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# Spatial and temporal variation of nitrogen exported by runoff from sandy agricultural soils

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**Abstract**

The eutrophication problem has drawn attention to nutrient leaching from agricultural soils, and an understanding of spatial and temporal variability is needed to develop decision-making tools. Thus, eleven sites were selected to monitor, over a two-year period, spatial and temporal variation of runoff discharge and various forms of N in surface runoff in sandy agricultural soils. Factors influencing the variation of runoff discharge and various forms of N in surface runoff were analyzed. Variation of annual rainfall was small among 11 sites, especially between 2001 and 2002. However, variation of annual discharge was significant among the sites. The results suggest that rainfall patterns and land use had significant effect on discharge. The concentrations of total N, total kjeldahl N (TKN), organic matter-associated N (OM-N), $NO_3^-$-N, and $NH_4^+$-N in the runoff ranged widely from 0.25 to 54.1, 0.15 to 20.3, 0.00 to 14.6, 0.00 to 45.3, and 0.00 to 19.7 mg/L, respectively. Spatial and temporal variations in the N concentration and runoff discharge were noted among the different sites. Annual loads of N in the runoff varied widely among monitoring sites and depend mainly on runoff discharge. High loads of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff either in citrus groves or on vegetable farms occurred from June to October for each year, which coincided with the rainy season in the region. This study found that N in surface runoff was related to rainfall intensity, soil N level, and fertilizer use.

**Key words**: citrus grove; nitrogen; rainfall; runoff; sandy agricultural soils; spatial and temporal variation; vegetable farms

## Introduction

The increased input of fertilizers and animal wastes has boosted agricultural crop production in many countries, but it has also contributed to the increased nitrogen export from agriculture to ground and surface waters (Hall and Risser, 1993; Neilsen *et al*., 1982; Schroder, 1985). Nitrogen in fertilizers is usually very soluble and relatively mobile in soil. Applied fertilizer-N not utilized by the crops can be lost to groundwater by leaching, to surface waters by surface runoff, or to air by denitrification and volatilization. Nitrogen leaching to groundwater is commonly associated with high application rates of fertilizer and manures. The leaching is enhanced in sandy soils, under wet climatic conditions or irrigation. Nitrogen loss to surface water may occur as direct runoff, or by infiltration through the root zone and discharge to surface water through seepage or tile drainage systems. In the former case, most of the N is in ammonium form. In the latter case, most of the N is in nitrate form (Zebarth *et al*., 1999). Nitrogen loss in the surface runoff is enhanced with reduced soil infiltration capacity.

Detailed information has been obtained on the factors that contribute to increased risk of nitrate leaching from agricultural soils to groundwater (Bergstrom, 1987; Johnson and Raun, 1995; Kalita and Kanwar, 1993; Strebel *et al*., 1989). However, this information is commonly available at a limited spatial scale, usually from point- or small-plot measurements. It is somewhat difficult to apply this information to large spatial scales. In contrast, surface-water studies are commonly performed on a watershed basis (Leon *et al*., 2001; Valiela and Bowen, 2002). There is an increasing demand for on-farm studies to supply information to help minimize nitrogen losses to the environment (Goss *et al*., 1995).

The crop-growing regions in Florida are characterized by sandy soils and unequally distributed high precipitation. Crops on the flatwoods soil are bedded to improve drainage and surface-water removal during intense storms. The sandy soils are very sensitive to the loss of N, because the soils usually contain low clay and organic matter for holding N. Many of the recent studies on N loss associated with Florida crop production have centered on the leaching and vertical transport of $NO_3^-$-N and $NH_4^+$-N caused by fertilization in sandy soils (He *et al*., 2000; McNeal *et al*., 1995; Paramasivam *et al*., 2001, 2002). Models have been applied to assess N leaching in field conditions (Harrison

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*et al*., 1999). Lesser attention, however, has been given to the loss from surface runoff. There is minimal quantitative data on concentrations and forms of N in the surface runoff from field-scale studies. This information is needed to better assess the environmental impact and the possible mechanisms of N loss to surface waters. The objectives of this study were to assess spatial and temporal variability of forms, concentrations and loads of N over a two-year period at four vegetable and seven citrus production sites in South Florida.

## 1 Materials and methods

Eleven field sites (seven in commercial citrus groves and four on vegetable production farms) in St. Lucie and Martin Counties, FL were selected to monitor N losses in surface runoff in 2001–2002 (Table 1). For vegetable farms, crops were grown from fall to next spring (end of August to March) followed by a four- to six- month summer fallow period. Polyethylene mulch was used as a bed cover during the growth season. SIGMA 900MAX portable autosamplers (American Sigma, Loveland, Co.) were installed at the drainage outlet for each site. Rainfall and runoff flow rate were recorded every ten minutes. All sites were distributed in the Indian River area of Florida with flat landscape (< 5% slope) and shallow water table, where the dominant hydrological pathways had an extensive network of artificial drainage ditches.

The soils of the experiment sites were representative of commercial citrus and vegetable production systems in the Indian River. They included Wabasso sand (sandy, siliceous, hyperthermic Alfic Alaquods), Waveland fine sand (sandy, siliceous, hyperthermic, ortstein Arenic Alaquods), Ankona sand (sandy, siliceous, hyperthermic, ortstein Arenic Ultic Alaquods), Winder variant sand (sandy, siliceous, hyperthermic Typic Glossaqualfs), and Nettles sand (sandy, siliceous, hyperthermic, ortstein Alfic Arenic Haplaquods). General characteristics of the study sites are given in Table 1.

For each field site, three composite soil samples were taken across each experimental field prior to the experiment. Each composite sample was composed of a mixture of four samples taken at a depth of 0–15 cm from four locations within each field. All soil samples were air-dried and ground to < 2 mm prior to chemical analysis. Soil pH, measured in water at a soil:water ratio of 1:1 using a pH/ion/conductivity meter (Accumet Model 50, Fisher Scientific, Norcross, GA) varied greatly across the 11 sites, and ranged from 4.4 to 8.1 (Table 1). Soil total N was determined using a CN-Analyzer (Vario MAX CN Macro Elemental Analyzer, Elemental Analysensystem GmbH, Hanau, Germany) and ranged from 0.27 to 1.84 g/kg. Particle composition of soil sample was determined using the micropipette method (Miller and Miller, 1987) and clay content of the soil samples ranged from 19 to 81 g/kg. Soil available N ($NH_4^+$-N and $NO_3^-$-N) was determined by shaking a 2.5 g air-dried soil sample in 25 ml of 2 mol/L KCl for 1 h (Mulvaney, 1996). Concentrations of $NH_4^+$-N and $NO_3^-$-N in the filtrate were analyzed using the Nitrate/Ammonium autoanalyzer (AA III, Bran Luebbe, Buffalo Grove, IL). The concentrations of soil available $NH_4^+$-N and $NO_3^-$-N were low and ranged from 0 to 2.25, 1.50 to 7.50 mg/kg, respectively.

Runoff samples from each field site were collected in 1-L bottles placed inside each autosampler during each rainfall event. The autosamplers were programmed so that six individual surface runoff samples were taken every 24 h. The autosamplers were checked daily to ensure proper performance and to collect surface runoff samples, if available. Data on rainfall and flow recorded in the autosamplers were transferred every week to a computer in the laboratory, using a data logger. Water samples collected from the autosamplers were immediately transported to laboratory.

Portions of the sub-samples were filtered through a Whatman #42 filter paper for measurement of N. The concentration of $NO_3^-$-N was measured within 24 h after sample collection using an Ion Chromatograph (DX 500; Dionex Corporation Sunnyvale, CA). $NH_4^+$-N and total kjeldahl N (TKN) in the runoff sample were measured using the nitrate/ammonium autoanalyzer (AA III, Bran Luebbe, Buffalo Grove, IL) following EPA method 351.3 (APHA *et al*., 1995). Total N in the runoff sample was calculated as the sum of the TKN and $NO_3^-$-N. Organic matter-associated N (OM-N) was calculated from the difference between the TKN and $NH_4^+$-N. The dissolved total N, TKN, $NO_3^-$-N, and $NH_4^+$-N loads in the runoff water were determined as a product of the N concentrations in runoff sample and each runoff water discharge.

**Table 1 Characteristics of the studied field sites**

| Site | Crop | Soil type | Site area ($m^2$) | pH ($H_2O$) | $NH_4^+$-N (mg/kg) | $NO_3^-$-N (mg/kg) | Total N (g/kg) | Clay (g/kg) | Annual N fertilizer ($kg/hm^2$) | Irrigation method | Permeability |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Citrus | Waveland fine sand | 2748 | 7.5 | 1.75 | 7.50 | 0.35 | 52 | 168 | Microjet | Moderately rapid |
| 2 | Citrus | Waveland fine sand | 2748 | 7.4 | 1.00 | 2.50 | 0.50 | 50 | 168 | Microjet | Moderately rapid |
| 3 | Citrus | Wabasso sand | 2978 | 5.0 | 1.50 | 2.75 | 0.27 | 19 | 168 | Microjet | Moderately slow |
| 4 | Citrus | Wabasso sand | 2978 | 5.1 | 1.50 | 1.50 | 0.33 | 25 | 168 | Microjet | Moderately slow |
| 5 | Vegetable | Nettles sand | 405 | 7.1 | 0.75 | 3.50 | 0.44 | 55 | 376 | Drip | Moderately rapid |
| 6 | Vegetable | Nettles sand | 405 | 7.6 | 2.25 | 5.50 | 0.36 | 52 | 393 | Drip | Moderately rapid |
| 7 | Vegetable | Wabasso sand | 570 | 7.3 | 0.00 | 2.25 | 0.37 | 43 | 274 | Seepage | Moderately slow |
| 8 | Vegetable | Wabasso sand | 570 | 7.2 | 0.50 | 2.00 | 0.36 | 45 | 291 | Seepage | Moderately slow |
| 9 | Citrus | Ankona sand | 1822 | 4.4 | 2.12 | 5.45 | 1.84 | 61 | 165 | Microjet | Moderately slow |
| 10 | Citrus | Winder variant sand | 6122 | 8.1 | 1.65 | 2.51 | 0.41 | 81 | 180 | Microjet | Slow |
| 11 | Citrus | Ankona sand | 2242 | 6.6 | 1.82 | 1.50 | 0.41 | 41 | 142 | Microjet | Moderate |

## 2 Results

### 2.1 Rainfall and runoff discharge

Annual rainfall for the 11 sites ranged from 1203 to 1572 mm and 1002 to 1363 mm for the years 2001 and 2002, and averaged 1314 mm and 1183 mm, respectively (Fig.1a, Table 2). Mean rainfall in 2001 was 11.1% more than that in 2002. Annual water discharge for the 11 sites varied greatly from 55 to 488 mm and 5 to 226 mm for the years 2001 and 2002, and averaged 208 mm and 102 mm, respectively. The mean water discharge in 2001 was 103.9% more than that in 2002. Variation of annual rainfall among different sites for the same observed year was small and was 6.4% and 9.1% for 2001 and 2002, respectively. However, variation of annual water discharge among different sites for the same observed year was great and was 61.1% and 52.0% for 2001 and 2002, respectively. Runoff coefficient, a ratio of water discharge to rainfall, was generally small, but it varied greatly in space and time. Mean of monthly runoff coefficient ranged from 0 to 0.85 (data not shown). Annual mean of runoff coefficient of the 11 sites varied greatly from 0.04 to 0.31 and 0.0 to 0.18 for 2001 and 2002 and averaged 0.15 and 0.086, respectively. The annual mean of runoff coefficient for 2001 was about two times of that for 2002.

Variations of annual rainfall among the 11 sites or between 2001 and 2002 were not significant. But, variations of either annual water discharge or runoff coefficient among the 11 sites or between the years 2001 and 2002 were noted. These results indicated that formation of runoff was variable and affected by other factors besides annual rainfall.

Rainfall and water discharge varied seasonally for all 11 sites (Fig.1). However, the seasonal variation of rainfall and water discharge was different in the observed years, and was more significant in 2001 than 2002. Seasonal distribution of rainfall was more even in 2002 than 2001. Seasonal variation of water discharge was more significant than rainfall.

Although there were correlation between monthly rainfall and monthly water discharge for each site, the correlation coefficients were generally small (Table 3). The correlation between annual rainfall and annual water discharge for all 11 sites was also small ($r$ = 0.5815, $n$ = 22). These results indicated that either annual rainfall or monthly rainfall could not clearly express the amount of surface runoff, and other factors, including rainfall intensity, rainfall pattern, and antecedent soil moisture before rain had a greater effect on surface runoff.

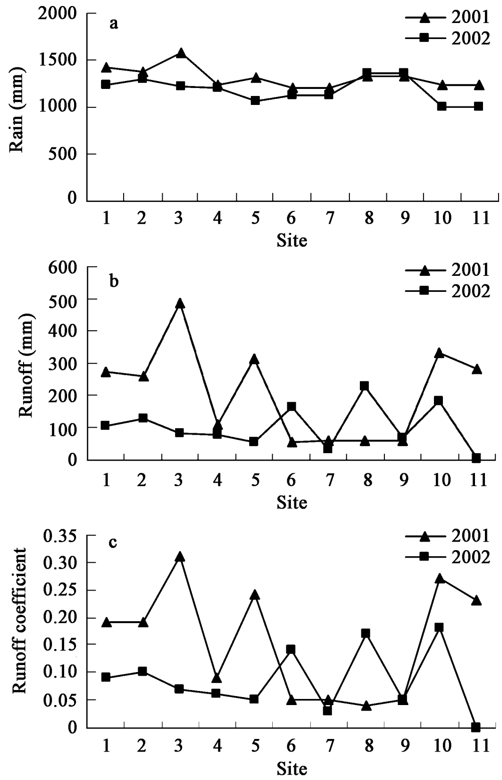


Fig. 1 Annual rainfall (a), runoff (b), and runoff coefficient (c) of 11 sites.

### 2.2 Nitrogen concentration and forms in runoff water

Total N concentration in the 1276 runoff water samples collected from the 11 sites during the two-year experiment varied widely from 0.25 to 54.1 mg/L with a mean concentration of 4.12 mg/L. The concentrations of TKN, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff water ranged from 0.15 to 20.3, 0.00 to 14.6, 0.00 to 45.3, and 0.00 to 19.7 mg/L, respectively. Most of the samples had moderate concentrations (1–5 mg/L) of total N, TKN, and OM-N. The proportions of samples with this concentration range were 67.95%, 74.53%, and 69.36%, respectively,

**Table 2 Mean of monthly rain and discharge of 11 sites**

| Month | Rain (mm) | | Discharge (m$^3$/hm$^2$) | |
|---|---|---|---|---|
| | 2001 | 2002 | 2001 | 2002 |
| 1 | 20.57±4.74 | 42.12±12.21 | 0.03±0.07 | 2.51±3.71 |
| 2 | 1.68±1.65 | 106.97±25.45 | 0.00±0.00 | 41.48±74.36 |
| 3 | 83.43±22.52 | 24.89±14.87 | 8.64±15.04 | 18.38±42.05 |
| 4 | 7.90±6.54 | 71.33±47.40 | 0.44±1.42 | 18.02±29.66 |
| 5 | 77.75±35.50 | 130.16±34.46 | 4.79±8.77 | 92.71±157.13 |
| 6 | 156.28±59.07 | 184.87±56.47 | 251.77±343.29 | 119.58±131.79 |
| 7 | 252.15±45.06 | 212.34±59.14 | 389.54±455.15 | 573.15±732.73 |
| 8 | 179.90±34.63 | 112.68±23.46 | 417.35±441.34 | 46.19±97.48 |
| 9 | 281.13±75.43 | 120.17±52.82 | 530.59±542.70 | 43.84±49.84 |
| 10 | 133.79±26.28 | 26.95±25.59 | 201.80±281.19 | 3.47±6.53 |
| 11 | 101.62±29.17 | 55.63±12.74 | 310.68±317.70 | 34.32±60.47 |
| 12 | 17.89±10.65 | 94.74±26.73 | 0.00±0.00 | 30.15±54.52 |

**Table 3 Correlation coefficients between monthly rain and discharge ($n$ = 24)**

| Site | Correlation coefficient ($r$) | Site | Correlation coefficient ($r$) |
|---|---|---|---|
| 1 | 0.5782 | 7 | 0.4042 |
| 2 | 0.6295 | 8 | 0.4565 |
| 3 | 0.8388 | 9 | 0.6617 |
| 4 | 0.6207 | 10 | 0.7487 |
| 5 | 0.4819 | 11 | 0.6624 |
| 6 | 0.2308 | | |

for the total N, TKN, and OM-N. The concentrations of $NO_3^-$-N and $NH_4^+$-N in the runoff water were generally lower; 70.30% and 91.30% of the samples had $NO_3^-$-N or $NH_4^+$-N below 1 mg/L. Few of the samples contained relatively higher N, about 2% of the samples had $NO_3^-$-N concentrations higher than 10 mg/L, a maximum limit of the contamination level for drinking water (USEPA, 1986; Canadian Water Quality Branch, 1995). In addition, the proportions of samples that had total N, TKN, OM-N, or $NH_4^+$-N higher than 10 mg/L were 7.37%, 3.06%, 1.49%, and 0.78%, respectively.

At the vegetable sites, the peaks of total N, $NO_3^-$-N, and $NH_4^+$-N concentrations occurred occasionally during September to May (data not shown), which correspond to the periods of the crop growing season and one to two months after each field's plastic cover was removed. The peak N in the runoff was probably associated with leaking of N from the fertilizer band in the crop growing season or higher $NH_4^+$-N and $NO_3^-$-N in surface soils in the period of one to two months after removal of the plastic cover. However, soil available N was generally low during June to September, because most of available N was leached downward in the two months after the removal of plastic cover. Consequently, the concentrations of N in runoff collected during June to September were low. At the citrus sites, peaks of total N, $NO_3^-$-N, and $NH_4^+$-N concentrations occurred occasionally during May to October, which coincided with the rainy season. Storms increased the amount of runoff, which increased N loss from the soils.

Proportions of total N as $NO_3^-$-N, $NH_4^+$-N, and OM-N in the runoff samples varied with the samples, ranged from 0 to 92.91%, 98.91%, and 100%, and averaged 23.86%, 8.62%, and 67.21%, respectively. The OM-N was the most important N form in the runoff water.

Both N and P contribute to accelerated eutrophication of surface water systems. The flourishing of phytoplanktons in a water system is limited by either N or P, depending on their availability and the N/P ratio (Phlips *et al*., 2002). In January 2001, the United States Environmental Protection Agency (USEPA, 2005) published sets of nutrient criteria for various types of water bodies within 17 eco-regions of the United States. The nutrient-water quality criteria of total N for Southern Florida Coastal Plain is 1.27 mg/L. A large portion of the surface runoff samples collected from this study had total N concentration above this critical level. This means that the runoff water from the agricultural fields could be a potential non-point source of N if it is allowed to drain directly to the surrounding water systems such as the Indian River Lagoon.

### 2.3 Spatial and temporal variation

Annual mean concentrations of total N, TKN, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff water varied with the tested sites and sampling years as shown in Table 4. Sites 1 and 5 had the lowest annual mean $NO_3^-$-N and $NH_4^+$-N, respectively, in the two-year experimental period. However, the sites that had the highest annual mean $NO_3^-$-N or $NH_4^+$-N varied with sampling year. In 2001, the

**Table 4 Annual mean concentrations of various N forms in runoff water from each field site**

| Site | TN (mg/L) | TKN (mg/L) | OM-N (mg/L) | $NO_3^-$-N (mg/L) | $NH_4^+$-N (mg/L) | $NO_3^-$-N/TN (%) | $NH_4^+$-N/TN (%) | OM-N/TN (%) |
|---|---|---|---|---|---|---|---|---|
| 2001 | | | | | | | | |
| 1 | 1.91de | 1.68e | 1.54de | 0.23c | 0.14bc | 8.79e | 7.67c | 83.54a |
| 2 | 2.74cd | 2.13de | 2.04d | 0.61c | 0.09c | 18.22de | 4.44c | 77.34ab |
| 3 | 4.59bc | 3.20cd | 2.10d | 1.39c | 1.10ab | 22.93cd | 10.94bc | 66.12bc |
| 4 | 3.90bc | 3.22cd | 1.53de | 0.68c | 1.70a | 12.39e | 18.59ab | 69.03abc |
| 5 | 0.48de | 0.35f | 0.31f | 0.14c | 0.04c | 16.21de | 7.50c | 76.29ab |
| 6 | 1.87de | 0.69ef | 0.35f | 1.18c | 0.34bc | 40.88a | 12.54abc | 46.97d |
| 7 | 6.42ab | 2.35cdf | 1.83de | 4.08ab | 0.50bc | 31.60abc | 15.12abc | 53.27cd |
| 8 | 2.13cd | 1.15ef | 0.69ef | 0.98c | 0.46bc | 31.69abc | 20.51a | 47.83d |
| 9 | 6.64ab | 4.53b | 4.02b | 2.10bc | 0.51bc | 24.54bc | 7.74c | 67.72abc |
| 10 | 4.81bc | 2.37cde | 2.17d | 2.43abc | 0.20bc | 35.96ab | 4.48c | 59.56cd |
| 11 | 7.74a | 6.26a | 5.76a | 1.48c | 0.50bc | 17.71de | 5.71c | 76.58ab |
| 2002 | | | | | | | | |
| 1 | 1.39de | 1.06ef | 0.90ef | 0.32c | 0.16bc | 11.62e | 9.98bc | 77.67ab |
| 2 | 3.05cd | 1.76de | 1.60de | 1.26c | 0.17bc | 19.20cd | 7.54c | 72.68abc |
| 3 | 3.69bc | 2.21cde | 2.03d | 1.21c | 0.17bc | 25.80bc | 4.76c | 67.11abc |
| 4 | 2.66cd | 2.02de | 1.89d | 0.65c | 0.13bc | 18.05de | 4.55c | 77.24ab |
| 5 | 1.43de | 0.94ef | 0.86ef | 0.49c | 0.08c | 35.55ab | 8.14c | 56.31cd |
| 7 | 3.63cd | 1.97de | 1.25de | 1.66c | 0.73bc | 40.77a | 10.48bc | 48.76d |
| 8 | 6.72ab | 2.19cde | 1.64de | 4.53a | 0.56bc | 40.52a | 10.56bc | 48.93d |
| 9 | 5.06ab | 3.47bc | 3.21bc | 1.60c | 0.26bc | 26.60bc | 4.66c | 68.74abc |
| 10 | 2.47cd | 1.65e | 1.42de | 0.82c | 0.23bc | 28.17abc | 9.92bc | 61.91bc |
| 11 | 4.53bc | 3.38bc | 3.07c | 0.93c | 0.32bc | 13.67de | 5.97c | 78.09ab |

No runoff samples were collected from site 6 in 2002; TN: total N; TKN: total Kjeldahl N; OM-N: organic matter-associated N. Duncan multiple range groupings, means for each P form with the same letter is not significantly different at $p < 0.05$.

highest of annual mean $NO_3^-$-N and $NH_4^+$-N occurred at site 7 and site 10, respectively. In 2002, the highest of annual mean $NO_3^-$-N and $NH_4^+$-N occurred at site 8 and site 7, respectively. The annual mean concentrations of the TKN and OM-N had similar patterns in both 2001 and 2002; the highest occurred at sites 9 and 11, and the lowest occurred at site 5. Variation of total N in the runoff among the 11 sites was greater than other N forms. In 2001, the highest annual mean total N occurred at sites 7, 9, and 11, and the lowest occurred at site 5; the highest was 16 times the lowest. In 2002, the highest annual mean total N occurred at sites 8 and 9, and the lowest occurred at sites 1 and 5. The difference of annual mean total N among the 11 sites was smaller in 2002 than 2001 (Table 4).

Because the proportion of total N as $NH_4^+$-N in runoff water was generally low, and the TKN and OM-N in the runoff water originated mainly from soils, the variation trend of TKN and OM-N concentrations in the runoff among the 11 sites was comparable between 2001 and 2002. Available N (KCl-extractable $NO_3^-$-N and $NH_4^+$-N) in soils before fertilization at all sites was very low (Table 1); and thus, fertilizers (ammonium nitrate) could be main source of $NO_3^-$-N and $NH_4^+$-N in the runoff water. The total N, being comprised of OM-N, $NO_3^-$-N and $NH_4^+$-N, was affected by both soil N levels and fertilizations and varied between 2001 and 2002.

Except for sites 5 and 10 in 2001, mean proportions of total N as organic (OM-N) in the runoff water was higher at citrus sites than vegetable sites, whereas mean proportions of total N as inorganic N ($NO_3^-$-N and $NH_4^+$-N) were higher at the vegetable sites than the citrus sites (Table 4). The differences in the mean concentrations of TKN, OM-N, and $NO_3^-$-N, and proportions of total N as $NO_3^-$-N, $NH_4^+$-N, and OM-N between the vegetable and the citrus sites were significant (Table 5). The concentrations of TKN, OM-N, and proportion of OM-N in the total N were greater at the citrus than the vegetable sites, while the mean $NO_3^-$-N concentration and proportions of $NO_3^-$-N and $NH_4^+$-N in the total N were greater at vegetable than citrus sites. The correlations of annual mean total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff water for each site with soil available N ($NO_3^-$-N and $NH_4^+$-N), total N, and fertilizer N rates were analyzed. There was a significant correlation between annual mean OM-N in the runoff water and soil total N ($r$ = 0.45*, $n$ = 21), suggesting that soil total N (mainly organic N) had a significant influence on organic N (OM-N) concentration in the runoff water. However, no significant correlations were found between the annual mean concentrations of $NO_3^-$-N and $NH_4^+$-N in the runoff water and soil available N ($NO_3^-$-N and $NH_4^+$-N) and fertilizer N rates, indicating that factors other rather than fertilizer N controlled the inorganic N loss from the soils. The differences in land use and soil permeability among the 11 sites may have a significant effect on $NO_3^-$-N and $NH_4^+$-N concentrations in the runoff water.

## 2.4 Discharge and nitrogen loads of runoff water

Because the mean annual rainfall for the 11 sites was higher in 2001 (1314 mm) than 2002 (1183 mm), annual runoff discharge, mean concentrations of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff water for most of the sites were greater in 2001 than 2002 (Table 6). Although differences in annual rainfall among the 11 sites were minimal, annual runoff discharge and N loads in the runoff varied widely among the 11 sites. The differences of annual runoff discharge and N loads in the runoff among the 11 sites was probably related to soil properties (permeability and total N), fertilizer rates, land use, and other unknown factors. In 2001, the highest runoff discharge (site 10) was about 10 times of the lowest discharge (site 5). Consequently, total N, OM-N, and $NO_3^-$-N loads in the runoff at site 10 were 90, 67, and 184 times, respectively, of those at site 5. However, the $NH_4^+$-N load in the runoff water in the 2001 was the highest at site 4 and the lowest at site 5 (Table 6); the highest was 171 times of the lowest. In 2002, annual discharge was highest at sites 1 and 7, and lowest at 8; the highest was about 48 times of the lowest. The highest loads of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N occurred at site 7, the lowest loads occurred at site 8 for total N, OM-N, and $NH_4^+$-N, and at sites 1 and 2 for $NO_3^-$-N. The highest loads of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff were 18, 37, 69, and 38 times the lowest loads.

Annual loads of total N, OM-N, and $NO_3^-$-N in the runoff were significantly correlated with annual runoff discharge ($r$ = 0.78***, 0.69***, 0.77***, respectively). The annual load of OM-N in the runoff was also slightly correlated with soil total N ($r$ = 0.40, $p$ = 0.073). However, no significant correlations were found between N loads and soil available N ($NO_3^-$-N, and $NH_4^+$-N) and fertilizer rates. Loads of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff varied with season. Monthly loads of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N in the runoff either from citrus or vegetable sites were high from June to October for each year, which coincided with rainy season. For other months, the loads of total N, OM-N, $NO_3^-$-N, and $NH_4^+$-N

**Table 5 Comparison of mean TN, TKN, OM-N, $NO_3^-$-N, and $NH_4^+$-N in runoff water collected during the two-year experiment between vegetable and citrus sites**

| Site | $n$ | TN (mg/L) | TKN (mg/L) | OM-N (mg/L) | $NO_3^-$-N (mg/L) | $NH_4^+$-N (mg/L) | $NO_3^-$-N/TN (%) | $NH_4^+$-N/TN (%) | OM-N/TN (%) |
|---|---|---|---|---|---|---|---|---|---|
| 2001 | | | | | | | | | |
| Vegetable farms | 161 | 4.22ab | 1.70c | 1.25c | 2.53a | 0.44a | 30.46b | 16.42a | 53.13b |
| Citrus groves | 588 | 4.68a | 3.24a | 2.68a | 1.44b | 0.56a | 22.39c | 7.84b | 69.77a |
| 2002 | | | | | | | | | |
| Vegetable farms | 66 | 4.19ab | 1.78bc | 1.29c | 2.40a | 0.50a | 39.33a | 9.90b | 50.76b |
| Citrus groves | 461 | 3.35b | 2.29b | 2.07b | 0.99b | 0.21a | 21.23c | 6.71b | 72.06a |

**Table 6 Annual loads of TN, OM-N, $NO_3^-$-N, and $NH_4^+$-N in runoff water collected from each field site during the two-year experiment**

| Site | Total N (kg/hm$^2$) | OM-N (kg/hm$^2$) | $NO_3^-$-N (kg/hm$^2$) | $NH_4^+$-N (kg/hm$^2$) | Runoff discharge (m$^3$/hm$^2$) |
|---|---|---|---|---|---|
| 2001 | | | | | |
| 1 | 1.04 | 0.85 | 0.08 | 0.11 | 594 |
| 2 | 2.07 | 1.50 | 0.51 | 0.06 | 603 |
| 3 | 9.32 | 5.52 | 2.17 | 1.63 | 2724 |
| 4 | 14.41 | 2.45 | 1.70 | 10.25 | 2599 |
| 5 | 0.26 | 0.17 | 0.06 | 0.02 | 548 |
| 6 | 1.07 | 0.20 | 0.67 | 0.20 | 571 |
| 7 | 5.78 | 2.94 | 1.90 | 0.95 | 3327 |
| 8 | 3.04 | 1.10 | 1.30 | 0.63 | 2831 |
| 9 | 23.31 | 12.97 | 8.71 | 1.63 | 3117 |
| 10 | 23.37 | 11.42 | 11.05 | 0.90 | 5263 |
| 11 | 8.76 | 7.25 | 0.99 | 0.52 | 1094 |
| 2002 | | | | | |
| 1 | 2.13 | 1.89 | 0.06 | 0.18 | 2263 |
| 2 | 0.90 | 0.75 | 0.05 | 0.08 | 698 |
| 3 | 4.79 | 2.06 | 1.76 | 0.21 | 1058 |
| 4 | 3.12 | 2.23 | 0.66 | 0.22 | 1267 |
| 5 | 1.45 | 0.59 | 0.68 | 0.19 | 1626 |
| 7 | 7.20 | 2.61 | 3.47 | 1.13 | 1832 |
| 8 | 0.39 | 0.07 | 0.29 | 0.03 | 47 |
| 9 | 2.80 | 1.66 | 1.00 | 0.14 | 542 |
| 10 | 1.83 | 0.83 | 0.77 | 0.24 | 833 |
| 11 | 3.94 | 2.44 | 0.93 | 0.26 | 777 |

No runoff samples were collected from site 6 in 2002.

in the runoff were generally low.

## 3 Discussion

Research during the last decade has shown that nutrient concentration in water discharge from a basin is a result of several interacting processes, including exchange between cycles in the soil, and aquatic, geological, and atmospheric environment. These can be categorized into: (1) nutrient release through mineralization, weathering, fertilization, atmospheric deposition, sewage effluents; (2) water transport; (3) transformation and immobilization including denitrification, sedimentation, and adsorption. The influence of different processes varies spatially and temporally, because they may be more or less favored by local conditions such as land use, topography, management, meteorology, or hydrology. Thus some researches have emphasized the importance of linking combinations of local characteristics to surface water quality (Osborne and Wiley, 1988), because this enables further understanding of simultaneous influences from several different processes.

Agricultural soils are often nutrient rich, drained, tilled and fertilized regularly, and periodically lack vegetation cover and uptake, and thereby increase nutrient leaching both on the surface and through the soil profile. In our study, both inorganic N and organic N were related to land use. Total N loads in surface runoff from the 11 field sites ranged from 0.26 to 23.4 kg/(hm$^2$·a), which is compatible with the reported range (Yan *et al*., 2005). Surface runoff N loads are mainly affected by N fertilizer rates (Yan *et al*., 2005), N-rich organic manure application (Smith *et al*., 2001), and soil type (Van Beek *et al*., 2004). Our study observed a slight correlation between N loads and soil total N, but no significant correlation was noted between N loads and N fertilizer rate or soil extractable inorganic N.

An important finding of the study is that the average nutrient losses varied greatly among sites studied. The main explanations for this variability were differences in water runoff, fertilizer use, soil type, and erosion (Vagstad *et al*., 2004). However, it was difficult to find general relations between the individual factors. For example, there was poor correlation between nitrogen losses and fertilizer use. Therefore, the results suggest that the observed variability in N losses cannot have been solely due to differences in farm management practices, although the studied sites do include a wide range of nutrient application levels, and other relevant factors. There is considerable spatial variation in the physical properties (soil, climate, hydrology, and topography) and the agricultural management of the sites, and the interaction between these properties and relative effects of these factors has an important impact on erosion and nutrient losses. In particular, hydrological processes may have a marked effect on N losses measured in the sites. The results indicate that significant differences in hydrological pathways (e.g. the relationship between fast- and slow-flow processes) lead to major site differences in nutrient inputs to surface waters and therefore also in the response to changes in field management practices. Agricultural practices such as crop rotation systems, nutrient inputs, and soil conservation measures obviously play a significant role in the site-specific effects. The interactions between agricultural practices and basic catchment characteristics, including hydrological processes, determine the final losses of nitrogen to surface waters; hence it is necessary to understand these interactions to efficiently manage diffuse losses of agricultural nutrients.

It was reported that land use, soil type, slope, and rainfall could influence runoff (Osborne and Wiley, 1988). In this study, though rainfall, soil texture, and slope for all sites were similar, they may not be the main factors causing difference of runoff volume among the sites. The influence of hydrometeorological variables needs to be explored for further understanding of flow generation and concentration variation in discharge from agricultural basins. Combination of patterns of rainfall and antecedent soil moisture may most often be more strongly related to the runoff. Kleinman *et al*. (2006) reported that elevated, antecedent soil moisture required less rain to generate runoff and greater runoff volumes, site hydrology, not chemistry, was primarily responsible for variations in N losses. Nutrient losses were significantly greater under the high intensity rainfall due to large runoff volumes. Overall, the drainage characteristics of soil were found to have a considerable influence on the potential of land to lose nutrients to water.

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