with prevention of soil erosion than they are in managing the storm water related pollution in runoff such as phosphorus (P) and fecal material. Compared to urban watersheds, the dynamics of pollutant drainage during storm water events in agricultural settings is not well understood.

Graywood Gully is one of the smallest catchments (38 ha) in the Conesus Lake watershed. Land use is mostly in agriculture (74%) with a single dairy farm operation with approximately 100 head of cattle and row crops including corn and beans. For the past 5 years, “Whole Farm Planning” has been instituted at Graywood Gully, and a myriad of structural and cultural Best Management Practices (BMPs) aimed at controlling nutrient and animal waste pollution have been implemented based on soil testing, evaluation of the P index, and field assessments (Makarewicz, 2009). The changes implemented, such as better manure management, installation of subsurface drainage, and

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the limiting of animal access to sites of potential runoff, were designed to improve the soil and the nutrient and microbial characteristics of the runoff from the dairy farm to Conesus Lake and are described in Simon and Makarewicz (2009). Stream discharge was monitored continuously for a 5-year period, and both weekly and event-driven sampling followed the chemistry and microbiology of the stream water. The microbial output or loading of the Graywood Gully watershed to Conesus Lake was monitored, allowing evaluation of the impacts of BMPs. Previously, we reported on the characteristics and microbiology of nonevent flows (Simon and Makarewicz, 2009). Here we describe the processes that take place during storm water events, examine the microbiology of events, and analyze the changes that the implementation of BMPs had on both stream microbiology and on the role of Graywood Gully as a source of microbial loading to Conesus Lake.

Methods

Graywood Gully discharge was monitored continuously at a point just upstream from its outlet into Conesus Lake with a differential pressure transducer (ISCO 720) attached to an ISCO continuously recording flow meter (Model 6700) equipped with an automatic sampler (Makarewicz et al., 2009). A water year (WY) was defined as the period from 1 September to 31 August of the following year. A total of 5 water years of daily discharge data was collected starting on 1 Sep 2002 and ending on 31 Aug 2007 (Makarewicz et al., 2009). Water samples were taken using two different methodologies: weekly manual grab and automated hydrometeorological event samples.

Flow or discharge from a stream can be considered as having two components: nonevents and events. The latter were associated with rain or snowmelt and were defined operationally as times during which there was a rapid rise (2.54 cm) in creek level over a short period of time (30 min). Upon reaching a discharge peak, the end of an event was defined by a leveling off of the descending limb of the stream hydrograph (Makarewicz et al., 2009). Events were sampled in two ways: (a) water was collected every 60 min during an event, and the samples prior to the event peak (event rise) and following the event peak (event fall) were pooled separately; and (b) a grab sample was taken during the event (event grab).

All water samples were analyzed at SUNY Brockport, a NELAC certified water chemistry laboratory. Procedures for the analysis of total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate (NO_3+NO_2), total Kjeldahl nitrogen (TKN), sodium, and total suspended solids (TSS) followed standard APHA (1999) methods and are reported in Makarewicz et al., 2009. Within 6 h of collection, water samples for microbial analyses were processed at SUNY Geneseo (USEPA 1999, 2000). Procedures for microbial analysis are described in Simon and Makarewicz (2009). Briefly, *Escherichia coli* and *Enterococcus* spp. were measured using membrane filter (MF) methods employing m-ColiBlue24 (Millipore®) medium (Grant, 1997) and m-Enterococcus Agar (Difco 0746), respectively. Additionally, the general microbial composition of stream waters was assessed by measuring total coliform and total heterotrophic bacteria (APHA, 1999).

To compare the chemistry of events and nonevents, only high flow samples were used for the analysis. High flow was defined as being $\geq 1/4$ the maximum flow in the period being analyzed. For example, in an analysis of WY 2 stream chemistry, nonevent flows were analyzed when their daily discharge flow was greater than 3073 m^3/day, a value which is 1/4 of the maximum nonevent flow during WY 2 of 15,954 m^3/day. For the same period, high flow event samples were analyzed when they were associated with a flow greater than 3981 m^3/day, a value that is 1/4 of the maximum event flow discharge of 12,293 m^3/day that occurred during WY 2. The standard of defining high flow as being $\geq 1/4$ of the maximum event flow was chosen after inspection of the flow patterns over the 5-year period of monitoring. This empirically chosen level insured that event flows analyzed were clearly associated with observable rain or snowmelt events. Additionally, placing a similar restriction ($\geq 1/4$ of the maximum flow) on nonevent flows restricted the analysis to times with substantial movement of water and not the low nonevent flow that is often seen for much of the year (Simon and Makarewicz, 2009). Using these limitations allows an analysis of water characteristics to be independent of any effect due to changes in stream velocity. Events were separated into "event grabs," days when events were sampled as they were taking place, and "event collected," where automated stream samplers collected events that were sandwiched between times of defined increases and decreases in stream height. Event grab samples were included when the discharge at the time the sample was taken exceeded 2835 m^3/day, a number which is 1/4 of maximum event grab discharge for the 5 water years of this study (11,314 m^3/day). All event-collected samples were used regardless of stream flow rates during the event.

Monthly microbial nonevent loading was calculated by multiplying the geometric mean (USEPA, 1999) of monthly *E. coli* counts times the nonevent water discharge for the appropriate month. Chemical and microbial data were analyzed directly without transformation. In the 5-year study there were a total of 72 events (ranging from 8 events in WY 5 to 20 events in WY 2). Because of sampling limitations, it was only possible to carry out microbial analysis on a total of 30 of these events. To characterize the microbiology of an event, the microbial counts carried out on samples collected before and after each event peak were averaged. Because the data are not normally distributed, the Mann Whitney Rank Sum Test was used for the comparison of chemistry and biology between events and nonevents.

Results

Events

Events occurred in both wet years (WY 2, WY 3) and dry years (WY 1, WY 4), and the wet/dry peak discharge varied by as much as a factor of 10 (Table 1). Some years (WY 4) had as few as eight events while others (WY 2) had as many as 18 (Fig. 1). Over 5 years of measurement, the average event lasted 84.3 h, although events as short as 7 h or as long as 339 h were recorded (Fig. 2a). More significantly, events of the same duration can differ dramatically in the m^3 discharge of water (Fig. 2b). As an example, the water discharge of events lasting between 20 and 29 h ranged from 159 m^3 to 9128 m^3, a 58-fold difference between minimum and maximum discharge.

While storm events transferred significant amounts of water from Graywood Gully to Conesus Lake, only 21% to 47% of the annual water loading was from event flow (Table 1). Interestingly, in both wet (WY 2) and dry (WY 4) years, daily nonevent discharge rates at times exceeded event discharge rates (Fig. 1). Previous analysis of the Graywood Gully nonevent stream temperatures indicated that there might be an additional groundwater contribution to Graywood water output (Noll and Magee, 2009; Simon and Makarewicz, 2009). However, analysis of water discharge from other Conesus Lake subwatersheds [Sutton Point Gully (68 ha), North McMillan Creek (1778 ha), data not shown] also demonstrated that yearly water loading and daily discharge rates from nonevents could exceed that for events.

Table 1
Yearly nonevent and event discharge (m^3) from Graywood Gully.

	WY 1	WY 2	WY 3	WY 4	WY 5
Event discharge	107,685	436,795	176,740	65,058	124,205
Nonevent discharge	121,998	616,429	668,751	214,823	336,007
Total discharge	229,683	1,053,224	845,491	279,881	460,212
Fraction event/total	0.47	0.42	0.21	0.23	0.27

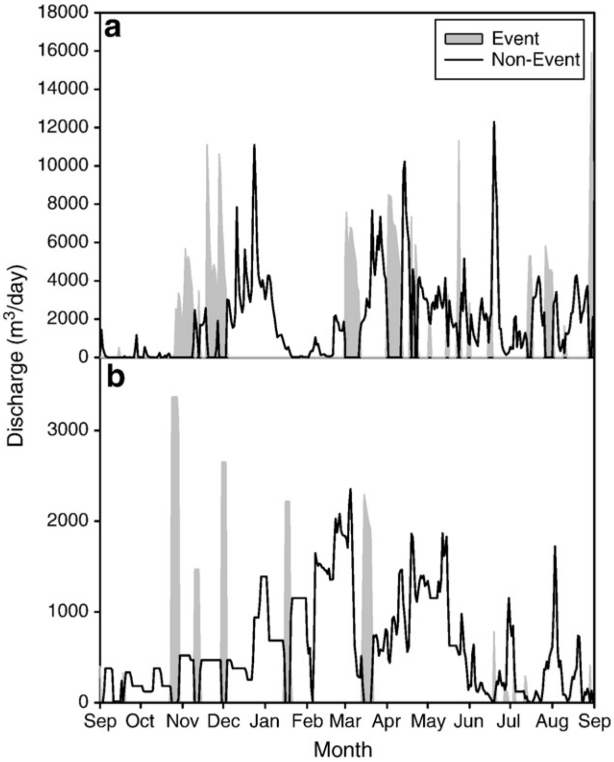

Fig. 1. Daily Graywood Gully event and nonevent water discharge (m^3/day) for (a) WY 2 and (b) WY 4.

Differences among events and high nonevent discharge

We compared water chemistry for nonevents and events of similar discharge for 2 water years (Table 2). Sodium concentrations were higher during high flow nonevents than during events, while TSS was higher during events (Fig. 3). In both cases the differences between events and nonevents were statistically significant ($p \leq 0.001$). Therefore under conditions of the same water flow, particulate fractions (TKN, TP, and TSS) were lost from the watershed at a significantly higher rate during events, while dissolved fractions (SRP, NO_3+NO_2, and sodium) were lost from the watershed during nonevents. However, there were some differences between water years. In WY 4, a relatively dry year, all dissolved fractions concentrations from the watersheds during nonevents were significantly higher than during events (Table 2). In WY 2, a relatively wet year, only the sodium concentration was significantly higher during nonevents than events.

Table 2
The maximum daily event and nonevent discharge (m^3/day) from Graywood Gully, and a comparison of the chemistry of event and nonevent discharges on days when the discharge is $\geq 1/4$ of the maximum daily discharge.

	WY 2			WY 4		
	Event	Nonevent	*p*	Event	Nonevent	*p*
Flow	12,293	15,954		2344	3372	
Na		↑	≤0.001		↑	≤0.001
TSS	↑		≤0.001	↑		≤0.001
NO_3	–	–	0.710		↑	0.007
TKN	↑		≤0.001	↑		≤0.001
SRP	–	–	0.054		↑	≤0.001
Total P	↑		≤0.001	↑		≤0.001

The arrow (↑) indicates that chemistry was elevated in either events or nonevents, while (–) indicates no significant difference. Probabilities (*p*) are from the Mann Whitney Rank Sum Test comparison of the concentrations in events and nonevents for all samples in any given water year.

There was a linear increase in both sodium and TSS loading with increasing discharge (Fig. 4), but the variability in TSS loading was as much as 100 fold. No influence of time of year (month), the year of study (e.g., before and after the implementation of BMPs), or "event intensity" (m^3/h) on the variability in TSS loading was observed.

Because events and nonevents were enriched in different chemical species, each flow differed in its annual contribution to the loading from Graywood Gully (Fig. 5). Events contributed the majority of TSS, TP, and TKN to the lake while nonevents contributed the majority of sodium, SRP, and NO_3+NO_2. Particularly striking is the fact that over a 5-year period, between 83% and 97% of TSS loading was a result of events while between 56% and 85% of NO_3+NO_2 loading was a result of nonevent flow.

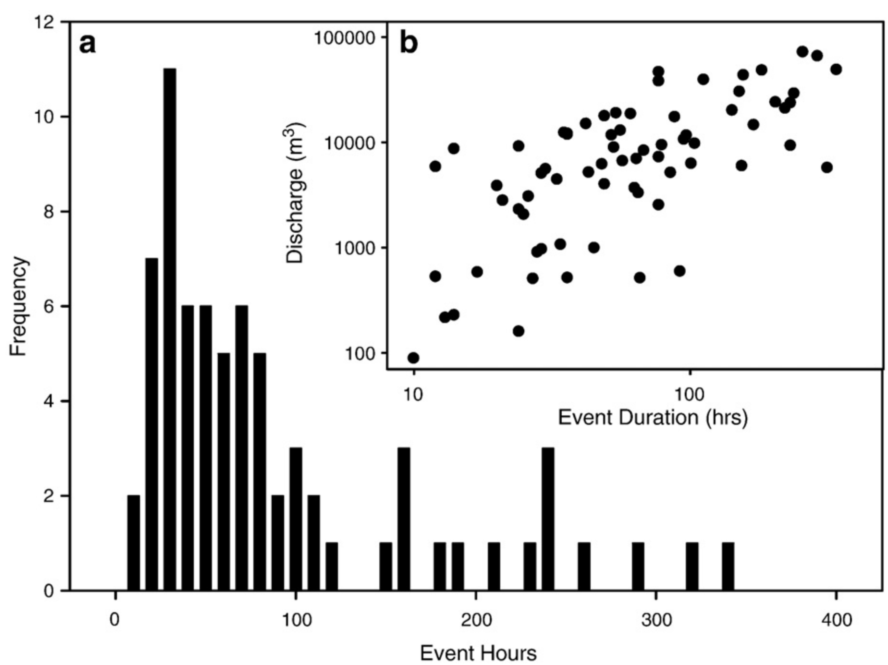

Fig. 2. Characteristics of storm water and snowmelt events in the Graywood Gully watershed. (a) Distribution of event duration in hours. (b) Total event discharge (m^3) as a function of event duration. All events for WY 1 through WY 5 are included.

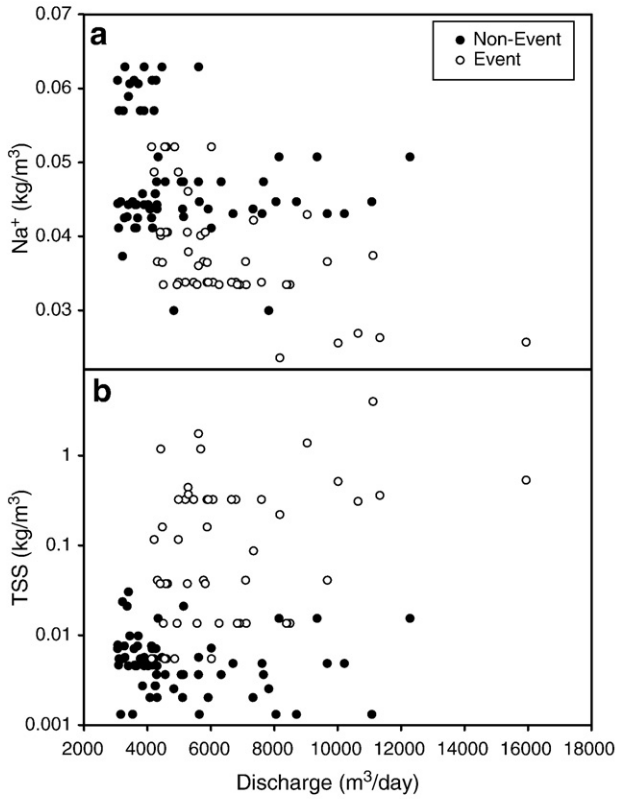

Fig. 3. Event and high nonevent flow concentrations (kg/m^3) of (a) sodium and (b) total suspended solids (TSS) as a function of water discharge/day (m^3) for WY 2.

Microbiology of events

Microbial levels during events were substantially higher than during high flow nonevents. For example, the median concentration of *E. coli* was 200 CFU/100 mL during high flow nonevents and 3660 CFU/100 mL for all samples taken during events. Irrespective of how the event sample was collected, *E. coli*, *Enterococcus*, total coliforms, and total heterotrophic bacteria levels in events were significantly elevated ($p<0.001$ to $p<0.02$) relative to those in high flow nonevent discharge (Fig. 6).

Monthly microbial nonevent loading of *E. coli* was compared to the *E. coli* loading from individual events (Fig. 7a). Individual events almost always contribute at least a 10-fold higher amount than is contributed monthly by nonevent flow. For the 5-year study period,

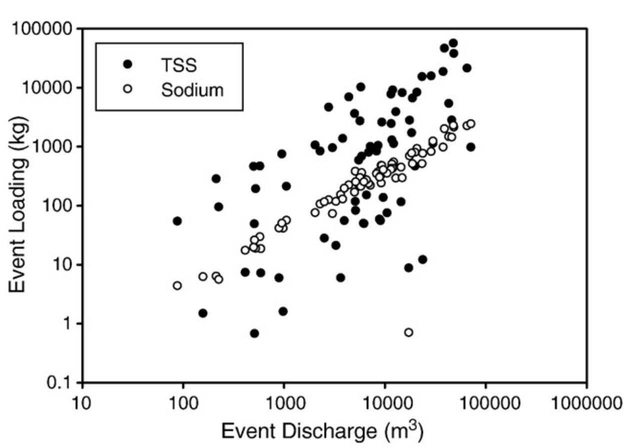

Fig. 4. Graywood Gully event loading of total suspended solids (TSS) and sodium as a function of the total discharge of the event. All events between WY 1 and WY 5 are included in the analysis.

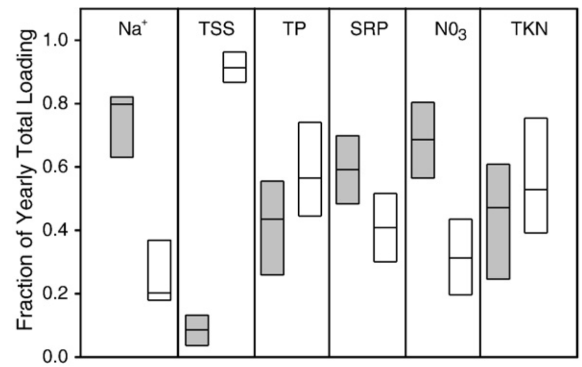

Fig. 5. Box plots of the median and range of the fraction of yearly total loading from events (light box) and nonevents (dark box) of sodium (Na^+), total suspended solids (TSS), total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate (NO_3) and total Kjeldahl nitrogen (TKN) in Graywood Gully over the 5 years of the study.

the median monthly ratios of event loading to nonevent loading for *E. coli* (8.70), total coliforms (4.45), and total heterotrophic bacteria (4.21) demonstrated that events represented the primary source of bacteria that exited from Graywood Gully (Fig. 7b). While the median ratio of event loading to nonevent loading for *Enterococcus* was only 1.05, the average ratio for *Enterococcus* was 31.9, and months with ratios of 670.0 had been measured.

Because the contribution of events was dependent on both the measured level of bacteria in the water and stream discharge, we compared microbial loading of *E. coli* and *Enterococcus* through time. Because of the extreme variability among events, there was no statistical significance in the event loading of *E. coli* or *Enterococcus* between WY 1 and WY 5 ($p=0.164$; $p=0.247$). However, an examination of the trends in the mean, median, maximum and minimum loading for the events of WY 1 through WY 5 (Fig. 8) showed that there was a downward trend in microbial loading in events that corresponded to the period when BMPs were implemented in Graywood Gulley. The mean *E. coli* loading decreased by almost 13 fold and there was a 28-fold decrease in the maximum event loading. The minimum event loading dropped between WY 1 and WY 2 and then remained around 1×10^{10} CFU *E. coli* from WY 3 to WY 5. Similarly, minimum *Enterococcus* loadings decreased from WY 1 to WY 3 followed by a rise in the mean, median, maximum, and minimum between WY 4 and WY 5.

Discussion

There have been previous studies on storm water flow from agricultural lands with either dairy farms or livestock (McFarland and Hauck 1999; Hively et al., 2005). These studies included those that have examined the relationship between management practices and the microbial quality of the water draining from these lands (Lewis et al., 2005; Collins et al., 2007). However, what makes the current study unique is that the detailed monitoring of water flow, chemistry, and microbiology over a period of 5 years (e.g., under different climate and seasonal conditions) has made it possible to examine the “fine structure” underlying long-term effects not only on the farm and drained watershed but also in the receiving body of water. Additionally, because of Graywood Gully watershed's small size (38 ha) and its significant slope (8% grade) to Conesus Lake, storm events are characterized by their rapid response times. Stream discharge at the mouth of Graywood Gully can increase rapidly and within hours of the beginning of a stormwater event. This means that the “signal” resulting from any agricultural improvements has a better chance of being seen.

The yearly contribution of water from Graywood Gully is only a small fraction of the total water entering Conesus Lake. Water from

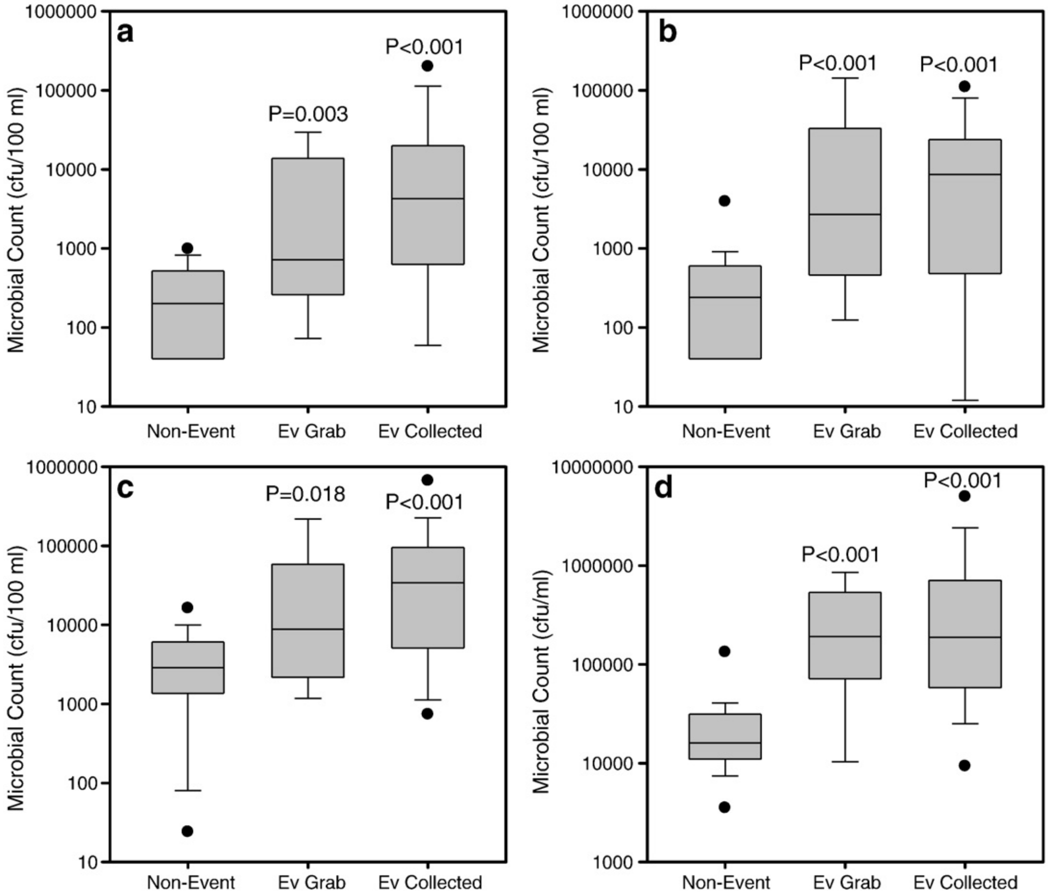

Fig. 6. The microbiology of Graywood Gully during times of event and high nonevent flow. (a) *Escherichia coli*, (b) *Enterococcus*, and (c) total coliforms are given in units of cfu/100 mL. Total heterotrophic bacteria (d) are in cfu/mL. Data for all 5 years of the study were combined. Events were separated into "event grabs" (EV Grab), days when events were sampled as they were taking place, and "event collected" (EV Collected), where automated stream samplers collected events samples. Data is given as the range and median with whiskers representing the 5th/95th percentile of all values. Individual points are outliers. Total samples: nonevent $N=31$; event grab $N=16$; event collected $N=78$.

stream flow undoubtedly influences the lakeshore in the area of the outlet, and any improvement in water quality will have its greatest impact at that location. However, when total nonevent microbial loading was examined (Simon and Makarewicz, 2009), the monthly contribution from Graywood Gully often exceeded North McMillan Creek, a watershed that is 48 times the size of Graywood Gully; loading of *E. coli* from Graywood Gully was 50–100 times greater than from Sutton Point Gully, a small agricultural watershed that does not house animals. In a similar fashion, loadings from events should make significant contributions to loading in Conesus Lake.

Significant variability in the number, intensity, and characteristics of events occurred over the study period. For example, some years had as few as 8 events (WY 5) or as many as 18 (WY 3) and the difference in annual Graywood Gully discharge was as much as five fold (Table 1). Seasonal, yearly, and long-term differences in climatological conditions can affect land characteristics such as soil moisture, and this in turn can have a significant effect on runoff potential and soil erosion, factors that determine both the occurrence and nature of events (Zollweg and Makarewicz, 2009). The origin and character of events are also influenced by human activities. In urban settings, the increasing areal component of pavement contributes to the intensity and chemistry of event runoff (Gilbert and Clausen, 2006). In an agricultural setting such as Graywood Gully, land management practices, such as the annual cycles of tillage, may have a major impact on soil erosion and nutrient loss as does the choice of crops that are chosen by the farmer (Makarewicz et al., 2009). Measures taken for erosion control, such as the installation of 6250 m of subsurface drainage tiles on the farm in the Graywood Gully watershed (Herendeen and Glazier, 2009), modulates water drainage patterns and influences storm water events and water chemistry (Noll and Magee, 2009).

Even though there can be nonevent daily discharges as large as those that occur during events, an event is distinguished from a nonevent by the rate of rise of stream level. Stream levels during events rose at rates ≥2.54 cm/30 min while high flow nonevent discharges were reached at a slower rate. In Graywood Gully, events primarily moved elements associated with particulate matter such as TSS, TKN, TP, and bacteria, while soluble components such as sodium, NO_3+NO_2, and SRP were elevated in water from nonevents (Fig. 3, Table 2). This implies that the pathways of water movement during events and nonevents were different. Storm events are associated with the rapid increase in flow across the land and thus move particulate materials, such as TSS from the surface (Renard et al., 1997) and contribute to surface erosion (Verstraeten and Poesen, 1999). However, the loss of dissolved substances during nonevent flows, even those with discharge rates matching events, is consistent with water percolating through the soil (groundwater) prior to entering the stream.

Event behavior in Graywood Gully differed from that in larger rivers. Richards et al. (2001), working on Ohio rivers such as the Maumee River, observed that event discharge rose first, followed by increased concentrations in particulate fractions (TSS, TP, and TKN), and lastly followed by peaks of dissolved substances (sodium and SRP) a day after the particulate levels increased. This did not happen in Graywood Gully because of the rapid time response and limited stream length (0.87 km). At Graywood Gully, particulates could be elevated in both event rise and event fall samples. Materials associated with water percolating through the soil contributed to the stream only after the event was over and were elevated in nonevent flow.

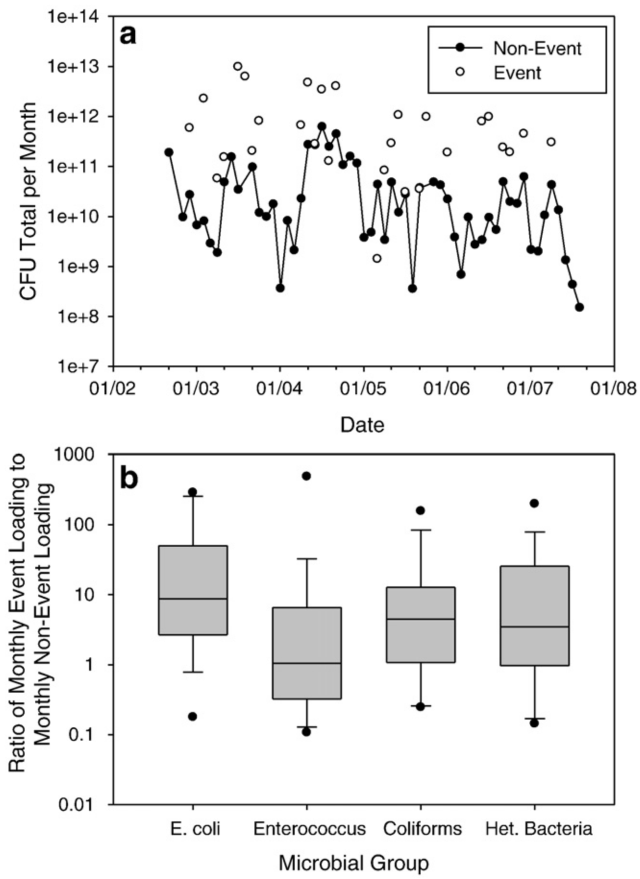

Fig. 7. Microbial loading during events and nonevents in Graywood Gully. (a) Monthly nonevent loading of *E. coli* was calculated by multiplying the geometric mean of the monthly *E. coli* level (CFU/100 mL) times the total monthly nonevent discharge. Individual event loadings were estimated based on event discharge and the *E. coli* level of event samples. Where an event was sampled as both an "event rise" and "event fall," the average of the two samples was used to characterize the event. (b) Box plots of the ratio of event loading to nonevent loading in months where the microbial characteristics of the event were measured for *E. coli* ($N = 30$), *Enterococcus* ($N = 28$), total coliforms (Coliforms) ($N = 30$) and total heterotrophic bacteria (Het. Bacteria) ($N = 25$). The median is given in each box and the whiskers represent the 5 and 95 percentile of the data.

Event and nonevents were responsible for loading different components to Conesus Lake. Events contributed the majority of particulates, especially TSS, whereas nonevents primarily loaded soluble components (Fig. 5). Because events were limited to an average of 48.7 days a year (range: 18.3–78.2 days) during the 5-year study, particulate loading was a dramatic and rapid affair; it is not surprising that it represented the most visible effect of an event. Nonevents contributed materials over most of the year (Simon and Makarewicz, 2009) and were responsible for the chemical species such as SRP that contributed significantly to the growth of metaphyton and plants in Conesus Lake (Bosch et al., 2009a,b).

Events were responsible for the majority of bacterial loading from the Graywood Gully watershed (Fig. 7b), and it was estimated that greater than 92% of all *E. coli* entered the lake through storm events. The calculation of *E. coli* event loading was based on complete estimates of nonevent loading, but only used incomplete event data because not all events were analyzed. While it is not clear how much the event contribution was underestimated, a doubling of the total event contribution would only raise the estimate from 92% to 96%.

Bacteria behave as particulates and are also associated with the particles in TSS that are capable of settling out of the water. For example, Fries et al. (2006) indicated that an average of 38% of the bacteria was associated with particles while Characklis et al. (2005) noted that an average of 20–35% of the organisms was associated with particles which rose to 30–55% in storm samples.

Has the microbial quality of water during events improved after the implementation of BMPs? This is a difficult question to answer because the nonevent presence of indicator bacteria themselves are seasonal (Simon and Makarewicz, 2009) and because there are a number of factors that contribute to the uncertainties in measuring the *E. coli* levels in storm water (McCarthy et al., 2007, 2008). These include the yearly differences in rainfall and soil conditions that would affect the extent and character of events. Complicating the assessment of BMP application in the Graywood Gully watershed is that event water and nonevent water may have different areal sources Noll and Magee, 2009). Nonevent water was clearly from the surface-defined topography of the watershed, while during events as much as 4% of flow from the watershed was actually from an adjacent watershed (Noll and Magee, 2009).

Because of the wide range of variability in the measured bacterial numbers in a single water year, the differences in the loading of bacteria that are indicators of fecal pollution between WY 1 and WY 5 were not statistically significant. However, the trends are encouraging. In WY 1, 97% of all *E. coli* were transported to the lake during events while the number was reduced to 84% in WY 5. Both the average and mean levels of *E. coli* during events have improved (Fig. 8). The maximum loading event seen each year for the 5 years of the study has decreased every year, and the overall improvement is by a factor of 29. The minimum loading event decreased dramatically between WY 1 and WY 2, likely due to the implementation of better manure management practices, but has been constant for the last 3 years of the study at about 1×10^{10} CFU/event. This lack of decrease in the minimum loading may be related to nondomestic animals within the watershed. Bacterial source tracking suggests that half of the *E. coli* that were recovered from Conesus Lake streams had wildlife as a source (Somarelli et al., 2007).

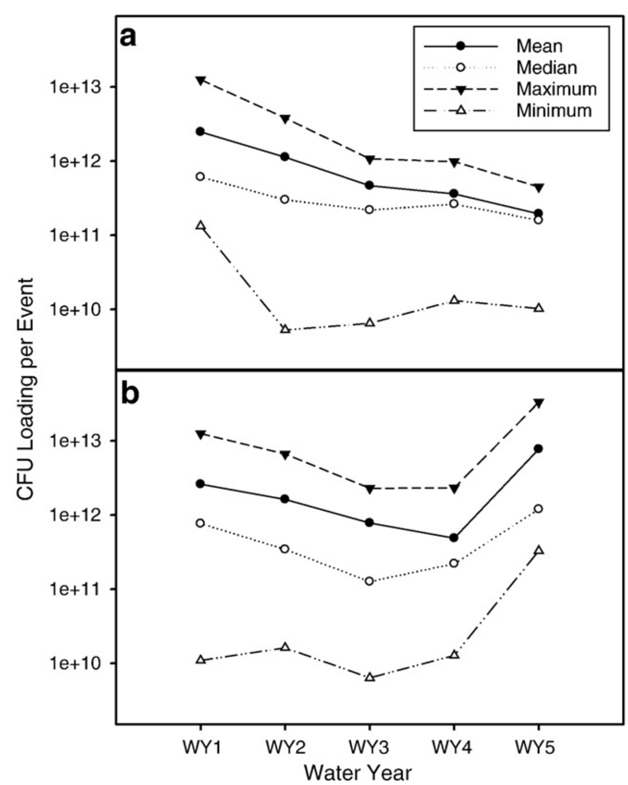

Fig. 8. Event loading of (a) *E. coli* and (b) *Enterococcus* in Graywood Gully WY 1 to WY 5. The mean, median, maximum and minimum loading was determined for events in each water year of the study.

The levels of *Enterococcus*, another indicator of fecal pollution, fell for the first 4 years of the study and then rose. The increase between WY 4 and WY 5 is difficult to explain without further investigation. However, the continuous decrease of *E. coli* loading at the same time suggests that elevated *Enterococcus* levels may come from a different source. This could be due to an increase in abundance of nondomestic animals. Canada goose (*Branta canadensis*) populations are increasing in the Finger Lakes region of New York State (Ankney, 1996) and such wildlife species often contribute more *Enterococcus* than *E. coli* in their feces (Middleton and Ambrose, 2005). Efforts to minimize microbial runoff from farms by the use of appropriate management practices may have a minimum level that can be achieved as events will always carry fecal matter from wildlife. This consideration may require us to rethink both how we evaluate and how we regulate the microbiology of recreational waters (Simon and Makarewicz, 2009).

Previously, we had demonstrated that a major decrease in bacterial levels in nonevent Graywood Gully stream water over a 5-year study period corresponded with the implementation of efficient manure management practices (Simon and Makarewicz, 2009). Although nonevents contributed a majority of the water from Graywood Gully, events delivered 92% of all *E. coli* to the downstream systems. Overall, it appears that the application of BMPs to the single farm in the Graywood Gully watershed has improved the microbiology of event and nonevent flows and consequently decreased the role that the watershed plays as a contributor of fecal pollution to Conesus Lake. Lastly, our data suggest that management practices that focus on decreasing particulate transport during storm water events are likely to be the most useful in managing microbial loss from agricultural landscapes.

Acknowledgments

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Spatial and temporal distribution of the cyanotoxin microcystin-LR in the Lake Ontario ecosystem: Coastal embayments, rivers, nearshore and offshore, and upland lakes

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ABSTRACT

Cyanotoxins, a group of hepatotoxins and neurotoxins produced by cyanobacteria, pose a health risk to those who use surface waters as sources for drinking water and for recreation. Little is known about the spatial and seasonal occurrence of cyanotoxins in Lake Ontario and other lakes and ponds within its watershed. Within the embayments, ponds, rivers, creeks, shoreside, and nearshore and offshore sites of Lake Ontario, microcystin-LR concentrations were low in May, increased through the summer, and reached a peak in September before decreasing in October. Considerable variability in microcystin-LR concentrations existed between and within habitat types within the Lake Ontario ecosystem. In general, the average microcystin-LR concentration was two orders of magnitude lower in embayment (mean = 0.084 μg/L), river (mean = 0.020 μg/L), and shoreside (mean = 0.052 μg/L) sites compared to upland lakes and ponds (mean = 1.136 μg/L). Concentrations in the nearshore sites (30-m depth) and offshore sites (100-m depth) were another order of magnitude lower (mean = 0.006 μg/L) than in the creek/river, bay/pond, and shoreside habitats. Only 0.3% (2 of 581) of the samples taken in Lake Ontario coastal waters exceeded the World Health Organization (WHO) Drinking Water Guideline of 1 μg microcystin/L for humans. In contrast, 20.4% (20 of 98) of the samples taken at upland lakes and ponds within the watershed of Lake Ontario exceeded WHO Guidelines. No significant relationship between nitrate and microcystin-LR concentrations was observed in Lake Ontario even though a significant positive relationship existed between phosphorus and phycocyanin and microcystin-LR concentrations. At an upland lake site (Conesus Lake) in the Ontario watershed, the development of a littoral *Microcystis* population was not observed despite high nutrient loading (P and N) into the nearshore zone, well-developed nearshore populations of filamentous *Spirogyra* and *Zygnema*, the occurrence of *Dreissena* spp., and the known occurrence of *Microcystis* and microcystin production in the pelagic waters of Conesus Lake.

Introduction

Cyanotoxins, a group of hepatotoxins and neurotoxins produced by cyanobacteria, pose a health risk to those who use surface waters as sources for drinking water and for recreation. Besides microcystin-LR, other cyanotoxins reported in the lower Great Lakes include anatoxin-a (Yang, 2005), cylindrospermopsin, dimethyl microcystin-LR, and microcystin-AR (Boyer, 2006). In the Great Lakes, cyanotoxin production has been associated with species of *Microcystis*, *Anabaena*, *Planktothrix*, *Cylindrospermopsis*, and *Aphanizomenon* (Rinta-Kanto and Wilhelm, 2006; Hong et al., 2006). However, morphologically similar cells of the same species of cyanobacteria are not always capable of producing toxins, and cells that are capable of producing toxins may not produce them (Ouellette and Wilhelm, 2003; Ouellette et al., 2006). In general, little is known about the spatial distribution of cyanotoxins, conditions that trigger toxic blooms, and whether cyanotoxins are a major health issue in coastal waters used for recreation and drinking water in the Great Lakes region.

In the upper Great Lakes, microcystins (MCYSTs) were observed in Saginaw Bay of Lake Huron (Vanderploeg et al., 2001) at a concentration (3.5 μg/L) exceeding the World Health Organization Drinking Water Guideline of 1 μg/L (WHO, 1998). In the lower Great Lakes, the western basin of Lake Erie experienced a large reoccurring toxic bloom of *Microcystis aeruginosa*, first observed in 1995 (Brittain

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et al., 2000) and often exceeding the WHO Guideline for microcystin (Boyer, 2006). However, species of *Planktothrix*, rather than *Microcystis*, produced microcystins at Sandusky Bay (Rinta-Kanto et al., 2005). On the eastern shore of Lake Erie, concentrations of microcystin at Presque Isle and Wendt Beach never exceeded 0.5 μg/L (Murphy et al., 2003). Suggested proximate and ultimate causes of these toxic blooms in the open waters of the western basin of Lake Erie included transport from the Maumee River (Boyer, 2006), selective filtration by zebra mussels (Vanderploeg et al., 2001), and changes in light and phosphate and nitrogen regimes (Sarnelle et al., 2005; Murphy and Brownlee, 1981).

Less information on cyanotoxins in Lake Ontario and lakes within its watershed is available. In 2001 microcystin concentration in Hamilton Harbor, a bay at the western end of Lake Ontario, was reported as high as 400 μg/L in surface scums (Murphy et al., 2003). High concentrations of nitrogen and perhaps higher levels of iron were suggested as possible causes of the toxic bloom. Boyer (2006) reported microcystin concentrations exceeding the WHO Drinking Water Guideline of 1 μg/L during August of 2003 in the eastern basin of Lake Ontario near Oswego, NY. In 2004 Makarewicz et al. (2006), reporting on 24 sites along the southern coast of Lake Ontario, noted that microcystin concentrations in the nearshore zone (30-m depth) never exceeded 0.008 μg/L while concentrations in bays and rivers of Lake Ontario were often higher but still lower than WHO Guidelines.

Clearly, cyanotoxins occur in Lake Ontario at concentrations occasionally well above the WHO Guidelines. However, little is known about the spatial and seasonal occurrence of cyanotoxins in Lake Ontario and other lakes and ponds, including Conesus Lake, within its watershed. At Conesus Lake, blooms of littoral filamentous algae have been a regular occurrence since the 1920s (Muenschner, 1927). In general, growth of periphytic algae on sediment and rocks in Conesus Lake (e.g., *Cladophora* sp.) is restricted to very shallow areas along the shore. Littoral filamentous algae are predominant as surface mats of metaphyton within macrophyte beds dominated by Eurasian watermilfoil (D'Aiuto et al., 2006). Zebra mussels (*Dreissena polymorpha*) do occur in the milfoil beds (Bosch et al., 2001) and may be a potential cause of the nuisance accumulations of littoral filamentous algae as they effectively recycle nutrients, increase the amount of light penetrating into the water, and selectively feed on diatoms and other algae that potentially compete for light, substrate, and dissolved nutrients (Pillsbury et al., 2002). The presence of zebra mussels has been correlated with the increased growth of littoral algae, including *Microcystis* sp. and rooted macrophytes in the Great Lakes (Pillsbury et al., 2002; Skubinna et al., 1995; Lowe and Pillsbury, 1995) and in some

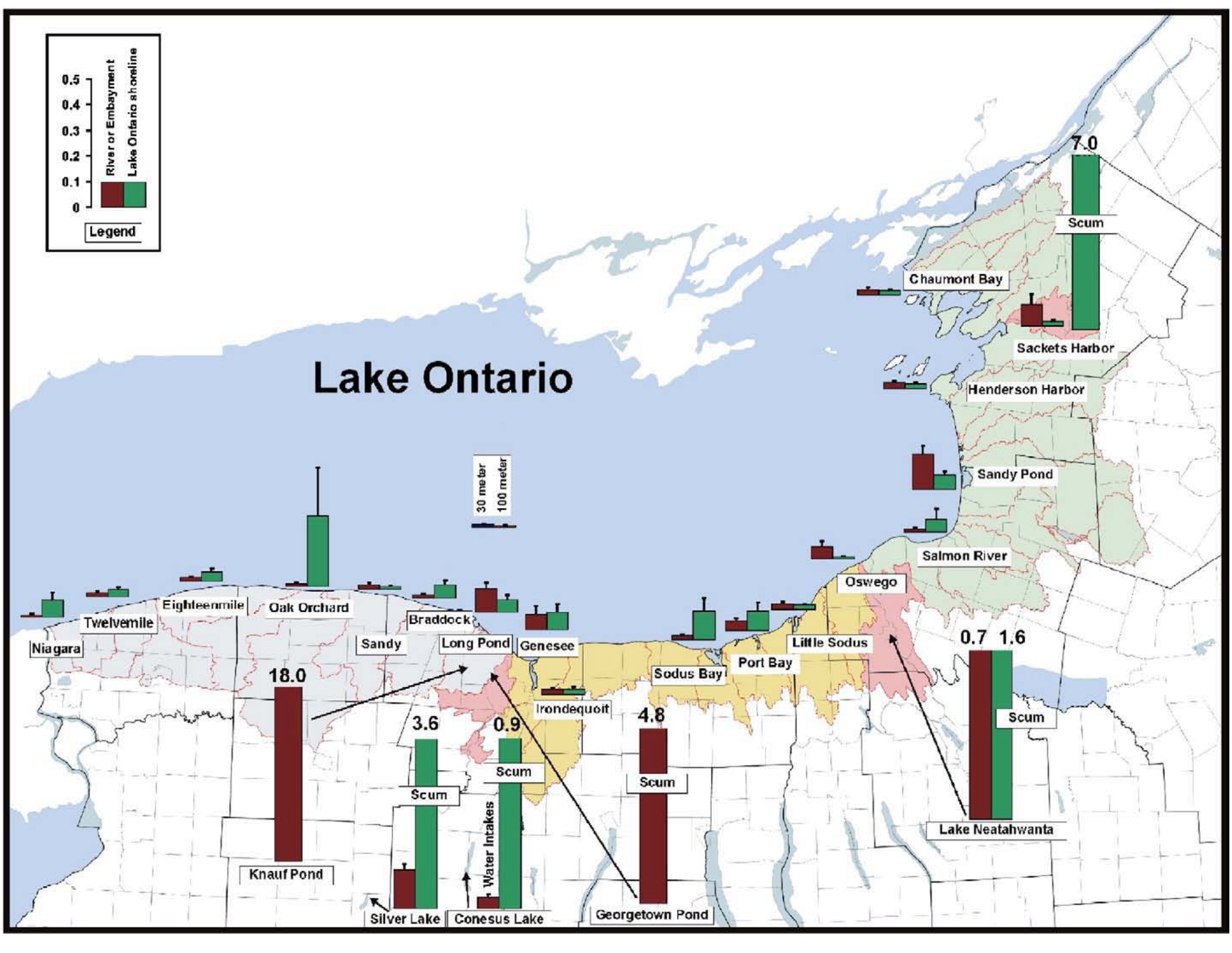

Fig. 1. Sampling locations and microcystin-LR concentrations (average ± S.E., μg/L) along the Lake Ontario shoreline and the associated rivers, embayments and ponds. The green vertical bar at each Lake Ontario site represents the "shoreside" sampling site. The red vertical bar at each Lake Ontario site represents samples taken in the creek, river or embayment. Vertical bars are to scale. Bars for Knauf and Georgetown Ponds, Lake Neatahwanta, Sackets Harbor "scum" and the Conesus and Silver Lake "scum" concentrations are not to scale with concentration listed above the bar. The vertical bars for the nearshore and offshore of Lake Ontario are labeled "30 m" and "100 m".

of the larger Finger Lakes of New York (Zhu et al., 2006). While the ecosystem dynamics in Conesus Lake may have been altered by the invasion of zebra mussels, the principal agent that drives the nuisance growth of littoral algae appears to be high nutrient levels (Bosch et al., 2009). Makarewicz et al. (2007) demonstrated that stream effluent entering Conesus Lake from watersheds that were primarily in agriculture contained high concentrations of soluble reactive phosphorus (P) and nitrate (N) that stimulated the growth of the *Zygnema* spp., *Spirogyra* spp. and other species of littoral algae, with biomass doubling times in the range of hours to days.

We expected that the high nutrient loading (P and N) to the nearshore of Conesus Lake, the well-developed populations of *Spirogyra* and *Zygnema* in the nearshore, the occurrence of *Microcystis* and the toxin microcystin in pelagic waters, and the occurrence of *Dreissena* spp. in Conesus Lake would produce environmental conditions required for the development of *Microcystis* and cyanotoxin production in the nearshore region. An opportunity thus existed at Conesus Lake through the USDA/CSREES watershed project (Makarewicz et al., 2009) to evaluate the impact of management practices on potential cyanotoxin occurrence in the nearshore of Conesus Lake where blooms of littoral filamentous algae have been a regular occurrence since the 1920s (Forest et al., 1978; Bosch et al., 2009). The present analysis assesses the spatial and seasonal distribution of microcystin-LR at 45 sites in the New York waters of Lake Ontario, and other lakes, including Conesus Lake, within its watershed.

Methods

A spatial sampling regime was undertaken which provided unprecedented spatial coverage of the New York coastline ranging 186 km from the Niagara River on the west to Chaumont Bay in eastern Lake Ontario from 2003 to 2006 (Fig. 1). Sites east of Port Bay

Table 1

Microcystin-LR and phycocyanin concentrations (μg/L) for Lake Ontario, Lake Ontario embayments, tributaries and ponds and the adjacent Lake Ontario shoreline water (SS) and selected New York State inland lakes from May through October 2003 to 2006.

	Latitude	Longitude	Microcystin-LR (μg/L) Mean ± S.E. (Min–Max)	Phycocyanin (μg/L) Mean ± S.E.	N
Offshore/Nearshore Sites			0.006 ± 0.001		
Lake Ontario 100 m	43.42514	77.89707	0.006 ± 0.001 (<0.003–0.032)	7.5 ± 1.0	33
Lake Ontario 30 m	43.37652	77.89393	0.007 ± 0.002 (<0.003–0.070)	6.9 ± 1.0	33
Bays/Harbors Sites			0.084 ± 0.037		
Sandy Pond	43.63191	76.18874	0.117 ± 0.028 (0.013–0.225)	64.0 ± 20.9	8
Long Pond North	43.29299	77.67775	0.082 ± 0.019 (0.004–0.795)	89.0 ± 14.4	46
Sackets Harbor	43.94893	76.12206	0.070 ± 0.037 (0.003–0.310)	17.3 ± 6.8	8
Long Pond South	43.28625	77.70528	0.053 ± 0.008 (0.008–0.246)	96.3 ± 15.9	39
Port Bay	43.27897	76.82813	0.032 ± 0.007 (0.006–0.117)	30.0 ± 8.1	16
Henderson Harbor	43.85051	76.20630	0.019 ± 0.006 (<0.003–0.050)	8.9 ± 0.9	8
Little Sodus Bay	43.34359	76.69892	0.018 ± 0.006 (<0.003–0.045)	12.5 ± 2.4	8
Chaumont Bay	44.02972	76.21444	0.016 ± 0.006 (<0.003–0.052)	12.0 ± 3.1	8
Irondequoit Bay	43.18579	77.52851	0.016 ± 0.006 (<0.003–0.093)	24.1 ± 6.8	16
Sodus Bay	43.26355	76.99359	0.013 ± 0.004 (<0.003–0.052)	10.0 ± 1.7	16
Braddock Bay	43.30765	77.70746	0.011 ± 0.004 (<0.003–0.072)	12.6 ± 1.9	16
Sackets Harbor scum	43.94893	76.12206	7.034 (7.034–7.034)	41540.0	1
Shoreside Sites			0.052 + 0.013		
Oak Orchard Creek SS	43.33694	78.17722	0.239 ± 0.162 (<0.003–2.545)	62.8 ± 24.8	16
Sodus Bay SS	43.27403	76.97469	0.094 ± 0.044 (0.003–0.543)	77.4 ± 50.2	16
Port Bay SS	43.30497	76.83757	0.064 ± 0.028 (0.004–0.434)	15.5 ± 3.7	16
Niagara River SS	43.27764	78.98757	0.058 ± 0.026 (<0.003–0.325)	32.8 ± 11.3	16
Genesee River SS	43.25946	77.60303	0.056 ± 0.030 (0.003–0.344)	29.0 ± 10.3	12
Sandy Pond SS	43.63102	76.19539	0.047 ± 0.013 (0.014–0.123)	20.0 ± 4.3	8
Braddock Bay SS	43.29827	77.68408	0.044 ± 0.015 (<0.003–0.225)	55.9 ± 17.0	16
Salmon River SS	43.55231	76.21347	0.043 ± 0.034 (0.003–0.282)	64.8 ± 52.6	8
Eighteenmile Creek SS	43.31884	78.83585	0.033 ± 0.012 (<0.003–0.133)	25.1 ± 8.8	12
Twelvemile Creek SS	43.31886	78.83585	0.022 ± 0.009 (<0.003–0.128)	13.7 ± 3.8	16
Irondequoit Bay SS	43.23644	77.53461	0.017 ± 0.007 (<0.003–0.063)	17.8 ± 3.9	12
Henderson Harbor SS	43.83843	76.29582	0.015 ± 0.005 (<0.003–0.044)	10.9 ± 1.8	8
Chaumont Bay SS	44.01126	76.28714	0.014 ± 0.002 (0.006–0.025)	12.2 ± 2.9	8
Little Sodus Bay SS	43.34359	76.69892	0.014 ± 0.005 (<0.003–0.042)	9.2 ± 1.3	8
Oswego River SS	43.45877	76.53444	0.008 ± 0.003 (<0.003–0.029)	10.3 ± 1.5	8
Sandy Creek SS	43.35331	77.89151	0.008 ± 0.002 (<0.003–0.067)	15.3 ± 4.1	33
River/Creek Sites			0.020 ± 0.004		
Genesee River	43.25943	77.60298	0.050 ± 0.026 (<0.003–0.435)	54.4 ± 26.1	16
Oswego River	43.45929	76.51156	0.042 ± 0.017 (0.003–0.139)	14.5 ± 3.7	8
Sandy Creek	43.35358	77.89128	0.016 ± 0.004 (<0.003–0.065)	15.8 ± 3.3	16
Twelvemile Creek	43.31886	78.83585	0.014 ± 0.004 (<0.003–0.071)	12.3 ± 4.0	16
Salmon River	43.57036	76.18706	0.011 ± 0.004 (<0.003–0.035)	11.7 ± 1.4	8
Oak Orchard Creek	43.37116	78.19146	0.011 ± 0.003 (<0.003–0.052)	10.7 ± 2.1	16
Niagara River	43.25979	79.05779	0.009 ± 0.003 (<0.003–0.044)	9.1 ± 1.4	16
Non-Lake Ontario Sites			1.136 ± 0.317		
Knauf Retention Pond	43.25258	77.71608	18.042 ± 2.353 (15.689–20.394)	7608.0 ± 0.0	2
Georgetown Pond	43.24339	77.71767	4.838 ± 2.580 (2.125–9.995)	1313 ± 678	3
Silver Lake surface scum	42.70203	78.02567	3.593 ± 1.092 (0.040–10.716)	6399.0 ± 6085.8	11
Lake Neatahwanta scum	43.31170	76.42645	1.595 (1.595–1.595)	–	1
Conesus Lake surface scum	43.26333	76.99318	0.918 ± 0.832 (0.008–5.070)	1360.1 ± 785.2	6
Lake Neatahwanta	43.31170	76.42645	0.696 ± 0.072 (0.163–1.126)	476.6 ± 161.5	18
Conesus Lake water plants	42.83161	77.70828	0.012 ± 0.003 (0.003–0.061)	19.2 ± 4.3	27
Silver Lake	42.70203	78.02567	0.042 ± 0.006 (0.003–0.150)	29.6 ± 4.1	30

For statistical analysis, non-detects were assigned a value of 0.001. The number of samples (n) taken varies from site to site. Some sites were routinely sampled (e.g., Lake Ontario) while other sites were occasionally sampled (see Methods).

were sampled in 2005 and 2006 only. Seasonal samples were taken monthly from June through September in every year with occasional samples from May and October. In general, water bodies were selected to cover a wide variety of types with respect to morphology, mixing behavior, trophic state, cyanobacteria dominance, and regional distribution. Forty-five sites were sampled within the Lake Ontario watershed (Table 1). Thirty embayments, streams, and Lake Ontario itself (e.g., Niagara River, Twelvemile Creek, Eighteenmile Creek, Oak Orchard Creek, Sandy Creek, Braddock Bay, Long Pond, Genesee River, Irondequoit Bay, Sodus and Port Bays, Sackets Harbor, Chaumont Bay, Oswego River, Sandy Pond) (Fig. 1) were sampled at a depth of 1 m (swimmable depth) approximately 2 to 10 m from land. Stream samples were taken in the mouth of the stream; embayment samples were generally taken near the outlet to Lake Ontario. Latitude and longitude are provided in Table 1. Samples taken along the Lake Ontario shoreline at a depth of 1 m were defined as shoreside samples. The areas within 30 m and beyond 30 m were defined, respectively, as nearshore and offshore (Hallermeier, 1981). Two open water sites due north of Hamlin Beach State Park were sampled biweekly at a 1-m depth for microcystin-LR in the nearshore (30-m water depth, 2.4 km from shoreline) and the offshore (100-m water depth, 7.8 km from the shoreline) zones. Biweekly sampling occurred at Long Pond until June after which weekly samples were taken. Samples were occasionally taken from two storm water retention ponds (twice) within the Lake Ontario watershed and from three upland lakes: Conesus Lake (weekly, 2004 and 2006), Silver Lake (biweekly, 2004 only), and Lake Neatahwanta (occasionally).

For monitoring purposes, up to 20 L (or until the filter was clogged) of water from a depth of 1 m were pumped in the field (Manostat peristaltic pump) through a 125-mm glass fiber filter (Whatman 934 AH, 1.5 μm particle retention), placed in a centrifuge tube, stored on dry ice for transport and stored at −20 °C in the laboratory. In the laboratory 15 mL of 50% methanol was added to the centrifuge tube containing the filter, sonicated (Model 100 Fisher Scientific Sonic Dismembrator) for 1 min with a 20 s on 20 s off pulsed cycle on ice to reduce degradation of MCYSTs from excessive heat, and centrifuged at 3000 rpm. The methanol supernatant was filtered through a 0.45-μm filter and placed in a 4.5-mL cryovial for storage at −20 °C until analysis.

Standard solutions of MCYST-LR were made from a stock concentration determined spectrophotometrically at a wavelength of 238 nm (Weller et al., 2001). MCYST-LR was determined by the *P*rotein *P*hosphatase *I*nhibition *A*ssay (PPIA) following Carmichael and An (1999). The PPIA was performed by placing replicate ($n=2$) 10-μL aliquots of standards (range 3 to 60 μg/L), blank, control activity check and samples onto a 96 well microtitre plate. If the relative percent deviation of duplicates exceeded 20%, the analysis was rerun. Protein phosphatase reagent [PP1 in a buffered solution containing equal portions of dithiothreitol (0.049%), bovine serum albumin (0.158%), and manganese chloride (0.031%)] was added (90 μL) to all wells except the blank, which received a similar reagent without the PP1. The microtitre plate was placed into an Elx-808 microplate reader (Biotek Instruments Inc.), shaken for 15 s, and transferred to an incubator for 5 min at 37 °C. The plate was removed and 100 μL of p-nitrophenylphosphate solution was added to all wells, placed back in the plate reader, and shaken for 15 s; absorbance was recorded at 405 nm. The plate was then replaced in the incubator for 1 h and removed; absorbance was read again at 405 nm. Detection limit of microcystin-LR was determined to be 0.003 μg/L (G.L. Boyer, Personal Communication, SUNY Environmental Science and Forestry, 2006). An interlaboratory analytical comparison on split samples of *Microcystis aeruginosa* (strain LB 2385 in BG11 medium) by PPIA analysis between laboratories at SUNY ESF (mean = 1.724 μg/L) and SUNY Brockport (mean = 1.818 μg/L) indicated excellent agreement (paired *t*-test, $P=0.66$).

At all sites the following were also analyzed: chlorophyll (Seabird CTD 25 fluorometry), turbidity (Seabird CTD 25), soluble reactive phosphorus [Technicon Autoanalyser (APHA, 1999, method SM 4500-P F)], total phosphorus [Technicon Autoanalyser, persulfate digestion (APHA 1999, SM 4500-P F)], and nitrate–nitrite [Technicon Autoanalyser (APHA, 1999, SM 4500-NO_3 F)]. Phycocyanin was also analyzed by fluorometry [Turner Designs TD-700 and Turner Designs Aquaflor (excitation 595 nm, emission 670 nm)] and standardized with phycocyanin from SIGMA Chemical. Comparison with a standardization procedure using cultures of *Microcystis aeruginosa* (UTEX LB 2665) suggests that our values may be 15 to 29% high. Culture phycocyanin concentration was determined by the trichromatic procedure (Bennett and Bogorad, 1973) using a Beckman DU 640 Spectrophotometer. Thus our measurements of phycocyanin are likely to be high estimates of pigment concentration but are good quantitative measurements of phycocyanin fluorescence. The SEABIRD CTD provided *in vivo* measurements of temperature, chlorophyll, PAR, dissolved oxygen, pH, conductivity, and turbidity. This instrument is calibrated in the lab each month for pH, dissolved oxygen, and chlorophyll using commercially available standards. The Water Quality Lab at SUNY Brockport is NELAC certified (ELAP #11439, EPA # NY 01449).

A Kolmogorov–Smirnov *D* test revealed that, even with log transformations, only some years of the 4-year microcystin (2003 and 2006) and phycocyanin (2004) data were normally distributed. Even so, the log transformed data were used to ensure near normal distribution (Zar, 1999). Moderate violations of parametric assumptions have little or no effect on substantive conclusions from Analysis of Variance (Cohen, 1969; Zar, 1999). A two-way ANOVA was used to test for differences in microcystin and phycocyanin concentrations among months and sites followed by post-hoc comparisons (Least Significant Difference, SPSS, Inc., Version 16). In addition, a one-way ANOVA allowed comparison of microcystin concentrations between habitats: bay/ponds, creeks/rivers, nearshore/offshore, shoreside sites, and upland ponds and lakes. Differences were considered significant at $p<0.05$.

Results

Ninety-eight samples were taken at inland lakes within the Lake Ontario watershed from 2003 to 2006; all had detectable levels (>0.003 μg/L) of MCYST. Of the 581 samples taken along coastal Lake Ontario during the 2003–2006 period, 91.6% had detectable levels of microcystin-LR. Significant differences were observed in both average monthly ($P=0.000$, two-way ANOVA) and average site microcystin-LR concentrations ($P=0.000$). Interaction effects were insignificant ($P=0.249$).

A post-hoc analysis (LSD) revealed that average concentrations (μg/L) of microcystin-LR in May (0.006), June (0.015), and October (0.006) were significantly lower than in July (0.037), August (0.043),

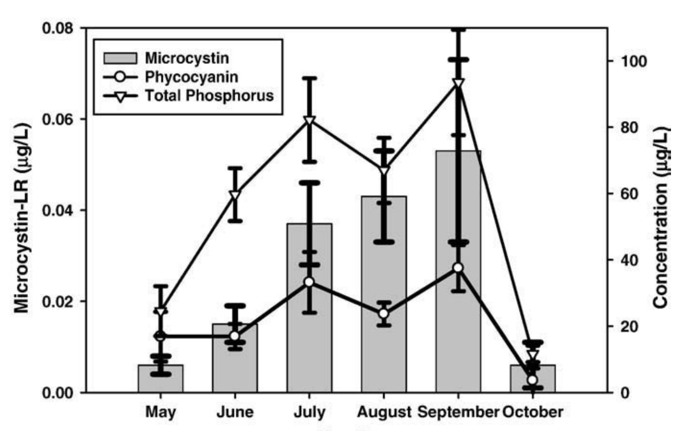

Fig. 2. Average monthly (±S.E.) microcystin-LR, total phosphorus and phycocyanin concentration (μg/L) at 37 sites in Lake Ontario from 2003–2006. Sites include streams, rivers, embayments, shoreside sites, and the nearshore and offshore zones. See Fig. 1 for location of sites.

and September (0.053) of the study period (Fig. 2). A similar post-hoc analysis indicated that average Lake Ontario offshore (100-m depth: mean = 0.006 µg/L) and nearshore (30 m: 0.007 µg/L) microcystin-LR concentrations, which were not significantly different from each other, were generally significantly lower than in embayments, streams, and rivers (Table 1, Fig. 1). The near and offshore sites of Lake Ontario had significantly lower concentrations than all but five of 18 embayment/river sites (Niagara River, Oak Orchard Creek, Braddock Bay, Salmon River, and Chaumont Bay) and all but four of 18 shoreside sites (Sandy Pond, Salmon River, Little Sodus Bay, and Twelvemile Creek) (Table 1).

Considering the bay/pond, creek/river, and the shoreside site habitats, Sandy Pond (mean = 0.117 µg/L) had significantly higher concentrations of microcystin-LR than all other bays except Long Pond North (Table 1), a barrier beach pond on the south shore of Lake Ontario. Similarly, the shoreside site at Oak Orchard Creek had a significantly higher concentration of microcystin-LR (mean = 0.239 µg/L) than the Sodus Bay shoreside site (0.094 µg/L); both had significantly higher concentrations than all other shoreside sites (Table 1). Of the rivers and creeks, the Genesee River had the highest average microcystin-LR concentration (mean = 0.050 µg/L) but was significantly different from only Oak Orchard Creek and the Niagara River, which had the lowest concentration of microcystin-LR (0.009 µg/L) (Table 1).

When the data were grouped by habitat types, significant differences (one-way ANOVA followed by post-hoc analysis by LSD) occurred between Lake Ontario bays (mean = 0.084 µg microcystin-LR/L), rivers/creeks (0.020 µg/L), shoreside sites (0.052 µg/L), nearshore and offshore sites (0.006 µg/L), and upland lakes and ponds (1.282 µg/L). The upland lakes and ponds were characterized by high concentrations of microcystin-LR in surface scums (e.g., Silver Lake 3.593 µg/L) but much lower concentrations 1 m below the surface (0.042 µg/L) (Table 1). At the Conesus Lake site, microcystin-LR was observed at levels well below the WHO Guideline over two summers in the epilimnion of the offshore waters (Table 1, Conesus Lake Water Plants) but occasionally at elevated levels in offshore surface scums (Table 1). For example, an offshore surficial toxic bloom of *Microcystis aeruginosa* occurred after a calm windless night during September 2004. Within 1 h of the wind increasing, the surface scum was dissipated, and the bloom was no longer detectable by our sampling techniques. Intensive sampling of the littoral filamentous algae of Conesus Lake over an 8-year period revealed a dominance of *Spirogyra* and *Zygnema* but little evidence of *Microcystis* spp. (Bosch et al., 2009). Two small retention ponds, Knauf and Georgetown, had very high levels of microcystin-LR in the water column (18.042 and 4.838 µg/L, respectively).

Considering the combined data with microcystin-LR concentrations above 0.1 µg/L from Lake Ontario, Lake Neatahwanta, Conesus Lake and other sites, there were significant correlations between microcystin-LR and phycocyanin ($r^2 = 0.76$, $P = 0.000$, $n = 62$) and microcystin-LR and total phosphorus concentrations ($r^2 = 0.70$, $P = 0.000$). The correlation between microcystin-LR and nitrate was not significant ($r^2 = 0.008$, $P = 0.478$, $n = 62$). However, a significant correlation also existed between phycocyanin and total phosphorus concentrations ($r^2 = 0.86$, $P = 0.000$, $n = 62$).

Discussion

In general, within the embayments/rivers and shoreside sites of Lake Ontario, microcystin-LR concentrations were low in May, increased through the summer, and reached a September peak before decreasing in October (Fig. 2). Average total phosphorus and phycocyanin followed a similar seasonal pattern (Fig. 2). A similar seasonal pattern was observed in Lake Suwa, Japan, but with much higher concentrations of microcystins (Park et al., 1997).

Only 0.3% (2 of 581) of the samples taken in Lake Ontario coastal waters exceeded the WHO Drinking Water Guideline of 1 µg microcystin/L for humans. On 19 Sept 2006 a concentration of 2.545 µg/L was observed at a depth of 1 m at the shoreside location at Oak Orchard Creek. At Sackets Harbor in August 2005, a surface scum was observed with a microcystin-LR concentration of 7.034 µg/L. *Microcystis aeruginosa* was the predominant member (96.4% of total abundance) of the phytoplankton community of the surface scum.

In general, at the Lake Ontario offshore (100-m depth) and the nearshore (30-m depth) sites, the maximum level of microcystin-LR observed never exceeded 0.032 µg/L (mean = 0.006 and 0.007 µg/L, respectively) while maximum levels along the shoreline (lake side), bays and rivers were often higher by an order of magnitude [e.g., 0.225 µg/L (Braddock Bay lake side), 0.435 µg/L (Genesee River), 0.795 µg/L (Long Pond North), 0.325 µg/L (Niagara River shoreside), 0.123 µg/L (Sandy Pond lake side), 0.310 µg/L (Sackets Harbor), 0.543 µg/L (Sodus Bay shoreside)] (Table 1, Fig. 1) but still substantially lower than the World Health Organization Guideline of 1.00 µg/L.

In contrast, 20.4% (20 of 98) of the samples taken at upland lakes and ponds within the watershed of Lake Ontario exceeded the WHO Drinking Water Guideline. Microcystin-LR concentrations as high as 5.070 µg/L in the offshore of Conesus Lake, a source of drinking water, 10.716 µg/L in Silver Lake in September 2004, and 1.595 µg/L in Lake Neatahwanta during July of 2004 were observed. Knauf (18.042 µg/L) and Georgetown Ponds (4.838 µg/L), both retention ponds receiving water from suburban home developments, had very high microcystin-LR concentrations during the summer.

Considerable variability in microcystin-LR concentrations existed between and within habitat types within the Lake Ontario ecosystem. In general, upland ponds and lakes had the highest average (mean = 1.126 µg/L) and the highest variability in microcystin-LR concentrations (range: 0.003 to 20.394 µg/L) compared to the shoreside sites, creeks/rivers, embayment/ponds, and the near/offshore waters of Lake Ontario (Table 1). Even though there were significant differences within a habitat, the concentration of microcystin-LR was two orders of magnitude lower (0.02 to 0.08 µg/L range of means) in embayment, river, and shoreside sites compared to upland lakes and ponds. Concentrations in the nearshore site (30-m depth) and offshore site (100-m depth) were another order of magnitude lower (mean = 0.006 µg/L) than in the creek/river, bay/pond, and shoreside habitats. In general, within the open waters of Lake Ontario, average concentrations of microcystin-LR at shoreside sites with samples at swimmable depth were significantly higher than at the nearshore or offshore waters of Lake Ontario. However, there were exceptions. Of the 36 stream, river, embayment and shoreside samples, only samples from the Niagara River, Oak Orchard Creek, Braddock Bay, Salmon River, Chaumont Bay, and the shoreside sites at Sandy Pond, Salmon River, Little Sodus Bay, and Twelvemile Creek were not significantly different from the near and offshore sites of Lake Ontario.

Sivonen (1990) and Murphy et al. (2003) suggested that the increased nitrogen in Lake Erie may contribute to the growth of toxic *Microcystis*. Orr and Jones (1998), working with N-limited *Microcystis* cultures, suggested that microcystin production is controlled by environmental effects on the rate of cell division, not through direct effect on the metabolic pathways of toxin production. Microcystin production decreased at a rate compared to that of cell division when the culture becomes nitrate limited. We observed no significant relationship between nitrate and microcystin-LR concentrations in P-limited Lake Ontario. Our work does support the hypothesis that the spatial differences observed in microcystin-LR concentration (above 0.1 µg/L) were positively related to variability in phosphorus concentrations. Total phosphorus concentrations did explain 70% of the variability in microcystin-LR concentrations and 86% of phycocyanin levels. However, the high loading of phosphorus from agriculturally dominated watersheds (Makarewicz et al., 2009) to

Conesus Lake did not lead to a toxic or nontoxic nearshore *Microcystis* bloom over an 8-year period. Historically, blooms of littoral filamentous algae (*Cladophora*, *Spirogyra*, and *Zygnema*) have been a regular occurrence in littoral of Conesus Lake since the 1920s (Muenschner, 1927; Savard and Bodine, 1971; Forest et al., 1978) with *Spirogyra* consistently present in shallow waters during spring and summer (Forest et al., 1978; D'Aiuto et al., 2006; Makarewicz et al., 2007; Bosch et al., 2009). While an ephemeral surface toxic bloom of *Microcystis aeruginosa* (microcystin-LR = 5.070 μg/L) was observed in the offshore open waters of Conesus Lake in late September, toxic or nontoxic blooms of *Microcystis* were not observed in the littoral zone.

We expected that the high nutrient loading (P and N) to the nearshore, the well-developed populations of *Spirogyra* and *Zygnema* in the nearshore, the occurrence of *Microcystis* and the toxin microcystin in pelagic waters, and the occurrence of *Dreissena* spp. in Conesus Lake would produce environmental conditions required for the development of *Microcystis* and cyanotoxin production in the nearshore region. Over an 8-year study period, the nearshore bloom of *Microcystis* never occurred. Thus an evaluation of the impact of agricultural runoff and best management practices on the production of toxic-producing blooms in the nearshore was not possible. Availability of micronutrients and macronutrients and toxic metal and UV stress are other possible factors that may induce production of cyanotoxins (Lyck, 2004; Humble et al., 1997; Twiss and Gouvea, 2006; Orr and Jones, 1998; Sivonen, 1990).

In lieu of direct measurements of microcystin concentrations, which are currently expensive and time-consuming, the World Health Organization (WHO, 2003) recommends the use of cell counts of cyanobacteria or chlorophyll *a* concentration as indirect indicators of potential cyanotoxins. However, chlorophyll *a* has proven not to be a good indicator of densities of potentially toxin-producing cyanobacteria in Lake Champlain (Watzin et al., 2006) or in Lake Ontario where only 25% of the variability in microcystin-LR concentration was explained. Fluorometric techniques focused on phycobilisomes offer potential for rapid screening of samples for cyanobacteria because they focus on detecting phycocyanin that is diagnostic for cyanobacteria (Downes and Hall, 1998; Leboulanger et al., 2002; Watzin et al., 2006). Phycocyanin explained 76% of the variation in microcystin-LR concentrations in Lake Ontario. However, the relationship between phycocyanin and microcystin concentrations was weak in Lake Champlain (M. Watzin, Personal Communication, University of Vermont). Phycobilins measured, via fluorescence-based probes, may provide a practical and efficient monitoring tool in tier-based monitoring systems for microcystins (e.g., Watzin et al., 2006) compared to time-consuming cell counts of cyanobacteria or the ineffective measure of chlorophyll *a* concentrations. More investigation on this aspect is necessary as the cellular content of phycobilin varies with taxa and the physiological state of the cell.

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Management of agricultural practices results in declines of filamentous algae in the lake littoral

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ABSTRACT

Filamentous algal cover was quantified during periods of peak biomass from 2001 to 2007 in six littoral macrophyte beds in Conesus Lake, New York (USA). Three of the study sites were adjacent to streams that drained sub-watersheds where extensive agricultural best management practices (BMPs) designed to reduce nutrient runoff were implemented beginning in 2003. Three other study sites were downstream from sub-watersheds where only a few or no BMPs were implemented by landowners. For the sites that received extensive management, comparisons of the Pre-BMP baseline period (2–3 yrs) to the Post-BMP period (4 yrs) revealed that algal cover was statistically lower than baseline in eight of eleven years (72.7%). For the three sites where limited or no management was implemented, the percent cover of filamentous algae was lower than Pre-BMP baseline levels in only three of twelve years (25%). Where major reductions in cover of filamentous algae occurred, positive relationships existed with summer stream loading of nitrate and soluble reactive phosphorus to the nearshore. In some cases only nitrate loading was significantly correlated with percent cover, indicating that the relative importance of nitrogen and phosphorus to algal growth near streams may be determined by the characteristics and land use within each sub-watershed. Agricultural BMPs targeting nutrient and suspended solid runoff can effectively reduce filamentous algal growth locally along the lake littoral zone on a time scale of months to a few years and with moderate commitment of resources. This work offers a new perspective for management of the growing problem of littoral algal growth in the embayments and drowned river mouths of the Great Lakes.

Introduction

The littoral algae of freshwater ecosystems constitute a community dominated by diatoms, filamentous chlorophytes, and cyanobacteria that live in association with a variety of substrates and at times form thick mats that float at the water's surface (Azim et al., 2005, Goldsborough et al., 2005). This community is an important component of the primary production and nutrient dynamics of lakes (Wetzel, 1996).

Littoral algae, including filamentous species such as *Spirogyra* spp., *Zygnema* spp. and *Cladophora* spp., have high light requirements for growth (Graham et al., 1995, Hill and Harvey, 1990), and their peak biomass is typically in summer when surface incident irradiance is high (Wu and Mitsch, 1998, Irfanullah and Moss, 2005). They also respond to nutrient enrichment (TENCate et al., 1991, Fried et al., 2003, Blumenshine et al., 1997) and represent an important sink of nitrogen (N) and phosphorus (P) in a variety of freshwater systems including streams (Biggs, 2000), wetlands (Wu and Mitsch, 1998, Gaiser et al., 2003), subtropical lakes (Havens et al., 1999), and temperate lakes (Irfanullah and Moss, 2005). Nutrient inputs from the catchment basin increase the biomass and productivity of littoral algae (e.g., Makarewicz et al., 2007, Hawes and Smith, 1993) and may shift the algal–macrophyte competitive balance, leading to a decline in macrophyte biomass in lakes undergoing eutrophication (Phillips et al., 1978). Nonpoint agricultural runoff can be a major cause of littoral algal blooms leading to water quality detriment in both lotic and lentic inland waters (Makarewicz et al., 2007, McDowell et al., 2004, Fried et al., 2003). In particular, the continued nutrient augmentation by artificial fertilizers and the ongoing practice of spreading manure by increasingly larger agricultural operations are major contributors to nutrient runoff from watersheds in agriculturally intensive landscapes

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(Schippers et al., 2006). With plenty of nutrients, ample light, and adequate shelter from winds and currents (often provided by macrophytes), littoral algae can bloom to nuisance levels (Fried et al., 2003, Phillips et al., 1978). In areas of restricted water flow (e.g., coves), the extensive die-off and decay that follow a bloom can produce anoxic conditions and blooms of anaerobic bacteria (Fried et al., 2003). Odors emanating from these sites are a major detriment to water quality and to recreational use of inland waters.

In Conesus Lake, western New York, blooms of littoral filamentous algae have been a regular occurrence since the 1920s (Muenschner, 1927). In the early 1970s and prior to the invasion of dreissenids, large masses of the green algal genus *Spirogyra* were consistently present in shallow waters during spring and summer wherever there was macrophyte substrate available for attachment (Forest et al., 1978). Forest et al., (1978) described an incident in which *Cladophora* “festooned” rooted aquatic plants near the mouth of a creek and actually attained comparable masses to macrophytes. The chlorophyte genera *Cladophora*, *Spirogyra,* and *Zygnema* were well represented by species in Conesus Lake decades ago (Savard and Bodine, 1971) and continue to be present today (Makarewicz et al., 2007, D'Aiuto et al., 2006, D'Aiuto, 2003). Even in the context of this history, there is reliable anecdotal evidence that over the last 10 years the biomass of littoral algae in Conesus Lake has reached unprecedented levels, and nuisance accumulations of filamentous algae have been documented within beds of the invasive Eurasian watermilfoil *Myriophyllum spicatum* since 1998 (Bosch et al., 2004, Fig. 1A and B).

In this study we surveyed the littoral filamentous algal communities growing within widely separated Eurasian watermilfoil beds associated with stream effluent from six sub-watersheds of Conesus Lake over seven summer growing seasons (2001–2007). Of the six sub-watersheds, three had received extensive agricultural best management practices (BMPs) designed to reduce nutrient runoff, beginning in 2003. In the three other sub-watersheds only a few or no BMPs were implemented by landowners. The study was conducted to evaluate the potential of watershed-based management in reducing nuisance growth of algae in the lake littoral. Our hypothesis is that filamentous algal cover will decrease in response to declines in nutrient loading downstream from sub-watersheds subject to extensive farm management (see Herendeen and Glazier, 2009), whereas algal cover will remain relatively unchanged downstream from sites where limited or no BMPs were implemented.

Background

With the exception of North McMillan Creek and North Gully, the same sub-watersheds of Conesus Lake, NY (42°46′46.01″N, 77°43′ 0.33″W) were used for this study (Graywod Gully, Sand Point Gully and Cottonwood Gully, Sutton Point Gully and Long Point Gully) and for the BMP nutrient impact studies of Makarewicz et al. (2009). North McMillan Creek was not included in this study because no resident filamentous algae or Eurasian milfoil populations were present in the littoral zone. The North Gully sub-watershed was

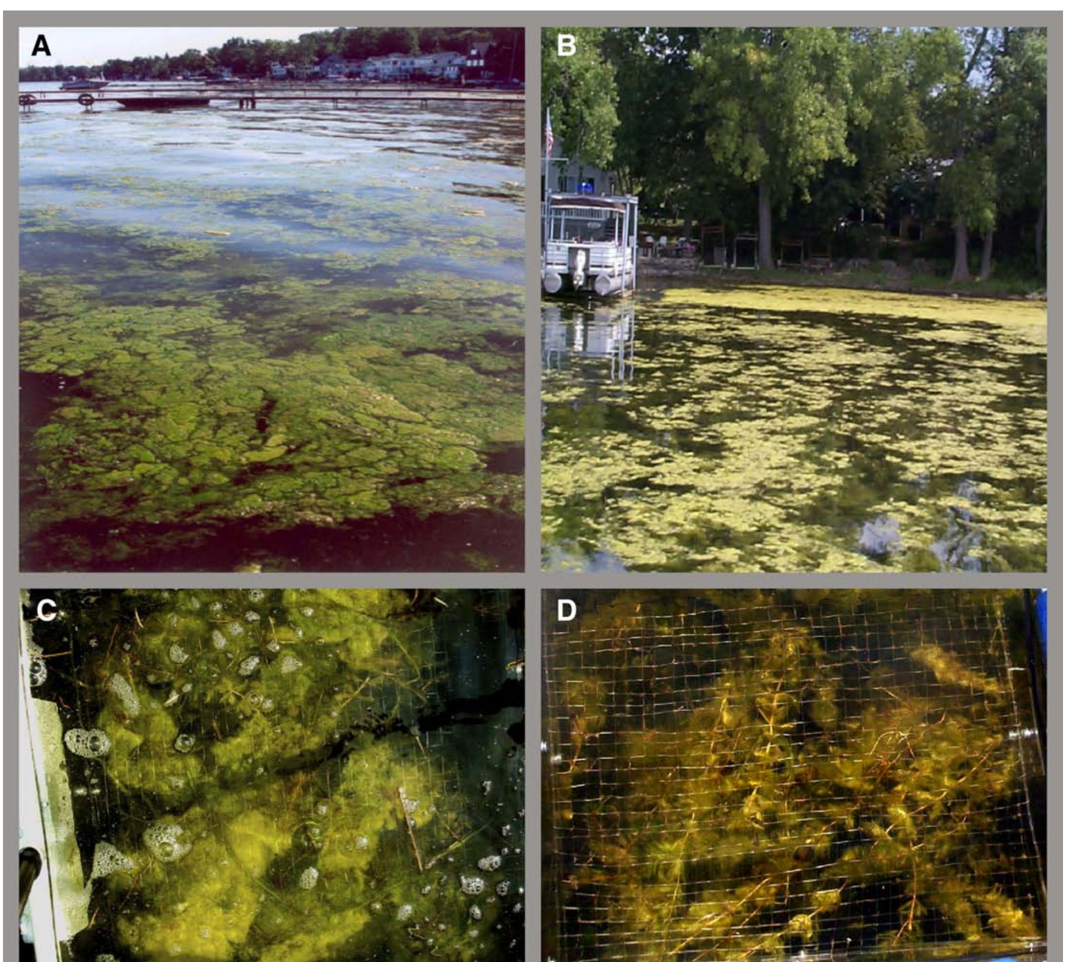

Fig. 1. Filamentous algal mats growing on macrophyte beds in Conesus Lake. (A) Surface growth in shoreline area near the Graywood Gully site in 2001, prior to the implementation of BMPs in the sub-watershed. (B) Second shoreline, near North Gully, with filamentous algae growing on Eurasian watermilfoil at depths of 1–2 m. (C, D) Surface quadrats showing high percent cover (C) and low cover of the watermilfoil canopy (D).

added to replace the North McMillan site. North Gully is a 735-ha sub-watershed that is approximately 45% in dairy and row crop agriculture. Makarewicz et al. (2002) found that total phosphorus runoff from North Gully increased by 150% during major hydro-meteorological events and attributed these increases primarily to agricultural sources. The tributary drains into a shallow cove that is covered by a dense and persistent macrophyte bed. Extensive filamentous algal cover has been observed at this site over the last 10 years. For descriptions of the other watersheds in this study please refer to Makarewicz et al. (2009).

Herendeen and Glazier (2009) describe in detail the BMPs applied to the sub-watersheds and the time line of implementation. A short summary of BMPs follows. In the Graywood Gully sub-watershed where row crops and dairy farming were present prior to 2003, application of a full spectrum of management practices was implemented as part of this project (fertilizer reduction, cover crops, contour strips, reduction in fall and winter manure spreading, various grass filters for runoff from bunker storage of silage and milk house wastes, cows and heifers fenced from the creek and pond). In Cottonwood Gully where row crops predominate, the BMPs were limited to two major efforts: the construction of three water and sediment control basins (gully plugs) and strip cropping designed to retain soils on the watershed. At Sand Point Gully, rotational grazing pens and water troughs were installed; cattle were fenced out of the creek starting in May 2004; and cultural management practices (i.e., changes in crop rotations, tillage practices etc.) were implemented as fallow land, wheat, and an alfalfa grass mix were converted to soybean production acreage starting in 2003.

Agricultural practices in the Sutton Point Gully, Long Point Gully, and North Gully sub-watersheds were modified to a lesser degree by landowners. Dairy cattle were removed from the Long Point Gully watershed in 2003 and a 37% reduction (76.7 ha) in crop acreage occurred by 2004, but manure injection continued throughout the study period (Makarewicz et al., 2009). No physical infrastructure improvements were implemented in the Sutton Point watershed until 2007 when gully plugs were added at the end of the project. However, a cultural BMP was implemented. A significant and increasing portion of the watershed has been placed into alfalfa/grass production since 2002 (37% in 2003 to 60.3% in 2007). At North Gully no major construction nor other farm management activities were initiated over the study period (N. Herendeen, Personal Communication, Cornell Cooperative Extension, P. Kanouse, Personal Communication, Soil and Water Conservation District).

Methods

The six study sites were selected on the basis of their proximity to streams that drain sub-watersheds with high agricultural land use (Makarewicz et al., 2009, D'Aiuto et al., 2006) and the persistence of high cover of filamentous algae on macrophytes (Makarewicz et al., 2007). Three transects were established at each site. Typically, one transect originated within a few meters of each stream and the other two about 100 and 150 m from the stream in the direction of the macrophyte bed (Cottonwood, Sand Point, North Gully and Long Point Cove, Fig. 2) or about 100 m on either side of any stream that was centrally located with respect to the bed (Graywood, Sutton Point; Fig. 2). In the Cottonwood study site, the northern transect was approximately 250 m to the north of the stream. This is more removed from the stream than any other transect in our study yet still well within its influence according to hydrodynamic simulations of major runoff events (Li et al., 2007) and firsthand observations. Except for North Gully, which was first sampled for algal cover in 2002, percent cover in the littoral zone was measured during the 2001–2003 growing seasons before BMPs were established in all sub-watersheds. Consistent with Makarewicz et al. (2009) and Lewis and Makarewicz (2009), the post-BMP period was defined as the 2004–2007 growing seasons.

Determination of percent cover

During the first two summers of this study (2001–2002), the percent cover was determined visually using a 0.25-m^2 PVC quadrat. The quadrat was a square made of PVC tubing and fitted with a grid made of monofilament line. The monofilament grid was held below the surface to integrate the top 10–15 cm of plant growth in the water column. Floats were attached to the sides of the PVC quadrat making it neutrally buoyant during deployment. Starting in 2003, percent cover was determined by digital image analysis. A five-megapixel digital camera was attached to a tripod mounted on corners of the quadrat, and photographs were taken by deploying the imaging system from the side of a small inflatable boat (Fig. 1C–D). The photographs were analyzed for percent surface cover using *ImageJ* (Rasband, 2007). To minimize the measurement error during the image analysis phase, a single investigator analyzed all of the digital images. Blind quality control tests indicated a high degree of reproducibility in the final measurements ($r^2 = 0.98$, $n = 14$ quadrats). Since both the *in situ* and the digital quadrat analyses are based on visual observations of quadrat samples, we saw no reason to expect a systematic error when comparing data obtained from the different methods.

Monitoring was conducted between June and early September each year. Quadrat measurements were initiated when monitoring observations indicated that a high surface biomass had developed. This occurred typically between July (earliest on the 28th) and August (latest 21st) each year. Follow-up qualitative surveys were conducted to evaluate the condition of the plants and possible changes in biomass. When growth continued later into the season, additional quadrat measurements were taken as necessary. At each study site, at least four and typically more than six replicate quadrats were measured over depths of 1 and 3 m within a few meters of three transects that were established perpendicular to the shoreline.

Trends were analyzed individually for each of the six sites by comparing algal cover from each Post-BMP year (2004–2007) to pooled data from the corresponding Pre-BMP baseline period (2001–2003). Sample sizes for each of the baseline years were different from each other. Pooling the whole data set would result in years with higher sample sizes having a greater influence on the distribution of data. We corrected for years with larger sample sizes by systematically selecting a subset equaling the data points for the year with the smallest sample size. For example, if year one and two had sample sizes of 12 and 24 quadrats, every other sample from year two was used for the compilation of data points. In determining Pre-BMP baseline cover, data from a particular site would be excluded if the macrophyte substrate did not develop to levels that approached typical biomass and distribution for a given site. For example, in 2002 metaphyton data were not incorporated into the analysis of Pre-BMP cover at Sutton Point Gully because the macrophyte bed never completely developed.

The percent cover data were significantly different from the normal even after they were transformed to $\log_{10}$ values. Therefore a Kruskal–Wallis One Way Analysis of Variance on Ranks was used to compare distributions of log-transformed data (*SigmaStat3.5*, SYSTAT Software Inc.). In cases where the ANOVA on Ranks produced statistically significant differences, a Dunn's Method of multiple comparisons versus a control group was used to compare each year to its corresponding Pre-BMP baseline. The output for a Dunn's Method *post hoc* test in the *Sigmastat* software package indicates whether a test *p*-value is less than 0.05, but no specific *p* values are reported. Therefore in the remainder of this paper, the results of all Dunn's Method tests are reported only as significant ($p<0.05$) or not significant ($p>0.05$). The percent cover data for each season was considered to be an independent replicate rather than a repeated measure for each site. This assumption is reasonable

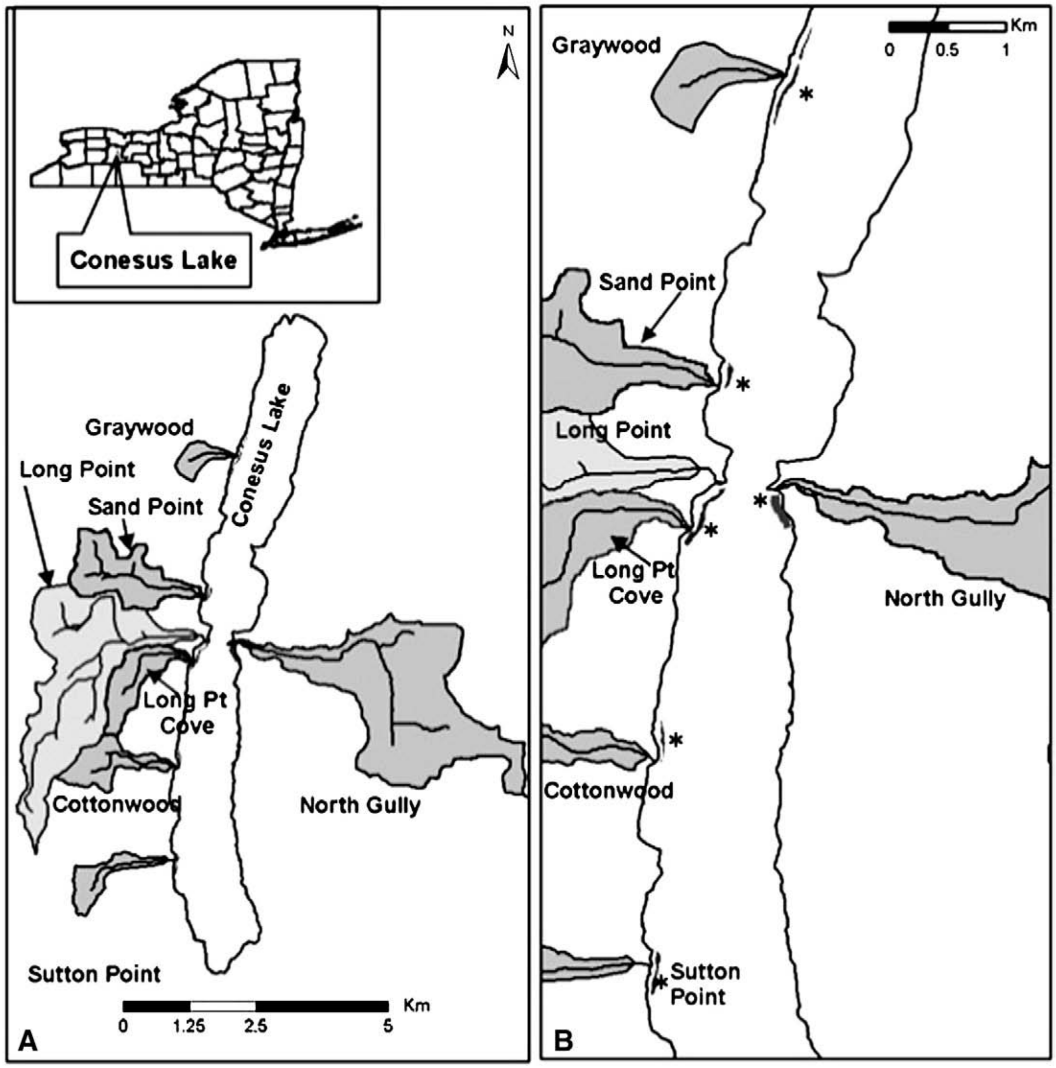

Fig. 2. Conesus Lake, New York (insert) showing the study sub-watersheds (A) and the macrophyte beds (highlighted with an asterisk) where filamentous algal cover was measured from 2001 to 2007 (B).

considering the rapid response of filamentous algae to nutrient delivery and the number of different variables that potentially affect the growth dynamics of this community.

Nitrate and soluble reactive phosphorus loading

Cottonwood Gully, Graywood Gully, and Sand Point Gully were monitored continuously from 1 Sept 2002 to 31 Aug 2007 along with Long Point Gully and Sutton Point Gully streams (Makarewicz et al., 2009). The North Gully tributary was sampled weekly during nonevent discharge and during most major rain events from 18 May 2004 to 4 Sept 2007. Makarewicz et al. (2009) provide details on the sampling procedures used to quantify stream discharge, analyte concentration, and analyte flux and report annual trends over the 5-year period for all but the North Gully tributary. For the present study, we determined total loading and concentrations of nitrate (NO_3-N) and soluble reactive phosphorus (SRP) for the summer period of highest filamentous algal biomass (July and August) from 2003 to 2007. Molar ratios for soluble N and P were calculated from average daily stream concentrations of NO_3-N and PO_4-P. Linear regression analyses were used to relate nutrient loading trends to trends in percent cover of filamentous algae (*SigmaStat3.5*, SYSTAT Software Inc.).

Two tributaries potentially affect the Long Point Cove (LPC) macrophyte bed and associated filamentous algal community. To the south there is an unnamed small gully and stream (referred to as LPC Gully) that drains the 114.2-ha LPC Gully sub-watershed (Fig. 2). Approximately 50% of the sub-watershed consists of either woodland cover or vacant fields and the remainder is in row crop agriculture. In the fall 2005, gully plugs and accompanying drainage tile were installed in a small field encompassing approximately 10% of the watershed just up-slope from the middle sections of the tributary (P. Kanouse, Personal Communication, Livingston County Soil and Water Conservation District). The remaining 40% of the watershed has been in row crops, primarily corn since 2003. To our knowledge there has been no other significant agricultural management in the area. North of the LPC sub-watershed is the 605-ha Long Point Gully sub-watershed. The latter feeds into a large tributary that drains into a prominent peninsula (Long Point), forming the northern boundary of Long Point Cove (Fig. 2). The three transect sites at Long Point Cove were located along the southern portion of the cove, one just a few meters to the north of LPC Gully. The other transects were 50 m and 100 m to the north. Our direct observations of sediment plumes during two major runoff events indicate that a portion of the Long Point Gully discharge plume (10–20%) flows southward into Long Point Cove. This loading is not likely to have as great an influence on our transect areas as LPC Gully. Unfortunately, the limited samples collected on discharge and analyte flux for LPC Gully are not sufficient for an analysis of loading trends.

Results

Dynamics of filamentous algal community

Blooms of filamentous algae, primarily *Spirogyra* and *Zygnema*, grew within the three-dimensional habitat comprised by macrophyte beds and as surface mats on macrophyte canopies throughout Conesus Lake (Fig. 1). Community development was highly seasonal. Initial bursts of biomass were sometimes evident in late June but more typically in early July within days to 1 to 2 weeks after major rain events. Surface cover increased soon thereafter and reached peak development in late July or August as biomass accumulated and as decreasing lake water levels exposed more of the filamentous algal growth on the surface. Within the general period of high biomass, there was moderate synchrony in the onset of peak percent cover between sites. The timing of collapse of the blooms was more variable. In some cases filaments turned brown (a sign of senescence) and began to decompose within a week or two of estimated peak surface cover. Community biomass decreased soon thereafter, although much of the degraded organic material could remain on the rooted macrophytes into winter. In other cases, healthy filaments, high biomass, and surface cover persisted into late August. There were differences in the extent of community development between years. In 2005 for example, North Gully was the only site that developed extensive cover, whereas 2007 was characterized by relatively high percent cover at all the study sites (Fig. 3B).

The spatial distribution of surface cover was very patchy within each study area, possibly due to variability in growth and spatial differences in conditions that promoted biomass accumulation (e.g., hydrodynamics). This patchiness is reflected in frequency distributions of percent cover that deviated from the normal distribution in almost all of the study sites. Typically the highest surface algal cover

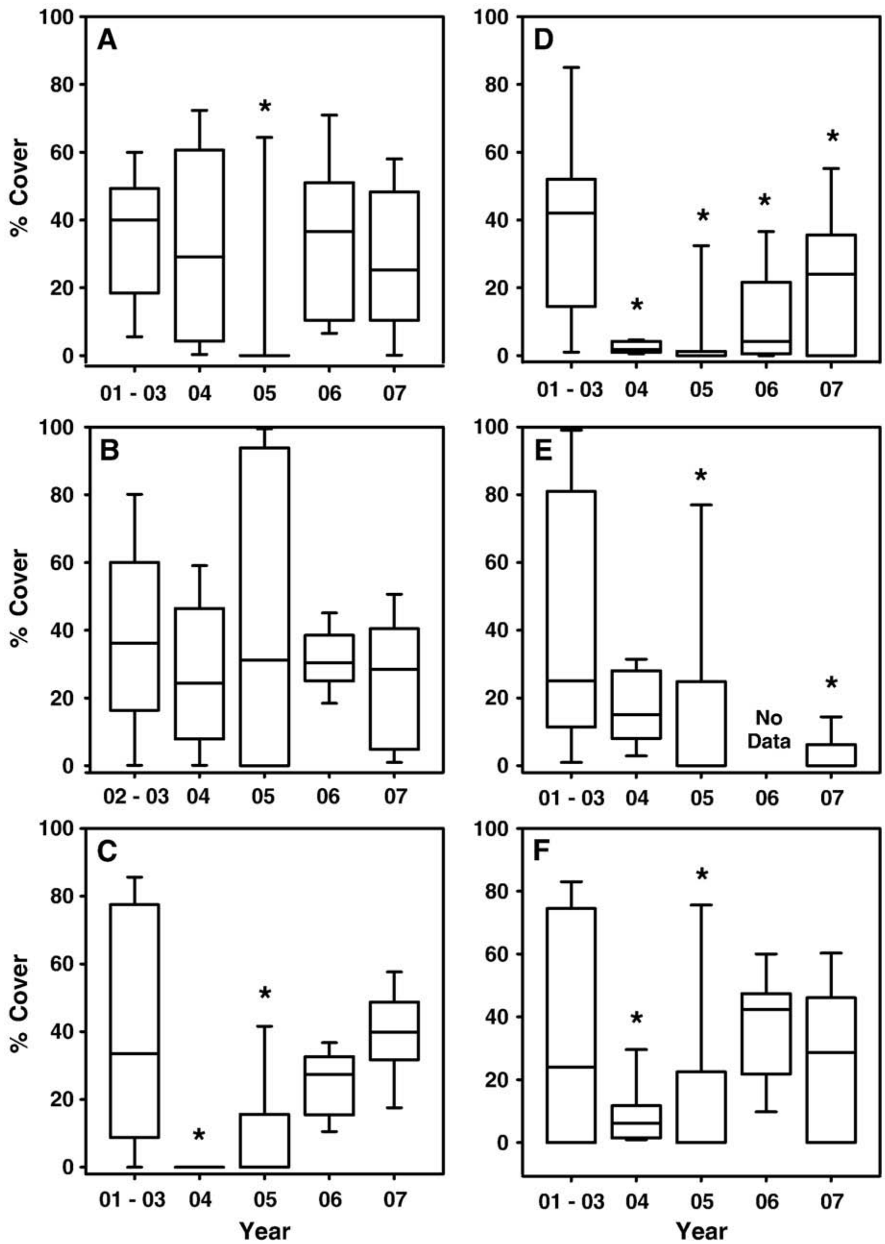

Fig. 3. Box plots of the surface cover of filamentous algae growing on watermilfoil-dominated beds in Conesus Lake: (A) Sutton Point, (B) North Gully, (C) Long Point Cove, (D) Cottonwood Gully, (E) Graywood Gully, and (F) Sand Point Gully. Cottonwood Gully, Graywood Gully, and Sand Point Gully received structural and cultural BMPs as part of this project. Comparisons between individual experimental years (Post-BMPs) and baseline (Pre-BMPs) were made using Kruskal–Wallis ANOVA on Ranks followed by Dunn's *post hoc* tests. The box plots show the 25%–75% confidence limits and the bars indicate the 10%–90% confidence limits. Asterisks indicate values that were significantly different from baseline.

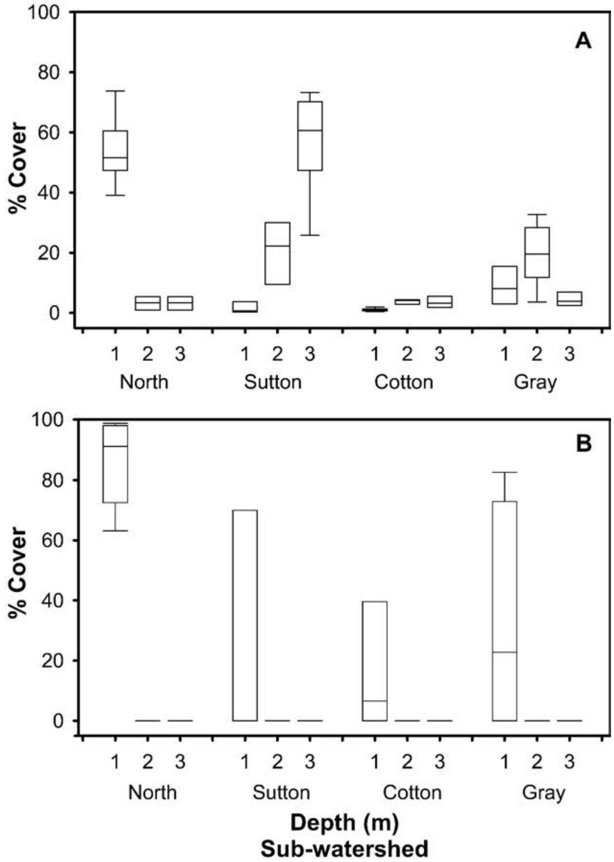

Fig. 4. The percent surface cover by filamentous algae along the shoreline in 2004–2005. (A) Trends for 2004 showing the high cover over a depth of 3 m at Sutton Point, corresponding to a distance of approximately 20 m from shore. At other sites the maxima were over 1–2 m. (B) In 2005 there were no littoral filamentous algae mats beyond a depth of 1 m.

occurred over depths of 1–2 m, less than 50 m from shore (Fig. 4). Algal cover was generally low or absent along the outer margins of the macrophyte bed (3-m depth, <75 m from the shoreline; Fig. 4A). One consistent exception to this trend was the central transect at Sutton Point, where the 3-m depth region was less than 20 m from the shoreline. Algal biomass accumulation over depths of 2 m (usually 25–50 m from the shoreline) was variable. In 2004 for example, we found very little growth over the 2-m portions of transects (Fig. 4B). However, it was more typical to find significant cover in this zone, and in some exceptional cases there was higher cover over the 2-m depths than closer to shore. Because of these inconsistencies, the analyses of percent cover trends described in the following sections are based on pooled data from depths of 1–2 m for all but the Sutton Point stream transect analysis, which incorporates the 3-m samples (Fig. 4).

Trends in algal cover and loading

In the Sutton Point Gully, North Gully, Long Point Gully and Long Point Cove sub-watersheds, only a few or no BMPs were implemented during the study period (Herendeen and Glazier, 2009). In general, the percent cover of filamentous algae from 2004 to 2007 (Post-BMP period) downstream from those sites was statistically lower than the 2001–2003 Pre-BMP baseline period only 25% of the time (Fig. 3A–C); that is, percent cover was not significantly different than 2001–2003 baseline 75% of the time. At North Gully, percent cover was not statistically different from baseline values in any of the years sampled (ANOVA, $p=0.13$). Statistically significant decreases from baseline (Pre-BMP) percent cover were observed at Sutton Point in 2005 (ANOVA, $p<0.001$, $df=4$, Dunn's Method) and at Long Point Cove in 2004 and 2005 (ANOVA, $p<0.001$, $df=4$, Dunn's Method) (see Fig. 3A–C). As mentioned earlier, data for 2002 were not incorporated into the analysis of Pre-BMP cover at Sutton Point Gully because the macrophyte bed never fully developed in 2002.

At Long Point Cove there was no measurable surface cover of filamentous algae in 2004 (Fig. 3C). The macrophyte bed developed late in the 2004 season (late July and August). Consequently, for most of the summer there was no macrophyte substrate to support algal surface growth. In 2005 the macrophyte bed had developed by June, and the percent cover of filamentous algae in August was high in some quadrats (78%) but still low overall, with growth absent from much of the habitat. Overall, trends in nutrient loading from Long Point Gully were not related to percent cover in Long Point Cove (Table 3). Analysis of these trends may be confounded by the potential impact of loading from the LPC Gully tributary (Fig. 2). In general, at the Long Point Cove, Sutton Point, and North Gully study sites there appeared to be no trend up or down in filamentous cover over the study period (Fig. 3A–C).

At Cottonwood Gully, Graywood Gully and Sand Point Gully, a number of BMPs were implemented during the study period (Herendeen and Glazier, 2009). The percent cover of filamentous algae at these locations from 2004 to 2007 (Post-BMP period) was statistically lower than the 2001–2003 baseline median for 72.7% of the time (Fig. 3D–F); that is, percent cover after the implementation of BMPs was not significantly lower than Pre-BMP baseline percent cover only 27.3% of the time. For example, Pre-BMP median cover values at Cottonwood were 71% in 2001 and 42% in 2003. In 2002 the median cover was only 3% and most of the area was free of filamentous algae. Because there was sufficient macrophyte biomass to provide ample substrate for algal growth, the data was not excluded from the analysis. Beginning in 2004, there was a statistically significant decrease in median cover (ANOVA, $p<0.001$, $df=4$, Dunn's Method) to 1.8% (Fig. 3D). Percent cover remained significantly lower in 2005 (median of zero), when most of the macrophytes were free of filamentous algae, and increased slightly but was still significantly lower in 2006 to a median cover of 4.1%, with a few quadrats reaching 46% cover. In 2007 there was an increase in biomass, but overall

Table 1
Loading of nitrate (NO_3-N+NO_2-N) and soluble reactive phosphorus (SRP) from five tributaries draining agricultural sub-watersheds around Conesus Lake.

Tributary	Year	Nitrate (kg)	SRP (kg)
Sutton Point Gully	2003	62.49	0.59
	2004	25.05	0.51
	2005	14.18	0.79
	2006	45.44	1.28
	2007	24.85	0.41
Long Point Gully	2003	984.32	4.91
	2004	840.89	4.67
	2005	862.23	7.86
	2006	2955.12	20.59
	2007	48.97	1.33
Cottonwood Gully	2003	256.87	3.12
	2004	69.31	1.80
	2005	82.81	1.92
	2006	69.49	1.79
	2007	33.84	1.33
Graywood Gully	2003	463.74	5.50
	2004	1627.49	15.31
	2005	54.45	1.43
	2006	32.25	1.28
	2007	10.32	0.29
Sand Point Gully	2003	276.38	3.01
	2004	80.76	1.58
	2005	95.10	2.89
	2006	413.01	1.72
	2007	8.57	0.72

The numbers are the sum of daily loading (kg/d) during July and August, the months of peak production for filamentous algae. Cottonwod Gully, Graywood Gully, and Sand Point Gully received BMPs as part of this project.

percent cover remained significantly lower than Pre-BMP baseline values (median cover 24.0%, many quadrats with >40% cover). Similarly at the managed Graywood Gully site, the median percent cover of filamentous algae was 25% for the Pre-BMP period (2001–2003) and 14% in 2004. In 2005 and 2007, the median values were 0% (Fig. 3E) and statistically lower than Pre-BMP baseline values in both sites (ANOVA, $p<001$, $df=3$, Dunn's Method). Data are not reported for 2006 as the Eurasian watermilfoil canopy collapsed by late July before the surface mats of algae had developed. Overall, median percent cover was lower than Pre-BMP baseline cover values in 2 of the 3 years surveyed. At Sand Point Gully, significantly lower percent cover of filamentous algae was observed in 2004 and 2005 (ANOVA, $p<0.001$, $df=4$, Dunn's Method), but in 2006–2007 cover increased to Pre-BMP values. In general, at Graywood Gully, Cottonwood Gully, and Sand Point Gully, there were significant reductions in percent cover of filamentous algae after the implementation of management practices in the respective sub-watershed (Fig. 3D–F).

The decreases in algal cover relative to Pre-BMP baseline values in Cottonwood and Graywood were generally consistent with trends in nutrient loading from the corresponding tributaries. At Cottonwood, where we saw the largest decreases in percent cover, July–August loading of NO_3 and SRP decreased by 73% in 2004 and remained low for the remainder of the study period (Table 1). In Graywood, with the exception of 2004, Post-BMP loading of NO_3 and SRP were at least 88% and 74% lower than 2003 values, respectively. In 2004 heavy rains produced unusually high discharge and nutrient loading from Graywood Gully, yet filamentous cover was low possibly due to high water movement and/or high water turbidity in the stream outlet area.

The relationship between the concentration of nutrients and algal cover was less consistent (Table 2). In particular, SRP concentrations did not follow any obvious trends that could be related to algal cover.

Table 2
Median concentrations of nitrate (NO_3-N+NO_2-N), soluble reactive phosphorus (SRP, PO_4-P), and N:P molar ratios for July–August loading.

Tributary	Year	NO_3-N concentration (mg/L)	PO_4-P concentration (μg/L)	N:P molar ratio
Sutton Point Gully	2003	2.30	37.06	161.43
	2004	0.75*	31.75	50.86
	2005	0.66*	40.93	36.12
	2006	1.04	30.87	71.87
	2007	1.88	42.64	107.25
North Gully	2004	0.39	14.70	60.36
	2005	0.34	12.17	58.71
	2006	0.35	13.50	62.85
	2007	0.25	19.29	31.23
Long Point Gully	2003	8.59	47.98	352.91
	2004	3.80	69.14	238.83
	2005	4.89*	58.85	163.45
	2006	2.80*	53.91	154.06
	2007	2.24*	60.80	81.45
Cottonwood Gully	2003	1.95	60.21	57.67
	2004	1.06*	64.67	37.04
	2005	1.35	59.64	51.60
	2006	1.29	43.33*	69.15
	2007	1.21*	52.20	48.37
Graywood Gully	2003	7.34	171.40	72.24
	2004	7.34	66.02*	201.92
	2005	4.59	139.12	76.84
	2006	1.37*	115.70	41.64
	2007	1.24*	85.95*	25.76
Sand Point Gully	2003	2.48	57.10	49.48
	2004	0.97	29.96*	72.97
	2005	1.27	67.75*	34.46
	2006	1.47	53.98	56.94
	2007	0.86*	72.21*	24.32

The Cottonwood Gully, Graywood Gully, and Sand Point Gully sub-watersheds received BMPs as part of this project. Asterisks indicate statistically significant differences (Kruskal–Wallis ANOVA on Ranks, Dunn's *post hoc*). Each year from 2004 to 2007 was compared to its corresponding 2003 (Pre-BMPs) values.

Table 3
Regression analyses comparing trends in July–August loading of nitrate (NO_3-N+NO_2-N) and soluble reactive phosphorus (SRP) from five tributaries and median percent cover of filamentous algae downstream in the littoral zone of Conesus Lake from 2003 to 2007.

Comparison	Regression coefficient (r^2)	Standard error	t	p
Sutton Point Gully nitrate × % cover	0.80	0.21	4.1	0.03
Sutton Point Gully SRP × % cover	0.00	0.01	0.1	0.92
Long Point Gully nitrate × % cover	0.00	18.4	−0.18	0.87
Long Point Gully SRP × % cover	0.00	0.04	−0.56	0.62
Cottonwood Gully nitrate × % cover	0.92	0.03	6.8	0.01
Cottonwood Gully SRP × % cover	0.82	0.01	4.4	0.02
Graywood Gully nitrate × % cover	0.64	<0.01	2.5	0.13
Graywood Gully SRP × % cover	0.70	0.34	2.8	0.11
Sand Point Gully nitrate × % cover	0.00	0.09	0.52	0.64
Sand Point Gully SRP × % cover	0.00	0.02	−0.52	0.64

Nitrate concentrations in 2005 were statistically lower than 2003 values in both Long Point Gully and Sutton Point Gully and the corresponding N:P ratios were reduced by more than 50% at both sites (Table 2). These decreases coincided with algal cover values that were also well below medians for the 2001–2003 Pre-BMP baseline period (Fig. 3).

Overall there were positive relationships ($r^2>0.64$) between percent algal cover and nutrient loading (kg) for the July–August period at Graywood (NO_3, SRP), Sutton (NO_3 only) and Cottonwood Gully (NO_3, SRP) (Table 3). At Sand Point and Long Point regression coefficients (r^2) were zero. At North Gully, loading was not measured for the entire July–August period. In terms of concentrations, only NO_3 from Cottonwood Gully was significantly related to filamentous cover ($r^2=0.84$, $p=0.02$).

Discussion

In recent years the growth of periphytic algae on sediment and rocks in Conesus Lake (e.g., *Cladophora sp.*) has generally been restricted to very shallow areas along the shore. Other species of littoral filamentous algae are predominant as epiphytes and surface mats of metaphyton within macrophyte beds dominated by Eurasian watermilfoil (D'Aiuto et al., 2006). The macrophyte beds provide a suitable habitat, may provide nutrients from the sediments to its epiphytes (Jackson et al., 1994), and in general, offer an expansive substrate that extends the area of filamentous algal growth well offshore.

Adult zebra mussels (*Dreissena polymorpha*), first noticed in Conesus Lake in 1994 with large populations evident by 1998, do occur in the milfoil beds (Bosch et al., 2001) and may be a potential cause of the nuisance accumulations of littoral filamentous algae as they effectively recycle nutrients, increase the amount of light penetrating into the water, and selectively feed on diatoms and other algae that potentially compete for light, substrate, and dissolved nutrients (Pillsbury et al., 2002). The presence of zebra mussels has been correlated with increased growth of littoral algae and rooted macrophytes in the Great Lakes (Pillsbury et al., 2002, Skubinna et al., 1995, Lowe and Pillsbury, 1995) and in some of the larger Finger Lakes of New York (Zhu et al., 2006). However, there are areas of the Conesus Lake littoral located away from streams that generally have much lower filamentous algal growth despite the presence of large zebra mussel populations locally (Bosch et al., 2004). Moreover, large populations of *Spirogyra*, *Zygnema*, and other filamentous algae existed in the littoral zone (Muenschner, 1927) and especially near streams (Forest et al., 1978) long before dreissenids arrived in Conesus Lake. There is some anecdotal evidence that filamentous algal biomass may have increased after the zebra mussel invasion, but no quantitative data exists to confirm this.

The present study was designed to determine whether agricultural BMPs in small sub-watersheds could decrease nutrient losses from the watershed and in turn reduce the growth of filamentous algae downstream in the lake littoral zone. It was not possible to make comparisons of percent cover or nutrient loading between sites because the individual sub-watersheds and shoreline areas had unique individual characteristics (i.e., they were not replicates). There was also regional variation in runoff and stream water chemistry due to microclimates and, despite being primarily in agriculture, land use varied in each of the sub-watersheds (e.g., dairy farming versus row crops). Finally, spatial and temporal variations in light availability, temperature, and other known forcing agents of algal growth could not be controlled. Despite these potentially confounding forces, we provide evidence of differences in site-specific trends that are best explained by changes in stream nutrient loading related to agricultural BMPs.

At the three littoral sites associated with sub-watersheds that received minimal or no BMPs (Long Point Gully, LPC Cove Gully, Sutton Point Gully and North Gully), algal cover was statistically lower than baseline in only three of twelve sample seasons. At two littoral sites located downstream from sub-watersheds in which extensive BMPs were implemented by our project, considerable reductions in filamentous algal cover were observed compared to pre-management levels. At the Graywood Gully site, the Pre-BMP median cover of 25% had decreased to 14% by 2004 (Post-BMP) and was statistically below these levels in 2005 and 2007. There was no data for 2006 due to the collapse of the macrophyte bed, which is discussed in Bosch et al., (2009). At the Cottonwood Gully site, the median algal cover decreased from a Pre-BMP median of 42% to median values of 0–4.1% Post-BMP (2004–2006). Surface cover was relatively high (median 24%) in 2007 at Cottonwood Gully but also in several other streams and may be related to conditions during this very dry summer (Herendeen and Glazier, 2009) during which stream chemistry was influenced by ground water (Makarewicz et al., 2009).

In general, trends in stream loading of NO_3 and SRP for July–August from 2003 to 2007 are significantly correlated with the annual loading trends (correlation coefficient = 0.83, $p > 0.01$ for NO_3 and 0.64, $p = 0.01$ for SRP) reported by Makarewicz et al. (2009), who concluded that agricultural BMPs were responsible for major decreases in nutrient loading from Graywood Gully, Cottonwood Gully, and Sand Point Gully. Sand Point Gully is an interesting situation as no major structural BMPs were implemented here, yet statistically significant reductions in percent cover of filamentous algae were observed in 2004 and 2005. At Sand Point Gully, rotational grazing pens and water troughs were installed, and cattle were fenced out of the creek starting in May 2003. Two gully plugs and tiles were also installed in a small portion of the watershed in November of 2002 prior to the beginning of this project. As indicated by Makarewicz et al. (2009), we did not expect a large impact of management practices as the structural BMPs implemented, rotational grazing and "gully plugs", accounted for less than 9.5% of the entire watershed. Starting in 2004 however, several cultural BMPs (manure injection, strip cropping) were implemented in significant portions of the watershed. Makarewicz et al. (2009) observed a significant 44% reduction in NO_3 concentrations by 2005 with no further significant changes over the study period. The reductions observed in percent cover of filamentous algae from a median of 42% during the Pre-BMP baseline period (2001 to 2003) to 6.2% (2004) and to 0% (2005) coincided with reductions in "annual" NO_3 values but not July–August NO_3 loading or concentrations. By 2006, percent cover of filamentous algae increased from 2004 and 2005 and coincided with a large increase in summer (July and August) NO_3 load but not with SRP.

While a number of variables may affect the growth of filamentous algae, the principal agent that drives the nuisance growth in Conesus Lake seems to be high nutrient loading. Makarewicz et al. (2007) demonstrated that stream effluent entering Conesus Lake from sub-watersheds that were primarily in agriculture contained high concentrations of SRP and NO_3 that stimulated the growth of littoral algae, with biomass doubling times in the range of hours to days. These results are in agreement with the general observation that areas of nuisance filamentous growth around Conesus Lake were associated with streams that drained predominately agricultural sub-watersheds (D'Aiuto et al., 2006). Experiments in flow-through chambers indicated that the biomass of filamentous algae increased by an average of 384% over a 3-day period when provided with stream water rich in P and N and only by an average of 206% when grown in lake water (Makarewicz et al., 2007). These observations are consistent with the studies of Wu and Mitsch (1998) who demonstrated that filamentous algae in freshwater wetlands form dense mats near areas of inflow with up to 86% cover. Furthermore, these dense assemblages acted as filters that limited the delivery of P to other areas and consequently restricted plant growth downstream from areas of inflow (Wu and Mitsch, 1998).

Stoichiometric N:P cellular ratios and their relationship to nutrient availability for plant growth in aquatic ecosystems have been studied extensively for microalgae but only recently in filamentous macroalgae. D'Aiuto et al. (2006) and Makarewicz et al. (2007), working primarily with species of *Spirogyra* and *Zygnema* in Conesus Lake, carried out a series of NO_3 and SRP enhancement experiments in which only SRP addition to ambient lake water promoted significant enhancement of filamentous algal growth. The molar ratios of soluble N:P were as low as 8.9 in many of these trials that resulted in vigorous growth. Such low ratios were rarely measured for the streams in the present study, giving further credence to the conclusion that P was the limiting nutrient for littoral algal growth. In contrast to the Conesus Lake results, studies by Townsend et al. (2008) on a tropical species of *Spirogyra* that inhabits streams and rivers of Australia indicated that both N and P could alternately limit growth. Optimal cellular N:P ratios for *S. fluviatilis* were 87:1, indicating a higher N requirement than predicted by the Redfield ratio of 16:1 that is typical of microalgae. Algal samples were collected from a variety of habitats, and among these approximately half appeared to be N limited. Locally high N concentrations in other habitats resulted in P limitation (Townsend et al., 2008).

While the present study did not test directly for P and N limitation, analysis of N and P loading relative to percent cover in different watersheds does explain some of the patterns in our data. For example, at Sand Point Gully percent cover decreased with decreasing levels of annual NO_3 and then increased in 2006 when summer NO_3 levels increased. Similarly at Sutton Point Gully, percent cover was positively related to NO_3 loading ($r^2 = 0.80$) but not to SRP. The lowest metaphyton percent cover measured for this site (2005) corresponded to the lowest July–August N:P molar ratios recorded in Sutton Point Gully. At Graywood and Cottonwood Gully, filamentous algal cover correlated positively with both P and N. At Long Point Gully, there was no significant correlation between percent cover and NO_3 or SRP loading, and this may represent the influence of Long Point Gully Cove where stream water chemistry was not measured. The possibility that N limitation may be controlling filamentous algal cover in some sub-watersheds is interesting and plausible. In addition to the work of Townsend et al., (2008) showing variations in nutrient limitation on *Spirogyra*, mesocosm experiments with benthic algae, organisms that are often a few centimeters below surficial filamentous algae in the water column, have demonstrated that large amounts of N and P are removed from the water column and that both may be limiting for growth (Havens et al., 1999, McDougal et al., 1997). The design and consequently the results of the watershed manipulation experiments reported here do not allow or indicate whether N or P or both were limiting to littoral algal growth in Conesus Lake while correlation analyses have to be viewed as suggestive. Without additional data we can only speculate on this topic, but it is possible that in Conesus Lake, the relative importance of N and P to growth of littoral algae near streams may be determined locally by the characteristics and land use within each individual sub-watershed.

The results of this study are consistent with previous work not only in showing that nutrient losses from the catchment basin can have a major effect on benthic algal growth in freshwater ecosystems but also in demonstrating the potential impact of agricultural runoff on aquatic plant growth (Beman et al., 2005, Makarewicz and Howell, 2007, McDowell et al., 2004). More importantly, evidence is presented indicating that agricultural BMPs targeting nutrient runoff can effectively reduce littoral zone filamentous algal growth on a time scale of months to a few years and with moderate commitment of resources. As such, this work offers a new perspective for management of the growing problem of littoral plant growth in the embayments and drowned river mouths of the Great Lakes (Makarewicz and Howell, 2007) and lakes in general that may be more feasible than conventional efforts that typically attempt to manage plant growth on a lake-wide scale.

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Responses of lake macrophyte beds dominated by Eurasian watermilfoil (*Myriophyllum spicatum*) to best management practices in agricultural sub-watersheds: Declines in biomass but not species dominance

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ABSTRACT

Long-term studies of macrophyte beds growing near streams in Conesus Lake, New York, have revealed a high biomass and continuing dominance of the invasive rooted species Eurasian watermilfoil (*Myriophyllum spicatum*). We tested whether agricultural best management practices (BMPs) designed to reduce tributary nutrient and soil loss from the watershed could reduce populations of Eurasian watermilfoil downstream in the lake littoral. Six macrophyte beds were monitored during a 3-year baseline period (2001–2003) prior to the implementation of BMPs and for a 4-year experimental period after a variety of agricultural BMPs were implemented in three sub-watersheds. For three macrophyte beds downstream from sub-watersheds managed as part of our project, quadrat biomass decreased by 30–50% and was statistically lower than Pre-BMP baseline values in 7 of 11 experimental sample years. Biomass loss primarily in the form of the dominant Eurasian watermilfoil ranged from 6.2 to 10 t wet weight for each bed. The declines in biomass coincided with significant annual and January–August decreases in the concentrations and fluxes of dissolved nutrients, total phosphorus, and total suspended solids in nearby streams. For three macrophyte beds downstream from watersheds in which landowners applied less extensive or no new agricultural management, biomass was statistically indistinguishable from Pre-BMP baseline values in all 12 experimental sample years. Milfoil remained the overwhelmingly dominant species at all sites during the entire study period. These results provide impetus for the use of watershed nutrient management to control the nuisance growth of Eurasian watermilfoil on a local scale in the lake littoral.

Introduction

The lake littoral, including the shoreline and shallow nearshore zones, serves as the interface between the drainage basin and the open waters that constitute most of the of the lake volume. In many lakes, the littoral region is dominated both structurally and ecologically by aquatic vascular plants, which play a fundamental role in ecosystem productivity by sequestering nutrients, stabilizing lake sediments, buffering wave action, and serving as food, shelter and refuge for fish fry, insects and other aquatic invertebrates (e.g., Wetzel 2001, Barko et al. 1991, Carpenter and Lodge 1986). Littoral aquatic plant communities, however, frequently develop excessive biomass, causing major changes in ecosystem processes and creating nuisance conditions that affect navigation and interfere with the recreational use and aesthetic quality of inland waters. Two of the major causes of use impairment by aquatic macrophyte communities are eutrophication and species invasions.

Over the last 60 years, the invasive freshwater macrophyte Eurasian watermilfoil (*Myriophyllum spicatum*) has spread throughout most of the U.S. and Canada, becoming one of the most ubiquitous and troublesome species in lake ecosystems (Smith and Barko 1990, Madsen et al. 2008). Eurasian watermilfoil is a submersed perennial vascular macrophyte that requires ample light (Skubinna et al. 1995, Madsen et al. 1991a) and proliferates in waters rich in nutrients (Wakeman and Les 1994a). Growing preferentially at depths of 1–4 m (Nichols and Shaw 1986), milfoil competes aggressively with native plant species for light and sediment nutrients (Wakeman and Les 1994b), forming densely branched canopies that can exceed 80% surface cover and effectively limit the growth of other species by shading. Overall, the presence of milfoil in most lakes is responsible for reducing biomass, species richness, and biodiversity of native

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vascular macrophytes (Madsen et al. 1991b, Bowman and Mantai 1993, Boylen et al. 1999) and associated invertebrate communities (Chruvelil et al. 2002).

Extensive efforts and resources have been dedicated to development of effective strategies for Eurasian watermilfoil management (see Nichols et al. 1988 and Madsen 1997 for reviews). These include mechanical treatment with weed harvesters and dredges, water level drawdown in reservoirs, shading, chemical control of internal loading (Mesner and Narf 1987), herbicide treatments (e.g., Getsinger et al. 2002), and biological controls, including pathogens such as fungi (Smith et al. 1989), herbivorous milfoil weevils *Euhrychiopsis lecontei* (Newman et al. 1996), and aquatic moths of the genus *Acentra* (Gross et al. 2001). These control techniques are generally short-lived, expensive (Smith and Barko 1990), and may be detrimental to non-target populations or the ecosystem. More importantly, they do not address what is fully recognized as the ultimate cause of excessive plant growth in inland waters: nutrient loading from the surrounding watershed (e.g., Carpenter et al. 1998, McDowell et al. 2004).

With the exception of the work of D'Aiuto et al. (2006) and Makarewicz et al. (2007), an extensive scientific literature search did not identify any studies in which the connection between runoff from the watershed and Eurasian watermilfoil growth was tested. This paucity of watershed nutrient management studies is not unreasonable, considering the number of reports indicating that Eurasian watermilfoil and many other macrophytes can potentially meet their nutrient requirements by uptake through the roots (e.g., Nichols and Keeney 1976a, Barko et al. 1991). Lake sediments accumulate vast and seemingly unlimited stores of nitrogen (N) and phosphorus (P) that can be readily available to rooted plants in various dissolved forms in pore or interstitial waters (Wakeman and Les 1994b). Carignan and Kalff (1980) showed that nine species of common aquatic macrophytes took all their P directly from the sediment when grown *in situ*. Similarly, Barko et al. (1991) concluded that given the large pools of P in most lake sediments, it is unlikely that this element would normally limit plant growth. However, plant shoots can also take up dissolved nutrients from the water and, on a weight-specific basis, the contribution of shoot uptake to growth in Eurasian watermilfoil can be comparable to that of root uptake (Nichols and Keeney 1976a). Smith and Adams (1986) determined that shoot uptake of dissolved P by Eurasian watermilfoil could account for approximately 27% of the annual P uptake. Nitrogen can also be obtained by submerged macrophytes from the water column and from sediments (Nichols and Keeney 1976b, Barko et al. 1991). Nichols and Keeney (1976a) found that the N requirements of Eurasian watermilfoil could be met by root uptake of NH_4 alone. At high concentrations in the water, however, shoot uptake of N as NH_4 and nitrate (NO_3) could be more important than root uptake.

Most experimentation on macrophyte nutrient uptake has been done in artificial habitats such as greenhouses and small ponds [see Carignan and Kalff (1980) and Anderson and Kalff (1986) for exceptions]. In natural aquatic habitats, the contribution of shoot and root uptake of N and P to plant growth may be more variable depending on the relative availability of these elements in the sediments and in the water column. Nitrogen availability in the sediments can at times be limiting for milfoil growth (Carignan 1985), as shown by Anderson and Kalff (1986) who found that biomass of plants grown in NH_4-fertilized lake sediments during late spring and summer increased at rates that were 30–40% higher than plants grown in ambient sediments. During such periods of sediment nutrient limitation of plant growth, nutrient absorption from the water column may be particularly important (Carignan 1982, 1985). Similarly, shoot absorption may be more important during periods of elevated nutrient concentrations in the water column, as might be the case near streams and other sources of drainage after precipitation events.

In Conesus Lake, New York, Eurasian watermilfoil dominates the submersed littoral vegetation, forming a dense band at depths of 1.5–3.5 m along the lake perimeter. This band is punctuated by a series of expansive and luxuriant beds that occupy the broad shallow margins of small peninsulas and adjacent coves associated with major tributary deltas (Bosch et al. 2000, D'Aiuto et al. 2006, Makarewicz et al. 2007). Research in the Conesus Lake watershed has shown that tributaries flowing from catchment basins with primarily agricultural land use (60–80%) deliver high fluxes and concentrations of soils and dissolved nutrients into the lake (D'Aiuto et al. 2006, Makarewicz et al. 2007). For example, in samples taken over a 3-day period in June 2002 from one such tributary, concentrations of NO_3+NO_2 were 13–240 times higher than in lake water, and soluble reactive phosphorus (SRP) concentrations were 3–24 times higher. Filamentous algal growth has been shown to nearly double in direct response to stream SRP loading (Makarewicz et al. 2007), and it is near these tributaries where the largest and most dense milfoil beds in Conesus Lake are located (Bosch et al. 2000, 2001).

This study was undertaken to determine if implementation of agricultural best management practices (BMPs) in small agricultural sub-watersheds could decrease nutrient loading by tributaries thereby reducing the biomass and dominance of Eurasian watermilfoil downstream in the Conesus Lake littoral zone. The quadrat biomass [dry weight (DW/m^2)], surface area (m^2), total bed biomass (kg), and species composition of six macrophyte beds dominated by Eurasian milfoil were measured during the height of the growing season from 2001 to 2003 prior to implementation of structural (e.g., gully plugs) and cultural (e.g., crop rotations) BMPs. Starting in summer 2003, BMPs were implemented in sub-watersheds upstream from three macrophyte beds (Herendeen and Glazier 2009), while limited or no improvements were made by landowners in four watersheds associated with three other large littoral macrophyte beds. All six beds were monitored during a Post-BMP experimental period from 2004 to 2007, and these data were compared to trends documented for the 2001–2003 Pre-BMP baseline period. Our hypothesis was that major reductions in tributary nutrient loading achieved through BMPs would lead to significant decreases in the biomass and dominance of Eurasian watermilfoil in associated macrophyte beds.

Background

Conesus Lake, the westernmost of New York's Finger Lakes, is a eutrophic glacial lake with an average depth of 10 m and a maximum depth of 20 m. Summer Secchi depths range between 1.5 and 3.5 m (SOCL 2001). The relatively high turbidity limits macrophyte distribution to a maximum depth of 4–5 m (Bosch et al., 2001). Approximately 15% of the lake bottom is shallower than 3 m and 61% is below 5 m.

According to Forest et al. (1978), the macrophyte flora of Conesus Lake was first characterized by W. C. Muenschner in 1927 for the New York State Department of Environmental Conservation. Muenschner (1927) judged several species to be “predominant,” including *Myriophyllum*, which he identified as *M. exalbescens*. Quantitative studies of the macrophyte communities were subsequently conducted by Forest and Mills (1971) and Forest et al. (1978), but it was Bosch et al. (2000, 2001) who first reported conclusively that the dominant milfoil in Conesus Lake was the invasive *M. spicatum* (Eurasian watermilfoil).

The littoral macrophyte beds chosen for this study were located downstream from agricultural sub-watersheds, including Sutton Point Gully, Long Point Gully, Cottonwood Gully, Graywood Gully and Sand Point Gully, that were part of the BMPs impact study described by Makarewicz et al. (2009) (Fig. 1). The North McMillan Creek sub-watershed that served as a forested reference watershed for Makarewicz et al. (2009) was not included in this study because the littoral macrophyte community near this stream was sparse and dominated by water celery (*Vallisneria americana*) rather than Eurasian watermilfoil. The McPhersons Point study site that is

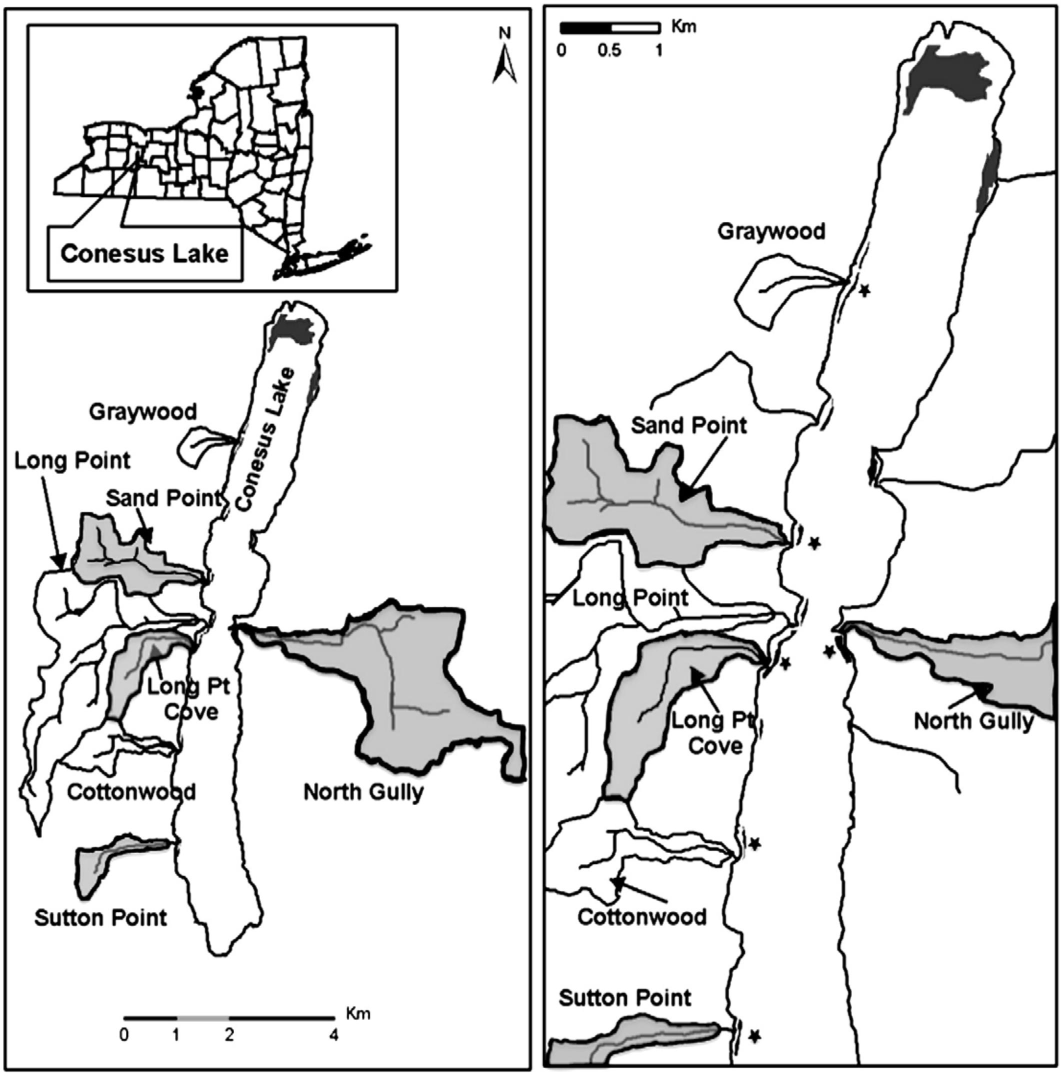

Fig. 1. The distribution of the largest macrophyte assemblages dominated by Eurasian watermilfoil in Conesus Lake. Project macrophyte beds and watersheds are identified by name.

associated with the North Gully tributary was added to replace the North McMillan site in this experiment (Fig. 1).

Quadrat biomass, standing crop, and species composition of macrophyte beds in Conesus Lake were studied extensively in the 1960s and 1970s (Forest and Mills 1971, Forest et al. 1978) and more recently by Bosch et al. (e.g., 2000, 2001) who worked in some of the sites used in the present study. More intensive sampling was implemented in 2001 to gather data in preparation for the BMP study that was initiated after the 2003 growing season. Sampling was continued through 2007. Thus the 2001–2003 growing seasons for the macrophyte beds are considered Pre-BMP baseline data that are compared to Post-BMP experimental years 2004–2007.

Agricultural BMPs implemented in the Sand Point Gully, Cottonwood Gully, and Graywood Gully sub-watersheds are described in detail by Herendeen and Glazier (2009) and Makarewicz et al. (2009) and summarized by Bosch et al. (2009). In sub-watersheds associated with the other three study macrophyte beds, either no major BMPs were undertaken (i.e., North Gully) or changes were made independently by landowners during the study period (Sutton Point Gully, Long Point Gully, Long Point Cove Gully) as follows. At the Sutton Point watershed an increasing proportion of the acreage was converted from corn production to alfalfa/grass after the 2003 macrophyte growing season from 37% in 2003 to 60.3% in 2007. At the 605-ha Long Point Gully sub-watershed, dairy cattle were removed in 2003 and a 37% reduction in crop acreage occurred by 2004 (Makarewicz et al. 2009). The Long Point Gully stream may strongly influence the northern portion of the Long Point Cove macrophyte bed that is located immediately to the south of this tributary (Fig. 1). A second gully/stream that drains into the southern portion of the Long Point Cove macrophyte bed may have a greater impact on the southern area of the bed, as proposed by Bosch et al. (2009) with respect to growth of metaphyton. Approximately 50% of the 114.2-ha Long Point Cove Gully sub-watershed consists of woodland cover or vacant fields; 36% of the watershed has been in row crops, primarily corn since 2003. The only significant BMPs implemented in this sub-watershed were the installation of gully plugs and drainage tile in about 11 ha near the middle reaches of the gully. This installation was completed in the fall 2005 (P. Kanouse, Personal Communication, Livingston County Soil and Water Conservation District).

Methods

Macrophyte collections

To characterize macrophyte communities associated with the six project watersheds, we conducted annual surveys of plant biomass, species composition, and bed distribution in early to mid August (2001–2007), which is considered the approximate peak of the growing season in the Finger Lakes region of New York (Zhu et al.

2006, Madsen et al. 2008). A stratified-random design for macrophyte sampling for each study site generally followed the recommendations of Madsen (1993).

For each site, three transects were established approximately perpendicular to the shoreline at different distances from the nearby stream. At the Cottonwood Gully, Sand Point Gully, Long Point Cove and North Gully macrophyte study sites, the stream was located near the southern or northern edge of the bed. Therefore one transect was positioned within a few meters of the stream and the other two were positioned incrementally about 75 m from the streamside transect along the length of the bed. For the Long Point Cove macrophyte bed, one transect was established near the southern margin of the bed adjacent to the Long Point Cove Gully tributary (Fig. 1). The two other transects were established at about 75 and 150 m to the north at a minimum distance of 300 m to the south of the Long Point Gully tributary. At the Graywood Gully and Sutton Point Gully sites, the stream was located near the middle of the bed. One transect was established directly off the stream; the two other transects were about 75 m to the north and south.

The dry weight per square meter of substrate (i.e., quadrat biomass) was determined along each transect using a quadrat sampling approach. Replicate quadrat samples (2–4, usually 3) were collected at depths of 1, 2, 3 and 4 m by S.C.U.B.A. divers or by snorkeling in the shallower depths. Typically, 36 quadrats were sampled for each bed, including 18 quadrats within the 1.5- to 3.5-m zone dominated by Eurasian watermilfoil. The quadrat was a square of PVC pipe with sides of 0.5 m (total surface area 0.25 m^2) that were held together by PVC elbows. One side of the quadrat was removed to facilitate its placement on the substrate around the plant stems. All vascular macrophytes rooted or attached within the quadrat were cut at the stem just above the sediment, placed inside a collection bag, and brought back to the laboratory for analysis. In the laboratory, zebra mussels and metaphyton attached to the plants were carefully removed. The macrophyte material was then drained well and blotted dry with paper towels before weighing. The wet weight of samples, sorted by individual species (Prescott 1980, Borman et al. 1998), was determined to the nearest 0.1 g with an electronic scale.

Results are reported as dry weight biomass. To convert quadrat biomass from wet weight to dry weight, we used conversion factors determined empirically for each of the six most common species in Conesus Lake (Table 1). The conversion factors were determined by drying pre-weighed bulk samples in an oven at 105 °C for approximately 48 h. The samples were then weighed a second time. A wet to dry conversion factor of 0.10 was used for all other species encountered (Forest et al. 1978).

In addition to the transect studies, the distribution of the Eurasian watermilfoil-dominated areas in each of the macrophyte beds was mapped using global positioning systems (GPS). To map the beds, a swimmer would visually track the perimeter of the milfoil growth zone while corresponding GPS coordinates were recorded from a closely trailing boat every few seconds using a Trimble Model TSC1 global positioning unit (Trimble Navigation Ltd.). Pathfinder and ArcView software (Esri Inc.) were used to determine the surface area of each bed and to superimpose the graphical data on a high-resolution topographic map of Conesus Lake and the surrounding watershed (Fig. 1). Eurasian watermilfoil was considered to be dominant in any region where it represented more than half of the individual plant stems. Typically the inner margin of the milfoil bed was in a region of sparse macrophyte growth so that any slight error in defining the inner margin would have little effect on estimates of bed biomass. Beyond the offshore margin of the Eurasian watermilfoil (>3.5 m), the macrophyte community was dominated by a very narrow and usually sparse strip of coontail (*Ceratophyllum demersum*) or curly-leaf pondweed (*Potamogeton crispus*). Consequently, the potential error in determining the offshore margin of the bed would also be small. The reproducibility of the surface area estimates was tested on two occasions in different years, and in both cases the corresponding surface area estimates as determined by two different sets of swimmers were within 2% of each other.

Table 1
Wet weight to dry weight conversion factors for abundant macrophyte species collected from beds dominated by Eurasian watermilfoil in Conesus Lake.

Species	Common name	Sample size	Dry/wet weight	Standard deviation
Myriophyllum spicatum	Eurasian milfoil	18	0.1795	0.0321
Vallisneria americana	Water celery	8	0.0366	0.0029
Ceratophyllum demersum	Coontail	7	0.1048	0.0181
Zosterella dubia	Water stargrass	8	0.1060	0.0312
Potamogeton crispus	Curly-leaf pondweed	8	0.0773	0.0021
Elodea canadensis	Common waterweed	8	0.0663	0.0117

For less common species, an average conversion factor of 0.10 was obtained from the literature (Forest et al. 1978).

Aquatic macrophyte biomass data were transformed to $\log_{10}$ values for statistical analysis (Madsen 1993). The transformed data were normally distributed in all but the Cottonwood Gully data set. However, half the data sets failed equality of variance tests. Therefore a Kruskal–Wallis One Way Analysis of Variance on Ranks was used to compare pooled Pre-BMP baseline biomass distribution (2001–2003) to each of the subsequent Post-BMP experimental years (2004–2007) for each site. In cases where the ANOVA on Ranks produced statistically significant differences, a Dunn's Method of multiple comparisons versus a control group (i.e., the baseline period) was used to identify the individual years that differed from the Pre-BMP distribution (*SigmaStat 3.5*, SYSTAT Software Inc.).

The total biomass of the milfoil-dominated zone for each bed was calculated by multiplying the area where milfoil was dominant times the average quadrat biomass collected in the 2- to 3-m zone. More spatially explicit estimates of biomass were not obtainable because fine scale contour maps were not available. By treating Pre-BMP years and Post-BMP years as comparable sets of replicates, we tested the hypothesis that total biomass in the milfoil-dominated zone Post-BMP were statistically lower than in Pre-BMP years using a One tail *t*-test assuming equal variances (Microsoft *Excel 12.0.1* Analysis Toolpack, Microsoft Corporation).

Plant biomass and nutrient loading

To determine whether decreases in plant biomass were consistent with decreased nutrient loading due to implementation of agricultural BMPs, biomass trends were compared to annual marginal mean loading trends for NO_3+NO_2, soluble reactive phosphorus (SRP), total phosphorus (TP) and total suspended solids (TSS) (Makarewicz et al. 2009). Additionally, here we report trends in total January–August loading of NO_3+NO_2 and SRP. The measurements are based on daily samples collected from November 2002 to September 2007 at the mouths of the Cottonwood Gully, Sand Point Gully, Graywood Gully, Sutton Point Gully, and Long Point Gully tributaries. Nutrient concentrations for the North Gully tributary were monitored weekly during nonevent discharge and during most major hydrometeorological events from 18 May 2004 to 4 Sep 2007. These results are reported by Bosch et al. (2009), who found no statistically significant differences between years. Data for the Long Point Cove Gully tributary were insufficient to construct an accurate analysis of loading trends. A detailed account of collection methods and determination of loading trends is provided by Makarewicz et al. (2009).

Results

Community structure and dynamics

There was a consistent depth-related pattern of macrophyte biomass distribution and species composition that created a loosely organized depth zonation in Conesus Lake. Eurasian watermilfoil was the dominant species at depths of approximately 1.5–3.5 m, forming a ring that almost completely encircled open waters of the lake. Near streams, alluvial deltas provided broad areas of shallow habitat. At depths of 2–3 m along these streamside areas, the macrophyte community consisted of dense and nearly monospecific assemblages of milfoil (Fig. 1) that attained dry weights as high as 1650 g/m^2. Depths of 4 m were usually on the upper slope just beyond the shallow nearshore shelf that constituted the Conesus Lake littoral zone. Many of the quadrats taken at 4 m were devoid of plants (Fig. 2). This trend reflected the patchy distribution and sparse biomass of curly-leaf pondweed, Eurasian watermilfoil, and especially coontail, the only species that were routinely found in deeper waters. Below 4 m there was little or no macrophyte biomass, presumably due to light limitation imposed by the highly turbid water column (SOCL 2001).

Water depths of 0.5–1.5 m (i.e., the shallow littoral, 1–20 m offshore depending on slope) were transitional nearshore habitats subjected to extremes in physical conditions. Consequently, biomass and species composition of the macrophyte community at these depths were highly variable temporally and spatially. Near some streams, the shallow littoral was dominated by water celery (*Vallisneria americana*), water stargrass (*Zosterella dubia*), common waterweed (*Elodea canadensis*), and in some exceptional cases by coontail. These species, and primarily water celery, seemed to replace milfoil in areas that were rich in very fine sediments (e.g., transects closest to North Gully and Sutton Point Gully). Populations of Eurasian watermilfoil were dominant in areas of more coarse sediments, including near outflows of Graywood, Cottonwood, and Long Point Cove Gully. These distribution patterns could vary between years, making the shallow littoral zone the most diverse and dynamic macrophyte habitat in Conesus Lake.

Trends in quadrat biomass, bed surface area, and total biomass

Macrophyte bed development during the growing season was variable both between sites and in different years. Our observations indicate that overwintering condition was a key determinant in macrophyte bed growth dynamics during the subsequent growing season. Beds typically persisted through winter as both roots and shoots. Shoot biomass increased rapidly in late spring and early summer and reached peak biomass in mid to late August. In this context, the winters of 2001–2002 in the North and Sutton Point, and 2002–2003 in the Cottonwood Gully, Sand Point and Graywood Gully

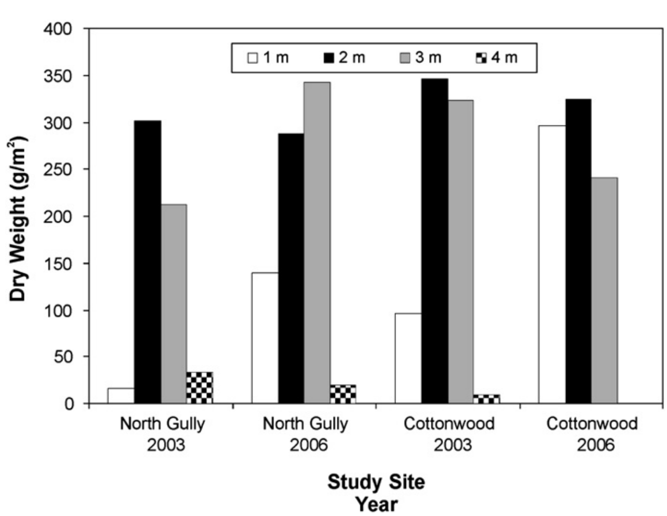

Fig. 2. The distribution of biomass (g/m^2) with depth at two representative study sites in Conesus Lake.

Table 2
Surface area (m^2) dominated by Eurasian watermilfoil within study macrophyte beds.

Study site	2000	2001	2002	2003	2004	2005	2006	2007
North Gully	23,192	25,783	12,004	19,760	30,099	21,798	22,560	27,850
Sutton Point Gully	Nd	8592	3688	11,819	11,909	11,995	7438	10,973
Long Point Cove	Nd	28,866	26,436	25,200	23,524	24,706	21,926	23,094
Cottonwood Gully	Nd	9387	7360	3750	9205	6880	5605	8100
Graywood Gully	Nd	16,496	17,336	8930	15,414	10,639	10,673	14,687
Sand Point Gully	9535	9781	7354	5310	8474	8349	9775	9684

Underlined values are the maximum surface areas in years where shoot die-off occurred in the winter. Thus macrophytes were slow to grow and developed incompletely during the following growing season. Nd = no data.

sites were exceptional. At these times, most of above sediment (shoot) biomass was lost despite the fact that the beds appeared healthy during the previous growing season. The beds overwintered as roots and the ground biomass developed slowly and incompletely in the following growing season (Table 2). This phenomenon was not observed in any of the other 6 sample years or in previous studies by Bosch et al. (2000, 2001). Thus for one growing season in most study beds, winter die-off was more important to milfoil growth and biomass than other variables (i.e., nutrient availability). Where winter shoot die-off occurred, we did not include biomass data for those growing seasons in our overall analyses of trends. Statistical comparisons of Pre- and Post-BMP trends were possible since at least 2 years of baseline (Pre-BMP) data remained per study site.

For the North Gully study site, overall median quadrat biomass for 2001 was 430 g/m^2 and individual quadrats consistently reached dry weights of more than 600 g/m^2 (equivalent milfoil wet weight of 3.3 kg). Overall, median North Gully quadrat biomass values for each year of the Post-BMP experimental period were lower than that of the baseline period, but the differences were not statistically significant (Fig. 3a). Post-BMP total bed biomass was 34% lower than Pre-BMP values, but was not statistically significant (Fig. 4; One tail t-test, $p = 0.07$). Surface area of the milfoil-dominated bed at North Gully remained relatively constant throughout the study (Table 2; One tail t-test, $p = 0.20$).

At the Sutton Point study site, there were no significant differences in quadrat biomass between Pre- and Post-BMP years (Fig. 3b; ANOVA on Ranks $p = 0.07$). In 2005 median quadrat biomass was 144.5 g/m^2, nearly 100 g/m^2 lower than in any other year. The macrophyte bed peaked in July 2005 after overwintering as roots and shoots. With the exception of 2006, when there was a slight but statistically insignificant decline, surface area of the milfoil-dominated zone at Sutton Point remained fairly constant at approximately 11,000 m^2 (Table 2; One tail t-test $p = 0.43$). Average biomass of the whole bed in 2004–2007 was only 2.7% lower than in the Pre-BMP baseline period (Fig. 4).

At the Long Point Cove macrophyte bed, median quadrat biomass was not statistically different from the Pre-BMP baseline period during any of the Post-BMP years (Fig. 3c; ANOVA $p = 0.34$). Biomass in 2004 and 2007, however, was 33% and 25% lower, respectively, than the Pre-BMP median of 206 g/m^2. These relatively low values were due largely to declines in milfoil populations in the northern transect located nearest to Long Point Gully rather than to changes throughout the macrophyte bed. In 2007 for example, median quadrat biomass at the 2-m depth of the northern transect nearest Long Point Gully was 105 g DW/ m^2, whereas comparable values in central and southern transects were 250 and 208 g DW/ m^2. Similarly, for the Post-BMP period, the surface area of the Long Point Cove macrophyte bed was significantly smaller than that of the Pre-BMP period (Table 2; One tail t-test, $p = 0.02$) and losses in surface area were primarily in the northern portion closest to the Long Point Gully tributary. Overall,

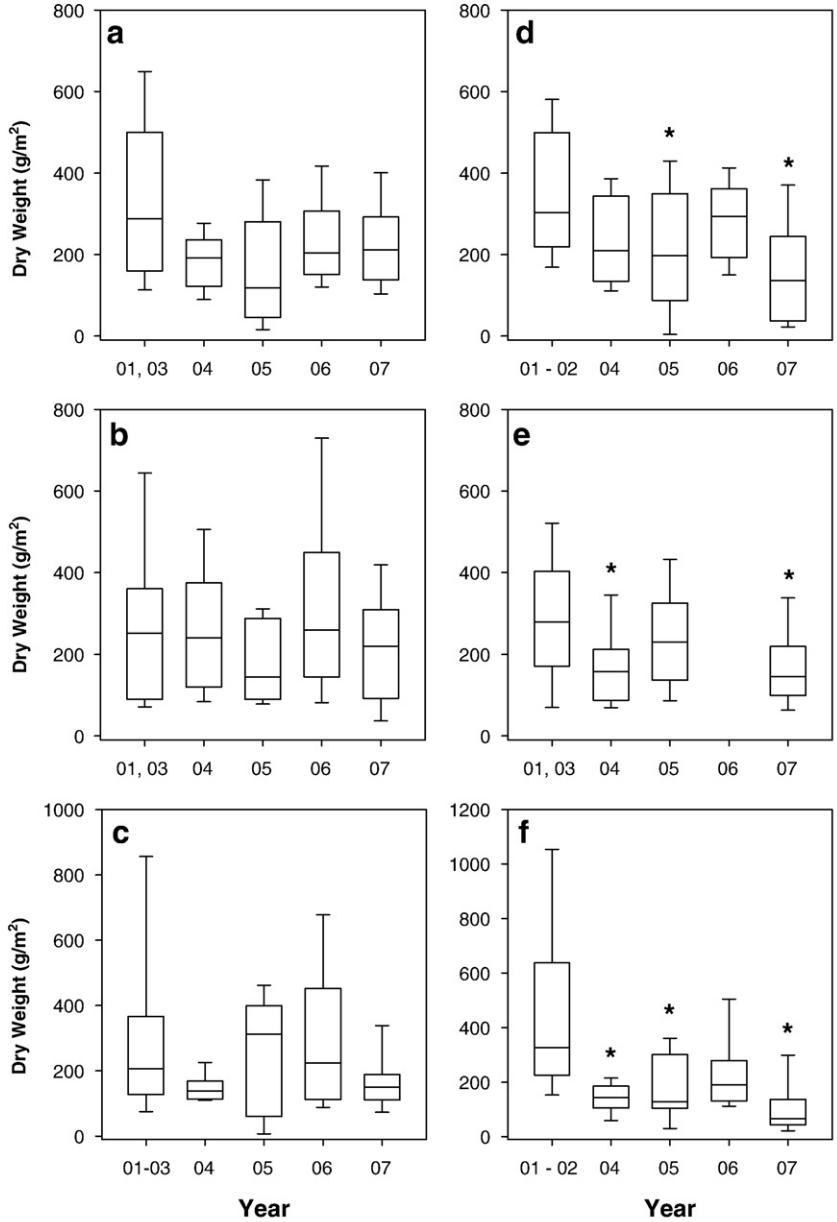

Fig. 3. Box plots showing the Pre- and Post-BMP macrophyte biomass (g/m^2) in the 2- and 3-m depth band of the littoral zone routinely dominated by Eurasian watermilfoil: (a) North Gully (b) Sutton Point Gully (c) Long Point Gully (d) Cottonwood Gully (e) Graywood Gully (f) Sand Point Gully. Each box represents the 25 and 75% confidence intervals and the capped lines are the 10–90% intervals. The single line across the bar is the median. Asterisks indicate years in which biomass was statistically different from the Pre-BMP years (2001–2003).

average bed biomass for the Post-BMP period was 14% lower than that of the Pre-BMP period, but these differences were not statistically significant (Fig. 4; One tail t-test $p = 0.26$).

Starting in the summer of 2003, a variety of BMPs were implemented in the Cottonwood, Sand Point, and Graywood Gully watersheds (Herendeen and Glazier 2009, Makarewicz et al. 2009). At Cottonwood Gully, quadrat biomass decreased from a Pre-BMP median of 303.1 g/m^2 to significantly lower values of 197.4 g/m^2 in 2005 (34.9% decrease) and 135.6 g/m^2 in 2007 (55% decrease, Fig. 3d). Median quadrat biomass in 2004 was 30.8% lower but not statistically different from Pre-BMP biomass. In 2006 median biomass was 296.7 g/m^2, nearly equal to the Pre-BMP median. Bed surface area was smallest in 2005 and 2006 (Table 2), but otherwise Pre-BMP and Post-BMP surface areas were not statistically different (One tail t-test, $p = 0.26$). Owing to the lower quadrat biomass, however, total bed biomass was statistically lower after BMPs were implemented in the Cottonwood Gully watershed (Fig. 4; One tail t-test $p = 0.03$).

At the Graywood Gully site there were statistically significant reductions in biomass during the Post-BMP period as well. Median quadrat biomass in 2004 and 2007 was approximately 50% and 54% lower than the Pre-BMP median of 318.3 g/m^2 (Fig. 3e). In 2005 median biomass was 230.6 g/m^2, 27% lower but not statistically different from the Pre-BMP median. The area of milfoil-dominated zone was statistically smaller by an average of 24% after BMPs were implemented (Table 2; One tail t-test, $p = 0.05$, df = 3). Areas vacated by milfoil were either colonized by coontail or remained unoccupied. In 2006 quadrat biomass data was extremely low because milfoil had collapsed in late July followed by a major die-off of plants in August. Similar widespread collapses have been observed by our research group in other locations around Conesus Lake, both in 2006 and in other years. Our observations indicate that the cause of the collapses may be the weight of zebra mussels attached to Eurasian watermilfoil stems combined with that of the metaphyton growing within the canopy. This increased weight appears to overcome the inherent

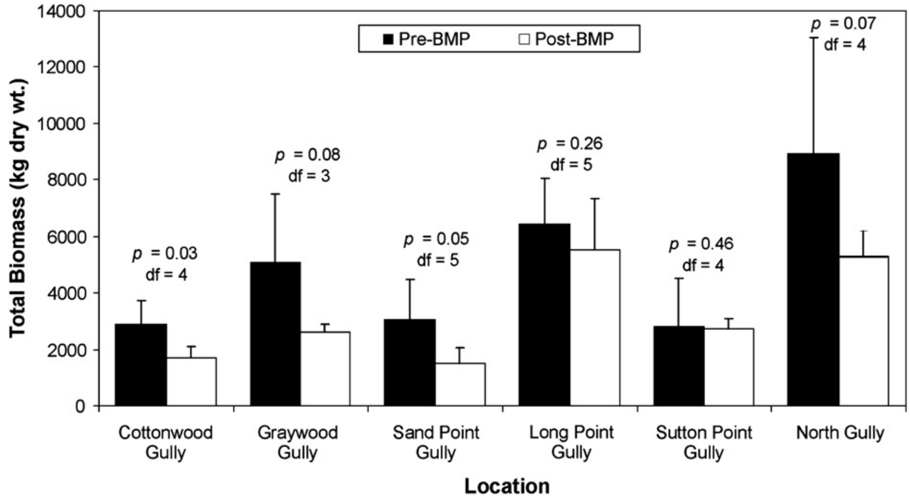

Fig. 4. Total macrophyte biomass in beds dominated by Eurasian watermilfoil.

buoyancy of the plants with the final collapse often triggered by wave action. Excluding 2006 data, average bed biomass decreased by about 48% under the influence of both a lower median quadrat biomass and smaller bed surface areas, but this difference was not statistically significant (Fig. 4, One tail t-test, $p = 0.08$, df = 3).

At Sand Point Gully, median macrophyte biomass in the 2- to 3-m zone was statistically lower for 2004–05 and in 2007 (Fig. 3f) with median decreases of 56%, 61%, and 80% from the Pre-BMP baseline median of 356.3 g/m^2, respectively. In 2006 the median quadrat biomass was 41% lower than Pre-BMP values, but that difference was not statistically significant. The Sand Point macrophyte bed remained fairly consistent in surface area throughout the study period (Table 2). Average bed biomass for the Post-BMP period, however, was 50% lower than for the Pre-BMP period (Fig. 4; One tail t-test, $p = 0.05$). These changes were largely driven by decreases in quadrat biomass rather than changes in the area covered by Eurasian watermilfoil.

Species composition

Milfoil was the overwhelmingly dominant macrophyte at depths of 1.5–3.5 m at all study sites, a pattern that reflects conditions in all of Conesus Lake as far back as the 1960s (Forest et al. 1978) (Table 3). But there are patches in some of the transects where milfoil is not dominant, for example near the stream at North Gully in 2003 and 2005. By 2007, however, milfoil was dominant in those transect areas and in the two other permanent transects established at North Gully.

Table 3
Percent (mean ± SD) of the macrophyte dry weight comprised by Eurasian watermilfoil at depths of 2 and 3 m in the six study sites.

Study site	2003	2004	2005	2006	2007
North Gully					
South, Central	99.6 ± 0.1	99.4 ± 0.8	95.9 ± 5.8	99.5 ± 0.7	100.0 ± 0.8
Stream	39.2 ± 22.6	ns	46.0 ± 44.0	ns	99.1 ± 1.4
Sutton Point Gully					
South, North	66.6 ± 5.8	65.4 ± 27.5	85.1 ± 15.8	85.1 ± 21.0	81.6 ± 14.0
Stream	34.8 ± 27.7	36.5 ± 31.3	0.5 ± 1.3	8.8 ± 12.0	98.5 ± 2.5
Long Point Cove	93.2 ± 11.5	99.5 ± 1.0	94.5 ± 7.7	99.9 ± 0.1	98.5 ± 2.3
Sand Point Gully	98.6 ± 1.4	91.4 ± 11.4	95.8 ± 5.9	99.5 ± 0.7	95.7 ± 7.4
Cottonwood Gully	95.8 ± 7.4	100.0 ± 0	95.4 ± 9.2	100.0 ± 0	95.7 ± 4.8
Graywood Gully					
South, Stream	55.7 ± 17.2	67.3 ± 16.5	76.6 ± 31.1	97.8 ± 0.7	92.6 ± 0.8
North	12.7 ± 22.0	23.3 ± 28.6	93.0 ± 6.0	ns	93.0 ± 7.1

In some transects, particularly those near streams, the dominant species were water celery and coontail rather than milfoil. Data for those transects are shown in separate rows. ns = no sample.

A similar pattern was evident in Graywood Gully downstream from the most intensively managed watershed. Some other beds, such as Cottonwood Gully, show strong and persisting milfoil dominance in the 2- to 3-m depth range throughout the study period. Eurasian watermilfoil continued to be dominant or became increasingly dominant in all of our study sites, regardless of intensity and type of BMPs implemented. This is true even in sites like Sand Point Gully,

Table 4
Total January–August loading of nitrate (NO_3+NO_2), total phosphorus and soluble reactive phosphorus from five tributaries draining agricultural sub-watersheds around Conesus Lake.

Study site/year	Nitrate	Total phosphorus	SRP
Cottonwood Gully			
2003	*3296.8 kg*	*88.5 kg*	*24.4 kg*
2004	17.3	1.5	−38.4
2005	33.8	101.7	28.6
2006	−61.5	−56.8	−54.6
2007	−38.1	−36.7	−25.8
Graywood Gully			
2003	*3194.6 kg*	*72.3 kg*	*38.8 kg*
2004	176.5	272.1	99.5
2005	74.9	110.6	119.7
2006	−65.2	−54.9	−53.0
2007	−55.9	−27.7	−12.4
Sand Point Gully			
2003	*3757.6 kg*	*149.8 kg*	*32.6 kg*
2004	−77.9	−40.5	−57.8
2005	−38.7	−37.1	−3.7
2006	−71.0	−86.9	−81.9
2007	−90.7	−85.3	−87.6
Long Point Gully			
2003	*9256.3 kg*	*245.6 kg*	*105.2 kg*
2004	86.6	−16.8	−19.7
2005	114.7	104.7	84.1
2006	17.1	−26.4	−41.9
2007	−52.3	−60.8	−63.5
Sutton Point Gully			
2003	*702.3 kg*	*8.9 kg*	*4.1 kg*
2004	184.6	348.9	394.5
2005	151.2	255.7	238.2
2006	−15.2	4.5	44.9
2007	32.5	76.6	101.5

The numbers corresponding to the year 2003 represent the loading in kg. The numbers for subsequent years are the percent difference from the 2003 value. Negative values indicate percent decreases; positive values are percent increases.

where significant decreases in milfoil biomass were recorded during all 4 years of the Post-BMP period.

Nutrient loading trends

The total sub-watershed loading of NO_3+NO_2, TP, and SRP for the January–August period is reported in kg for 2003 and as a percent of the 2003 values for 2004–2007 (Table 4). Loading at Cottonwood Gully and Graywood Gully was markedly lower than baseline in 2006 and 2007 for all analytes, while loading by Sand Point Gully was lower than baseline beginning in 2004. In contrast, for Sutton Point, values were consistently above baseline with the exception of NO_3 loading in 2006, which was 15.2% lower than baseline. Values for Long Point Gully were higher or only slightly lower than baseline every year except in 2007, when decreases of 52.3–63.5% relative to baseline were detected.

Discussion

The biomass growth of Eurasian watermilfoil in temperate lakes is known to be highly variable spatially, both in different lake habitats and within an assemblage. It is also variable between years and on a seasonal scale, with biomass peaking in early to late summer (Little et al. 1997). As in other lakes, dynamics of Eurasian watermilfoil in Conesus Lake were highly variable. An additional source of variability was also seen as a result of unusually large-scale winter losses of shoot biomass that occurred during the winters of 2002 and 2003. These die-offs were followed by a slow recovery and incomplete growth of milfoil beds from overwintering roots during the following growing season. Despite the natural variability in Eurasian watermilfoil populations, statistically significant trends of lower biomass were observed. Specifically, at the 2- to 3-m depths dominated by Eurasian watermilfoil downstream from agricultural watersheds where major BMPs were implemented (i.e., Sand Point Gully, Graywood Gully, Cottonwood Gully), median biomass densities were significantly lower than Pre-BMP values in 7 of 11 sample years. In contrast, among three macrophyte beds associated with four watersheds where minor or no BMPs were introduced (Sutton Point Gully, North Gully, Long Point Cove Gully, Long Point Gully), none of the 12 Post-BMP sample years showed statistically lower quadrat biomass. At Sand Point Gully and Graywood Gully, median quadrat biomass declined by more than 50% of Pre-BMP values by the 2004 growing season, and in both cases there was high statistical confidence in these differences. At Cottonwood Gully the 2004 median biomass was approximately 31% lower than the Pre-BMP median (2001–2002), but the reduced values were not statistically lower until 2005 and then again in 2007. Overall, the declines in quadrat biomass associated with some declines in surface area translate into an average total decrease of 30–42% in the milfoil bed biomass and equivalent dry weight losses of 1.1–1.8 t (ca. 6.1–10.0 t wet weight) in each bed.

The reductions in quadrat biomass, bed area, and bed biomass are consistent with changes in tributary nutrient loading brought about by the implementation of BMPs upstream in the corresponding watersheds. Makarewicz et al. (2009) reported statistically significant annual decreases in nutrient concentrations and fluxes for Sand Point Gully, Graywood Gully, and Cottonwood Gully from 2004 to 2007 when compared to Pre-BMP loading (September 2002 through August 2003) for each site. For example, in the Graywood Gully tributary after winter surface manure application was effectively halted and other management practices were in place, NO_3 fluxes decreased by 25% within 1 year and remained on average about 66% lower than Pre-BMP in subsequent years. Fluxes of SRP decreased by about 17–39%, but these differences were not statistically significant. Significant decreases in SRP, however, were observed during the winter and spring (Lewis and Makarewicz 2009) and in the January–August period in this study (Table 4). The lack of significance observed by Makarewicz et al. (2009) is likely due to P inputs from the "extended watershed" as discussed by Noll and Magee (2009). Between September 2004 and August 2007, TP and TSS were statistically lower by an average of 31% and 51%, respectively.

Tributary nutrient loading patterns analyzed here (January–August) and in an accompanying metaphyton study (July–August) (Bosch et al. 2009) are generally consistent with annual nutrient loading trends reported by Makarewicz et al. (2009). January–August fluxes of NO_3, TP, and SRP decreased markedly beginning in 2004 for Sand Point Gully and in 2006 for Cottonwood Gully and Graywood Gully (Table 4). In contrast, fluxes in 2004–2007 (Post-BMP years) were comparable or higher than in 2003 (Pre-BMP year) for Sutton Point Gully. Makarewicz et al. (2009) report statistically significant annual decreases in NO_3+NO_2 loading for Sutton Point Gully and in NO_3+NO_2, TP, and SRP for Long Point Gully starting in 2004. These trends were not reflected in the January–August loading data or in the macrophyte data, with the following exceptions. In 2006 and 2007 there were annual and January–August reductions in loading from Long Point Gully that may have affected milfoil growth in the northern portion of the Long Point Cove bed. By 2007, areal coverage of milfoil had declined by 14% compared to 2001–2003 values, and median quadrat biomass along the northern transect closest to Long Point Gully was 66% lower than in the two southerly transects combined. We know that a shift in agricultural operations from dairy to row crops took place in the Long Point Gully watershed after 2003 (See Makarewicz et al. 2009) and that new management practices, including manure injection, were implemented during this transition. Reductions in nutrient loading, first detected in 2004, were evidently brought about by concomitant changes in farming practices. Ultimately, such changes may be responsible for statistically significant decreases in both macrophyte bed surface area and biomass seen in the northern portion of Long Point Cove which are masked in the statistical analysis of whole-bed trends.

Results of this study are important to our understanding of how changes in nutrient delivery from the watershed can affect littoral macrophyte beds dominated by the invasive Eurasian watermilfoil. In their studies of N utilization by milfoil, Nichols and Keeney (1976a) suggested that highly concentrated storm effluents entering Lake Wingra, WI, could provide an important source of N that would be available to the plants by absorption through the shoots. This process might be particularly important as a supplement to root uptake during the late spring and summer growth periods when N availability in the sediment can be limiting (Anderson and Kalff 1986). It is unclear whether the same applies to P. Barko et al. (1991), Smith and Barko (1990), and many others have provided evidence that phosphorus supplies in lake sediments may be so high as to never be limiting for the growth of rooted macrophytes. On the other hand, there is good evidence that sediment P concentrations can decline significantly during the growing season in dense macrophyte assemblages (Carignan 1985). Under these circumstances, shoot uptake of SRP could provide an important alternate or supplementary reservoir. Consequently, a response to P loading as suggested by D'Aiuto et al. (2006) might not be unexpected. In fact, Shuskey et al. (2009) demonstrate that growth of Conesus Lake Eurasian milfoil was stimulated by shoot uptake of nutrients.

Conesus Lake is a turbid eutrophic lake. Nonetheless, concentrations of N and P in the water are often low. Surface NO_3 concentrations during the summer season are less than 60 μg/L, while SRP concentrations are less than 10 μg/L (SOCL 2001). During major runoff events in the May–August macrophyte growing season, streams draining agricultural watersheds around Conesus Lake deliver NO_3 and SRP at concentrations that are typically two to three and one to two orders of magnitude, respectively, higher than in lake concentrations (e.g., D'Aiuto et al. 2006, Makarewicz et al. 2007). Dissolved P entering through streams has a strong influence on littoral macroalgae, in some cases doubling their biomass in a matter of hours in comparison to incubations in lake water (Makarewicz et al. 2007). Particulate loads from agricultural watersheds are also high. For example, Makarewicz et al. (2009) reported Pre-BMP averages for TSS concentrations of 8.8 mg/L, total annual loads of more than 2.4 t for Sand Point Gully, and 9.5 mg/L and 1.67 t for Cottonwood Gully. Dilution of stream runoff in the littoral would yield lower concentrations of suspended solids and dissolved nutrients, but macrophyte beds reduce current velocities and slow the dissipation of inflowing waters (Madsen et al. 2004), thus increasing nutrient retention and sediment deposition near stream mouths (Losee and Wetzel 1993). This has been shown specifically for Sand Point Gully and Cottonwood Gully in Conesus Lake using hydrodynamic models of plume dispersion (Li et al. 2007).

The extent to which changes in Eurasian watermilfoil populations reported here result from decreases in stream loading of sediments and/or dissolved nutrients cannot be determined from our data. The fact that changes in biomass were evident within a year after BMPs were implemented is an indication that water column nutrients may have been important, since shoot uptake of nutrients from the water can occur rapidly (e.g., Smith and Adams 1986). Conversely, because Eurasian watermilfoil roots penetrate as deep as 60 cm into the sediment and the highest root biomass may be 10–20 cm below the sediment water interface (Carignan 1985), it is unlikely that any freshly settled nutrients within the macrophyte beds would be transported to those depths within such a limited time frame. However, the possibility that sediment nutrients in the milfoil rooting depths might have decreased 2 or 3 years after the implementation of BMPs cannot be precluded.

In conclusion, the data presented in this study indicate that agricultural BMPs implemented at the level of the sub-watershed brought about reductions in the population biomass (g/m^2) and bed biomass (kg) but not in the community dominance of Eurasian watermilfoil in Conesus Lake. The response times of the macrophytes to upland management appeared within a few years. While the spatial scale of these changes is only on the order of one to a few hectares in the littoral zone near stream mouths, this is a region where the public comes in daily contact with the lake and where improvements could significantly enhance the recreational value and aesthetic quality of the habitat. These results should provide an impetus for the use of a traditional approach, agricultural nutrient management, either singly or in combination with other management approaches, to control the growth of Eurasian watermilfoil on a local scale in the lake littoral.

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In situ experimental studies of Eurasian Watermilfoil (*Myriophyllum spicatum*) downstream from agricultural watersheds: Nutrient loading, foliar uptake, and growth

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ABSTRACT

Recent studies in Conesus Lake, New York, documented significant decreases in the biomass of Eurasian watermilfoil (*Myriophyllum spicatum*) near the mouths of streams draining sub-watersheds where reductions in nutrient loading occurred as a result of the implementation of agricultural Best Management Practices (BMPs). In situ experiments were conducted to further investigate the relationship between stream loading, foliar uptake, and growth of Eurasian watermilfoil. In two of three experiments, plants cropped to a height of approximately 50 cm had the lowest growth (g/m^2) downstream from a sub-watershed where major BMPs had been implemented (80% and 0%). In sub-watersheds where minimal or no BMPs were introduced, plants showed significantly higher growth as biomass increased (216% and 22%). In a second set of experiments, shoots of Eurasian watermilfoil plants were incubated for 24 h in ambient lake water and in lake water with enriched concentrations of nitrate and soluble reactive phosphorus comparable to rain event stream effluent concentrations and then allowed to grow in situ for a 2-week experimental period. For all experiments combined, the shoot biomass increased significantly in the enhanced nutrient treatments when compared to the ambient treatment at the Sand Point macrophyte bed (reduced loading) but not at the Eagle Point macrophyte bed (high loading). Overall, the results indicate that foliar uptake of nutrients in stream effluent can contribute to the growth of Eurasian watermilfoil and reinforce the hypothesis that reductions in stream loading through agricultural BMPs can help reduce macrophyte growth in the lake littoral.

Introduction

The invasive rooted macrophyte *Myriophyllum spicatum*, commonly known as Eurasian watermilfoil, is a dominant species in the nearshore regions of many North American lakes from Florida to Quebec in the east and from California to British Columbia in the west (Smith and Barko 1990). Eurasian watermilfoil grows vigorously in fertile shallow waters of the lake littoral, forming dense and expansive beds that suppress the growth and diversity of native macrophytes (Madsen et al., 1991). Excessive growth and canopy formation by Eurasian watermilfoil, a characteristic of shallow areas of many lakes (e.g., Madsen et al., 2008), reduces the aesthetic appeal of inland waters and is a detriment to recreational activities.

The littoral zone occupied by Eurasian watermilfoil and other submersed macrophytes serves as the interface between the land of the drainage basin and the open waters of the lake. As such, macrophytes have a key role in the nutrient and energy dynamics of lake ecosystems. Macrophytes contribute significantly to primary production in shallow lakes (Smith and Barko 1990, Wetzel 2001) and provide vast amounts of nutrients and organic compounds to the lake ecosystem, particularly at the time of die-off (Carpenter 1980). Macrophyte beds also provide a substrate for epiphytic diatoms and for filamentous green algae such as *Zygnema* and *Cladophora* (Pillsbury et al., 2002, D'Aiuto et al., 2006, Bosch et al., 2009a) and serve as a habitat for many animal species, including invertebrates and fish (e.g., Sloey et al., 1997, Pratt and Smokorowski 2003). Watermilfoil-dominated macrophyte beds also provide a favorable habitat for large populations of the invasive zebra mussel *Dreissena polymorpha* (Zhu et al., 2006). The benthic adult zebra mussels filter algae out of the water column and by their sheer numbers and filtering capacity increase light penetration, promoting higher macrophyte growth and a greater maximum depth of habitat (Skubinna et al., 1995, Zhu et al., 2006). Zebra mussel feeding also redirects energy and nutrients from the water column to nearshore benthic habitats (Holland et al., 1995, Hecky et al., 2004). Thus, under the ecological conditions that prevail in many North American lakes, the littoral zone macrophyte beds may have a more prominent role as a sink of nutrients and energy in lake ecosystems by sequestering materials from the open waters and by intercepting runoff from nonpoint sources (Hecky et al., 2004).

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Conesus Lake, the westernmost of the Finger Lakes of New York, is a relatively shallow, eutrophic lake (average depth 11.5 m, maximum 20 m) with a biologically rich littoral community dominated for decades by Eurasian watermilfoil (Forest et al., 1978, Bosch et al., 2009b). Some of the most dense and largest watermilfoil-dominated macrophyte beds in Conesus Lake occur near streams that drain agricultural sub-watersheds. A link between stream loading to the nearshore and macrophyte standing crop has been proposed, whereby land use type and stream runoff influence Eurasian watermilfoil growth (Makarewicz et al., 2007, D'Aiuto et al., 2006). This link between land use of a watershed and impacts on downstream aquatic systems, including water chemistry, bacteria, metaphyton, and macrophytes has been the focus of The Conesus Lake Watershed Project (Makarewicz 2009) and this issue. For example, Makarewicz et al. (2009) have demonstrated significant decreases in nutrient loading from sub-watersheds in which agricultural best management practices (BMPs) were implemented. Simon and Makarewicz (2009a and b) and Bosch et al. (2009a), working in the same sub-watersheds, observed major decreases in fecal indicators and biomass of filamentous algae associated with the implementation of BMPs and the resultant nutrient reduction to downstream systems.

Working with Eurasian milfoil in the nearshore of Conesus Lake, Bosch et al. (2009b) indicated that reductions in stream water nitrogen (N) and phosphorus (P) runoff achieved over 5 years through agricultural BMPs were related to reductions of up to 50% in Eurasian watermilfoil density and bed standing crops in the neighboring shoreline area. Since sediment nutrient availability is generally considered to promote macrophyte growth (e.g., Anderson and Kalff 1986), the significant decrease in Eurasian milfoil growth being associated with a decrease in stream nutrients, rather than sediment concentrations, was surprising and suggested that foliar uptake was playing a role. Also during the summer growth phase, the streams that drain the various Conesus Lake sub-watersheds had annual minimum daily nonevent flows punctuated by intense rain events (thunderstorms) that delivered significant amounts of nutrients and soil (Makarewicz et al., 2001). There is a possibility that rain events may be stimulating the growth of Eurasian milfoil, especially if foliar uptake of nutrients is occurring. Finally, Bosch et al. (2009b) demonstrated a reduction in "community" biomass of Eurasian milfoil over a 5-year period; that is, reductions in plant growth were suggested but not demonstrated in a single growing season.

In this follow-up study to Bosch et al. (2009b), we used short-term manipulative field experiments to examine two different aspects of the relationship between stream loading and Eurasian watermilfoil growth in the nearshore of Conesus Lake, New York. We tested the hypothesis that watermilfoil plants would produce less biomass (less growth) downstream from a managed watershed (Sand Point Gully, less nutrient loading) than downstream from watersheds that received limited or no BMPs (Eagle Point Gully and Long Point Gully, greater nutrient loading) within a single growing season. A

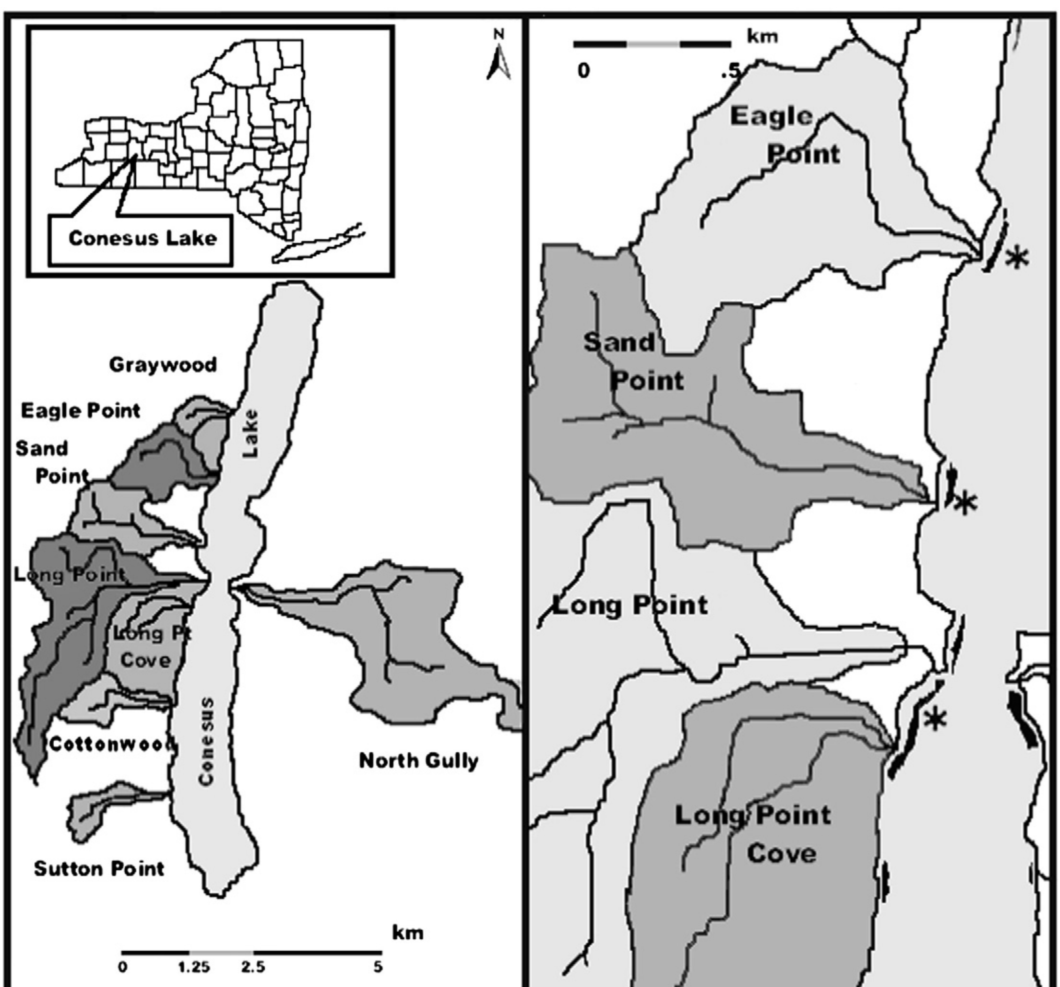

Fig. 1. Location of Eagle Point, Long Point, and Sand Point in the Conesus Lake watershed showing project sub-watersheds and study sites where experiments were conducted (*). Dark areas indicate macrophyte beds dominated by watermilfoil. Inset shows the location of Conesus Lake in New York State, USA.

second set of field experiments at the Sand Point and Eagle Point sites tested whether pulses of dissolved nutrients in the water column, simulating a 24-hour stream runoff event, would enhance the growth of individual watermilfoil plants. Results from these experiments should help us to better understand the influence of land use and stream runoff events on Eurasian watermilfoil growth and clarify the potential role of watershed practices in managing the growth of this invasive species.

Methods

Field experiments were conducted at water depths of 1.5–3 m in three Eurasian watermilfoil-dominated macrophyte beds (>70% of the macrophyte biomass) (Bosch et al., 2009b) during the summer of 2008 in Conesus Lake, New York. The three beds were located alongside the mouths of streams (Sand Point Gully, Eagle Point Gully, and Long Point Cove; Fig. 1) draining agricultural sub-watersheds that were part of a study on the effectiveness of agricultural BMPs in reducing nutrient and soil runoff to downstream aquatic habitats [see Makarewicz (2009) for a description of the project]. In 2003 and early 2004, BMPs were established in the Sand Point Gully sub-watershed (Fig. 1). Major BMPs implemented included structural (e.g., rotational grazing pens) and cultural (e.g., crop rotation) management practices. Manuring practices were changed in 2004 from surface spreading to injection in approximately 25% of the sub-watershed area (Herendeen and Glazier 2009). In the Eagle Point Gully sub-watershed (non-managed site), landowners did not implement major structural or cultural BMPs from 2003–2008 (Peter Kanouse, Personal Communication, Livingston County Soil & Water Conservation District).

The third location was Long Point Cove. Two streams draining two separate sub-watersheds influence the Long Point Cove macrophyte bed (Fig. 1; Bosch et al., 2009b). To the south, the small Long Point Cove Gully stream drains a 114.2-ha sub-watershed in which the principal land use is 50% woodland cover or vacant fields and 36% row crops, primarily corn. BMPs implemented by farmers in the fall of 2005 in ~11 ha of this watershed were gully plugs and drainage tiles (Bosch et al., 2009b). To the north of the macrophyte bed, the Long Point Gully stream drains a large heavily agricultural watershed. No major structural BMPs were established within this 605-ha watershed; however, dairy cattle were removed in 2003 and a 37% reduction in crop acreage occurred over the study period (Makarewicz et al., 2009). Despite these changes in farming practices, nutrient loading at Long Point Cove was still elevated as compared to Sand Point Gully (Makarewicz et al., 2009).

Stream nutrient loading

For Eagle Point Gully, water samples were analyzed for nitrate ($NO_3 + NO_2$, from here on indicated as NO_3–N), total phosphorus (TP), and soluble reactive phosphorus (SRP) 1 day per week and during runoff events from May 2005 to June 2007 as part of the larger sub-watershed study but not reported on by Makarewicz et al. (2009). Daily nutrient runoff from the Eagle Point Gully sub-watershed was calculated by estimating daily discharge and multiplying by nutrient concentration (Makarewicz et al., 2009). Nutrient loading at Sand Point Gully was measured daily and is reported by Makarewicz et al. (2009). *T*-tests were used to compare event loading and total loading (event and daily data combined) between Sand Point Gully and Eagle Point Gully. These data were used to help illustrate the differences in nutrient loading between the two sub-watersheds.

Runoff event

For Conesus Lake, hydrodynamic models have illustrated that stream water carrying sediment and nutrients is focused over macrophyte beds after a rain event (Li et al., 2008). To quantify the effect that event runoff has on nutrient concentrations within a macrophyte bed, we sampled beds in the nearshore downstream from Sand Point Gully on 8 Aug 2008 immediately following a rain event of 50 mm over 2.3 h (National Weather Service spotter located within watershed). Water samples were collected along four 30 m transects during this moderate runoff event (Fig. 2). One transect ran directly from the mouth of the stream out into the open waters of the lake; parallel transects were located every 10 m north of the stream along the macrophyte bed extending from the shore eastward into the lake. Along each transect, water samples were collected at the nearshore edge, center, and offshore edge of the macrophyte bed. Water samples were collected just beneath the surface of the water, filtered on site (0.4-μm polycarbonate filter), stored in ice, and analyzed within 24 h for NO_3–N and SRP.

Macrophyte regrowth experiments

The hypothesis that individual plants receiving greater amounts of nutrients from sub-watersheds with limited or no BMPs will grow more than plants downstream from a comparable sub-watershed where BMPs were established was tested. This was tested in a series of three experiments at the Sand Point Gully, the Eagle Point Gully (one experiment), and the Long Point Cove macrophyte beds (two experiments). The second and third cropping experiments were conducted at Long Point and not at Eagle Point, as sunfish nests and changes in patchy watermilfoil distribution made it difficult to identify cropping areas that would be comparable to those at Sand Point in terms of size and distance from stream.

Within each nearshore bed, replicate 10×10 m plots ($n=2$–4) were selected at the depth of maximum Eurasian watermilfoil biomass (1.5–3 m; Bosch et al., 2009b). At the start of each experiment, S.C.U.B.A. divers cropped plants within each plot by hand or with a machete to an initial height of approximately 50 cm, which was typically 1/3–1/2 of plant height. To determine initial biomass, we sampled each plot by harvesting the above-sediment biomass in 3–4 replicate quadrats (0.25 m^2). After a 2-week period of

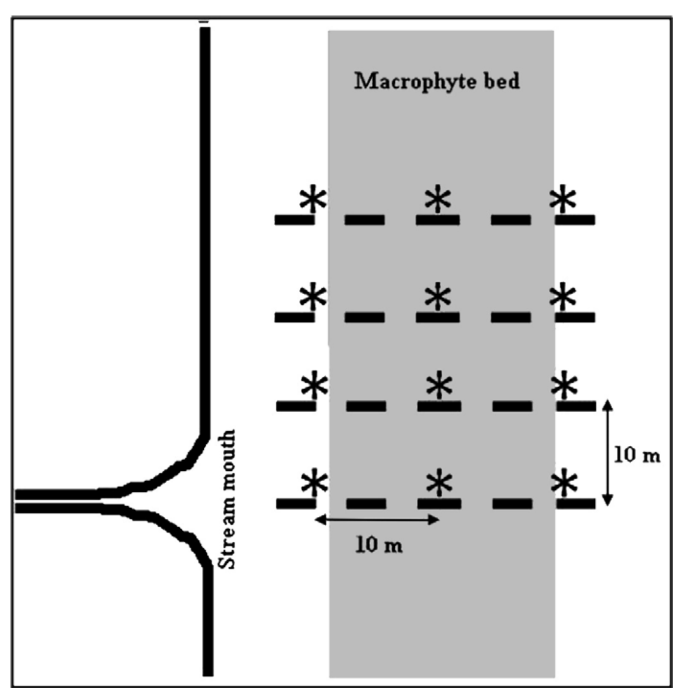

Fig. 2. Sampling grid used to measure nutrient concentrations within the macrophyte beds utilized in the studies immediately following a rain event of 50 mm on 8 Aug 2008. Water samples (*) were collected along four 30-m transects.

regrowth, divers, being careful to avoid sampling in previously harvested areas, collected four additional replicate, 0.25-m^2 quadrats from each of the 10×10 m plots. Regrowth within each cropped plot was determined by quantifying the change in biomass density within each plot. Because the samples were dominated by Eurasian watermilfoil (>98% of the macrophyte biomass), the dried biomass was reported as total grams dry weight per square meter (g DW/m^2) rather than as species-specific weights. *T*-tests compared percent biomass of regrowth between sites using Sigma Stat 3.5 software (Copyright 2006 Systat Software, Inc) with a threshold of $\alpha = 0.05$ for statistical significance.

Responses to water column nutrient enrichment

The hypothesis that pulses of dissolved nutrients in the water column, simulating a 24-hour stream runoff event, would enhance the growth of watermilfoil plants over a 2-week experimental period was tested between June through August 2008 at Sand Point Gully and Eagle Point Gully at depths of 3 m. Our approach measured the growth of individual plants after receiving artificial enrichments of NO_3–N and SRP but not dissolved ammonia (NH_4). Nichols and Keeney (1976a) have shown ammonia to be a preferred source of N for foliar uptake below a ratio of approximately 4:1 (NO_3–N:NH_4–N). This ratio in stream water entering Conesus Lake was much higher (e.g., 10.6:1 for Sand Point Gully, 40:1 for Long Point Gully; Makarewicz and Bosch, unpublished data). Furthermore, seasonal trends and concentrations of NH_4 in stream runoff were unchanged by BMPs at Sand Point Gully and other sub-watersheds while NO_3–N declined by approximately 40%, showing potential as a management tool for watermilfoil growth.

Divers tagged individual plants with numbered Velcro strips and measured plant size to the nearest cm. Each plant was sketched in situ, so that the number, relative location, and length of all parts including branches above the sediment were recorded. Plant height was taken vertically from the surface of the sediment to the apical tip of the main shoot. Any length of stem that ran horizontally on the sediment surface and was anchored by adventitious roots was not included. Since watermilfoil plants can grow as multiple shoots from a single root system, care was taken to examine the base of each shoot to identify the exact branching pattern. Plants with multiple shoots were excluded from the experiments.

Groups of three tagged plants, considered analytical replicates, were then enclosed in individual clear, plastic, 4-liter Zip-Loc bags (R. C. Johnson Co.) containing ambient lake water. A portion of the zip seal was replaced with Velcro to avoid damage to the stems but still provided an adequate seal separating the shoots from both the sediment and surrounding water. N as $NaNO_3$ and P as KH_2PO_4 were injected into each of the 3–7 replicated treatment bags to achieve the desired final nutrient concentrations comparable to historical stream nutrient loading data (Makarewicz et al., 2009). Control bags were injected with ambient lake water through self-sealing ports to reduce leakage. After receiving injections, the contents of each bag were agitated and allowed to mix for several minutes. Water subsamples were then drawn from each bag by syringe to determine initial nutrient concentrations.

After a 24-hour incubation period, the plastic bags enclosing the plants were removed, and plants were allowed to grow for 2 weeks under ambient lake conditions before harvesting. Harvested plants were again sketched and measured to the nearest cm in the laboratory. To quantify biomass increase, new biomass was separated from initial biomass on the basis of the linear dimensions and the drawings of plants obtained at the start of each experiment. Any plant with obvious signs of damage due to the bag or tags (broken stems/shoots or decrease in height) was excluded from the final analysis.

For statistical comparisons, all data were converted to percent growth to correct for size-specific differences in growth rates that might be inherent in the life history of watermilfoil and for any advantage that the larger surface area of bigger plants might provide in terms of nutrient absorption. *T*-tests were first used to compare the growth rates of the enriched and ambient treatments for individual experiments at each site. We also examined the differences between treatments at each site by a two-way ANOVA using time and treatment (nutrient enrichment) as the two factors (Sigma Stat 3.5 software, $\alpha = 0.05$).

Plant and nutrient sample processing

All plant samples were rinsed in tap water to remove epiphytes while zebra mussels attached to plant stems were removed by hand prior to any weight measurements. Quadrat samples were sorted to individual species and dried for 24 h at 105 °C before weighing (Denver Instruments Model P-603D). For nutrient analysis, each water sample collected during experiments was filtered through a 0.4-µm pore membrane filter, stored in ice, and analyzed within 6 h of collection at SUNY Brockport's NELAC certified water quality laboratory using standard methods (APHA 1999, see Makarewicz et al., 2009 for details).

Results

Stream loading

From June 2005 to July 2007, the average and maximum loadings of NO_3–N and TP due to rain events at the Eagle Point stream were 50–100% higher than at Sand Point ($p < 0.0$, *t*-test; Fig. 3). Differences in rain event SRP loading were not significant ($p = 0.278$, *t*-test). No differences were detected in summer nonevent loading of NO_3–N and TP between the Eagle Point Gully and Sand Point Gully streams; however, nonevent SRP loading at Eagle Point was significantly higher than at Sand Point ($p = 0.014$, *t*-test). For Long Point Gully, annual loadings of SRP, TP and NO_3–N were approximately 2, 2.7, and 8.5 times higher than that of Sand Point Gully in 2007 despite the fact that BMPs implemented by landowners had brought about reductions in nutrient loading (Makarewicz et al., 2009).

The effect of watershed runoff on nearshore nutrient chemistry

Water samples collected in the Sand Point macrophyte bed after a moderate runoff event from the nearby stream showed a strong influence of stream loading on nutrient concentrations within the beds (Table 1). After the runoff event, concentrations of NO_3–N were 2.6 mg

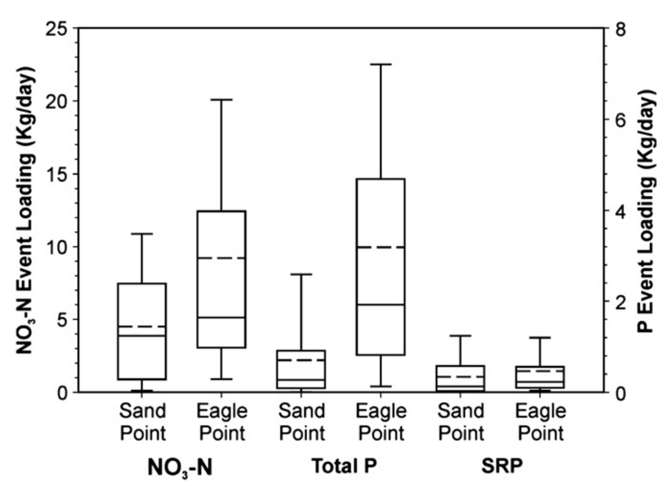

Fig. 3. Comparison of event loading (kg/day) of nitrate (NO_3–N), total phosphorus (TP), and soluble reactive phosphorus (SRP) for Sand Point Gully and Eagle Point Gully from May 2005 to June 2007. The solid horizontal line across the boxes is the median and the dashed line represents the mean. Each box encompasses the 25th–75th confidence intervals and the capped lines are the 10-90% intervals.

Table 1
Nutrient concentration profile for water within the Sand Point macrophyte bed after the 8 Aug 2008 rainfall event (total rainfall was 50 mm).

Position within the macrophyte bed		Distance from stream along shore (m)			
		0 m	10 m	20 m	30 m
Nearshore edge	SRP	<0.04	<0.04	0.09	0.9
	NO_3–N	2.2	4.3	1.9	0.9
Central	SRP	0.08	<0.04	0.09	0.9
	NO_3–N	2.2	5.3	No data	1.9
Offshore edge	SRP	<0.04	<0.04	<0.04	0.1
	NO_3–N	2.6	3.4	2.1	2.6

Soluble reactive phosphorus (SRP, µg P/L), nitrate (NO_3–N mg/L). 0 m = stream mouth. Due to a sampling error, NO_3–N concentrations were not recorded for the Central 20-m sampling location.

N/L at a distance of 30 m from the stream mouth and on the outer edge of the macrophyte bed. Closer to the stream, a maximum concentration of 5.3 mg N/L was recorded and was comparable to those reported for Sand Point Gully stream water (Makarewicz et al., 2009). In contrast, average epilimnion NO_3–N concentrations were generally below 0.5 mg NO_3–N/L in the pelagic during the summer and were non-detectable in the epilimnion by fall (Makarewicz et al., 2001). In contrast to the NO_3–N data, SRP concentrations were low (<0.04–0.9 µg P/L) throughout the sampling area.

Macrophyte regrowth experiments

Plot biomass at Eagle Point Gully more than tripled over a 2-week period (10 June to 27 July) reaching an average final quadrat biomass of 168.8 g DW/m^2 in June. This was equivalent to a 216% increase over the 2-week experiment ($n = 8$ quadrats for each two plots, Table 2) and a rate of growth of 8.2 g DW/m^2/day. Eurasian watermilfoil comprised 94% of the plant biomass at the start of the experiment and 98% at the end. Plot biomass at Sand Point Gully increased by only 34 g DW/m^2 over the same 2-week period. This was equivalent to a 78.5% increase in the quadrat biomass (Table 2) and a rate of increase of 2.4 g/m^2/day. Eurasian watermilfoil comprised 100% of the quadrat biomass at both the start and end of the experiment at Sand Point. This increase in macrophyte biomass at Sand Point Gully was statistically lower than the increase at Eagle Point ($p = 0.026$, t-test). No observable runoff occurred at either of the two streams during the experimental time period.

In a second experiment conducted in July, the amount of growth after cropping was 50.6% at Long Point Cove and 35.2% at Sand Point (Table 2), and the differences between sites were not statistically significant ($p = 0.195$, t-test). Growth rates were 1.66 and 1.02 g/m^2/day, respectively. This represented almost a 50% decrease in growth rate for Sand Point when compared to the rate in June. Eurasian watermilfoil represented 100% and 99% of the total biomass at the start of the experiment for Long Point and Sand Point, respectively, and 100% for both sites at the end of the experiment. Heavy rains (11.56 cm) occurred during 7 of the 14 days in this experiment, delivering elevated nutrient levels to the macrophyte bed.

Table 2
Mean biomass (g DW/m^2) measurements and t-test results for regrowth experiments conducted at the Sand Point Gully (moderate loading), Eagle Point Gully (high loading), and Long Point Gully (high loading) macrophyte beds.

Start date	Site	Initial biomass	Final biomass	% Biomass regrowth	n	df	p
June 2008	Eagle Point	53.4 ± 12.2	168 ± 38.1	216 ± 1.10	2	2	0.026
	Sand Point	43.3 ± 3.92	77.3 ± 2.54	78.5 ± 22.2	2		
July 2008	Long Point	51.0 ± 9.18	74.3 ± 6.12	50.6 ± 4.09	3	4	0.195
	Sand Point	43.3 ± 4.67	57.6 ± 2.75	35.2 ± 12.5	3		
August 2008	Long Point	53.7 ± 9.13	65.5 ± 6.63	22.2 ± 9.89	4	5	0.042
	Sand Point	56.1 ± 11.3	29.6 ± 2.33	0.00 ± 8.57	4		

Values are mean ± S.E.

In a third experiment initiated in August at the Long Point Cove and Sand Point macrophyte beds, the amount of growth after cropping was lower than in both June and July experiments for both sites (Table 2). At Long Point Cove the representation of watermilfoil in the plots declined radically from more than 90% to 48% by the end of the experiment as Eelgrass (*Vallisneria americana*) dominated the sampling area. At Sand Point the watermilfoil composition of the quadrats remained near 100%. Cropped areas within the macrophyte beds at Sand Point exhibited zero regrowth. A statistically significant difference was detected between the amount of macrophyte regrowth at Long Point and Sand Point ($p = 0.042$, t-test).

Seasonal changes in the amount of regrowth at each site were examined using one-way ANOVAs with each experiment serving as a replicate. The macrophyte bed at Sand Point exhibited significantly more growth after cropping in June than in July and August (Table 2, $p = 0.008$, one-way ANOVA). Although no statistical differences in the amount of regrowth were detected between July and August for the macrophyte bed at Long Point, the average growth after cropping in July was more than double than in August (Table 3).

Responses to water column nutrient enrichment

The initial concentrations of NO_3–N (mg/L) and SRP (µg P/L) in our nutrient enrichment experiments were, respectively, 3.90 ± 1.52 (mean ± S.E.) and 0.19 ± 0.33 for the ambient treatments and 78.6 ± 47.2 and 1.09 ± 0.59 for the enriched treatments. The concentrations were variable between experiments, but within experiments there were large and ecologically relevant differences between treatments.

Nutrient enrichment did not result in significant increase of plant growth in either the Eagle Point or Sand Point macrophyte beds in June or in the Eagle Point macrophyte bed in August (Table 3). Although no significant differences could be detected in the July experiments, the average percent growth in the enriched treatment was nearly 4× higher than in the ambient control treatment at Eagle Point and 2× higher at Sand Point.

At Sand Point Gully, the percent growth was higher in the enriched treatment in both the July and August experiments (30–60% more than ambient controls). However, only the August experiment exhibited a statistically significant difference, with plants receiving nutrient enrichment increasing in biomass by 60%, while plants incubated in ambient water increased only by 31% over the 2-week experiment (Table 3, $p = 0.007$, t-test).

At Eagle Point the August experiment was started a week earlier but overlapped with the above August experiment at Sand Point. Unlike at Sand Point Gully, the percent biomass increase at Eagle Point

Table 3
Summary data and results of t-tests comparing growth of plants incubated in ambient lake water to plants incubated in nutrient enriched lake water for experiments conducted during 2008 at Eagle Point Gully and Sand Point Gully.

Dates	Site	Treatment	SRP (µg P/L)	Nitrate (mg NO_3–N/L)	% Biomass growth	n	df	p
Jun	Eagle	Ambient	1.5 ± 0.2	0.03 ± 0.25	114 ± 95.5	2	5	0.794
		Enriched	56 ± 20	0.45 ± 0.12	95.6 ± 26.3	5		
	Sand	Ambient	5.5 ± 3.4	0.04 ± 0.00	36.7 ± 17.1	2	5	0.445
		Enriched	37 ± 12	0.30 ± 0.09	97.8 ± 43.7	5		
Jul	Eagle	Ambient	2.2 ± 0.9	0.04 ± 0.04	53.3 ± 24.0	3	7	0.254
		Enriched	42 ± 7.3	1.05 ± 0.16	222 ± 91.9	6		
	Sand	Ambient	2.8 ± 2.3	0.02 ± 0.02	49.8 ± 6.37	7	12	0.262
		Enriched	39 ± 17	1.05 ± 0.35	103 ± 33.9	7		
	Sand	Ambient	6.5 ± 0.7	0.01 ± 0.01	59.9 ± 12.4	7	12	0.110
		Enriched	120 ± 11	1.53 ± 0.13	87.2 ± 9.85	7		
Aug	Eagle	Ambient	3.9 ± 1.2	0.04 ± 0.01	59.7 ± 14.1	7	11	0.858
		Enriched	105 ± 1.2	1.53 ± 0.47	63.5 ± 15.5	6		
	Sand	Ambient	3.9 ± 1.2	0.04 ± 0.01	31.3 ± 6.91	7	12	0.007
		Enriched	105 ± 18	1.53 ± 0.19	59.0 ± 5.06	7		

Soluble reactive phosphorus = SRP. Values are mean ± S.E.

Table 4
Summary data for nutrient enrichment experiments (A) and results of two-way ANOVA using time (month of individual experiments) and treatment (nutrient enrichment) as the two factors (B).

A. Macrophyte bed	Percent biomass growth (mean ± S.E.)			
	Ambient controls		Nutrient enrichment	
Eagle Point	64.4 ± 11.4		111.0 ± 27.5	
Sand Point	45.3 ± 6.01		85.20 ± 11.9	
B. Factor	**Sum of squares**	***df***	***F***	***p***
Eagle Pt. time	13,837.980	2	0.630	0.538
Eagle Pt. treatment	4128.069	1	0.376	0.544
Eagle Pt. time × treatment	19,849.737	2	0.904	0.414
Sand Pt. time	8096.523	2	1.744	0.188
Sand Pt. treatment	13,728.712	1	5.915	0.020
Sand Pt. time × treatment	1138.673	2	0.245	0.784

Percent growth represents the average increase in plant biomass for the experiments conducted within each macrophyte bed. Experiments were conducted during 2008 at Eagle Point Gully and Sand Point Gully.

was nearly identical in both the ambient controls (60%) and the enhanced nutrient conditions (64%, Table 3).

Two-way ANOVAs were used to test for consistent differences in plant growth between the ambient and enriched treatments at each site. For the experiments at Sand Point Gully, nutrient enrichment resulted in plant growth (mean = 85%) that was significantly higher ($p = 0.02$) compared to plants incubated in ambient nutrient levels (mean = 45%, Table 4). Although the differences between nutrient treatments at Eagle Point were not significant ($p = 0.54$, two-way ANOVA), plants in nutrient enriched conditions experienced higher growth (mean = 111%) than plants incubated in ambient lake water (mean = 64% growth, Table 4). For both Sand Point and Eagle Point there were no significant differences in growth between experiments (i.e. time effect $p = 0.188$ and 0.538, respectively) and no significant interaction of time × treatment ($p = 0.784$ and 0.414, respectively), indicating that effect of nutrient enrichment was not influenced by the time of the growing season in which an experiment was conducted.

Discussion

Tributaries flowing into Conesus Lake from catchment basins with primarily agricultural land use (60–80%) deliver high concentrations and flux of dissolved nutrients into the lake littoral (Makarewicz et al., 2009). For example, concentrations of NO_3–N were 13–24 times and SRP concentrations were 3–24 times higher than in lake water in one Conesus Lake stream over a 3-day period in June 2002. Total loading from Long Point Gully from 1 Sep 2006 to 31 Aug 2007 was approximately 2800 kg for TSS, 3580 kg of NO_3–N, and 21.2 kg of SRP (Makarewicz et al., 2009). The growth of filamentous algae nearly doubled in direct response to stream SRP loading (D'Aiuto et al., 2006, Makarewicz et al., 2007), and it is near these tributaries, where nutrient levels are elevated (e.g., Table 2), that the largest and most dense Eurasian watermilfoil beds in Conesus Lake are located (Bosch et al., 2009b).

Reductions in nutrients and soil delivered to downstream systems achieved by implementation of agricultural BMPs (Makarewicz et al., 2009) have had a local effect on macrophyte communities in Conesus Lake. In three beds downstream from sub-watersheds where major BMPs were implemented, including the Sand Point study site, quadrat biomass decreased by 30–50% within 1 or 2 years of BMPs implementation and was statistically lower than pre-BMP values in 7 of 11 sample years (Bosch et al., 2009b). In three macrophyte beds downstream from sub-watersheds where minimal or no BMPs were introduced (e.g., Long Point Cove), biomass was statistically indistinguishable from Pre-BMP values in 12 experimental sample years.

The pathway(s) by which the reported decreases in stream nutrient loading might have caused declines in Eurasian watermilfoil biomass remains uncertain. Numerous studies point to sediment pore water as the primary source of P and N for rooted aquatic plants (e.g., Carignan 1982, Chambers and Prepas 1990, Smith and Barko 1990, Barko et al., 1991). However, uptake from the water column through the foliage is also possible. For example, in experiments with whole watermilfoil plants grown in the laboratory, the contribution of root and shoot to NH_4 uptake depended on the relative concentrations in the water column and in the sediment (Nichols and Keeney 1976b). If present in concentrations of about 0.1 mg/L NH_4–N, the water could be an important source of N (Nichols and Keeney 1976a). Similarly for SRP, shoot uptake by Eurasian watermilfoil plants collected from January to November by Smith and Adams (1986) in Lake Wingra, WI, was approximately 27% of the total P uptake.

Field studies have provided some important insights on how ecological conditions influence the balance between sediment and water column uptake. Carignan (1985) found that in the course of the growing season, root uptake by Eurasian watermilfoil and other macrophytes caused major reductions in pore water concentrations of SRP and NH_4. Anderson and Kalff (1986) subsequently documented limitation of Eurasian watermilfoil growth by sediment NH_4 supplies in situ. Similar experiments in which SRP was added to the sediments have provided mixed results. Carignan and Kalff (1980) reported that root uptake accounted for 70–90% of total P uptake when sediment concentrations were 9× greater than water concentrations, but shoot uptake accounted for 70% of total P uptake in other experiments when concentrations were equal (Carignan 1985). Experiments conducted by Anderson and Kalff (1986) and by Zhu et al. (2008) showed that supplies of sediment SRP were not limiting for the growth of Eurasian watermilfoil in situ, once again supporting the primacy of sediment P as strongly advocated by Barko et al. (1991). It may be that, as reported by Nichols and Keeney (1976a) and Carignan (1982), the importance of root and shoot N and P uptake to the nutrition of Eurasian watermilfoil depends on the relative concentrations of water column and pore water nutrients. Thus, plants near nutrient sources of runoff, such as streams draining agricultural lands where concentrations of dissolved nutrients can be high, may meet a higher proportion of their requirements from supplies in the water column. This hypothesis is not supported by experiments in running waters of the South Saskatchewan River (Chambers et al., 1989) where *Potamogeton crispus* grown under various combinations of sediment and water column N and P concentrations relied almost exclusively on root uptake. However, we are not aware of any studies in which comparable results were reported for lake populations of Eurasian watermilfoil.

It seems unlikely that reductions in sediment nutrients could have been a major factor in the biomass losses of Eurasian watermilfoil beds bordering managed tributaries in Conesus Lake because the changes occurred in a relatively short period of time (less than 5 years) (Bosch et al., 2009b). Assuming sedimentation rates are equal to the maximum of 2.0 mm/year reported by Rooney et al. (2003) for littoral macrophyte beds, it would be roughly 24 years before reductions in particulate nutrient loading might be evident at the typical rooting depths of 50–100 mm reported for Eurasian watermilfoil plants (Carignan 1985). Sedimentation rates in our sites would have to be 5× the reported maximum to affect nutrient concentrations at rooting depths within 5 years.

In the present study, macrophyte regrowth was compared between sites receiving high loads of nutrients from tributaries (Eagle Point Gully, Long Point Gully) to a site in which BMPs established in 2003–2004 had reduced nutrient loading to relatively low levels (Sand Point Gully). In two of the three cropping experiments, watermilfoil biomass regrowth was statistically higher at the sites receiving higher loads of stream nutrients [Eagle Point Gully (this study) and Long Point Gully (Makarewicz et al., 2009)]. In one experiment carried out from 16–30 July, there were no significant differences in growth between the two sites receiving stream water from the Sand Point Gully (BMPs, low nutrient loads) and Eagle Point Gully (higher nutrient loads) watersheds. There were 7 days of moderate to heavy rain in 7 of the 14 days of the experiment, delivering elevated nutrient levels to the macrophyte bed. Perhaps plants at both sites were saturated with water column nutrients, and coupled with low incident light, these conditions were partially responsible for the low growth and absence of enrichment in this experiment.

A general trend of decreasing amounts of regrowth from June to August (Table 2) was observed. This trend is consistent with results from temperate Fish Lake where growth of Eurasian watermilfoil plants was high during early summer and decreased in July and August (Lillie et al., 1997) and with the studies by Carignan (1985) who found that supplies of pore water nutrients decreased during the macrophyte growing season. We cannot discount the potential importance of several other variables in determining the results of the cropping experiments. Nutrients in groundwater flow entering the littoral zone may have been a factor (Lodge et al., 1989). Moreover, there is evidence that the congeneric species *M. heterophyllum* can store large amounts of P absorbed from the water column for subsequent use (i.e., luxury consumption) (Chagnon and Baker 1979). If Eurasian watermilfoil is capable of luxury consumption, then growth might be temporally separated from peaks in nutrient acquisition, confounding any attempts to correlate short-term patterns of nutrient loading with plant growth. Overall, the finding that rates of regrowth were generally lower at Sand Point than in sites with higher nutrient loading is consistent with the results of the multiyear experiment carried out by Bosch et al. (2009b), who indicated that reductions of nutrient loading from agricultural watersheds achieved by establishing BMPs lead to decreases in Eurasian watermilfoil growth and biomass in the nearshore of Conesus Lake.

Neither the results of the short-term cropping study nor the multiyear experimental study by Bosch et al. (2009b) provides any insight on whether reductions in Eurasian watermilfoil biomass downstream from managed agricultural watersheds were the result of decreases in water column or sediment pore water nutrient supplies, or both. The in situ nutrient enrichment experiments were designed specifically to test for growth of individual plants in response to enhanced water column NO_3–N and SRP. Results compiled for all five experiments indicate that the dry weight biomass of individual plants at Sand Point increased statistically relative to controls, whereas no statistical differences were detected for the four experiments conducted at Eagle Point. While these results are not conclusive, they provide credible evidence that the growth of Eurasian watermilfoil plants can be influenced by foliar uptake of dissolved nutrients in stream runoff. More research is required to evaluate why enrichment of growth was significant at the managed Sand Point watershed (moderate loading) and not significant at the Eagle Point site (higher loading). Saturation levels may have been reached at Eagle Point as both pore and stream water concentrations of nutrients were already high and plants were not responding to enhanced nutrient levels.

The extent to which stream loading can fuel macrophyte bed growth locally, the relative importance of dissolved and particulate nutrient loading, and the ability of macrophytes to alter or create their own microhabitat within the nearshore in this process remain uncertain. For example, macrophyte beds, once developed, can significantly alter the hydrodynamics of the littoral zone of lakes by reducing flow velocities and dampening wave action (Wetzel and Corners 1979, Madsen et al., 2001). The magnitude of these reductions depends on the dominant species and density of the plant community (Petticrew and Kalff 1992). Losee and Wetzel (1993) documented median flow velocities of 0.06 cm/s within macrophyte beds compared to 0.20 cm/s immediately outside the beds. Such reductions in water movement limit the re-suspension of sediments while increasing sedimentation rates of fine inorganic and organic particles and associated nutrients (Rooney et al., 2003). In mesotrophic Lake Memphremagog (Canada-Vermont), maximum sediment accumulation rates were 5× higher in dense macrophyte beds than in nearby offshore areas (2 and 0.4 mm/year, respectively). Here sedimentation rates were positively correlated with P deposition rates, indicating the potential for nutrients to accumulate in littoral sediments occupied by macrophyte beds (Rooney et al., 2003).

Although sedimentation rates in Conesus Lake have not been measured, Li et al. (2008) used hydrodynamic modeling to study the dispersal of stream plumes in littoral areas containing beds of Eurasian watermilfoil. The model predicts that macrophyte beds have the potential to reduce plume flow currents by as much as 74%, an estimate that is consistent with the 70% reduction in median values reported in field studies by Losee and Wetzel (1988, 1993). Such high reductions in stream plume velocity should increase sedimentation rates within the macrophyte beds as shown by Rooney et al. (2003) and also prolong the exposure of plants to waters laden with dissolved nutrients (Li et al., 2008). While the significance of this stream plume capture phenomenon remains unknown, the potential for fertilization of the littoral macrophyte beds must be high, as indicated by the amount of sediment and dissolved nutrients delivered by streams into Conesus Lake. Loading by the Sand Point Gully tributary has decreased significantly since agricultural BMPs were established in the watershed in 2003. However, even the relatively moderate load of sediment and dissolved nutrients in the stream (904 kg TSS, 420 kg NO_3–N and 8.76 kg of SRP) (Makarewicz et al., 2009) contributes significantly to nutrient levels in the macrophyte beds, as indicated by high water column concentrations of NO_3–N within the Sand Point bed after a moderate rain event on 8 Aug 2008 (see event concentrations in Table 1 and ambient concentrations in Table 3). Elevated levels of SRP were not observed in the macrophyte bed after this event, possibly due to rapid uptake by benthic algae, phytoplankton (Howard-Williams 1981), and macrophytes.

This study is unique in testing the effects of stream loading and foliar uptake on Eurasian watermilfoil growth experimentally and in situ. Clearly, additional studies are required to fully understand the dynamics of this system. Our experimental data do indicate that short-term macrophyte growth can be influenced by the levels of nutrients being lost from a sub-watershed. In a sub-watershed where major BMPs were implemented and nutrient loading was lower, macrophyte growth was significantly lower than downstream from sub-watersheds where nutrient loading was higher. Also, foliar uptake of nutrients by Eurasian watermilfoil was observed, indicating that macrophyte growth could be affected by fluctuations in stream concentrations of nutrients. As Noll et al. (2009) suggest, the resultant decrease in rooted macrophytes may play a pivotal ecological role in the nearshore zone by accelerating the decrease in macrophyte biomass. As the beds decrease in size and biomass so does the ability to reduce the velocity of the stream water entering the nearshore (Li et al., 2008) allowing N- and P-laden water and suspended particles to move off shore and not accumulate within the beds. Combined with the likely reduction in the organic fraction of P (Noll et al., 2009) and reduction in N and other nutrients in stream water from the sub-watershed due to implementation of BMPs (Makarewicz et al., 2009), the physical loss of the macrophyte bed itself may serve to accelerate the decrease in macrophyte loss within the system with time.

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