Agron. Sustain. Dev. 30 (2010) 631–648
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DOI: 10.1051/agro/2010001

Available online at:
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Review article

Sustainable cow-calf operations and water quality: A review

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(Accepted 18 December 2009)

Abstract – As animal agriculture has evolved to larger production operations in subtropical regions of United States, the problems associated with manure handling, storage and disposal have grown significantly. Understanding the interaction effects of sustainable cow farming with water-table management, nutrient dynamics and water quality in pastures could be the key to reducing nutrients in runoff. Soils do not contribute equally to nutrient export from watersheds or have the same potential to transport nutrient to runoff nor would soil test levels accurately predict total dissolved nutrients. Better understanding of soil nutrient dynamics and crop nutrient changes resulting from different management systems should allow us to predict potential impact on adjacent surface waters. In many states, these issues are critical and of increasing importance among environmentalists, ranchers, and public officials particularly in the case of N and P. One of the first steps in assessing N or P level on any farm is to consider total N or total P inputs and outputs. In Florida, reduction of P transport to receiving water bodies is the primary focus of several studies because P has been found to be the limiting nutrient for eutrophication in many aquatic systems. Long-term monitoring of the changes in soil nutrients, especially soil P would enable us to predict soil chemical or physical deterioration under continuous forage-livestock cultivation and to adopt measures to correct them before they actually happen. Despite substantial measurements using both laboratory and field techniques, little is known about the spatial and temporal variability of nutrient dynamics across the entire landscape, especially in agricultural landscapes with cow-calf operations.

bahiagrass / cow-calf / groundwater / surface water / groundwater / sustainability / nutrient cycling / plant uptake / water quality / subtropics / BMPs / eutrophication / trophic state index (TSI)

Contents

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Article published by EDP Sciences

1. INTRODUCTION

Beef cattle (*Bos taurus*) pastures in subtropical regions of United States and other parts of the world are typically dominated by subtropical and tropical grasses such as bahiagrass (*Paspalum notatum,* Flügge) or bermudagrass (*Cynodon dactylon,* L). The establishment and maintenance of persistent grass-legume pastures is a key option to increase productivity and profitability of beef cattle production systems. The development of effective grass-legume pastures for beef cattle production is a good option to improve nutritional value of the pastures, minimize N or P fertilizers input, and better manage nutrient cycling to enhance water quality. The greatest environmental concern with many grazing areas in Florida is level of soil P due to P accumulation in soil, and the subsequent loss of sediment-bound and soluble P in runoff. Our ability to estimate the levels and changes of soil P and other crop nutrients in subtropical beef cattle pastures has the potential to improve our understanding of P dynamics and nutrient cycling at the landscape level.

Throughout the southeastern United States, grazinglands have considerable variability in soils, climate, and growing season, which not only affect the types of forage that can be grown, but also the overall environmental and biodiversity management. Grazing animals affect the movement and utilization of nutrients through the soil and plant system, and thus the fertility of pasture soils (Haynes and Williams, 1993). Different pasture species may affect nutrient use and turnover due to seasonal timing of growth, root type, and forage types (Haynes, 1981; Wedin and Tilman, 1990; Stout et al., 1997). Nutrient availability at watershed scale may control pasture growth and thus the number of domestic animals that can be supported. Increased nutrient availability, through fertilizer applications, and the subsequent increases in pasture production offer the potential for increased animal production (Haynes and Williams, 1993). Society relies on adequate freshwater resources to support populations of people, agriculture, industry, wildlife habitat, aquatic ecosystems, and a healthy environment. Consequently, the interaction of pasture management and hydrology is important issue to environmentalists, ranchers, and public officials because it may affect nutrient dynamics and water quality.

Forage-beef cattle research programs must adopt an integrated approach that will lead to the development of appropriate sustainable pasture technologies that optimize beef cattle ranching profitability. Thus, both actual and perceived environmental problems associated with beef cattle production systems need to be addressed when new management systems are being developed. A key issue to be evaluated is how different livestock management practices impact the environment, including water quality, flora and fauna biodiversity, and soil and landscape integrity. Another equally important issue concerns the balance of fertility management for forage-livestock agro-ecosystem that may result in increased nutrient use efficiency and, therefore, less likelihood of nutrient loss to the environment due to leaching and/or runoff. Additionally, there is a heightened likelihood of P losses from over fertilized pastures through surface water runoff or percolation past the root zone (Gburek and Sharpley, 1998; Stout et al., 2000).

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) 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.

Recent assessments of water quality status have identified eutrophication as one of the major causes of water quality “impairment” not only in the United States, but also around the world. In most cases, eutrophication has accelerated by increased inputs of P and/or N due to intensification of crop and animal production systems since the early 1990’s. The current high demand for quality protein and fiber production because of increasing world population has resulted in an intensification of agricultural production systems. As animal-based agriculture has evolved to larger production operations in subtropical region of United States, the problems associated with

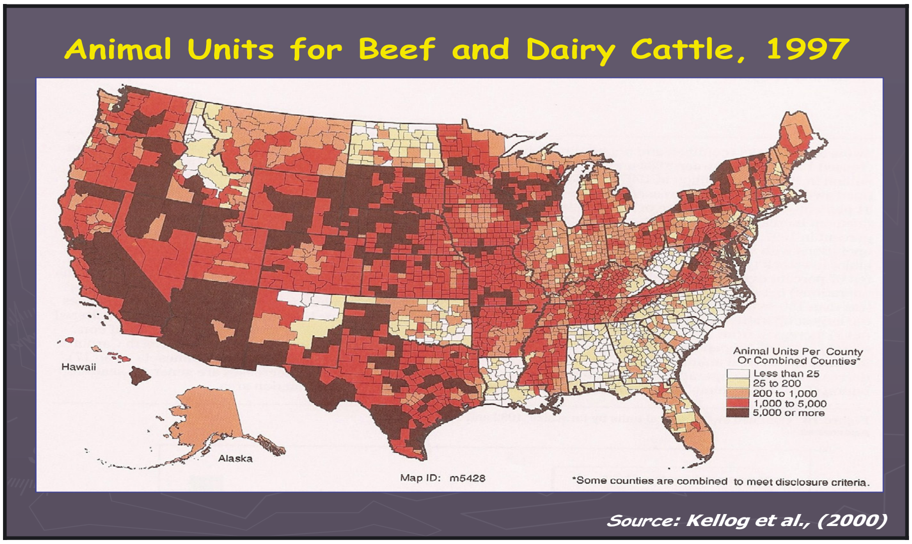

Figure 1. Spatial distribution of animal units for beef and dairy cattle in the United States (*Source: Kellog et al., 2000*).

manure handling, storage and disposal have grown significantly. This review article examines the following two key questions: (1) are forage-based animal production systems as suggested by regulators the major sources of non-point source nutrients pollution that are contributing to the degradation of water quality in lakes, reservoirs, rivers, and ground water aquifers? and (2) is properly managed cow-calf operations in subtropical agro-ecosystem would not likely be the major contributors to excess loads of N or P in surface water and/or shallow groundwater?

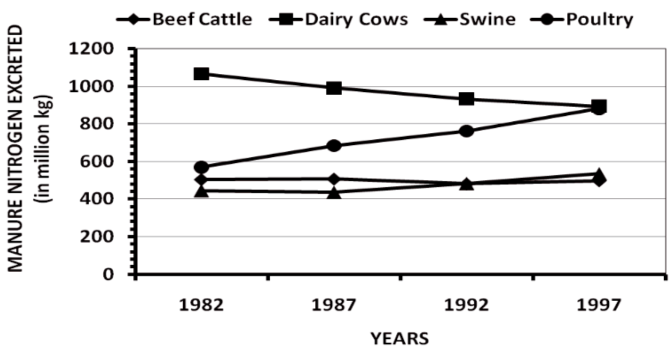

Figure 2. Temporal distribution of manure nitrogen excreted in the United States (*Source: Kellog et al., 2000*).

1.1. Animal numbers, livestock operations and animal manure production in the United States

Data from the census of agriculture were used by Kellog et al. (2000) to make estimates of livestock populations in the United States. A census of agricultural producers is being conducted every five years by the United States Department of Agriculture National Agricultural Statistics Service. The basic building block of the estimation process is an animal unit. For the purposes of this paper, an animal unit represents 1000 pounds or 450 kilograms of live animal weight. An example of animal units for beef and dairy cattle spatial distribution in the United States is shown in Figure 1. For more detailed information related to spatial and temporal trends of animal units in the United States, the published report by Kellog et al. (2000) is highly suggested. They also reported the estimated amount nutrients (i.e., N and P excreted) based on manure production associated for livestock animals units.

The amount of manure N and P excreted in the United States are shown in Figures 2 and 3. The annual amount of manure N and P production from beef cattle operations remained constant from 1982 to 1997 while manure N and manure P production for dairy cattle had shown slightly decreasing trend from 1982 to 1997. On the contrary, the manure N and manure P production from poultry operations showed a remarkable increasing trend from 1982 to 1997 (Figs. 2 and 3).

1.2. Overview: cow-calf operation and management in Florida

Eleven million ha of grazingland in the subtropical (23.5–30° N Lat) United States supports about 30% of the U.S. beef cow herd. Florida's beef production ranks 10th among beef producing states in the United States and 4th nationally among states in number of herds with more than 500 brood cows. Florida's beef cattle had sales of more than \$443 million in 2004. The majority of Florida's cow herd is located in the Kissimmee/Lake Okeechobee watershed, a place where the public is becoming more concerned about high levels of P entering the lake and subsequently flowing out into waterways and the Everglades.

Florida is a large state with a considerable variability in soils and climate. In north Florida, there are some clay-loam

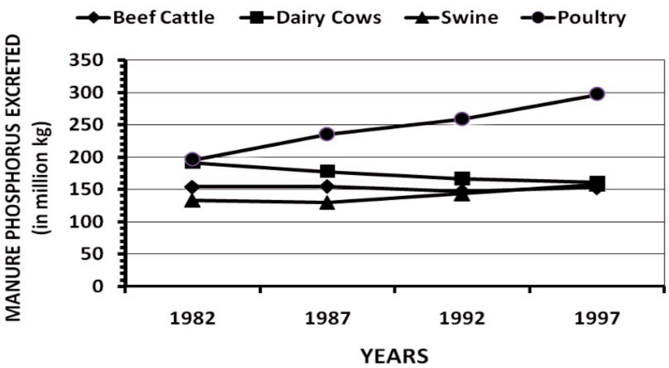

Figure 3. Temporal distribution of manure phosphorus excreted in the United States (*Source: Kellog et al., 2000*).

soils with good moisture-holding capacity that are quite productive (Chambliss et al., 2001). Coming down the peninsula, soils are dominated by sandy ridges and flatwoods. In general, the flatwood soils with their higher moisture-holding capacity are more productive than the upland deep, droughty sands. The warm growing season is longer in south Florida than in north Florida while winter temperatures are usually lower in north Florida than in south Florida. These differences in climate, soils and length of growing season affect the types of forage that can be grown. Nevertheless, Florida's relatively mild climate, together with more than 127 cm of annual rainfall, affords a better opportunity for nearly 12 months of grazing than in any other state except Hawaii (Chambliss et al., 2001).

1.2.1. Pasture management: grazing and fertilization (typical in Florida ranches)

Bahiagrass is a common pasture used for beef cattle across Florida. Fertility and management practices have been based on University of Florida's recommendations as described by Chambliss (1999). Pastures are being grazed during spring of the year. After the start of summer rainy season, pastures that are to be hayed are being dropped out of the grazing cycle (usually starting in July). Pasture fields with bahiagrass are normally fertilized in the spring with 90 kg N ha^{-1} and 45 kg K_2O ha^{-1}. Grazing cattle at the United States Department of Agriculture Subtropical Agricultural Research Station in Brookville, FL and other Florida ranches are being rotated among pastures on a 3-day grazing interval with 24 days of rest between pastures. The average number of grazing cattle was about 2.9 animal units per hectare and grazing days of about 5.5 on monthly basis as shown in Table I. In addition, the number of days grazed each month, average number of animals per hectare and estimated total feces excreted along with the estimated total N in feces and from urine are also shown in Table I.

1.3. Eutrophication associated with animal-based agriculture: overview

Eutrophication is frequently a result of nutrient pollution, such as the release of sewage effluent, urban stormwater runoff, and run-off carrying excess fertilizers into natural waters. Nutrients (e.g., N or P) and other pollutants may enter from a number of sources (Fig. 4). However, it may also occur naturally in situations where nutrients accumulate (e.g. depositional environments) or where they flow into systems on an ephemeral basis. Eutrophication generally promotes excessive plant algal growth and decay, favors certain weedy species over others, and is likely to cause severe reductions in water quality (Schindler, 1974).

In aquatic environments, enhanced growth of choking aquatic vegetation or phytoplankton (that is, an algal bloom) disrupts normal functioning of the ecosystem, causing a variety of problems such as a lack of oxygen in the water, needed for fish and shellfish to survive (Fig. 5). The water then becomes cloudy, colored a shade of green, yellow, brown, or red. Human society is impacted as well; eutrophication decreases the resource value of rivers, lakes, and estuaries such that recreation, fishing, hunting, and aesthetic enjoyment are hindered. Health-related problems can occur where eutrophic conditions interfere with drinking water treatment. Many drinking water supplies throughout the world may experience periodic massive surface blooms of *cyanobacteria* (Kotak et al., 1993). These blooms contribute to a wide range of water-related problems including summer fish kills (Fig. 5), and unpalatability of drinking water (Palmstrom et al., 1988).

Increased loss of nutrients in agricultural runoff has potentially serious ecological and public health implications (Hooda et al., 2000). Nitrogen and P are particularly important as both are implicated in aquatic eutrophication (Levine and Schindler, 1989). Eutrophication and the associated ecological effects result in a general decline in overall water quality, restricting its use for general and drinking purposes (USEPA, 1988; Sharpley and Withers, 1994).

2. FORAGE-BASED COW-CALF OPERATION: EFFECT ON THE ENVIRONMENT AND WATER QUALITY

Beef cattle operations have been suggested as one of the major sources of non-point source P and N pollution that are contributing to the degradation of water quality in lakes, reservoirs, rivers, and ground water aquifers in Florida (Allen et al., 1976, 1982; Bogges et al., 1995; Edwards et al., 2000). Cattle manure contains appreciable amounts of N 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). 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). Additionally, there is a heightened likelihood of N and P losses from overfertilized

Table I. Monthly grazing activity and estimates of nitrogen contributions from cattle feces and urine (*Source: Sigua et al., 2009).*

Months	Average days grazed per pasture	Animal unit per month[1]	Total feces excreted[2]	Total feces nitrogen excreted[3]	Total urine nitrogen excreted[4]	Total nitrogen after losses[5]
			$\text{kg ha}^{-1}\ \text{month}^{-1}$			
January	13.8	1.0	913	5.0	2.8	2.3
February	9.4	0.8	669	3.7	2.0	1.7
March	13.5	0.9	753	4.1	2.5	1.9
April	12.0	0.7	619	3.4	1.9	1.6
May	12.7	0.7	654	3.6	1.9	1.7
June	12.0	0.9	748	4.1	2.5	1.9
July	6.9	0.7	628	3.4	1.9	1.6
August	9.1	0.8	734	4.0	2.5	1.9
September	8.2	1.1	947	5.2	3.0	2.5
October	6.7	1.1	976	5.3	3.1	2.5
November	6.6	0.8	705	3.9	2.2	1.8
December	9.4	1.2	1037	5.7	3.4	2.7
Totals	**–**	**10.8**	**9347**	**51.2**	**27.8**	**24.1**

[1] Animal units per month (450 kg cow/calf unit).

[2] Total feces excreted (kg as excreted) = [(number of AUM × total annual animal feces excretion/12) × total feces excretion (as excreted) per animal per year] = 10.4 metric tons (Kellogg et al., 2000).

[3] Total nitrogen excreted = total feces excreted × percent total nitrogen in feces (0.55%; Kellogg et al., 2000).

[4] Total urine nitrogen excreted = based on urine N (g day per animal with live weight of 381 kg). Values on this table were adjusted for 450 kg cow-calf unit (Yan et al., 2007).

[5] Total nitrogen after losses = based on volatilization correction of 70% (during and after animal excretion).

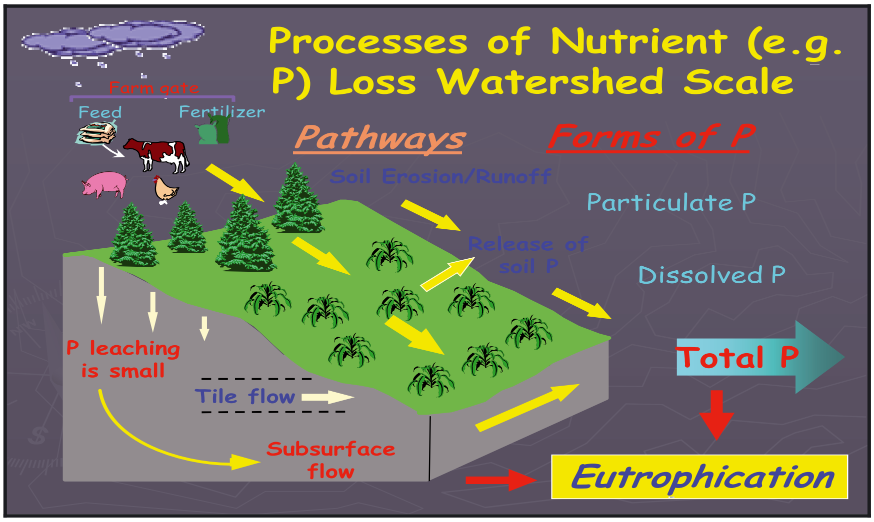

Figure 4. Nutrient losses from farm gate on watershed scale causing eutrophication.

pastures through surface water runoff or percolation past the root zone (Schmidt and Sturgul, 1989; Gburek and Sharpley, 1998; Stout et al., 1998, 2000).

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 Florida aquatic systems (Botcher et al., 1999; Sigua et al., 2000; Sigua and Steward, 2000; Sigua and Tweedale, 2003). Recently, Sigua et al. (2004, 2006) found that the levels of soil P varied widely with different pasture management. Water quality in lakes associated with cattle production was "good", equivalent to 30–46 trophic state index (TSI) based upon the Florida Water Quality Standard. These findings indicate that properly managed livestock operations may not be major contributors to excess loads of nutrients (especially P) in surface water (Sigua et al. 2006). In another study in south Florida, Arthington et al. (2003) reported that the presence of beef cattle at three stocking rates (1.5, 2.6, and 3.5 ha per cow) had no impact on nutrient loads (P and N) in surface runoff water compared with pastures containing no cattle. However, these studies should not be considered as definitive for the region because of the wide range in management options (fertilization,

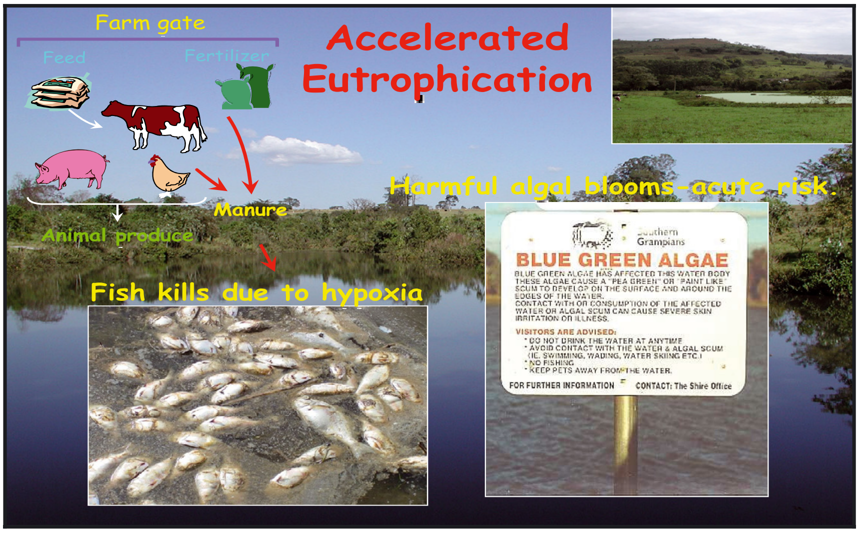

Figure 5. Effect of accelerated eutrophication from farm gate to harmful algal blooms and fish kills due to hypoxia.

stocking rate, grazing system, forage type, etc.) that are used in beef cattle production systems. Whether or not nutrient 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 defined (Gary et al., 1983; Edwards et al., 2000; Sigua et al., 2004). Concern for losses of soil P by overland flow were noted when soil P exceeded 150 mg kg^{-1} in the upper 20-cm of soil (Johnson and Eckert, 1995; Sharpley et al., 1996). 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 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 better 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).

2.1. Impact of grazing cattle, cattle movement, and grazing behavior on water quality and the soil nutrient dynamics around and beneath cattle congregation sites

Understanding cattle movement in pasture situations is critical to assess their impact on agro-ecosystems. Movement of free-ranging cattle varies due to spatial arrangement of forage resources within pastures (Senft et al., 1985) and the proximity of water (Holechek, 1988; Ganskopp, 2001), minerals (Martin and Ward, 1973), and shade to grazing sites. The breed of the animal also affects livestock distribution pattern (Herbel et al., 1967). Hammond and Olson (1994) and Bowers et al. (1995) reported that temperate British breeds (Angus and Hereford) of *Bos taurus* cows grazed less during the day than tropically adapted Senepol cows, but compensated for reduced grazing activity during the hotter parts of the day by increasing time spent grazing at night. Grazing animals congregate close to the shade and watering areas during the warmer periods of the day (Mathews et al., 1994, 1999). White et al. (2001) claimed that there was a correlation between time spent in a particular area and the number of excretions and this behavior could lead to an increase in the concentration of soil nutrients close to shade and water. Sigua (2004) demonstrated that concentrations of total inorganic N, total P and the degree of soil compaction varied significantly among different animal congregation sites. The highest concentrations of total inorganic N and total P were found at the shade and mineral feeder sites, respectively. The most compacted soil was at the mineral feeders' site. Although the levels of total inorganic N and total P were high near the center of the congregation sites, their levels did not increase with soil depth and their concentrations decreased almost linearly away from the center of the congregation sites. Soil compaction tended to decrease away from the center of the mineral feeder sites, but not at the water trough or shade sites. This study suggest that congregation sites in beef cattle operations in Florida are not as nutrient rich as suspected, and may not contribute more nutrients to surface and groundwater supply under Florida conditions because P levels at the center of sites were below 150 mg kg^{-1} (Sigua, 2004). This concentration of soil P should not be considered an absolute maximum number for soil P to become harmful to water quality and the environment, but rather a good indicator of P accumulation in the soil. Furthermore, since there was no evidence of a vertical build up or horizontal movement of inorganic N or total P in the landscape, Sigua et al. (2005) surmised that cattle congregation sites may not be considered a substantial source of nutrients at the watershed level.

Grazing animals impact the movement and cycling of nutrients through the soil and plant system, and thus on the fertility of pasture soils (Haynes, 1981; Haynes and Williams, 1993). Grazing can accelerate and alter the timing of nutrient transfers, and increase amounts of nutrients cycled from plant to soil (Klemmendson and Tiedemann, 1995). Long periods of time, position of shade and water resources for grazing cattle can influence the spatial distribution of soil biochemical properties including soil organic C and N, particulate organic C and N, microbial biomass, and net N mineralization (Ruess and McNaughton, 1987; Kieft, 1994; Kieft et al., 1998; Franzluebbers et al. 2000). Long-term intensive grazing may decrease the input of organic matter into soils in the immediate vicinity of individual plants and eventually reduce nutrient concentrations beneath plants by limiting availability of photosynthesis and/or meristematic tissues necessary for growth (Milchunas and Lauenroth, 1993; Briske and Richards, 1995). Thrash (1997) reported that concentration of large herbivores around the troughs negatively impacted the infiltration rate of the soils, with implications for the rate of soil loss and the soil moisture regime. Elsewhere, grazing, trampling, and dung deposition by large herbivores often result in a zone of decreasing impact on many vegetation and soil parameters including herbaceous vegetation basal cover, soil bulk density, and penetrability away from water points (Andrew and Lange, 1986; Thrash et al., 1991). The effect of trampling appears to be less severe on vegetated grasslands than on poor or bare soil (Warren et al., 1986). Studies on grazing and soil compaction generally find that exposure to livestock grazing compacts soil and that soil compaction increases with grazing intensity. This pattern is reflected in reviews of the scientific literature on the subject (Fleischner, 1994; Lauenroth et al., 1994; Kauffman and Krueger, 1984).

2.2. Effects of forage type and harvest method on above ground net production, N and P uptake, and soil C, P, and N dynamics in subtropical pastures

The productivity of any ecosystem depends on the amounts of nutrients stored in various compartments, such as vegetation, litter, soil, and animal biomass, and on the rates of nutrient cycling and transfer among those compartments. The cycling of nutrients in a given ecosystem is affected by a combination of biological and physical processes (Holt and Coventry, 1990). The relative importance of these processes varies considerably between ecosystems as a result of differences between climate, soils, vegetation, and management practices. Nutrient dynamics in various agro-ecosystems are continually evolving in response to changing management practices. Soil dynamics may continue to change in response to external abiotic perturbations such as global changes in temperature, precipitation, and CO_2 concentration (Janzen et al., 1997, 1998).

Different pasture species affect nutrient use and turnover due to seasonal timing of growth, root type, and forage types (Wedin and Tilman, 1990; Stout et al., 1997). Other researchers (Anderson and Coleman, 1985; Dormaar, 1992) also have attributed the amount of grassland soil organic matter to the amount of root biomass. Vegetation grazed by livestock is rapidly decomposed during digestion and many nutrients are returned to the soil in readily available forms in feces and urine (Laurenroth et al., 1994).

3. CASE STUDIES: EFFECTS OF GRAZING ON SURFACE WATER/SHALLOW GROUNDWATER QUALITY AND SOIL QUALITY (FLORIDA EXPERIENCES)

3.1. Case study #1: surface water quality

3.1.1. Experimental methods

The lakes that we studied were adjacent to or within about 14-km away from the USDA-ARS, Subtropical Agricultural Research Station (STARS), Brooksville, FL (Fig. 6). These lakes are associated with forage-based beef cattle operations. The lakes were (1) Lake Lindsey (28°37.76'N, 82°21.98'W), adjacent to STARS; (2) Spring Lake (28°29.58'N; 82°17.67'W), about 10 km away from STARS; and (3) Bystre Lake (28°32.62'N; 82°19.57'W), about 14 km away from STARS.

Monthly water quality monitoring of lakes associated with beef cattle pastures was begun in 1993 and continued until 2002 by the field staff of the Southwest Florida Water Management District (SWFWMD). Monthly water samples were taken directly from the lakes using a water (Van Dorn) grab sampler. Water quality parameters monitored were Ca, Cl, NO_2 + NO_3-N, NH_4-N, total N, total P, K Mg, Na, Fe, and pH. All sampling, sample preservation and transport, and chain of custody procedures were performed in accordance with an EPA-approved quality assurance plan with existing quality assurance requirements (USEPA, 1979; APHA, 1992). The SWFWMD Analytical Laboratory, using EPA-approved analytical methods, performed the chemical analyses of water samples from the lakes (USEPA, 1979).

Trophic state index development and calculation. Lake trophic state index (TSI) is understood to be the biological response of a lake to forcing factors such as nutrient additions. Nutrients promote growth of microscopic plant cells (phytoplankton) that are fed upon by microscopic animals (zooplankton). The TSI of Carlson (1983) uses algal biomass as the basis for trophic state classification (e.g. oligotrophic, mesotrophic, eutrophic, and hypereutrophic).

The Florida trophic state index of Brezonik (1984), which was derived using data from 313 Florida Lakes, was modified in the study. The first step involved in assigning a TSI value was to assess the current nutrient status based on TN/TP ratio of the lake. The TN to TP ratios were classified into three categories namely: N limited (TN/TP < 10:1); P limited (TN/TP > 30:1); and balanced (10:1 ≤ TN/TP ≤ 30:1). The TSI of each lake that we studied was calculated by entering key water quality parameters: total P (TP, ug L^{-1}); total N (TN, mg L^{-1}); chlorophyll a (CHL, mg m^{-3}) for the measurements of planktonic algae density, and Secchi depth (SD, m) for measuring water transparency into an empirical formula (Eqs. (1) to (3)). Equation (1) (P-limited) was used to calculate the TSI values for Lake Lindsey and Spring Lake while equation (2) (N-limited) was used to calculate the TSI value for Bystere Lake. A novel paper on trophic state index for lakes written by Carlson (1977) was an excellent reference to explain the mathematical derivations of the

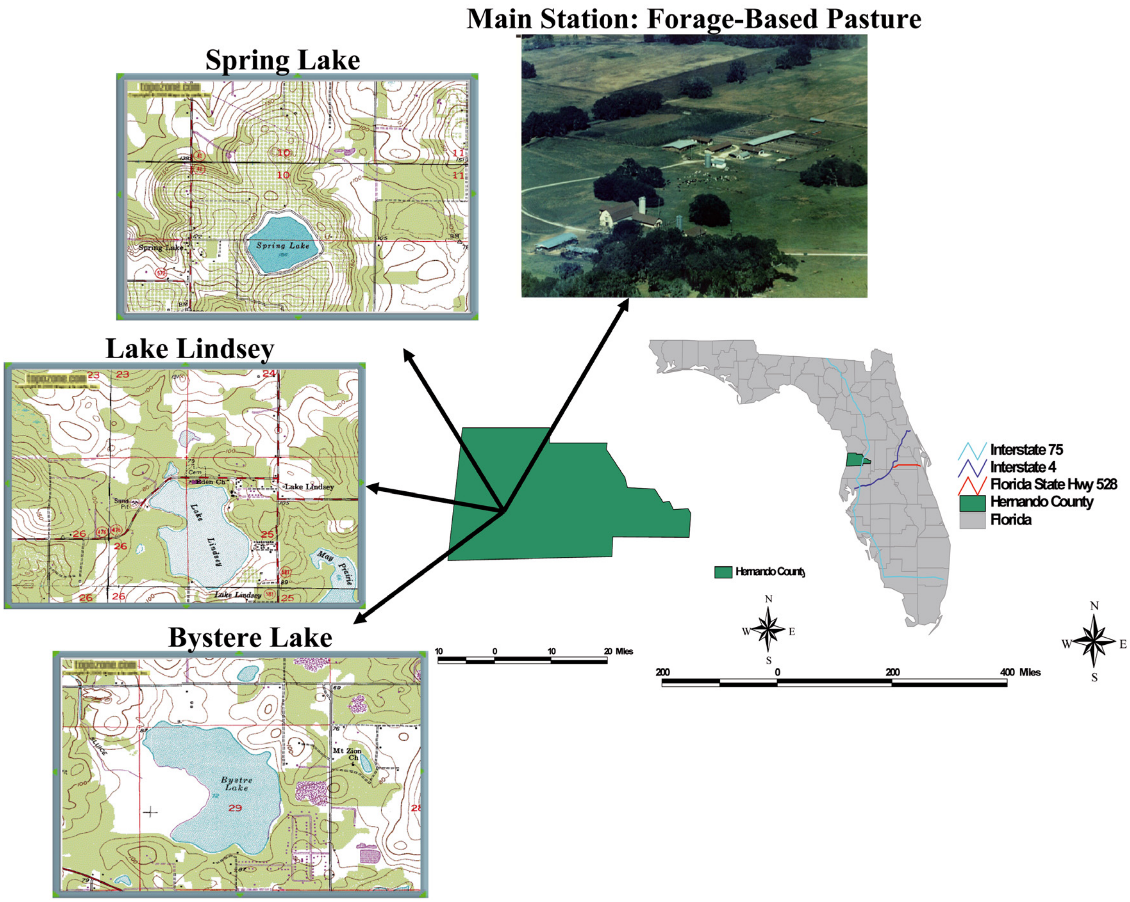

Figure 6. Location of the study sites and aerial view of the Subtropical Agricultural Research Station, Brooksville, Hernando County, Florida, USA.

different equations listed below.

I. **PHOSPHORUS - LIMITED LAKES (TN/TP > 30:1) (1)**
TSI (AVG) = 1/3 [TSI (CHL) + TSI (SD) + TSI (TP)]
where: TSI (CHL) = 16.8 + 14.4 * ln (CHL), (mg m^{-3})
TSI (SD) = 60.0 – 30.0 * ln (SD), (m)
TSI (TP) = – 23.8 + 23.6 * ln (TP), (ug L^{-1})

II. **NITROGEN - LIMITED LAKES (TN/TP < 10:1) (2)**
TSI (AVG) = 1/3 [TSI (CHL) + TSI (SD) + TSI (TN)]
where: TSI (CHL) = 16.8 + 14.4 * ln (CHL), (mg m^{-3})
TSI (SD) = 60.0 – 30.0 * ln (SD), (m)
TSI (TN) = 59.6 + 21.5 * ln (TN), (ug L^{-1})

III. **NUTRIENT - BALANCED LAKES**
(10:1 ≤ TN/TP ≤ 30:1) (3)
TSI (AVG) = 1/3 [TSI (CHL) + TSI (SD) + (0.5 TSI (TP) + TSI (TN))]
where: TSI (CHL) = 16.8 + 14.4 * ln (CHL), (mg m^{-3})
TSI (TN) = 56 + 19.8 * ln (TN), (ug L^{-1})
TSI (TP) = – 18.4 + 18.6 * ln (TP), (ug L^{-1})
TSI (SD) = 60.0 – 30.0 * ln (SD), (m)

3.1.2. Highlights of research results

Status of water quality in lakes. Assessment of water quality data from 1993 to 2002 confirms that water–quality variations (temporal and spatial) existed in lakes associated with beef cattle pasture systems in Central Florida. The lakes were found to differ from each other in Ca, NO_2+NO_3-N, TN, TP, K, Mg, Na, and Fe. Significant temporal variations were observed for NH_4-N, TP, and Fe while significant interaction effects (lakes × year) were only noted for NH_4-N, TP, Mg, Na, and Fe.

The levels of NO_2+NO_3-N and NH_4-N in lakes did not show any significant differences from 1993 to 2002 while TN of Bystere Lake declined from 1.12 to 0.76 mg L^{-1} between 1993 and 2002. Lake Lindsey's TN in 1993 was about 0.82 mg L^{-1} and 0.80 mg L^{-1} in 2002 while Spring Lake's TN in 1993 and 2002 were 0.65 and 0.78 mg L^{-1}, respectively. The levels of TP in Bystere Lake between 1993 and 2002 increased from 0.08 to 0.34 mg L^{-1} while levels of TP in Lake Lindsey did not change from 1993 to 2002. A decline of TP was noted in Spring Lake from 1993 (0.19 mg L^{-1}) to 2002 (0.01 mg L^{-1}). With the continuous conversion of cropland and pastureland to residential use (although at slow pace), contribution of nutrients from anthropogenic sources is becoming a big concern environmentally over time for lakes associated with forage-based pasture systems.

Total nitrogen (TN)/total phosphorus (TP) ratio. Nitrogen and P are the primary crop nutrients that can impact the environment. When applied in excess of crop needs, nutrients can run off into surface waters resulting in excessive aquatic plant growth and toxicity to certain fish species. The TN/TP ratio may be a useful method to establish the N and P reduction targets in the environment (Sigua et al., 2000). The ratio of TN to TP is one of the important components in calculating the TSI of lakes.

Several studies have shown that a TN/TP ratio $\leq$ 10:1 appears to favor algal blooms, especially blue-green algae, which are capable of fixing atmospheric N (Schindler, 1974; Chiandini and Vighi, 1974; Sakamoto, 1966; Sigua et al., 2000). Figure 7 shows the TN/TP ratio of the lakes that were associated with beef cattle operations. Lake Lindsey and Spring Lake can be classified as P-limited lakes with TN/TP ratios of 41:1 and 51:1, respectively. Bystere Lake with a TN/TP ratio of 9:1 was classified as an N-limited lake and may have higher probability for algal bloom compared with Lake Lindsey and Spring Lake because of its higher P levels. From 1993 to 2002, the TP in Bystere Lake increased from 0.08 mg L^{-1} to 0.34 mg L^{-1}.

Trophic state index of lakes (TSI). The Florida TSI was devised to integrate different but related measures of lake productivity or potential productivity, into a single number that ranges from 0 to 100. The measures included in the calculation of TSI are water transparency (Secchi depth), chlorophyll *a* (measurement of algae content), TN, and TP. The Florida TSI for Lake Lindsey, Spring Lake, and Bystre Lake were 35, 30, and 46, respectively (Fig. 8). Based on this, the TSI of these lakes can be classified as "good" according to Florida water quality standard (TSI of 0–59 = "good"; TSI of 60 to 69 = "fair"; and TSI of 70 to 100 = "poor"). Although the TSI levels of the three lakes did not show any significant change from 1993 to 2002, TSI levels increased numerically for all lakes (Fig. 8). This is reflected in a change in the trophic status of Bystere Lake. Lake Lindsey with TSI of 31 and 38 in 1993 and 2002, respectively, remained within the mesotrophic classification, while Spring Lake with TSI of 25 and 26 in 1993 and 2002, respectively, remained in the oligotrophic category. Lake Lindsey (mesotrophic lake) would normally have moderate nutrient concentrations with moderate growth of algae and/or aquatic macrophytes and with clear water (visible depth of 2.4 to 3.9 m).

Oligotrophic lake such as Spring Lake would normally have less abundance of aquatic macrophytes and algae, or both because nutrients are typically in short supply. Oligotrophic lakes tend to have water clarity greater than 3.9 m due to low amounts of free-floating algae in the water column.

Bystere Lake, which was at the upper end of the mesotrophic range in 1993 (TSI of 49), shifted into the slightly eutrophic state in 2002 with a TSI value slightly above 50. Eutrophic lakes normally have green, cloudy water, indicating lots of algal growth in the water. Water clarity of most eutrophic lakes generally ranges from 0.9 to 2.4 m. Generally, water quality in Lake Lindsey and Spring Lake was consistently good (1993–2002) while water quality of Bystere Lake ranged from good in 1993 to fair in 2002 (Fig. 8).

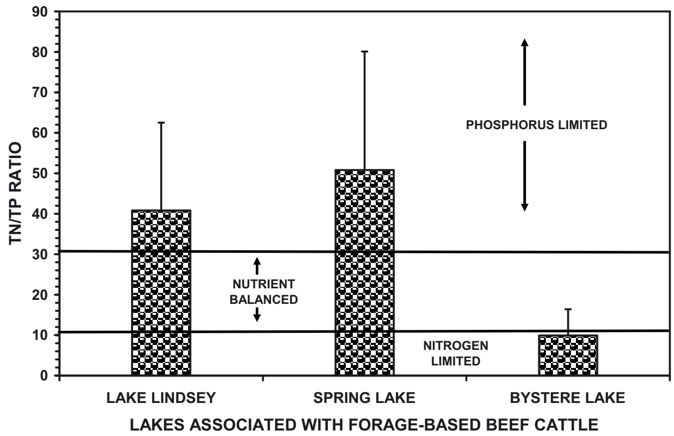

Figure 7. Total nitrogen (TN) to total phosphorus (TP) ratio of lakes associated with forage-based beef cattle pasture system (*Source: Sigua et al., 2006c).*

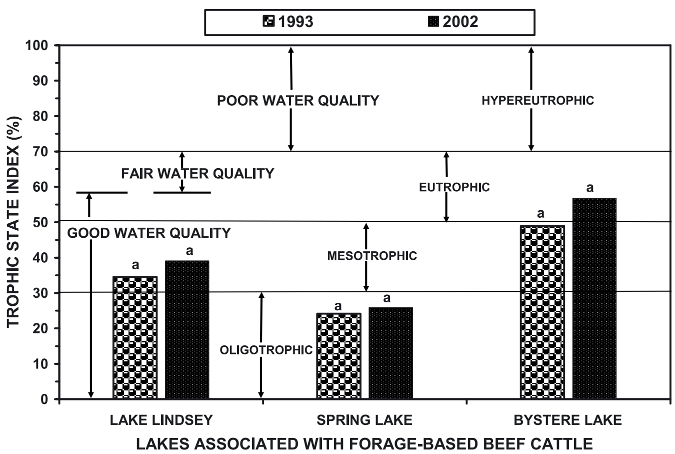

Figure 8. Trophic state index for lakes with forage-based beef cattle pasture system. Trophic state index is sginficantly different ($P \leqslant 0.05$) when superscripts located at top bars are different (*Source: Sigua et al., 2006c).*

3.2. *Case study #2: shallow groundwater quality*

3.2.1. *Experimental methods*

Instrumentation and water sample collection. Two adjacent 8-ha pasture fields were instrumented with a pair of shallow wells placed at different landscape positions. The different landscape positions are top slope (TS; 10–20% slope, 2 ha; middle slope (MS; 5–10% slope, 2 ha and bottom slope (BS; 0–5% slope, 2 ha). The wells were constructed of 5 cm schedule 40 PVC pipe and had 15 cm of slotted well screening at the bottom. During installation of wells, sand was placed around the slotted screen, and bentonite clay was used to backfill to the soil surface to prevent surface water or runoff from moving down the outside of the PVC pipe and contaminating groundwater samples. A centralized battery-operated peristaltic pump was used to collect water samples. Wells were completely evacuated during the sampling process to ensure that water for the next sampling would be fresh groundwater. Water samples were collected from the groundwater wells every two weeks. However, there were periods when ground water levels were below the bottom level of the wells and samples could not be obtained. In addition to ground water samples, surface water samples were collected in the pasture bottoms or the seep area when present, by taking composite grab samples on the same schedule. The seep area, which is located at the lower end of BS is a remnant of a sinkhole formation and became a small scale lake with varying levels of surface water. The seep 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 samples were analyzed for NO_3-N and NH_4-N using a Flow Injector Analyzer according to standard methods (APHA, 1989).

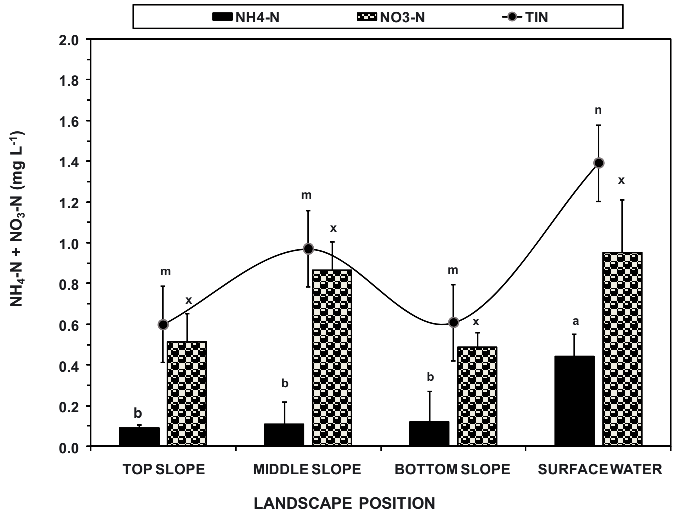

Figure 9. Average concentrations of NH_4-N, NO_3-N and TIN in shallow groundwater at different landscape positions. Line above the bars and across the line represents standard error of the mean. Means of NH_4-N (a & b), NO_3-N(x & y) and TIN (m & n) in shallow groundwater are significantly different ($P \leqslant 0.05$) when superscript located at top bar and line is different (*Source: Sigua et al., 2009a*).

3.2.2. Highlights of research results

3.2.2.1. ***Concentration of NH_4-N, NO_3-N and TIN in surface and ground water***

Concentrations of NH_4-N, NO_3-N, and TIN in shallow groundwater did not vary with landscape positions (Fig. 9). However, concentrations NH_4-N, NO_3-N, and TIN in the water samples collected from the seep area were significantly ($P \leqslant 0.05$) higher when compared to their average concentrations in water samples collected from the different landscape positions (Fig. 9). Averaged across year, concentration of TIN ranged from 0.5 to 1.5 mg L^{-1}. The highest TIN concentration occurred ($P \leqslant 0.05$) in the surface water while the concentrations from the shallow groundwater wells (BS-0.6 mg L^{-1}, MS-0.9 mg L^{-1}, and TS-0.6 mg L^{-1}) were similar to each other and lower than the seepage area (Fig. 9).

Average concentrations of NO_3-N (0.4 to 0.9 mg L^{-1}) among the different sites were well below the maximum, of 10 mg L^{-1}, set for drinking water (Fig. 9). On the average, the concentrations of NO_3-N did not vary significantly with landscape positions, and as with TIN, the levels were significantly lower than surface water from seepage area (Fig. 9). The maximum NO_3-N concentrations (averaged across landscape position) in shallow groundwater for 2004, 2005 and 2006 were also below the drinking water standards for NO_3-N. Other summary statistics for the levels of NO_3-N, NH_4-N and TIN in shallow groundwater are shown in Table II.

Similar trends in landscape positions were found for average concentrations of NH_4-N (Fig. 9). Again, the concentrations of NH_4-N in shallow groundwater did not vary significantly among top slope, middle slope, and bottom slope wells. These levels of NH_4-N were lower than that of the surface water (0.5 mg L^{-1}). Annual average concentrations of NO_3-N, NH_4-N and TIN in shallow groundwater of pastures associated with beef cattle operations did not vary significantly with the different year of sampling. The highest concentration of TIN was in 2005 followed by 2006 and 2004 (Tab. II).

Average concentrations of NO_3-N in surface water and shallow groundwater (0.4 to 0.9 mg L^{-1}) among the different sites did not exceed the drinking water standard for NO_3-N (10 mg L^{-1}). Nitrate levels in excess of 10 mg L^{-1} in drinking water can cause health problems for human infants, infant chickens and pigs, and both infant and adult sheep, cattle and horses. The relatively constant concentration of NO_3-N in the shallow groundwater over the three years (2004–2006) could be due to the combined effects of precipitation, N fertilization, and leaching from plant decomposition and animal feces and urine. Uptake of N by the actively growing bahiagrass reduced the amount of mineral N remaining vulnerable to leaching during the growing season (Decau et al., 2003).

3.2.2.2. ***Concentration of total phosphorus and degree of phosphorus saturation in soils***

Concentrations of total P in soils varied significantly ($P \leqslant 0.001$) with landscape position and sampling depth, but there was no interaction effect of landscape position and sampling depth (Tab. III). Soil samples from the seep area had the lowest concentration of total P when compared with other landscape positions. Soils from the middle slope (9.2 ± 1.8 mg kg^{-1}) had the greatest concentration of total P followed by top slope (5.9 ± 1.8 mg kg^{-1}) and bottom slope (5.7 ± 1.5 mg kg^{-1}). Averaged across years, total P in the soil was about 9.1 mg kg^{-1}.

Table II. Summary statistics for the concentration of inorganic nitrogen in shallow groundwater under bahiagrass pastures associated with cow-calf operations.

Statistical parameters	NO_3-N	NH_4-N	Total inorganic N
	2004		
Number of samples (n) (N)	40	40	40
Mean (mg L^{-1})	0.7	0.1	0.8
Median (mg L^{-1})	0.4	0.04	0.6
Mode (mg L^{-1})	0.2	0.04	0.2
Maximum (mg L^{-1})	4.7	1.3	5.9
Minimum (mg L^{-1})	0.2	0.03	0.2
Std. error mean	0.2	0.03	0.2
Variance	0.9	0.04	0.9
Skewness	3.2	4.5	3.2
	2005		
Number of samples (n)	45	45	45
Mean (mg L^{-1})	0.8	0.3	1.03
Median (mg L^{-1})	0.3	0.1	0.7
Mode (mg L^{-1})	0.1	0.1	0.3
Maximum (mg L^{-1})	4.1	2.8	6.9
Minimum (mg L^{-1})	0.1	0.1	0.2
Std. error mean	0.1	0.1	0.1
Variance	0.8	0.2	1.0
Skewness	2.1	4.6	2.1
	2006		
Number of samples (n)	10	10	10
Mean (mg L^{-1})	0.7	0.2	0.8
Median (mg L^{-1})	0.4	0.1	0.6
Mode (mg L^{-1})	-	0.1	-
Maximum (mg L^{-1})	2.7	0.4	3.1
Minimum (mg L^{-1})	0.1	0.1	0.2
Std. error mean	0.2	0.03	0.2
Variance	0.6	0.01	0.6
Skewness	0.8	1.9	1.8

Source: Sigua et al., 2009a.

Degree of P saturation in the soils varied significantly ($P \leqslant 0.001$) with landscape position and sampling depth, but was not affected significantly by the interaction of landscape position and sampling depth (Tab. III). The middle slope position (19.9 ± 4.9 %) had the highest degree of P saturation followed by top slope, bottom slope and seep area. Soils collected at sampling depth of 0–20 cm (20.9 ± 6.1 %) had significantly higher degree of P saturation than soils collected between 20 and 100 cm.

There was a significant ($P \leqslant 0.05$) decrease in the average concentrations of total P with increasing sampling depth (Tab. III). The upper two depths (0–20 cm and 20–40 cm) had the highest concentrations while the lowest amount of total P was found in the lowest sampling depth of 60–100 cm (Tab. III). These results suggest that there had been little movement of total P into the soil pedon since average degree of P saturation in the upper 20 cm was 21% while degree of P saturation at lower soil depth (60–100 cm) was about 3%.

Results indicate that current pasture management including cattle rotation in terms of grazing days and current fertilizer (inorganic + manures + urine) application rates for bahiagrass pastures offer little potential for negatively impacting the environment. Properly managed livestock operations contribute negligible loads of total P to shallow groundwater and surface water. Overall, there was no buildup of soil total P in bahiagrass-based pasture. Therefore, results of this study may help to renew the focus on improving fertilizer efficiency in subtropical beef cattle systems, and maintaining a balance of P removed to P added to ensure healthy forage growth and minimize P runoff.

Table III. Average concentration (± std. error of mean) and F-values of total phosphorus and degree of phosphorus saturation in soils at various landscape positions and soil depths.

Soil parameters	Total phosphorus	Degree of phosphorus saturation
	— mg kg^{-1}—	—— % ——
A. Landscape position		
1. Top slope	5.91 ± 1.77b*	14.79 ± 4.57a
2. Middle slope	9.19 ± 1.77a	19.92 ± 4.97a
3. Bottom slope	5.67 ± 1.53b	7.86 ± 2.14b
4. Seep area	0.38 ± 0.13c	1.25 ± 0.73c
$LSD_{(0.05)}$	2.62	6.35
B. Soil depth (cm)		
1. 0–20	7.05 ± 1.86a	20.93 ± 6.11a
2. 20–40	9.05 ± 2.18a	13.77 ± 3.22b
3. 40–60	3.24 ± 0.94b	5.64 ± 1.67c
4. 60–100	1.81 ± 0.62b	3.47 ± 1.06c
$LSD_{(0.05)}$	2.62	6.35
	F-values	**F-values**
Landscape position (LP)	17.37***	14.77**
Soil depth (SD)	14.66**	14.24**
LP × SD	2.13^{ns}	2.28^{ns}

* Means in columns within each subheading followed by common letter(s) are not significantly different from each other at $P \leqslant 0.05$.
** Significant at $P \leqslant 0.001$; ns – not significant.

3.3. Changes in soil P, K, Ca, Mg and pH associated with cow-calf operation

During the last 15 years (1988–2002), concentrations for P, K, Ca, Mg, and soil pH have declined by about 7%, 38%, 46%, 61%, and 23% in pastures with bahiagrass and 27%, 55%, 76%, 56%, and 22% for pasture fields with bahiagrass + Rhizoma peanuts that were grazed in spring and hayed in early fall, respectively (Tab. IV). However, the levels of Ca:Mg ratio in fields with grazed and hayed bahiagrass and grazed and hayed Rhizoma peanuts had increased by about 42% and 29%, respectively. It is worth mentioning that depletion rates of P, K, Ca, and Mg were greater in pasture fields with bahiagrass + Rhizoma peanuts than in pastures with bahiagrass suggesting that the former has greater nutritional demands.

Long-term monitoring on the changes in soil P and other crop nutrients in subtropical foraged-based beef cattle pastures enabled to predict soil chemical deterioration and/or soil accumulation of nutrients that could occur under continuous forage-livestock cultivation and to adopt measures to correct them before they actually happen. In addition, long-term

Table IV. Changes (%) in Mehlich-1 extractable soil P and other nutrients under different pasture management in STARS, Brooksville, FL in 2002 relative to the 1988 levels of Mehlich-1 extractable soil P and other nutrient (*Source: Sigua et al., 2006b*).

Soil nutrients	Pasture management	
	BG+GZHY (%)	RP-G+GZHY (%)
P	–6.6	–26.9
K	–38.2	–55.1
Ca	–45.8	–75.6
Mg	–61.2	–56.0
Ca/Mg ratio	+42.1	+29.0
pH	–22.1	–21.9

BG – bahiagrass; RP-G – rhizoma peanuts-mixed grass; GZHY – grazing + haying.

monitoring also provided answers about the nutrient dynamics/cycling and turnover in legume-grass mixtures and the efficiency of fertilizer use. The knowledge that was gained on the relationship of temporal and spatial changes in soil nutrient levels in forage-based beef cattle pasture should indeed provide insights for improved grazing management, which could be both economically and environmentally safe.

4. MANAGEMENT OPTIONS: MANAGING PROPERLY COW-CALF OPERATIONS TO IMPROVE WATER QUALITY AND PASTURE SUSTAINABILITY

4.1. Improving pasture sustainability

The cow-calf (*Bos taurus*) industry in subtropical United States and other parts of the world depends almost totally on grazed pasture areas. Thus, the establishment of complete, uniform stand of bahiagrass in a short time period is economically vital. Failure to obtain a high-quality bahiagrass stand early means the loss of not only the initial investment costs, but also both production and its cash value. Forage production often requires significant inputs of lime, N fertilizer and less frequently of P and K fertilizers. Domestic wastewater sludge or sewage sludge, composted urban plant debris, waste lime, phosphogypsum and dredged materials are examples of materials that can be used for fertilizing and liming pastures. Beef cattle producers throughout the United States need better forage management systems to reduce input costs and protect environmental quality.

The ability to reuse lake-dredged and domestic sewage sludge materials for agricultural purposes is important because it reduces the need for offshore disposal and provides an alternative to disposal of these materials in landfills that are already overtaxed. Often these materials can be obtained at little or no cost to the farmers or landowners. Thus, forage production offers an alternative to waste management since nutrients in the lake-dredged materials and biosolids are recycled into crops that are not directly consumed by humans. Results have shown the favorable influence that biosolids and lake-dredged materials had on bahiagrass during its early establishment in sandy subtropical beef cattle pasture areas in south central Florida. Some of the promising effects of added biosolids and lake-dredged materials on soil quality and on early establishment of bahiagrass are summarized below.

4.1.1. Biosolids as nutrient source

Biosolids usually are applied at agronomic rates designed to supply crops with adequate N nutrition. Biosolids contain a substantial amount of N (typically 3 to 6 % by weight). The N is not immediately available to crops, but is released slowly by biological activity. Since biosolids are produce and handled by different processes at different treatment plants, it is important to know if those treatment processes affect how much N becomes available to plants. Nutrients in municipal residuals produced annually in the United States account for about 2.5% of the total N, 6% of the P, and 0.5% of the K applied on farms each year (Muse et al., 1991).

The field experiment was conducted at the University of Florida Agricultural Research and Education Center, Ona, FL (27°26'N, 82°55'W) on a Pomona fine sandy soil. With the exception of the control, bahiagrass plots received annual biosolids and chemical fertilizers applications to supply 90 or 180 kg total N ha^{-1} yr^{-1} from 1997 to 2000. Land application of biosolids and fertilizer ceased in 2001 season. In early April 1998, 1999, and 2000, plots were mowed to 5-cm stubble and treated with the respective N source amendments. The experimental design was three randomized complete blocks with nine N-source treatments: ammonium nitrate (AMN), slurry biosolids of pH 7 (SBS7), slurry biosolids of pH 11 (SBS11), lime-stabilized cake biosolids (CBS), each applied to supply 90 or 180 kg N ha^{-1}, and a nonfertilized control (Control). Application rates of biosolids were calculated based on the concentration of total solids in materials as determined by the American Public Health Association SM 2540G methods (APHA, 1989) and N in solids. The actual amount of biosolids applications was based on the amount required to supply 90 and 180 kg N ha^{-1}. Sewage sludge materials were weighed in buckets and uniformly applied to respective bahiagrass plots.

Forage yield of bahiagrass was significantly ($P \leqslant 0.001$) affected by the different biosolids in all years (1998 to 2002), but not by the interaction effects of year × treatments. Although yield trend was declining from 1988 to 2002, forage yield of bahiagrass that received biosolids were consistently and significantly ($P \leqslant 0.05$) greater than the forage yield of the unfertilized bahiagrass (Tab. V). The bahiagrass fertilized with SBS11-180 had the greatest forage yield in 1998 (5.1 ± 0.4 Mg ha^{-1}), 1999 (4.6 ± 0.2 Mg ha^{-1}), 2000 (4.5 ± 0.2 Mg ha^{-1}), and in 2002 (3.3 ± 0.6 Mg ha^{-1}). Forage yield of bahiagrass fertilized with AMN-90 and AMN-180 was significantly greater than those of the unfertilized bahiagrass in 1998 and 1999, but not in 2000 and in 2002. Although SBS11-180 had the greatest residual effect (170%) in 2002, CBS-90 and CBS-180 had more pronounced effects when compared with the other sewage sludge sources because their relative impact on

Table V. Comparison on forage yield (Mg ha^{-1}; mean ± S.D.) of bahiagrass among years with repeated application of biosolids (1998, 1999, and 2000) and with no biosolids application (2002) *(Source: Sigua, 2008).*

Nitrogen sources*	With sewage sludge			Without sewage sludge
	1998	1999	2000	2002
Control	2.4 ± 0.5d**	1.8 ± 0.2c	1.4 ± 0.3d	1.2 ± 0.2c
AMN-90	4.3 ± 0.2ab	3.7 ± 0.1b	2.1 ± 0.1cd	2.1 ± 0.3bc
AMN-180	4.7 ± 0.4a	4.7 ± 0.02a	3.2 ± 0.3b	2.2 ± 0.4b
SBS7-90	4.4 ± 0.4ab	3.1 ± 0.3b	2.2 ± 0.4bcd	2.5 ± 0.5ab
SBS7-180	5.0 ± 0.5a	5.1 ± 0.2a	2.6 ± 0.2bc	2.3 ± 0.5b
SBS11-90	4.1 ± 0.5abc	3.3 ± 0.3b	1.9 ± 0.3cd	1.9 ± 0.2bc
SBS11-180	5.1 ± 0.4a	4.6 ± 0.2a	4.5 ± 0.2a	3.3 ± 0.6a
CBS-90	2.9 ± 0.4cd	2.2 ± 0.2c	1.8 ± 0.6cd	2.5 ± 0.5ab
CBS-180	3.3 ± 0.3bcd	3.3 ± 0.1b	2.7 ± 0.2bc	2.5 ± 0.5ab

* AMN – ammonium nitrate; SBS7 – slurry biosolids of pH 7; SBS11 – slurry biosolids of pH 11; CBS7 – limed-stabilized cake biosolids; 90–90 kg N ha^{-1}; 180–180 kg N ha^{-1}.

** Mean values in each column followed by the same letter(s) are not different ($P > 0.05$) according to the Duncan's multiple range test.

forage yield compared with the control between years with (1997–2000) and without (2002) sewage sludge applications increased from 30% to 110% and 70% to 110%, or net increases of 267% and 57% in forage yield change, respectively.

The residual effects of applied sewage sludge on bahiagrass yield expressed as percent forage yield change over the unfertilized bahiagrass are shown in Figure 8. Residual effects of AMN-90 (–6%), AMN-180 (–31%), SBS7-80 (–21%), and SBS7-180 (–17%) declined (negative) with time, but the residual effects of applied SBS11-180 (+13%), CBS-90 (+267%), and CBS-180 (+57%) were positive over time although sewage sludge application ceased after harvest in 2000. The percent forage yield change of bahiagrass fertilized with SBS11-180, CBS-90, and CBS-180 during years when sewage sludges were applied (1998–2000) were 150%, 30%, and 70% compared with percent forage yield change of 170%, 110%, and 110% in 2002 (when sewage sludge applications ceased), respectively.

The residual effects on forage yield of applied CBS-90 (+267%) and CBS-180 (+57%) relative to the control increased with time although biosolids applications ceased after the 2000 harvest season. This was probably due to the higher concentration of organic N in addition to the liming property of CBS. Liming the field could have some direct and indirect effects on forage productivity and on the nutrient status of the soils. Perhaps the single direct benefit of liming is the reduction in acidity and solubility of aluminum and manganese (Peevy et al., 1972). Some of the indirect benefits of liming pasture fields among others would include: enhancing P and microelement availability, nitrification, N fixation, and improving soil physical conditions (Nelson, 1980; Tisdale and Nelson, 1975; Russell, 1973). Dried and composted biosolids have slower rates of N release and in case of CBS with much higher solids concentration (500 000 mg L^{-1}); more N will be released in the second, third, or even the fifth year after the initial application due to higher amount of organic N than ammonium N. The proportions of ammonium and organic N in biosolids vary with the stabilization process.

4.1.2. Lake-dredged materials as nutrient source

The field study was adjacent to the Coleman Landing spoil disposal site in Sumter County, FL. Each plot (961 m^2) was excavated to a depth of about 28 cm, and existing natural soil (NS) and organic materials were completely removed. Excavated NS materials were placed at the south end of the test plots. Existing vegetation from each plot was totally removed prior to backfilling each plot with different ratios of NS and lake-dredge materials (LDM): (100% NS + 0% LDM); (75% NS + 25% LDM); (50% NS + 50% LDM); (25% NS + 75% LDM); and (0% NS + 100% LDM). These ratios of NS to LDM represent the treatment combinations of LDM0; LDM25; LDM50; LDM75; and LDM100, respectively. Natural soils that were excavated were backfilled to each plot along with lake-dredged materials that were hauled from the adjacent settling pond. The total amount of lake-dredged materials and natural soils that was placed back on each test plot was in accordance with the different ratios of lake-dredged materials and natural soils that were described above. After mixing the natural soils and lake-dredged materials, each of the test plots was disked to a uniform depth of 28 cm. Plots were disked in an alternate direction until lake-dredged materials and natural soils were uniformly mixed. Each plot was seeded with bahiagrass at a rate of 6 kg plot^{-1}, followed by dragging a section of chain link fence across each test plot to ensure that bahiagrass seeds were in good contact with the natural soils and lake-dredged materials. Field layout was based on the principle of a completely randomized block design with four replications.

The forage yield of bahiagrass at 112, 238, and 546 Julian days after seeding are shown in Figure 10. Forage yield of bahiagrass varied significantly ($P \leqslant 0.001$) among plots with lake-dredged materials additions. The greatest forage yield of 673 ± 233 kg ha^{-1} at Julian day 112 was from plots amended with 50% lake-dredged materials while bahiagrass in plots amended with 100% lake-dredged materials and 75% lake-dredged materials had the highest forage yield at Julian days 238 and 546 with average forage yield of 3349 ± 174 and 4109 ± 220 kg ha^{-1}, respectively (Fig. 10). The lowest forage

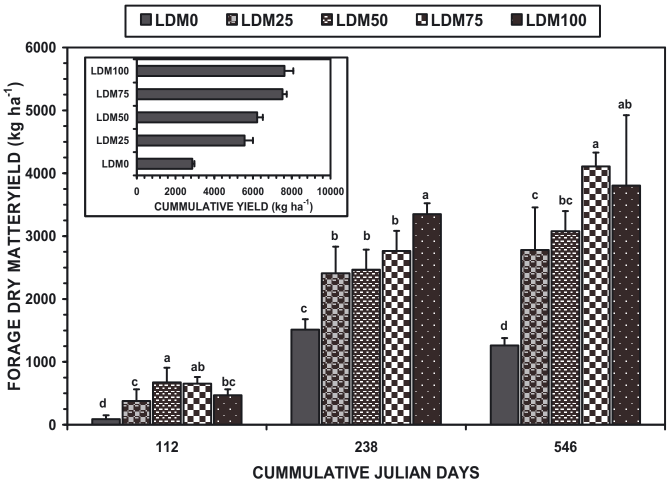

Figure 10. Forage yield of bahiagrass (Julian days 112 to 546) as affected by varying levels of dredged materials application. Forage yield from plots with or without lake-dredged materials are significantly different ($P \leqslant 0.05$) at Julian days 112, 238, and 546 when superscripts located at top of bars are different *(Source: Sigua, 2008).*

yield of 89 ± 63, 1513 ± 166, and 1263 ± 116 kg ha^{-1} were from the control plots for Julian days 112, 238, and 546, respectively (Fig. 10). The average forage yield increase of bahiagrass in plots amended with lake-dredged materials (averaged across treatments) was 512%, 82%, and 173% when compared with bahiagrass in control plots with 0% lake-dredged materials for Julian days 112, 238, and 546, respectively (Fig. 10). These data show the favorable influence that lake-dredged materials had on forage yield of bahiagrass during its early establishment in subtropical beef cattle pastures.

Mean forage yield of bahiagrass during Julian day 112 in plots with 50% lake-dredged materials of 673 ± 233 kg ha^{-1} was not significantly different from that in plots with 75% lake-dredged materials (654 ± 106 kg ha^{-1}), but was greater than that in plots with 25% lake-dredged materials (378 ± 185 kg ha^{-1}) and 0% lake-dredged materials (Fig. 10). For Julian day 238, the greatest forage yield among plots amended with lake-dredged materials was from plots with 100% lake-dredged materials (3349 ± 174 kg ha^{-1}). The lowest forage yield of 1513 ± 166 kg ha^{-1} was from plots with 0% lake-dredged materials. Mean forage yield of bahiagrass in plots with 50% lake-dredged materials of 2467 ± 320 kg ha^{-1} was not significantly different from that in plots with 75% lake-dredged materials (2467 ± 320 kg ha^{-1}) and 25% lake-dredged materials (2409 ± 423 kg ha^{-1}), but was greater than that in plots with 0% lake-dredged materials (Fig. 10).

4.1.3. *Using legumes to enhance nitrogen fertility in bahiagrass pastures*

Beef cattle pastures in the United States and other parts of the world are typically dominated by tropical grasses such as bahiagrass or bermudagrass; drawbacks to these grasses include limited growth during the winter, relatively low nutritional value, and high N fertilizer costs. The development of effective grass-legume pastures for beef cattle production would improve nutritional value of the pastures, minimize N fertilizer input, and better manage nutrient cycling to enhance water quality. Improving the nutritional value of these grass pastures by the addition of legume will improve beef cattle gains in this region and impact both quantity and cost of the red meat supply in the United States and other parts of the world.

The establishment and maintenance of persistent grass-legume pastures is a key option to increase productivity and profitability of beef cattle production systems. Despite the great potential to improve sustainability of animal production, the adoption of technologies has been limited and slow. The integration of legumes (*Arachis glabrata*, Benth) and non-legumes (bahiagrass, *Paspalum notatum*, Flugge) can produce synergistic effects that can minimize external inputs, particularly N fertilizers in agricultural ecosystems. There is a need to carry out research that will help to understand the different factors affecting the fixation and release of biologically fixed N to non-legume crops. The integration of legumes and non-legumes (grass) can produce synergistic effects that can minimize external inputs, particularly N fertilizers in agricultural ecosystems (Saikia and Jain, 2007; Rochester et al., 2001; Omay et al., 1998; Keisling et al., 1994; Oyer and Touchton, 1990).

Symbiotic N_2 fixation may provide N for the growth of legumes and eventually to intercropped plants, such as grasses through the transfer of N from the legumes. An increased understanding of the mechanisms of N transfer in grass-legume mixtures will help increase forage productivity. Nitrogen transfer if properly documented and quantified can lead to improved management systems and increased productivity of grass-legume mixtures (e.g., bahiagrass-peanut mixtures), while minimizing the use of inorganic N fertilizers. This will increase profits and decrease the impact of agriculture on the environment.

4.1.4. *Managing nutrients across paddocks to improve water quality*

4.1.4.1. ***Nutrient balance***

Nutrient balance in the ecosystem involves profitability of the agricultural enterprise and commitments to resource management to maintain quality of air, water and land resources. The role of nutrient management in livestock systems takes on new meaning as producers and the public together consider economic and noneconomic issues (Nelson, 1999). The intensification of livestock production with its associated increased demand for fodder has encouraged farmers to rely more heavily on chemical fertilizers and imported feeds, and very often the waste is considered as a disposal problem rather than useful source of plant nutrients (Hooda et al., 2000). It should be noted that for a farm to be sustainable, its P or N budget should balance, at least after soil reserves are brought up to

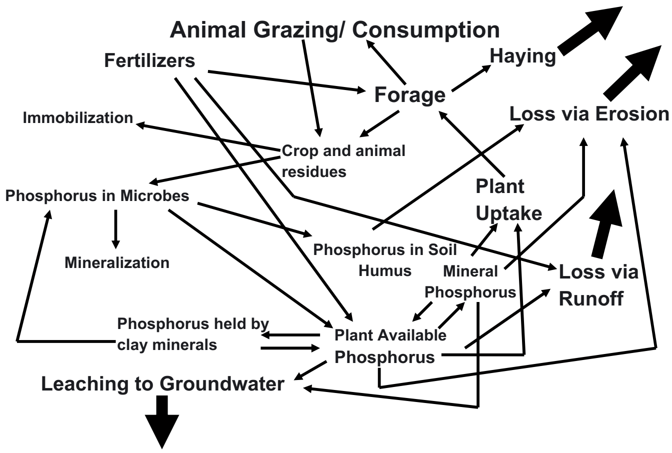

Figure 11. Generalized schematic showing different phosphorus compartments in forage-based pasture system (*modified from Sigua et al., 2006c*). (➡ Loss or export from the system).

desired levels for sustainable production. If there is a net loss of P or N, the farm's soils will eventually become depleted and if there is an excess, the likelihood of pollution is greater (Van Horn et al., 1996). Effective use and cycling of P or N is critical for pasture productivity and environmental stability. Phosphorus or N (not shown) cycling in pastures is complex and interrelated and pasture management practices influence the interactions and transformations occurring within the P cycle (Fig. 11).

Increased loss of nutrients in agricultural runoff has potentially serious ecological and public health implications (Hooda et al., 2000). Nitrogen and P are particularly important as both are implicated in aquatic eutrophication (Levine and Schindler, 1989). Any approach that controls N and/or P losses from agriculture to water must begin with the long-term objective of increasing N and/or P efficiency by attempting to balance N or P inputs with N or P outputs within a watershed. Reducing N or P loss in agricultural runoff may be brought about by Best Management Practices (BMPs) that control the source and transport of N and/or P (Tab. VI).

A review paper published by Shigaki et al. (2006) on animal-based agriculture options for the future contained excellent discussion on the source and transport management of nutrients in the watershed. Source management attempts to minimize the build-up in the soil above levels sufficient for optimum crop growth while transport management refers to efforts to control the movement of nutrients from soils to sensitive locations such as bodies of fresh water. As shown in Table VI, there are several measures available to minimize the potential for nutrient losses in agricultural runoff, which address sources and transport of P and/or N. Important measures to be considered are those that attempt to decrease the surplus of nutrients (P or N) in localized areas. Shigaki et al. (2006) suggested the following measures: (1) dietary P reduction; (2) feed additives that enhance P utilization by animals; (3) alternative uses for manure other than land application; and (4) transporting manure to P- or N-deficient areas.

4.1.4.2. *Water quality best management practices (BMP) for Florida cow-calf operations*

Although cow-calf operations in Florida are generally low-intensity agriculture with relatively low levels of pollutant discharged off-site, certain sites and management practices may contribute to violations of State water quality standards. Under these situations, cattle ranches may contribute elevated levels of P, N, sediments, bacteria and oxygen-demanding organic materials. The potential for discharges from cow-calf operations to cause water quality violations varies greatly, depending on soil type, slope, drainage features, stocking rate, nutrient management and other factors. In general, areas where cattle tend to congregate or have access to water bodies may have the greatest potential to contribute to water pollution (Sigua and Coleman, 2007). Spearheaded by the Florida Cattlemen's Association (FCA) and drawn up in a unique partnership between producers and regulators, the BMP manual serves as both roadmap and vehicle to enhance and protect water quality in Florida.

In 1997, the Florida Cattlemen Association began drafting a common sense, economically-viable guidelines for production practices designed to protect water bodies and maintain compliance with the State of Florida water quality standards. The manual describes the water quality best management practices (BMPs) for beef cow-calf operations in Florida. The different practices are specifically targeted for beef cow-calf operations in Florida and the activities that normally occur in conjunction with beef cattle production. These are not rules or regulations, but voluntary best management practices for Florida ranches that consider good water quality conditions.

The manual is heralded as a unique consensus document that outlines common sense, economically and technically feasible production and management practices that enhance and protect Florida's water resources. It is designed specifically for Florida's cow-calf operations; it does not apply to concentrated animal feeding operations (CAFOs), which generally require a permit (see the 107-page BMP manual at http://jefferson.ifas.ufl.edu/old/agpages/cow_calf_bmp/BMPManual.pdf). Although the manual was designed for statewide use in Florida, the general premise and principles can be adopted elsewhere.

5. SUMMARY AND CONCLUSIONS

Current pasture management including cattle rotation in terms of grazing days and current fertilizer (inorganic + manures + urine) application rates for bahiagrass pastures in subtropical regions of USA offer little potential for negatively impacting the environment. Properly managed livestock operations contribute negligible loads of total P and N to shallow groundwater and surface water. Overall, there was no buildup of soil total P and N in bahiagrass-based pasture. These observations may help to renew the focus on improving fertilizer efficiency in subtropical beef cattle systems, and maintaining a balance of P and/or N removed to P and/or N added to ensure healthy forage growth and minimize P or N runoff.

Table VI. Modified best management practices for the control of diffuse sources of agricultural nutrients (*Source: Shigaki et al., 2006*).

I. Source BMPs – Practices that minimize nutrients loss at the origin
1. Attempts to match animal requirements for N or P with feeds P or N
2. Enzyme added to feeds to increase nutrient utilization by animals
3. Test soils and manures to optimize N and P management
4. Chemically treat manure to reduce P solubility (e.g., alum, fly ash, etc.)
5. Biologically treat manure (e.g., microbial enhancement)
6. Calibrate fertilizer and manure application equipment
7. Apply proper amount of fertilizer
8. Careful timing of fertilizer application to avoid imminent heavy rainfalls
II. Transport BMPs – Practices that minimize transport of nutrients
1. Minimize erosion, runoff and leaching of nutrients
2. Use cover crops to protect soil surface from erosion
3. Install filter strips and other conservation buffers to trap eroded P
4. Manage riparian zones, grassed waterways and wetlands to trap eroded nutrients
5. Stream bank fencing to exclude animal from water course
III. Source and transport BMPs – Systems approach to minimize nutrient losses
1. Retain crop residues and reduce tillage to minimize erosion and runoff
2. Proper grazing management to minimize erosion and runoff
3. Install and maintain manure handling systems
4. Implement nutrient management plan for the farms

Contrary to early perception, forage-based animal production systems with grazing are not likely one of the major sources of non-point source P pollution that are contributing to the degradation of water quality in lakes, reservoirs, rivers, and ground water aquifers, but perennially grass-covered pastures are associated with a number of environmental benefits. Continuous grass cover leads to the accumulation of soil organic matter, sequestering carbon in the soil and thereby reducing the potential CO_2 accumulation in the atmosphere. The increase in soil organic matter is also related to soil quality, with improvements in soil structure, aeration and microbial activity. Effective use and cycling of N or P is critical for pasture productivity and environmental stability. In addition to speeding up N or P recycling from the grass, grazing animals can also increase N or P losses in the system by increasing leaching potential due to concentrating N into small volumes of soil under dung and urine patches, redistributing N or P around the landscape, and removal of N or P in the form of animal products.

The overall goal efforts to reduce N or P losses from animal-based agriculture should be to balance off-farm P inputs in feed and fertilizer with outputs to the environment. Source and transport control strategies can provide the basis to increase N and P efficiency in agricultural systems.

Overall,

- Forage-based animal production systems as suggested by regulators are not the major sources of non-point source nutrients pollution that are contributing to the degradation of water quality in lakes, reservoirs, rivers, and ground water aquifers; and
- Properly managed cow-calf operations in subtropical agroecosystem would not likely be the major contributors to excess loads of N or P in surface water and/or shallow groundwater.

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