Nutrient Transport in a Restored Riparian Wetland

G. Vellidis,* R. Lowrance, P. Gay, and R. K. Hubbard

ABSTRACT

We determined the water quality effect of a restored forested riparian wetland adjacent to a manure application area and a heavily fertilized pasture in the Georgia Coastal Plain. The buffer system was managed based on USDA recommendations and averaged 38 m in width. Water quality and hydrology data were collected from 1991–1999. A nitrate plume in shallow ground water with concentrations exceeding 10 mg NO_3–N L^{-1} moved into the restored forested riparian wetland. Along most of the plume front, concentrations were less than 4 mg NO_3–N L^{-1} within 25 m. Two preferential flow paths associated with past hydrologic modifications to the site allowed the nitrate plume to progress further into the restored forested riparian wetland. Surface runoff total N, dissolved reactive phosphorus (DRP), and total P concentrations averaged 8.63 mg N L^{-1}, 1.37 mg P L^{-1}, and 1.48 mg P L^{-1}, respectively, at the field edge and were reduced to 4.18 mg N L^{-1}, 0.31 mg P L^{-1}, and 0.36 mg P L^{-1}, respectively, at the restored forested riparian wetland outlet. Water and nutrient mass balance showed that retention and removal rates for nitrogen species ranged from a high of 78% for nitrate to a low of 52% for ammonium. Retention rates for both DRP and total P were 66%. Most of the N retention and removal was accounted for by denitrification. Mean annual concentrations of total N and total P leaving the restored forested riparian wetland were 1.98 mg N L^{-1} and 0.24 mg P L^{-1}, respectively.

Section 208 of the United States Federal Water Pollution Control Act Amendments of 1972, Public Law no. 92-500, specifies the development of water quality management plans to control nonpoint-source pollution from agriculture, including manure application sites. As a result, considerable research has been devoted to this topic and numerous strategies have been evaluated. Coastal Plain streams in Georgia, Florida, and other states often violate water quality standards for low dissolved oxygen. In the Georgia Coastal Plain 90% of monitored streams violated water quality standards for dissolved oxygen (Suwannee River Water Management District, 2000). These dissolved oxygen violations are thought to be due to nutrient enrichment mostly from nonpoint sources of N and P in these streams (Georgia Department of Natural Resources Environmental Protection Division, 2000). If the USEPA requires states to implement nutrient (N and P) water quality criteria in the next two years, most of these same streams and many others will violate criteria for total N and total P (USEPA, 2000).

Surface runoff and subsurface flow from farm animal waste land application sites can provide significant loadings of nutrients to receiving waters unless appropriate management techniques are employed. Grass buffer zones or vegetative filter strips (VFS) have been investigated as a means of reducing nutrient loadings in streams by minimizing pollutant concentration in manure-polluted runoff (Doyle et al., 1977; Dickey and Vanderholm, 1981; Overcash et al., 1981). The buffer zones reduce runoff by increasing infiltration, which may increase the nutrient loading of ground water. Recent studies have focused on the use of grass and forest buffers as treatment areas for liquid manure (Hubbard et al., 1998b). Dillaha et al. (1989) and Magette et al. (1989) found that VFS with buffer strip to waste area length ratios of 0.25 and 0.5 were effective in removing sediment from runoff, but did not reliably reduce nutrient losses from agricultural areas. A buffer strip to waste area length ratio of 1.0 is usually required to reduce nutrient loading, thus making the cost of the buffer zone a major factor in the cost–benefit analysis of land application waste treatment (Bingham et al., 1980).

Riparian ecosystems can also be used to control nonpoint pollution. Lowrance et al. (1984, 1985) and Peterjohn and Correll (1984) demonstrated that riparian forest ecosystems of coastal plain agricultural watersheds are excellent nutrient sinks that buffer the nutrient discharge from surrounding agroecosystems. Lowrance et al. (1984), Peterjohn and Correll (1984), and Lowrance (1992) showed that nutrient uptake and removal by soil and vegetation in the riparian forest ecosystem prevented agricultural upland outputs from reaching stream channels. They concluded that the riparian ecosystem can serve as both a short- and long-term nutrient filter and sink if aboveground vegetative biomass is periodically harvested to ensure a net uptake of nutrients. Forested riparian wetlands have also been shown to function as nutrient sinks and filters for land-treated waste application of municipal sewage (Turner et al., 1976; Sloey et al., 1978). Scientific studies of restoration efforts are relatively recent and few in number and causal relationships between restoration techniques and long-term responses of particular ecosystem process are poorly understood (Kusler and Kentula, 1990).

Information on storage and removal of nutrients migrating through the riparian zone is still lacking for riparian areas during the first years following restoration. Although mature riparian forests have been shown to be excellent nutrient sinks and buffers, little research has been conducted on the effectiveness of newly established riparian buffers. Licht and Schnoor (1990) used densely planted poplar (*Populus* spp.) trees to provide a riparian buffer strip for conventional row crop agricultural land in Iowa. They found that in some instances, the reestablished riparian forest decreased nitrate con-

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Abbreviations: DRP, dissolved reactive phosphorus; LIFE, low-impact flow event; TKN, total Kjeldahl nitrogen.

centrations in the shallow soil profile from 25 to 5 mg NO_3–N kg^{-1} dry soil during the first year of reestablishment. Clausen et al. (2000) studied nutrient transport and developed N budgets for a restored fescue (*Festuca* spp.) buffer in Connecticut. They found that loads and concentrations of nitrate, total Kjeldahl nitrogen (TKN), and total P were reduced in runoff and that the concentration of nitrate in ground water was reduced by 35% compared with the control, which was an unrestored riparian corn (*Zea mays* L.) field. Oelbermann and Gordon (2000) found that a 12-year-old riparian forest buffer restored with northern hardwood species in Ontario provided large quantities of N in litterfall to the adjacent stream. They hypothesized that the restored forest buffer converted inorganic N inputs to organic N outputs to the stream, an observation made for mature buffers by Lowrance et al. (1983).

Although a number of studies of riparian forest buffer restoration were begun in the 1990s (Schultz et al., 1995; Schuetz et al., 1994; Vellidis et al., 1994), and some components of these studies have been reported (Tufekcioglu et al., 2001, Lee et al., 1999., Lowrance et al., 1995), there are still very few estimates of the ability of restored riparian forest buffers to remove nutrients from agricultural runoff and shallow ground water. Plot-scale studies of mature forest buffers have provided guidance on the use of buffers for water quality renovation, but studies are needed that examine the effects of riparian buffer restoration on stream nutrient levels (Dosskey, 2000) and compare these stream nutrient levels to suggested water quality standards.

Because much of the emphasis in buffer zone studies has been on N, there are still substantial unanswered questions about the effects of forest buffers and wetlands on P transport. The function of riparian buffers in control of P is particularly important in manure application areas that might already be overloaded with P and because the near-stream area is known to be the most critical for P discharge (Heathwaite et al., 2000; Smil, 2000; Sharpley et al., 1999; Hubbard et al., 1998a).

The purpose of this study was to determine the effects of a restored forested riparian wetland buffer system on surface and subsurface transport of nitrogen and phosphorus entering the buffer from adjacent upland agricultural production sites, including a liquid manure application area and a pasture. The site was also used for studies of denitrification removal of nitrate (Lowrance et al., 1995) and herbicide transport through the riparian buffer (Vellidis et al., 2002).

MATERIALS AND METHODS

Study Site

The study was done at a research site on the Animal and Dairy Science Research Farm that is known as the Dairy Wetland. The site is on the Tifton Campus of the University of Georgia, which is located in the Tifton–Vidalia Upland portion of the Gulf-Atlantic Coastal Plain in the headwaters of the Suwannee River basin (Fig. 1). The climate of the Tifton–Vidalia Upland is humid subtropical providing abundant rainfall and a long growing season. Average monthly temperatures range from 11°C in January to 27°C in July and August with a 47-yr mean annual temperature of 19.2°C (Batten, 1980). The average frost-free season is 253 d. Precipitation follows a definite seasonal pattern with generally low rainfall from September through November and an increase in precipitation in December through early May. Rainfall typically decreases again in May and early June. Summer thunderstorms and tropical depressions cause July and August to be wetter months on average. Average annual precipitation for the study period (1991–1999) was 1210 mm.

Fig. 1. The Dairy Wetland research site is located near Tifton, Georgia, within the Suwannee River basin of the U.S. Coastal Plain.

Because of both a plinthic soil horizon (irreversibly hardened mixture of iron sesquioxides and quartz) beginning at a depth of 1 to 1.5 m and the presence of the Hawthorn Formation, a geologic formation that limits deep recharge to the regional aquifer system, most of the excess precipitation in the Tifton–Vidalia Upland moves either laterally in shallow saturated and unsaturated flow or moves in surface runoff during storm events. It is common for the lateral saturated flow to form hillside seeps that can combine with direct surface runoff. The general hydrology of the Tifton–Vidalia Upland is reflected at the Dairy Wetland and makes this region and the particular site ideal for the study of surface runoff and shallow subsurface transport of agricultural pollutants into riparian ecosystems. The soil at the Dairy Wetland is an Alapaha loamy sand (loamy, siliceous, subactive, thermic Arenic Plinthic Paleaquult). The soil of the adjacent upland area is a Tifton loamy sand (fine-loamy, kaolinitic, thermic Plinthic Kandiudult) (Calhoun, 1983).

On the upland southwest of the Dairy Wetland (Fig. 2), a 5.6-ha center pivot irrigation system applies liquid manure derived from flush cleaning of a 120-dairy cow facility at a nitrogen application rate of 600 kg N ha^{-1} yr^{-1} (Vellidis et al., 1996). Manure application began in July 1991. Liquid manure is applied year-round on a biweekly schedule and results in an additional 760 mm yr^{-1} of water to the field. The cropping system under the pivot is overseeding of ‘Abruzzi’ rye (*Secale cereale* L.) into ‘Tifton 44’ bermudagrass [*Cynodon dactylon* (L.) Pers.] sod in the fall, followed by minimum tillage planting

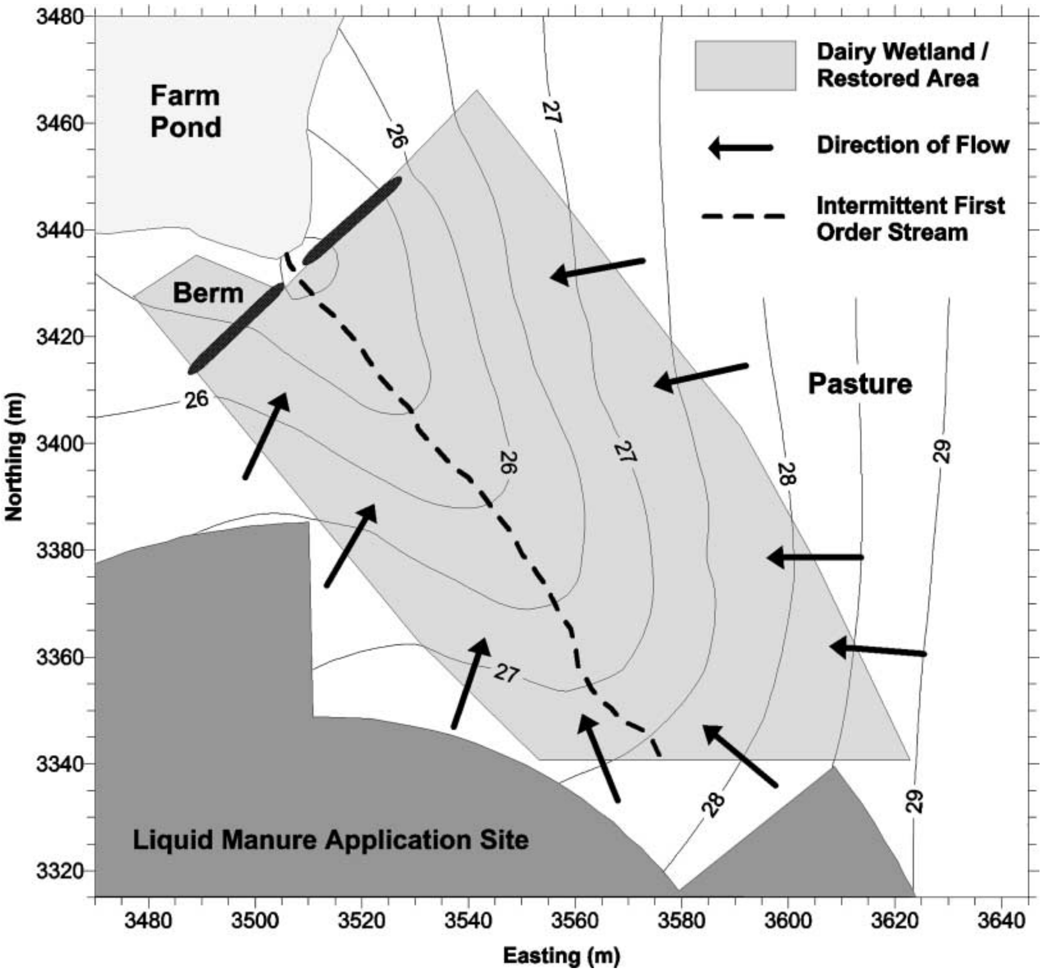

Fig. 2. Topography and hydrography of the Dairy Wetland (shaded light gray) and the surrounding uplands. Elevations are given in meters above an arbitrary reference point. Berms along the northern edge of the Dairy Wetland route surface flow to a flume at the outlet.

of silage corn into the bermudagrass and rye stubble in the spring, followed by summer crops of hay or silage from the residual bermudagrass. During the summer, the hay is harvested on a monthly schedule. The year-round sod cover minimizes erosion and reduces runoff by promoting infiltration.

A 1.5-ha section of the land application site drains downslope directly into the Dairy Wetland. Because the arc of the pivot extends over the Dairy Wetland, the last five sprinklers on the pivot, beginning with the last tower and including the overhang, were equipped with solenoid valves that are closed when the pivot traverses the site. This prevents direct application of liquid manure onto the Dairy Wetland. A narrow wooden bridge was constructed to allow the last tower of the pivot to traverse the site without affecting surface hydrology. The 1-ha upland pasture on the east side of the Dairy Wetland receives inorganic fertilizer at a rate of 300 kg N ha^{-1} yr^{-1} and 150 kg P ha^{-1} yr^{-1} as recommended by the University of Georgia Cooperative Extension Service. Surface runoff and shallow ground water from these upland areas flow into the Dairy Wetland (Fig. 2).

Dairy Wetland Hydrology

The Dairy Wetland, approximately 1 ha in size, is located on an intermittent first-order stream. The upland areas surrounding the Dairy Wetland typically have a plinthic layer at a depth of 1 to 1.5 m. As in the wetland, the plinthite acts as an aquitard and, during periods of high rainfall, is responsible for the formation of transient perched water tables. From May to December, surface runoff generally occurs during intense rainfall events. During the winter and early spring months, when the soil profile is often saturated, ground water seeps along the southern perimeter of the wetland. The seepage, in combination with frequent runoff events, results in stream flow through the wetland during the winter and early spring. The boundaries of the restored area (Fig. 2) are similar to the boundaries of the area delineated by the USDA Natural Resources Conservation Service as a wetland based on hydrology, hydric soils, and wetland vegetation. This fits the general pattern of the Coastal Plain, where all or parts of riparian forests are frequently delineated as wetlands.

Dairy Wetland Restoration

The mature riparian forest of the Dairy Wetland was logged in 1985 and replaced with a wet pasture. Ditches were dug after logging to facilitate drainage into a farm pond constructed downslope. Two smaller ditches in the headland met to form a larger ditch that carried intermittent streamflow. Over the next five years, the ditches filled in with eroded sediment.

Restoration began in February 1991, when a three-zone riparian buffer system, as prescribed by USDA Forest Service specifications (Welsch, 1991), was established by planting hardwoods in Zone 1 and slash pine in Zone 2 (Fig. 3) (Vellidis et al., 1993). Zone 1 is a 10-m-wide band of trees with mixed hardwoods in rows including swamp blackgum (*Nyssa sylvat-*

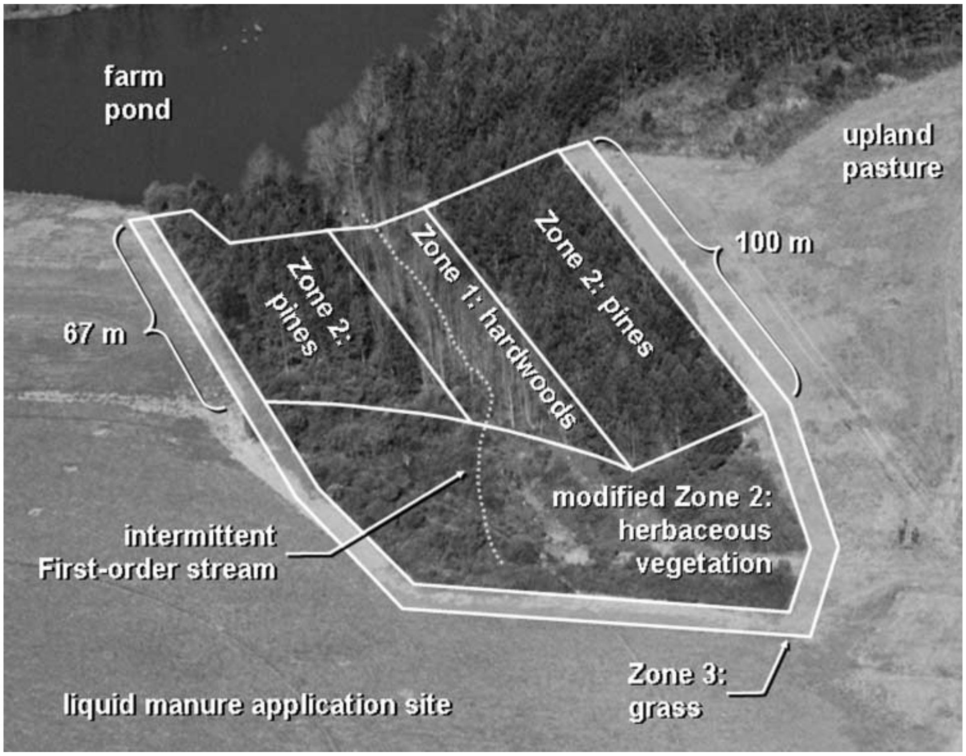

Fig. 3. Perspective view of the Dairy Wetland and the surrounding uplands showing how the three-zone riparian buffer system was implemented during restoration of the site.

ica Marshall), green ash (*Fraxinus pennsylvanica* Marshall), and yellow poplar (*Liriodendron tulipifera* L.). Zone 2 is a 20-m-wide band of slash pine (*Pinus elliottii* Engelm.) in rows. Rows were about 2 m apart. No trees were planted under the overhang of the pivot. This area is considered to be a modified Zone 2. Vegetation in the modified Zone 2 was all volunteer and consisted of a mixture of silverling (*Baccharis halimifolia* L.), black willow (*Salix nigra* Marshall), grasses (especially crowngrass [*Paspalum* spp.]), and other herbaceous species. On the west, south, and east sides of the Dairy Wetland, Zone 3 is an 8-m-wide strip of 'Tifton 44' bermudagrass. The entire three zone buffer averages 38 m in width.

Instrumentation

Instrumentation at the Dairy Wetland was initially developed to examine the fate and transport of nutrients moving downslope in surface runoff or shallow ground water flow from the upland liquid manure application site. Shallow ground water monitoring wells and surface runoff collectors were installed during 1991. In 1993, an herbicide transport study began on the eastern side of the Dairy Wetland (Vellidis et al., 2002). This resulted in the modification and expansion of the monitoring well network.

Monitoring Well Network

The original nutrient monitoring well network consisted of 63 PVC wells, 50 mm in diameter, and screened with a 0.25-mm well screen from 0.1 to 0.8 m below the soil surface. Twenty-three wells in six transects were installed on the west slope, 18 wells in three transects were installed on the south slope, and 22 wells in six transects were installed on the east slope of the Dairy Wetland. Within a transect, wells were installed 5 m apart near the upland–wetland interface and 10 m apart within the wetland. These wells were installed using the methods described by Vellidis et al. (1994). Henceforth they are referred to as "original shallow wells" (0.1–0.8 m).

Beginning in January 1993 a network of 72 monitoring wells appropriate for pesticide transport studies was established on the east slope of the Dairy Wetland, replacing the 22 original east slope nutrient wells. The network consisted of 36 shallow and 36 deep wells. Shallow wells were screened from 0.1 to 0.6 m below the soil surface while deep wells were screened from 0.6 to 2.0 m below the soil surface. Henceforth these wells are referred to as "new shallow wells" (0.1–0.6 m) and "deep wells" (0.6–2.0 m). Wells were assembled from threaded PVC pipe and screened with a 0.25-mm well screen. The wells were installed using the methods described by Lowrance et al. (1997). Six well transects were installed with each transect containing 12 wells—a shallow and deep well pair at each of six positions within the transect. The transect positions, in meters from the upslope edge of the Zone 3 grass buffer were: 0, 8 (the upslope edge of the Zone 2 managed forest buffer [pines]), 13, 18, 28 (upslope edge of the Zone 1 forest buffer [hardwoods]), and 38 m (near the stream channel) (Fig. 4).

Well water samples were collected biweekly from January 1992 through December 1997. Samples were collected monthly from January 1998 through December 1999. Before sampling each well, the depth to ground water was measured manually and one well volume was removed and discarded. Well samples were collected into chemically clean bottles with Teflon-lined caps, stored in coolers in the field, and transported to a refrigerator at the lab within 2 h of collection.

Surface Runoff Collector Network

Surface flow through the Dairy Wetland was monitored with a large H-flume at the wetland outlet and two types of runoff collectors. Paired gutter and flume collectors installed on the west and south slopes of the site were later supple-

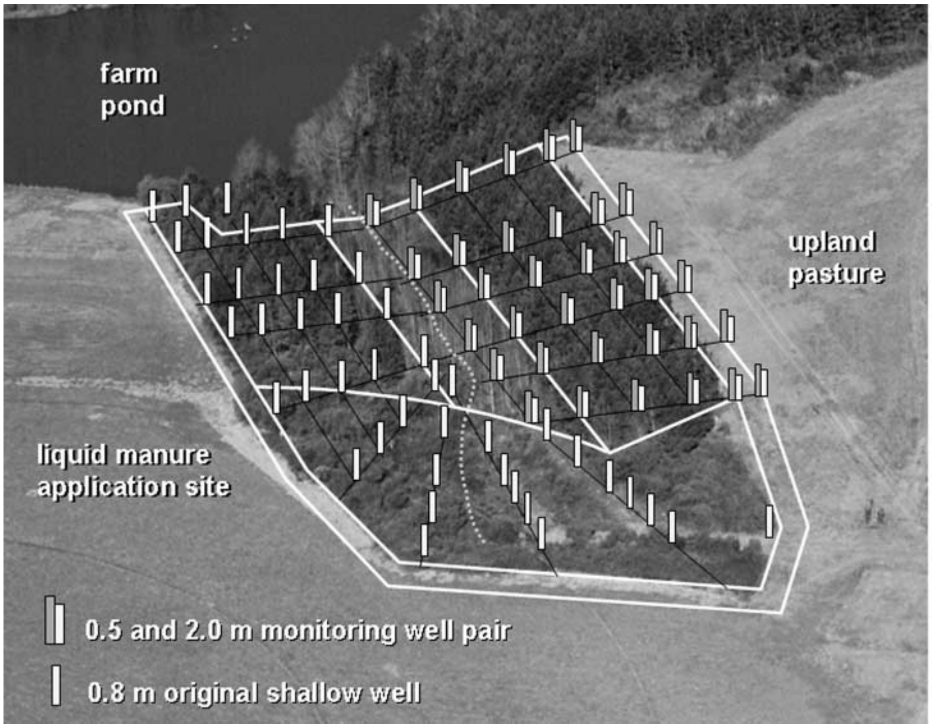

Fig. 4. Perspective view of the Dairy Wetland and the surrounding uplands showing the ground water monitoring well network consisting of 42 original shallow wells and 72 new wells (36 pairs of deep and shallow wells). The boundaries of the three zones are also shown.

mented with low-impact flow event runoff samplers installed on the east slope of the Dairy Wetland. The runoff collectors were designed to provide specific information on nutrient uptake and removal by wetland soil and vegetation types as nutrient fronts from the upland migrated through the wetland. The outlet flume provided data on the overall effectiveness of the Dairy Wetland in attenuating nonpoint source pollution in intermittent streamflow.

Gutter and Flume Runoff Collectors. In 1992, a total of four gutters were installed in pairs on the west and south slopes, with each pair consisting of an upslope and a downslope gutter. The gutters on the west slope sampled water entering and leaving the forested Zone 2—a distance of approximately 25 m. The gutters on the south slope sampled water entering and leaving the modified Zone 2—a distance of approximately 20 m. Because the upslope gutter of this pair was located within the seepage area at the southern perimeter of the Dairy Wetland, some of the overland flow it intercepted originated as seepage. Runoff was collected in a 3.6-m-long galvanized sheet metal gutter, passed through a 200-mm modified Tucson flume, and redistributed through a 3.6-m-long slotted gutter.

A 450-mm H-flume installed at the wetland outlet was used to measure surface water quality and quantity discharged into the farm pond. Tapered earthen berms approximately 10 m long were constructed on either side of the H-flume to route surface flow (Fig. 2).

The five flumes were equipped with 5-FW1 strip chart stage recorders (Belfort Instrument, Baltimore, MD) and 96-h charts. Discharge was determined by manually digitizing runoff event hydrographs and integrating beneath the resulting curves to obtain total flow volume per event. Composite water samples of runoff events were collected from the four modified Tucson flumes and the H-flume with battery-powered peristaltic pumps using the methods described by Vellidis et al. (1994). During runoff events, the pumps were switched on by electro-optic liquid level switches installed in the flume stilling wells. Multiple events in a day were not collected separately. The 13-L composite sample jars were returned to the laboratory and a well-mixed 500-mL subsample was poured from the jar into a 500-mL glass bottle for storage at 4°C until analyzed. If less than 500 mL was collected, the entire sample was stored.

Low-Impact Flow Event Runoff Samplers. During 1993, as part of the herbicide study, three transects of four 0.3-m-wide low-impact flow event (LIFE) samplers (Sheridan et al., 1996) were installed to sample surface runoff on the east slope of the Dairy Wetland (Vellidis et al., 2002). In each transect, the collectors were installed at the upslope edge of Zones 1, 2, and 3 and at the midpoint of Zone 2 (Fig. 5). Two transects had instruments that retained 10% of the collected sample. The third transect had instruments that retained 1% of the collected sample. This design ensures that a measurable volume can be collected over a wide range of runoff events. The 10% collection is made by splitting the flow into 10 pathways at the back of the collector and collecting flow from one pathway. The 1% sample is collected by connecting two 10% samples in series. The sample receptacle is large enough to contain runoff from approximately a 10-year return interval event in the 1% samplers (Vellidis et al., 2002). The samplers were positioned so as not to interfere with surface runoff collection at the next zonal interface. The three samplers at the upslope end of Zone 3 are designated Position 1; the upslope end of Zone 2 is Position 2. The middle of Zone 2 is Position 3 and the upslope end of Zone 1 is Position 4 (Fig. 5). Having two types of samplers (10 and 1%) allowed both large and small runoff events to be sampled and runoff volumes obtained. Samples from all collectors that had volumes greater than 100 mL were used for each surface runoff event.

Surface runoff sample volumes were measured and subsamples collected for analysis on the work day following each runoff event. Multiple events in a day were not collected separately. Samples were collected by pumping the receptacles with a peristaltic pump while agitating the sample by mixing with

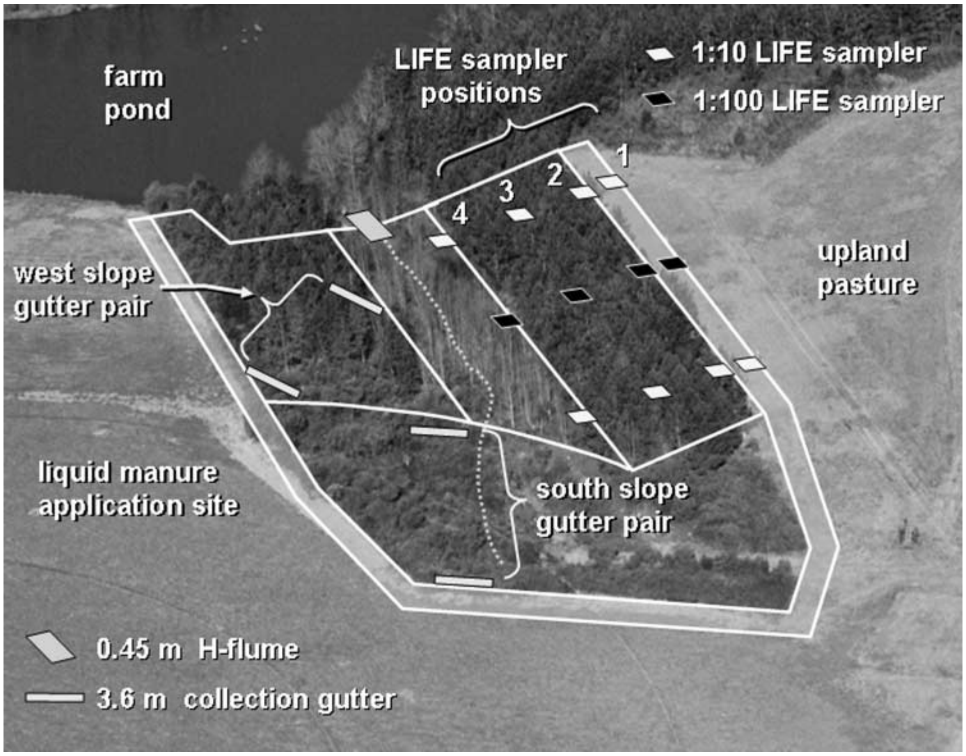

Fig. 5. Perspective view of the Dairy Wetland and the surrounding uplands showing the location of the paired 3.6-m collection gutters, the 12 low-impact flow event (LIFE) surface runoff collectors, the H-flume, and the boundaries of the three zones.

the inlet line of the pump. Samples were collected into chemically clean glass bottles fitted with Teflon-lined caps. Surface runoff samples were stored in coolers in the field and transported to lab refrigerators (4°C) within 2 h of collection.

All ground water and surface runoff samples were analyzed for NO_3–N, NH_4–N, dissolved reactive phosphorus (DRP), and Cl^- with USEPA-approved colorimetric techniques (Clesceri et al., 1998). The TKN was quantified with digestion and titration techniques adapted from USEPA-approved methods (Clesceri et al., 1998). Total P was quantified with digestion and USEPA-approved colorimetric techniques (Clesceri et al., 1998). Total N was the sum of nitrate plus TKN.

Data Analysis

For both ground water and surface runoff, data were summarized seasonally with winter consisting of data from January, February, and March; spring consisting of April, May, and June; summer consisting of July, August, and September; and autumn consisting of October, November, and December.

Shallow Ground Water

For each monitoring well, mean seasonal concentration data were developed from the entire period of record (1992–1999). These seasonal means for each of the 114 wells were used to develop nutrient surface concentration maps and loading rates.

Surface concentration maps were developed with the kriging geostatistical gridding method. Kriging is a flexible gridding method that incorporates anisotropy and underlying trends in an efficient and natural manner. Kriging attempts to express trends suggested in the data, so that, for example, high points might be connected along a ridge rather than isolated by bull's eye–type contours (Deutsch and Journel, 1992; Cressie, 1991; Journel, 1989; Isaaks and Srivastava, 1989; Journel and Huijbregts, 1978).

To quantify nitrogen attenuation in shallow ground water within the Dairy Wetland, seasonal NO_3–N loads were developed for wells along the perimeter of the wetland and for downslope wells near the stream. The loads were calculated by applying average seasonal concentrations for the wells to Darcian flows calculated through the saturated thickness of the soil profile within the effective sampling depth of the wells (Bouwer, 1978) and applied to the contributing area assigned to each well (Fig. 6). Bail tests (Freeze and Cherry, 1979) of each well conducted during autumn 1998 and winter and spring 1999, and the Hvorslev (1951) method were used to determine saturated hydraulic conductivity of the soil around each well. Hydraulic gradient was calculated using water table elevations of the perimeter wells and the downslope wells near the stream. Loads were calculated only for the autumn, winter, and spring seasons as bail tests could not be performed during the summer. Inputs to the Dairy Wetland were calculated based on the upper tiers of wells along the perimeter of the riparian zone. Ground water outputs from the wetland were calculated from the near-stream tiers of wells.

Surface Runoff

Concentration data were tested for normality using the univariate procedure in the Statistical Analysis System (SAS Institute, 1999). The concentration data were not normally distributed, nor were the log-transformed concentration data. Therefore, typical analysis of variance was not used. Instead, the NPAR1WAY procedure of SAS with the Kruskal–Wallis test was used. NPAR1WAY is a nonparametric procedure that tests whether the distribution of a variable has the same location parameter across different groups. The Kruskal–Wallis procedure tests the null hypothesis that the groups are not different from each other by testing whether the rank sums are significantly different based on a chi-square distribution

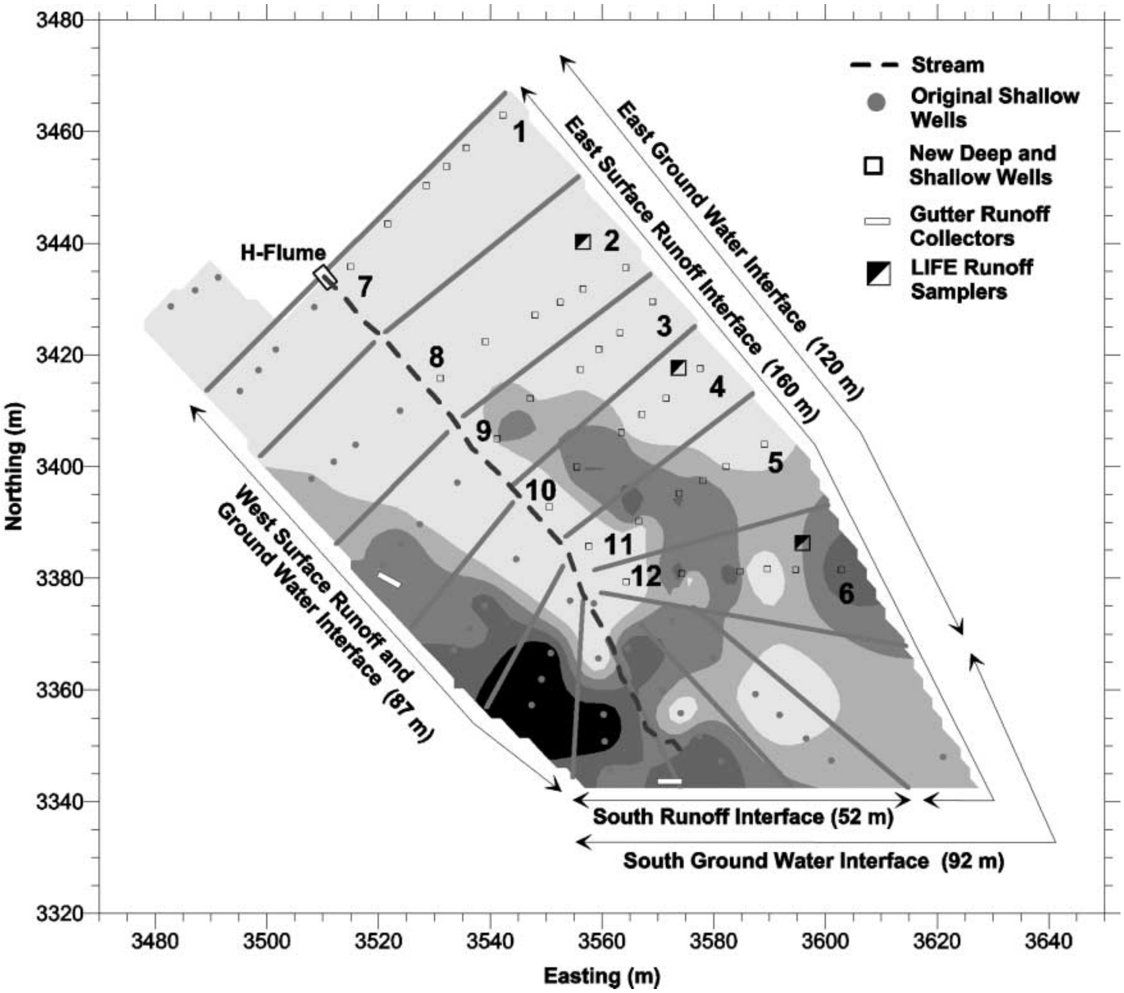

Fig. 6. Map of the Dairy Wetland showing the contributing areas assigned to each well to calculate ground water loads and the surface runoff and ground water perimeter interface lengths. The map is superimposed on a shallow ground water nitrate contour map that is discussed later.

(Sokal and Rohlf, 1981). The nonparametric tests were done on data grouped by position (distance from field) and zone. The calculated means were not subjected to a parametric analysis of variance test, therefore, means separation tests are not used.

Nutrient loads entering the Dairy Wetland in surface runoff were estimated by calculating a load per unit length of interface (g^{-1}) for the west, south, and east slopes and applying that load to the total length of the perimeter. The length of interface was 87, 52, and 160 m for the west, south, and east slopes, respectively (Fig. 6).

Nutrient loads leaving the Dairy Wetland in surface flow were calculated by applying measured concentrations to flows measured at the wetland outlet H-flume. Outputs were calculated from both storm events and baseflow with most baseflow occurring during the winter and early spring. For all runoff collectors, individual runoff event loads were calculated by multiplying concentration by volume of each measured runoff event. Annual loads were calculated by summing individual event loads during each year. Mean annual loads were determined by averaging available annual loads.

Water and Nutrient Mass Balances

Water and nutrient mass balances were calculated from entering surface runoff, entering subsurface flow, precipitation, exiting streamflow, exiting subsurface flow, and evapotranspiration. The length of record was different for the gutter and flume samplers and the LIFE samplers. Therefore, all mass balance components were available only for the 1993 to 1994 time period when the LIFE collectors were in place. Although the time periods for different components of the mass balance differ, we used mean annual loads for inputs and outputs to construct an average mass balance for the Dairy Wetland. Mean annual subsurface flow and surface runoff inputs and outputs were calculated as described above. Data from two National Atmospheric Deposition Program (NADP) network collectors (GA50 and GA99) were used to estimate precipitation amounts, nitrogen deposition, and chloride deposition for the site for 1992–1999 (http://nadp.sws.uiuc.edu/nadpdata/; verified 15 Oct. 2002). The NADP collectors are located about 3 km from the Dairy Wetland. Data collected on Little River watershed in 1979–1981 were used to estimate the total P input to the dairy wetland in precipitation (Lowrance et al., 1985). Evapotranspiration losses were estimated from earlier studies of water balance in riparian buffers on the Little River watershed (Lowrance et al., 1983). Balances were calculated by subtracting water volume or nutrient mass measured leaving the Dairy Wetland from water volume or nutrient mass measured entering the wetland. Percent retention and removal was defined as the difference divided by water volume or nutrient mass measured entering the wetland. After estimates were made for all the mass balance compo-

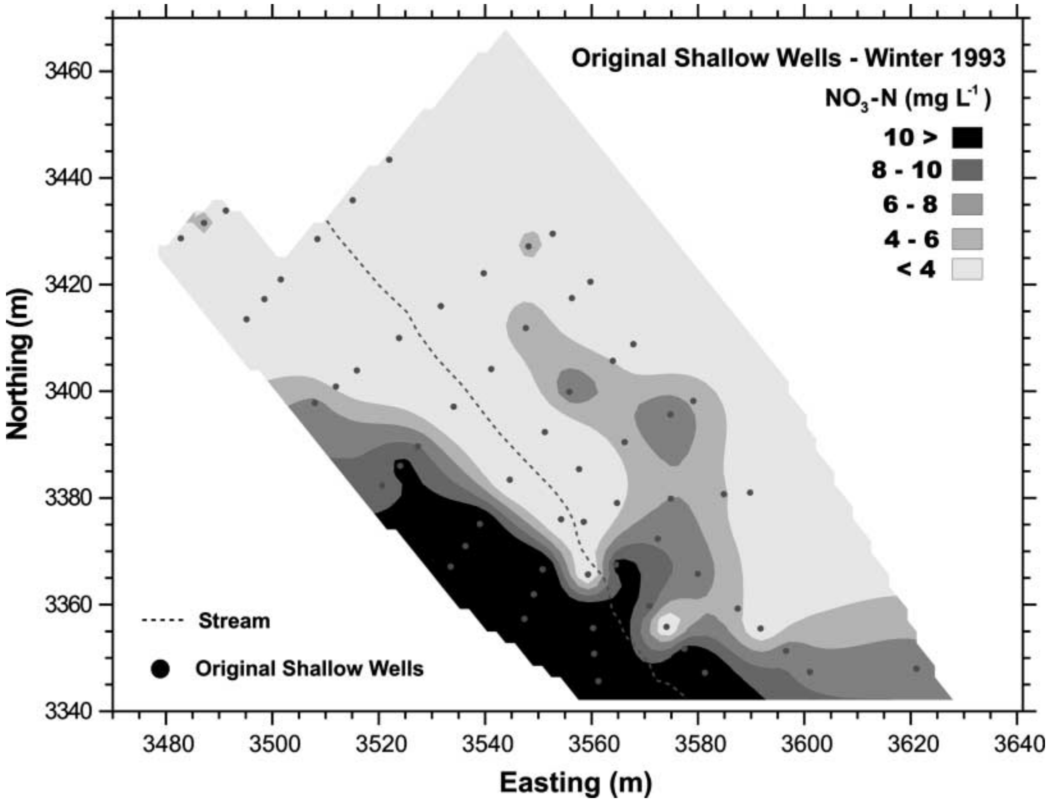

Fig. 7. Surface map of mean NO_3–N concentrations in the shallow ground water of the Dairy Wetland during winter (January–March) 1993. The data were collected with the original 63 shallow wells (0.1–0.8 m). A preferential flow path associated with past hydrologic modification allows nitrate to move much deeper into the wetland than expected. Similar maps were developed for autumn 1992 and spring 1993.

nents, the chloride (Cl) inputs and outputs were balanced by increasing Cl output. This adjustment was then applied to both water and nutrients to obtain a conservative estimate of nutrient retention and removal.

RESULTS AND DISCUSSION

Shallow Ground Water

Comparisons of mean nitrate concentrations in tiers of perimeter wells versus tiers of near-stream wells showed that the wetland was very effectively attenuating nutrient concentrations (Vellidis et al., 2001). Surface concentration maps, however, showed distinct preferential flow paths through the wetland that permitted nitrate plumes to partially bypass the attenuating capacity of the biologically active root zone. In winter 1993 there was a nitrate plume in the original shallow wells (0.1–0.8 m) that moved into the wetland along the southwest perimeter, which received subsurface and surface flow from the adjacent liquid manure land application site (Fig. 7). Along most of the plume front, nitrate was attenuated rapidly and concentrations were less than 4 mg NO_3–N L^{-1} before the stream channel (broken line). However, at the southwest corner of the wetland, a preferential flow path allows the nutrient plume to progress well into the wetland. The location of the plume coincides with the approximate location of one of the old drainage ditches. Similar nitrate maps were developed for winter and spring 1992. Summer and fall water tables were too low to sample reliably for most wells during 1991 and 1992.

After winter 1993, mean seasonal nitrate concentrations from the original shallow wells (0.1–0.8 m) and the new shallow wells (0.1–0.6 m) were combined to characterize shallow ground water (Fig. 8). As in the previous figure, higher concentrations of nitrate were found along the southwest perimeter of the Dairy Wetland and incoming concentrations were higher during winter and spring when subsurface flow was prevalent. However, the preferential flow plume seen in Fig. 8 is truncated and does not extend as far north as in Fig. 7. With the exception of the plume, and an isolated area of higher concentrations in the center of the wetland, concentrations were less than 4 mg NO_3–N L^{-1} within 20 m of the wetland perimeter.

The surface contour map of mean seasonal nitrate concentrations in the deep wells (0.6–2.0 m) (Fig. 9) shows a consistent plume of approximately the same size and shape and in the same location as for the original shallow wells (Fig. 7). This indicates that the plume may have been in ground water below the sampling depth of the new shallow wells (0.1–0.6 m). A surface contour map of combined original shallow wells (0.1–0.8 m) and deep wells (0.6–2.0 m) (Fig. 10) clearly showed the nitrate plume. The plume persisted throughout the year, and was fed from the southwest and southeast edges of the wetland—the approximate location of the old ditches.

Loads

The majority of the ground water load data are discussed in the subsequent Mass Balance section. This

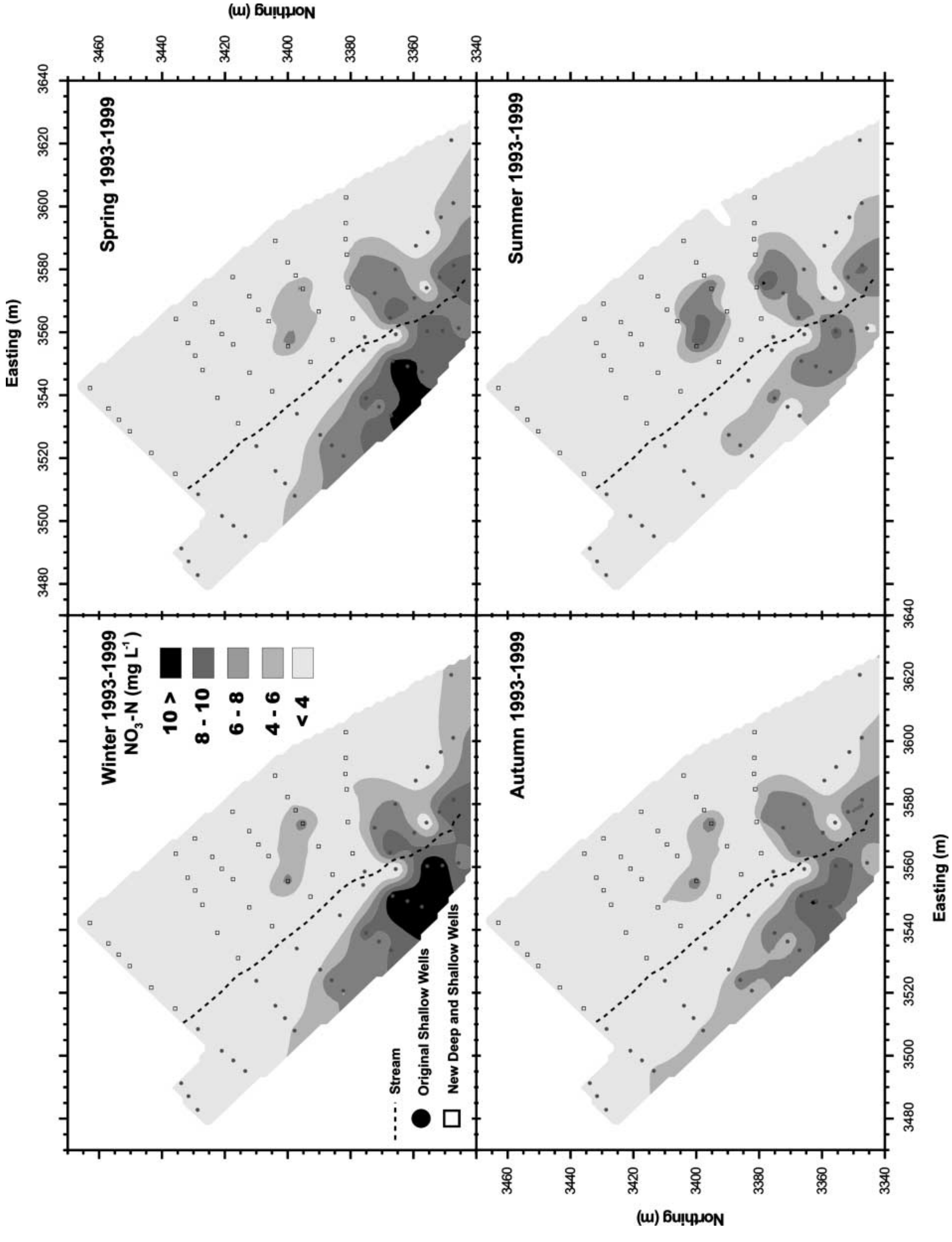

Fig. 8. Surface map of mean seasonal NO_3–N concentrations in the shallow ground water of the Dairy Wetland for the period of 1993–1999. The maps were created by pooling data from the 42 original shallow wells (0.1–0.8 m) and 36 new shallow (0.1–0.6 m) wells. The preferential flow path appears truncated.

section will instead focus on observations pertaining to preferential flow. Table 1 presents mean seasonal and annual nitrate loads in ground water sampled by the deep wells (0.6–2.0 m) in the edge-of-field and near-stream tiers of wells. Mean annual loads were estimated for winter, spring, and autumn only, because hydraulic conductivities were not available for summer. Nevertheless, annual estimates should accurately reflect actual annual loads as little shallow ground water movement occurred in the Dairy Wetland during most summers. Table 1 also presents the percent of a tier's total load attributed to each well and its contributing area (Fig. 6). Along the east slope of the wetland, 4513 g NO_3–N yr^{-1} or 77.6% of the estimated annual nitrate load entered

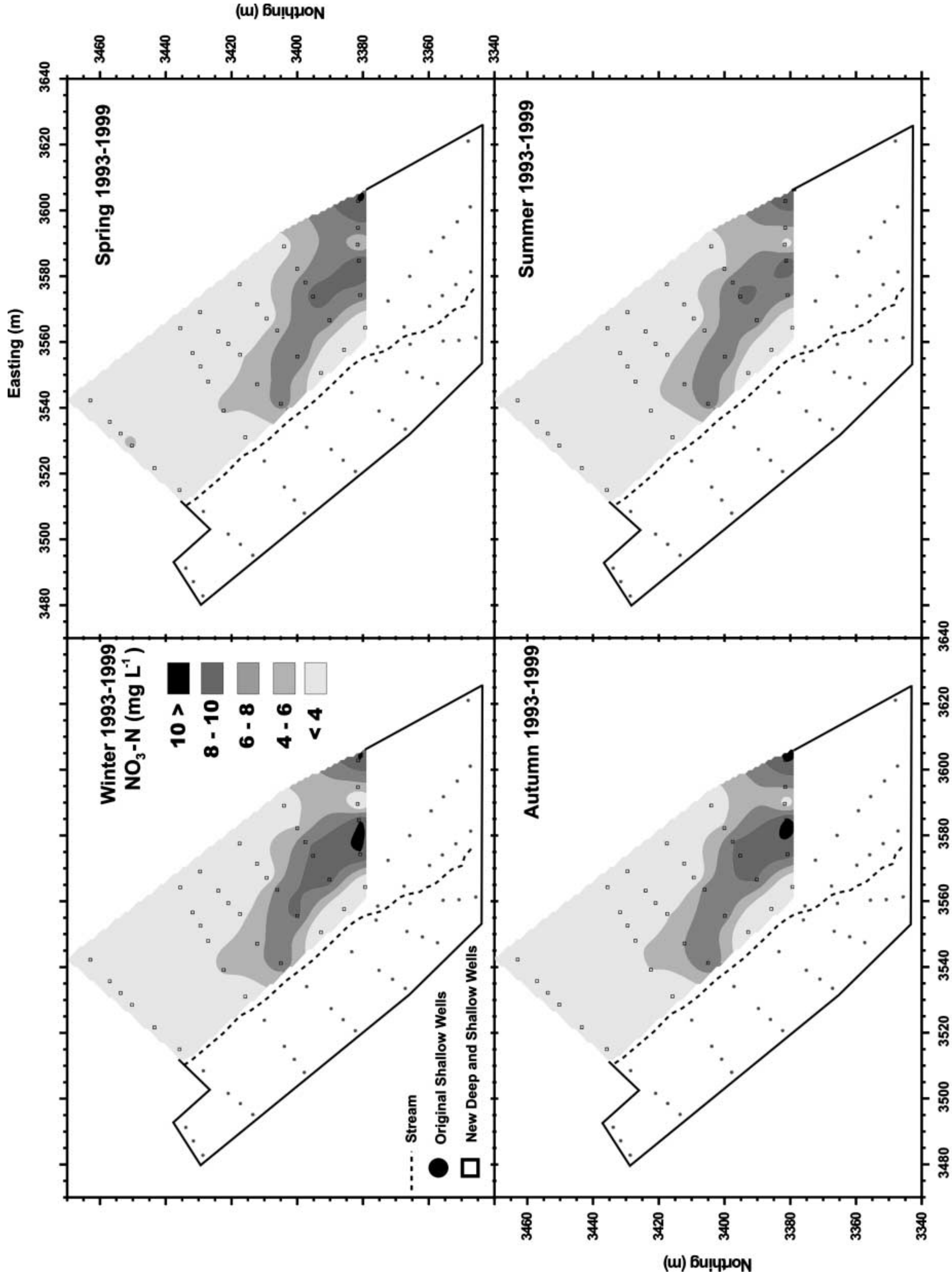

Fig. 9. Surface map of mean seasonal NO_3–N concentrations in the shallow ground water of the eastern slope of the Dairy Wetland for the period of 1993–1999. The maps were created with data from the 36 deep wells (0.6–2.0 m). A preferential flow path originating from the southeast and eventually coinciding with the preferential flow path shown in Fig. 6 is clearly seen.

the wetland from the contributing area assigned to the deeper well at Position 6 (Table 1). At the stream, 309 g NO_3–N yr^{-1} or 83.6% of the estimated annual nitrate load passed through the contributing area assigned to the deeper well at Position 9. These positions coincide with the high-concentration nitrate plume seen in Fig. 6 through 8. The corresponding new shallow well (0.1–0.6 m) load for Well Position 6 was 50 g NO_3–N yr^{-1} (25.8% of tier) and Well Position 9 was 10 g NO_3–N yr^{-1} (8.6% of tier). Combined shallow and deep load for Well Position 6 was 4563 g NO_3–N yr^{-1}, of which 98.9% moved in the 0.6- to 2.0-m section of the soil

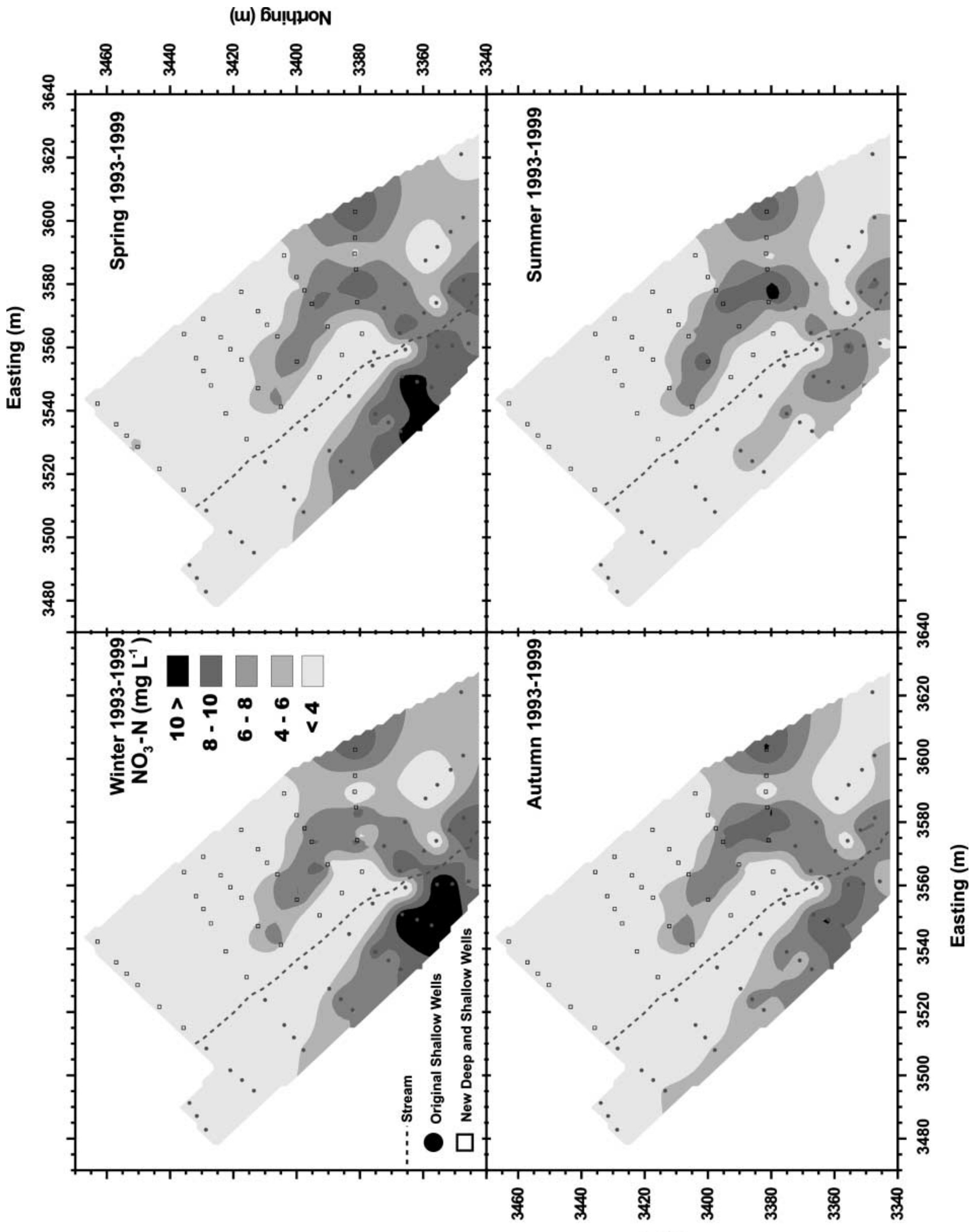

Fig. 10. Surface map of mean seasonal NO_3–N concentrations in the shallow ground water of the Dairy Wetland for the period 1993–1999. The maps were created by pooling data from the 42 original shallow wells (0.1–0.8 m) and 36 deep wells (0.6–2.0 m). This map shows preferential flow paths originating in the southwestern and southeastern areas of the Dairy Wetland.

profile. Similarly, 96.9% (319 g NO_3–N yr^{-1}) of the combined load for Well Position 9 moved in the deeper section of the soil profile.

In contrast, the highest-percent load attributed to a well-contributing area on the south and west slopes (original wells: 0.1–0.8 m) did not exceed 49%. Two wells on the southern perimeter and two wells on the western perimeter had loading rates between 1200 and 1650 g NO_3–N yr^{-1}. The remaining five wells had loading rates that ranged between 200 and 600 g NO_3–N yr^{-1}. Near-stream loadings on the south slope ranged from 24 to 470 g NO_3–N yr^{-1} while on the west slope they

Table 1. Comparison of mean seasonal NO_3–N loads in ground water between the tier of deep wells (0.6–2.0 m) at the edge of the field and the tier of deep wells (0.6–2.0 m) nearest the stream.

	Load				Load			
Well†	Winter	Spring	Autumn	Total	Winter	Spring	Autumn	Total
	g				% of total for tier			
			Deep edge-of-field wells					
1	39	40	24	103	1.6	2.1	1.7	1.8
2	34	6	5	44	1.4	0.3	0.3	0.8
3	118	137	68	323	4.9	7.0	4.7	5.6
4	216	127	123	466	9.0	6.5	8.5	8.0
5	152	133	82	367	6.3	6.8	5.7	6.3
6	1847	1518	1147	4513	76.7	77.5	79.2	77.6
Total	2406	1961	1449	5816	100	100	100	100
			Deep near-stream wells					
7	5	2	0.3	8	3.3	2.0	0.3	2.1
8	24	9	2	34	14.9	7.5	1.8	9.2
9	126	90	94	309	79.9	77.9	96.5	83.6
10	1	10	1	13	0.8	9.0	0.9	3.4
11	0.2	0.5	0.1	1	0.1	0.4	0.1	0.2
12	1	4	0.4	6	0.9	3.3	0.4	1.5
Total	157	116	98	371	100	100	100	100

† Location of wells is shown in Fig. 6.

were less than 50 g NO_3–N yr^{-1}, with the exception of Well Position 13. Here, the loading rate was 115 g NO_3–N yr^{-1}. Paradoxically, mean seasonal nitrate concentrations at this well were much less than 4 mg NO_3–N L^{-1}. Further examination of the data showed that saturated hydraulic conductivities for this contributing area were 37% higher than the other wells in the near-stream tier. Thus, even though mean concentrations were low, the transport capacity of the soil profile was high.

Surface Runoff

Mean nutrient concentrations in surface runoff as measured by the trough (west and south slopes) and dustpan (east slope) collectors and the H-flume at the Dairy Wetland outlet are presented in Table 2. Significance level based on the Kruskal–Wallis test is presented for each grouping of surface runoff collectors. Differences with a probability level greater than 0.01 are counted as no significant difference for this test.

Nitrate, total N, DRP, and total P showed consistent concentration reductions from upslope to downslope positions in the runoff troughs. Ammonium increased slightly but significantly for one trough pair and decreased significantly for the other pair. The TKN was lower in the downslope position for both the south and west troughs but the differences were not significant for the west troughs.

The LIFE samplers had less reduction of N concentrations from the field edge to the downslope position than in the runoff troughs. Nitrate and TKN were not significantly different but ammonium was significantly lower in the downslope position. Both DRP and total P were lower at Position 4 (downslope) than Position 1 (upslope) for the LIFE samplers.

This study provides the first example for a restored Coastal Plain buffer, where it was possible to compare the surface runoff concentrations from the field edge to outputs in streamflow. Most results were as expected with edge of field concentrations significantly higher than the streamflow outputs. The exception was ammonium, for which streamflow output concentrations (1.20 mg NH_4–N L^{-1}) were greater than the surface runoff input concentrations (0.96 mg NH_4–N L^{-1}).

Water and Nutrient Mass Balances

An annual water balance and nutrient mass balance of the Dairy Wetland is presented in Table 3. Water and nutrient loadings to the west and south edges of the Dairy Wetland can be attributed to the liquid manure land application site, while loadings to the east edge can be attributed to the pasture. With the exception of

Table 2. Mean nutrient concentrations in surface runoff (followed by standard error and number of observations in parentheses) as measured by the trough (west and south slopes) and low-impact flow event (LIFE) samplers (east slope) collectors and the H-flume at the Dairy Wetland outlet. Surface runoff data were analyzed for significant differences with the Kruskal–Wallis test.

	Mean nutrient concentration						
Position	NO_3–N	NH_4–N	TKN†	Total N	Cl	DRP‡	Total P
	mg L^{-1}						
			Trough collectors				
F4, south upslope	1.18 (0.16, 226)	0.96 (0.27, 223)	7.51 (0.69, 109)	8.48 (0.71, 109)	23.0 (0.90, 225)	2.00 (0.11, 225)	2.01 (0.18, 85)
F1, south downslope	0.58 (0.15, 398)	1.03 (0.26, 398)	4.94 (0.46, 202)	5.25 (0.46, 202)	23.7 (0.56, 399)	0.70 (0.08, 399)	0.94 (0.13, 140)
Significance level	<0.0001	0.0001	<0.0001	<0.0001	0.18 (NS§)	<0.0001	<0.0001
F3, west upslope	0.86 (0.18, 73)	1.19 (0.43, 72)	12.99 (2.88, 41)	13.6 (1.89, 41)	19.1 (1.71, 72)	2.14 (0.51, 73)	2.30 (0.47, 34)
F2, west downslope	0.31 (0.06, 181)	0.35 (0.09, 180)	5.95 (0.58, 113)	6.19 (0.59, 113)	27.2 (1.1, 180)	0.53 (0.13, 181)	0.33 (0.05, 88)
Significance level	<0.0001	<0.0001	0.07 (NS)	0.008	<0.0001	<0.0001	<0.0001
			LIFE samplers				
1, Field edge	1.07 (0.15, 193)	0.89 (0.20, 198)	7.70 (1.95, 102)	6.62 (1.02, 94)	14.5 (0.63, 195)	0.36 (0.05, 198)	0.76 (0.26, 102)
2, Upslope edge of Zone 2	2.32 (0.62, 138)	1.54 (0.46, 139)	6.71 (0.85, 72)	7.98 (1.07, 66)	15.6 (0.81, 139)	0.22 (0.04, 140)	0.42 (0.04, 72)
3, Middle of Zone 3	2.22 (0.42, 175)	0.89 (0.33, 178)	5.24 (1.01, 87)	6.86 (1.13, 81)	13.8 (0.59, 175)	0.07 (0.01, 178)	0.33 (0.05, 88)
4, Upslope edge of Zone 1	0.66 (0.07, 186)	0.42 (0.11, 187)	6.83 (1.24, 96)	7.37 (1.30, 91)	12.0 (0.38, 185)	0.06 (0.01, 187)	0.52 (0.11, 96)
Significance level (between Positions 1 and 4)	0.67 (NS)	0.008	0.016 (NS)	0.076 (NS)	0.0015	<0.0001	0.007
			Cumulative				
All field edge	1.09 (0.10, 492)	0.96 (0.16, 493)	8.49 (0.57, 252)	8.63 (0.71, 244)	19.1 (0.5, 492)	1.37 (0.10, 496)	1.48 (0.16, 221)
H-flume (wetland outlet)	0.50 (0.15, 223)	1.20 (0.38, 223)	3.78 (0.46, 115)	4.18 (0.47, 115)	18.8 (0.5, 222)	0.31 (0.05, 223)	0.36 (0.03, 67)
Significance level	<0.0001	<0.0001	<0.0001	<0.0001	0.07 (NS)	<0.0001	<0.0001

† Total Kjeldahl nitrogen.
‡ Dissolved reactive phosphorus.
§ Not significant.

NO_3–N, surface runoff dominates nutrient mass entering the Dairy Wetland. Entering water volumes are dominated by runoff and precipitation. The highest influx of water in surface runoff was along the south edge of the wetland, with approximately 109 m^3 yr^{-1}. This was much higher than the comparable values of 22 and 41 m^3 yr^{-1} along the west and east edges, respectively. The difference can be attributed to increased seepage and runoff resulting from the land application of liquid dairy manure onto the upland crop production area. Incoming surface runoff nitrate, DRP, and total P loads, which were at least three to five times higher along the south edge than the west and east edges, were also increased due to the large increase in water input.

Incoming ground water volumes are similar from the three sides with 7.4, 8.3, and 9.3 m^3 yr^{-1} entering from the west, south, and east, respectively. Although the original shallow wells (0.1–0.8 m) on the west and south are sampling water from about the same depth as the new shallow wells (0.1–0.6 m), nitrate loads are much higher in the west and south and more closely resemble loads from the deep wells (0.6–2.0 m). This may be explained by the fact that the wells on the west and south perimeters were in relatively lower positions in the landscape (the majority are between 26 and 28 m elevation; Fig. 2) than the perimeter wells on the east (most between 27 and 28 m). As was evidenced by the seeps along the south perimeter, the depth of the transmissive soil profile decreased in lower portions of the landscape and deeper ground water was forced closer to the surface.

Water left the Dairy Wetland primarily as surface flow through the H-flume or evapotranspiration. Ground water accounted for only 5% of water leaving (Table 3). Retention and removal rates ranged from a high of 83% for nitrate to a low of 24% for chloride. If not all outputs from the system were accounted for at the H-flume, there would be an apparent retention of water and chloride in the system. Not accounting for all outputs would lead to overestimates of nutrient retention and removal. If all inputs and outputs of chloride are estimated correctly, the mass of chloride entering the system should equal the mass of chloride leaving the system. Yet, our mass balance showed 75.3 kg of Cl retained by the system. Since our water volume balance also showed approximately the same retention rate, we attributed the difference to measurement errors at the H-flume. These errors probably occurred because of missed flows when the H-flume was submerged during a few large storm events in the summers of 1994 and 1996. To provide a

Table 3. Measured annual water balance and nutrient mass balance in the Dairy Wetland.

		Nutrients						
Budget item and interface length	**Water**	**NO_3–N**	**NH_4–N**	**TKN†**	**Total N**	**Cl**	**DRP‡**	**Total P**
	m^3 yr^{-1}	kg yr^{-1}						
		Entering Dairy Wetland						
Entering with runoff								
West edge, 87 m	**1 963**	**1.1**	**0.5**	**14.6**	**15.7**	**32.9**	**2.1**	**3.3**
South Edge, 52 m	**5 690**	**11.4**	**3.3**	**43.7**	**55.2**	**118.7**	**7.7**	**12.1**
East Edge, 160 m	**6 595**	**2.4**	**3.8**	**31.8**	**34.2**	**102.1**	**2.1**	**2.5**
Total, 299 m	**14 248**	**14.9**	**7.6**	**90.1**	**105.1**	**253.7**	**11.9**	**17.9**
Entering with ground water								
West edge, 87 m	**643**	**3.9**	**0.3**	**3.2**	**7.1**	**20.9**	**0.1**	**0.9**
South edge, 92 m	**766**	**4.1**	**0.1**	**2.3**	**6.4**	**14.4**	**0.1**	**1.6**
East edge deep, 120 m	**671**	**5.8**	**0.1**	**0.6**	**6.4**	**13.7**	**0.0**	**0.0**
East edge shallow, 120 m	**446**	**0.2**	**0.1**	**1.0**	**1.2**	**9.2**	**0.0**	**0.0**
Total, 299 m	**2 527**	**14.0**	**0.6**	**7.1**	**21.1**	**58.2**	**0.2**	**2.5**
Precipitation	**12 100**	**1.7**	**8.1**	**11.2**	**12.9**	**3.6**	**0.4**	**0.4**
Total entering	**28 900**	**30.6**	**16.3**	**108.4**	**139.1**	**315.5**	**12.5**	**20.8**
		Leaving Dairy Wetland						
Leaving with runoff								
H-flume, peaks	**5 914**	**1.3**	**2.4**	**20.7**	**22.0**	**97.9**	**1.8**	**2.3**
H-flume, base flow	**5 201**	**2.2**	**3.0**	**12.8**	**14.9**	**101.2**	**1.3**	**1.6**
Total	**11 115**	**3.5**	**5.4**	**33.5**	**36.9**	**199.1**	**3.1**	**3.9**
Leaving with ground water								
West stream, 73 m	**633**	**0.2**	**0.4**	**4.0**	**4.2**	**29.6**	**0.1**	**1.3**
South stream, 21 m	**159**	**0.9**	**0.0**	**0.5**	**1.4**	**3.5**	**0.0**	**0.2**
East stream deep, 78 m	**124**	**0.4**	**0.1**	**0.1**	**0.5**	**2.7**	**0.0**	**0.0**
East stream shallow, 78 m	**314**	**0.1**	**0.1**	**0.6**	**0.8**	**5.3**	**0.0**	
Total, 172 m	**1 230**	**1.6**	**0.6**	**5.2**	**6.9**	**41.1**	**0.1**	**1.5**
Evapotranspiration	**9 780**							
Total leaving	**22 130**	**5.1**	**6.0**	**38.7**	**43.8**	**240.2**	**3.2**	**5.4**
		Differences						
Difference§	**6 800**	**25.5**	**10.3**	**69.7**	**95.3**	**75.3**	**9.3**	**15.4**
Retention and removal, %¶	**23**	**83**	**63**	**64**	**69**	**24**	**74**	**74**
Balanced total leaving#	**29 060**	**6.7**	**7.9**	**50.8**	**57.5**	**315.5**	**4.2**	**7.1**
Balanced difference	**−160**	**23.9**	**8.4**	**57.6**	**81.6**	**0.0**	**8.3**	**13.7**
Balanced retention and removal, %	**−1**	**78**	**52**	**53**	**59**	**0**	**66**	**66**

† Total Kjeldahl nitrogen.

‡ Dissolved reactive phosphorus.

§ Defined as water volume or nutrient mass measured entering the wetland minus water volume or nutrient mass measured leaving the Dairy Wetland (input − output).

¶ Defined as difference divided by water volume or nutrient mass measured entering the wetland ([input − output]/input).

 # Water volume and nutrient mass leaving increased by 31.3% to balance Cl entering and leaving the Dairy Wetland.

more conservative estimate of the nutrient retention and removal in the Dairy Wetland we balanced the Cl budget and applied this correction to water and nutrient export through the H-flume. This correction increased the mass of water, Cl, and nutrients leaving the system by 31.3%, the amount needed to balance the Cl budget. This increase exactly balanced the chloride budget and balanced the water budget within <1% (Table 3). These conservative estimates of mass retention and removal for nitrate, ammonium, total N, DRP, and total P provide estimates of the minimum percent nutrient retention and removal by the Dairy Wetland (bottom three rows of Table 3).

Final retention and removal rates for nitrogen species ranged from a high of approximately 78% for nitrate to a low of 52% for ammonium. Final retention rates for both DRP and total P were 66%. Although measurement of all processes of N and P retention and removal was beyond the scope of this study, denitrification losses of N were measured from the site during the first two years of the Dairy Wetland restoration (April 1991–May 1993) (Lowrance et al., 1995). The average annual denitrification rate based on 2480 intact cores taken in the top 24 cm of soil was 68 kg N ha^{-1} yr^{-1}. The denitrification estimate was about 71% of the unbalanced N retention and removal and 83% of the balanced N retention and removal. The remainder of the N retention and removal and most of the P retention would be accounted for by vegetation uptake and soil storage of N and P. Although neither mechanisms of P retention nor vegetation uptake of N and P were the subject of this study, other studies provide some information on the possible mechanisms of nutrient retention. Studies of other riparian forests near the Dairy Wetland showed that N and P storage in woody vegetation would account for 63% of the N retention and 28% of the P retention found in the Dairy Wetland (Lowrance et al., 1984). Future studies of the Dairy Wetland will involve harvesting of whole trees to determine nutrient accumulation in woody biomass. Nitrate and DRP retention rates (80 and 74%, respectively) were reported for a riparian wetland receiving storm runoff from a golf course in the Sandhills region of South Carolina (Casey and Klaine, 2001). These rates are similar to the Dairy Wetland rates. In a companion study of the South Carolina riparian wetland, Casey et al. (2001) reported that denitrification was important for N removal and that soil sorption of P could account for the extensive phosphate retention found.

Mean annual concentrations of total N and total P leaving the site determined by dividing the balanced mean annual load leaving by the balanced mean annual volume leaving were 1.98 and 0.24 mg L^{-1}, respectively. These can be compared with similar calculations for the large watershed (1593 ha) that contains the Dairy Wetland (Lowrance et al., 1985). Flow-weighted concentrations for this watershed were 1.01 and 0.27 mg L^{-1}, respectively, for total N and total P (Lowrance et al., 1985). Although for different time periods, this comparison indicates that the watershed of the Dairy Wetland that includes the liquid manure application area, the adjacent fertilized pasture, and the Dairy Wetland itself has N loads and flow-weighted concentrations that are about twice that of the larger watershed and that the P loads and flow-weighted concentrations are about the same as the larger watershed.

IMPLICATIONS FOR RIPARIAN BUFFER RESTORATION

The Dairy Wetland restored forested riparian wetland was very effective in reducing both concentrations and loads of N and P derived from the surrounding agricultural land. The results demonstrate that within the first eight years following restoration, restored areas can retain large masses and high percentages of the nutrients entering. Due to its headwater location and the application of instrumentation to monitor most inputs and outputs from the Dairy Wetland, it was possible to construct more detailed and more complete water and nutrient budgets than for some previous studies. As with other studies, the amount of nitrate retained and removed or transformed is very high. Unlike some earlier studies that had less complete accounting of nutrient inputs and outputs, especially in surface runoff, the Dairy Wetland had a very high percent retention of P. Percent retention for total P was greater than percent retention and removal for total N (66% for total P, 59% for total N). Because most cropping systems and disposal areas have received manure application based on N use or removal, there is considerable concern over P transport from manure use systems. These results indicate that a wetland riparian forest buffer is effective in retaining P and that coupling such landscape features with manure land application areas can be an effective means of reducing P discharges to receiving waters. A number of factors lead to the conclusion that the mass and percent retention and removal of nutrients in this forested wetland riparian system were near the maximum for a forested riparian buffer in the Coastal Plain. Although shallow ditches were installed after the area was logged in 1986, the site was not heavily affected hydrologically. In addition, when restored to a forest and grass buffer in 1991, the site had only been out of its original forest vegetation for five to six years. Until more comprehensive mass balance numbers are available, we recommend that the percent retention and removal values from this study be used as the maximum values to expect during the first decade of restoring riparian areas also delineated as wetlands in the southeastern U.S. Coastal Plain.

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