Nitrate-nitrogen and dissolved reactive phosphorus in subsurface drainage from managed turfgrass

K.W. King, K.L. Hughes, J.C. Balogh, N.R. Fausey, and R.D. Harmel

ABSTRACT: Recent evidence suggests that turfgrass nutrients in runoff and subsurface flow pose potential risks to surface water quality. Research on water quality associated with turfgrass has generally focused on surface runoff, not subsurface flows. Quantifying the delivery of nutrients, nitrate nitrogen (NO_3-N) and dissolved reactive phosphorus, to streams from subsurface drainage features on managed turf sites, and relating the transport to fertility management and season is important for many urban managers, especially those under regulatory scrutiny. NO_3-N and dissolved reactive P concentrations from two French drains located on the Morris Williams' Municipal Golf Course in Austin, Texas, were measured over a four-year period (March, 1999 to March, 2003). Time series statistics were used to analyze and relate NO_3-N and dissolved reactive P concentrations to weather and management. A weak statistical relationship (r^2 = 0.55) was detected between discharge and NO_3-N concentration at one of the two drains. The relationship between discharge and dissolved reactive P concentration was not significant. Median NO_3-N concentrations from the two drains were 1.27 mg L^{-1} and 0.32 mg $L^{-1.}$ NO_3-N loading from the drains was 2.7 kg ha^{-1} (2.4 lb ac^{-1}). The NO_3-N concentrations and load from the turf area were approximately 10 percent of those values reported for typical row crop agriculture. Median dissolved reactive P concentrations were 0.11 mg L^{-1} and 0.09 mg L^{-1} while dissolved reactive P loading was 0.46 kg ha^{-1}. Significant (α = 0.05) seasonal tendencies were found with respect to NO_3-N and dissolved reactive P. A strong correlation was evident between the timing of peak NO_3-N losses and nitrogen application; and between the timing of peak NO_3-N losses and air temperature. There was a similar correlation between the timing of peak dissolved reactive P losses and phosphorus application. Our results suggest NO_3-N transport in subsurface drainage from this golf course is not a water quality issue. However, our findings suggest significant dissolved reactive P transport through the drains and a need for an integrated (turf, nutrients, and water) management plan that includes consideration of subsurface drainage fluxes.

Keywords: Golf course, nutrients, time series, urban, water quality

Environmentally sound management of golf course, parkland, and municipal turfgrass provides both public and private facilities with environmental, cultural, and economic benefits (Balogh et al., 1992a; Beard, 2000). Maintaining suitable turfgrass for golf courses and other recreational facilities is an important component of urban and suburban land use management. Knowledge of water and soil quality in urban watersheds is important from a regulatory and environmental perspective. Quantifying the potential for input of nutrients (nitrogen and phosphorus) to streams from subsurface drainage features on managed turf sites is important for many urban managers, especially those potentially subject to environmental regulations.

Irrigation, drainage, and nutrient requirements are integrated in the management of turfgrass on golf course facilities. Tile drains and French drains are typically used to control subsurface drainage, especially on greens and fairways with shallow water tables or fine soil texture (Pira, 1997). Tile drains are defined as "drains constructed by laying drain tile with unsealed joints in the bottom of a trench which is then refilled," whereas French drains are a "type of drain consisting of an excavated trench, refilled with pervious materials such as coarse sand, gravel, or crushed stones, through whose voids water percolates and flows toward an outlet" (ASAE Standard S526.2, 2001). Tile drainage and other subsurface drainage features are considered essential by turfgrass managers to maintain water tables at depths necessary for healthy plant growth; maintain sufficient water and air in soil void space to stimulate essential microbial activity; avoid rutting and soil compaction by maintenance equipment; and to allow site use soon after heavy rains, especially on golf courses and other recreational areas (Pira, 1997).

Subsurface drainage reduces surface runoff by increasing the subsurface movement of excess water and facilitating infiltration. Nitrogen and phosphorus transport through subsurface drainage systems may become a component of surface runoff, if the drainage water discharges directly into surface water or onto the surface offsite or downslope. Subsurface drains conveying water directly into a stream or pond will bypass natural and managed filtering processes, including upland and riparian buffer zones. Considerable research on the relationship of nutrient delivery to surface water from agricultural fields and watersheds with a significant component of subsurface drainage is available (Randall et al., 2003; Daniel et al., 1998; Sims et al., 1998). This research has demonstrated that a significant portion of nitrogen and, to a lesser extent, phosphorous movement to surface water can be directly attributed to lateral flow enhanced by subsurface drainage features.

Leaching losses of nutrients from turf have been documented (Morton et al., 1998; Petrovic 1990; and Guillard and Kopp, 2004), however, transport through subsurface flow from turfgrass systems has not been well characterized. Information is not available on the impact of turf physiology, management practices, and local weather conditions on lateral/drain hydrology and water quality.

Kevin W. King and **Norman R. Fausey** work for the U.S. Department of Agriculture Agricultural Research Service (USDA-ARS), and **Kerry L. Hughes** works for The University in Columbus, Ohio. **James C. Balogh** works for Spectrum Research Inc. in Duluth, Minnesota. **R. Daren Harmel** works for the USDA-ARS in Temple, Texas.

Reprinted from the *Journal of Soil and Water Conservation*
Volume 61, Number 1
Copyright © 2006 Soil and Water Conservation Society

The objective of this study was to quantify nitrate-nitrogen (NO_3-N) and dissolved reactive phosphorus fluxes from two French drains on a municipal golf course and relate the findings to fertility management practices and seasonal climatic patterns.

Materials and Methods

Experimental site. A sub-area of the Morris Williams' Municipal Golf Course located in Austin, Texas (Figure 1) was selected for this study. Specifically, the area of interest for this study is the 15th fairway, tee, and green (Figure 2). The primary soil in the fairway is a Travis gravelly loamy sand over sandy clay (fine, mixed, thermic Ultic Paleustalfs) with a small portion of Houston Black clay (fine, montmorillonitic, thermic Udic Haplusterts) located south of the stream (SCS, 1974). Travis soils are not as deep as the other soils in this location and have low to moderate potential for runoff. In contrast, the clayey Houston soils have a greater potential for runoff and preferential flow due to high shrink/swell characteristics.

During the study period (March, 1999 to March, 2003), management practices were typical of municipal courses in the southern United States. Fairways and greens were seeded with a hybrid bermudagrass cultivar. Greens were over-seeded in late fall with perennial ryegrass (*Lolium perenne* L.). The golf course was irrigated with a mixture of potable water from the city and water pumped from an onsite reservoir. Irrigation was applied on an "as needed" basis, determined by course personnel, to replace evapotranspiration losses.

Fertilizer was applied by both dry broadcast and spray techniques throughout the year (Figure 3) as a combination of organic, bio-stimulant, slow release, and fast release formulations. Annual average commercial fertilizer application rates for greens, fairways, and tees were determined from course records (Table 1). Average annual N application mass for the study area (1.07 ha, 2.64 ac) was 103.3 kg ha^{-1} (92.3 lb ac^{-1}), while P applications totaled 21.8 kg ha^{-1} (19.5 lb ac^{-1}). No efforts were made to quantify the mass or decomposition of grass clippings dropped back on the course after mowing.

The climate in Austin is characterized by long, hot summers and short, mild winters. Austin averages 273 growing season days per year, generally lasting from mid-March to mid-November. Thunderstorms during the summer generate short intense rainfalls. Moisture in the form of frozen precipitation can occur but is generally negligible. The 30-year normal precipitation (Figure 4) is 810 mm (NOAA, 1993). Normal daily temperatures range from an average minimum of 4°C (39°F) in January to an average maximum of 35°C (95°F) in August.

Data collection. The 15th hole (Figure 2) was divided into two segments based on surface topography. The two sites were designated as Site 3 and Site 4 (Figure 2). Site 3 drained the fairway (0.28 ha, 0.69 ac) south of the stream and green (0.05 ha, 0.12 ac) of hole number 15 (total area = 0.33 ha, 0.82 ac). Site 4 drained the fairway (0.72 ha, 1.78 ac) north of the stream and tee (0.02 ha, 0.05 ac) area of hole number 15 (total area = 0.74

Figure 1
Layout of Morris Williams Municipal Golf Course.

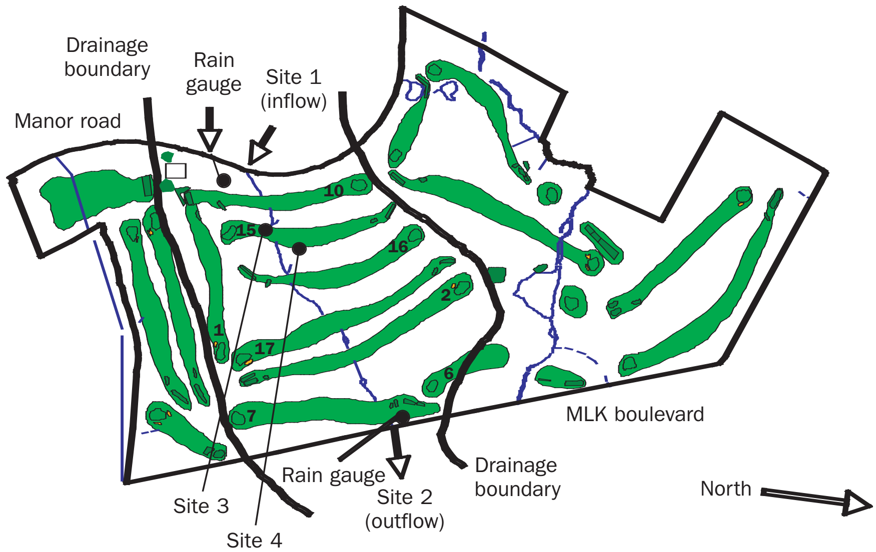


Figure 2
Drainage network and sampling sites for the 15th hole at Morris Williams Municipal Golf Course. A culvert connects the two stream segments.

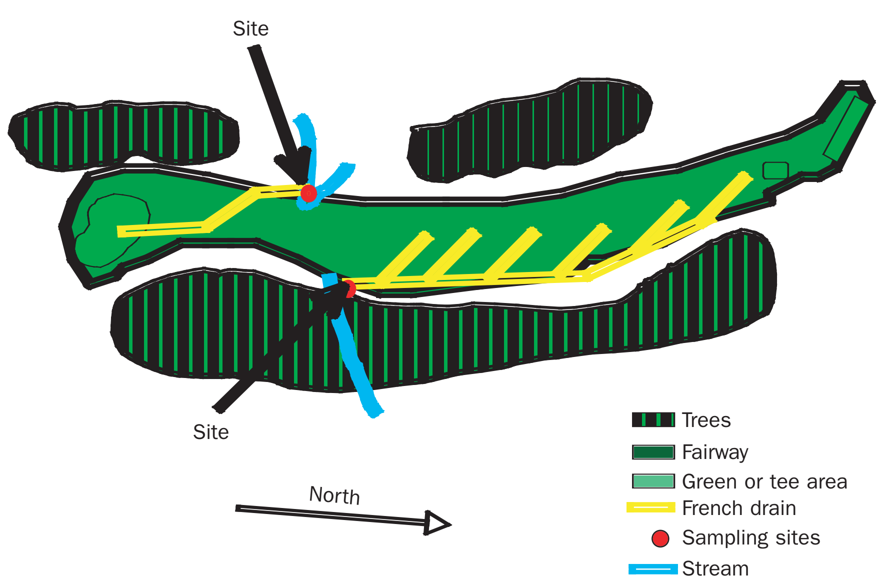


Figure 3
Four year variation of commercial nitrogen (a) and phosphorus (b) fertilizer application at Morris Williams Municipal Golf Course. Fertilizer application data was totaled by month and averaged over the study area. Gray bars represent 25th and 75th percentiles, line represents the median of the data points in each month.

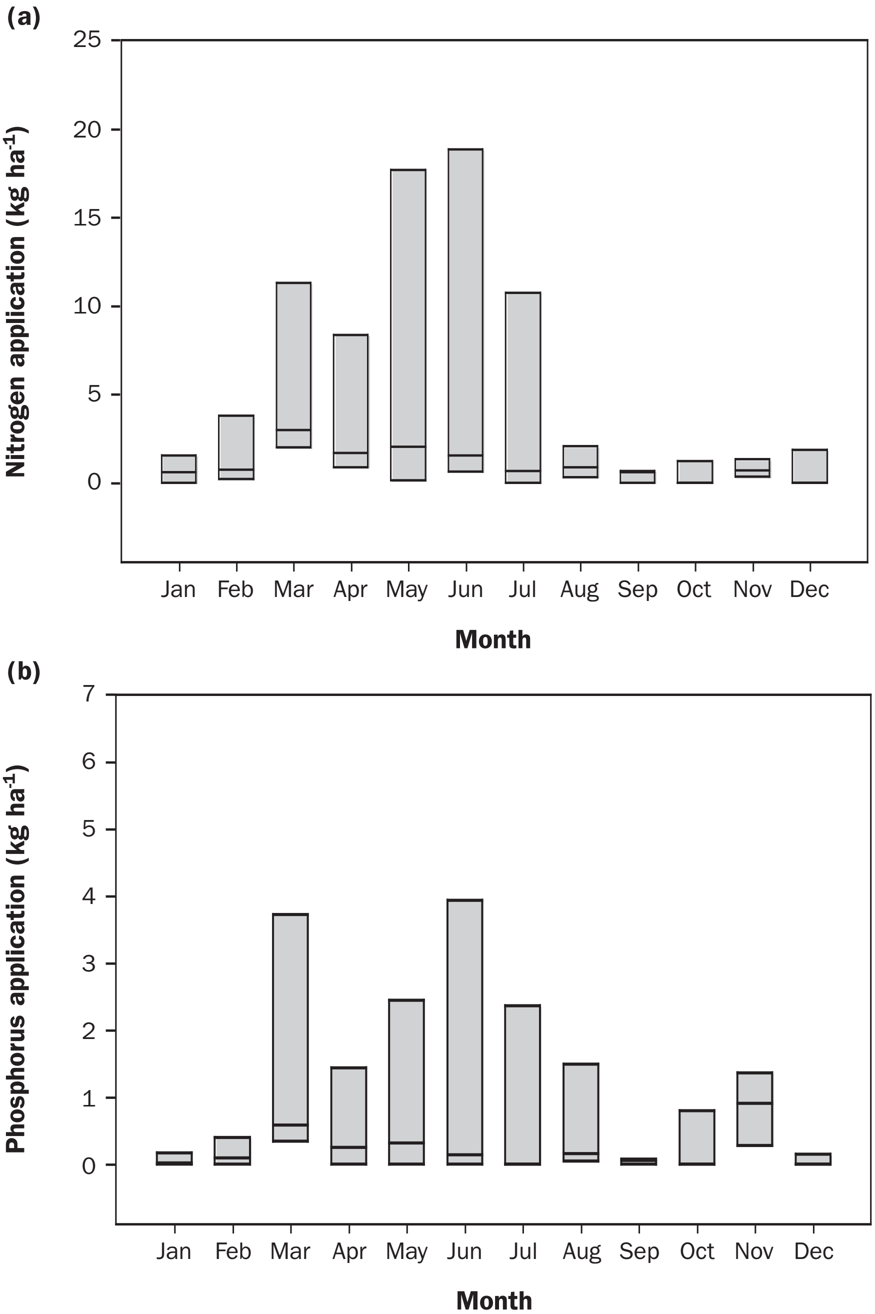


Table 1. Reported annual average commercial fertilizer application at Morris Williams Municipal Golf Course.

	Nitrogen (kg ha^{-1})	Phosphorus (kg ha^{-1})
Greens	407.6	133.2
Tees	139.0	32.0
Fairways	87.4	16.0

ha, 1.83 ac). The French drains were 0.6 m (2 ft) deep and 0.3 m (1 ft) wide. The fill material was 'pea' gravel overlaid by approximately 0.15 m (0.5 ft) of sod. Nutrient concentrations in the drainage water from the two sites were measured daily from March, 1999 to March, 2003 using automated samplers programmed to collect one sample every 24-hours. Weekly sampling from subsurface drainage has been shown to produce a 92 percent probability of capturing the pollutant mass to within ± 15 percent (Wang et al., 2003). Quantifying subsurface flow from the French drains at the sampling sites was accomplished by forcing the drainage water through a sharp crested V-notch weir (Figure 5). Water level above the bottom of the V-notch (water quantity) was recorded at 15-minute intervals for a period of 13-months at Site 3 and two-years at Site 4 using ISCO 730 bubbler module technology. Bubbler module failure and cost prohibited a longer period of discharge sampling. Precipitation was recorded with tipping bucket rain gauges located at the inflow and outflow of the course (Figure 1). Periodic soil sampling of the area was not possible because of destruction and play interruption.

All samples were analyzed colorimetrically for NO_3+NO_2-N (hereafter referred to as NO_3-N) and PO_4-P (hereafter referred to as DRP) concentrations using a Technicon Autoanalyzer IIC and methods published by Technicon Industrial Systems (1973a; 1973b; 1976). The samples were unfiltered and non-digested. Sediment in the collected samples was negligible.

Statistical analysis. Concentration data for NO_3-N and dissolved reactive P were evaluated using Minitab® (1998). The normality of the data was investigated using the Anderson-Darling statistic. Data that were not normally distributed, were compared using the Mann Whitney nonparametric test (H_0: median (x_1) = median (x_2); H_A: median (x_1) ≠ median (x_2)). Spearman's rank correlation coefficient (α = 0.05) was calculated for data from Site 3 and Site 4 for each chemical component to assess whether peak concentration at one sampling site was associated with peak concentrations at the other sampling location (H_0: x_1 and x_2 are correlated; H_A: x_1 and x_2 are uncorrelated).

Univariate time series analysis of the NO_3-N and dissolved reactive P concentration data from Site 3 and Site 4 was also conducted. Autocorrelation analyses (determining the correlation between observations within a single time series, separated by different time lags) of the data were conducted, and the significance of the lagged autocorrelations was tested using the Ljung-Box Q statistic (H_0: the autocorrelations for all lags up to k lags are zero, H_A: at least one autocorrelation is not equal to zero (Cromwell et al., 1994a). This analysis indicated whether the data was random or nonrandom. Nonrandom data is an indication that either a periodic cycle is present and/or an increasing or decreasing trend exists. The statistical significance of trends in the time series data was tested using the Mann-Kendall nonparametric test for trends (H_0: no trend; H_A: data follows a trend) (Salas, 1993). Monthly averages specific to each site were compared using the multiple range test (H_0: mean (x_1) = mean (x_2); H_A: mean (x_1) $\neq$ mean (x_2)). Cross correlation (up to lag = 12) analyses between univariate time series was completed to test for significance using the Pierce-Haugh test (H_0: x(t) does not cause y(t); H_A: x(t) does cause y(t)) (Cromwell et al., 1994b; Chatfield, 1989).

Figure 4
Four year monthly variation in measured precipitation (box plots) and normal precipitation (line) for study area. Gray bars represent 25th and 75th percentiles, line represents the median of the data points in each month.

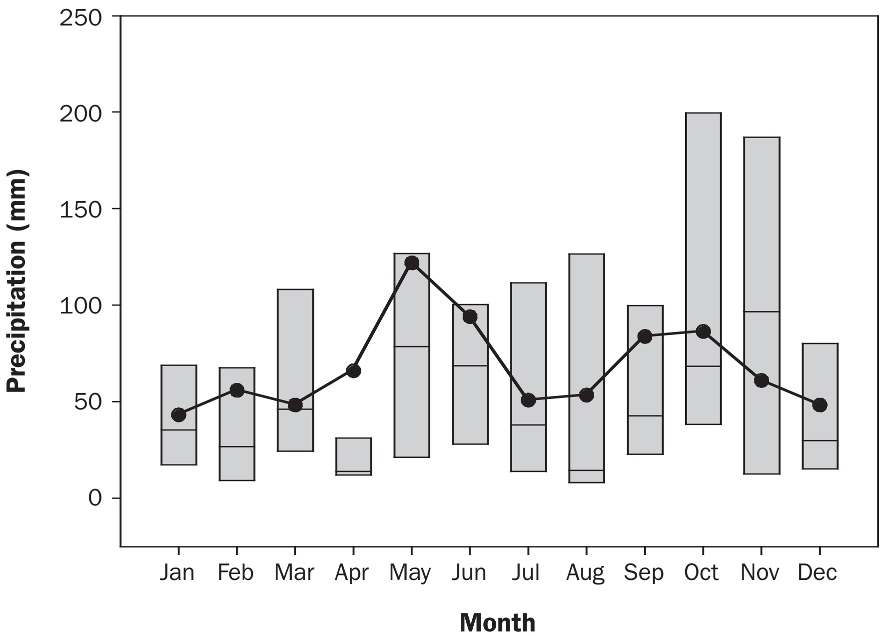
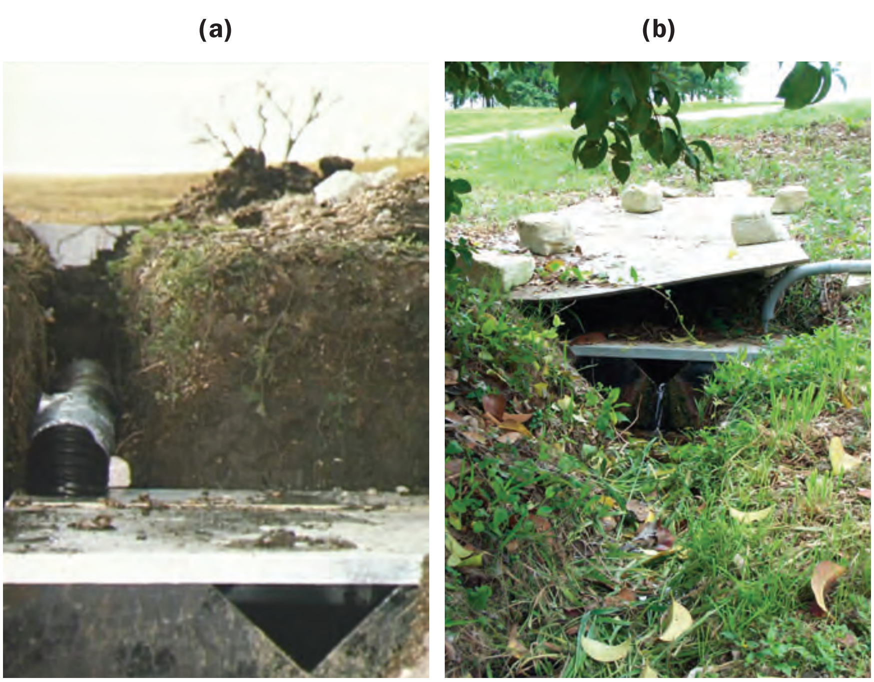


Figure 5
V-notch weirs installed at a) Site 3 and b) Site 4.



Results and Discussion

Concentrations. NO_3-N and dissolved reactive P concentrations recorded at Site 3 and Site 4 were not normally distributed (α < 0.05), therefore, median concentrations will be discussed. For the four-year study period, measured median NO_3-N concentration at Site 3 was 1.27 mg L^{-1} while median dissolved reactive P concentration was 0.11 mg L^{-1} (Table 2). Measured median concentrations at Site 4 were 0.32 mg L^{-1} NO_3-N and 0.09 mg L^{-1} dissolved reactive P (Table 2). NO_3-N and dissolved reactive P concentrations from Site 3 were significantly (α = 0.05) greater than concentrations detected at Site 4. Greater NO_3-N and dissolved reactive P concentrations measured at Site 3 are indicative of greater and more frequent fertilizer applications to greens compared to fairways. There was a weak relationship (r^2 = 0.55) between daily discharge and NO_3-N concentration at Site 4; however, no relationship was detected for dissolved reactive P and discharge at Site 4. A similar analysis conducted for Site 3 showed no relationship between drainage discharge and NO_3-N or dissolved reactive P. Concentrations of NO_3-N and dissolved reactive P at Site 3 and Site 4 were generally positively correlated through time (Spearman's rho, α = 0.05). This correlation suggests that weather factors influence the timing of nutrient movement in the French drain (constant across the experimental site), although the magnitude of concentration in the drainage water may have been influenced by management (different for each sampling site).

The greatest NO_3-N concentration recorded in the drainage water was 3.94 mg L^{-1} (August 10, 2000) at Site 3 and 3.07 mg L^{-1} (July 4, 2002) at Site 4. The measured concentrations are comparable to previous

Table 2. Statistical distribution of measured daily nutrient concentrations (mg L^{-1}) in lateral flow.

	Site 3 Lateral flow concentrations (mg L^{-1}) (n = 1339)	
	NO_3-N	Dissolved reactive P
25th percentile	0.687	0.09
Median	1.27	0.11
75th percentile	1.58	0.15
Maximum	3.94	0.99
Mean	1.15	0.13
Variance	0.42	0.007
Standard skewness	-1.02	46.2
Standard kurtosis	-3.65	142
	Site 4 Lateral flow concentrations (mg L^{-1}) (n = 1461)	
	NO_3-N	Dissolved reactive P
25th percentile	0.20	0.07
Median	0.32	0.09
75th percentile	0.64	0.11
Maximum	3.07	0.62
Mean	0.47	0.09
Variance	0.18	0.003
Standard skewness	28.1	46.1
Standard kurtosis	30.6	129

studies conducted on predominately grassed systems (Mitchell et al., 2000; Randall et al., 1997). Mitchell et al. (2000) reported mean NO_3-N concentrations in a tile draining a permanently grassed area was 1.0 mg L^{-1}, while Randall et al. (1997) found flow weighted average NO_3-N concentrations in subsurface drainage water for alfalfa and grass of 2 mg L^{-1}. In contrast, peak NO_3-N values in agricultural drainage water are often reported to exceed 10 mg L^{-1} (Fenelon and Moore, 1998; Mitchell et al., 2000; Tomer et al., 2003; Randall et al., 2003). The greater concentrations measured in agriculture drainage water was attributed to larger fertilizer amounts and less frequent applications.

The greatest dissolved reactive P concentration recorded during the period of study was 0.99 mg L^{-1} (November 21, 2001) at Site 3 and 0.62 mg L^{-1} (September 7, 2002) at Site 4. Eutrophication may occur with dissolved reactive P concentrations of 0.01 mg L^{-1} and total phosphorus concentrations of 0.02 mg L^{-1} (Sharpley and Rekolainen, 1997). Total phosphorus (dissolved reactive P plus sediment bound P) was not analyzed in this study because the source of water was subsurface drainage with negligible sediment. The median dissolved reactive P values measured during this study are comparable to results reported for subsurface drainage water from corn (0.005-0.1 mg L^{-1}) (Kladivko et al., 1991). Movement of phosphorus through the soil (vertical and lateral) is generally not considered a problem (Gilliam et al., 1985); however, on sand and sandy loam textured soils (often found on greens) the potential for phosphorus to move increases (Mansell et al., 1985; Reddy et al., 1978). Phosphorus has also been shown to move into tile drainage through macropores and/or when leaching conditions are favorable (Geohring et al., 2001). dissolved reactive P concentrations up to 3 mg L^{-1} have been reported in leachate from golf green lysimeters (Shuman et al., 2000).

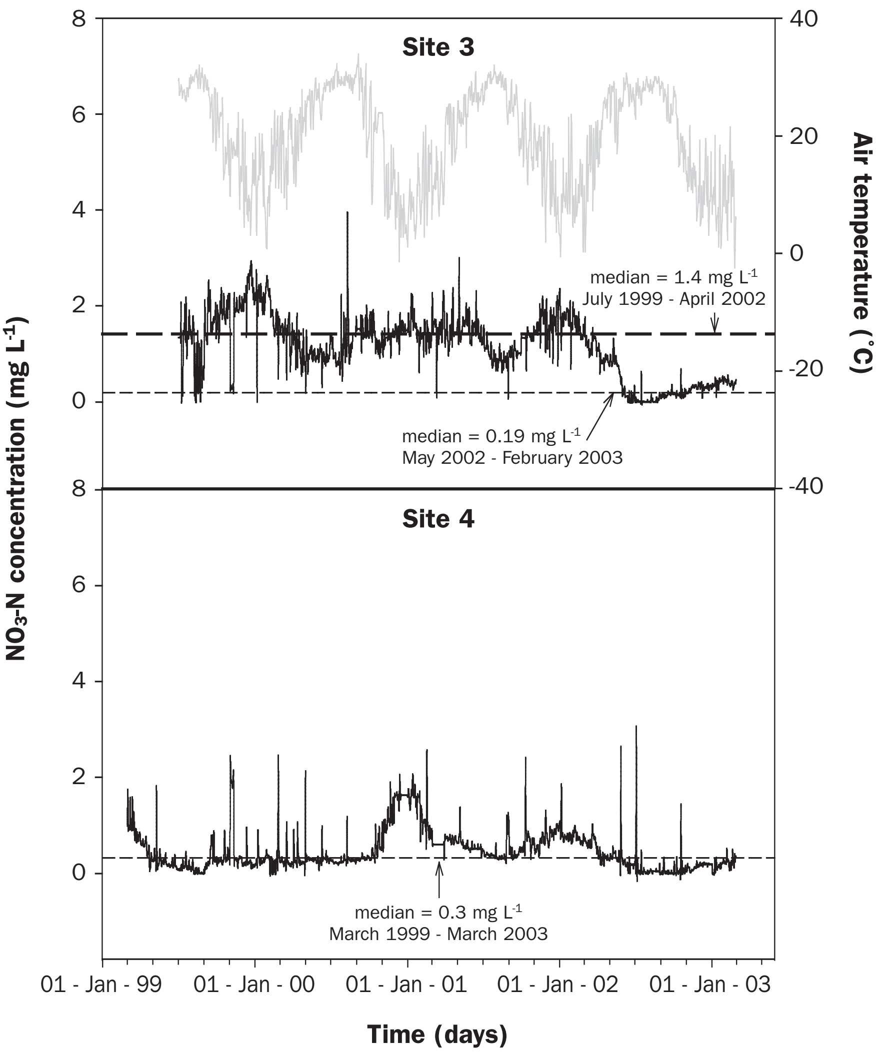
Nutrient losses. Based on flow measurements (13-months at Site 3 and two-years at Site 4), median daily flow at Site 3 was 0.6 m^3 (1.8 m^3 ha^{-1}; 0.2 mm volumetric depth), while median daily flow at Site 4 was 8.2 m^3 (11.1 m^3 ha^{-1}; 1.4 mm volumetric depth). No significant relationship between monthly discharge from the French drains and precipitation (Figure 4) was detected (Site 3 r^2 = 0.25; Site 4 r^2 = 0.13). Flow from the French drains was continuous throughout the year, most likely a result of irrigation (Table 3). The difference in flow magnitudes at Site 3 and Site 4 could result from differences in topography, contributing area, length and depth of drain, and/or seepage from an unknown source contributing to Site 4 (although a hydrogeologic survey of the site was not conducted). In the case of seepage from an unknown source contributing to Site 4, measured concentrations at that site may also be impacted. The measured nutrient load attributed to the course at Site 4 may be either over- or under-estimated depending on the volume and nutrient concentration of the seepage water. The estimated average annual combined load of NO_3-N in the drainage water associated with Site 3 (0.77 kg ha^{-1}) and Site 4 (1.92 kg ha^{-1}, 1.7 lb ac^{-1}) was 2.7 kg ha^{-1} (2.4 lb ac^{-1}) (~~2.6~~ percent of the amount applied on the study area). This amount is comparable to, but less than, the value of 3.8 kg ha^{-1} yr^{-1} (3.4 lb ac^{-1} yr^{-1}) reported by Mitchell et al. (2000) on a grass system in Illinois and the value of 10.7 kg ha^{-1} yr^{-1} (9.6 lb ac^{-1} yr^{-1})documented by Ruz-Jerez et al. (1995) for intensively managed ryegrass in New Zealand. In contrast, the average NO_3-N loading from corn and corn/soybean crop production systems is reported to be in the range of 5 to 100 kg ha^{-1} yr^{-1} (4.5 to 89.3 lb ac^{-1} yr^{-1}) (Kladivko et al., 1991; Kladivko et al., 1999; Mitchell et al., 2000; Gentry et al., 2000; Randall et al., 2003).

The estimated average annual combined dissolved reactive P load transported through the French drains at Site 3 (0.08 kg ha^{-1} (0.7 lb ac^{-1}) and Site 4 (0.38 kg ha^{-1}) was 0.46 kg ha^{-1} (an amount equivalent to 2.1 percent of the applied). This amount is considerably greater than loadings recorded from drainage water on a corn production system (0.04 kg ha^{-1} yr^{-1}, 0.04 lb ac^{-1} yr^{-1}; Kladivko et al., 1991). Dissolved reactive P losses in subsurface drainage water can be substantial when conditions for leaching are favorable or promoted or when preferential flow is present (Duxbury and Perverly, 1978; Geohring et al.,

Table 3. Average annual amounts and measured nitrate-nitrogen (NO_3-N) and dissolved reactive phosphorus (P) concentrations (n = 212) for irrigation water.

	Irrigation (mm)	**NO_3-N (mg L^{-1})**	**Dissolved reactive P (mg L^{-1})**
Greens	1040		
Tees	815	0.18	0.05
Fairways	455		

Figure 6
Time series plot of NO_3-N concentration and air temperature data for the study period at Site 3 and Site 4.



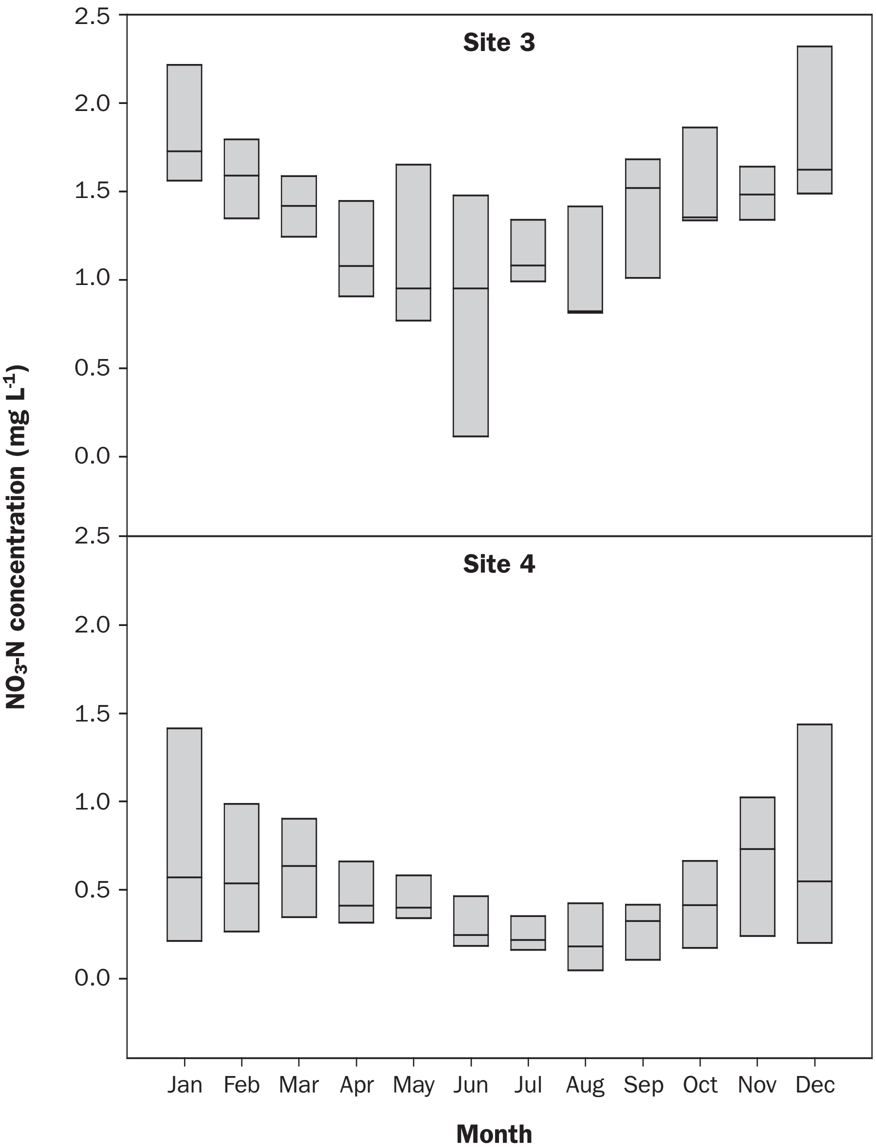
2001; Jamieson et al., 2003). The Houston Black soil present in this study area is susceptible to preferential flow (Allen et al., 2005) and may explain the greater transport of dissolved reactive P. In addition leaching can be substantial in sandy soils like those found in the green.

Time series analysis. **NO_3-N.** Autocorrelation of daily and monthly NO_3-N data for Site 3 and Site 4 showed a statistically significant (α = 0.05) nonrandom relation (indication that a periodic cycle and/or increasing or decreasing trends are present) between NO_3-N concentration and time. No significant trends (α = 0.05) were detected in the NO_3-N concentration data at Site 3 or Site 4, although at the end of the data collection period (May, 2002 to March, 2003) there appeared to be a decrease in the amount of NO_3-N detected in the subsurface drainage water from Site 3 (Figure 6). Site 3 median NO_3-N concentration measured between July, 1999 and April, 2002 (1.4 mg L^{-1}) was significantly (α = 0.05) different from the median measured concentration between May, 2002 and March, 2003 (0.19 mg L^{-1}). No explanation for the difference was apparent, however, it should be noted that 2002 precipitation (492 mm) was less than the average annual rainfall for the site (810 mm) (Figure 4).

An annual cycle was detected for NO_3-N concentrations in the subsurface drainage water at both Sites 3 and 4. Monthly variation of the NO_3-N data (Figure 7) showed that the NO_3-N concentration peaked during the months of October to March (Site 3 and 4). Comparison of the monthly means using multiple range analysis confirmed that significantly (α = 0.05) higher NO_3-N concentrations were detected during the months of December and January when compared to the spring and summer (April to August).

Seasonal trends in NO_3-N concentration were attributed to temperature and plant physiology. Hydrology would generally be expected to be related to concentration, however, as previously noted only a weak relationship between discharge and concentration was detected at Site 4. The managed grasses are primarily a hybrid bermudagrass cultivar. Bermudagrass has an optimal growth temperature between 27 and 38°C (80°F and 100°F) (Beard, 1973; Troughton, 1957) and often becomes dormant when temperatures above 15°C (59°F) are not sustained (DiPaola and Beard, 1992). During the fall and winter months (periods of peak NO_3-N concentration) ambient air temperatures are often below this dormancy threshold temperature point. Only the greens are overseeded with a non-dormant ryegrass, therefore any residual/excess fertilizer present on fairway or tee areas will not be utilized by the turfgrass and becomes available for leaching or lateral flow. Fast release fertilizers were generally applied in the green up phases (early spring) while slow release formulations were applied in the summer and early fall. Compounding the dormancy of the grass is the reduced microbial activity (less N sequestration), also a result of lower temperatures. The seasonal pattern of NO_3-N concentrations correlates well with ambient air temperatures (Figure 6). This seasonal pattern was confirmed using cross correlation analysis; the greatest cross correlation between NO_3-N concentration at Site 3 and temperature was negative and occurred at a lag of one month and the greatest cross correlation for NO_3-N concentration at Site 4 and temperature was negative and occurred at zero lag (a negative sign indicates that concentration is high when temperature is low, a zero lag indicates that low temperature and high concentrations occur during the same month and a lag of one indicates that low temperature precedes high concentration by one month). This analysis indicated that temperature was a predictor of current NO_3-N concentration in the subsurface drainage water at Site 4 while at Site 3, temperature was a predictor of the NO_3-N concentration in the subsurface

Figure 7
Monthly variation in NO_3-N concentration from four years of data at Site 3 and Site 4. Gray bars represent 25th and 75th percentiles, line represents the median of the data points in each month.



drainage water during the following month. The seasonal tendencies are similar to results found in row crop production agriculture (Kladivko et al., 1999; Mitchell et al., 2000; Randall et al., 2003).

Nitrogen application to Site 3 and Site 4 occurs throughout the year (Figure 3), with most nitrogen applied from March to June. Cross correlation between monthly NO_3-N losses and nitrogen application showed that NO_3-N in the subsurface drainage peaked seven months after the spring application (Figure 7). Because of the dynamic nature and mobility of nitrogen, it is unlikely that the peak losses during winter are related to the bulk applications during the spring. The losses are more likely a result of natural factors and management. The findings suggest that NO_3-N losses are a result of natural factors such as temperature (reduced microbial activity during the winter) and plant uptake (dormancy during the winter) in combination with management (nutrient application during dormant periods).

Dissolved reactive phosphorus. A time series plot of the dissolved reactive P concentration data from Site 3 and 4 is shown in Figure 8. Autocorrelation analysis of daily dissolved reactive P data for Site 3 and Site 4 showed a statistically significant (α = 0.05) nonrandom relation between dissolved reactive P concentration and time. An annual cycle was detected for dissolved reactive P concentration in the subsurface drainage water at both Site 3 and 4. The dissolved reactive P concentration was greater during the months of August to November (Site 3) (Figure 9) and August to December (Site 4). Comparison of the monthly means using multiple range analysis confirmed that significantly (α = 0.05) more dissolved reactive P was detected at Site 3 during the months of October and November than in the winter and spring (February to May). Similar results were confirmed using multiple range tests for Site 4. However, no significant cross correlations were found between dissolved reactive P concentrations in the subsurface drainage water and temperature or precipitation.

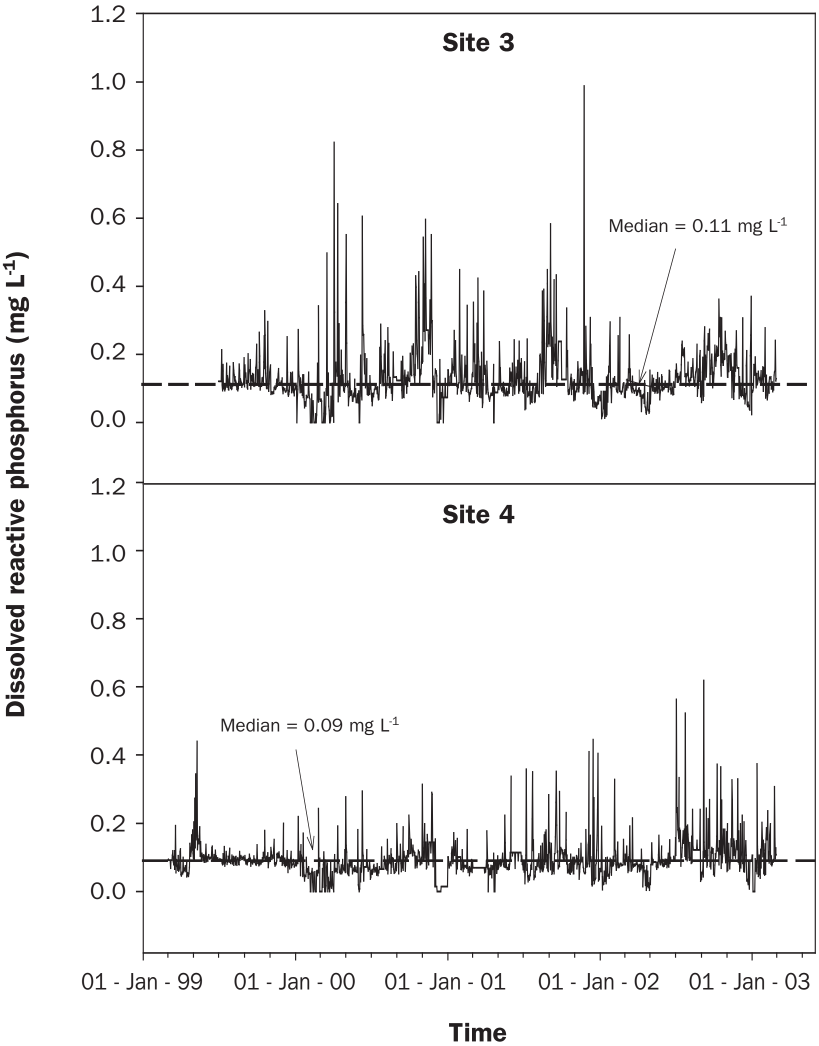
Most of the phosphorus application to Site 3 and Site 4 occurred between March and August, during the active growing season of the grasses on the fairway. The timing of dissolved reactive P transport through the French drains was significantly cross-correlated to the application of phosphorus fertilizer (α = 0.05). No relationship between concentration and discharge was detected (Site 3 r^2 = 0.04; Site 4 r^2 = 0.03). The greatest cross correlations occurred with phosphorus application at a lag of 6 months, indicating that dissolved reactive P movement occurs primarily after the growing season. One driver for dissolved reactive P movement appears to be plant physiology. During periods of green up and growth, the turf is utilizing phosphorus, thus reducing the amount available for transport. Historically golf courses apply phosphorus regularly to establish and maintain root density and vigor. Periodic soil testing would likely confirm dissolved reactive P concentrations in the profile in excess of plant requirements, however, regular soil testing was not permitted because of destruction and play interruption.

Summary and Conclusion

NO_3-N and dissolved reactive P concentrations were measured for four-years from two sites within the Morris Williams' Municipal Golf Course located in Austin, TX. Based on these measurements, and subsequent analysis, the following summary and conclusions can be drawn.

- NO_3-N concentrations (1.27 mg L^{-1} at

Figure 8
Time series plot of dissolved reactive phosphorus (DRP) concentration for the study period at Site 3 and Site 4.



Site 3; 0.32 mg L^{-1} at Site 4) and load (2.7 kg ha^{-1}, 2.4 lb ac^{-1}) transported through the subsurface drainage water were approximately 1/10th the concentration and load typically reported for tile drainage from row crop agriculture.

• A strong seasonal pattern was detected; NO_3-N was present in greater concentrations in the subsurface drainage water during the winter months (periods of greater rainfall, turfgrass dormancy, and reduced microbial activity) than during the spring and summer months.

• The greatest concentrations of NO_3-N (3.9 mg L^{-1} at Site 3; 3.1 mg L^{-1} at Site 4) measured in the drainage water occurred seven to nine months after application.

• Seasonal differences in losses of dissolved reactive P were not as pronounced as those observed for NO_3-N. The greatest dissolved reactive P concentrations (0.99 mg L^{-1} at Site 3; 0.62 mg L^{-1} at Site 4) in subsurface drainage water were measured during the fall. Only small dissolved reactive P concentrations were detected during the winter and spring months.

• Dissolved reactive P concentrations in the subsurface drainage water were greater than concentrations measured in tile drains from agriculture and could pose a potential threat of eutrophication to a surface water system.

• The timing of NO_3-N and dissolved reactive P movement through subsurface drainage from golf course turf appeared to be dependent on climatic factors (temperature and precipitation) and turf management factors (magnitude and timing of applications).

• The magnitude of NO_3-N and dissolved reactive P concentrations was dependent on the frequency and amount of fertility management practices. At the more intensively managed site (Site 3), consistently higher NO_3-N and dissolved reactive P concentrations were detected in the drainage water than were measured from the less intensively managed site (Site 4).

Current environmental management practices for turfgrass systems are often focused on surface losses. Based on the measurements taken in this study, a similar emphasis should be given to the potential for subsurface movement of nutrients that eventually return to the surface water, particularly dissolved reactive P. The implementation of integrated management plans that account for turf quality, hydrology, and nutrient interactions should focus not only on potential surface movement but also on potential subsurface and lateral transport. As dissolved reactive P levels in the soil increase, more dissolved reactive P may be available for leaching from the system. Thus, turfgrass managers should apply phosphorus only after determining the requirement for sustaining plant growth.

Endnotes

Trade names are included for the benefit of the reader and do not imply endorsement by U.S. Department of Agriculture. This research was partially funded by the U.S. Golf Association, Green Section.

References Cited

Allen, P.M., R.D. Harmel, J.G. Arnold, B. Plant, J. Yelderman, and K.W. King. 2005. Field data and flow system response in clay (Vertisol) shale terrain, north central Texas. Hydrological Processes. U.S. Department of Interior 10.1002/hyp.5782.

American Society of Agricultural Engineers (ASAE). S526.2. 2001. Soil and water terminology. Pp. 983-1002. *In:* American Society of Agricultural Engineers Standard. St. Joseph, Michigan.

Balogh, J.C., V.A. Gibeault, W.J. Walker, M.P. Kenna, and J.T. Snow. 1992a. Background and overview of environmental issues. Pp. 1-37. *In:* Golf course management and construction: Environmental issues. J.C. Balogh and W.J. Walker (eds.) Lewis Publishers, Inc. Chelsea, Michigan.

Beard, J.B. 1973. Turfgrass: Science and culture. Prentice-Hall. Englewood Cliffs, New Jersey.

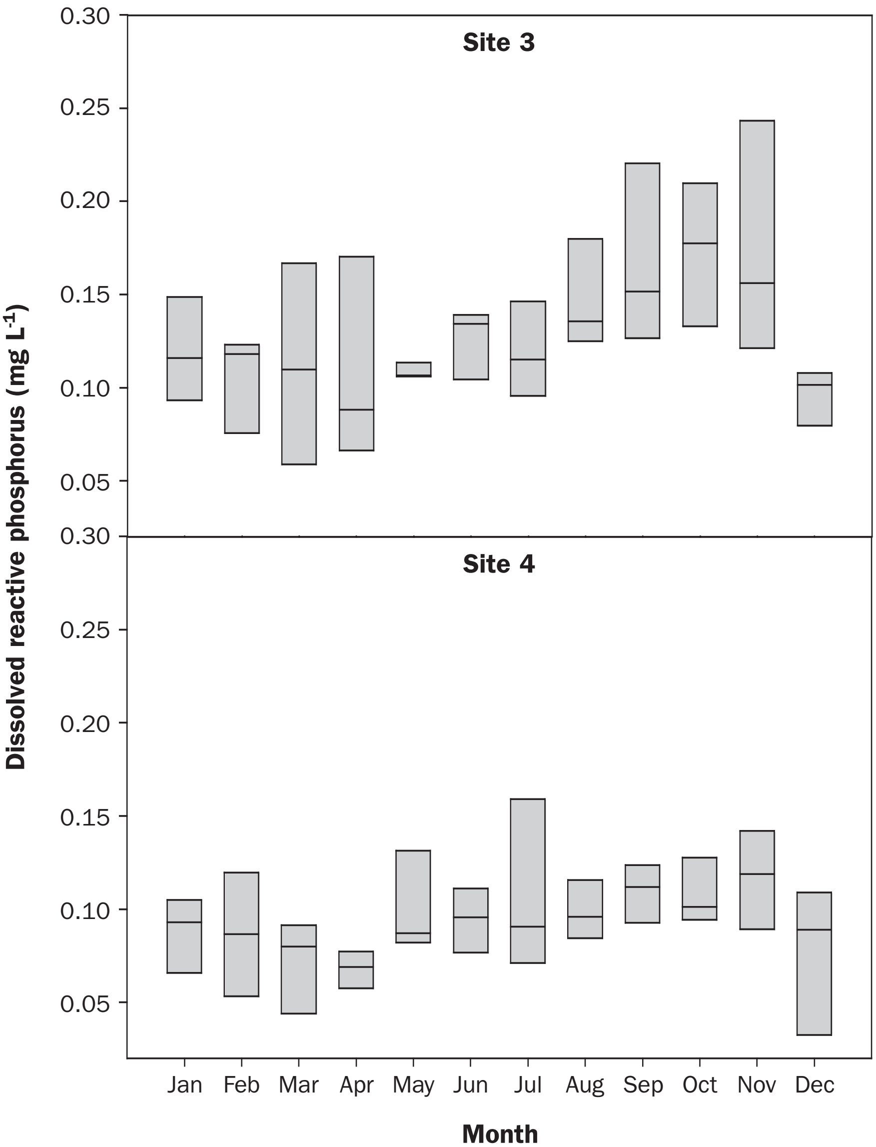
Beard, J.B. 2000. Turfgrass benefits and the golf environment. Pp. 36-44. *In:* Fate and management of turfgrass chemicals, ACS Symposium Series No. 743. J.M. Clark and M.P. Kenna (eds.) American Chemical Society, Washington, D.C.

Chatfield, C. 1989. The analysis of time series: An introduction. Chapman and Hall, London, England.

Cromwell, J.B., W.C. Labys, and M. Terraza. 1994a. Univariate tests for time series models. Sage Publications, Thousand Oaks, California.

Figure 9

Monthly variation in dissolved reactive phosphorus concentration from four years of data at Site 3 and Site 4. Gray bars represent 25th and 75th percentiles, line represents the median of the data points in each month.



Cromwell, J.B., M.J. Hannan, W.C. Labys, and M. Terraza. 1994b. Multivariate tests for time series models. Sage Publications, Thousand Oaks, California.

Daniel, T.C., A.N. Sharpley, and J.L. Lemunyon. 1998. Agricultural phosphorus and eutrophication: A symposium overview. Journal of Environmental Quality 27:251-257.

DiPaola, J.M. and J.B. Beard. 1992. Physiological effects of temperature stress. Pp. 231-267. *In:* Turfgrass - Agronomy Monograph No. 32. D.V. Waddington, R.N. Carrow, and R.C. Shearman (eds.) Agronomy Society of America, Crop Science Society of America, and Soil Science Society of America publishers. Madison, Wisconsin.

Duxbury, J.M. and J.H. Perverly. 1978. Nitrogen and phosphorus losses from organic soils. Journal of Environmental Quality 7:566-570.

Fenelon, J.M. and R.C. Moore. 1998. Transport of agrichemicals to ground and surface water in a small central Indiana watershed. Journal of Environmental Quality 27:884-894.

Gentry, L.E., M.B. David, K.M. Smith-Starks, and D.A. Kovacic. 2000. Nitrogen fertilizer and herbicide transport from tile drained fields. Journal of Environmental Quality 29(1):232-240.

Geohring, L.D., O.V. McHugh, M.T. Walter, T.S. Steenhuis, M.S. Akhtar, and M.F. Walter. 2001. Phosphorus transport into subsurface drains by macropores after manure applications: Implications for best manure management practices. Soil Science 166(12):896-909.

Gilliam, J.W., T.J. Logan, and F.E. Broadbent. 1985. Fertilizer use in relation to the environment. Pp. 561-588. *In:* Fertilizer technology and use. O.P. Engelsted (ed.) Soil Science Society of America. Madison, Wisconsin.

Guillard, K. and K.L. Kopp. 2004. Nitrogen fertilizer form and associated nitrate leaching from cool-season lawn turf. Journal of Environmental Quality 33(5):18221827.

Jamieson, A., C.A. Madramootoo, and P. Enright. 2003. Phosphorus losses in surface and subsurface runoff from a snowmelt event on an agricultural field in Quebec. Canadian Biosystems Engineering 45:1.1-1.7.

Kladivko, E.J., G.E. Van Scoyoc, E.J. Monke, K.M. Oates, and W. Pask. 1991. Pesticide and nutrient movement into subsurface tile drains on a silt loam soil in Indiana. Journal of Environmental Quality 20:264-270.

Kladivko, E.J., J. Grochulska, R.F. Turco, G.E. Van Scoyoc, and J.D. Engel. 1999. Pesticide and nitrate transport into subsurface drains of different spacings. Journal of Environmental Quality 28:997-1004.

Mansell, R.S., P.J. McKenna, E. Flaig, and M. Hall. 1985. Phosphate movement in columns of sandy soil from a wastewater-irrigated site. Soil Science 140:59-68.

Minitab, Inc. 1998. Minitab statistical software, Release No. 12. User's guide: Data analysis and quality tools. Minitab Inc., State College, Pennsylvannia.

Mitchell, J.K., G.F. McIsaac, S.E. Walker, and M.C. Hirschi. 2000. Nitrate in river and subsurface drainage flows from an east central Illinois watershed. Transactions of the ASAE 43:337-342.

Morton, T.G., A.J. Gold, and W.M Sullivan. 1988. Influence of overwatering and fertilization on nitrogen losses from home lawns. Journal of Environmental Quality 17(1):124-130.

National Oceanic and Atmospheric Administration (NOAA). 1993. Local climatological data: Annual summary with comparative data. Austin, Texas.

Petrovic, A.M. 1990. The fate of nitrogenous fertilizers applied to turfgrass. Journal of Environmental Quality 19(1):1-14.

Pira, E. 1997. A guide to golf course irrigation system design and drainage. Ann Arbor Press Inc., Chelsea, Michigan.

Randall, G.W., D.R. Huggins, M.P. Russelle, D.J. Fuchs, W.W. Nelson, and J.L. Anderson, 1997. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. Journal of Environmental Quality 26:240-1247.

Randall, G.W., J.A. Vetsch, and J.R. Huffman. 2003. Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by time of nitrogen application and use of nitrapyrin. Journal of Environmental Quality 32:1764-1772.

Reddy, G.Y., E.O, McLean, G.D. Hoyt, and T.J. Logan. 1978. Effects of soil, cover crop, and nutrient source on amounts and forms of phosphorus movement under simulated rainfall conditions. Journal of Environmental Quality 7:50-54.

Ruz-Jerez, B.E., R.E. White, and P.R. Ball. 1995. A comparison of nitrate leaching under clover-based pastures and nitrogen fertilized grass grazed by sheep. Journal of Agricultural Science 125:361-369.

Salas, J.D. 1993. Analysis and modeling of hydrologic time series. Pp. 19.1-19.72. *In:* Handbook of Hydrology. D.R. Maidment (ed) McGraw Hill, New York, New York.

Sharpley, A.N. and S. Rekolainen. 1997. Phosphorus in agriculture and its environmental implications. Pp. 1-53. *In:* Phosphorus loss from soil to water. H. Tunney, O.T. Carson, P.C. Brooks, and A.E. Johnston (eds.) CAB International, Wallingford, England.

Shuman, L.M., A.E. Smith, and D.C. Bridges. 2000. Potential movement of nutrients and pesticides following application to Golf Courses. Pp. 78-93. *In:* Fate and management of turfgrass chemicals. American Chemical Society Symposium Series No. 743. J.M. Clark and M.P. Kenna (eds.) American Chemical Society, Washington, D.C.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. Phosphorus loss in agricultural drainage: Historical perspective and current research. Journal of Environmental Quality 27(2):277-293.

Technicon Industrial Systems. 1973a. Nitrate and nitrite in water and wastewater. Industrial method No. 100-70w. Technicon Instruments Corp., Tarrytown, New York.

Technicon Industrial Systems. 1973b. Ammonia in water and wastewater. Industrial method No. 98-70w. Technicon Instruments Corp., Tarrytown, New York.

Technicon Industrial Systems. 1976. Individual/simultaneous determination of nitrogen and/or phosphorus in BD acid digest. Industrial method No. 344-74a. Technicon Instruments Corporation, Tarrytown, New York.

Tomer, M.D., D.W. Meek, D.B. Jaynes, and J.L. Hatfield. 2003. Evaluation of nitrate nitrogen fluxes from a tile-drained watershed in central Iowa. Journal of Environmental Quality 32:642-653.

Troughton, A. 1957. The underground organs of herbage grasses. Bulletin No. 44. Commonwealth Bureau of Pastures and Field Crops. Commonwealth Agricultural Bureau. Farmham Royal, Bucks, England.

U.S. Environmental Protection Agency (USEPA). 1986. Quality criteria for water. U.S. Environmental Protection Agency, Washington D.C. Report EPA 440/5-86-001.

U.S. Soil Conservation Service (SCS). 1974. Soil Survey of Travis County Texas. U.S. Department of Agriculture Soil Conservation Service and Texas Agricultural Experiment Station. U.S. Government Printing Office. Washington, D.C.

Wang, X., J.R. Frankenberger, and E.J. Kladivko. 2003. Estimating nitrate-N losses from subsurface drains using variable water sampling frequencies. Transactions of the ASAE 46(4):1033-1040.