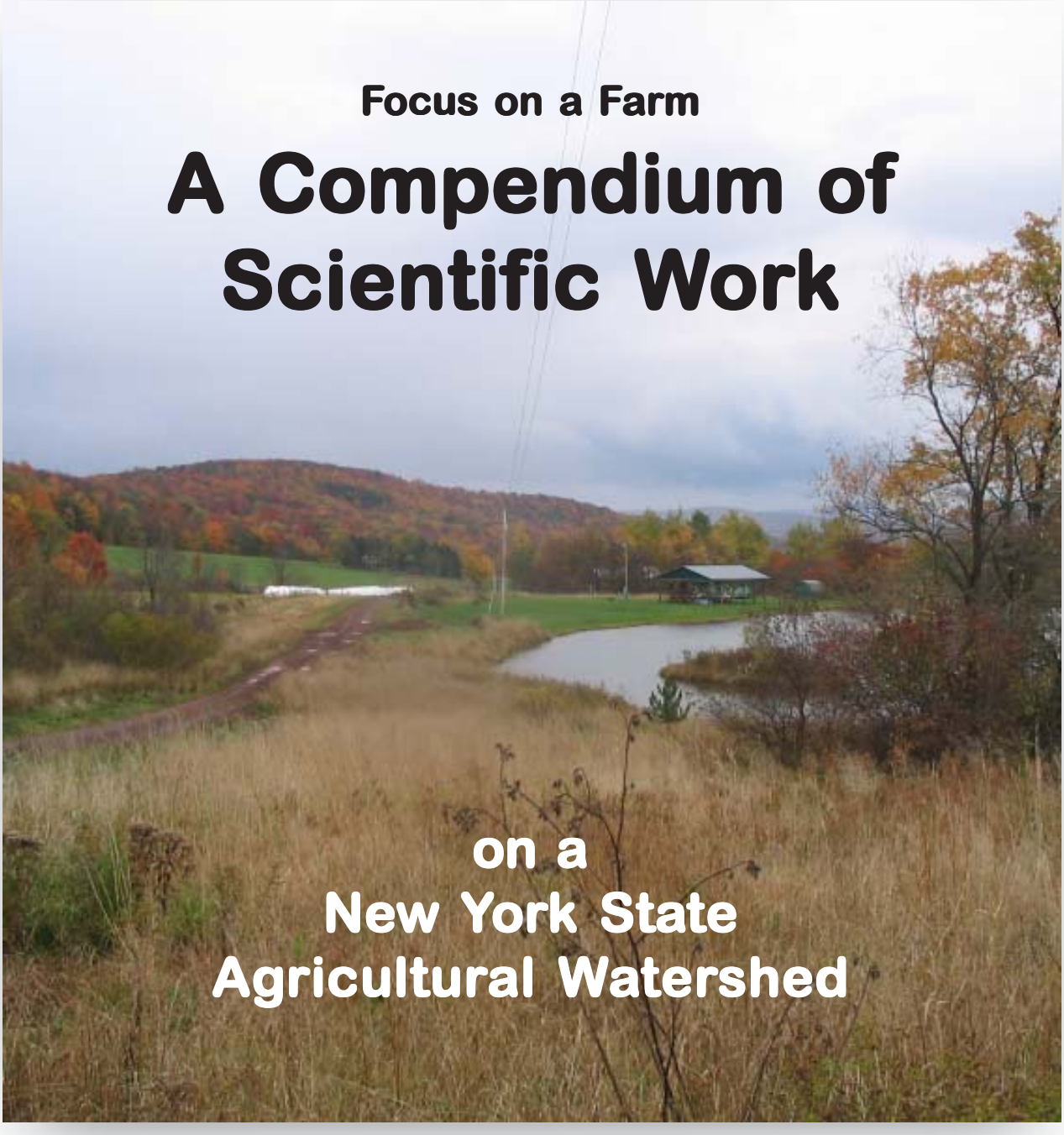

Focus on a Farm

A Compendium of Scientific Work

on a New York State Agricultural Watershed

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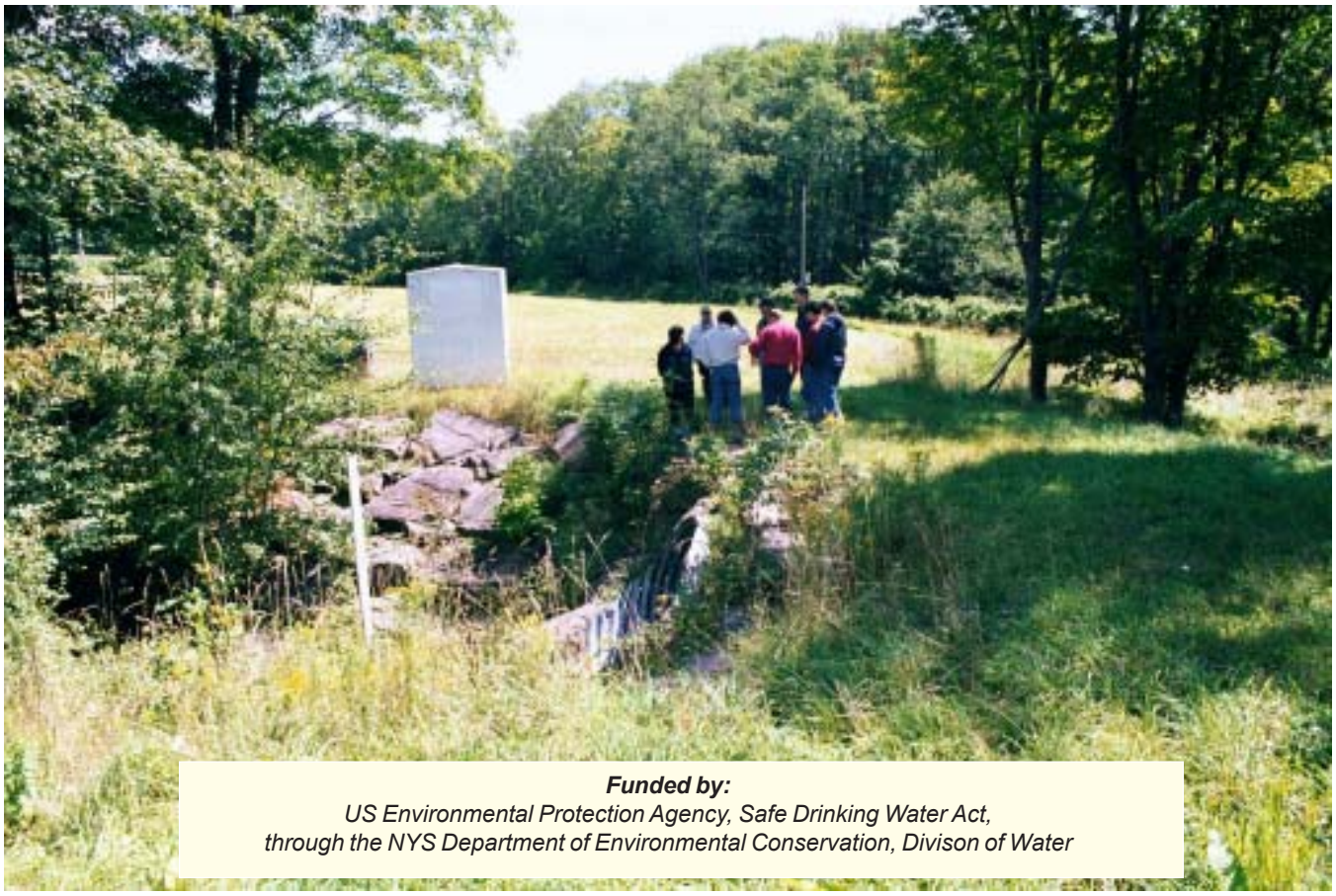

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Executive Summary

This document integrates and synthesizes multiple reports describing completed studies on a farm and nearby forested site in the Catskills region of upstate New York. Delaware County, through its Action Plan (DCAP), and the Watershed Agricultural Council (WAC), together with their watershed partners, have made a substantial investment in comprehensive scientific and technical work on two small watersheds — one devoted to farming and the other serving as a forested control basin with very little human activity. Much of this work is monitoring and assessment in the fullest sense, and represents a solid and comprehensive body of understanding that is of substantial interest and applicability, especially for the planning and management of phosphorus.

Paired Watersheds

Two basins were selected in 1992 as a pair after an exhaustive exploration by researchers from the New York State Department of Environmental Conservation (DEC). In order to enable detained water quality monitoring, several critical criteria for the two basins had to be met. Most importantly, a farm was needed that had a drainage basin with as few confounding factors as was feasible. Ideally this required finding a basin which constituted essentially the farm with no upstream influences. Second, the farm should serve as representative of the small dairy farms typically found in the New York City Watershed area and the pollution problems generally associated with them. Having located such a farm, it was then desirable that a second basin chosen as a "control" be located nearby and be of comparable size to that of the farm basin, with similar topography and pattern of precipitation. The R Farm and the Shaw Road sites meet these criteria.

Monitoring Sites

Staff of the DEC installed automated monitoring stations on the outlets of the two tributaries draining the two sites. Monitoring to evaluate nutrients and sediment in the tributaries began in June 1993. A total of 7,225 samples were analyzed for load calculations during the study period. Of these, 4,473 were collected at the farm and 2,752 were collected at the control site. A little more than 1,300 samples were collected over a two year period prior to the adoption and implementation of management practices on the farm to establish baseline conditions. After a 17 month period during which best management practices (BMPs) were installed, monitoring resumed in November 1996 and ended in November 2006. An additional 5,900 samples were collected during the ten-year post-BMP period.

In conjunction with the DEC monitoring, multiple research projects have conducted field and laboratory studies to gain understanding of the sources and movement of nutrients and sediment especially as influenced by the management practices on the farm. Cornell scientists have tested several models to take advantage of the rich data set available from the DEC. This work confirms that repeated applications of manure, and the consequent enrichment of the soil, greatly enhances the movement of phosphorus to the stream.

In terms of reducing the risk of phosphorus moving to streams and quantifying the reductions, one of the most dramatic accomplishments was the precision feeding research

conducted by Cornell scientists (the storage of manure is thought to have contributed greatly to reducing loads from the farm as well). Their monitoring of the gains due to reducing the imports of phosphorus in feed to the farm could reduce the amount of phosphorus in manure by 30 percent or more.

The time series of data obtained from the two monitoring stations at the outlets to the two basins, when inspected by eye, did not obviously manifest clear improvements in water quality following the management practices. By developing and applying statistical methods using the paired data sets, the DEC researchers and their colleagues were able to show significant reductions in farm losses of sediment and nutrients. This was confirmed by biological assessment of the farm stream. However, the farm stream has nutrient and sediment levels that remain substantially higher than that of the control site. There may be a level below which it is unreasonable to seek further improvements in water quality that drain farms devoted to animal production.

This study well demonstrates the scientific and statistical power of monitoring paired sites representing a farm, without confounding factors, and a control site having minimal human influences. In this regard, this study is unique.

Protecting the New York City Drinking Water Supply

Background

New York City's three major systems of drinking water supply, the Catskill, Delaware, and Croton, are located within a 125-mile radius to the north and northeast of the City, and provide water for nine million people (Fig 1). The total watershed area of 1,950 square miles contains 19 reservoirs, and covers parts of eight New York counties. New York City has long maintained a high quality of drinking water supply drawn from this unique system. It has been satisfying water quality and watershed management criteria required by the U.S. Environmental Protection Agency (EPA) for its Filtration Avoidance Determination (FAD) since 1993. However, excessive nutrients entering New York City's reservoirs from various land uses can threaten the quality of the water supply. Eutrophication of a surface drinking water system always raises concern because of the potential health risks associated with certain byproducts that are formed when water containing high levels of algae and other suspended organic matter is disinfected.

Eutrophication of the Cannonsville Reservoir (Fig. 2), the City's third largest reservoir, has been a problem since not long after its completion in 1965. Studies conducted in the early years of the 1970s confirmed that the reservoir was "in advanced stages of eutrophication" (Brown et al. 1980). A major land use in the Cannonsville watershed is farming; farms located there are predominantly dairy and livestock enterprises. In late 1977, the United States Department of Agriculture (USDA) and the EPA created a Model Implementation Program (MIP). Its purpose was to implement land management practices and relate them to water quality changes (Brown et al, 1983). In 1978, the two agencies selected the Cannonsville Reservoir basin as one of seven MIP demonstration sites. As phosphorus (P) is the limiting nutrient for algal growth in Cannonsville, (Wood, 1979; Longabucco and Rafferty, 1998; New York City Department of Environmental Protection, 2001), the objective was to control excess P losses in barnyard runoff, and to a lesser extent, in cropland runoff, and evaluate the effects on reservoir water quality (Brown et al. 1983). This major implementation, monitoring and modeling effort continued through the early 1980s and produced valuable information on P losses from dairy barnyards (Robillard and Walter, 1983b),

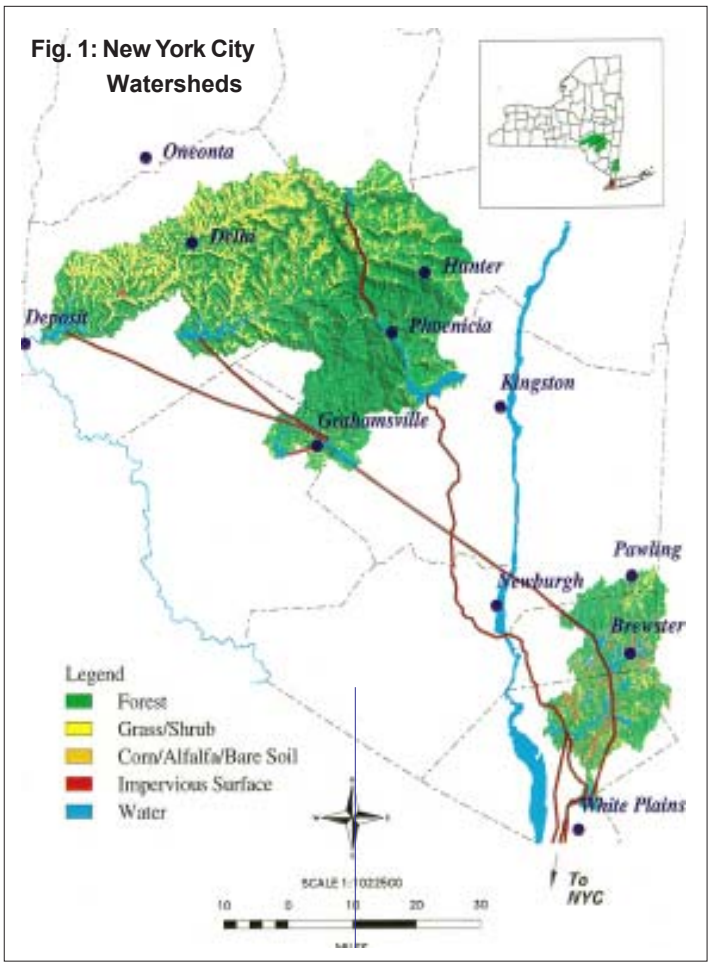
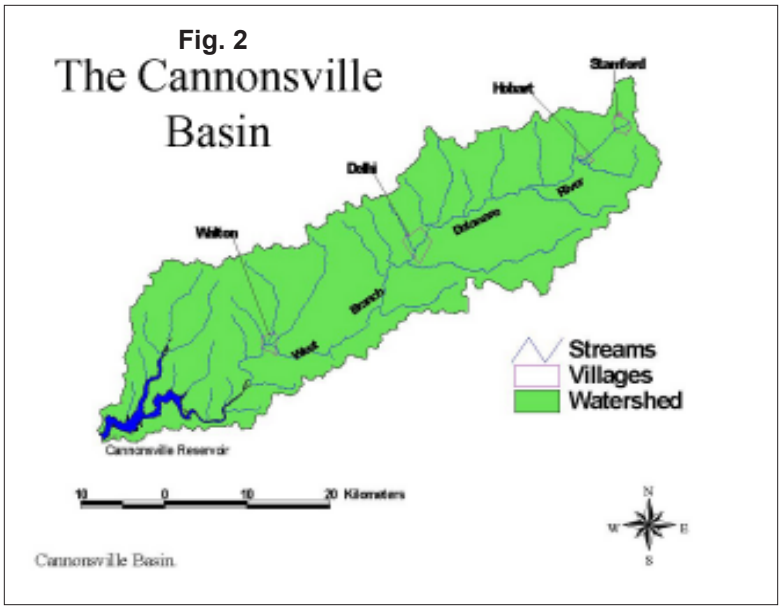


Fig. 1: New York City Watersheds

manure spreading schedules (Robillard and Walter, 1983a), riverine phosphorus transport (Brown et al. 1983), modeling of streamflow and nutrient transport (Haith et al. 1983), and factors affecting eutrophication of the Cannonsville Reservoir (Brown et al. 1986).

The Federal Safe Drinking Water Act (SDWA) requires filtration of water supply systems that draw water from surface sources unless they meet criteria specified under the Surface Water Treatment Rule (SWTR). A water supplier that meets the criteria may be granted "Filtration Avoidance". Currently, the New York City (NYC) system meets the filtration avoidance criteria. However, a potentially critical exception is the Cannonsville reservoir. The Cannonsville Reservoir has had a long history of eutrophication due to excess loading of phosphorus from the West Branch of the Delaware River associated primarily with dairy agriculture and point source discharges (Brown et al., 1986; Brown et al., 1989; Effler and Bader, 1998). In the past, as now, nonpoint sources were identified as a major cause of water quality impairment in the reservoir. Farming is a primary contributor of nonpoint source phosphorus in the Reservoir basin (Brown et al., 1989; New York City Department of Environmental Protection, 2001) and has been estimated to account for about 70 percent of the total P load entering the Cannonsville Reservoir (Longabucco and Rafferty, 1998; Longabucco, 2001). Dairy farming is the main type of farming in the Cannonsville Reservoir Basin. These farms grow forages and purchase concentrate feeds, and are typical of New York State dairies, in that more P is imported in the form of feed and fertilizer than is exported in milk, meat, or crops sold. This therefore results in a net annual accumulation of P on the farm ranging from 19 to 41 kg/cow (Klausner, 1993; Klausner et al., 1998).

Fig. 2



Major farm sources of P include land spreading of manure, barnyard runoff, and over-fertilization of cropland. Thus, there is a continual threat not only of P, but sediment, pathogens and other nutrients as well, entering and degrading the unfiltered drinking water supplies from agricultural and other land uses.

In order to avoid the need for a costly filtration system, NYC opted to implement agricultural nonpoint source management measures, together with urban nonpoint source and more stringent point source controls, in its watershed.

Watershed Agricultural Program

The New York State Department of Agriculture and Markets, and New York City, in association with the principal agencies concerned with the New York city Watershed, established a NYC Watershed Agricultural Program (WAP) in 1992. WAP is founded on a non-regulatory foundation relying on voluntary participation of farmers with the aid of financial and supportive incentives. Senior staff, representing Cornell Cooperative Extension, the Soil Conservation Service, and the Soil and Water Conservation Districts, proposed and developed the concept of Whole Farm Planning (WFP). The underlying WFP goal is to systematically plan operations on the farm by integrating environmental and business objectives. A key point is to take explicitly into account

water quality aims and farm economics. WFP was adopted by the WAP as the primary means of protecting NYC water supplies from farm-related pathogen sources and other nonpoint source pollution, as well as maintaining a viable agricultural community in the watershed. Subsequently, the economic objectives were dropped.

To protect the water supply from excessive nutrient loads (as well as waterborne pathogens), the New York City Department of Environmental Protection (DEP) organizes and funds the voluntary incentive-based WAP. Administered by the Watershed Agricultural Council, which was incorporated in 1993, and with the collaboration from farm owners, the Program has implemented BMPs on more than 85 percent of the farms located within the Delaware/Catskill region of the water supply basin. The BMPs were designed to reduce losses of nutrients (P and nitrogen), sediment and pathogens from farmlands; they were typically improvements to infrastructure and management. A farmer-oriented WFP process in which both the environmental and economic viability of the farm were considered, determined each set of BMPs.

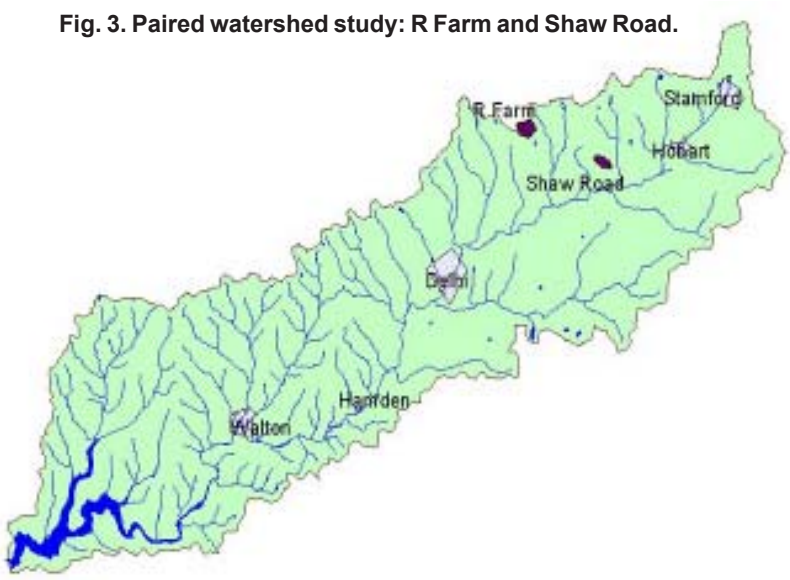
WAP launched the WFP program through a carefully planned experimental Phase I. The objectives were to test WFP on selected farms as pilots and demonstrations. This Phase I began in 1992. Ten demonstration farms were selected on which to develop, test and demonstrate the WFP method. A particular challenge was to establish a sound monitoring program to assess the effects on water quality of the management practices adopted in the WFP.

Assessing improvements in water quality following the implementation of BMPs on operating farms is notoriously difficult. A first requirement is to conduct multiple years of pre- and post-BMP implementation monitoring (USEPA, 2000). Natural variability fluctuates widely from year-to-year. Therefore at least two to three years of monitoring is desirable prior to the implementation of the management practices to be assessed, and at least that many following the implementation. In many cases there may be significant lags expected between changes on the land and effects in water quality. Hence the monitoring may be conducted over many years depending upon the circumstances. Account must also be taken of factors not directly related to the management practices that may also cause changes in water quality. Such changes can include: varying animal numbers; cropping patterns or land uses; amount of impervious areas; stream discharge; precipitation; ground water table depth; or other climatic or hydrologic variables (USEPA, 2000).

Scientific Context for the R Farm Study

With the support and encouragement of the Watershed Agricultural Council (WAC), scientific staff of the DEC established a long-term paired watershed study in 1993 (Fig. 3) on one of the ten demonstration farms, known as the R farm, to evaluate changes in nutrient and sediment loading attributable to implementation of BMPs. These BMPs included manure storage and management, rotational grazing, and improved infrastructure.

Fig. 3. Paired watershed study: R Farm and Shaw Road.



A particular difficulty in assessing the effects of management practices on receiving waters is the confounding effects of upstream influences. These, together with the variability of the weather, can render it very difficult to distinguish the water quality effects (the signal) from all the non-relevant factors (noise). The DEC investigators defeated both difficulties by finding a watershed drained only the farm being treated and hence had no upstream confounding influences (Longabucco and Rafferty, 1995). This

representative upland farm is located in the watershed of the West Branch of the Delaware River (WBDR), which is the primary tributary of Cannonsville Reservoir. To address the issue of climate variability, the DEC researchers utilized a nearby non-agricultural catchment in the vicinity of the R Farm, thus creating a paired watershed monitoring design, with the R Farm as the treatment watershed and the forested watershed as the control.

Automated stream water monitoring provided data to calculate nutrient and sediment loads from the farm watershed for all runoff events and baseflow periods during a two-year pre-treatment period and a ten-year post-treatment period. Statistical control for inter-annual climatic variability influencing the farm was provided by matched monitoring of pollutant loads from a nearby 90-ha forested watershed.

Scientifically sound quantification of the effects of farm management programs on water quality is critical to agencies responsible for water resource protection. During the past decade, the Watershed Agricultural Program has invested many millions of dollars in farm BMPs to more securely protect the integrity of the New York City water supply. This investment has been especially important in the Cannonsville Reservoir Basin. It has historically been the reservoir with by far the lowest quality in the Catskill-Delaware system part of the Watershed and includes most of the dairy farms still producing in the entire watershed area. Since the establishment of the paired watershed study in 1993 other complementary scientific investigations have focused on the farm and forested site with the purpose of establishing a better understanding of the processes governing losses of pollutants to water. Therefore substantial research has been conducted to document claimable reductions in stream water loads resulting from agricultural BMPs implemented as part of an effort to control eutrophication of Cannonsville Reservoir. The findings from this body of research are summarized here.

Summary of the Paired Watershed Study

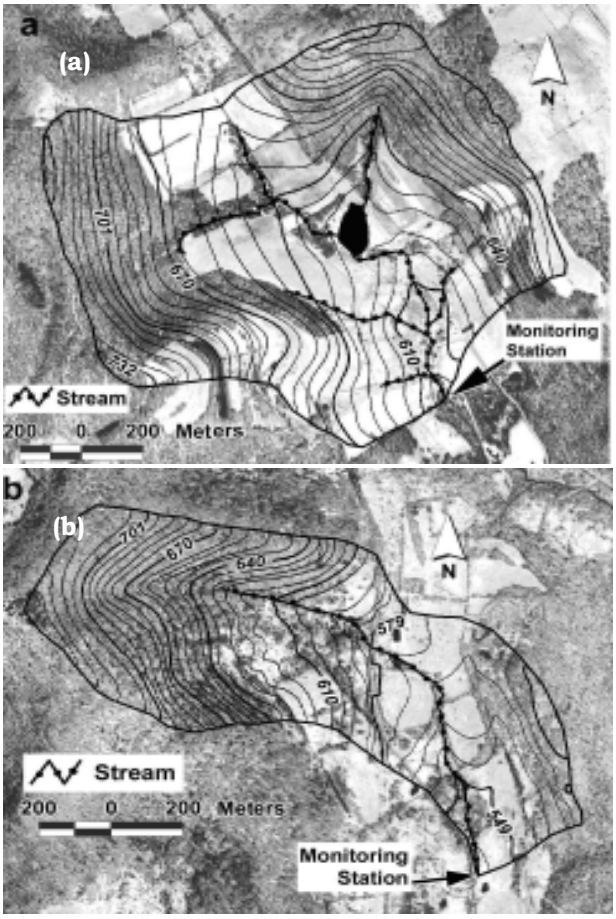
(from Bishop et al. 2005 and Bishop et al. 2007)

Study Sites

The 160-ha R farm (Fig. 4a) is occupied by a third-generation dairy farm that maintains approximately 80 milking cows and 35 heifers. The farm is typical of upland dairy operations of the Catskills region of New York in that the barn is located in the valley bottom, close to a central stream. Land use on the watershed is 53 percent deciduous forest, 25 percent improved pasture and hay, 7 percent corn rotation, 13 percent unimproved pasture, and 2 percent impermeable areas. Deciduous forest and unimproved pasture largely dominate the upper slopes of the watershed, while crop fields and improved pasture tend to be located on the lower slopes. Impermeable surfaces such as barnyards, roads, and farm buildings are mostly near the stream. During the grazing season (early May to late October) cows frequently cross the stream and saturated areas to reach pasture.

The control watershed (Fig. 4b), located 6.4 km east of the farm, covers about 90 ha and is comprised of mostly forest and old fields. A nonfarm control was selected because no significant changes in the watershed were expected during the study period. In contrast, working farms may modify operational practices or go out of business altogether over the course of a long-term study and cannot be relied on to provide the consistent control necessary for describing natural environmental variability. The nonfarm control watershed contains several seasonal residences and one septic system, but has no recent history of manure application and no significant anthropogenic P inputs, aside from atmospheric deposition. Land use is 78 percent deciduous forest, 22 percent shrub and grasses, and <1 percent impermeable areas.

Fig. 4. Maps of (a) farm site and (b) control site, showing watershed boundaries, 6-m (20-ft) contours, perrenial stream channel, and monitoring stations.



Monitoring Scheme

An automated stream monitoring station was established at the outflow of each watershed. It consisted of a heated shelter housing a refrigerated automatic sampler and a data logger, and two parallel pipes containing sensing equipment and sampling lines, through which all stream flow was channeled. A 43-cm-diameter pipe provided accurate measurements of the low to moderate discharges that occur most of the year, and a 2.1-m-diameter culvert pipe effectively handled high flows. Covered with soil and stone to form a dike, the pipes remained ice-free, thereby increasing accuracy

during freezing weather. Flow was recorded every 10 minutes by the data logger, using input from combination level–velocity electromagnetic sensors located in the pipes. Stream flow rating curves were developed for each site through manual stream gauging and were updated annually. Heated and unheated rain gauges were installed at the sites to differentiate between rain and snowfall precipitation.

Water samples were collected at least weekly during baseflow periods, and more frequently during runoff events. Event-based sample collection was triggered by the onset of precipitation in the case of rainfall events, and a rise in stream stage of at least 0.03 m. Frequency of sample collection was directly related to the rate of stream rise and fall, up to a maximum rate of six samples per hour. The number of samples collected per event typically ranged from three to more than twenty, depending on event magnitude and duration, which ranged from several hours to several days. All events that occurred during the six-year study period were sampled.

Stream-water samples were analyzed by the New York State Department of Health in Albany, NY, for total phosphorus (TP), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), nitrate + nitrite (NO_x), total ammonia (T-NH_3), total Kjeldahl nitrogen (TKN), and total organic carbon (TOC). Particulate P (PP) was computed as the difference between TP and TDP. Stream loads were calculated on both daily and event bases.

Farm Pollutant Sources and BMPS

At the beginning of the study a number of pollution sources were identified on the R farm that could be corrected or reduced through application of BMPs. These sources included manure spreading on snow-covered and frozen ground, concentrated manure spreading on a few fields, barnyard runoff, livestock access to the stream, unsuitable grazing areas, streambank erosion, silage leachate and milkhouse wastewater reaching the stream, and farm road erosion. Prior to any changes being made on the farm, both farm and control sites were monitored for two years (June 1993–May 1995) in order to establish the pre-treatment (pre-BMP) relationship between the farm and control watersheds in terms of hydrologic response and pollutant loading.

Following these two years during which farm operation remained essentially constant, an extensive program of BMPs was implemented on the farm watershed. A site-specific WFP developed by a WAP planning team and the collaborating farm family identified target areas for control of nutrients, sediments, and pathogens. The resulting array of BMPs included physical changes to farm infrastructure as well as organizational changes to farm management, and affected both pollutant source areas and transport processes across much of the farm landscape. Stream monitoring was suspended during the implementation period (June 1995 through October 1996), and then resumed for the post-treatment (post-BMP) period (November 1996 through October 2006).

Before implementation of BMPs, manure produced by the dairy herd was spread daily. Limited access to hillside slopes during the winter months often required spreading or stockpiling manure on near-stream fields. After construction of a 2300-m^3 manure storage lagoon spreading was eliminated, with the exception of about one load per week of heifer manure during the high-runoff winter and early spring months. Manure application on hydrologically active areas and on fields with high STP was reduced through adoption of a farm nutrient management plan containing field-specific timing and rate recommendations, and by improvement of certain farm roads, which allowed access to upper slopes that had received little manure in the past. The suitability of several fields for manure spreading was enhanced by construction of upslope diversion ditches to prevent runoff from entering fields, and installation of tile drains to accelerate drying of fields and reduce the frequency of field runoff production. (Although tile drains were considered to be a BMP at the beginning of the study, the WAP no longer promotes this practice due to the associated


Streambank Erosion

risk of nutrient loss to preferential flow pathways.)

Erosion potential was reduced by decreasing the amount of time corn was in rotation, by implementing contour strip-cropping on one field, and by improving drainage to reduce field runoff production. The WFP also led to improved pasture management and reduced potential for overgrazing and associated soil erosion through the establishment of a rotational grazing system. A spring development project supplied water for grazing cows, and encouraged their movement away from streamside areas. Cows were fenced out of several hydrologically active areas where they had previously grazed. Near the barn, several stream crossings and roadways were fenced and improved to keep the cows out of the stream. Although the main barnyard had been paved with concrete before the study period, barnyard water management was improved, and a grassed filter area was constructed to intercept barnyard runoff that had previously flowed directly into the stream. A portion of the main stream, which had run quite close to the barnyard, was diverted to a new channel a short distance to the west. Bagged silage was relocated away from the streamside, and milkhouse waste, which had previously drained directly to the stream, was routed to the manure storage lagoon. Finally, careful and detailed records of farm activities, such as location and amount of manure spreading, were kept in order to relate changes in water quality to changes in farm practices. A complete list of Phase 1 BMPs is given in Table 1.

Several additional BMPs were installed on the farm after the original implementation took place. A streamside shed that the farmer had begun using as a shelter area for dry cows and heifers in the late 1990s was identified as a high contributing source area. Located just upstream of the monitoring station, a cattle path led from the shed down a steep, eroded slope through the stream and up the opposite bank to a pasture area. In summer 2001 this area was revegetated and fenced to protect the stream bank. A cattle lane and crossing was also built to provide safe access across the stream to pasture.

Beginning in January 2001, the farm was selected for participation in a pilot program of precision feeding aimed at reducing phosphorus importation on dairy farms from purchased feed. Feeds and homegrown forages were analyzed for their protein, carbohydrate and mineral contents and the nutritional needs of the herd were determined using the cuNMPS (Cornell University Nutrient Management Planning System) software. Diets were adjusted and as a result, P imported onto the farm in purchased feed was reduced by 30 percent. This directly translated into a 30 percent reduction in P excreted by the cows (Cerosaletti et al. 2004).

Table 1. Best management practices (BMPs) implemented on the farm between June 1995 and October 1996.

Near-barn BMPs

- Installation of manure storage lagoon
- Barnyard improvements including water management
- Filter area established for barnyard runoff
- Stream corridor relocated away from barnyard
- Grazing cows excluded from stream/swale areas
- Milkhouse washwater diverted to manure storage
- Relocation of silage storage bag away from stream
- Improvement of stream crossings and roadways

Watershed-scale BMPs

- Access roads constructed to allow manure spreading on upper slopes
- Distributed manure according to nutrient management plan
- Fencing improvements to support rotational grazing
- Spring development to supply drinking water away from the stream
- Diversion ditches to improve field drainage
- Subsurface drainage to reduce field saturation and runoff

Installation of another spring development and remote watering system was performed in summer 2002 on the farm, resulting in less cattle traffic in and around the stream.

As it was expected that the new management practices described above would lead to additional improvements in water quality on the farm, the post-implementation period was split into Phase 1 (initial round of BMPs), and Phase 2 (BMPs since April 2001). This start date for Phase 2 represents the time manure produced under the precision feeding program and stored in the manure lagoon


Cattle path and stream crossing

would first be applied to the farm's fields. Analyses of data then focus on changes in water quality between the pre-BMP period and Phase 1, and changes between Phase 1 and Phase 2.

Analysis Of Data

Changes in event loading between the pre- and post-BMP periods were determined through use of multivariate regression and analysis of covariance (ANCOVA). A complete discussion of the development of the statistical model may be found Bishop et al. (2005).

Nonevent, baseflow periods were analyzed by comparing stream water concentrations from the pre-BMP to those of the Phase 1 and Phase 2 post-BMP periods. Mean sample concentrations for each of the three study periods were calculated and compared for significant differences using a one-sided t test. As with the event data, the natural log transformation was employed to improve the normality of the baseflow concentration data prior to testing for significance. Differences in the geometric mean baseflow concentrations and their respective 95 percent confidence intervals were used to estimate full year reductions in baseflow concentrations.

Event Load Reductions

i) BMP Treatment Effects: Comparing Pre-BMP to Phase 1 Post-BMP

The magnitude of Phase 1 post-BMP event load reductions, as well as 95 percent confidence intervals were computed for all analytes. When analyzed on a full year basis, without separation into seasons, all analytes, with the exception of NO_X and TKN, showed significant reductions in event loads after implementation of Phase 1 BMPs. Reductions ranged from 22 percent in TOC loads to 41 percent in TDP loads. NO_X loads actually increased when compared to pre-BMP levels and TKN displayed a non-significant increase of 1 percent. Seasonally, most analytes showed significant reductions in winter and summer. No significant changes in fall event loads were noted for any analytes. Spring event loads were similarly unaffected with the exception of a 38 percent reduction in TDP.

ii) BMP Treatment Effects: Comparing Phase 1 to Phase 2 Post-BMP

Event loads were analyzed to determine if additional reductions occurred after the second round of practices were installed on the farm. In the full year PP, TDP and T-NH_3 decreased 23 percent, 12 percent and 43 percent, respectively, relative to loading in Phase 1. NO_X decreased by about the same percentage (26 percent) it increased between the pre-BMP period and Phase 1 (20 percent), and, thus, was essentially unchanged from the beginning of the study. Seasonally, reductions in summer loads were noted for PP, TDP, TSS, T-NH_3 and NO_X. PP and TSS showed significant decreases in fall loads. NO_X and TKN both increased significantly in winter when compared to Phase 1. It is unclear why winter event loads of these analytes would increase in Phase 2. It remains to be determined if some aspect of farm management changed in Phase 2 that would contribute to winter increases of nitrogen.

iii) Effects of Seasons on Event Loading and BMP Performance

Phase 1 BMPs significantly reduced full year event loads of PP, TDP, TSS, T-NH_3 and TOC by amounts ranging from 20–41 percent. When examined seasonally, TDP load reductions were evident in all seasons except the fall. PP, T-NH_3, and TOC load reductions were significant only in the winter and summer seasons. TSS displayed a significant seasonal reduction only in the winter. NO_X loads significantly increased in the Phase 1 post-BMP period in the full year and winter season. TKN, which is a measure of the reduced forms of nitrogen in surface water, principally ammonium and amino forms of organic nitrogen, appeared unchanged in the full year and in all seasons.

Additional reductions were observed after Phase 2 BMPs were installed. TDP and T-NH_3 displayed significant full year reductions in event loads of 12 percent and 43 percent, respectively. NO_X decreased by 26 percent in the full year, but this nearly offset the increase of 20 percent observed between the pre-BMP and Phase 1 periods. Significant reductions were noted for the first time in the fall season (PP – 41 percent and TSS – 47 percent). Further summer reductions were observed for PP, TDP, TSS, and T-NH_3. NO_X was reduced by 53 percent in the summer, although it increased by 33 percent in the winter. TKN increased in winter as well. Spring saw reductions in PP of 46 percent, in TSS of 49 percent, and in NO_X of 45 percent. The effects of seasonal differences in BMP performance and hydrology on results are discussed below.

- **Summer (15 June–30 September)**

Loadings in the summer season appear most affected by BMPs implemented in Phase 1 and Phase 2. After Phase 1, TDP and PP summer event loads were reduced by 51 percent and 44 percent, respectively, and after Phase 2, by 30 percent and 38 percent, respectively. T-NH_3 summer event loads exhibited >50 percent reductions after each phase of BMPs. Significant reductions after Phase 2 were also observed in TSS and NO_X. In the dry summertime, upper watershed slopes did not usually saturate, and nutrient and sediment loads were produced mainly from near-stream, impermeable, and slope-break sources. BMPs that would operate mostly in these areas included Phase 1 and 2 exclusion of cows from the stream corridor, relocation of the silage storage bag away from the stream bank, implementation of rotational grazing, improved pasture management, Phase 2 remediation of the dry cow loafing area and stream crossing improvement, and somewhat reduced manure spreading during summer months.

- **Fall (1 October–14 December)**

Significant event load reductions after Phase 1 were not observed during the fall season for any analytes. Increased fall manure spreading in the post-BMP period when the farmer emptied the manure storage lagoon in preparation for the winter may have offset any phosphorus and nitrogen reductions attributable to other BMPs implemented on the farm. At this time of year there is little crop growth to utilize nutrients added to the soil, thus manure applied to the land would be expected to be available for loss during runoff events. The fall reductions observed after Phase 2 in PP (41 percent) and TSS (47 percent) may be somewhat attributable to the protection and revegetation of the dry cow loafing area near the stream, practices that would be expected to reduce losses of particulate fractions.

- **Winter (15 December–13 April)**

Reductions in winter phosphorus and organic carbon event loads in the Phase 1 post-BMP period were most likely largely attributable to storage of manure and minimal spreading from mid-December to mid-April. Sediment reductions may be linked to decreased farm vehicle traffic and farm road disturbance associated with minimal manure spreading. Decreases in winter ammonia loads appeared to be largely offset by increases in nitrate loading, and suggests a transformation of nitrogen forms through nitrification. The reduction in ammonia loading is perhaps due to the lack of fresh manure being applied daily to snow and frozen ground and subjected to runoff processes. In this case, surface-applied manure in cold weather would tend to retain nitrogen as ammonia, instead of being converted to nitrate, which occurs in the soil under warmer conditions. Ammonia-nitrogen contained in the large amounts of manure applied in the fall, when the storage was emptied, was likely converted to nitrate in the soil. This could have still been available for loss during winter runoff events, nitrogen being more mobile in the soil than phosphorus. In addition, a portion of the ammonia was no doubt lost from the manure through volatilization during agitation of the storage, and subsequent spreading on fields. Thus, unlike phosphorus, winter loads of nitrogen appear unaffected, and in one case increased (NO_X in Phase 2 by 33 percent), by the BMPs installed in Phase 1 and Phase 2.

- **Spring (14 April–14 June)**

Spring TDP event loads were reduced by 38 Percent in the Phase 1 post-BMP period. Manure was heavily surface-applied in the spring months to empty the storage after winter, with some being incorporated into the soil during tillage. Losses from manure-spread fields and increased sediment availability resulting from spring tillage and increased farm traffic would potentially mask clear-cut reductions in sediment and nutrient loadings. It is encouraging that TDP, the most important nutrient contributing to eutrophication, was significantly reduced in springtime as a result of the Phase 1 BMPs. This may be a result of barnyard water

management practices, improved field drainage, and manure spreading schedules that more evenly distributed manure over the farm. All of these practices may be expected to reduce event loadings of dissolved nutrients, but not necessarily the particulate fractions. Phase 2 BMPs had a significant effect on spring event loadings of NO_X, which was reduced by 45 percent, and TSS, which was reduced by 49 percent. However, as NO_X exhibited a non-significant increase of 40 percent after Phase 1 BMPs, the overall change in nitrate event loading from the pre-BMP period may be considered negligible.

Baseflow (Non-Event) Reductions

As with the analysis of event periods, it is difficult to determine effects of BMPs on water quality based on annual loads due to potential masking by interannual environmental variability. While there appears to be differences in annual farm loads between the pre- and post-BMP periods, it makes more sense to examine baseflow sample concentrations for any significant changes during the study period.

In the pre-BMP period, there were 125 baseflow samples collected; in Phase 1, there were 178 samples; in Phase 2 there were 255 for phosphorus forms and sediment, and 141 for nitrogen forms and TOC. Like the event analysis, concentrations were compared for both the full year and seasonally.

When comparing Phase 1 to the pre-BMP period, baseflow concentrations of all three forms of P and T-NH_3 were significantly reduced in the full year and all seasons; TSS was significantly reduced in the full year and spring season; NOX was significantly reduced in the summer season, and significantly *increased* in the winter and spring seasons; and TKN was significantly increased in the full year, fall and spring seasons. Changes in mean baseflow concentrations between Phase 1 and Phase 2 of the post-BMP period included significant reductions in full-year TSS, summer TDP, SRP, TSS and NO_X, and fall PP and TSS. Significant increases in full-year TKN and TOC, summer T-NH_3 and TKN, winter SRP and TKN, and spring T-NH_3 were also observed between Phase 1 and Phase 2 baseflow concentrations.

Between the pre-BMP period and Phase 1, substantial reductions of 51 percent to 71 percent were observed in full-year concentrations of ammonia and the three forms of phosphorus. TSS was reduced to a lesser extent after Phase 1, and TKN actually increased by 15 percent. When comparing Phase 1 concentrations to Phase 2, TSS and NO_X decreased by 22 percent and 35 percent, respectively, TKN increased again, and TOC exhibited a 16 percent increase. The overall change in NO_X would appear slight, as this analyte increased after Phase 1, though not significantly ($p > 0.05$), then decreased after Phase 2. Changes in TOC are negligible, as the pre-BMP and Phase 2 concentrations are the same. It is unclear why TKN would increase throughout the post-BMP period, when ammonia decreased significantly, as TKN is a measure of ammonia and organic forms of nitrogen. It must be presumed that for some reason certain organic forms increased enough to more than offset observed decreases in ammonia.

The significant reductions observed in post-BMP baseflow concentrations of phosphorus, sediment and ammonia would be expected to result in proportionally reduced baseflow loads. Pollutants in baseflow are typically derived from point discharges, leaching from field soils in subsurface flow, release from disturbed stream banks and resuspended bed sediments, and direct activity by cattle in the stream. For dissolved analytes, much of the reduction may be attributed to the elimination of the daily milkhouse waste discharge to the stream as well as decreased manure deposition near-stream. The reductions in particulate forms are likely due to the exclusion of livestock from the stream and associated reductions in direct manure deposition, stream bank erosion, and sediment resuspension and transport.

Total Farm Reductions

The overall effect of BMPs on the farm may be estimated by adding the event reductions to the baseflow reductions. Table 2 shows the fraction of annual loads delivered during events and baseflow periods, significant reductions ($p < 0.05$) after Phase 1 and Phase 2 BMPs for both event and baseflow loads, and the combined effect of these reductions on the total annual loading. Loads of ammonia and dissolved phosphorus exhibited the greatest reductions as a result of the BMPs implemented under Whole Farm Planning. Farm losses reduced by 50 percent or more can be considered to be quite substantial and would be expected to have positive effects on receiving water bodies if achievable on other farms in the watershed. Particulate P and sediment losses were reduced by 49 percent and 45 percent, respectively. While not quite as large as the decreases in ammonia and TDP, these reductions

may contribute to lessening eutrophication, turbidity and sedimentation in receiving water bodes. Reductions in NO_X of 23 percent and TOC of 5 percent were smaller, and TKN increased by 17 percent. These differences would be expected to have little effect on receiving waters.

Certain changes in farm practices occurring in the post-BMP period may have counteracted the effect of BMPs to some degree. These included a gradual increase in herd size of about 30 percent and intensified use by cows of the streamside loafing yard that created a concentrated nutrient-loading source area not far upstream of the monitoring station. In addition, none of the Phase 1 BMPs altered either the amount of P imported onto the farm as feed or fertilizer or the amount exported as products. Therefore, as the mass balance of P on the farm did not change appreciably during the first four years of the post-BMP period, presumably any reductions observed in stream losses of P resulted from more of it being retained on the farm. This outcome has the potential of accelerating net accumulation of P in the farm soils and eventually raising STP levels to the point of saturation of soil P binding capacity. Studies indicate this saturation point represents a threshold of STP above which TDP concentrations in runoff can increase sharply (e.g., Beauchemin and Simard, 1999; McDowell and Sharpley, 2001), an effect that, in the absence of measures to reduce P inputs, would be expected to lead to increased loading of dissolved phosphorus from the farm in the future.

Beginning in 2001, the second phase of BMPs

Table 2. Overall effects of BMPs on annual farm loads.

	Avg % of Annual Load*	% Reduction Phase 1	Phase 2	Total
PP				
Event	90	34	23	44
Baseflow	10	51	0	5
Total				**49**
TDP				
Event	68	41	12	33
Baseflow	32	60	0	19
Total				**52**
TSS				
Event	93	28	25	43
Baseflow	7	16	22	2
Total				**45**
NOX				
Event	51	-20§	26	6
Baseflow	49	0	35	17
Total				**23**
T-NH3				
Event	66	33	43	41
Baseflow	34	68	0	23
Total				**64**
TKN				
Event	65	0	0	0
Baseflow	35	-15	-28	-17
Total				**-17**
TOC				
Event	55	22	0	12
Baseflow	45	0	-16	-7
Total				**5**

* Average partitioning of loads in the entire 10-year post-BMP period between events and baseflow.
§ Negative value indicates a percent increase in loading.

implemented on the farm not only corrected the concentrated nutrient source area, but addressed an identified P imbalance resulting from more P being imported, largely as purchased feeds, than was exported in the form of milk, meat, or crops. The farm watershed P mass balance was improved with institution of a precision feeding program, which lowered imports of dietary P by an average of 25 percent and reduced excretion of P in manure by 33 percent (Cerosaletti et al. 2004). Reductions of this magnitude in the amount of manurial P applied to the farm soils should slow the rate of soil P accumulation and continue to reduce losses of P in runoff waters. The observed Phase 2 reductions in TDP and PP may be somewhat attributable to the institution of precision feeding, although seasonally, reductions due to this practice would be expected to be associated more with runoff losses during fall and spring when most of the manure is now spread, not in summer when the greatest reductions in both TDP and PP actually occurred.

Our study was somewhat unique in its characterization of the changes in water quality from a single farm, so findings from other BMP effectiveness studies that monitored larger watersheds may not be directly comparable. One study of interest (Brannan et al., 2000), however, demonstrated reductions of 35 percent in PP loading and 4 percent in TDP loading in a 10-year evaluation of animal waste practices (including manure storage, spreading schedules, and stream fencing) implemented in a 331-ha Virginia watershed containing two dairy farms. In the same study, the authors reported PP load reductions of 70 percent, but TDP load increases of 117 percent in a nearby 462-ha agricultural watershed composed mostly of cropland that received BMPs including nutrient management plans based on nitrogen needs, and field erosion control practices. Conversion of organic P to inorganic P in the manure storage and application of manure at rates based on nitrogen needs of crops, which typically result in over-fertilization of P, were given as factors that could explain the ineffectiveness of the program in reducing TDP loads. The BMPs evaluated in our study produced overall PP reductions comparable with those Brannan et al. (2000) reported for the first watershed and about half of that observed in the second watershed, but were much more successful in reducing TDP loading. Findings of Brannan et al. (2000) may constitute evidence of the eventual P saturation of soil and subsequent release of dissolved P in runoff that is postulated to occur when conservation and nutrient management practices are implemented in the absence of efforts to improve whole-farm mass balance of P.

The effects of the BMPs implemented under the Whole Farm Planning program on nitrogen losses were mixed. The two main components of N in manure are organic N and ammonia (Collins et al, 1995). In fresh manure, the inorganic portion is commonly in the form of ammonium N. Storage of manure, especially in slurry form, generally results in conversion of organic N to ammonium through ammonification (Brannan et al. 2000). Loss to the atmosphere can occur through volatilization of ammonia N from either the storage or from surface-applied manure. Ammonia N is converted to nitrate by soil bacteria when manure is incorporated into the soil. If application is in excess of crop needs, nitrate is quickly lost in surface and subsurface runoff. While manure storage has the benefit of producing more plant-available N by transforming organic N to inorganic forms, if crop needs are small or absent at time of application as they are in the fall season when the storage is emptied, there is more potential for loss to the environment. This may explain the apparent increases seen in NO_X loading after Phase 1. Ammonia loadings decreased, presumably through loss to the atmosphere and conversion to nitrate, and nitrates increased due to excess amounts in relation to plant needs. Brannan et al. (2000) reported results similar to ours in that reductions in ammonia concentrations of 30 - 70 percent were measured in their three study watersheds and nitrate loading showed the smallest reductions due to BMPs.

Phosphorus: Critical Nutrient and Major Pollutant

P is a critical nutrient for plants and for animals. It influences the growth and yield of crops and the ability of the plants to withstand stresses from climate and disease-causing organisms (Bundy et al, 2005). Deficiencies of P during the period of early growth of plants are especially critical since they cannot be remedied by an adequate availability of P for later growth. As Bundy et al (2005) describe, insufficient P in the soil can dramatically reduce crop yields. Such deficiencies can readily occur. A survey conducted by the Potash and Phosphate Institute, cited by Bundy et al, found that nearly 40 percent of the soils tested in New York had medium or lower levels of P. At such levels, crops would be expected to respond to additions of P to the soils. Conversely, a survey by Kellog, et al. (2000), found that about half the counties of New York have soils to which excess amounts of manure have been applied, leading to elevated P levels in the soil.

Levels of soil P conventionally have been determined from the viewpoint of the agronomic needs of the crop. Unfortunately, the determination of levels of P in the soil remains of limited value for estimating the associated losses of P to ground or surface waters from that soil. As Maguire, et al. (2005) note, P is conveyed to watercourses by the erosion of soil particles and by dissolved P in surface runoff. Until recently, pathways that transfer P from soils to freshwaters were the most studied. However, it is now realized that P can also leach down through the soil (Maguire and Sims, 2002).

Maguire et al. (2005) provide a good review of soil testing methods for P and state that the basic principles were set out by Cline in 1944. A key difficulty in determining representative soil P levels is the high degree of variability that can be experienced in the field even within a short distance. Such variability merits the application of a formal sampling strategy—random, stratified random, or systematic sampling design. Whether or not the field is customarily ploughed, or if the objective is to track the potential leaching of P, will determine the desirability of sampling in the soil column. With respect to ascertaining the potential risk to water quality, a key aspect of the sampling is to determine the "environmental threshold". The environmental threshold method is quick and much less costly than an alternative approach which develops a so-called P index (Heathwaite et al. 2005). The P index has been used mainly for planning agronomic nutrient management. Original indexes were calculated by assuming the site factors comprising the index were additive. In reality, the factors could interact. A remedy for this defect was sought by distinguishing the various factors, P applications and soil P and soil erosion, surface runoff, and subsurface runoff. Further refinements to the index included separating particulate and soluble losses of P to take into account land management on these different forms of P (Bryant et al. 2002). However, as Heathwaite et al. (2005) acknowledge, "P indices are not designed to quantify P loss" to waterbodies.

Setting a threshold for soil test P is a simple and relatively inexpensive first means to assess the loss of P from a field (Maguire et al. 2005). This threshold can initially be set as the optimum level of soil test P for agronomic purposes. However, this level can be easily reached in soils on farms producing animal manures. Limiting the applications of manures on such farms may cause hardship for the farmer. Hence the threshold may be set above the agronomic level but below an estimated level at which undesirable losses of P may occur (Weld et al, 2001). However, just what the threshold level should be is likely to remain uncertain in practice. Although it has been determined that loses of P are

related to the level of soil test P, other factors may still influence losses such as the addition of manures or fertilizers prior to runoff events (Mullins et al. 2005).

A larger challenge is the need to understand the transport of P especially at scales larger than the plot or field. As Gburek et al. (2005) state, there is a need to link the P observed at the plot scale, to that observed at the catchment outlet. These linkages are usually represented by a mathematical model at an appropriate scale. As Heathcote et al. (2005) observe, all models are incorrect but can be useful. However, that usefulness can only be affirmed by the validation of real observations obtained through monitoring of the waterbodies affected. Heathcote et al. further assert that "A major limitation to the representation, calibration, validation of nonpoint source models is the lack of stream flow monitoring of P concentrations and loads for a wide range (geographically and land use dominated) of watersheds."

Determination of the Significance of Groundwater P

As noted earlier, expectations to the contrary, leaching and subsurface flows of P is considered a potential contributor to the loading of P in streams. To verify or refute this hypothesis, a groundwater study was conducted, including systematic sampling of groundwater on the R Farm.

The project samples shallow ground water near where it discharges to streams to check if there is any phosphorus present and to examine relationships between phosphorus concentrations and land use. It was deemed important to evaluate potential nonfarm as well as farm sources of P in groundwater. Hence, land use categories of interest are urban with sanitary sewers (in villages), forest, active farm, former farm, and rural residential with septic systems. Table 3 counts cumulative samples collected through December.

As of late December, laboratories had provided phosphorus results through July and all other results through June or July. All laboratory results in hand and essential field measurement data from all sampling periods have been transferred from laboratory submissions and field logs into a consistent database that is being maintained routinely.

Work completed included a sampling period at R Farm in October and a sampling period at all other ongoing sites.

Sites under consideration are italicized in the site inventory in table 4. The project's funds and calendar are unlikely to support adding any additional sites, thus the current list will probably be the final list and some of the cells in Table 4 will be unrepresented. There will be no additional sampling at the Shaw Road site.

With the October samples, the R farm site has a complete annual cycle of coverage. Sampling will be reduced to every other sampling period. The H Farm, Walton, Hamden, and Trout Creek sites will remain on a 4-7 week intervals due to their shorter periods of record. Extending from the initial R Farm and Shaw Road to the current roster of seven sites, each has multiple sample collection points. Each successive site's detailed sampling design reflects experience from all prior sites.

P losses from Farm and Forest Catchments

Scott (1998), characterized P loading from a 16.6 ha forested catchment area of the R Farm using automated sampling equipment. In a diversion ditch installed below the forested area, Scott measured an average soluble reactive phosphorus concentration of 0.0127 mg L^{-1} in runoff. This concentration was higher than expected. Scott suggested the hypothesis that residual P previously applied to the area when it was managed as pasture, accounted for the unexpectedly elevated levels. It is also possible that some of the P was released from the edge of the field where Scott collected his samples, or from soils of the diversion ditch itself which had been constructed only a few months prior to sampling (Hively, 2004).

Hively and an associate, Nagle (Hively, 2004), also sampled the same forested catchment. They found that humus and mineral soil samples differed according to their location, soil type, forest cover, and the presence of burrowing worms. They failed to detect noticeable differences in soil or humus P between newer and older growth areas as discussed in Scott et al (2001a). An experimental application of simulated rainfall on a site within the same area of forest on the farm produced an average dissolved phosphorus concentration in resultant runoff of 0.007 mg L^{-1}. This concentration is close to that obtained by Scott. These results are comparable to those obtained in the forested area in the Shaw Road watershed. An average annual dissolved phosphorus concentration of 0.013 mg L^{-1} was observed during flow-weighted event sampling and

Table 3. Sample counts.

Period	Site	Wells & springs	Tile Drains	Streams & ditches	Field Blanks	Lab Blanks	Total
Nov 2004	*R Farm*	5	0	2	1		8
	Shaw Road	1	0	1			2
Dec 2004	*R Farm*	6	0	2	1		9
Feb 2005	*R Farm*	6	1	2	1	*Walton*	11
Mar 2005	*R Farm*	6	1	2	1	*Ithaca*	11
	Hamden	1	0	1	1		3
Apr 2005	R Farm	4	1	2	1	Food Club distilled water	9
	Hamden	1	0	1	1		3
May 2005	R Farm	5	1	2	1	Ithaca (2, one unfiltered)	11
	Hamden	3	0	1	1		5
July 2005	R Farm	5	1	2	1	Ithaca, Walton, Food Club distilled	12
	Hamden	3		1	1		5
	Shaw Road	1		1			2
Aug 2005	R Farm	all wells dry	tile drain dry	1 (+ RFS2 dry)		Ithaca (filtered)	2
	Hamden	2 (+1 dry)		dry			2
	H Farm	3		1	1		5
Sep 2005	R Farm	1 (+4 dry)	tile drain dry	1 (+ RFS2 dry)	0	Ithaca (filtered & unfilt)	4
	Hamden	2 (+1 dry)		dry			2
	H Farm	3		1			4
	Walton	2		1	1		4
	Trout Creek	2		1			3
Oct 2005	R Farm	4	1	2	1	0	8
Nov 2005	Hamden	3		1		Ithaca (filtered)	5
	H Farm	2		2			4
	Walton	2		1	1		4
	Trout Creek	2		1			3
Cumulative		**75**	**6**	**33**	**15**	**12**	**141**

Note: Work at R Farm and Shaw Road sites was partially funded under a parallel project through March, and became part of this project on April 1, 2005. The Hamden, H Farm, Trout Creek, and Walton sites are entirely under this project. Italicized samples were also reported in the R Farm parallel project's quarterly reports and final report.

Table 4: Ongoing and candidate well sites.

Land Cover	Upland	Lowland
Mature forest	Ongoing R Farm subwatershed and . Shaw Road subwatershed (NYS WRI) Spring at H Farm (NYS WRI)	*Need candidate. Beerston?*
Former farmland	*Need candidate*	H Farm. Former livestock farm along East Brook Road, Town of Walton
Active farm	Ongoing R Farm (NYS WRI)	Ongoing L Farm (Cornell BEE) *Addition: Farm in broad West Branch Delaware River valley?*
Built up areas served by septic systems	Hamlet of Trout Creek (NYS WRI)	Hamden hamlet in West Branch Delaware River valley
Urban area with sanitary sewers (to isolate non-wastewater aspects)	(None present in basin)	Streamside linear park in V. Walton (NYS WRI) *Manufacturing plant in Hobart?*

0.007 mg L^{-1} during sampling of baseflow (Personal communication, P.L. Bishop, 2003).

Scott also sampled a 2.24 ha intensively grazed pasture on the R Farm. Using automated sampling methods, he found that 37.1 percent of the annual soluble reactive P load was transported in a tile drain, with the remaining 62.9 percent delivered as surface runoff (Scott et al., 1998). He estimated that the mean annual concentration of TDP in the tile drain was 0.043 mg L^{-1}, and the corresponding concentration in surface runoff was 0.504 mg L^{-1}. An experimental application of rainfall on the same pasture just after it had been grazed, produced an average dissolved P concentration in runoff of 0.64 mg L^{-1}. A simulation of rainfall on a nearby area of pasture under regrowth, produced an average dissolved P concentration of 0.37 mg L^{-1} in the runoff. As a result of his observations, Scott (1998) suggested installing upslope tile drains and drainage ditches rather than in-field tile lines. Such a practice may be as effective in preventing soil saturation, but avoids the short-circuiting produced by macropore flow directly into drainage tiles.

Experimental Determination of P Runoff Across the R Farm using Rainfall Simulation

Hively (2004) measured the transport of P in runoff from nine selected sites across the R. Farm. For this purpose he used the National Phosphorus Project rainfall simulator. The sites were chosen as appearing to be likely to produce runoff. The nine sites are described in Table 5. Rainfall was simulated on each plot. When runoff first occurred the time from the beginning of the rainfall was recorded. Flow-weighted samples were obtained from each plot. After the first day of sampling, the plots were allowed to return to field capacity. On the second day the sampling was repeated.

Results

In the first simulations, the time that elapsed from the onset of rainfall varied across the plots by an order of magnitude. However, on the second day when the soils in the plots were all approximately in a similar condition of field capacity, the times taken for runoff to occur were much more uniform. As Hively notes, this finding supports the obvious hypothesis that runoff is induced by saturated soils. Soils with higher moisture levels will produce runoff more quickly than drier soils.

As expected, the concentrations of total suspended solids, total P, particulate P, and total dissolved P, varied widely. The highest concentrations, by an order of magnitude, were those measured form the barnyard. For example, the total dissolved P in the sampled runoff from the barnyard was 11.6 mg L^{-1}. Pathway plot was the second highest. This plot had by far the highest loading of total suspended

Table 5. Nine simulted rainfall applicator sites sampled summer 2001, representing potential different P loading sources within the dairy farm watershed (Hively 2004).

Site Code	Site Name	Site Description	Cover[a](%)	Cover type	Slope (%)
HYD	Heifer barnyard	Heavy manure deposits	10	Some debris from fed hay	11
PTH	Cow path	Leading up from heifer stream crossing	50	Sparse close-cropped grass	15
GRN	Grazing North	Intensive rotational grazing not yet grazed in 2001	100	Improved pasture grasses, lush growth	10
GRS	Grazing South	Intensive rotational grazing recently grazed	100	Improved pasture grasses, lush regrowth	10
HAY	Hayfield	Recently cut hayfield (lst st cut, little regrowth	80	Cut stems of hay grasses	10
PAS	Pasture	Swale area in extensively grazed hillside pasture	100	Unimproved pasture grasses and weeds	13
SMZ	Spring in maize	Hillside spring area in plowed cornfield	15	Tilled soil, some stones	13
SHP	Spring in heifer pasture	Hillside spring area in heifer pasture	85	Nutsedge and swamp grasses	14
FOR	Forest	Hardwood forest, flowpath at base of slope	75	Forest floor herbs and fallen leaves	7

solids in its runoff. The lowest concentrations, were those obtained in the runoff from the forested plot (Table 6). For the sites to which manure had not been freshly applied, Morgan's soil test phosphorus was found to predict the concentrations of TDP quite well [TDP (mg L^{-1}) = 0.0056 + 0.0180 * STP (mg kg^{-1}): adjusted R^2 = 84 percent]. This relationship failed on sites that were manured. As would be expected, the concentrations of suspended solids were higher on those sites with little groundcover compared to those sites with vegetation.

With respect to loads, only four of the nine sites produced significant loads: the barnyard, cow pathway, maize field and a grazing pasture (Table 7).

Table 6. Average concentrations (mg L^{-1}) of total suspended solids (TSS), total phosphorus (TP), particulate phosphorus (PP), and total dissolved phosphorus (TDP) in composite (0 to 30 min) and equilibrium flow (at 30 min) runoff samples (Hively 2004).

	30-minute Flow Composite					Equilibrium Flow (t=30 min)				
Site[1]	TSS	TP	PP	TDP	SRP	TSS	TP	PP	TDP	SRP
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
HYD	375[a]	13.16[a]	1.57[a]	11.60[a]	8.83[a]	280	13.74	1.87	11.87	8.92
PTH	540[a]	0.99[b]	0.81[a,b]	0.18[c]	0.23[c]	450	0.73	0.57	0.16	0.23
GRN	23[b]	0.58[c,d]	0.21[c]	0.37[c,d]	0.33[b,c]	21	0.54	0.19	0.36	0.32
GRS	33[b]	0.95[b]	0.31[c]	0.64[b]	0.46[b,c]	25	0.83	0.22	0.61	0.44
HAY	35[b]	0.68[b,c]	0.26[c]	0.43[b,c]	0.37[b,c]	16	0.67	0.19	0.48	0.40
PAS	16[b]	0.25[c,d]	0.13[c]	0.12[e]	0.09[c,d]	22	0.22	0.10	0.11	0.10
SMZ	615[a]	0.62[b,c,d]	0.51[b,c]	0.11[e]	0.14[c,d]	560	0.59	0.51	0.08	0.15
SHP	137[b]	0.30[c,d]	0.28[c]	0.020[e]	0.01[d]	100	0.24	0.22	0.020	0.01
FOR	72[b]	0.19[d]	0.18[c]	0.007[e]	0.01[d]	54	0.08	0.07	0.007	0.01

[a,b]Letters indicate significantly different treatment means, as determined by Tukeys mutliple comparison, a = 5%
[1]Refer to Table 5 for site code descriptions.

This finding would appear to identify which areas on the farm are likely sources of P and solids. Further as Hively observes, few of the soils measured on the farm exceeded 25 mg kg^{-1} Morgan's STP. He concludes "that any observed R Farm watershed event above 0.5 mg L^{-1} can be attributed to runoff from freshly manured hydrologic source areas, including field spread manure, recently grazed fields, barnyards and roadways, or areas where cows have direct access to the stream."

Table 7. Loads of total suspended solids (TSS), total phosphorus (TP), particulate phosphorus (PP) delivered in runoff from the nine rainfall simulation sites during a 25-minute simulated rainfall event on the first day of rainfall application, under dry summer conditions (Hively 2004).

Site Code	TSS mg/plot[b]	TP mg/plot	TDP mg/plot	PP mg/plot
HYD	3730	110	112	0
PTH	12675	29	7	22
GRN	0	0	0	0
GRS	0	0	0	0
HAY	0	0	0	0
PAS	0	0	0	0
SMZ	3282	3	0	2
SHP	4550	10	1	9
FOR	0	0	0	0

Hively confirmed the above results with extended field studies of the farm over a two year period by collecting 78 soil samples. The samples were collected to represent the varied conditions across the farm (Fig 5). He observed elevated levels of TDP adjacent to the barnyard, on roadways, and in tile drainage. Concentrations were generally low in the upper parts of the farm watershed area. This finding corresponded to measurements of STP in stream sediments which were low in the upper reaches but were found to be increasing in the lower stream reach and downstream of the barnyard area. Through his samples, Hively particularly sought to determine the relative magnitudes of P loadings from ten "source" areas, including roadways, barnyards, barnyard filter areas, stream crossings, swamp and forested areas. He found considerable variation as he did with his rainfall simulations. He confirmed the major conclusion that impermeable areas, such as roadways, are a most critical source of P in the stream especially in summertime. It is therefore essential to prevent drainage water flowing across such areas. Diversion ditches and roadway berms would well serve this key objective if the stream is to adequately protected. In the upstream areas, Hively suggested there was considerable buffering capacity that would protect the stream. However, further downstream and below the barnyard, enrichment of soils adjacent to the stream would cause higher levels of P in the stream. He suggested that it would seem advisable to periodically remove sediments and soils which had accumulated very high levels of P in them.

Precision Feeding to Reduce Excretion of P by Dairy Cows

(Cerosaletti et al. 2004)

R Farm herd size at the time of the study was 78 adult cows and is tie-stall-barn, component-fed, with all feeds, including purchased concentrates, being fed separately to each individual cow. Lactating cow diets, forage quality, cow performance, and manure nutrient composition was monitored for 28 months on the farm, beginning in December 1999. Feed intake, feed quality, and production data were collected at monthly intervals in conjunction with production test day. Data collected included daily temperature, humidity, and daily bulk-tank, mean herd milk production per cow. All lactating cows were scored for body condition every 3 months during the study. Individual cow fecal samples were obtained to determine fecal P content. Lactating cow diets were modeled and P intakes and excretions were predicted using the Cornell Net Carbohydrate and Protein System (CNCPS) version 4.0 (Fox et al., 2000) for each production group. Strategies were developed to reduce P intakes as close to requirement as possible by manipulating purchased feed P sources. Dietary strategies were developed in close cooperation with the farm owners and their feed industry representatives. Intervention strategies were implemented from November 2000 through January 2001, and changes in nutrient intakes and excretions, feed costs, milk production, and reproductive performance were monitored. Milk production was monitored short-term before and after implementation of dietary changes on the basis of daily milk sold per cow. Long-term milk production trends were monitored as monthly herd average milk production per cow, with the data standardized to 150 DIM and four percent FCM to control for stage of lactation and energy content variability.

Whole-farm nutrient import and export data were collected for the years 2000 (pre-implementation) and 2001 (post-implementation) to compute whole-farm mass nutrient balances for N, P, and K. These data were summarized for the pre- and post-implementation periods using the procedure of Klausner et al. (1997) to calculate whole-farm mass nutrient balances, where the whole-farm mass balance is equal to the difference between farm nutrient imports (concentrates, crops, bedding, animals, fertilizer, and, in the case of N, symbiotic fixation by legumes) and farm nutrient exports (milk, animals, and crops). Nutrient import and export amounts were determined from farm records. In the case of crops and bedding, nutrient content values were determined via analysis. Nutrient content values suggested by Klausner et al. (1997) were used for animals. Nitrogen fixation was determined using the equation of Klausner et al. (1997).

Crop yields were determined from farm records and percentage of CP was determined from forage analysis. An adjusted whole-farm mass balance was calculated for both herds for the post-implementation year, primarily to account for atypical crop purchases on both farms in the post-implementation year due to pest-induced crop loss, but also to remove variation in mass balance between years within the herd from other sources, except for purchased concentrate feed. The adjusted mass balance was calculated by setting all nutrient imports and exports in the post-implementation year equal to those in the pre-implementation year, except for purchased concentrate amounts and N, P, and K densities, which were adjusted to actual post-implementation purchased concentrate feeding rates per cow and N, P, and K densities.

Results and Discussion

Prior to any dietary adjustments, P intakes exceeded P requirements by 32 percent (Table 8). In the herd, no consistent trend in P intake across production level was detected.

Table 8. Phosphorus intake as a percentage of requirement for project herds prior to dietary adjustment (Cerosaletti, et al. 2004).

	Milk production level, kg/d per cow				
Item	<23	23–32	32–41	>41	Mean
R Farm	154	171	174	161	165

Even though dietary P intakes were well in excess of requirement, limited amounts of mineral P were fed even before dietary adjustments were made (Table 9). Most of the dietary P in this herd was from forages and from by-product feeds used in the grain mix, in particular wheat middlings. This feed was used heavily in this pelleted grain mixture not only to reduce feed cost, but also to obtain a good-quality pellet.

Table 9. Diet for R Farm for January 2001 test[1] before and after dietary adjustments; herd average production of 32.7 kg/d per cow (Cerosaletti et al. 2004).

Item	High-P diet	Low-P diet
	kg of DM/d	
1st cut grass round bale silage	3.8	3.8
3rd cut grass round bale silage	1.6	1.6
2nd cut grass hay	3.3	3.3
Corn silage	2.9	2.9
Corn meal	2.4	4.0
High P grain mix	8.1	
Low P grain mix		5.7
Total DMI	22.1	21.3
Metabolizable energy balance, %	106	103
Days to gain 1 condition score	231	441
Metabolizable protein balance, %	102	101
Urea cost, Mcal/d	1.11	0.60
Dietary CP density, %	18.3	17.6
Dietary P density, %	0.56	0.44
P intake, % of requirement	152	112

[1]Month that reduced-P diets were initiated.

The higher forage P levels observed are of biological importance in balancing diets from a nutrient management perspective. These results point to the importance of analyzing forages accurately for nutrient content in balancing dairy rations. In this study, relying on average tabular values for forage P content would have resulted in an underestimation of forage P content and an oversupplementation of ration P. Higher forage P content provides an opportunity to reduce supplemental (imported) ration P. In some instances, P supplied by forages and grains alone (with no mineral P) may meet or exceed animal P requirements. In these cases, further reductions in P imports will need to come from reductions in the amount of purchased concentrate fed.

From an agronomic perspective, higher forage P content represents an opportunity to remove more soil P in herbage biomass, which is important in P-based nutrient management planning, where manure spreading rates may be limited to crop P removal. Higher forage P removal rates could, over a long time period, result in a slower rate of soil P accumulation, or even a decline in soil P. Additionally, homegrown forage P represents a recycling of P within the farm and the opportunity to export P in the form of milk every time the P cycle is turned.

Dietary P intakes were still in excess of requirement after implementation of reduced-P diets (Table 10), due largely to high P levels in the forages fed to this herd. Many of the forages in herd A contained P levels that were higher than the NRC (2001) recommended total dietary P levels for cows at similar levels of milk production and DMI. Additional P reductions might have been achieved by replacing more wheat middlings in the protein mixture with a lower P feed, such as corn meal or citrus pulp. This was not done for cost and palatability reasons.

Table 10. Predicted reductions in P intakes and manure P excretions after implementing P reduced diets in R Farm herd (Cerosaletti et al. 2004).

	Milk production level, kg/d per cow				
Item	<23	23–32	32–41	>41	Average
P intake, % of requirement	125	121	118	108	118
Reduction, g/d	16	37	50	57	40
Reduction, % of intake	18	29	32	33	28
Reduction, % of excretion	22	36	40	44	36

By implementing P-reducing strategies in the diets, predicted P intakes and excretions were reduced by an average of 25 percent (Table 10). Absolute P reductions averaged 40 g/d per cow (14.6 kg/yr per cow). These reductions increased with increasing production level, a result of the greater total daily P intake of higher-producing cattle. When

P reductions are expressed as a percentage of P intakes and excretions, a similar trend holds for the herd. Measured fecal P concentration decreased significantly (1.32 percent of DM before implementation vs. 0.88 percent after implementation, $P < 0.001$, $n = 90$), by an average 33 percent.

Short- and long-term milk production data before and after dietary adjustments (Table 11) suggest that no dramatic short-term drops in milk production have occurred as a result of reducing P in dairy diets.

Table 11. Short and long term milk production before and after implementation of reduced P diets (Cerosaletti et al. 2004).

Item	Mean daily herd milk weights[1] Before	After	Mean monthly milk weights[2] Before	After
No. of observations	10	10	14	14
Milk, kg/d per cow	28.6	30.4	31.8	32.2

[1]Mean weights for 10 d immediately pre- and post-implementation.
[2]Standardized to 150 DIM, 4% FCM.

The herd experienced a milk production increase immediately after implementing dietary changes, which may be attributable to increased intake of rumen-fermentable carbohydrate and/or metabolizable protein, although diets were not balanced to support such increases.

Whole-farm mass nutrient balances were determined for both pre- and post-implementation years (Table 12). Initial nutrient balances are consistent with those reported for typical northeastern US commercial dairy farms (Tylutki and Fox, 1997; Cerosaletti et al., 1998; Klausner et al., 1998), as well as for dairy farms in The Netherlands (Valk et al., 2000). Through dietary manipulation of concentrate P sources alone, the amount of P remaining on the farms was reduced to 49 percent in the adjusted balances, with reductions of 65 percent in the amount of P remaining on the farm. The percentage of P remaining was reduced to less than 45 percent.

The reductions in P intakes and manure P excretions obtained in this study have major implications for farm management. Reductions in manure P excretions can result in less crop acreage needed to recycle manure P under a P-based nutrient management plan (manure P application restricted to crop P removal). Powell et al. (2001) estimated that reducing dietary P density from 0.48 to 0.38 percent of DM would reduce manure P excretion enough to lower crop acreage required to recycle manure P by 44 percent. Crop fields that have a very high soil test P level resulting from excessive fertilizer and manure P application may also be restricted from further application of manure when nutrient management plans are designed according to USDA-NRCS (1999) guidelines. Reducing manure P content via dietary manipulation can significantly slow the rate of soil P accumulation, and in some cases, even bring it to equilibrium (Powell et al. 2001).

Table 12. Whole-farm mass nutrient balances before and after dietary adjustments (Cerosaletti et al. 2004).

Item	P before	after	adjusted[1]
Imports			
Purchased concentrate	2067	1165	1123
Purchased crops	—	52	—
Bedding	5	5	5
Animals	5	12	5
Fertilizer	203	203	203
Total	2280	1437	1336
Exports			
Milk	728	735	728
Animals sold	98	59	98
Crops sold	—	32	—
Total	826	826	826
Difference[2]			
Total	1454	611	510
Per mature cow[3]	19	8	7
% of imports	64	43	38

[1]Values in this column are the same as the "before" year with the exception of purchased concentrate amounts and densities which were adjusted to "after" year purchased concentrate feeding rates per cow and densities.
[2] Amount remaining on farm.
[3]Number of cows: before 78, after, 81, adjusted, 78.

On a basin-wide scale, the P reductions obtained in this study have major implications. There are presently between 7000 and 8000 mature dairy cows in the Cannonsville Reservoir Basin. Should an average reduction in P intake and manure P excretion of 25 g/d per cow be achieved across all lactating cows in the basin, then the total feed P imports into the basin and P excreted in the dairy manure produced in the basin would be reduced by 64,000 to 73,000 kg/yr. This reduction is similar to the 50,000-kg average annual total phosphorus load to the Cannonsville Reservoir (Longabucco and Rafferty, 1998; Longabucco, 2001). Reductions of this magnitude represent a substantial reduction in P

loading on agricultural soils via manure application. Phosphorus loading (both dissolved and total P) of watercourses has been shown to be highly correlated to dairy manure production and application (McFarland and Hauck, 1999).

Results

The results of this field study indicate that large reductions in feed P imports, whole-farm mass P balance, and manure P excretions can be achieved through dietary manipulation on commercial dairy farms in the northeastern US. Such reductions will be highly variable from farm to farm, based on variable levels of dietary P intake, forage P content, and willingness of the farmer to adopt reduced P diets. Reducing dietary P levels closer to requirement will require frequent and accurate feed analysis, quantification of DMI, and ration management to ensure that formulated diets are mixed and delivered to the cows properly. Strategies to reduce P intake by manipulating purchased concentrate will need to take into consideration mineral and by-product feed P sources. Choice of forages may also play a role on some farms. Forages high in P content may make it difficult to reduce P intakes to requirement; however, homegrown forages must be recognized as a recycled source of P on the farm, and must not contribute to a mass nutrient imbalance. Reduced P dietary strategies will also need to consider structural, animal, people, and economic issues. Even modest dietary P reductions can have a major impact on watershed-scale P imports in watersheds that have large agricultural land use and where dairy farming constitutes a substantial portion of the agriculture.

Development of Watershed Data Sets and Modeling

During his research on the R-Farm, Dean Hively compiled data sets for watershed physiography, land use, farm management, precipitation and temperature, and monitoring station records. These data were used in the estimation of phosphorus loading coefficients in hydrological modeling. A weather station was installed in 2002 to collect data for future modeling (Hively, 2004).

Modeling: SWAT and GWLF

The Soil and Water Assessment Tool (SWAT) model was applied to the Shaw Road watershed (Remnek, 2003), and a comparison was made between SWAT and Generalized Watershed Loading Function (GWLF) by applying these models to R-Farm (Cerucci and Pacenka, 2003). Results indicated that neither of these models performed better than the fully distributed SMDR model for analysis of phosphorus loading at a small spatial scale. SMDR was preferred over the SWAT and GWLF for analysis on R-Farm. [Kim and Steenhuis developed their variable source area models of storm runoff (Kim and Steenhuis, 2001a), soil erosion and deposition (Kim and Steenhuis, 2001b) during several rainfall events in the summer of 1993 and 1994 using the R-Farm data as the basis of modeling.]

Marcelo Cerucci, NYS WRI, performed the GWLF simulations at the Shaw Road and R Farm sites and the SWAT hydrology simulations at the R Farm site. Marcelo wrote much of the first draft of the narrative for this report. Alexi Remnek, Cornell Civil and Environmental Engineering (CEE), performed the HSPF simulations at Shaw Road. Ziyin Shen of Cornell CEE performed the SWAT simulations at Shaw Road. Shen also provided the basic methodology for using SSURGO soils data with SWAT2000, which ordinarily only uses STATSGO soil data. Eliza Bettinger assisted in the initial representation of the sites with ArcView.

Hydrologic Model Applications

Hydrologic models for the prediction of nonpoint source pollution have been extensively applied and tested in many large-scale watersheds (Corwin, 1996; Young, 1989). However, the need for more information about the performance of BMPs has created a demand for testing these models for small scale watersheds.

The modeling tools considered in this analysis are the USDA's SWAT (Arnold et al. 1995) and GWLF (Haith et at. 1987). These models provide time series of stream flow, sediment and phosphorus loads as outputs, which were compared against observed data collected at the research site. The comparison between the simulated loads and observed data help to form a basis to evaluate models performance, and their future use as management tools to assess the impacts of management practices in terms of phosphorus loads.

Other hydrological and water quality modeling work in the full Cannonsville Reservoir basin includes that by New York City (New York City Department of Environmental Protection, 1998 and 2001) and a project at Cornell University parallel to the one supporting this work (personal communications, B. A. Tolson and C. A. Shoemaker, 2002) (this needs to be updated). These two projects have been applying a custom version of GWLF and the 2000 version of SWAT, respectively. Prior hydrology modeling work at the same small watershed modeled used a GIS-based Soil Moisture Routing Model (Frankenberger et. al., 1999). This work was was initiated under the Delaware County Action Plan for Phosphorus Reduction (Delaware County Board of Supervisors, 1999 and 2002).

There are hundreds to thousands of parameters that can be adjusted in the SWAT calibration process to fit to observed streamflow, sediment concentration, and nutrient concentrations, depending on how many sub-watersheds, land cover types, and soil types are represented. GWLF calibration involves a few dozen parameters. This illustrative simulation represents the results of a small time investment typical of comparative model application. This level of effort was adequate to help choose between SWAT and GWLF for more detailed application that is ongoing.

Summary of SWAT

The Soil and Water Assessment Tool (SWAT) is a continuous time simulation model able to predict the impacts of environmental characteristics and management practices on water quality over time. SWAT was developed by USDA (Arnold et al., 1995). SWAT requires specific information about weather, soil properties, topography, vegetation, and land management practices occurring in the watershed.

SWAT can simulate hydrology, sediment, nutrients and chemicals for several sub-basins draining to a common outlet. The basic simulation unit of SWAT is the hydrological response unit (HRU). Each sub-basin may contain one or more HRUs depending on the resolution of spatial data. Each HRU is defined according to the land use and soil type distribution over the landscape. SWAT simulations are performed for each HRU and summarized for individual sub-basins. Water and other substances leaving an upstream sub-basin via a channel may be routed through a cascade of downstream sub-basin channels until the most downstream sub-basin's final outlet.

Hydrology is the driving force behind SWAT computations. Surface runoff is computed using the Curve Number Method (USDA Soil Conservation Service, 1972), and the computation of erosion is performed using USDA's Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975). SWAT can operate linked to a GIS-ArcView interface that automatically delineates watersheds and creates sub-basin and HRU input files (Neitsch et al., 1999).

Summary of GWLF

The Generalized Watershed Loading Functions (GWLF) code was developed by Dr. Douglas Haith (Haith and Shoemaker, 1987). A much simpler model than SWAT, Haith's version 2.0 of GWLF simulates in parallel several source areas each containing one homogeneous land cover that has a particular effect on runoff volume or sediment and nutrient loads. The flows and loads from each source area are added, weighted by relative area, to represent the flow and loads from the entire watershed, without routing through channels. Although GWLF allows the subdivision of the watershed in source areas representing different land cover it does not have the same spatially distributed character as SWAT. GWLF has a relatively simple structure compared to SWAT, and limited parameters to represent management practices, groundwater movement, sediment loads, and dissolved and particulate nutrients.

The GWLF model simulates particulate nitrogen and phosphorus loads from each source area by multiplying solid phase concentrations provided in input data times sediment mass fluxes derived using the Universal Soil Loss Equation (Wischmeier, 1978) adjusted using a sediment delivery ratio. Dissolved nutrient loads from each source area are obtained by multiplying simulated runoff volumes by dissolved concentrations provided in input data.

Runoff and infiltration partitioning between rainfall and snowmelt are computed with the SCS curve number method. Evapotranspiration, limited by moisture in soil storage, returns some water to the atmosphere. Infiltrated water that does not evaporate eventually enters a linear groundwater reservoir to eventually discharge to the basin outlet. Streamflow at the outlet blends runoff with discharge from groundwater. The water balance is computed daily for unsaturated and shallow saturated zones.

Research Site

As described earlier, the R farm is located in the upper west region of the Cannonsville watershed. The basin containing most of the farm covers an area of 160 ha, consisting of land covers hay, corn, pasture and forest. Other geographic features present in the farm basin are a stream, a pond and a barnyard (Fig. 6). Soils are derived primarily from glacial till and contain fragipans usually a meter or less below the surface. In order to apply nonpoint source models to a given site, parameter values representing the distinct land covers and soils have to be estimated from independent data or calibrated.

Land cover data were initially provided by the farm owners and complemented by color infrared Digital Orthophoto Quadrangles (DOQQ) taken in December 2000 by the US Geological Survey for New York State and field observations by Cornell University personnel from 2000 through 2002 using a Global Positioning System (GPS) unit to record locations. The land cover of a given field may change over time according to crop rotations and changes in management practices. There were extensive changes in structural and non-structural management practices under the Whole Farm Plan for the site, one of the earliest participants in the WAP.

Besides land cover, soil data are also important in deriving parameter values for the models. The Delaware County Soil Survey Geographic (SSURGO) database, and the State Soil Geographic (STATSGO) database were used to obtain parameter values such as the soil hydrologic group for each soil map unit. SSURGO and STASGO databases consist of polygons representing distinct soil types, defined as map units (MUIDs), and a relational database defining properties for each of one or more soil layers making up a map unit.

Weather data are another necessary input for the models. Rainfall volume, snow melt volume, and air temperature in the basin affect hydrology and sediment and nutrient loads over time. In order for the model predictions to be comparable with data collected in the research site, the weather data used in the simulations must be representative of the actual precipitation and temperature that occurred at the site. Some of the weather data had to be drawn from a meteorological station at the village of Delhi several miles away, having a lower elevation and different wind exposure (valley instead of the farm's ridge top). These were provided by the Northeast Regional Climate Center (NRCC) at Cornell University. When possible, measurements from a NYS DEC meteorological station located onsite were used. The onsite meteorological data have limitations for use in hydrology modeling, since the station was not installed at a standard location or height. (Its role is to trigger an automated water sampler to get a baseflow sample when it starts raining and before the stream hydrograph begins to rise.)

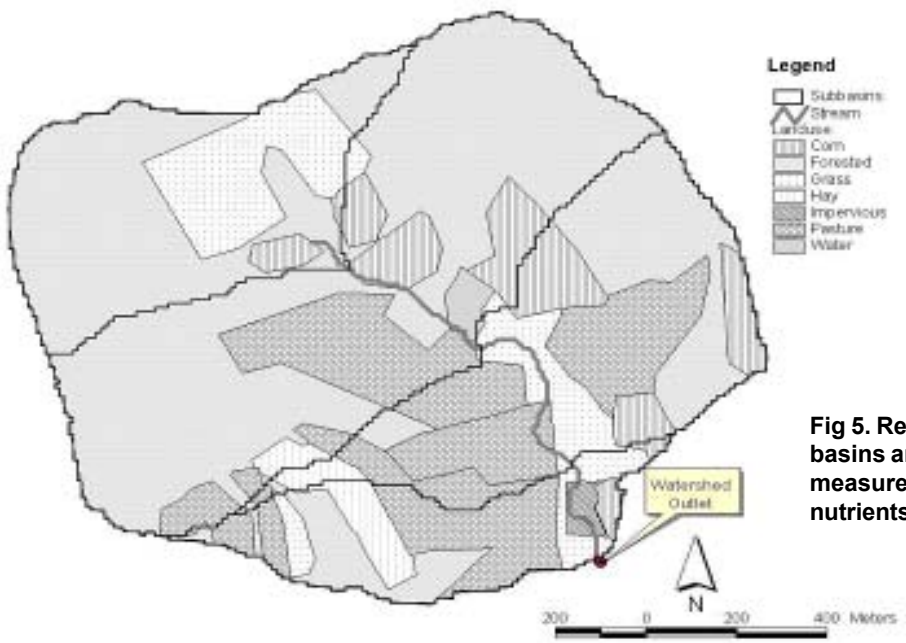


Fig 5. Research site land use, sub-basins and outlet where DEC measures flow and instream nutrients.

Besides the data for model parameterization described above, data for model calibration and validation are also used in the analysis. NYS DEC operates an automated stream gage and water quality monitoring station at the outlet of the drainage area to measure stream flow and concentrations of sediment and nutrients in the outflow from the watershed.

Streamflow and water quality data have been collected over two periods. The first period of data collection dates from June 1993 to May 1995, and consists of measurements before changes in farm management practices were implemented. A window period from June 1995 to May 1996 without measurements reflects the period when the new management practices were being implemented. Implementation involved extensive short-term disturbances from construction of structural management practices including relocating a section of the stream. The second, post-implementation monitoring period covers from June 1996 until this writing and afterward. (A second round of notable management practice changes occurred in 2002, after the period covered by this paper.)

Application of SWAT and GWLF to the Research Site

SWAT Application

The modeling of R Farm was divided according to the two monitoring periods, before the management practices were implemented and after. There is a one year gap between the periods. The modeling effort with SWAT presented in this article consists only of simulations for the first period.

Many parameter values for SWAT simulations were generated using the SWAT2000 ArcView interface, called AVSWAT. The watershed draining to the outlet of the basin, which corresponds to the same site where streamflow and water quality measurements were made, was defined based upon the topography of the terrain. Automatic watershed delineation tools available within the interface were used to obtain the watershed boundary and its geometric parameters. The watershed delineation was based on USGS 7.5 minute quadrangle 10-meter resolution Digital Elevation Models (DEM) provided by NYS DEC. The watershed was later subdivided into four sub-watersheds linked by the stream. This subdivision was made in order to better incorporate the farm pond in the modeling process and to have an estimate of the different contributions from the source areas.

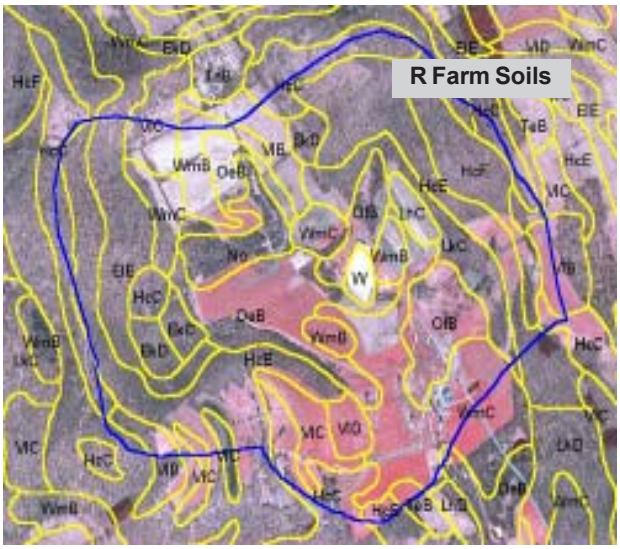
Once the watershed and sub-watersheds were delineated, the land coverage map and SSURGO soil database were used as a base to define the Hydrologic Response Units (HRUs).

A key in SWAT application to nutrients in livestock agriculture areas is the estimation of manure application per land cover type. For this comparative model application, Cannonsville basin averages estimated by Tolson (personal communication) were used. (In future work at the farm, actual manure records of the farm will be used.)

After the input files are generated, SWAT was run for the period of January 1990 to May 1998. Although the period of record for the analysis of phosphorus dates from June 1993 to May 1995, three extra years of simulation before 1993 were performed. This is done to allow the model to create realistic soil water and nutrient state estimates for the start date of the calibration period (June 1993); no soil state measurements were made at the time water monitoring began.

The results of R Farm simulations using SWAT2000 are presented next. Four different outputs are provided: Streamflow, sediment loads, particulate and dissolved phosphorus for the entire period.

SWAT simulations of stream flow are consistent with the measured data when averaged by month (Fig. 6). It is not possible to calibrate daily streamflow reliably at this site due to limitations of onsite precipitation measurements; storms that occur at the site are not all represented in simulation input data from the Delhi meteorological



station, and storms in the Delhi record do not necessarily occur at the site. Monthly averaging eliminated many of the storm timing discrepancies since they appear to be random.

The coefficient of determination of 0.75 indicates that SWAT is reproducing measured seasonal variations in streamflow, and a percent water balance error of 10.4 percent indicates that most of the measured streamflow produced in the watershed was represented for the period of simulation.

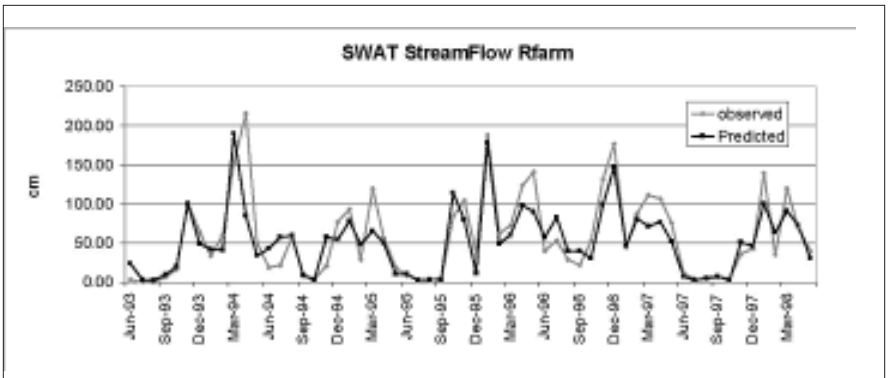


Fig. 6. Streamflow output from SWAT June 1993 to May 1998.

SWAT produces a lower-than-measured streamflow for this basin when calibrated to reproduce seasonal variability. It is possible to eliminate the water budget difference by calibrating different input parameters such soil moisture storage capacity or potential evapotranspiration. The model seems to under-predict streamflow during the spring months and over-predict streamflow in a few summer months. This problem may be related to the evapotranspiration methods used in SWAT.

The simulation of sediment loads from R Farm resulted in a good representation of the total sediment produced. The difference between monitored and simulated cumulative sediment load was 6.91 percent, which means that most sediment produced on the farm and reaching the stream is eventually represented by the model (Fig. 7). The low value for the coefficient of determination demonstrates that the seasonality of sediment loads is not being well represented. The peaks of sediment are not being captured by SWAT, and the model seems to be overestimating the sediment loads in low flow periods. These issues also occurred in a parallel application of SWAT2000 to the entire Cannonsville watershed (Tolson, personal communication), and are being addressed in that project.

Phosphorus forms in SWAT are divided into organic and inorganic. Organic phosphorus is mostly in particulate form, while inorganic phosphorus is more commonly found dissolved in surface runoff, baseflow or groundwater. SWAT's "organic" phosphorus is most comparable to "particulate phosphorus" reported in stream sampling data and SWAT's "inorganic phosphorus" is most comparable to "total dissolved phosphorus" reported

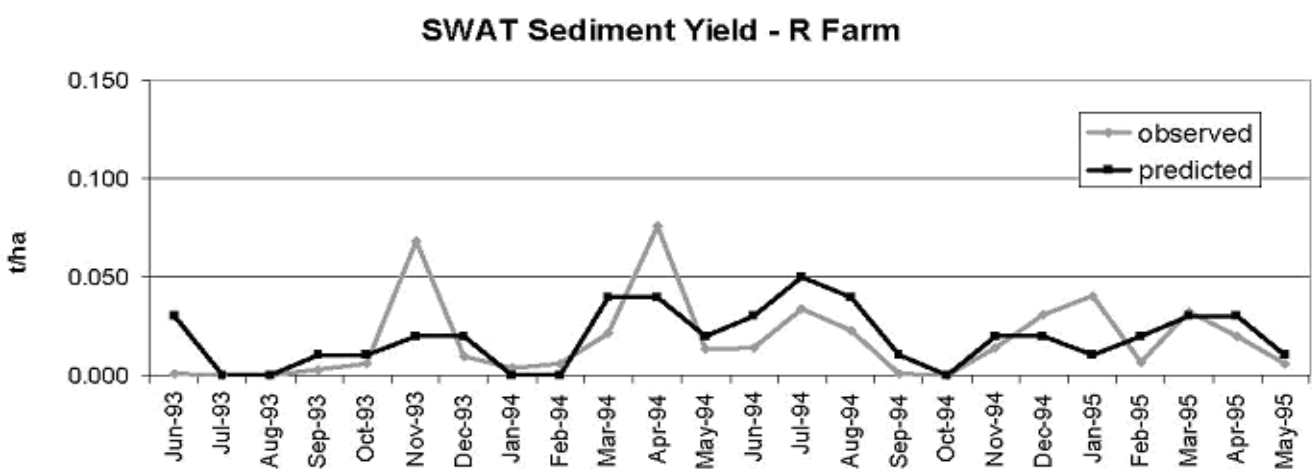


Figure 7. Sediment output from SWAT June 1993 to May 1995.

from sampling. The results of the simulations for period 1 are presented next (Figs. 8 and 9).

Particulate phosphorus (Fig. 8) is associated with sediment, and as a result, the output of particulate phosphorus will have a similar trend as the sediment loads produced in SWAT. The model is not capturing the peaks of particulate phosphorus, and overestimating the loads in low flow periods. However, the results of particulate phosphorus are better than results for sediment. The coefficient of determination obtained for this period of simulation was 0.41 and the cumulative difference was - 5.45 percent, which corresponds to a 3.3 kg/yr difference.

The simulation of dissolved phosphorus in SWAT provides a better fit to the observed data than particulate phosphorus. The coefficient of determination obtained by comparing observed with predicted dissolved phosphorus is 0.62. This means that the seasonal trends of dissolved phosphorus have been captured by SWAT. Differently from particulate phosphorus and sediment, the peaks of dissolved phosphorus have been represented well in the simulations (Fig. 9).

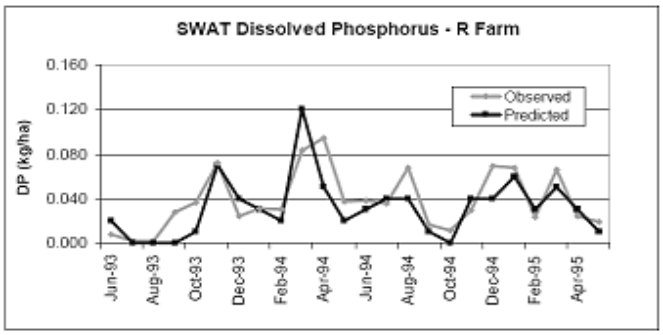


Figure 9. Dissolved phosphorus output from SWAT June 1993 to May 1995.

GWLF application

Coming from a time before GIS software became widely used, this version of GWLF does not have an integrated GIS interface to aid in deriving parameter values. The parameter values for this application of GWLF were obtained manually using ArcView GIS, land cover as provided by the farm owners and complemented by color infrared DOQQ, a USGS STATSGO digital soil map and tables, and a 10 m resolution DEM for the area. The watershed draining to the outlet of the basin was defined using the same 10 m DEM, and employing the same methods used in the SWAT delineation.

GWLF reproduces streamflow seasonality as well as SWAT did for the site (Fig. 10). The coefficient of determination obtained was 0.75. The cumulative water budget difference was 4.13 percent.

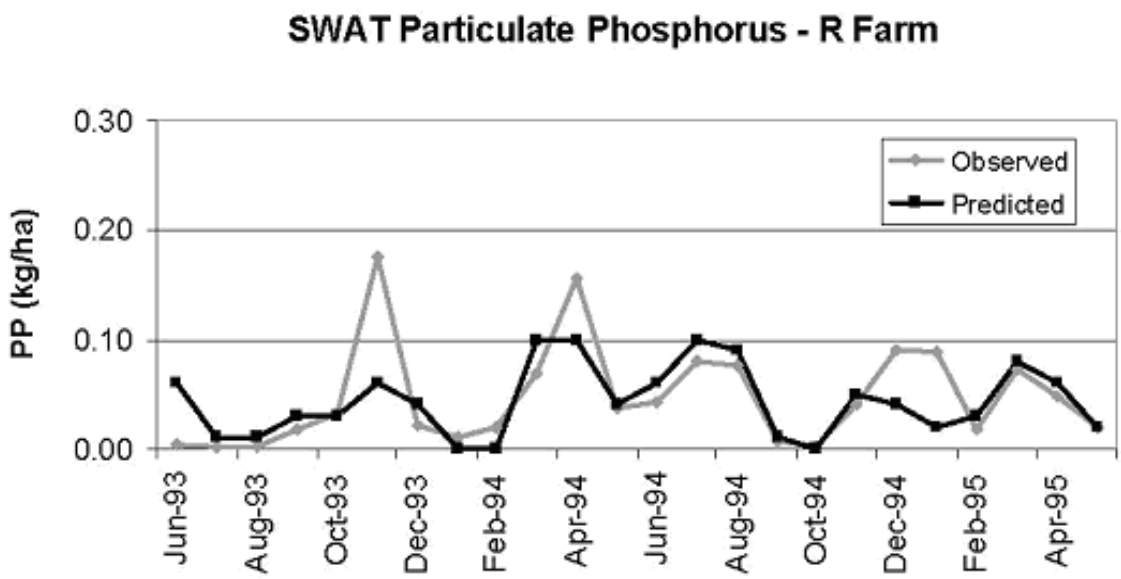


Figure 8. Particulate phosphorus output from SWAT June 1993 to May 1995.

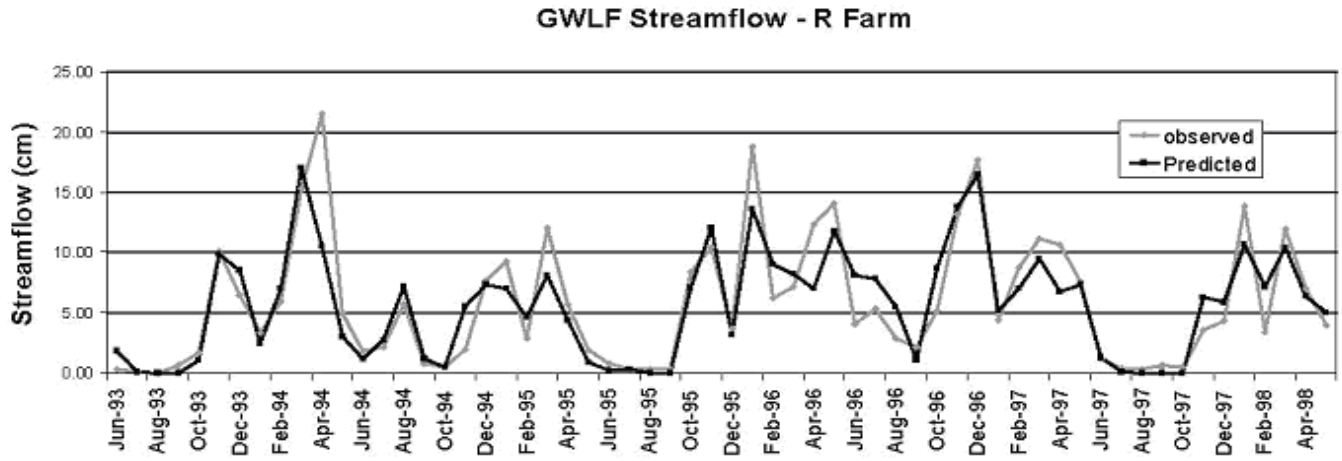


Fig. 10. R Farm streamflow simulations with GWLF from June 1993 to May 1998.

GWLF represents water quality in rudimentary ways, adequate for some purposes but not others. Unlike SWAT, which traces the application of fertilizer and manure to soil through the soil overland or via subsurface, GWLF represents water quality using a few concentration values specified in input data. For example, dissolved phosphorus load on a day from one land use are is the product of a concentration value in input, the simulated runoff depth for the day, and the area occupied by the land use type. Particulate phosphorus load is the product of computed sediment load and a mass/mass concentration in input data, per land use and optionally varying per month.

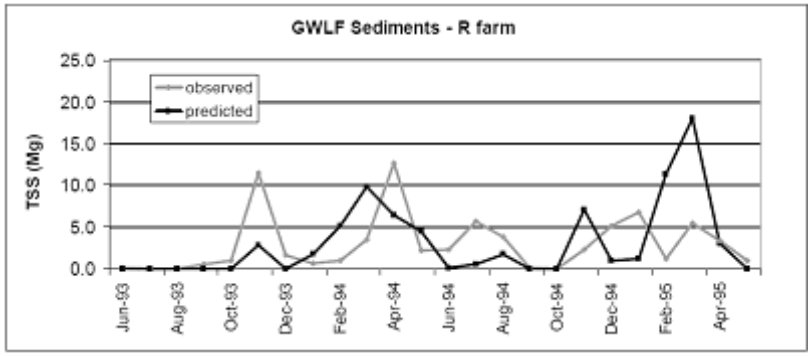


Figure 11. R Farm sediment simulations with GWLF from Jun 1993 to May 1995.

Probably due to the constraints of its simple structure, GWLF's simulation of sediment and particulate phosphorus did not provide a good reproduction of these variables at the farm scale. The seasonality of sediment loads and particulate phosphorus is not well represented. The values of the total loads are closer to the observed values, and are obtained through calibration of the few input parameters allowed. The amount of sediment leaving the watershed can be calibrated using the sediment delivery ratio parameter, which determines the percentage of eroded soil that will reach the water body and become sediment (Figs. 11 and 12).

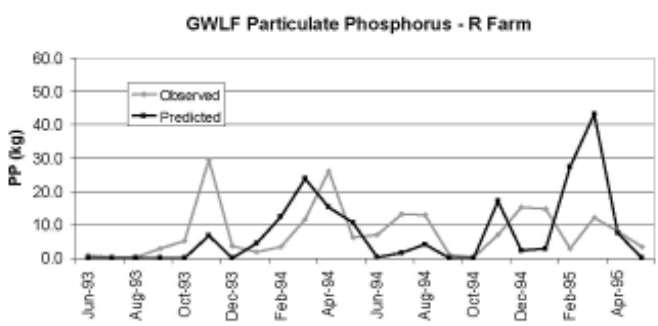


Fig. 12. R Farm particulate phosphorus with GWLF from June 1993 to May 1995.

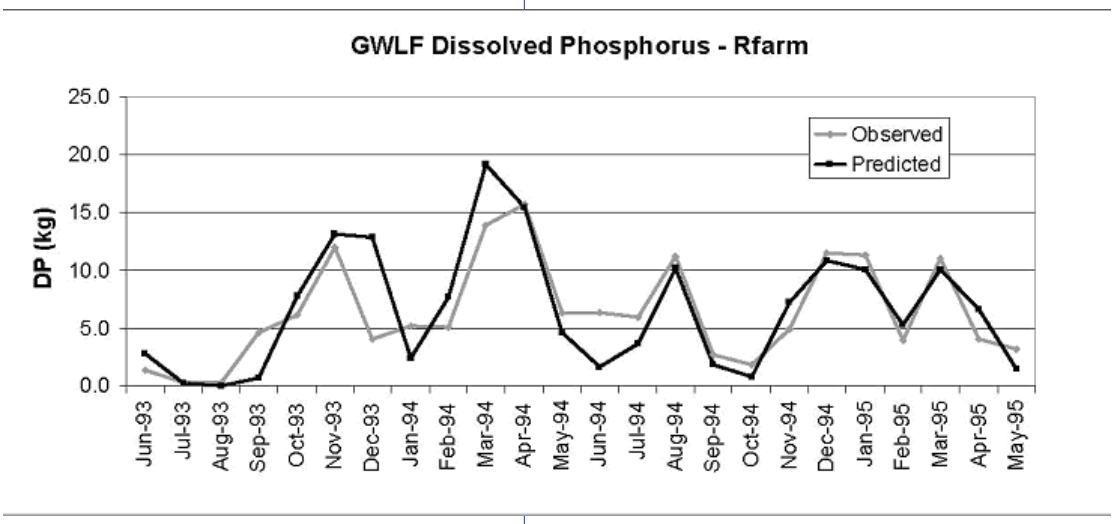


Fig. 13. R Farm dissolved phosphorus with GWLF from June 1993 to May 1995.

Although GWLF lacks precision in representing the loads of sediment and phosphorus at the farm scale, the simulations of dissolved phosphorus can be considered very satisfactory (Fig. 13). The coefficient of determination for dissolved phosphorus is 0.7 and the cumulative mass balance difference from observations was less than one percent for period one.

Interpretation and Conclusions for SWAT and GWLF

SWAT provided good estimates of flow and dissolved phosphorus yielded by the farm watershed. The simulations of sediment and particulate phosphorus did not capture the peaks; however, they provided a reasonable representation of the total values and the monthly events. SWAT allowed R Farm watershed to be subdivided into smaller watersheds. This distributed parameter approach can be used to better understand the sources of phosphorus within the farm.

SWAT requires many parameter values; its application was facilitated by its AVSWAT interface. The input files were automatically generated by the interface, using parameter values gathered from SSURGO soil database, land use, and other default values available in SWAT database.

GWLF simulations resulting from a small effort also provided a good representation of the flow and dissolved phosphorus, however they did not represent the loads of sediment and particulate phosphorus properly on a monthly time scale. New York City's variant of GWLF addressed this using an algorithm change, relaxing an undocumented assumption about the starting month of a simulation (Schneiderman et al. 2002).

GWLF has a very simple structure. This fact allowed the model to be easily applied at R Farm. Although the simplicity of GWLF and the reduced number of parameter necessary to perform simulations constitute an advantage of this model, it may also lead to a misrepresentation of some processes. The lack of equations and parameters representing the movement of eroded soil and associated pollutants makes GWLF a limited tool for assessing monthly loads of sediment and particulate phosphorus at the farm scale. In addition, the inputs for GWLF have to be generated manually.

The performance of the models could be evaluated according to the time scale of the results. On an annual time scale, both SWAT and GWLF provide good representations of flow, sediment, and particulate and dissolved phosphorus. On a monthly time scale, both SWAT and GWLF provide good estimates of flow and dissolved phosphorus, however, SWAT provides better monthly outputs of sediment and particulate phosphorus for this small-scale watershed.

Another advantage of SWAT over GWLF is the possibility of representing management practices in the simulations. SWAT is able to represent the changes of management practices in direct terms

(for example changes in rates and timings of manure application) and to identify differences in phosphorus sources and critical areas due to changes in management practices. It also allows features such as the pond located in the farm and stream processes to be simulated explicitly.

This work was considered adequate to choose SWAT2000 for continuation. Data about the farm are adequate to meet most of SWAT2000's input requirements for a more thorough application, which will include representations of the drastic change in manure application patterns and concentrated source emissions already accomplished. (Meteorology data shortcomings are being addressed with a second weather station installed in December 2002.) SWAT's difficulties in representing sediment appear to be addressable via corrections to its source code and more thorough calibration (Tolson, personal communication).

Modeling: Soil Moisture Distribution and Routing Model (SMDR)

Initially developed by Jane Frankenberger in the Department of Biological and Environmental Engineering for her dissertation (1996), the SMDR simulates the hydrology of watersheds with shallow sloping soils (Frankenberger et al. 1999). The model utilizes elevation, soil, and land use GIS data to predict the spatial distribution of soil moisture, evapotranspiration, surface runoff and interflow on a watershed. An updated version of the model (Gérard-Marchant, 2003) was applied to the R-Farm watershed to estimate daily distribution of soil saturation degree and runoff volumes (Hively, 2004). A preliminary distributed model for estimating total dissolved phosphorus from the R-Farm watershed was developed on the estimation given by the updated SMDR model, together with other data (Gérard-Marchant et al., 2003). Gérard-Marchant et al developed a spatially distributed model of total dissolved phosphorus (TDP) loading using 10-m raster maps covering the farm watershed (2003). They calculated separately TDP transport in baseflow and in surface runoff from manure-covered and non-manure-covered areas. Based on Morgan's soil P test maps, the pattern of manure application, and land use, landscape characteristics were defined for watershed landscape components. Field records were used to estimate the spatial distribution of manure applications.

SMDR Model Results

Simulations were conducted for the 30-month period, from 07/01/1996 to 12/31/1998. For the summer, hydrograph peaks were correctly timed but their intensity underestimated. Conversely, during the winter, high flow events were often overestimated, and did not always correspond to the actual day of the peak. The authors attributed these differences to the crude snowmelts estimates provided by the temperature-index method used.

TDP Transport Model Results

Daily TDP loads were simulated from the sum of overland flow from soils, transport by baseflow, and overland flow from manure. The resultant loads were compared to the daily observed loads recorded by the monitoring station. The TDP loading was substantially overestimated during the winter periods. The authors concluded this was due to the overestimation of surface runoff mentioned, attributable to poor estimation of snowmelt and rain/snow cover interaction. In the summer period, the total TDP load observed was 9.1 kg and the simulated one 5.9 kg. This underestimation was imputed to an underestimation of streamflows caused by potential evapotranspiration being overestimated. Results however supported the conclusion that total TDP loads to the streams appear to be largely controlled by the soil P transported by overland flow. The contribution of manure P to total TDP loads appeared to be negligible. This conclusion endorses the effectiveness of the current manure management plan on the farm.

Conclusions

From Field Scale Monitoring

Samples of runoff across the farm clearly showed that the levels of phosphorus observed related directly to the applications of manure. Morgan's soil test for phosphorus was found to predict the concentrations of total dissolved phosphorus quite well. An experimental study of runoff conducted on nine distinct sites across the farm showed that only four of the sites produced significant loads of phosphorus in runoff produced by rain simulation. The four sites were the barnyard, a cowpath, a ploughed cornfield, and a hillside pasture in Spring. Intensively grazed areas, a hayfield, a swaled pasture and a forested area all did not produce significant loads. The concentration of total dissolved phosphorus in runoff from the barnyard was observed to be 11.6 mg L^{-1}. This compared with an average concentration of only 0.007 mg L^{-1} measured in the forested area. This result emphasizes the critical importance of preventing runoff from barnyard areas. Concentrations of runoff from all non-forested sites on the farm were found to carry substantially higher concentrations of phosphorus than that from the forested area. In-field tile lines were also found to be significant sources of total dissolved phosphorus. The remedy suggested for the latter problem was to install upslope tile drains and drainage ditches.

From Precision Feeding

A Best Practice: One of the most effective management practices implemented on the farm was precision feeding. This remarkable study showed that the import of phosphorus to the farm, the excretion of phosphorus in manure, and the whole farm mass balance for phosphorus could all be significantly altered. The merit of this practice is amplified by its low cost of implementation, compared to structural management practices, and by the economic gain its adoption can provide to the farmer.

From Models

SWAT and GWLF were both applied to the R Farm. Both models provided reasonable estimates of flow and dissolved phosphorus. However, both models were much less successful in representing the loads of sediment and particulate phosphorus. Of the two models GWLF is easier to apply having a very simple structure and far fewer parameter requirements than SWAT. However, SWAT is superior in representing sediment and particulate phosphorus loads. SWAT can better represent management practices in its simulations. Hence SWAT seems a feasible tool for evaluating management practices and their impact on water quality. Cornell scientists also developed and applied the Soil Moisture Distribution and Routing Model (SMDR). A conclusion of this study was that total dissolved phosphorus from manure appeared to be negligible. This conclusion supports the effectiveness of current manure management on the farm.

From the Paired Watershed Study

Implementation of BMPs on the R farm resulted in a large number of changes to many aspects of the farm's infrastructure and management, and the observed reductions in nutrients and sediment loads are probably attributable to all of them. The small-scale watershed monitoring approach was an effective method for evaluating treatments that affected loading processes throughout the farm landscape. The results of the study quantitatively demonstrate that dairy farm BMPs can succeed in reducing losses during runoff events as well as

baseflow periods. Overall, decreases in farm loads were estimated to be 64 percent for ammonia, 52 percent for dissolved phosphorus, 49 percent for particulate phosphorus, 45 percent for suspended sediment, and 23 percent for nitrite+nitrate.

Multivariate ANCOVA was an effective technique for evaluating paired watershed event load data. The multivariate model improved on previous methods of paired watershed data analysis by including several covariates to control for environmental variability. The nonfarm watershed proved to be a satisfactory control for farm loads as long as additional terms, including the ratio of watershed event flow volumes, were also incorporated as covariates. Separation of the data into seasons before analysis allowed a more precise evaluation of changes in event loads and their relation to specific BMPs when compared with the full-year analysis, despite the smaller sample size.

The nonfarm site was a valuable alternative to monitoring a second farm in this long-term paired watershed study, as it can be difficult to find a control farm that maintains consistent operation and management throughout the study period, especially in areas like the New York City water supply basin where 85 percent of the dairy farms are participating in the WFP program. Use of a stable control site to represent environmental variation along with other covariates from the treatment watershed was critical for successfully determining the seasonal effects of BMPs on farm loading.

While the monitored farm has been more intensively managed, from an environmental perspective, than most other farms in the region that have adopted BMPs, our findings provide evidence that the Watershed Agricultural Program has reduced phosphorus and ammonia loading to one of New York City's water supplies, Cannonsville Reservoir. Presumably, initial decreases in phosphorus stream losses were a consequence of greater retention of P within the farm watershed, an outcome that could eventually lead to saturation of soil with P. Reductions in dissolved P occurring later in the study may be attributable to the implementation of precision feeding. Management programs that combine effective conservation and nutrient management measures with practices designed to improve farm P mass balance and slow net soil accumulation would appear to have the best chance of protecting water quality over the long term. The observed decreases in ammonia loads may be a result of both loss to the atmosphere through volatilization and conversion to nitrate in the soil with subsequent increases in loads of this nitrogen form during certain seasons. While nitrogen is not as important in this freshwater system as phosphorus, in ocean and estuarine systems where excess nitrogen is typically the nutrient of concern for eutrophication, other farm management practices that reduce inputs of N or utilize it more efficiently on the farm, may be of more value than the ones implemented in this project.

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