Environ Monit Assess (2014) 186:1317–1326
DOI 10.1007/s10661-013-3482-y

Groundwater phosphorus in forage-based landscape with cow-calf operation

Gilbert C. Sigua · Chad C. Chase Jr.

Received: 28 February 2013 / Accepted: 30 September 2013 / Published online: 24 October 2013

Abstract Forage-based cow-calf operations may have detrimental impacts on the chemical status of groundwater and streams and consequently on the ecological and environmental status of surrounding ecosystems. Assessing and controlling phosphorus (P) inputs are, thus, considered the key to reducing eutrophication and managing ecological integrity. In this paper, we monitored and evaluated P concentrations of groundwater (GW) compared to the concentration of surface water (SW) P in forage-based landscape with managed cow-calf operations for 3 years (2007–2009). Groundwater samples were collected from three landscape locations along the slope gradient (GW1 10–30 % slope, GW2 5–10 % slope, and GW3 0–5 % slope). Surface water samples were collected from the seepage area (SW 0 % slope) located at the bottom of the landscape. Of the total P collected (averaged across year) in the landscape, 62.64 % was observed from the seepage area or SW compared with 37.36 % from GW (GW1=8.01 %; GW2=10.92 %; GW3=18.43 %). Phosphorus in GW ranged from 0.02 to 0.20 mg L^{-1} while P concentration in SW ranged from 0.25 to 0.71 mg L^{-1}. The 3-year average of P in GW of 0.09 mg L^{-1} was lower than the recommended goal or the Florida's numeric nutrients standards (NNS) of 0.12 mg P L^{-1}. The 3-year average of P concentration in SW of 0.45 mg L^{-1} was about fourfold higher than the Florida's NNS value. Results suggest that cow-calf operation in pasture-based landscape would contribute more P to SW than in the GW. The risk of GW contamination by P from animal agriculture production system is limited, while the solid forms of P subject to loss via soil erosion could be the major water quality risk from P.

Keywords Groundwater · Phosphorus · Beef cattle · Landscape · Bahiagrass · Erosion · MCL

G. C. Sigua (✉)
United States Department of Agriculture,
Agricultural Research Service,
Coastal Plains Soil Water & Plant Research Center,
Florence, SC 29501, USA
e-mail: gilbert.sigua@ars.usda.gov

C. C. Chase Jr.
United States Department of Agriculture, Agricultural Research Service, U.S. Meat Animal Research Center,
Clay Center, NE, USA

Introduction

Phosphorus (P) fertilization is a vital component of productive farming. Phosphorus is an essential macronutrient that is required to meet global food requirements and make crop and livestock production profitable (Hedley and Sharpley 1998). While adequate levels of P soils are essential to grow crops, P has the potential to induce eutrophication in our water systems. Recent assessments of water quality status have identified eutrophication as one of the major causes of water quality "impairment" not only in the US, but also around the world (Sigua 2010). Phosphorus is considered to be the limiting nutrient in most freshwater ecosystems (Sundareshwar et al. 2003; Sondregaard and Jeppesen 2007). Controlling P inputs is thus considered the key to reducing eutrophication and managing ecological integrity (Daniel et al. 1998). In most cases, eutrophication has accelerated by increased inputs of P due to intensification of crop and animal production systems since the early 1990s. Cattle manure contains appreciable amounts of nitrogen and P (0.6 and 0.2 %, respectively), and portions of these components can be transported into receiving waters during severe rainstorms (Khaleel et al. 1980). Sharpley et al. (2001) argued that the overall goal of efforts to reduce P loss to water should involve balancing P inputs and outputs at farm and watershed levels. Conservation practices should be targeted to relatively small, but critical watershed areas for P export.

Reduction of P transport to receiving water bodies has been the primary focus of several studies because P has been found to be the limiting nutrient for eutrophication in many aquatic systems (Botcher et al. 1999; Sigua et al. 2000; Sigua and Tweedale 2003). Elsewhere, studies of both large (Asmussen et al. 1975; Bogges et al. 1995; Edwards et al. 2000) and small watersheds (Romkens et al. 1973; Hubbard and Sheridan 1983) have been performed to answer questions regarding the net effect of agricultural practices on water quality with time or relative to weather, fertility, or cropping practices. Work in other regions of the country has shown that when grazing animals become concentrated near water bodies, or when they have unrestricted long-term access to streams for watering, sediment and nutrient loading can be high (Thurow 1991; Brooks et al. 1997; Sigua and Coleman 2007). Additionally, there is a heightened likelihood of P losses from over fertilized pastures through surface water (SW) runoff or percolation past the root zone (Gburek and Sharpley 1998; Stout et al. 2000).

One of the first steps in assessing the P level on any farm is to consider total P inputs and outputs. Very little attention to date has been paid to evaluating transfers of P via groundwater (GW) in landscape with cow-calf operations (Sigua et al. 2009). An interest in resource balances in agricultural science dates back to an early experiment in 1930 using balance sheets to show how farm manure and other sources of P supply (air, rain, and soil) had satisfied crop needs (Scoones and Toulmin 1999). Subsequently, the approaches to input–output analysis became a major focus of systems ecology beginning in the 1950s, when energy, mineral P, and other cycles were identified (Odum 1988). Understanding the effects of water-table management, P dynamics and water quality in pastures is the key to reducing P in runoff. Sharpley (1997) noted that all soils do not contribute equally to P export from watersheds or have the same potential to transport P to runoff. In their studies, Coale and Olear (1996) observed that soil test for P levels did not accurately predict total dissolved P. Better understanding of soil P dynamics and other crop nutrient changes resulting from different management systems should allow us to predict potential impact on adjacent surface waters. These issues are critical and of increasing importance among environmentalists, ranchers, and public officials in the state (Sigua et al. 2006).

Relatively, little information exists regarding possible magnitudes of P losses from grazed pastures. Whether or not P losses from grazed pastures are significantly greater than background losses and how these losses are affected by soil, forage management, or stocking density are not well understood (Gary et al. 1983; Edwards et al. 2000; Sigua et al. 2004; Sigua et al. 2006). A long-term quantitative assessment of soil chemical properties may serve as an indicator of a soil's capacity for sustainable production of crops and animals in an economically sound, socially acceptable, and environmentally friendly manner (Lemunyon and Gilbert 1993; Sharpley et al. 1996). We hypothesized that properly managed cow-calf operations would not be major contributors to excessive concentrations of P in SW and GW in subtropical pastures. To verify our hypothesis, we examined the concentration of total P in SW and GW beneath bahiagrass-based pastures with cow-calf operations for 3 years (2007–2009).

Materials and methods

Site description

This study (2007–2009) was conducted at the Land Use Unit (28°4′22.8″–28°4′38.2″N; 82°20′7.7″–82°20′31.1″W) of the Subtropical Agricultural Research Station located 7 miles north of Brooksville, FL. The research station has three major pasture units with a combined total area of about 1,558 ha with 1,295 ha in permanent pastures. Cattle used for nutritional, reproductive, and genetic research on the station include about 500 heads of breeding females with a total inventory of about 1,000 head of cows, calves, and bulls. Cattle production at the station is forage-based with bahiagrass as the predominant forage species (approximately 1,000 ha). Most of the bahiagrass pastures have been established for over 30 years.

The soils at the study site are described as loamy, siliceous, hyperthermic family of the Grossarenic Paleudults (Hyde et al. 1977), slopes up to 30 %, and are consistently north facing. Forage production potential of the soils in the station is generally low to medium; the main limitation being soil water availability. The study area is well drained with average soil permeability ranging from 0.004 to 0.014 cm s^{-1}. Other properties of soils at the study site were included in Table 1.

The highest average temperature occurs during August, although highs in the mid-30 °C range occur regularly from May through September. The lowest average temperature of 14 °C occurs during January, but frosts are frequent during the winter months. The total annual precipitation in 2007, 2008, and 2009 were 106.65, 111.18, and 116.51 cm at the study site, respectively.

Pasture management, fertilization, and grazing day intervals

At the beginning of 1990s, bahiagrass pastures were fertilized annually in the early spring (March) with 77 kg N, 10 kg P, and 37 kg K ha^{-1} based on the revised fertilizer recommendation suggested by Chambliss (1999).

Historically, grazing cattle were rotated among pastures to allow rest periods of 2–4 weeks based on herbage mass. The timing of movement for rotationally grazed cattle was determined by the herd manager's perception of herbage mass based on plant height and not based on pasture measurement (Williams and Hammond 1999). Starting in 2000, cattle were rotated twice weekly (3- or 4-day grazing period). We anticipated 24 days of rest between pastures. During this study, the average number of 3.2 cow-calf pair per hectare grazed each pasture for about 10 days each month. Table 2 shows the average number of days grazed for each month (4 years average). This cattle grazing rotation yielded an average annual total manure excretion of about 9,347 kg ha^{-1} or about 17.8 kg ha^{-1} of total P from manure excretions (Sigua 2010).

Instrumentation and water sample collection

Two adjacent 8-ha pasture fields were instrumented with a pair of shallow wells placed at different landscape positions (Fig. 1). The different landscape positions are GW1 (10–20 % slope), GW2 (5–10 % slope), GW3 (0–5 % slope), and seepage area, SW (0 % slope). The wells were constructed of 5 cm schedule 40 PVC pipe and had 15 cm of slotted well screen at the bottom. During installation of wells, sand was placed around the slotted screen, and bentonite clay was used to backfill the soil surface to prevent SW or runoff from moving down the outside of the PVC pipe and contaminating GW samples. A centralized battery-operated peristaltic pump was used to collect water samples (Fig. 2). Wells were completely evacuated during the sampling process to ensure that water for the next sampling would be fresh GW (Hubbard et al. 1986). Water samples were collected from the GW wells every 2 weeks. However, there were periods when GW levels were below the bottom level of the wells and samples could not be obtained. Actual date of sample collection and sampling frequency were shown in Fig. 3.

In addition to GW samples, SW samples were collected in the pasture bottoms or the seep area when present by taking composite grab samples on the same schedule. The seepage area, which is located at the lower end of bottom slope is a remnant of a sinkhole formation and became a small-scale lake with varying levels of SW. The seepage area of about 2 ha in size is where runoff and seepage from higher parts of pasture converge.

Water sample handling and analyses

Water samples were transported to the laboratory following collection and refrigerated at 4 °C. Water

Table 1 Selected properties of soils at the study site

Landscape location (% slope)	pH (mg kg^{-1})	TIN	TP	Organic carbon (g kg^{-1})	Al (mg kg^{-1})	Fe	Sand (g kg^{-1})	Silt	Clay
a. Top (10–20 %)	5.9	6.2	5.9	5.6	120.2	2.2	86.2	5.9	7.9
b. Middle (5–10 %)	5.8	2.6	9.2	5.0	106.6	2.4	87.3	6.9	5.8
c. Bottom (0–5 %)	5.8	1.4	5.7	3.9	114.6	2.5	88.3	6.4	5.2

Table 2 Monthly summary of grazing activity at the study site

Months	Average days grazed per pasture	Average number of animals per ha	Animal unit per month[a]	Monthly manure phosphorus excreted[b]
January	13.8	2.6	1.0	1.73
February	9.4	2.5	0.8	1.27
March	13.5	2.1	0.9	1.43
April	12.0	2.0	0.7	1.18
May	12.7	2.1	0.7	1.24
June	12.0	2.4	0.9	1.42
July	6.9	3.3	0.7	1.19
August	9.1	3.6	0.8	1.39
September	8.2	4.8	1.1	1.80
October	6.7	5.3	1.1	1.85
November	6.6	3.6	0.8	1.34
December	9.4	3.7	1.2	1.97
Average	10.0	3.2		1.48
Total	–	–	10.8	17.81

[a] Animal units per month, *AUM* (450 kg cow/calf unit)

[b] Total manure excreted (kg as excreted) = [(number of AUM* total annual animal manure excretion/12) total manure excretion (as excreted) per animal per year = 10.4 mT (Kellogg et al. 2000)

samples were analyzed for total P using a Flow Injector P analyzer according to standard methods (APHA 1989).

Data reduction and statistical analysis

Concentrations of P in SW and GW beneath a bahiagrass-based pasture at four different landscape positions in 2007, 2008, and 2009 were analyzed statistically using the SAS PROC MIXED model (SAS Institute 2000). Where the *F* test indicated a significant ($p \leq 0.05$) effect, means were separated following the method of Duncan's multiple range test using appropriate error mean squares (SAS Institute 2000). The data were sorted by landscape position when there were differences in the concentration of P between SW and GW. Separation of the data by year was done to determine if total P concentrations were increasing with time (SAS Institute 2000).

Results

The concentration of P in landscape with cow-calf operations varied significantly with year (Y, $p \leq 0.0001$), type of water (T, $p \leq 0.01$), and location (L, $p \leq 0.0001$) and with the interactions of Y × T ($p \leq 0.001$) and Y × L

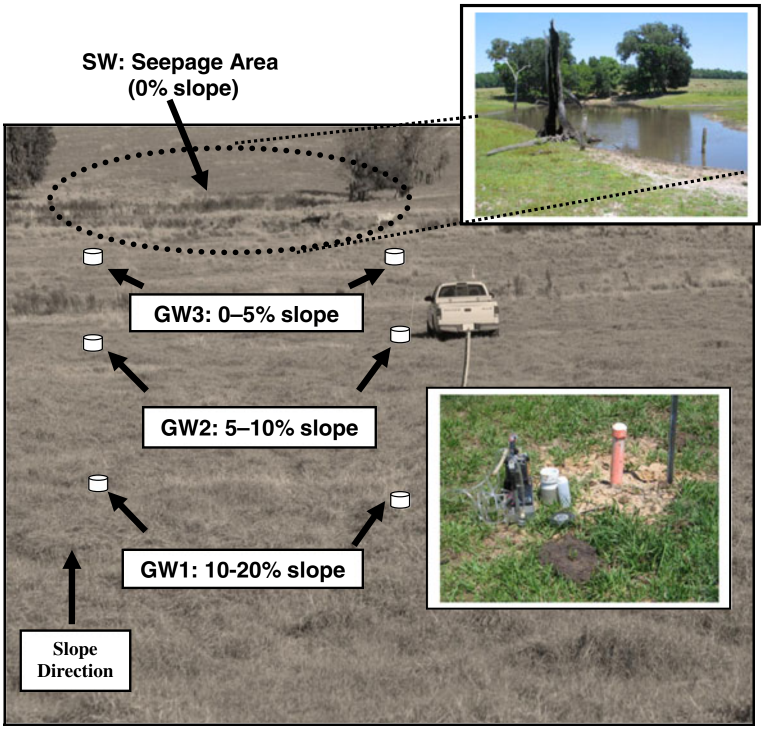
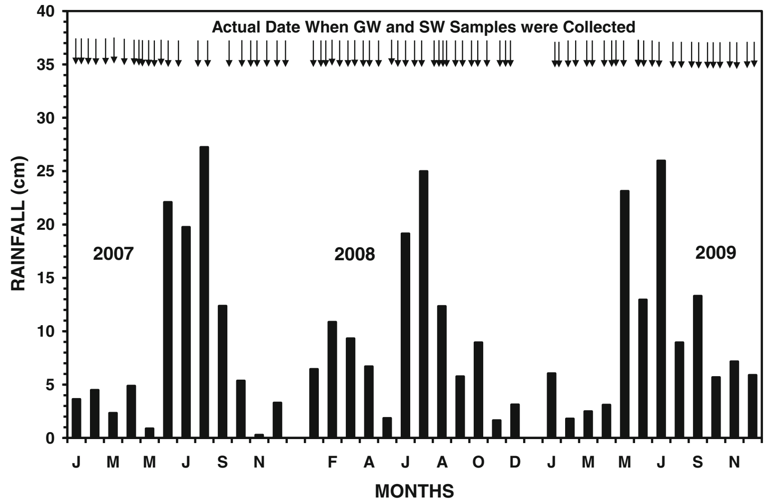


Fig. 1 Location of study area showing collection sites for surface and ground water

Fig. 2 Monthly rainfall distribution in the study area and actual date of sample collections (indicated by *arrows*)



($p \leq 0.01$) (Table 3). Groundwater samples at any given year (i.e., 2007, 2008, or 2009) had significantly ($p \leq 0.05$) lower concentration of P (0.09 mg L^{-1}) than the concentration of P in SW (0.55 mg L^{-1}). Of the total P collected (averaged across year), 62.64 % was observed from the seepage area or SW compared with 37.36 % from GW (GW1=8.01 %; GW2=10.92 %; GW3=18.43 %).

On the effect of landscape location, water samples collected from wells located near the bottom (GW3, 0–5 % slope) had the greatest amount of P followed by GW2 (5–10 % slope) and GW1 (10–20 % slope) with mean P concentrations of 0.13, 0.08, and 0.06 mg L^{-1}, respectively (Fig. 3). The concentration of P from these three landscape locations did not differ significantly ($p \leq 0.05$) from each other, but significantly ($p \leq 0.05$) lower than the concentration of P in the seepage area (0.46 mg L^{-1}) at the bottom of the landscape (Fig. 3). Of the total concentration of P among landscape locations, about 62.64 % was found in the SW, 18.43 % was from GW3, 10.92 % from GW2, and 8.01 % was from GW1 (Fig. 3). Results suggest an increasing concentration of P with decreasing slope position.

On the interaction effect of year and type of water, concentration of P in GW at any given year was significantly lower ($p \leq 0.05$) than the concentration of P in SW (Fig. 4). In 2007, the average concentration of P in GW was about 0.14 mg L^{-1} compared with 0.71 mg L^{-1} in SW. In 2008, the average concentration of P in GW was about 0.06 mg L^{-1} while concentration

Fig. 3 Concentration of P in GW and SW. *Dotted line* represents Florida's numeric nutrients standards (0.12 mg P L^{-1})

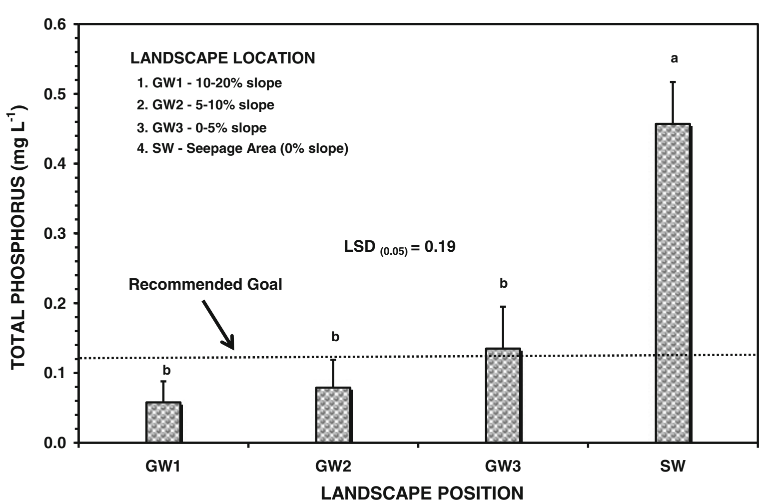


Table 3 Annual average of total phosphorus in surface and ground water

Year	Type	Location	N	Total phosphorus (mg L^{-1})
1. 2007	Ground	GW1	16	0.02±0.01a[a]
		GW2	14	0.16±0.09a
		GW3	23	0.20±0.06a
	Surface	SW	25	0.71±0.16a
LSD $_{(0.05)}$				0.71
2. 2008	Ground	GW1	15	0.12±0.06b
		GW2	14	0.04±0.02b
		GW3	19	0.05±0.02b
	Surface	SW	25	0.43±0.11a
LSD $_{(0.05)}$				0.24
3. 2009	Ground	GW1	16	0.02±0.001b
		GW2	14	0.07±0.04ab
		GW3	23	0.16±0.06ab
	Surface	SW	25	0.25±0.09a
LSD $_{(0.05)}$				0.25
Sources of variation		*F* values		
Year effect (Y)		7.21***[b]		
Type (T)		47.5*		
Landscape location (L)		10.23***		
Y x T		5.3**		
Y x L		5.2**		

[a]Means in column within each year followed by common letter(s) are not significantly different from each other at $p \leq 0.05$

[b]*$p \leq 0.01$; **$p \leq 0.001$; ***$p \leq 0.0001$e, *ns* not significant

of P in SW was about 0.43 mg L^{-1}. In 2009, the average concentration of P in GW was about 0.11 mg L^{-1} while concentration of P in SW was about 0.25 mg L^{-1}.

Mean annual concentration of P (averaged across water type and landscape locations) revealed that the greatest amount of P (0.38 mg L^{-1}) was in 2007 followed by 2008 (0.17 mg L^{-1}) and 2009 (0.16 mg L^{-1}). The annual average of P concentrations for 2007, 2008, and 2009 in pasture with cow-calf operations were all above Florida's NNS value of 0.12 mg P L^{-1} (Florida Department Environmental Protection 2013). Although a downward trend was noted on the concentration of P from 2007 to 2009, the average P levels were somewhat problematic in the long term because they were higher than Florida's NNS (Fig. 4).

As shown in Table 4, summary statistics of P concentrations varied significantly among years and among water types. In 2007, P concentrations in GW ranged from 0.0001 to 0.66 mg L^{-1} compared with P concentrations in SW that ranged from 0.07 to 0.36 mg L^{-1}. The P concentrations in GW and SW ranged from 0.00 to 0.58 and 0.03 to 1.39 mg L^{-1}, respectively, in 2008. In 2009, the maximum and minimum concentrations of P in GW were 0.97 and 0.0005 mg L^{-1} compared with the maximum and minimum of P in SW of 0.004 and 1.26 mg L^{-1}, respectively. Median concentrations of P in GW and SW for 2007, 2008, and 2009 were 0.04, 0.01, and 0.02 mg L^{-1} and 0.71, 0.22, and 0.05 mg L^{-1}, respectively (Table 4).

Discussion

Results of our 3-year study did not fully support our hypothesis that properly managed cow-calf operations would not be major contributors to excessive concentrations of P in SW and GW. Interestingly, the highest concentrations of P in our study were found at the seepage area located at the bottom of the forage-based landscape, while the lowest concentration of P was found at the highest slope location in the landscape (GW1). These results can be attributed to the grazing activities, as animals tend to graze more at the bottom slope, near the seepage area. In our pastures, slope position was confounded with mineral, water source, and to some degree shade, which were located at the

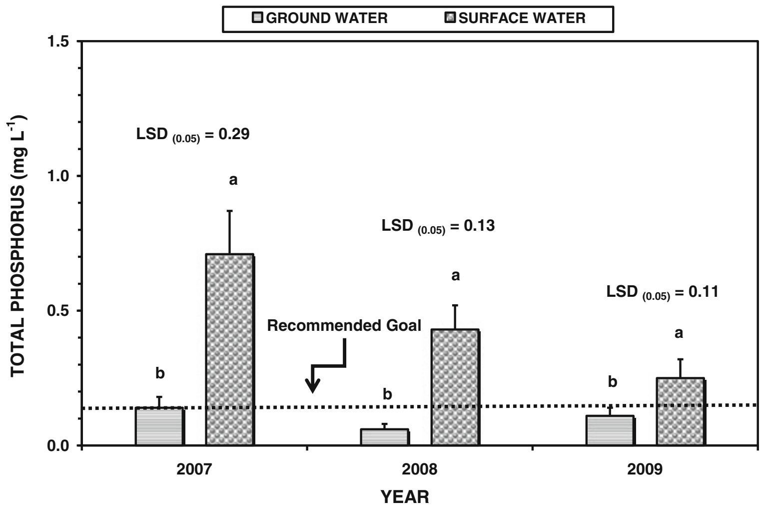


Fig. 4 Three-year (2007–2009) comparative average of P concentrations in GW and SW. *Bar graphs* with different superscript for each year were not significantly different at $p \leq 0.05$. *Dotted line* represents Florida's numeric nutrients standards (0.12 mg P L^{-1})

bottom slope of the landscape. White et al. (2001) claimed that there was a correlation between times spent in a particular area and the number of excretions received, and this behavior could lead to an increase in the concentration of P close to shade and water. Sigua and Coleman (2007) also reported that the concentrations of P in soils, plants, SW, and shallow groundwater varied significantly among the different congregation sites that would include water troughs, mineral feeders, and trees/shades on bahiagrass pastures in Florida. Nonuniform grazing distribution by livestock on landscapes can be caused by many variables such as water location (Ganskopp 2001), minerals (Martin and Ward 1973), herbage mass (Senft et al. 1983), and terrain slope, which exist at a variety of scales.

Livestock grazing plays an important role in soil and water dynamics because of the return of P through animal excretion. Nutrients in excreta/urine can be lost via soil erosion, surface runoff, animal ingestion (Boddey et al. 2004; Yan et al. 2007), leaching to groundwater (Tamminga 2006), and haying (Sigua et al. 2006). As nutrients accumulate in the soil, subsequent off-site loss can occur. The degree to which pastures can cause water quality concerns varies greatly depending on a number of factors such as soil type, fertilization regimen, stocking rate, and environment conditions (Silveira et al. 2010). We observed consistently that our animals tended to graze more at the bottom than in the middle slope or top slope of our pastures. When livestock tend to graze some pastured areas more than others like in our study (bottom slope > middle slope > top slope), soil and water physical and chemical attributes can change with time. Animals tend to deposit more excreta in loafing areas near shade and water troughs that are located at the bottom of the pasture landscape and dietary supplement sources (Peterson and Gerrish 1996) instead of uniform deposition across the pasture. Animal feces deposition is on the soil surface, which makes P transport easier and faster. Observations of animal movement based on actual positions of the test animals within the pasture at 8:30 a.m. and at 2:00 p.m. on daily basis disclosed that 40 to 50 % of the time that cows in herds was grazing at the bottom slope near the seepage area of the pasture (unpublished data).

Our results have shown that the 3-year average of P in GW of 0.09 mg L^{-1} was lower than the recommended goal or the Florida's NNS value of 0.12 mg P L^{-1} (Florida Department Environmental Protection 2013). The 3-year average of P concentration in SW of 0.45 mg L^{-1} was fourfold higher than Florida's NNS value. Results suggest that cow-calf operation in pasture-based landscape would contribute more P to SW than in the GW. The higher concentration of P in SW than that in GW could be attributed to the interactions between P and clay minerals. It was reported that P does not leach very much except in very sandy soils. Because P is strongly adsorbed to soil particles, P leaching would occur only when the percentage P saturation of the soil is increased to very high levels through continued applications of P exceeding crop requirement (Heckrath et al. 1995; Owens and Shipitalo 2006).

Fertilizer and manure P inputs to the soil are retained by smaller particles, so the added P is not redistributed uniformly through the whole profile (House et al. 1998). As P accumulates in the soils in response to excessive

Table 4 Summary statistics for the annual average concentration of total phosphorus in surface water and groundwater beneath pastures associated with managed cow-calf operations

Statistical parameters	Shallow groundwater	Surface water
2007		
Number of samples (n)	33	23
Mean (mg L^{-1})	0.14	0.71
Median (mg L^{-1})	0.04	0.36
Mode (mg L^{-1})	0.0008	0.07
Maximum (mg L^{-1})	0.66	2.63
Minimum (mg L^{-1})	0.0001	0.07
Std. Error Mean	0.04	0.16
Variance	0.57	0.60
Skewness	1.52	1.45
2008		
Number of samples (n)	56	25
Mean (mg L^{-1})	0.06	0.43
Median (mg L^{-1})	0.01	0.22
Mode (mg L^{-1})	0.01	0.02
Maximum (mg L^{-1})	0.58	1.39
Minimum (mg L^{-1})	0.00	0.03
Std. error Mean	0.02	0.09
Variance	0.02	0.19
Skewness	3.26	1.02
2009		
Number of samples (n)	53	26
Mean (mg L^{-1})	0.11	0.25
Median (mg L^{-1})	0.02	0.05
Mode (mg L^{-1})	0.006	0.04
Maximum (mg L^{-1})	0.97	1.26
Minimum (mg L^{-1})	0.0005	0.004
Std. Error Mean	0.03	0.07
Variance	0.05	0.14
Skewness	2.32	1.42

fertilizer, or animal manure, P may become susceptible to transport via surface runoff and subsurface leaching. Thus, P applied to the soil as fertilizers or in the form of manure in pasture tends to stay in the topsoil and not be lost to either subsurface drainage waters. The large majority of non-point source P lost to surface waters is attached to eroded soil particles. The smaller, lighter particles are transported to greater distances and are more likely to enter surface waters (House et al. 1998). The higher the concentration of P in topsoil the larger the amount of P in the receiving surface waters. Based on the results of our study, potential P delivery from agricultural field to surface or ground water could be affected by four factors, namely the (1) amount of P adsorbed to eroding sediments, (2) amount of soluble P in runoff water, (3) amount of soluble P in leaching water, and (4) amount of P losses related to the type of P-containing fertilizers or manures applied. Several process-based models have been developed to quantify P transport (Sharpley et al. 1992). For example, agricultural nonpoint pollution source assesses nutrient loading on watershed scales using individual runoff events (Young et al. 1995), soil and water assessment tool was developed to predict the impact of land and water management on sediment and chemical transport in large watersheds (Arnold et al. 1998), and erosion–productivity impact calculator has also been used to quantify sediment and nutrient transport (Sharpley and Williams 1990).

Conclusion

Results of this study may help to renew the focus on improving inorganic fertilizer efficiency in subtropical beef cattle systems, and maintaining a balance of P removed to P added to ensure healthy forage growth and minimize phosphorus runoff. New knowledge based on the whole-farm approach is desirable to identify pastureland at risk of degradation and to prescribe treatments or management practices needed to protect the natural resources while maintaining an economically and environmentally viable operation. Therefore, a better understanding of soil P dynamics, P use efficiency, and other crop nutrient changes in bahiagrass pasture with cow-calf management systems should allow us to better predict the least risk of P losses to adjacent SW and GW. Based on the concentrations of P in GW (0.09 mg L^{-1}) that were observed in our study, potential leaching of P under grazed bahiagrass pastures would not be harmful to water quality. However, the average concentration of P in SW of 0.45 mg L^{-1} is above the Florida's NNS (0.12 mg P L^{-1}), and this level could pose a harmful effect on the environment and therefore would require a continued and long-term monitoring and assessment.

To effectively implement any BMP, it is necessary to recognize the potential impact of agricultural P on SW and understand that different landscape locations and varying hydrologic conditions in forage-based landscape could affect the spatial and temporal variations of

P losses at the watershed scale. If we understand where and how P is getting into our waters, we can implement BMPs to reduce or even eliminate P as a potential pollutant from our water supply. For the purposes of reducing and/or eliminating P in getting into our waters, the following BMPs for P in the environment as suggested by Walker (2013) are indeed worth pursuing: (a) reducing direct runoff reduces P runoff, (b) reduce soil erosion, (c) use care when applying manure, especially near water, (d) know your soil and manure P levels through testing and analysis and match fertilizer and manure P and forage needs, and (e) do not over-apply fertilizer or manure P on sites adjacent to rivers, streams, lakes, or near sinkholes.

References

APHA. (1989). *Standard Methods for the examination of water and wastewater* (17th ed.). Washington, D.C: Am. Public Health Assoc.

Arnold, J. G., Srinivasa, R., Mutlah, R. S., & Williams, J. R. (1998). Large area hydrologic modeling and assessment Part 1: model development. *Journal of the American Water Resources Association, 49*, 1523–1527.

Asmussen, L. E., Sheridan, J. M., & Allison, H. D. (1975). *Water quality inventory of the Southern Coastal Plain and Atlantic Coastal Flatwoods of Georgia. USDA-ARS Publ. ARS-S-49*. Washington, DC: U.S. Government Printing Office.

Boddey, R. M., Macedo, R., Tarre, R. M., Ferreira, E., de Oliveira, E. C., Rezende, C., et al. (2004). Nitrogen cycling in *Brachiara* pastures: the key to understanding the process of pasture decline. *Agriculture, Ecosystems & Environment, 103*, 389–403.

Bogges, C. F., Flaig, E. G., & Fluck, R. C. (1995). Phosphorus budget basin relationships for Lake Okeechobee tributary basins. *Ecological Engineering, 5*, 143–162.

Botcher, A. B., Tremwel, T. K., & Campbell, K. L. (1999). Phosphorus management in flatwood (Spodosols) soils. In K. R. Reddy et al. (Eds.), *Phosphorus Biogeochemistry in Subtropical Ecosystems* (pp. 405–423). Boca Raton, FL: Lewis Publishers.

Brooks, K. N., Folliot, P. N., Gregersen, N. H. M., & DeBano, L. F. (1997). *Hydrology and the management of watersheds* (2nd ed.). Ames, IA: Iowa State University Press.

Chambliss, C. G. (1999). Florida Forage Handbook. Univ. Florida Coop. Ext. Serv. SP253.

Coale, F. J., & Olear, J. A. (1996). *The relationship between soil test phosphorus level and the concentration of dissolved and potentially transportable phosphorus in field drainage water. Technical Report*. Edgewater, Maryland: Chesapeake Research Consortium, Inc.

Daniel, T. C., Sharpley, A. N., & Lemunyon, J. L. (1998). Agricultural phosphorus and eutrophication: a symposium overview. *Journal of Environmental Quality, 27*, 251–257.

Edwards, D. R., Hutchens, T. K., Rhodes, R. W., Larson, B. T., & Dunn, L. (2000). Quality of runoff from plots with simulated grazing. *Journal of the American Water Resources Association, 36*, 1063–1073.

Florida Department Environmental Protection (2013). Implementation of Florida's numeric nutrient standards. Tallahassee, FL. p. 82. http://www.dep.state.fl.us/water/wqssp/nutrients/.

Ganskopp, D. (2001). Manipulating cattle distribution with salt and water in large arid-land pastures: a GPS/GIS assessment. *Applied Animal Behavior Science, 73*, 251–262.

Gary, H. L., Johnson, S. R., & Ponce, S. L. (1983). Cattle grazing impact on surface water quality in a Colorado Front Range Stream. *Journal of Soil and Water Conservation, 38*(2), 124–128.

Gburek, W. J., & Sharpley, A. N. (1998). Hydrologic controls on phosphorus loss from upland agricultural watersheds. *Journal of Environmental Quality, 27*, 267–277.

Heckrath, G. P., Brookes, C., Poulton, P. R., & Goulding, K. W. T. (1995). Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. *Journal of Environmental Quality, 24*, 904–910.

Hedley, M. J., & Sharpley, A. N. (1998). Strategies for global nutrient cycling. In I. Curie (Ed.), *Long-term nutrient needs for New Zealand's primary industries: Global supply, production requirements, and environmental constraints* (pp. 70–95). Palmerston North, New Zealand: The Fert. Lime Res. Ctr., Massey.

House, W. A., Jickells, T. D., Edwards, A. C., Praska, K. E., & Denison, F. H. (1998). Reactions of phosphorus with sediments in fresh and marine waters. *Soil and Water Conservation, 14*, 139–146.

Hubbard, R. K., & Sheridan, J. M. (1983). Water and nitrate–nitrogen losses from a small, upland, coastal plain watershed. *Journal of Environmental Quality, 12*, 291–295.

Hubbard, R. K., Gascho, G. J., Hook, J. E., & Knisel, W. G. (1986). Nitrate movement into shallow groundwater through a coastal plain sand. *Transactions of ASAE, 29*(6), 1564–1571.

Hyde, A. G., Law, L. Jr., Weatherspoon, R. L., Cheney, M. D., & Eckenrode, J. J. (1977). Soil survey of Hernando County, FL. USDA-NRCS, Washington, DC and University of Florida, Gainesville, FL. 152 p.

Kellogg, R. L., Lander, C. H., Moffit, D. C., & Gollehon, N. (2000). Manure nutrients relative to the capacity of cropland and pastureland to assimilate nutrients: spatial and temporal trends for the United States. USDA-NRCS, Washington, DC, 39 p.

Khaleel, R., Reddy, K. R., & Overcash, M. R. (1980). Transport of potential pollutants in runoff water from land areas receiving animal wastes: a review. *Water Research, 14*, 421–426.

Lemunyon, J. L., & Gilbert, R. G. (1993). The concept and need for a phosphorus tool. *Journal of Production Agriculture, 6*, 483–486.

Martin, S. C., & Ward, D. E. (1973). Salt and meal-salt help distribute cattle use on semi-desert range. *Journal of Range Management, 26*, 94–97.

Odum, E. (1988). Energy flow in ecosystems: a historical review. *American Zoologist, 8*, 11–18.

Owens, L. B., & Shipitalo, M. J. (2006). Surface and subsurface phosphorus losses from fertilized pasture systems in Ohio. *Journal of Environmental Quality, 35*, 1101–1109.

Peterson, P. R. and Gerrish, J. R. (1996). Grazing systems and spatial distribution of nutrients in pastures: livestock management considerations. In: Joost, R. E., & Robert, C. A.

(eds.), Nutrient cycling in forage systems. Potash Phosphate Inst. Found. Agron. Res. Manhattan, KS, p. 203–212.

Romkens, J. J. M., Nelson, W., & Mannering, J. V. (1973). Nitrogen and phosphorus composition of surface runoff as affected by tillage method. *Journal of Environmental Quality, 2*, 292–295.

SAS Institute. (2000). *SAS/STAT User's Guide. Release 6.03*. Cary, North Carolina: SAS Institute. 494 p.

Scoones, I., and Toulmin, C. (1999). Soil nutrient budgets and balances: What use for policy? Managing Africa's Soils No. 6. Africa. p. 23.

Senft, R. L., Rittenhouse, L. R., & Woodmanse, R. G. (1983). The use of regression models to predict spatial patterns of cattle behavior. *Journal of Range Management, 36*(5), 553–557.

Sharpley, A. N. (1997). *Dispelling common myths about phosphorus in agriculture and the environment. Technical Report, Watershed Science Institute, USDA Natural Resources Conservation Service, Aiken Center*. Burlington, Vermont: University of Vermont.

Sharpley, A.N. and Williams, J.R. (1990). EPIC-Erosion/ Productivity Impact Calculator: 1. Model Documentation. USDA Agric. Res. Service, Tech. Bull. No. 1768.

Sharpley, A. N., Smith, S. J., & Jones, O. R. (1992). The transport of bioavailable phosphorus in agricultural runoff. *Journal of Environmental Quality, 21*, 30–35.

Sharpley, A. N., Daniel, T. C., Sims, J. T., & Pote, D. H. (1996). Determining environmentally sound soil phosphorus levels. *Journal of Soil and Water Conservation, 51*, 160–166.

Sharpley, A. N., McDowell, R. W., & Kleinman, J. A. (2001). Phosphorus loss from land to water: integrating agricultural and environmental management. *Plant and Soil, 237*, 287–307.

Sigua, G. C. (2010). Sustainable cow-calf operations and water quality: a review. *Agronomy for Sustainable Development Journal, 30*(3), 631–648.

Sigua, G. C., & Coleman, S. W. (2007). Sustainable management of nutrients in forage-based pasture soils: effect of animal congregation sites. *Journal of Soils and Sediments, 6*(4), 249–253.

Sigua, G. C., & Tweedale, W. A. (2003). Watershed scale assessment of nitrogen and phosphorus loadings in the Indian River Lagoon Basin, FL. *Journal of Environmental Management, 67*(4), 361–370.

Sigua, G. C., Steward, J. S., & Tweedale, W. A. (2000). Water quality monitoring and biological integrity assessment in the Indian River Lagoon, Florida I. Status and trends (1988–1994). *Journal of Environmental Management, 25*, 199–209.

Sigua, G. C., Williams, M. J., & Coleman, S. W. (2004). Levels and changes of soil phosphorus in the subtropical beef cattle pastures. *Communications in Soil Science and Plant Analysis, 35*(7&8), 975–990.

Sigua, G. C., Williams, M. J., Coleman, S. W., & Starks, R. (2006). Nitrogen and phosphorus status of soils and trophic state of lakes associated with forage-based beef cattle operations in Florida. *Journal of Environmental Quality, 35*, 240–252.

Sigua, G. C., Hubbard, R. K., & Coleman, S. W. (2009). Quantifying phosphorus levels in soils, plants, surface water, and shallow groundwater in a bahiagrass pasture of southern Florida. *Environmental Science and Pollution Research, 17*, 210–219.

Silveira, M. L., Vendramini, J. M., & Sollenberger, L. E. (2010). Phosphorus management and water quality problems in grazing land ecosystems. *International Journal of Agronomy*. doi:10.1155/2010/517603.

Sondregaard, M., & Jeppesen, E. (2007). Anthropogenic impacts on lake and stream ecosystems, and approaches to restoration. *Journal of Applied Ecology, 44*, 1089–1094.

Stout, W. L., Weaver, S. R., Bureck, W. J., Folmar, G. J., & Schnabel, R. R. (2000). Water quality implications of dairy slurry applied to cut pastures in northeast USA. *Soil Use and Management, 16*, 189–193.

Sundareshwar, P. V., Morris, J. T., Koepfler, E. K., & Fornwalt, B. (2003). Phosphorus limitation of coastal ecosystem processes. *Science, 299*, 563.

Tamminga, S. (2006). Environmental impacts of beef cattle. pp. 1–11. In The John M. Airy Symposium: Visions for animal agriculture and environment. Kansas City, MO.

Thurow, T. L. (1991). Hydrology and erosion. In R. K. Heitschmidt & J. W. Stuth (Eds.), *Grazing management: an ecological perspective* (pp. 141–160). Portland, OR: Timber Press.

Walker, F. (2013). Best management practices for phosphorus in the environment. Agricultural Extension Service. University of Tennessee. p. 14. https://utextension.tennessee.edu/publications/Documents/PB1645.pdf.

White, S. L., Sheffield, R. E., Washburn, S. P., King, L. D., & Green, J. T., Jr. (2001). Spatial and time distribution of dairy cattle excreta in an intensive pasture systems. *Journal of Environmental Quality, 30*, 2180–2187.

Williams, M. J., & Hammond, A. C. (1999). Rotational versus continuous intensive stocking management of bahiagrass pastures for cows and calves. *Agronomy Journal, 91*, 11–16.

Yan, T., Frost, J. P., Keady, T. W. J., Agnew, R. E., & Mayne, C. S. (2007). Prediction of nitrogen excretion and urine of beef cattle offered diets containing grass silage. *Journal of Animal Science, 85*, 1982–1989.

Young, R. A., Onstad, C. A. and Bosch. (1995). AGNPS: An agricultural nonpoint source model. In computer models of watershed hydrology. V. P. Singh (ed), Highlands Ranch. CO: Water Resources Publ., pp. 1001–1020