Evaluating the Effectiveness of Best Management Practices to Reduce Nutrient Loadings from Beef Cattle Ranches to Lake Okeechobee
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DEMONSTRATION OF WATER QUALITY BEST MANAGEMENT PRACTICES FOR BEEF CATTLE RANCHING IN THE LAKE OKEECHOBEE BASIN

SUMMARY OF PROJECT ACCOMPLISHMENTS

I. Project Background and Goals

Lake Okeechobee is a large, multi-functional lake located at the center of the Kissimmee-Okeechobee-Everglades aquatic ecosystem. The lake provides regional flood protection, water supply for agricultural, urban and natural areas, and is a critical habitat for fish, birds and other wildlife, including the federally endangered Everglades Snail Kite. The 1997 Lake Okeechobee Surface Water Improvement and Management (SWIM) Plan found that excessive phosphorus loading is one of the most serious problems facing the lake. Frequent algal blooms, detrimental changes in biological communities, and impaired use of the water resources are among the documented adverse effects of excessive phosphorus loading. The purpose of this interdisciplinary research project was to evaluate the effectiveness of three Best Management Practices (BMPs) to reduce phosphorus (P) loadings from beef cattle ranches in the Lake Okeechobee basin. These BMPS included the use of soil amendments to retain P, ditch fencing and culvert crossings (DFCC) to keep cattle out of waterways, and wetland water retention (WWR) to increase the retention of water and P on ranches.

Nutrient management assessment were conducted by the Natural Resource Conservation Service and the Florida Department of Agriculture and Consumer Services for a group of potential cooperators. These assessments, as well as other site characteristics (such as suitability for hydrologic monitoring and commitment of the ranch owner) were used to select the final cooperators to participate in this project. As a result of these assessments the soil amendments BMP was evaluated using soils from Beaty Ranch, and the DFCC and WWR BMPs were evaluated at Peleaz Ranch, both in the Okeechobee basin.

II. Soil Amendments BMP Evaluation

1. Laboratory Column Studies

The use of soil amendments is one of several best management practices (BMPs) to reduce edge-of-field P losses, which can impair water quality. Numerous amendment studies have been conducted throughout Florida over the years by several investigators, utilizing a wide variety of amendments. Interpreting the results of these studies is complicated by the wide variety of amendments, amendment rates, soils, P sources, and P loss mechanism(s) investigated. The purpose of this effort was to conduct a systematic evaluation of numerous soil amendments using standardized protocols to provide directly comparable results upon which to judge amendment effectiveness. The protocols included standard total elemental analysis of each amendment, short-term lab equilibrations, small column leaching studies, and simulated rainfall studies. Amendments were applied to a composite soil, representing multiple samples of surface soil from an expected field demonstration site (Beaty ranch) in the Lake Okeechobee watershed. Amendments included water treatment residuals (Fe-, Al-, and Ca-based WTRs), industrial by-products produced or marketed in Florida (slag, silica-rich, and humate materials), and agricultural amendments (lime and gypsum).

Results of the evaluation protocols allowed deselecting of most amendments, and identified the one or two amendments worthy of field investigation. A summary of the pertinent criteria used to select or deselect amendments is given below:

- DuPont Fe-"humate" - deselected because of minimal P sorption capacity.
- Coal slag - despite good adsorption and leaching control properties, the material was deselected because of troublesome trace element contents, especially Mo and As, and because the rates of coal slag required for P control could detrimentally affect plant growth through effects on soil salinity and pH.

- Pro-sil - despite effective P sorption, effective leaching control, and moderate runoff control, the material was deselected because the rates required for P control can raise soil pH excessively which, when combined with a moderately high Mo content, could create an undesirable soil environment for pasture grass growth and grass quality that may threaten livestock health (molybdenosis).
- Gypsum - very effective at controlling P leaching, but ineffective at P sorption and P runoff control. Also deselected because rates necessary for P control may result in soil salinities incompatible with good pasture grass growth.
- Lime and Ca-WTR – behaved essentially the same in all protocols. Likely effective in initially acid soils requiring pH adjustment, but not in soils with pH values □7, where lime solubility is limited. In soils where soil pH is already near neutral (heavily manured Lake Okeechobee soils), little liming agent would be recommended for most pasture grasses. Both liming agents were, thus, deselected for field evaluation.
- Vigiron (Fe-WTR) - moderately effective at sorbing P and reducing leaching, but only fair in controlling P runoff. Deselected because it contains moderately high concentrations of Mo and As, and may release immobilized P under reducing conditions.
- dinoSoil - high rate (1%) only slightly effective at sorbing P and reducing leaching, but a top performer in runoff simulations. High cost (~$145/T), however, likely makes the amendment impractical for large scale use.
- Manatee and Okeechobee Al-WTRs - effective P sorbers, but ineffective at controlling P leaching when soluble P is below the zone of amendment incorporation. Very effective at controlling P leaching when soluble P is made to contact WTRs (amendment incorporation, or soluble P added after amendment addition). The Al-WTRs dominated the best performers in runoff simulations. The Okeechobee material was uniformly better when applied at 1%, whereas the Manatee material requires rates at 2.5%. The Okeechobee material is locally available, but its low solids content (~9%) creates handling and transportation problems. Additionally, annual WTR production is estimated at only 250-300 tons, which severely limits the acreage that can be amended. Relatively high rates (~25 T/A) of the Manatee material are needed, but the material's dry condition makes handling/application easy, and annual production is much greater (~4000 Mg) than at Okeechobee. Either (or both) materials are suitable for field-testing: Okeechobee material at 0.1 and 1%, or Okeechobee at 1% and Manatee material at 2.5%, but the Manatee material is recommended. Method of WTR application – surface applied or soil incorporated - should be tested.

Thus, the two treatments recommended for further testing using rainfall simulation studies were Manatee Al-WTR at 2.5%, surface applied and incorporated to 5 cm. Complete details regarding the laboratory column study of soil amendments are included in Appendix A.

2. Rainfall Simulation Studies

The objective of this effort was to evaluate the effect of Al-WTR on P loss from a manure impacted soil using a rainfall simulation protocol. Soil was removed from the field site as 0–10 and 10–20 cm depths. Both depths contained high concentrations of water-soluble P and Mehlich-1 P; approximately 18 and 950 mg P kg-1, respectively. After air drying and sieving, the soil was placed in rainfall simulation boxes (100 cm x 30 cm x 20 cm) designed to collect runoff, subsurface flow, and leachate. An Al-WTR was either surface applied or incorporated to 10 or 20 cm depths at a rate of 2.5% of soil dry weight. The soil was then sprigged with stargrass (*Cynodon nlemfuensis*). Rainfall simulations were run six times at 3 wk intervals. Runoff was collected for 30 min after initial runoff began. Subsurface flow and leachate were collected (depths of 10 and 20 cm, respectively) after runoff ceased.

When Al-WTR was surface-applied, the SP concentration in runoff was reduced by approximately 75% compared to untreated soil; however, SP concentrations in subsurfaceflow and leachate did not decrease. When Al-WTR was incorporated into the soil at depths of 0–10 or 0–20 cm, runoff SP concentrations were reduced by approximately 45%. Incorporation of Al-WTR to a depth of 10 cm decreased SP concentrations in subsurface flow and leachate by 37 and 11%, respectively. However, with incorporation of Al-WTR to a depth of 20 cm, both subsurface flow and leachate SP concentrations were reduced by approximately 90%. The incorporated Al-WTR reduced soil waterextractable P (WEP) by approximately 70%. However, Mehlich-1 P concentrations were not affected by the incorporation of Al-WTR in the soil. Care must be taken to ensure complete incorporation of Al-WTR throughout the P-impacted layer, as Al-WTR is only effective in reducing SP concentrations when it is in contact with the impacted soil. Shoot and root growth of stargrass were not adversely affected by the Al-WTR applied at a rate of 2.5% of soil weight. Complete details of the rainfall simulation studies can be found in Appendix B.

III. Ditch Fencing and Culvert Crossing BMP Evaluation

The ditch fencing and culvert cattle crossing (DFCC) BMP was implemented at a commercial cow-calf ranch (Pelaez & Sons Ranch) by fencing a 170 m section of the ranch's principal drainage ditch and installing a culvert crossing to provide a way for the cattle to cross over the ditch (Figures 1, 2 and 3, Appendix C) without entering it. The concentrations and loadings of Dissolved Organic Nitrogen (DON), Total Nitrogen (TN), and Total Phosphorus (TP) were compared for one pre-BMP period (wet period of 2005: June-Oct), and three post-BMP periods (wet periods of 2006-08: June-Oct) between the upstream and downstream of the BMP site. The downstream minus upstream loadings indicates whether the 170 m ditch section was a sink or a source of N and P. Positive values indicate a net contribution of N and P from the ditch while negative values indicate that the section is removing N and P. The positive values indicate addition of N and P from cattle crossing activity or stream soil and vegetation. Negative values on the other hand indicate that cattle crossing activity did not add N and P and/or that the N and P are removed/retained by the physical, chemical and biological processes.

During the pre-BMP period (2005), downstream P loading was 123 kg higher than the upstream P loading indicating addition of P from the fencing site. During the post-BMP periods of 2006 and 2008, downstream P loadings were 17 and 88 kg lower than the upstream P loadings indicating reduction/retention of P in the ditch section in contrast to the observed addition of P during the pre-BMP period. Downstream P loading was 35% higher than the upstream during the pre-BMP period while downstream P loadings were 32 and 11% lower during the post-BMP periods of 2006 and 2008, respectively. Reduction in N loading was also measured at the BMP site during the 2006 and 2008 wet seasons. The post-BMP period of 2007 was an exception. Unusually dry conditions during 2007 resulted in the addition of N and P at the BMP site which was likely due to the mineralization of resident P and N from soil and aquatic vegetation in the 170 m ditch section. Despite the addition of N and P during the post-BMP period of 2007, computed averages over the three post-BMP periods indicate a net reduction in N and P as a result of BMP implementation. Average upstream and downstream TP loadings were 295 and 264 kg for the three post-BMP periods indicating 10% reduction of TP at the BMP site. Likewise, average upstream and downstream TN loadings from the BMP site were 675 and 601 kg for the three post-BMP periods indicating 11% reduction of TN at the BMP site. Collection of additional data for another pre-BMP period may reduce the uncertainty related to the use of only one year of pre-BMP data and inaccuracies associated with flow measurements.

A refereed journal article by Shukla, Goswami, Graham, Hodges, and Knowles is under preparation. A draft copy of the publication is provided in Appendix C. It includes a detailed analysis of the effectiveness of the BMP with regards to water quality and economics.. The BMP effectiveness discussion includes a detailed statistical analysis.

IV. Wetland Water Retention (WWR) BMP Evaluation

Wetland water retention (WWR) BMP was implemented at two wetlands (wetlands 1 and 4) at Pelaez & Sons ranch (Figure 1, Appendix D). The BMP implementation involved installing a riser board structure at the outlet of the wetland and adding boards until the desired water retention level was met (Figure 2, Appendix D). Flow-weighted concentrations of TN and TP at the wetland outlets were compared for the pre- and post-BMP periods to evaluate the effectiveness of the BMP.

For wetland 1, two years of pre-BMP (June 2005-May 2006, pre-BMP1 and June 2006-May 2007, pre-BMP2) and two years of post-BMP (June 2007-May 2008, post-BMP1 and June 2008-May 2009, post-BMP2) data were used. To evaluate the WWR at wetland 4, June 2005-May 2006 was used as the pre-

BMP period while June 2006-May 2007, June 2007-May 2008, and June 2008-May 2009 were the three post-BMP periods (post-BMP1, post-BMP2, and post-BMP3, respectively). (Table 1 in appendix D)

At wetland 1, the TN and TP loads for post-BMP1 were less than those during the two pre-BMP periods. However, the reductions during post-BMP1 period could not be attributed entirely to the BMP since the flow volume during the post-BMP1 period (June 2007-May 2008) was low due to dry weather conditions. For post-BMP2 (June 2008-May 2009), TN and TP loads were almost twice than those during pre-BMP1. Unusually high P loadings were likely due to combination of high runoff volume and availability of P within the drainage area. Drought conditions in 2007 resulted in mineralization of soil and plant P which was available to move with overland flow in 2008. This overland flow was substantial due to the building of the water table by several rainfall events in July and the occurrence of tropical storm Fay (August 2008) during the middle of the wet season which resulted in large flows from the wetland that moved available P from the surface generating high P loadings. The average TN (304 kg), and TP (93 kg) loads for the two post-BMP periods were higher than the average TN (161 kg), and TP (47 kg) loads for the two pre-BMP periods. Given the data for the pre-BMP and post-BMP periods, the BMP did not seem to be effective in reducing the N and P loads at wetland 1. Large rainfall variability masked the BMP results. Longer term data will be needed to better evaluate this BMP at wetland 1.

For wetland 4, individual TN and TP loads during the three post-BMP periods were less than those during the pre-BMP period. Due to relatively low rainfall during post-BMP1 and post-BMP2, both flow volume and nutrient loads were less than those observed during the pre-BMP period. Due to this marked difference in rainfall, it was difficult to evaluate the BMP based on post-BMP1 and post BMP2 periods. In contrast, the pre-BMP and post-BMP3 periods had similar rainfall and flow volumes therefore providing a better comparison for BMP evaluation than post-BMP1 and post-BMP2. The N and P loads were lower during post-BMP3 compared to pre-BMP indicating that water retention at this wetland reduced both the flow volume and the TN and TP loading. Taking into account all three post-BMP periods, the average TN (97 kg), and TP (57 kg) loads were lower than the TN (319 kg) and TP (182 kg) loads during the pre-BMP period.

Average TN and TP loadings (combined for both wetlands) were calculated based on average loadings from all pre- and post-BMP periods for the two sites. This showed net reductions of N and P during the post-BMP period. There were three pre-BMP periods and five post-BMP periods in total for the two wetlands. On average there were 16% reduction in TN loadings (Pre-BMP: 214 kg and post-BMP: 180 kg), and 23% reduction in TP loadings (pre-BMP: 92 kg and post-BMP: 71 kg) during the post-BMP periods. The hydrologic and water quality monitoring at the two sites is continuing to better quantify the WWR BMP. A detailed analysis of the WWR BMP results has been presented in Appendix D.

V. **Hydrologic Model Evaluation**

Two hydrologic models, Watershed Assessment Model (WAM) and Agricultural Catchment Research Unit (ACRU) have been used for evaluating the WWR BMP in the ranch. Both of these models are being tested using observed hydrologic and water quality data. The ability of these models to simulate the effects of the WWR BMP is being evaluated. Preliminary results of the two models have been presented in Appendix F. Detailed results will be presented in the final report.

VI. **Economic Analyses**

Economic analyses of the BMP effectives are presented in appendices C and D as subsections.

VII. **BMP Education**

A number of presentations were made at local, state, and national scale to disseminate the project results. A brief account of these presentations is presented below.

- Presentations at the Greater Everglades Ecosystem Restoration (GEER) (2003) and USDA-CSREES National Water Quality Conference (2004).
- Two presentations at South Florida Beef Cattle Workshop (2003 and 2005).

- Three presentations at growers' sites on the benefits of the water quality BMPs I collaboration with World Wildlife Fund (WWF), and state agencies. .
- A presentation at the 2004 Gator Day in Tallahassee on the cow-calf BMP research and extension programs at UF to educate state legislators.
- A presentation at a Watershed Water Quality in-service training for county extension faculty from a variety of discipline (2004).
- Presentation at inter-agency meeting in Okeechobee (2008). The meeting was attended by FDEP, SFWMD, FDACS, USDA-NRCS, environmental organizations, ranchers, and other stakeholders.
- Presentation at inter-agency meeting in Okeechobee (2009). The meeting was attended by FDEP, SFWMD, FDACS, USDA-NRCS, environmental organizations, ranchers, and other stakeholders;
- Meeting at SWFREC-IFAS, Immokalee attended by South Florida Water Management District (SFWMD) (2009).
- Presentation of the BMPs in the UF-IFAS Lake Okeechobee Watershed Group meeting at Water Institute, University of Florida, Gainesville (2009).
- Evaluation of ranchland best management practices for phosphorus discharge in the Lake Okeechobee. University of Florida Water Institute, Gainesville, FL (2008).
- Effects of wetland hydrologic restoration on nitrogen and phosphorus discharges in the Lake Okeechobee basin. American Society of Agricultural and Biological Engineers (ASABE) Annual International Meeting, Providence, RI (2008).
- Evaluation of cattle-crossing BMP for phosphorus discharge at a beef-cattle ranch in the Lake Okeechobee watershed. ASABE Annual International Meeting, Providence, RI (2008).
- Effects of water retention on nitrogen and phosphorus loadings from two drained wetlands in the Lake Okeechobee Basin. ASABE Annual International Meeting, Reno, NV (2009).

APPENDIX A

Introduction

The use of soil amendments is one of several best management practices (BMPs) to reduce edge-of-field P losses, which can impair water quality. Soil amendments are intended to reduce P concentrations in soil solutions and, thereby, reduce P available for various loss mechanisms, including runoff and leaching. Numerous amendment studies have been conducted throughout Florida over the years by several investigators (e.g., Allen, 1988; Anderson, 1995; Alcordo, et al., 2001; Matichenkov et al., 2001), utilizing a wide variety of amendments. Interpreting the results of these studies is complicated by the wide variety of amendments, amendment rates, soils, P sources, and P loss mechanism(s) investigated. The purpose of our work was to conduct a systematic evaluation of numerous soil amendments using standardized protocols to provide directly comparable results upon which to judge amendment effectiveness. The protocols included standard total elemental analysis of each amendment, short-term lab equilibrations, small column leaching studies, and simulated rainfall studies. Amendments were applied to a composite soil, representing multiple samples of surface soil from the expected field demonstration site on the Beaty Ranch. Amendments included water treatment residuals (Fe-, Al-, and Ca-based WTRs), industrial by-products produced or marketed in Florida (slag, silica-rich, and humate materials), and agricultural amendments (lime and gypsum).

Amendment Selection Criteria

Best management practices to reduce P impacts on water quality can be categorized into methods that: 1) Reduce P inputs, 2) Increase P retention by soil, 3) Reduce P solubility in soil, and 4) Remove P from water escaping the watershed soils before the P-laddened water reaches a significant water body. This project focused on reducing soluble P concentrations in the soil solutions of watershed soils using methods 2 and 3, but a permanent solution to P management will likely have to involve a multi-faceted approach (all methods). To be effective, such methods must permanently immobilize P in the soils, not merely delay P mobility. Thus, practices (e.g., use of soil amendments) that alter soil properties to increase P retention or reduce P solubility must also decrease P release (desorption). Further, such alterations must be expected to be permanent, less the P be re-solubilized (mobilized) in the future under normal environmental conditions.

Best management practices must also be matched to the chemical, physical, and hydrologic characteristics of the problematic watershed. Thus, if the soils of the watershed are not already impacted by P (high in soluble P) but are expected to receive large inputs of P (e.g., via surface applications of manure), amendments or other practices that seek to reduce P inputs and/or the solubility of the P inputs can be effective. If the soil is already impacted and high concentrations of soluble P exist throughout the soil profile, practices that only increase P retention in the zone of incorporation may be ineffective at reducing P losses from other parts of the soil profile. Fine-textured soils may promote P retention and slow P leaching, but encourage surface runoff of soluble or particulate P. Flat, coarse-textured soils may minimize surface runoff, but may allow significant P leaching to ground waters. The presence of slowly permeable layers (e.g., spodic horizons), particularly if shallow in the profile, can alter the hydraulic characteristics of even coarse-textured soils and promote “sub-surface runoff”, or surface runoff in high water table periods (e.g., rainy season). The choice of amendment to use is influenced by all these factors and, thus, is complex. This complexity confounds extrapolation of laboratory studies designed to accentuate one P retention (or P loss) mechanism to the multitude of field conditions possible. Practical considerations such as amendment availability, rate necessary, and cost must also be included. Further, other contaminants (e.g., trace elements, excessive salinity or acidity/alkalinity) in amendments must be considered so solving the “P problem” does not lead to other water quality or soil/plant problems.

Previous Approaches (examples)

1. Reduce the solubility of P added in land-applied wastes.

a. Surface waste/amendment applications

Moore and colleagues (e.g., Moore et al., 2000; Moore and Miller, 1994; Shreve et al., 1995) have done extensive work that documents effective control of P solubility by Al added to poultry manure. Sufficient addition of Al sulfate (alum), to yield a 1:1 molar ratio of Al to P in the manure, significantly reduced P solubility and dramatically reduced P in runoff from manured sites.

b. Incorporated waste/amendment applications

Elliott et al. (2001) co-applied Al treated water treatment residuals (Al-WTR) with several biosolids, fertilizer, and two manures. They demonstrated almost complete control of P leaching through Florida sands initially low in P, regardless of P source because soluble P levels were dramatically reduced in the soil/amendment mixtures.

2. Increase soil retention of P

a. Increased P adsorption

Laboratory studies (O'Connor et al., 2002) showed that Al-WTRs adsorb large amounts of P, and that poorly P-sorbing Florida soils could be made to adsorb significantly more P when they were amended with modest amounts of Al-WTRs. The P retained by Al-WTR or Al-WTR-amended soils was essentially irreversibly bound, barring unrealistic changes in environmental conditions (very low pH). Iron-based WTRs, or salts, can also effectively sorb P, but P release can occur under reducing conditions (Ann et al., 2000a,b).

Surface applications/incorporation of Al-WTRs were much less effective in controlling P leaching in another Florida sand that was already impacted by long-term manure additions (Lane, 2002). Soluble P concentrations were high below the zone of amendment incorporation, contributing to P loss, and high soluble organic carbon concentrations appeared to reduce P binding to the WTR.

b. Increase soil solid phase control

Soil retention of P can also be increased by causing secondary solid phases (precipitates) of P to form. The process chemistry is similar to that used to remove P from municipal wastewaters (Jenkins and Hermanowicz, 1991). Whereas P solubility in acid systems is typically amenable to control with Fe and Al (to cause insoluble Fe- and/or Al-P compounds to form), high pH soils typically limit P solubility via precipitation of various Ca-P compounds (Allen, 1988; Anderson, 1995). Soils of south Florida impacted by heavy manure applications often exhibit high pH values (≥7), despite low natural soil pH values of 4.5 to 5.5. Thus, heavily manure-impacted soils can be more amenable to P control through the use of Ca-containing amendments (e.g., lime, gypsum) than to control with Fe- or Al- containing amendments. However, if the high soil pH values revert to the natural low soil pH values, the solubility of Ca-P compounds is predicted to increase and P would be released.

Choice of amendment and amendment rates

For this study, the choice of amendments and amendment rates to evaluate was influenced by both chemical and historical considerations. Initially, practical considerations of ready availability, application practicality, and cost were ignored. Refinements to the list of amendments for further evaluation, however, involved practical considerations.

1. Chemical considerations

As described above, P solubility is generally limited by Fe and Al in acid soils and by Ca in high pH soils. Thus, sources of Fe and Al as well as sources of Ca (or high pH) were evaluated. The rates of each amendment were roughly based on the presumed chemistry of the resulting (insoluble) metal salts. Thus, P precipitated as insoluble Fe- and/or Al-phosphates (e.g., strengite = $FePO_4$; variscite = Al PO_4) usually have molar ratios of Fe (or Al) to P of 1:1. The P concentration (moles) to be immobilized can be chosen as soluble P or, more commonly, total P in the soil. The concentration (moles) of metal (Fe or Al) needed to react with the P should be based on the “reactive” metal concentration in the amendments. Total metal concentrations in amendments other than soluble metal salts typically overestimate the concentration of metal truly available for reaction with P. O’Connor and Elliott (2001) showed that amorphous (oxalate extractable) metals in Fe and Al amendments better described amendment reactivity toward P than total metal content.

The ratio of metal (Ca) to P in the host of Ca-phosphates that can precipitate in soils varies widely. A common assumption, however, is that P will eventually be found in various apatite mineral forms, in which the Ca:P molar ratio is 5:3 [e.g., hydroxyapatite = $Ca_5(PO_4)_3(OH)$]. Total Ca and P concentrations (moles) are typically used to calculate the amount of amendment (Ca source) necessary to react with P in the soil. Sometimes, the soil is assumed sufficient to supply the necessary Ca, but soil pH must be raised to values where the solubility of Ca-P is low (pH values >7). In this case, the liming value of the amendment and the initial soil pH is considered, rather than the Ca concentration of the amendment.

2. Historical considerations

In this context, historical considerations refer to previous studies (published and unpublished) in which various rates of various amendments have been utilized. Particular attention was paid to literature (or results) generated by Florida investigators.

An excellent discussion of the use of chemical soil amendments to control P in Florida, specifically the Lake Okeechobee watershed, is given by Allen (1988). He calculated amounts of Ca-based amendments needed to control P losses from dairies assuming ideal chemistries (target Ca:P molar ratio of 5:3), but noted the possible interference of such soluble constituents as Mg and organic C on precipitation kinetics and purity of the solid phases formed. Greater quantities of Ca are necessary in such situations. Allen (1988) called for detailed soil chemistry studies to verify the calculations and to quantify interferences and actual reaction rates and reaction products.

Anderson et al. (1995) conducted lab incubations (equilibrations) with manure-loaded soils (Spodosols) amended with Ca-, Fe- and Al-salts, alone or in combination, under both aerobic and anaerobic conditions. Amendment rates were targeted to include metal:P molar ratios indicative of pure metal-P solids, but included rates on either side of the “ideal” rate necessary to attain the target ratios. Gypsum ($CaSO_4$) rates were 0, 4, 8, and 16 g kg^{-1}soil; $FeSO_4$ and $Al_2(SO_4)_3$ rates were 0, 50, 100, 250, and 1000 mg Fe (or Al) kg^{-1} soil; and $CaCO_3$ rates (not given) were chosen as the amounts necessary to raise the pH values of individual soils to 7-7.5. Anderson et al. (1995) concluded that lime could be an effective amendment if sufficient material was added to raise (and maintain) soil pH values in the 7-7.5 range. Iron and Al salts effectively increased soil retention of P, but the authors expressed concern about possible Al toxicities and cost of both metal salts. Anderson et al. (1995) favored gypsum as a soil amendment, especially in anaerobic systems and in Spodosols heavily impacted by dairy manure. Gypsum rates as great as 100 mg kg^{-1} soil were effective, although there were unexplained impacts on soil microbial activity at the highest rate.

Alcordo et al. (2001) conducted greenhouse leaching studies with Ap, E, and Bh horizons of beef cattle pasture soil (Immokalee series) amended with additional P fertilizer and a single Ca rate (800 kg Ca ha^{-1}) added as gypsum or lime products. No justification was given for the Ca rate and final soil pH values were not in the 7-7.5 range identified as useful by Anderson et al. (1995). A Ca loading of ~2.4 g Ca kg^{-1} soil can be calculated for the Alcordo et al. (2001) treatment, which is similar to the 8 g gypsum kg^{-1} soil (~2 g Ca kg^{-1} soil) rate of Anderson et al. (1995). Alcordo et al. (2001) found no benefit of gypsum products in reducing P leaching, but significant benefits of lime.

Rechcigl et al. (2000) evaluated limestone and gypsum as amendments in a field study to reduce P leaching and runoff from beef cattle pastures. Amendments were applied at 2 and 4 Mg ha^{-1}, roughly equivalent to 500 and 1000 kg Ca (from gypsum) ha^{-1} and 800 and 1600 kg Ca (from lime) ha^{-1}. The field rates were, thus, similar to the greenhouse rates used by Alcordo et al. (2001). Rechcigl et al. (2000) reported reduced soil water P concentrations in both the gypsum and lime treatments, but no effect on P concentrations in runoff. Boruvka and Rechcigl (2003) reported that lime was more effective than dolomite or gypsum at increasing P retention by the Ap horizon of a Spodosol (initial pH 4.3), but noted that the effect must be accompanied by an increase in soil pH (≥ 7) to increase P retention.

Stout et al. (2000) applied gypsum (10 and 20 g kg^{-1} soil) to 3 manure-impacted soils from Pennsylvania, and evaluated the impact on surface runoff of P. Gypsum significantly reduced runoff P in grass-covered runoff boxes (where dissolved P predominated), but failed to reduce dissolved P in runoff from bare soil (where particulate P predominates). Leaching losses of P were not evaluated.

Several industrial by-products (e.g., coal combustion by-products, steel processing sludge, bauxite mining residuals, and fertilizer production slag) have been examined as amendments to control P solubility in soil (Peters and Basta, 1996; Stout et al., 1998; Stout et al., 2000; Matichenkov et al., 2001; and Callahan et al., 2002). Amendment rates typically ranged from 5 to 80 g amendment kg^{-1} soil, simulating field application rates of about 3 to 50 Mg ha^{-1}. Effectiveness and practicality varied with application rate, soil condition (pH), and P control mechanism investigated. Only the Matichenkov et al. (2001) study was conducted using Florida soils and will be detailed here.

Matichenkov et al. (2001) evaluated Si-rich materials both for their abilities to increase soil retention of P and to improve Bahiagrass growth. Only the retention impacts are addressed here. Amendments included a slag by-product of the electric production of P fertilizer and Pro-Sil, a by-product of steel processing. Both amendments were applied to Florida soils at the equivalent of 10 Mg ha^{-1} (estimated to = 5 g amendment kg^{-1} soil). Effects of the slag material on (increased) P retention and (decreased) P leaching were marginal, whereas the effects of Pro-sil were dramatic.

Elliott et al. (2002) conducted lab equilibration and column leaching studies with various water treatment residuals (WTRs) produced in Florida. Applications of WTRs, notably the Fe- and Al-WTRs, to Florida soils that sorbed P poorly increased P retention and decreased P leaching. Amendment rates examined varied from 0.1 to 10% by weight, but practically effective rates were typically 1 to 5% by weight (1 to 50 g WTR kg^{-1} soil, and ~20 to 100 Mg amendment ha^{-1}). Brown and Sartain (2000) conducted greenhouse studies with a Fe-WTR (Vigiron) applied at 2.5% by weight to golf green mix (85% sand), and studied impacts on Bermuda grass growth and fertilizer-P leaching. Leaching of P was minimal in the presence of Fe-WTR, with no impacts on grass growth or nutrition.

dinoSoil™ (Leonardite, an oxidized form of lignite) has been championed as an amendment to improve soil quality and plant growth and to retard soil P loss (M. Hougland, personal communication, 2003). Rates as low as 1000 lbs per acre of dinoSoil (0.05% by weight) are recommended.

3. Final Selection

Based on the chemical and historical considerations described above, 10 materials were selected for evaluation as amendments at various rates of application (Table 1). The list included two Fe-"humates" (a Fe-WTR, "Vigiron") and a Ti-mine waste (Fe-"humate"), two Al-WTRs, one Ca-WTR, a coal combustion slag, a Si-rich material (Pro-sil), a Leonardite material (dinoSoil), and two agricultural materials (lime and gypsum). All materials are produced or marketed in Florida, and most have been evaluated to some degree by Florida researchers as amendments for P-impacted soils or waters. Amendment rates were initially chosen to represent wide ranges that encompassed rates reportedly effective at controlling P solubility/mobility. The list of amendments and range of amendment rates was expected to narrow as amendment effectiveness was tested in the various standardized protocols. Ultimately, only 1 or 2 amendment/amendment rate combinations will be recommended for field scale evaluation.

Table 1. Amendments and amendment rates selected for evaluation.

		Rates		
Amendment	**Source**	**g** material **kg**$^{-1}$ soil	**%** by wt.	**T/A***
Fe-WTR	Vigiron, Tampa, FL	0, 1, 5, 10,50,100	0 to 10	0 to 100
Fe-“humate”	Dupont, Starke, FL	“	“	“
Al-WTR	Manatee County, FL	“	“	“
Al-WTR	Okeechobee, FL	“	“	“
Pro-sil	Pro-Chem (PA)	0, 1, 2.5, 5, 10, 25	0 to 2.5	0 to 25
Coal slag	Nutrasource, Tampa, FL	0, 1, 5, 10, 50,100	0 to 10	0 to 100
Gypsum	Nutrasource, Orlando, FL	“	“	“
Ca-WTR	Bradenton, FL	0, 0.5, 1, 2, 5, 10	0 to 1	0 to 10
Lime	Franklin Minerals, Ocala, FL	“	“	“
dinoSoil	Leonardite (Texas)	0, 1, 5, 10, 50,100	0 to 10	0 to 100

*Approximated assuming uniform mixing with soil to a depth of 15 cm and a bulk density of 1.3 g cm^{-3}, which yields 10^3 tons of soil per “acre-furrow-slice”.

Standardized Protocols

Amendment and soil analyses

Amendments were air-dried and ground to pass a 2-mm sieve, and then digested using EPA Method 3050B (USEPA, 1995). Digests were analyzed for P, Fe, Al, Ca, Mg, Mn, S, Cu, As, Se, and Mo using inductively coupled plasma spectrometry (ICP) or graphite furnace atomic absorption spectrometry (GFAA). Percent solids, pH, and % organic matter were determined on materials “as is” (at their native moisture contents) using standard methods (Hanlon et al., 1997; Sparks, 1996). Carbon and N contents of the materials were determined by combustion at 1010 degrees Celsius using a Carlo Erba NA-1500 CNS analyzer. Detailed P chemistry of the materials was determined using sequential analysis (Chang et al., 1983), Mehlich-1 extraction (Hanlon et al., 1997), and oxalate extraction for P, Fe, and Al (McKeague et al., 1971). All analyses were conducted in triplicate, and in accordance with typical QA/QC procedures, which included use of certified standards to verify methods.

Similar analyses were conducted on soil from the expected field site (Beaty Ranch). Soil samples were randomly collected from 10 locations within the field site from the 0-15 cm depth and thoroughly mixed to yield a “composite” soil for use in the protocols involving soil. In addition to the analyses identified above for the amendments, the soil was analyzed for water extractable P using a 1:10 soil:water suspension equilibrated for 24 h, followed by filtration (0.45 micron), and P analysis by a colorimetric method (Murphy and Riley, 1962).

Selected properties of the amendments are given in Table 2. Total elemental analyses are, in general, consistent with values expected for the various materials. Thus, the two Al-WTRs contain elevated total Al contents, the Fe-WTR is high in total Fe, and the Ca-sources (Ca-WTR, gypsum, and lime) contain abundant Ca. The Fe-“humate” from Dupont contains ~10 times more Al than Fe, and the total Fe concentration of the Dupont material is only about ¼ that of the Fe-WTR. The coal slag contains abundant Fe and Al, suggesting that it may serve to immobilize P primarily through reactions with Fe and Al. The slag also has a low pH (3.69), which could affect the pH of poorly buffered soils when applied at high rates. Pro-sil is championed as a Si-rich material, but it also contains appreciable total Fe and Al (~5% by weight), and Ca (~24%), which are expected to influence P solubility. The Pro-sil material we received had an extraordinarily high pH (>11), whereas data provided by the producer suggest the pH is 7.6, but that the pH can vary within piles (B. Ande, 2003, personal communication). A material pH of >11 could be problematic if the liming value of the material was not considered in land application practices, particularly if the material was applied as surface applications to acid-tolerant (“acid-loving”) plants like

Bahiagrass. Bahiagrass growth can be severely impacted when soil pH values exceed ~7. The dinoSoil material reportedly contains abundant “humates” in combination with aluminosilicate clay (montmorillonite). The total analysis of Table 2 confirm the high total Fe and Al concentrations expected for aluminosilicate clay, but an organic matter content (as measured by loss on ignition) of only ~8%. The low pH of the dinoSoil material could alter the pH of poorly buffered soil if the material was applied at high rates. All amendments contain some P, but the total concentrations are usually low and none exceeded 3.12 g kg^{-1} (~0.3%).

Form	Total Elemental (g kg^{-1}, unless otherwise noted)												% Solids	% Org Matter (LOI)	pH ‡
	C (%)	N (%)	C:N	Fe	Al	Ca	Mg	P	Mn	S	Cu	Zn			
†M-Al-WTR	12.7	0.60	21.1	2.97	78.1	1.09	0.24	2.79	0.04	7.26	0.06	0.02	80.6	26.4	5.04
†O- Al-WTR	19.0	1.17	16.2	5.33	145	5.91	2.43	1.91	0.05	10.5	<0.01	<0.01	9.00	39.0	6.82
Fe-WTR	12.4	0.81	15.2	232	5.04	22.0	0.63	3.12	0.60	4.48	0.48	0.03	77.9	24.8	6.07
Ca-WTR	11.6	0.07	161	0.37	0.60	321	8.61	0.03	0.01	1.08	<0.01	<0.01	99.6	1.92	8.88
Coal slag	26.5	0.31	86.6	88.6	49.7	11.4	2.25	0.27	0.14	40.6	0.08	0.44	93.5	37.4	3.69
Pro-sil	0.85	0.03	29.3	33.5	15.2	240	56.8	0.12	8.22	0.80	0.04	0.04	99.4	1.76	11.3
Dupont	32.2	0.59	54.1	4.85	49.0	0.20	0.49	1.02	0.02	4.67	0.02	0.01	66.3	52.6	3.59
Gypsum	0.80	0.06	13.1	0.50	0.86	267	1.38	0.19	0.01	195	<0.01	<0.01	77.8	1.73	8.30
Lime	11.9	0.06	199	0.50	0.35	347	2.58	0.45	0.03	0.05	<0.01	<0.01	92.8	0.58	8.92
dinoSoil	2.68	0.14	19.6	41.9	66.6	10.2	7.93	0.17	0.35	14.0	0.03	0.09	91.3	7.83	3.63

Table 2. Selected properties of amendments.
†M-Al- WTR = Manatee County Al- WTR; O-Al-WTR = Okeechobee Al- WTR
‡ At solid: solution ratio of 1: 2

Additional phosphorus characterization data for the amendments are given in Table 3. The sequential analysis data suggest that little of the total P in any amendment is readily soluble (KCl-extractable values < 3 mg kg^{-1}) and Mehlich-1-extractable values < 10 mg kg^{-1}. Much of the sequentially extracted P was found in the HCl fraction (Ca and Mg-associated “forms”) and the residue fraction (non-labile “forms”).

This distribution is expected for the Ca-dominated amendments (gypsum, lime, and Ca-WTR). The lack of dominance of NaOH-Pi (Fe and Al-associated P “forms”) in the Fe and Al-WTRs may appear incongruent, but likely reflects the addition of liming agents to promote Fe- or Al-hydroxyoxide formation in drinking water treatment operations and the recalcitrance of P residues to extraction by the less stringent reagents that characterize the various “forms” of solid-P.

Table 3. Phosphorus characterization of amendments.

Form	Sequentially extracted P (mg kg^{-1})						Mehlich-I P (mgkg^{-1})	Oxalate Extractable (g kg^{-1}) ‡			PSI $ (%)
	KCl	NaOH Pi	NaOH Po	HCl	Residue	Sum		P	Fe	Al	
†M- Al- WTR	2.07 ± 0.11	397 ± 10	105 ± 1	2200 ± 110	301 ± 69	3000 ± 189	8.1 ± 0.1	3.02 ± 0.02	3.32 ± 0.05	109 ± 3.8	2.39
†O- Al- WTR	2.86 ± 0.32	66.9 ± 8.8	80.7 ± 4.3	195 ± 12	1680 ± 5	2030 ± 6	0.2 ± 0.0	0.61 ± 0.01	0.78 ± 0.00	73.7 ± 1.9	0.72
Fe- WTR	0.56 ± 0.15	320 ± 3	354 ± 113	857 ± 70	2010 ± 73	3540 ± 32	5.4 ± 0.2	0.90 ± 0.00	76.2 ± 1.0	1.28 ± 0.02	2.07
Ca-WTR	1.71 ± 0.22	1.59 ± 0.07	1.28 ± 1.28	0.98 ± 0.08	46.8 ± 0.1	52.4 ± 1.7	0.1 ± 0.0	0.02 ± 0.00	0.39 ± 0.01	0.51 ± 0.01	2.56
Coal slag	2.16 ± 0.22	30.0 ± 2.3	0.92 ± 0.92	135 ± 6	197 ± 18	365 ± 15	3.3 ± 0.0	0.07 ± 0.00	27.6 ± 1.4	6.49 ± 0.02	0.33
Pro-sil	1.16 ± 0.11	1.27 ± 0.08	0.08 ± 0.08	1.76 ± 0.18	69.7 ± 1.8	74.0 ± 1.9	0.1 ± 0.0	0.03 ± 0.00	28.3 ± 0.7	4.11 ± 0.03	0.16
Dupont	1.87 ± 0.11	141 ± 0	0.00 ± 0.00	521 ± 7	247 ± 12	911 ± 5	5.7 ± 0.0	0.64 ± 0.03	2.76 ± 0.18	27.4 ± 0.4	1.93
Gypsum	2.07 ± 0.11	0.00 ± 0.00	0.25 ± 0.00	135 ± 5	42.7 ± 0.5	180 ± 5	9.0 ± 0.2	0.10 ± 0.02	0.19 ± 0.01	0.04 ± 0.01	65.7
Lime	1.95 ± 0.00	2.01 ± 0.17	0.00 ± 0.00	145 ± 17	314 ± 21	463 ± 4	0.6 ± 0.0	0.22 ± 0.00	0.05 ± 0.00	0.01 ± 0.00	497
dinoSoil	0.71 ± 0.16	44.8 ± 0.0	150 ± 2	20.5 ± 0.3	63.5 ± 2.1	280 ± 2	10.9 ± 0.0	0.12 ± 0.00	2.48 ± 0.03	2.21 ± 0.03	2.91

† M-Al-WTR = Manatee Al-WTR; O-Al-WTR = Okeechobee Al-WTR

‡ At solid: solution ratio of 1:60

$ Phosphorus saturation index [(oxalate P/oxalate Fe + oxalate Al)*100]; elemental concentrations in moles.

Oxalate-extractable P, Fe, and Al values were used to characterize the extent to which amendment total P (and Fe and Al) was associated with amorphous Fe and Al hydroxyoxides (McKeague et al., 1971). In almost all amendments, amorphous Fe and Al solids (and associated P) appear to constitute about ⅓ to ½ of total elemental contents, indirectly confirming the prevalence of recalcitrant (residue fraction) P-solids. The exception is the Manatee Al-WTR in which essentially all of the total P, Fe, and Al is oxalate-extractable. The data suggest that the Fe and Al in Manatee-Al-WTR should be highly labile (reactive) toward soluble P in soils. The phosphorus saturation index (PSI) has been suggested as an *a priori* measure of lability of P in solids and a measure of likely P-sorption capacity of the solid (Elliott et al., 2002). Materials with very low PSI values are expected to have large P-sorption capacities; very large PSI values are associated with materials of limited P-sorption and even P release. The index applies only to materials dominated by Fe and Al hydoxyoxides, so values for Ca-dominated materials are not meaningful. Using this concept (PSI), the Okeechobee Al-WTR is identified as a likely good sorbent for P, but the Manatee Al-WTR and Vigiron Fe-WTR are also identified as potentially useful P-sorbents.

Some trace elements (Cu and Zn) were analyzed by normal ICP techniques in the total elemental digests (Table 2). Additional trace elements of significant environmental concern (As, Se, and Mo) were determined using techniques (GFAA, hydride generation, etc.) with lower detection limits. Results of these analyses are given in Table 4. The intent was to identify potential trace element contamination problems that could arise from adding significant amounts of the amendments to agricultural land. Arsenic (As), selenium (Se), and molybdenum (Mo) are common constitutes of industrial wastes, and As and Mo are of particular concern in Florida.

Trace metal concentrations have been identified for “exceptional quality (EQ)” biosolids (Epstein, 2003), which have no limitations to land application. The EQ limits for As and Se are 41 and 36 mg kg^{-1}, respectively. There currently is no EQ value for Mo, but O’Connor et al. (2001) suggested a value of 40 mg kg^{-1}. Such EQ materials may be land applied at rates comparable to the amendment rates examined herein. Ceiling concentrations of trace metals in biosolids represent the maximum trace metal concentrations in biosolids that can be land-applied. Ceiling concentrations for As, Se, and Mo are 75, 100, and 75 mg kg^{-1}, respectively. Most amendments had As, Se, and Mo concentrations well below even the criteria for EQ biosolids, so the amendments are expected to represent minimal trace element risk to humans, animals, or the environment even when the amendments are land-applied at high rates.

Table 4. Trace metal analysis of amendments.

Source	**Form**	**Total Elemental (mg kg^{-1})**		
		As	**Se**	**Mo**
Manatee County	Al- WTR	9.48	1.72	19.7
Okeechobee Utilities Auth.	Al- WTR	12.5	2.32	< 2.21
Vigiron	Fe- WTR	43.9	2.32	70.8
Bradenton, FL	Ca- WTR	0.32	0.06	0.28
Nutrasource	Coal slag	51.3	22.4	173
Pro Chem	Pro-sil	1.36	3.83	41.8
Dupont	Dupont	1.77	13.3	< 0.30
Nutrasource	Gypsum	0.11	2.77	1.65
Franklin Minerals	Lime	1.92	0.78	0.62
dino-Soil	Leonardite	15.9	1.03	< 0.23

Exceptions include Vigiron (Fe-WTR), the coal combustion slag, and possibly Pro-sil. Arsenic concentrations in Vigiron and the coal combustion slag exceed EQ standards, but not the ceiling concentrations. The selenium concentration in Vigiron was low, but the Mo concentration exceeded EQ standards, and approached the ceiling concentration. Special attention to environmental concerns (As) and animal health issues (Mo) are apparently appropriate if Vigiron is used as a soil amendment, especially at high rates of application. Selenium concentrations in all amendments were generally low, but the value for the coal combustion slag (22.4 mg kg^{-1}) is noteworthy. The coal combustion slag also contains Mo at a concentration (173 mg kg^{-1}) that exceeds both EQ and ceiling concentrations for biosolids. A biosolids containing this much Mo could not be legally land-applied at any rate. The trace element content of the coal combustion slag appears to eliminate it as a viable soil amendment. Pro-sil contains a relatively high Mo concentration (41.8 mg kg^{-1}), which dictates careful monitoring of pasture grass Mo concentrations if Pro-sil is applied at high rates. The high pH of Pro-sil (Table 2) can be expected to exacerbate the Mo hazard if sufficient Pro-sil is applied to raise soil pH above pH 7 (O’Connor et al., 2001).

Selected properties of the composite soil representing the surface 15 cm depth from the Beaty Ranch are given in Table 5. Soil maps suggest the site soil could be classified as either the Immokalee (sandy, siliceous hyperthermic, Arenic Alaquods) or the Myakka series (sandy, siliceous hyperthermic, Aeric Alaquods). The composite soil is slightly acidic, low in organic matter, and reflects years of manure-P input in elevated readily soluble (KCl-, water-, and Mehlich-1- extractable) P values (Table 5). The PSI for the soil suggests that the amorphous Fe and Al hydroxyoxides in the soil are nearly saturated with P, and that little additional P-retention capacity on these solids exists.

Table 5. Selected properties of Beaty Ranch composite soil.

Source	Form	pH †	EC *u*S/cm	% Org Matter (LOI)	Sequentially extracted P (mg kg^{-1})						Total P	Mehlich 1 P	Water P	Oxalate Extractable (mg kg^{-1}) ‡			PSI $ (%)
					KCl	NaOH Pi	NaOH Po	HCl	Residue	Sum	(mg kg^{-1})			P	Fe	Al	
Beaty Ranch	soil	6.40 ± 0.00	76.5 ± 1.9	2.3 ± 0.1	12.3 ± 0.3	50.1 ± 1.5	49.1 ± 1.8	73.1 ± 1.7	17.1 ± 0.2	202 ± 5.4	209 ± 4.3	116 ± 3.4	8.29 ± 0.10	129 ± 7.8	10.7 ± 2.4	135 ± 8.4	80.4

† At solid:solution ratio of 1:2

‡ At solid:solution ratio of 1:60

$ Phosphorus saturation index [(oxalate P/oxalate Fe + oxalate Al)*100]; elemental concentrations in moles.

Lab equilibration study

A lab equilibration (incubation) study was conducted on the composite Beaty soil and amendments. Multiple rates of each amendment (Fig. 1) were added to the soil along with sufficient background electrolyte to achieve a 1:2 solids:solution ratio. The suspensions were reacted for 40 h on an orbital shaker (250 rpm). Amendment effectiveness was evaluated by comparing the amount of soluble P remaining in the solution phase of the equilibrated suspension to that in the control suspension (no added amendment). Results are given in Fig. 1, where the percent of soluble P remaining in solution is plotted as a function of the amendment treatments. The initial P in the control sample was 3.25 mg kg^{-1} soil.

The Fe-humate material derived from Ti mine waste (Dupont) sorbed the least P compared to all other treatments (Fig.1). This material also reduced the pH of the suspension to 4.48 at the highest application rate (10%) compared to a pH of 6.15 for the control. The other Fe-humate material (Vigiron Fe-WTR) was more effective at increasing P sorption, particularly at rates ≥5% (Fig. 1), and had little impact on the pH of the suspension. The Okeechobee Al- WTR was very effective at sorbing P, with < 10% (<0.3 mg P kg^{-1}) of the original P left in solution for all rates ≥ 0.5%. The Manatee Al- WTR was less effective than the Okeechobee Al-WTR, and rates ≥ 5.0% were required to reduce the original P in solution to < 15% of the original. Although the Okeechobee Al-WTR appears to offer outstanding sorptive capacity, it has limitations due to its low solids content (9%), which could complicate material transportation and handling.

The Pro-sil material reduced soluble P effectively, with < 20% of the original P being left in solution when amendment rates were ≥ 1.0%. However, this material impacted suspension pH, increasing pH one unit at the 0.1% amendment rate and four units at the 2.5% rate. The coal slag material had a smaller impact on pH, reducing pH one unit at the 10% amendment rate, while reducing the original amount of P in solution to < 5% of the initial value at amendment rates ≥ 5%. Coal slag was the most effective material for reducing soluble P of all the materials investigated when applied at high rates. However, at the 10% amendment rate, this material increased the EC (soil salinity) to 3360 *u*S cm^{-1}, which is double that of the control (1560 *u*S cm^{-1}). The increase in EC for dinoSoil was comparable to that of the coal slag material, with an EC value of 3540 *u*S cm^{-1} for the 10 % amendment rate. The pH decrease with increasing rates for dinoSoil was also comparable to the coal slag material, reducing pH one unit at the 10% amendment rate. However, dinoSoil did not sorb P as effectively as the coal slag; ~11% of the original P was left in solution at the 10% amendment rate.

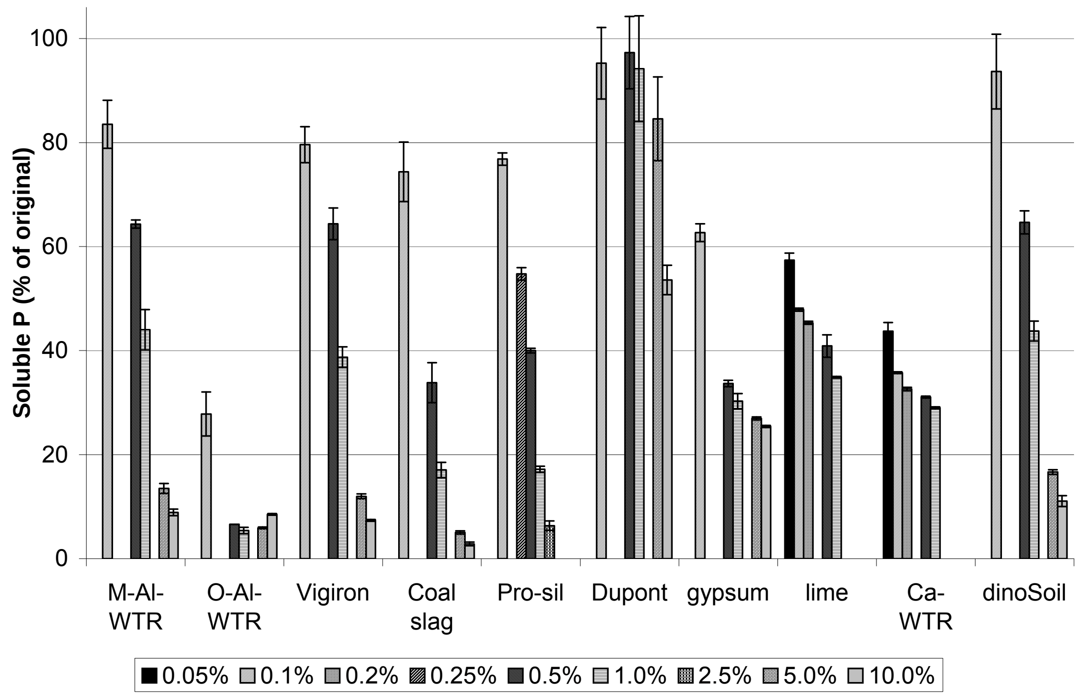

Figure 1. Percent of initial soluble P in solution after 24 h reaction. Amendment rates (key) are % by weight.

Gypsum caused a large increase in suspension EC; at the 0.5% amendment rate, the EC was 3800 *u*S cm^{-1}, and at the 10% amendment rate, the EC increased to 4640 *u*S cm^{-1}. Such elevated soil salinities could impair grass growth. No amendment rate of gypsum reduced soluble P below 25% of the initial concentration.

The lime and Ca-WTR produced similar results for both soluble P reduction and impact on suspension pH. The Ca- WTR consistently sorbed slightly more P than agricultural lime at the same amendment rate. The 1.0% rate of Ca- WTR reduced the amount of soluble P to ~ 30% of the initial P concentration. Neither amendment raised the pH of the system above pH 7.00 at the greatest (1.0%) amendment rate tested.

Results from the lab equilibration study suggested that no further work with the Fe-“humate” from Dupont was necessary. Agricultural lime and the Ca-WTR gave essentially the same results, so the lime material was dropped from further study. The Fe- and Al-based materials were clearly the best at reducing soluble P in the Beaty soil, especially at rates of 5% or more. Other materials (other than Dupont and lime) were maintained in the testing for completeness.

Small column leaching study

The small column study was designed to evaluate amendment effectiveness at reducing P leaching in a controlled laboratory setting. Soil used in the columns was the composite material collected from the Beaty Ranch. Soil (320 g) was packed into PVC columns (5 cm diameter x 17 cm long) to a depth of 13 cm at a bulk density of ~1.26 g cm^{-3}. Columns were equipped with a 2 cm drainage hole at the base, covered with screening to block soil loss. The study included 75 columns (8 amendments x 3 rates x 3 replicates + 3 controls). Amendments and amendment rates utilized are given in Table 6. Sixty mL of tap water (adjusted to pH 5) was added to each column, and allowed to infiltrate before amendments were applied to the soil surface. Columns were supported on racks, and loosely covered with clear plastic wrap to reduce moisture loss. The amended columns sat for 4 days before the first leaching event. Tap water (pH 5) equivalent to ~5 cm irrigation was added to each column weekly for a total of 8 weeks (~ 40 cm equivalent total), and resulted in a total of ~8 pore volumes of leachate being collected. Leachate was collected and its volume recorded. Leachate was analyzed for pH, EC, and soluble reactive P (Murphy and Riley, 1962) within 24 h of collection.

Table 6. Small column leaching study amendments and rates (by weight.).

Amendment	Rate (%)
Manatee Al-WTR	1.0, 5.0, 10.0
Okeechobee Al-WTR	0.1, 0.25, 0.5
Vigiron	1.0, 5.0, 10.0
Ca-WTR	0.1, 0.5, 2.0
Coal slag	0.5, 1.0, 5.0
Pro-sil	0.5, 1.0, 2.5
gypsum	0.5, 2.0, 10.0
dinoSoil	0.1, 1.0, 10.0

Amendment effectiveness in this study was quantified as the amount of P cumulatively leached in the 8 leachings, compared to the mass of P leached in the control treatment. Amendment impacts on leachate pH and EC were also of concern (data not presented).

Amendment impact on leachate pH was minor, and all leachate pH values were within ~0.3 pH units of the control, which averaged about 7.4. Leachate EC (a measure of leachate salinity) was generally unaffected by amendments, except for the highest rate (10%) of dinoSoil and most of the gypsum treatments. Gypsum is the most soluble of the amendments evaluated and was expected to furnish soluble Ca^{++} for reaction with soluble P throughout the column (Anderson et al., 1995). Most leachate EC values for the gypsum treatments were about 4 times (~2500 $\mu S\ cm^{-1}$) the value for the control throughout the leaching study. Only the smallest gypsum treatment (0.5%) resulted in smaller leachate EC values (~1500 $\mu S\ cm^{-1}$) in the last few leaching events. The elevated and constant EC values suggest that gypsum was able to maintain a relatively constant Ca^{++} activity and a constant potential for precipitation of Ca-P solid.

Leachate P concentrations (data not shown) in the gypsum treatments were nearly constant (~10 $\mu g\ L^{-1}$) for each leaching event, also suggesting solid phase control of P solubility. Control treatment leachate soluble P concentrations averaged ~30 $\mu g\ L^{-1}$ until leaching #8, when it decreased to ~20 $\mu g\ L^{-1}$. There was a correspondingly relatively constant reduction in the total mass of P leached in the gypsum treatments (Fig. 2). Gypsum was the most effective amendment in reducing P leaching loss, averaging ~35% of that lost from the controls. Other Ca-source amendments (e.g., Ca-WTR, Pro-sil, and coal slag) were not as effective, presumably because of limited solubility at the high pH of the soil.

The Al-WTRs were largely ineffective at controlling P leaching loss, but the Fe-WTR was superior to the Al-WTRs (Fig.2). Because the WTRs (surface applied) were not in direct contact with the majority of soluble P in the soil, effectiveness was limited by amendment solubility (releasing soluble Fe or Al) to react with soluble P beneath the zone of application. Direct adsorption of P onto the WTRs was minimized in the leaching protocol used herein. Very different results were observed when the Manatee Al-WTR was applied (surface or mixed with surface) to the surface of a P-deficient soil that was then loaded with P as fertilizer, poultry manure, or biosolids as surface applications (O'Connor and Elliott, 2001). Soluble P mobilized by leaching irrigations then passed through reactive sites on the Al-WTR or Al-WTR-amended soil, and leachate soluble P was significantly reduced (minimally, 50%) at Al-WTR rates of ≥ 2.5% (O'Connor and Elliott, 2001). Thus, Al-WTR can effectively reduce loss of soluble P added subsequently to, or in immediate contact with, amendment, but is much less effective at reducing loss of soluble P not in contact with the amendment. dinoSoil was about as effective as the Fe-WTR, and only reduced P loss by ~40% at the greatest application rate (Fig. 2).

Limited analyses of the selected leachates for trace elements (As, Mo, Se) revealed insignificant leaching of the potentially troublesome metals (data not presented).

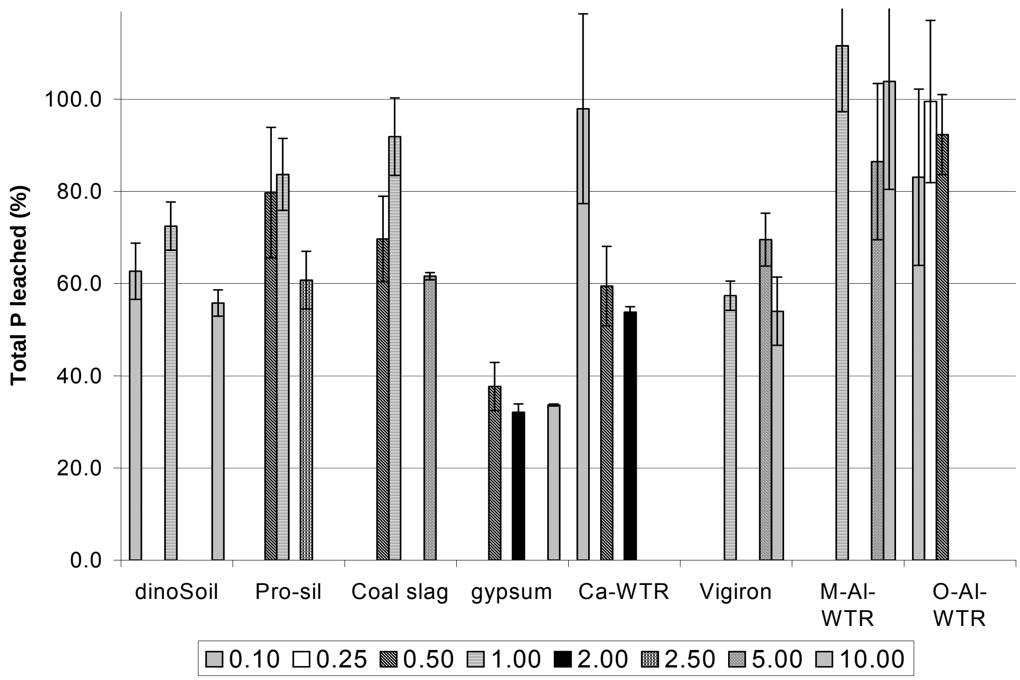

Figure 2. Percent P leached (100% = P leached in control) after 8 pore volumes of leachate. Amendment rates (key) are % by weight.

Simulated rainfall study

The final protocol used to evaluate amendment effectiveness was a simulated rainfall study using equipment and procedures specified by the National P Project protocol (National Phosphorus Research Project, 2001). The protocol specifies dimensions of runoff boxes (1 m long, 20 cm wide, and 7.5 cm deep), rainfall intensity (7.1 cm h^{-1},~equivalent to a 10-y, 24-h rain, applied from a height of 3 m above the soil surface), and soil packing and surface slope (3 degrees). The design was modified slightly in our experiments to quantify leaching of P in addition to runoff P by adding a second box under the first in a double-decker design. This design allowed collection of runoff and leachate simultaneously.

Air-dried soil was added to the top box and tapped to produce a depth of 5 cm and a bulk density of 1.4 g cm^{-3}. The soil was wetted to near field capacity and allowed to sit for 24 h. Amendments were surface-applied at the chosen rates (Table 7), and the boxes allowed to sit for another 48 h before beginning the first rainfall event.

Rainfall was applied using a TeeJet™ HH-SS50WSQ nozzle. The nozzle was ~3 m above the soil surface during rainfall events. The operating pressure was ~4 psi, and produced a discharge of ~210 cm sec^{-1} (7.1 cm h^{-1}). Tap water was adjusted to pH 5 with 1M HCl to mimic rainfall pH in Florida. Runoff was collected for 30 min after runoff began for each box. Following the completion of the first runoff cycle, the boxes were stored on racks for 48 h before the second cycle began. The same pattern was followed for a third rainfall event.

Table 7. Rainfall simulator study amendments and rates (% by weight.).

Amendment	Rate (%)
Manatee Al-WTR	1.0 and 2.5
Okeechobee Al-WTR	0.1 and 1.0
Vigiron	1.0 and 2.5
Ca-WTR	0.1 and 1.0
Pro-sil	0.5 and 1.0
Gypsum	0.1 and 1.0
dinoSoil	1.0

The collected runoff from each box was weighed, and a 1 L sub-sample collected under constant vortex to promote uniformity of the sample. A second portion of the runoff sample was filtered (0.45 μm) using a vacuum pump to obtain ~100 mL sub-sample. The leachate sample was thoroughly mixed and sub-sampled (250 mL) for later analysis. All samples were refrigerated until P analyses were performed, usually, the next day. Filtered runoff samples were analyzed for pH, EC, soluble reactive P (SRP), and total dissolved P (TDP) (Eaton et al., 1995). Analyses performed on the sediment-laddened runoff samples included pH, EC, total and volatile solids, total P (Eaton et al., 1995), and biologically available P (BAP) (Pierzynski, 2000). Soluble P, pH, and EC were measured on leachate samples. All P analyses were conducted using the ascorbic acid method (Murphy and Riley, 1962).

Total average runoff volumes ranged from 16.1 to 24.3 L (Fig. 3), with the dinoSoil treatment yielding the greatest volume. DinoSoil contains montmorillonite clay and, when applied at the 1% rate, created a nearly impervious layer that sealed the soil surface.

There was no leachate from 2 of the 3 replicates of the dinoSoil treatments, confirming the sealing of the soil surface suggested by the runoff data. The average leachate volumes for the other treatments ranged from 3.2 to 7.6 L (Fig. 4). In most cases, leachate volumes tended to decrease as amendment rates increased, except for Pro-sil treatments where leachate volume increased with amendment rate.

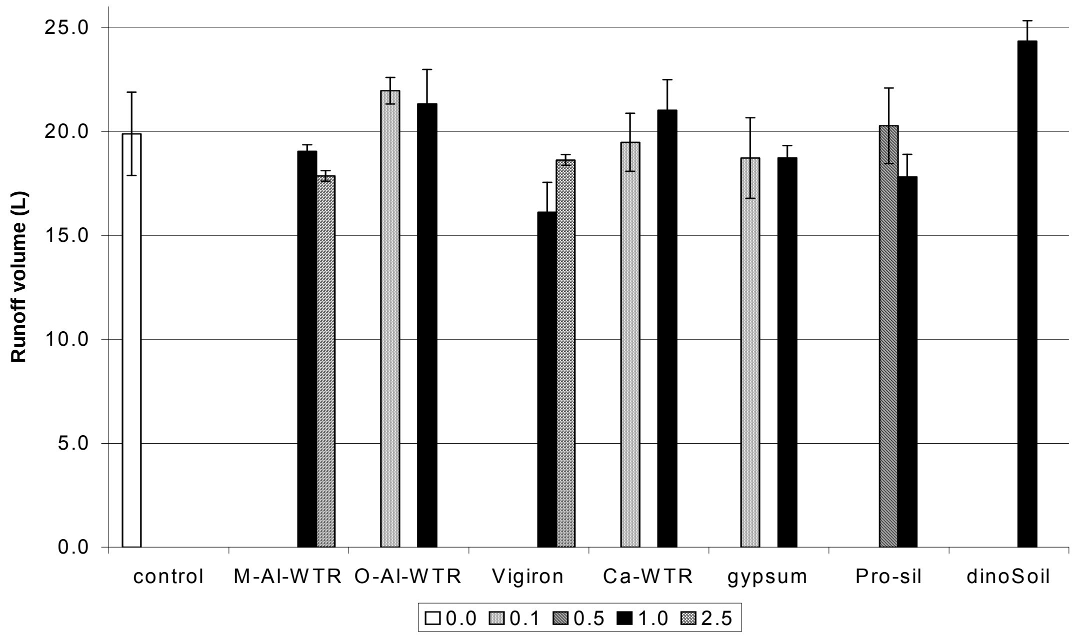

Figure 3. Total runoff volume (L) collected from three rainfall events. Amendment rates (key) are % by weight.

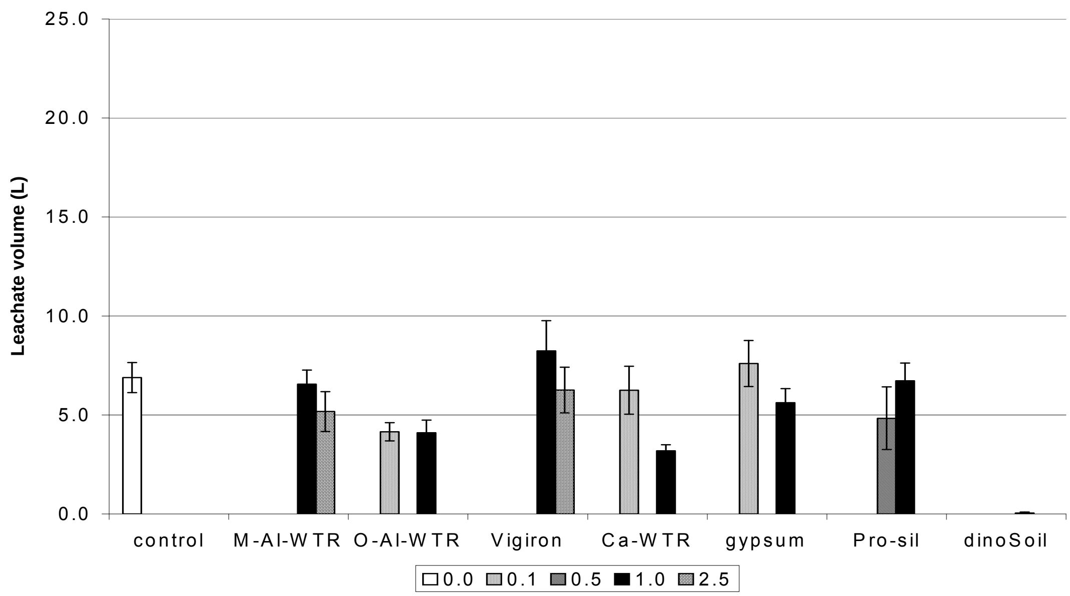

Figure 4. Total leachate volume (L) collected from three rainfall events. Amendment rates (key) are in % by weight.

The average runoff SRP in the control was 0.63 mg P, and the range for the treated soils was 0.18 to 2.14 mg (Fig. 5). All amendments applied at 1% reduced runoff SRP compared to the control, with the exception of the gypsum treatment. dinoSoil reduced runoff SRP by >60%, and both Al-WTRs reduced runoff SRP by ~50%. The Ca-WTR and Vigiron (Fe-WTR) reduced runoff SRP by ~40%, and Pro-sil reduced runoff SRP by ~20% compared to the control. Data for runoff total dissolved P (TDP) were similar to the runoff SRP data, and are not presented. The similarities suggest little contribution of organic P (detected in TDP, but not runoff SRP) to soluble P in runoff.

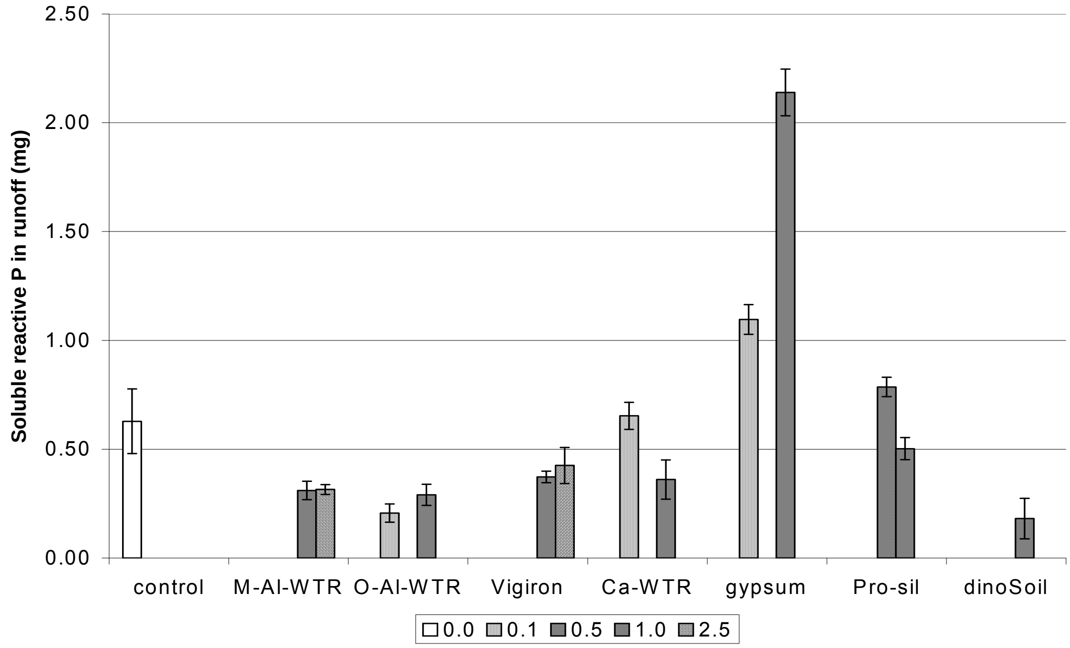

Figure 5. Soluble reactive P in runoff (mg, total of 3 rainfall events). Amendment rates (key) are in % by weight.

Runoff total P (soluble + particulate P) in the control was 14.3 mg P, and the range for the amendment treatments was 3.1 to 65.6 (Fig. 6). The greatest value was measured for dinoSoil, reflecting the easily transported colloidal nature of the material and its inherent total P content (Table 1). The two Al-WTRs, Vigiron, and Pro-sil effectively reduced runoff total P.

Phosphorus extracted by the Fe-strip method is referred to as biologically available P (BAP), as it has been correlated with algae-available P in runoff and sediments. Values of BAP, thus, should represent environmentally significant P better than total runoff P, as total runoff P includes non-labile P. The amount of runoff BAP in the control was 1.8 mg P, and ranged from 0.23 to 4.9 mg P for the amendments (data not shown). The dinoSoil BAP value was dramatically lower than runoff total P, suggesting that much of the runoff total P was non-labile. The two WTRs were again the most effective in reducing BAP compared to the control, although Pro-sil (~59%), Vigiron (~45%), and dinoSoil (~45%) were also effective.

Soluble reactive P (SRP) in the leachate from the control was 6.6 mg P, compared to SRP values in the amendment treatments, which ranged from 3.6 to 9.4 mg P (Fig. 7). The Ca-amendments (gypsum and Ca-WTR) and Vigiron (at the 1% rate) were the most effective amendments in reducing leachate SRP. dinoSoil allowed minimal leaching and SRP in the leachate, but promoted excessive surface runoff via its sealing effect on the soil. The two Al-WTRs and Pro-sil were largely ineffective at controlling leachate soluble P, especially at the common 1% amendment rate.

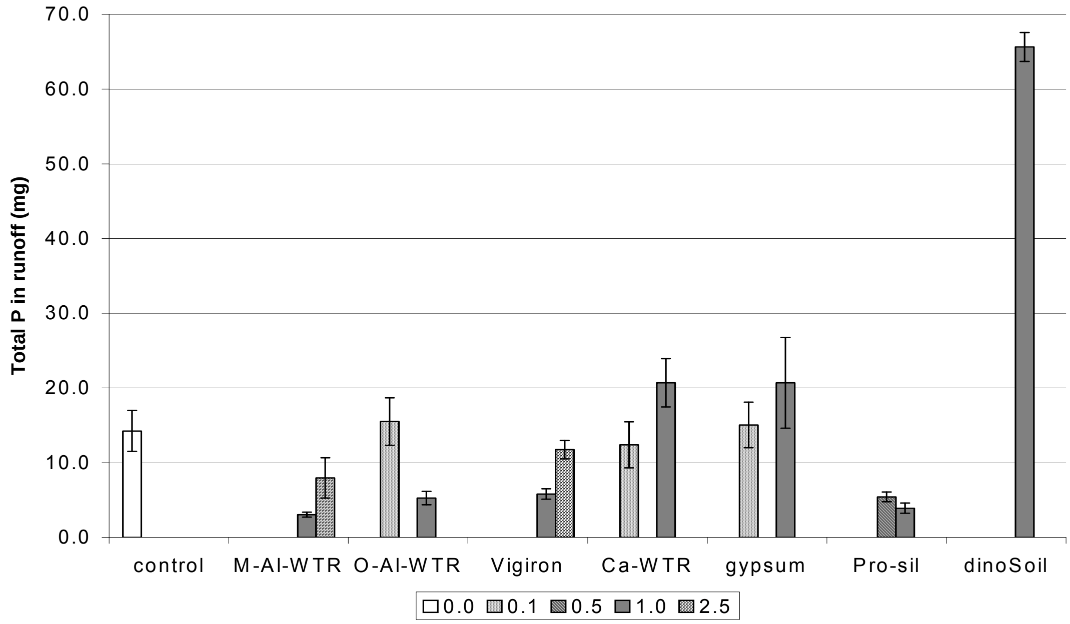

Figure 6. Total P in runoff (mg, total of 3 rainfall events). Amendment rates (key) are in % by weight.

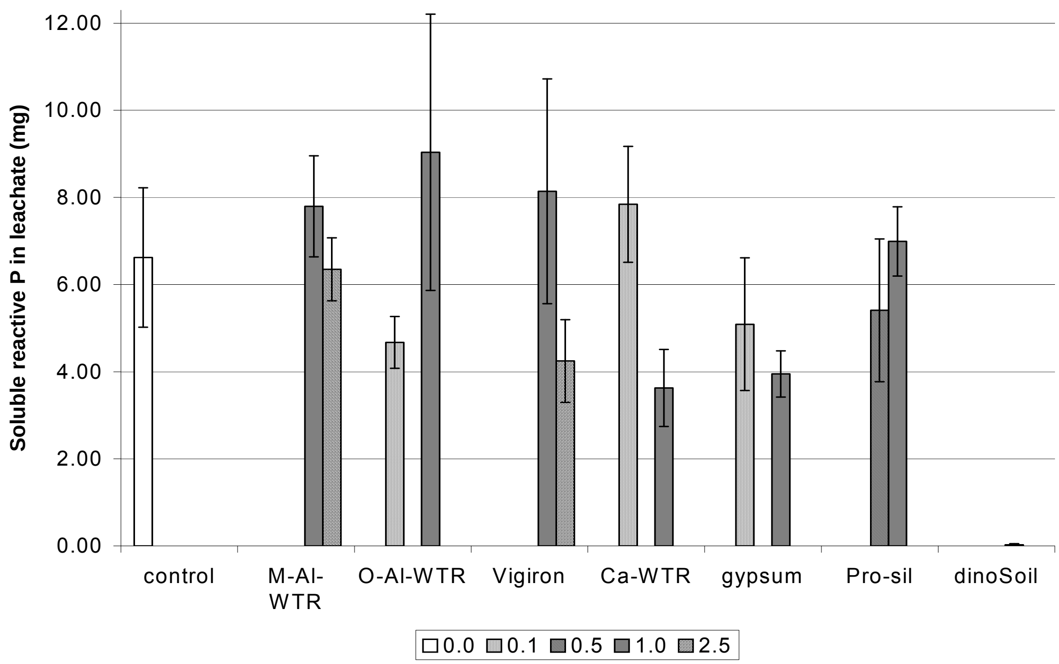

Figure 7. Soluble reactive P in leachate (mg, total of 3 rainfall events). Amendment rates (key) are in % by weight.

The various runoff and leachate parameters were statistically analyzed using a ranking scheme intended to identify the most effective treatment(s). The rankings and associated LSD values for each parameter ranking are given in Table 8. The total column tallies ranking points for all parameters measured in both runoff and leachate. The Total ranking identifies the two WTRs and dinoSoil as the best amendments. The LSD analyses for individual runoff P parameter rankings generally confirm the superiority of the three amendments as well, although only dinoSoil significantly reduced leachate P compared to the control.

Recall that dinoSoil essentially sealed the soil surface so little rainfall infiltrated and, thus, minimized leaching. There was minimal effect of amendment rate among the Al-WTRs; however, the low rate of Okeechobee Al-WTR (0.1%) was about as effective as either Al-WTR applied at 1%, or the Manatee Al-WTR applied at 2.5%.

Table 8. Rankings and associated LSD values for various P parameters.

Amendment	**rate**	**leachate**		**runoff**			
		SRP	SRP	TDP	TP	BAP	Total
	%						
control	0.0	9ab	10cde	11cd	9bc	12bc	51(12)*
M-Al-WTR[†]	1.0	11ab	4fg	4efg	1e	2g	22(1)
M-Al-WTR[†]	2.5	8ab	5fg	3fg	6cde	3fg	25(4)
O-Al-WTR[†]	0.1	5b	2g	2fg	11bc	4fg	24(2t)
O-Al-WTR[†]	1.0	14a	3fg	6efg	3de	1g	27(5)
Vigiron	1.0	13ab	7fg	8def	5de	8def	41(10)
Vigiron	2.5	4bc	8ef	10cd	7cd	11cd	40(8t)
Ca-WTR	0.1	12ab	11cd	9de	8cd	10cd	50(11)
Ca-WTR	1.0	2bc	6fg	5efg	13b	9cde	35(7)
Pro-sil	0.5	7ab	12c	12bc	4de	5efg	40(8t)
Pro-sil	1.0	10ab	9def	7def	2e	6efg	34(6)
gypsum	0.1	6ab	13b	13b	10bc	13b	55(13)
gypsum	1.0	3bc	14a	14a	12b	14a	57(14)
dinoSoil	1.0	1c	1g	1g	14a	7def	24(2t)
*cumulative points (actual ranking). Note lower cumulative points represents less P loss.							

[†] M-Al-WTR = Manatee Al-WTR; O-Al-WTR = Okeechobee Al-WTR

Selection Summary

Ten materials (Table 1) were selected initially for evaluation as possible amendments to control soluble P in Beaty Ranch soil. A series of evaluation protocols allowed deselecting most amendments and identified the one or two amendments worthy of field investigation. A summary of the pertinent criteria used to select or deselect amendments is given below:

1. DuPont Fe-"humate" - deselected because of minimal P sorption capacity (Fig.1).

2. Coal slag - despite good adsorption and leaching control properties, the material is deselected because of troublesome trace element contents, especially Mo and As (Table 4), and because the rates of coal slag required for P control could detrimentally affect soil pH and EC.

3. Pro-sil - despite effective P sorption, effective leaching control, and moderate runoff control, the material is deselected because the rates required for P control can raise soil pH excessively which, when combined with a moderately high Mo content, could create an undesirable soil environment for pasture grass growth and grass quality that may threaten livestock health (molybdenosis).

4. Gypsum - very effective at controlling P leaching, but ineffective at P sorption and P runoff control. Also deselected because rates necessary for P control may result in soil salinity incompatible with good pasture grass growth.

5. Lime and Ca-WTR – behaved essentially the same in all protocols. Likely effective in initially acid soils requiring pH adjustment, but not in soils with pH values ≥7, where lime solubility is limited. Because the pH of the Beaty Ranch field site is already 6.4, little (or no) liming agent would be recommended (for most pasture grasses) and low rates of

amendments are expected to be ineffective. Both liming agents are, thus, deselected for field evaluation.

6. Vigiron (Fe-WTR) - moderately effective at sorbing P and reducing leaching, but only fair in controlling P runoff. Deselected because it contains moderately high concentrations of Mo and As, and may release immobilized P under reducing conditions.

7. dinoSoil - high rate (1%) slightly effective at sorbing P and reducing leaching, but a top performer in runoff simulations. High cost (~$145/T), however, likely makes the amendment impractical for large scale use.

8. Manatee and Okeechobee Al-WTRs - effective P sorbers, but ineffective at controlling P leaching when soluble P is below the zone of amendment incorporation. Very effective at controlling P leaching when soluble P is made to contact WTRs (amendment incorporation, or soluble P added after amendment addition). The Al-WTRs dominated the best materials in runoff simulations. The Okeechobee material was uniformly better when applied at 1%, whereas the Manatee material requires rates ≥2.5%. The Okeechobee material is locally available, but its low solids content (~9%) create handling and transportation problems. Total annual production of the Okeechobee residual is estimated at only 250 to 300 Mg (~tons), which limits its use to minimal acreage at the 1% (10 T/A) rate. Relatively high rates (~25 T/A) of the Manatee material are needed to be effective, but the material's dry condition makes handling/application easy, and large quantities of the material are available (~4000 Mg/y). Thus, we recommend field evaluation of the Manatee Al-WTR at the 2.5% rate; the material should be surface applied on one plot and incorporated (to 5 cm) on the other plot.

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APPENDIX B

Soil Amendments Rainfall Simulator Evaluation

Introduction

Al-WTR from Manatee County was evaluated as an amendment applied to a manure-impacted soil from the Lake Okeechobee Basin in a rainfall simulation study. The Al-WTR was applied to the soil via surface application and incorporation into the soil at a rate of 2.5% of dry soil weight as shown in Table 1. Surface application (T1) of Al-WTR reduced soluble phosphorus (SP) concentrations in runoff by 77% but did not reduce the P concentrations in the subsurface flow or the leachate. Incorporation of Al-WTR (T2 and T3) reduced surface runoff P concentration by approximately 45%. Phosphorus concentrations for subsurface flow were reduced by 37% by mixing the Al-WTR in the top 10 cm of soil (T2) and by 90% by mixing the Al-WTR with the whole soil layer (T3). Phosphorus concentrations for leachate were reduced by 11% by mixing the Al-WTR in the top 10 cm of soil (T2) and by 94% by mixing the Al-WTR with the whole soil layer (T3). Although shoot biomass (forage yield) varied considerably, there was no trend that suggested an adverse effect of Al-WTR on either shoot or root yield (Table 2).

Trends in biomass P and N concentrations for the six simulation events are shown in Figures 1 and 2, respectively. Values are averaged over all treatments since there were no significant differences between treatments. Pre-treatment plant material contained approximately 3,200 µg P g^{-1} and approximately 500 µg N g^{-1}. Biomass P concentration remained relatively stable during the six simulation events because of the high P concentrations existing in the soil. However, biomass N concentrations decreased over the study period because no additional N was added to the soil. When averaged over all simulation events, P and N concentrations in both shoot and root biomass were not affect by any of the Al-WTR treatments (Figures 3 and 4).

Aluminum concentration in the shoots did not significantly increase between simulation events although there was high variability for simulation 6 which resulted in a slightly higher average concentration for that time period (Figure 5). The variability observed in simulation 6 can attributed to shoot contamination when cutting shoots to the soil surface. This is supported by a large standard deviation observed for simulation 6. Since this was the last harvest, shoots were cut down to the soil surface in order to get a measure of total above-ground biomass. This apparently resulted in inclusion of some soil particles in the biomass sample. When averaged over all simulation events, there were no effects of Al-WTR treatments on biomass Al concentrations. Biomass Al concentrations were in the low range of concentrations reported in the literature for forages. Aluminum concentrations in the roots were highly variable. As with the shoots, Al contamination is believed to be a factor because of the high level of variability in the results even though roots were washed multiple times to remove soil and WTR particles. The standard deviation was so high that there was no significant difference among treatments (P>0.05). There was no visual evidence of any adverse effects of Al-WTR on root growth.

To achieve the best results for reducing P loss in both surface runoff and subsurface flow/leachate from highly impacted soils, Al-WTR (2.5 % of dry weight of soil) should be first mixed with the impacted soil depth to reduce subsurface flow/leachate P loss AND then added to the soil surface to minimize P loss in runoff. For an un-impacted area with low initial soil P concentration that is going to be used for manure application, surface application of WTR would likely be sufficient to minimize P loss. Application of Al-WTR at 2.5% is not expected to adversely affect forage yield or quality of stargrass based on the uniform values of yield and P, N, and Al concentrations between treatments. Full details of this study are report in the Master's Thesis written by T.J. Rew that is included as Attachment 1.

Table 1. Treatments used in rainfall simulation study.

Treatment	Descriptions
C1†	No WTR applied
T1†	WTR surface applied.
T2†	WTR incorporated into 0-10 cm soil layer.
T3‡	WTR incorporated into 0-20 cm soil layer.
C2‡	No WTR applied

†0-10cm and 10-20cm soil layers placed in box in sequence.
**0-10 and 10-20 cm soil layers mixed prior to placement in box.

Table 2. Stargrass shoot and root biomass prior to each simulation event as influenced by Al-WTR application.

Treatment	Shoots (g dry weight)								Roots
	Sim 1	Sim 2	Sim 3	Sim 4	Sim 5	Sim 6†	Avg	Final‡	
C1	35a	17a	38a	16a	8a	n/a	23	57a	34a
T1	17b	8b	24b	19a	4b	n/a	14	41a	30a
T2	35a	21a	35ab	18a	7a	n/a	23	50a	32a
T3	31a	14a	30ab	17a	7a	n/a	20	56a	34a
C2	14b	6b	28ab	21a	9a	n/a	15	47a	29a

† No harvest due to slow winter growth.
‡ Final = All above-ground biomass harvested after simulation 7.
Treatments with the same letter are not significantly different *P*<0.05.

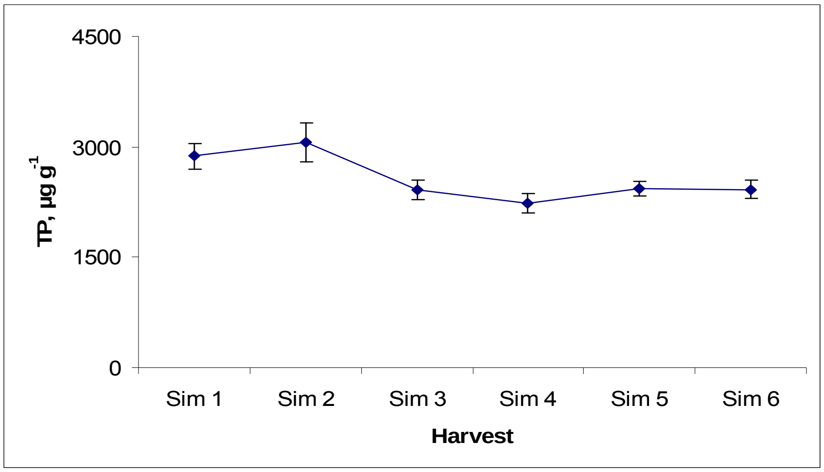

Figure 1. Average phosphorus concentrations and standard deviations per simulation in stargrass as affected by WTR.

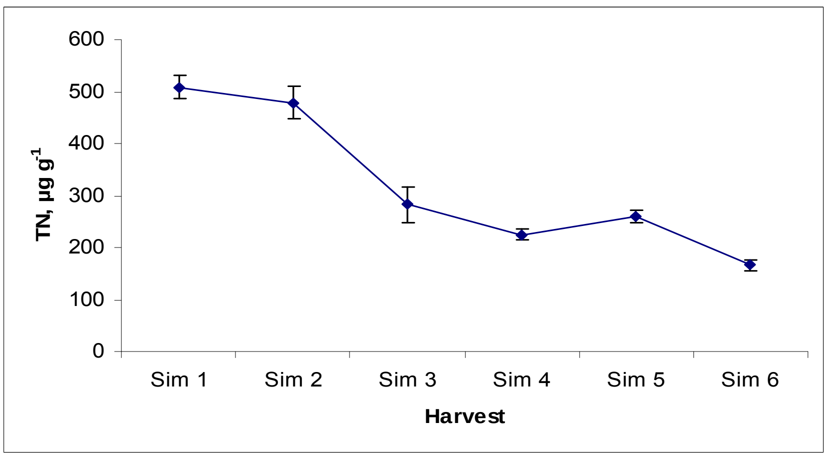

Figure 2. Average nitrogen concentrations and standard deviations per simulation in stargrass as affected by WTR.

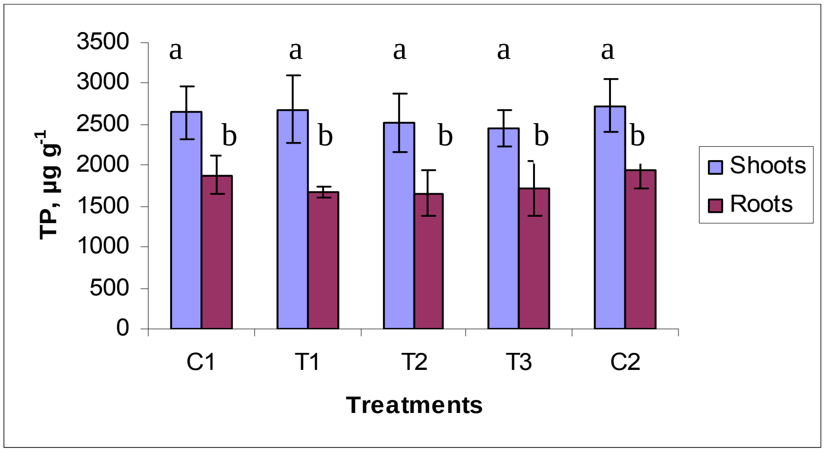

Figure 3. Average phosphorus concentrations and standard deviations in stargrass in treatments.

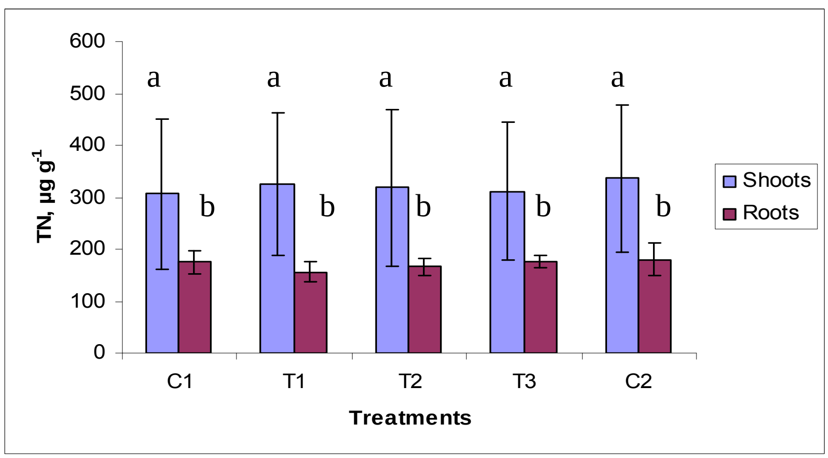

Figure 4. Average shoot and root nitrogen concentrations and standard deviations per treatment in stargrass as affected by WTR.

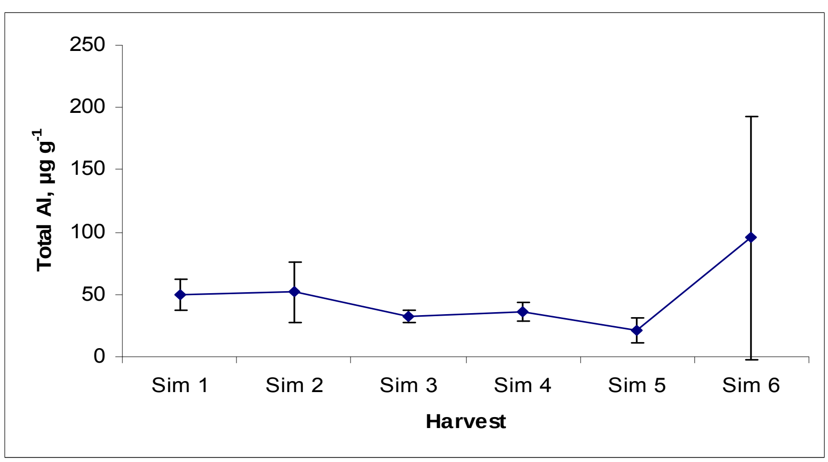

Figure 5. Average shoot aluminum concentrations and standard deviations per simulation in stargrass as affected by WTR.

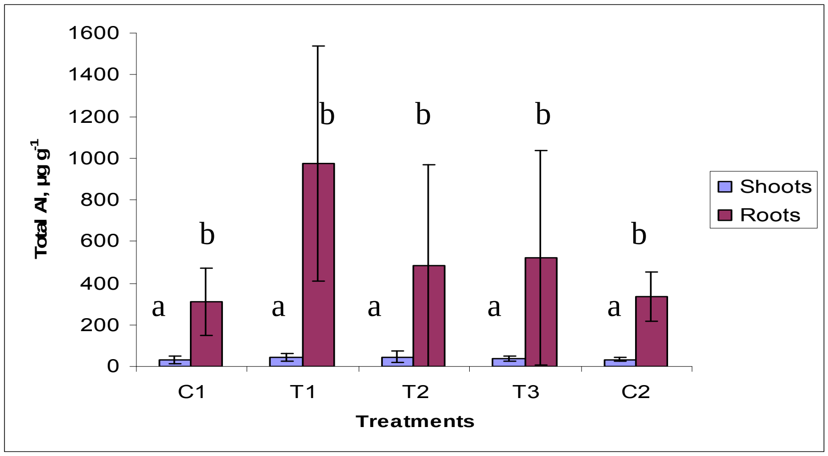

Figure 6. Average aluminum concentrations and standard deviations per treatment in stargrass as affected by WTR.

APPENDIX C

Evaluation of Ditch Fencing and Culvert Crossing Best Management Practice for Phosphorus and Nitrogen Discharges in the Lake Okeechobee Basin

ABSTRACT

Cow-calf operations are a source of excessive Nitrogen (N) and Phosphorus (P) loadings in the Lake Okeechobee basin, Florida (USA). A study was conducted at a beef-cattle ranch to evaluate ditch fencing and culvert crossing (DFCC) as a best management practice (BMP) implemented to improve downstream water quality. The BMP was implemented within a ditch section of 170 m in the ranch. The concentrations and loadings of Dissolved Organic Nitrogen (DON), Total Nitrogen (TN), and Total Phosphorus (TP) were compared for a pre-BMP period (June-October, 2005) versus a post-BMP period (June-October, 2006-08) between the upstream and downstream nutrient loads from the BMP site. During the pre-BMP period, P loading was 123.10 kg higher at downstream as compared to that at upstream from the BMP site. During the post-BMP periods in 2006 and 2008, downstream loadings of P were 17.31 and 88.03 kg lower as compared to those upstream from the BMP site. Downstream P loading was 35% higher than that for the upstream during the pre-BMP period while downstream P loadings were 32 and 11% lower during the post-BMP periods of 2006 and 2008, respectively. There were net reductions of N loadings at the BMP site during the 2006 and 2008. Unusually dry conditions during 2007 resulted in the addition of P and N at the BMP site, probably due to the release of P and N from soil and aquatic plants. Average upstream and downstream TP loadings were 294.53 and 263.64 kg for the three post-BMP periods indicating 10% reduction of TP. On average, upstream and downstream TN loadings from the BMP site were 674.65 and 600.85 kg for the three post-BMP periods indicating an 11% reduction of TN. Overall, the BMP is cost-effective in improving water quality downstream, with average capital costs for nutrient removal estimated at \$7.74 to \$22.05 per kg phosphorous, and \$3.23 to \$9.23 per kg nitrogen. These costs are an order of magnitude lower than the commonly used stormwater treatment areas.

Keywords: Ditch fencing and culvert crossing BMP, water quality, N and P loadings, macrophytal activity, denitrification

1. Introduction

Lake Okeechobee is a large, multi-functional lake located at the center of the Kissimmee-Okeechobee-Everglades aquatic ecosystem in south-central Florida. Excessive phosphorus (P) loading is one of the serious problems facing the lake (Boggess et al., 1995; Bottcher et al., 1995; Rice et al., 2002). Concentration of total phosphorus (TP) in the lake water is more than double the goal of 40 parts per billion set by the Florida Department of Environmental Protection (FDEP) to prevent an imbalance to the lake's flora and fauna (Thomas, 2003; FEEP, 2008). Runoff from dairies and cow-calf operations is considered to be the primary source of external P loadings to Lake Okeechobee. The cooperation of the dairy industry in implementing the agricultural Best Management Practices (BMPs) combined with a dairy buy-out program in which approximately 14,000 dairy cows were relocated outside of the Lake Okeechobee drainage basin (Schmitz et al., 1995), has resulted in substantial reductions of P runoff from dairies. Cow-calf operations are the most important remaining source of external P loadings to the lake due to the vast area under this land use.

A manual on water quality BMPs for cow-calf operations was published by the Florida Cattlemen's Association in 1999 in cooperation with the Institute of Food and Agricultural Services, University of Florida (UF/IFAS), Florida Department of Agriculture and Consumer Services (FDACS), FDEP, and South Florida Water Management District (SFWMD). An updated version of the manual was published in 2008 by the FDACS. The purpose of this manual was to educate and encourage voluntary compliance of a variety of BMPs for cow-calf operations. However, water quality benefits of most of the BMPs in the manual have not yet been quantified. There is a need to quantify the effects of these BMPs for making basin-wide plans for achieving the Total Maximum Daily Load (TMDL). Keeping cattle out of waterways with ditch fencing and culvert crossings within a ranch is one such BMP which is promising with regards to reducing N and P loadings in the basin. Cattle exclusion has been used in the literature to refer to a variety of management practices including the ditch fencing and culvert crossing (DFCC) BMP and/or providing alternate drinking water source for cattle, in order to eliminate the direct deposition of cattle feces and urine into ditches, streams, creeks, or rivers.

The ditch fencing and culvert crossing BMP seems to be an effective measure to improve the ground and surface water quality. Grazing cattle in pastures with unfenced streams and ranch drainage network contribute significant loads of nutrients to surface waters (Byers et al., 2005). Line et al. (2000) conducted a study to estimate the effects of cattle exclusion on water quality by fencing a 335 m long riparian corridor in North Carolina. Water samples were collected before (pre-BMP: 81 weeks) and after (post-BMP: 137 weeks) BMP installation. They found that post-BMP mean weekly loadings of Total Kjeldahl Nitrogen (TKN), Total Phosphorus (TP), Total Suspended Solids (TSS), and total solids (TS) were reduced by 79, 76, 82, and 83%, respectively and that these reductions were statistically significant ($p < 0.05$). However, the decrease in loads of weekly Nitrogen Dioxide (NO_2-N) and Nitrate-N (NO_3-N) were not statistically significant. About 90% of loadings for all nutrients except NO_2-N and NO_3-N, occurred during storms for both pre- and post-BMP periods. In the case of NO_2-N and NO_3-N, considerably larger load contributions were from groundwater while run-off contributed only 45-58% of the total loads. Line et al. (2000) noted that NO_2-N and NO_3-N contributions to groundwater from a waste application field and a waste storage pond at the site probably masked any reduction caused due to the BMP. Sheffield et al. (1997) evaluated alternative water sources as an option to reduce cattle entry to a stream in Virginia. Their study involved different durations and seasons for the pre-BMP (August 1994-April 1995) and post-BMP (April 1995-October 1995) monitoring. Sheffield et al. (1997) concluded that off-stream water sources reduced the in-stream cattle traffic which eventually reduced the TSS, Total Nitrogen (TN), and TP concentrations by 90, 54, and 81%, respectively. Concentrations of NO_3-N and orthophosphates (PO_4) were increased during the post-BMP period. The reason given by the authors for the increase in NO_3-N loads during post-BMP period was the high air temperature which increased the concentrations of the soluble NO_3-N during the post-BMP period (which included the record highs in the summer of 1995). The greater soil temperature during the post-BMP period might have increased the nitrification process, thereby, increasing the NO_3-N level (Sheffield et al., 1997).

A study was conducted by Meals et al. (2001) to evaluate the effectiveness of livestock exclusion along with stream bank restoration, and riparian zone protection in the Lake Champlain watershed located partly in Vermont/New York (USA) and Quebec (Canada). The study results indicated reductions in TP concentration (25%) and load (42%) in the treated watershed compared to a control watershed as the combined effects of livestock exclusion, streambank restoration, and riparian zone protection. Galeone et al. (2006) conducted a study in Lancaster, Pennsylvania on the effects of stream bank fencing on surface water quality. The results based on the effects of the BMP on water quality between the control and treatment watersheds suggested significant decrease in NO_3-N, NO_2-N, ammonia (NH_3-N), TKN, and TP loads.

In a review, Dillaha (2007) noted that in all the studies conducted on cattle exclusion there were significant experimental limitations. However, these studies provided some relevant information that could be helpful in estimating the effectiveness of the BMP. One of the limitations of some studies was that different seasons were used for pre- and post-BMP periods. For example, in the study conducted by Sheffield et al., (1997), the pre-BMP period was during winter, and the post-BMP period was during summer. For the earlier mentioned study by Galeone et al. (2006), Dillaha (2007) noted that several factors other than the BMP might have resulted in reductions in flows and P concentrations: pre-BMP rainfall was 11% higher than post-BMP period which resulted in reduced stream flow during post-BMP period; N and P fertilizer applications decreased by 27-30% during post-BMP period; and the number of livestock decreased by 50% during post-BMP period. In the study conducted by Line et al. (2000), the pre-BMP period had more storms as compared to that during the post-BMP period. The larger storms during the pre-BMP period were expected to produce greater loadings to the streams than those during the post-BMP period. In the study conducted by Sheffield et al. (1997), post-BMP rainfall was 54% higher than that during the pre-BMP period. Variability in factors such as climatic, fertilizer, and cattle density observed in these studies has limited the ability to attribute the water quality improvements to cattle exclusion BMPs.

In light of the above limitations of cattle exclusion studies and regional differences in weather, soil and hydrologic factors, conclusions drawn from one soil-hydrologic region may not be applicable universally. For example, soils and hydrology of south Florida are quite different from other regions within U.S. due to poorly drained sandy soils, shallow water table and nearly flat topography. Furthermore, as opposed to natural streams passing through the ranchland in other states, most of the ranches in south Florida have drainage ditches that drain the area to make it suitable for improved pasture and cattle production. Due to relatively flat topography, flow rates in the ditches are slow. The flows in the ditches mainly occur during the wet period (June-October). This is due to the fact that 70% of the total annual rainfall occurs during the wet period in this region (Shukla et al., 2008). Given such differences in soils, climate, hydrology, and landscape, cattle exclusion studies conducted elsewhere in the U.S. have limited applicability to south Florida. Many of the studies conducted on cattle exclusion evaluated the BMP effects with regards to nutrient concentrations and not on the loadings. However, given that water quality goals for the basins require load reductions, BMPs need to be evaluated with respect to loadings. The goal of this study was to evaluate the effectiveness of DFCC BMP in a ranch located within the Lake Okeechobee basin in south Florida for reducing N and P loadings.

2. Materials and Methods

2.1. Site description

The study site is a commercial cow-calf ranch located in Southwest Okeechobee County, Florida. The site encompasses an area of 249.8 ha within the ranch. Soils in the area are typically poorly drained and highly sandy (NRCS, 2003). The land use within the ranch includes improved pastures, upland hardwood forests, and non-forested wetlands. The ranch is dominated by improved pastures laced with shallow ditches for drainage. Native habitats in the ranch include seasonal wetlands, sawgrass marshes, palm-oak hammocks and live oak woodlands. The three major improved forage types are bahia, floralta, and stargrass.

The surface flow (drainage and runoff) from the ranch moves in a southerly direction and is discharged to the Kissimmee River which eventually empties into Lake Okeechobee. The ranch can be divided into two sub-watersheds, termed as sites 1 and 2. Discharges from sites 1 and 2 combine and flow through the main drainage ditch (170 m) and exit the ranch.

2.2. Stream fencing and culvert crossing BMP

A pre- and post-BMP monitoring design was used to evaluate the DFCC BMP at the main drainage ditch which drains the entire ranch. The BMP involved the installation of a crossover culvert and fencing to route the cattle over the ditch section instead of through it (Figure 1). Before the BMP implementation, there existed an old cattle crossing pathway almost midway in the ditch, which over the years resulted in erosion of the ditch bank (Figure 2). The eroded soil was deposited in the ditch bed resulting in an elevated patch of land in the middle of the ditch. The drainage ditch was also one of the deepest ditches in the ranch which resulted in availability of some water most of the year. The main drainage ditch provided drinking water as well as an area for the cattle to cool off. The culvert for the BMP was constructed during 2004 (prior to the pre-BMP monitoring period in 2005) and the fencing was installed in January 2006 (Figure 3).

Trapezoidal flumes (F1, F2, and F3, respectively) were installed at the outlets of the sites 1 and 2, and the outlet of the ranch to measure the flow volumes. Henceforth, the upstream and downstream ends of the main ditch (170 m) (BMP site) will be referred to as inflow and outflow points, respectively. By comparing P and N loadings at inflow and outflow points, the P and N (mass) addition/removal at the BMP site could be quantified.

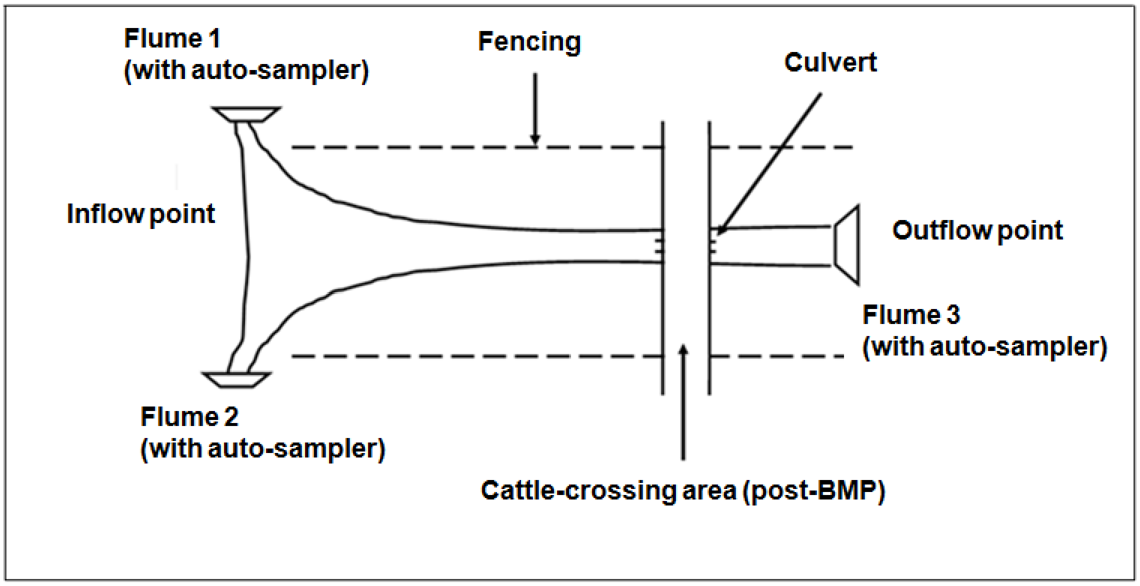

Figure 1. Inflow and outflow monitoring system for the Ditch Fencing Cattle Crossing BMP.

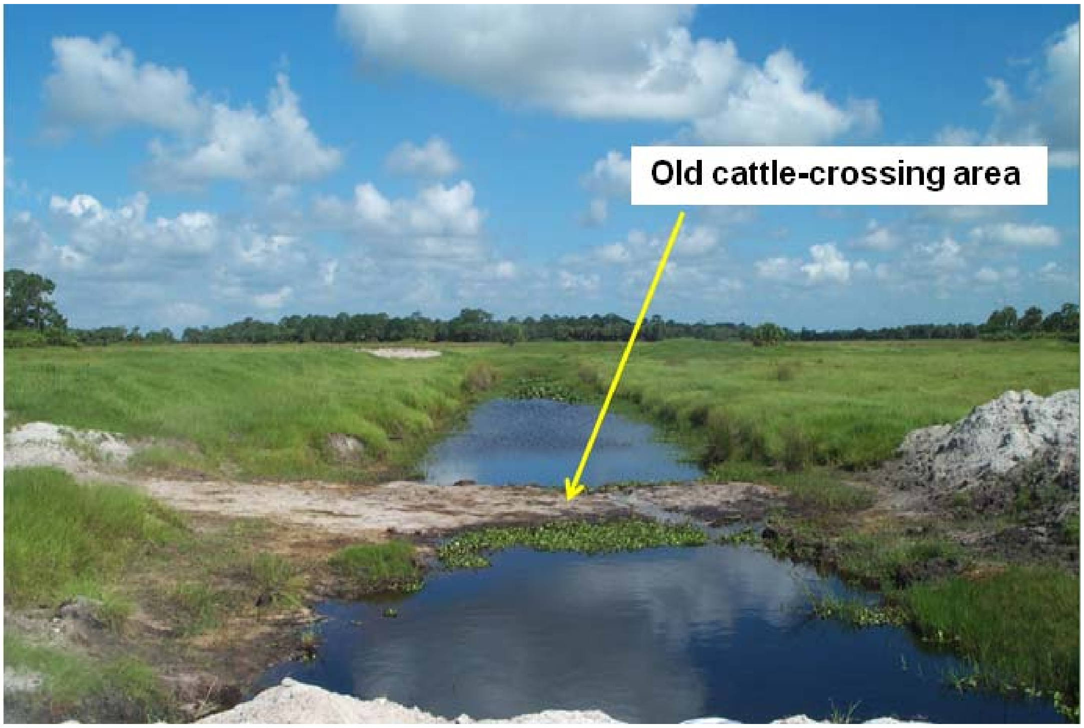

Figure 2. Ditch section showing the old cattle-crossing area during pre-BMP period.

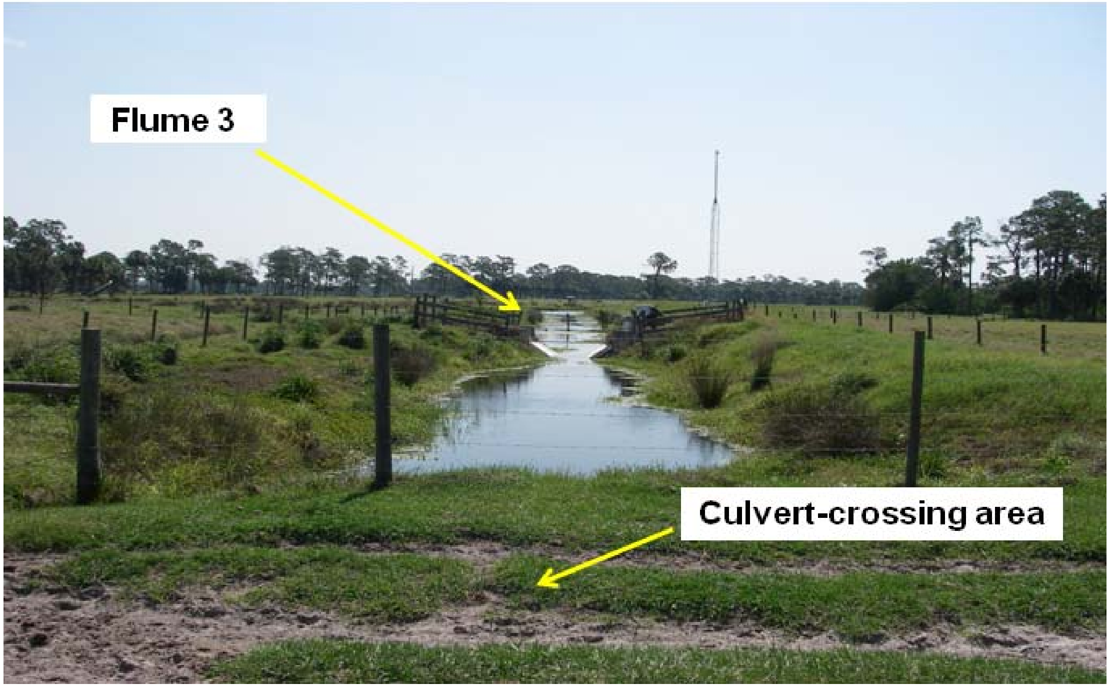

Figure 3. Ditch fencing and culvert crossing BMP site showing the culvert cattle-crossing area and the flume (Flume 3) at the outflow point.

2.3. Hydrologic and water quality monitoring

Each of the three flumes (F1, F2, and F3) was equipped with digital shaft encoders to measure the upstream and downstream heads at the flumes. Acoustic Doppler Velocimeters (ADV) were also installed in the ditches in August 2007 to measure the flow velocity. Flow rates were measured at an interval of 15 min (900 sec). To determine the load for each 15-minute interval, the flow rate (L/sec) was multiplied by the nutrient concentration (mg/L) and the duration (900 sec). To determine the total load for an entire monitoring period, the loads for all 15 minute period segments were summed together.

A Weather Station was installed at the ranch to monitor rainfall, temperature, solar radiation, wind speed, and relative humidity. Auto-samplers were installed near each flume to collect surface water samples using a combination of fixed time and head based sampling. The samples were sent to the Analytical Research Laboratory (ARL) at the University of Florida, Gainesville, FL, where they were analyzed for TP, TKN, NH_3-N, and NO_3-N. Table 1 shows the hydrologic and water quality parameters, sampling locations, methods and frequency for the study. The wet season (June-October) of 2005 was considered as the pre-BMP period. The wet seasons (June-October) of 2006, 2007, and 2008 were the three post-BMP periods (post-BMP1, post-BMP2, and post-BMP3, respectively) after the fencing at the BMP site was installed in early 2006.

Table 1. Hydrologic and water quality variables measured at the study site.

Data category	Parameter	Location	Measurement method	Sampling frequency
Hydrology and weather related	Upstream and downstream hydraulic heads at the flume	Sites 1, 2, ranch outlet	Shaft encoder connected to a datalogger	15 min
		Sites 1, 2, ranch	Manual gage	Weekly

		outlet		
	Rainfall	Site 1	Automatic rain gage connected to a datalogger	15 min
		Site 1	Manual gage	
	Rainfall, temperature, solar radiation, wind speed, and relative humidity	Weather station	Automatic (datalogger)	15 min
Surface water quality	NH_3-N, NO_3-N, TKN, TP	Sites 1, 2, ranch outlet	Automatic (Auto sampler)	Fixed time and flume head based
	NH_3-N, NO_3-N, TKN, TP	Sites 1, 2, ranch outlet	Manual (grab sample)	Monthly

2.4. Land use management data collection

Ranch management data were collected to evaluate the effects of pre and post-BMP ranch management activities on water quality. The data on ranch management included pasture type and improvement, fertilizer application rates, and animal (cattle) stocking rate. Equal amounts of NPK fertilizer (20-5-5) were applied at the three sites each year during the entire study period. There were more head of cattle present at the study site during post-BMP periods as compared to the pre-BMP period (Table 2). Therefore, the estimates of the reductions in P and N loadings at the BMP site, if observed, should be conservative.

Table 2. Animal stocking and fertilizer application rates for the study site.

Periods	NPK (20-5-5) application amount (kg/ha)*	No. of cattle
Pre-BMP (2005)	392	114
Post-BMP1 (2006)	392	167
Post BMP2 (2007)	392	138
Post BMP3 (2008)	392	138

*The fertilizer was applied in September of each year.

2.5. Determination of loadings at the BMP site by mass balance for P and N

The mass-balance for P and N loads for the ditch section for the pre-BMP period can be represented using the water discharge and chemical concentration data:

$$M_d = M_u + M_c + M_{gw} + M_p \qquad (1)$$

where M_d is the mass at the outflow point, M_u is the mass entering through inflow point, M_c is the contribution of P and N due to cattle traffic, and M_{gw} is the groundwater P and N addition. M_p is the addition or removal of P and N by physical/biological/chemical processes. Examples of these processes include retention and release of P and N from soil and aquatic vegetation. For the post-BMP period, the N and P contribution by the cattle traffic (the term M_c in Eq. 1) was eliminated.

The groundwater contribution of flow to the ditch section was estimated using the Dupuit's equation (Delleur, 2007). Estimated groundwater flow, and the P and N concentrations in the 10 wells near the ditch section were used to estimate the groundwater contribution of P and N to the ditch section. (Please refer to Appendix E for detailed materials and methods for the groundwater contribution study). Total Phosphorus, Dissolved Organic Nitrogen (DON), and TN loads were compared between inflow and outflow points to study the BMP effectiveness. Dissolved Organic Nitrogen was calculated as the difference between TKN and NH_3-N, and TN comprises TKN and NO_3-N.

2.6. Comparison of P and N concentrations for time-specific water samples for the inflow and outflow

Flows from F1 and F2 converge at inflow point which is about 15 m downstream from both the flumes. The actual concentrations of P and N species at the inflow point were dependent on the flow volumes and the mass of P and N contributed by the two flumes (F1 and F2) to the inflow point. To compare the concentrations of N and P species between the inflow and outflow points, water samples collected from the three flumes (F1, F2, and F3) at the same or similar sampling times (time-specific samples) were selected and the respective P and N concentrations were examined. The P and N concentrations and the corresponding flow volumes for F1 and F2 were used to calculate the total P and N mass contributed to the inflow point. Each pair of water samples collected from F1 and F2 at similar sampling times were used to determine the concentrations of P and N species for that sampling time for inflow. The combined mass of an individual P or N species from both flumes (F1 and F2) divided by the combined flow volumes from both flumes yielded the concentration of that particular species at the inflow point for the time when the water samples were collected. This concentration was compared to the corresponding concentration of that species at outflow for the similar sampling time. These samples were termed as ‘time-specific’ because it was crucial to select the water samples which were collected at same or similar times from numerous samples during the study period. On average, 25 such pairs of time-specific samples were selected for each year for comparison of inflow and outflow P and N concentrations.

2.7. Statistical analyses to study the effects of the BMP

Two types of statistical analyses were conducted to study the effectiveness of the BMP for P and N. First, analyses were conducted to compare the TP and TN loadings for the pre- and post-BMP periods using an Analysis of Covariance (ANCOVA) model. The loads (TP and TN) were calculated using the concentrations and the corresponding flow volumes. Point-wise comparisons of TP and TN concentrations between the inflow and outflow points for time-specific water samples were also made using paired t tests.

A heterogeneous slopes Analysis of Covariance (ANCOVA) was used to model the daily loading at F3 (outflow point) as a function of year, month, and total daily loadings at inflow point (i.e. the combined loadings at F1 and F2). There were 153 days (during June-Oct) monitored each year for a combined 612 days (during 2005-2008) but there were 171 days with no flow either at inflow or outflow flumes. The months of June and July in 2006 and June in 2008 had only 5 or fewer days when flow occurred and so these three months were excluded from the analysis. A heterogeneous slopes model was used because the test for equal slopes rejected (TP: $F_{16,373} = 45.23$, $p < 0.0001$; TN: $F_{16,373} = 19.5$, $p < 0.0001$). The observed standard deviations for the months that were retained in the model ranged from 0.0172 in October 2006 to 11.9141 in August of 2008 for daily TP loadings at F3 and from 0.0510 in October 2006 to 31.5420 in August of 2008 for daily TN loadings at F3. The variances of the residuals from the ANCOVA models tended to fall into three groupings of similar values for TP and four groupings for TN, and therefore, weighted ANCOVA models that also incorporated heterogeneous variances for the groups was used.

$$Y_{ij} = \beta_{0,ij} + \beta_{1,ij} X_{ij} + \varepsilon_{ij}, \qquad i = 2005, \dots, 2008; \; j = 6, 7, 8, 9, 10$$

$$Y_{ij} \; X_{ij} \; \beta_{0,ij} \; Y \; X \; \beta_{1,ij} \; \varepsilon_{ij} \; (ij)^{th}$$

For determining the effect of the BMP, pair-wise contrasts of the pre- and post-BMP mean monthly loadings were done for August, September, and October. In addition, the yearly average of the adjusted means over the three months (August, September, and October) in pre-BMP period (2005) was also compared to the yearly averages (for the same three months) in each of the subsequent three years (post-BMP periods).. The comparison of different months across years could be problematic since the result of the comparison depends strongly on the value of the loadings at F1 and F2 (X_{ij}) used to estimate the mean loading at F3 ($E[Y]_{ij} = \mu_{ij}$). It was decided to use the average value of X_{ij} since it was generally within the range of X_{ij} for any given month-year combination. X_{ij} For TP and TN, these means were 3.04 and 8.64 kg/day, respectively. To control for the multiplicity of contrasts, Fisher's protected LSD procedure was used since all tests were planned *a priori*. All analyses were done using SAS v9.2 (SAS Institute, Cary, NC).

2.8 Economic Analysis

Best management practices (BMP) studied as a part of this project were evaluated for capital costs of implementation and cost-effectiveness of nutrient removal. Expenses for construction of structural improvements included materials, labor, and contract services, as reported by co-investigators and the cooperating ranch owner. Capital costs were amortized over assumed useful life of 20 years at a 5 percent annual interest rate. Management activities during the 2004-06 period were tracked for 17 management units (pastures) ranging in size from 3.9 to 56 hectares, representing a total of 546 hectares. The management information was aggregated to reflect the experimental water basins monitored by the project. Production records were provided by the cooperator for animal stocking, forage fertilization, and supplemental feeding. Forages managed included Floralta, stargrass and bahiagrass. BMPs were assessed regarding their economic effectiveness for nutrient removal. Potential impacts of BMP's on cow herd health and performance and other non-market values were also assessed through interviews with ranch personnel.

3. Results and Discussion

3.1. Analyses of Phosphorus and Nitrogen loadings and concentrations

The groundwater contributions of P and N (Table 3) to the ditch section estimated using the Dupuit equation (Please refer to Appendix E for detailed results of the groundwater contribution study), and the loadings at the inflow and outflow points, were used in Eq. 1 to determine the loads contributed or removed at the BMP site.

Table 3. Phosphorus and Nitrogen contributed by groundwater to the ditch section at the BMP site.

Period	Dissolved Organic Nitrogen (kg)	Total Nitrogen (kg)	Total Phosphorus (kg)
Pre-BMP (2005)	6.06	9.66	0.95
Post-BMP1 (2006)	1.36	2.27	0.14
Post-BMP2 (2007)	0.92	1.43	0.19
Post-BMP3 (2008)	3.85	5.62	0.49

There were increases in TP and DON loadings at outflow as compared to those at inflow during pre-BMP period (Table 4). TP and DON loads increased by 35 and 3%, respectively at the outflow point. This indicated addition of TP (0.80 kg/day) and DON (0.19 kg/day) at the cattle crossing site due to the cattle traffic or other sources during the pre-BMP period. During post-BMP1 and post-BMP3 periods, P and N were removed at the BMP site. During the post-BMP periods, the cattle traffic through the ditch, a source of P and N, was eliminated which potentially reduced P and N loadings at outflow for the post-BMP1 and post-BMP3 periods. TP loads at outflow decreased by 32 and 11% during the post-BMP1 and post-BMP3 periods, respectively. For post-BMP2, N and P species were added within the BMP site. Post-BMP2 period experienced severe drought conditions in south Florida, resulting in drastically reduced flow. This reduced flow contributed small nutrient loadings. These small loadings and potential release of P and N from dead aquatic vegetation could have potentially masked the effects of the BMP during post-BMP2.

Table 4. Phosphorus and Nitrogen loadings for the pre- and post-BMP periods at the inflow and outflow sites.

Period	Dissolved Organic Nitrogen (kg)			Total Nitrogen (kg)			Total Phosphorus (kg)		
	Inflow	Outflow	% change	Inflow	Outflow	% change	Inflow	Outflow	% change

Pre-BMP	947.00	975.94	3.06	1485.00	1345.34	-9.40	350.00	473.05	35.16
Post-BMP1	79.88	73.61	-7.85	102.72	90.47	-11.93	54.31	37.00	-31.87
Post-BMP2	32.70	35.22	7.71	39.23	40.69	3.72	18.96	31.62	66.77
Post-BMP3	1513.00	1355.15	-10.43	1882.00	1671.38	-11.19	810.32	722.29	-10.85

During the pre-BMP period, there were increases in the flow-weighted concentrations of TP and DON at the outflow in comparison to those at inflow (Table 5), which was expected due to P and N contributions from the cattle traffic. For post-BMP1 and post-BMP3, there were decreases in P and N flow-weighted concentrations at the outflow. During post-BMP2, there were increases in P flow-weighted concentrations. Although flow-weighted TN concentrations at the outflow were always less in comparison to the inflow, these differences were on average higher during the post-BMP1 and post-BMP3 periods.

Table 5. Flow weighted concentrations (mg/L) of Nitrogen and Phosphorus for the pre- and post-BMP periods at the inflow and outflow sites.

Period	Dissolved Organic Nitrogen (kg)			Total Nitrogen (kg)			Total Phosphorus (kg)		
	Inflow	Outflow	% change	Inflow	Outflow	% change	Inflow	Outflow	% change
Pre-BMP	2.09	2.16	3.35	3.27	2.97	-9.17	0.77	1.04	35.06
Post-BMP1	2.03	1.91	-5.91	2.61	2.33	-10.73	1.38	0.95	-31.16
Post-BMP2	2.33	2.31	-0.86	2.80	2.65	-5.36	1.39	2.24	61.15
Post-BMP3	2.99	2.54	-15.05	3.72	3.14	-15.59	1.52	1.35	-11.18

The release of P and N at the BMP site during post-BMP2 may be explained in the context of the macrophytes present in the ditches. Phosphorus and N are released to water when the macrophytes decompose (Ogwada, 1984; Chimney and Pietro, 2006). Aquatic macrophytes present in the ditches and wetlands of the Okeechobee Basin function as temporary storage units for P and N, that upon decomposition release nutrients to the water (Reddy et al., 1995; Reddy et al., 1999).

There was high variability in rainfall for the monitoring periods (June-October: 2005-08) at the study site. Pre-BMP and Post-BMP3 had higher rainfall (1116 and 1124 mm, respectively) than the post-BMP1 and post-BMP2 periods (633 and 755 mm, respectively). Even though rainfall amounts were similar for post-BMP1 and post-BMP2, the unusually wet conditions due to Hurricane Wilma during October, 2005 extending to 2006 generated higher flow at the sites in 2006 (post-BMP1) in comparison to that during 2007 (post-BMP2). Post-BMP2 received considerably less rainfall in south Florida with almost no flow from November 2006 to June 2007 (seven months prior to post-BMP2 period). Lake Okeechobee experienced the lowest water level since 1931 in July 2007 (SWFWMD, 2008). The dry condition resulted in low moisture content in the ditch soil and decomposition of aquatic vegetation. This resulted in the release of P and N, and subsequent increase in P and N loadings at the outflow. The additions of TP (12.66 kg), DON (2.52 kg), and TN (1.46 kg) within the BMP site in 2007 can mostly be attributed to decomposition and subsequent release of P and N from macrophytes. The small loads generated upstream from the BMP site due to low flow and the relatively high contribution of P and N within the BMP site by the macrophytes potentially masked any reductions in P and N from the exclusion of cattle from the ditch.

During pre-BMP, post-BMP1, and post-BMP3 periods, there were reductions in N loadings at outflow as compared to inflow. The in-ditch losses/uptake of N has also been indicated by Reddy et al. (1999). Reddy et al. (1995) found that a significant portion of N added to wetlands and streams/ditches was lost from the wetland-stream systems. The climate of south Florida is conducive to denitrification with its high temperature and saturated soil due to high water table conditions during summer (Seitzinger, 1988; Gale et al., 1993; Martin and Reddy; 1997). Gordon et al. (1986) in their study in the wetlands in the Florida Everglades found that the highest denitrifification occurred between temperatures of 26 to 32 ^{o}C. The study site experienced this temperature range for most of the time during daytime for the study period.

The mean daily temperature during June-October for 2005-08 at the study site was 25.8 ^{o}C. This might explain the disappearance of considerable N during the pre-BMP period and two post-BMP periods (2006 and 2008). The unsaturated conditions during post-BMP2 (2007) may have reduced the denitrification considerably.

To compare the mean P and N concentrations between inflow and outflow points, time-specific samples were selected from the three flumes during the wet period (June-October) for each year. Mean TP concentration for the time-specific samples at outflow was higher than that at inflow during pre-BMP period (Table 6). Time-specific samples enabled better comparison of inflow and outflow P and N concentrations based on pairs of concentration values from inflow and outflow points. For Post-BMP1 and post-BMP3 periods, mean TP concentrations for the time-specific samples at outflow were lower than those at inflow point indicating reduction in P concentrations at the BMP site. During post-BMP2, mean TP concentration for the time-specific samples at outflow was higher than that at inflow likely due to the contribution by the macrophytes.

There were increases in DON and TN concentrations at outflow in the time-specific samples for all the periods except post-BMP3 even though flow-weighted concentrations of DON and TN were lower at outflow during the three post-BMP periods. Evaluation of the effect of the BMP on N loading and concentration is more complex than that for P loading and concentration due to the complex nature of nitrogen dynamics and processes combined with weather and hydrologic variability (e.g. rainfall, flow, and soil moisture).

Table 6. Mean (Arithmetic) Phosphorus and Nitrogen concentrations for the time-specific samples for the pre-BMP and post-BMP periods.

Period	Dissolved Organic Nitrogen (kg)			Total Nitrogen (kg)			Total Phosphorus (kg)		
	Inflow	Outflow	% change	Inflow	Outflow	% change	Inflow	Outflow	% change
Pre-BMP	2.96	3.30	11.49	3.50	3.55	1.43	0.54	0.66	22.22
Post-BMP1	1.98	2.42	22.22	3.01	3.25	7.97	2.28	1.77	-22.37
Post-BMP2	3.20	3.55	10.94	3.88	4.01	3.35	1.51	1.86	23.18
Post-BMP3	3.56	3.29	-7.58	4.24	3.85	-9.20	1.69	1.56	-7.69

3.2. Statistical analyses for the effectiveness of the BMP

3.2.1. Statistical analyses for P and N loadings using ANCOVA model

3.2.1.1. Total Phosphorus loadings

The estimated annual mean daily TP loadings averaged over the three months (August, September, and October) at outflow (F3) ranged from 1.46 kg in 2006 to a high of 3.24 kg in 2008. Tests comparing the annual mean daily loading over the three wet months of 2005 versus the same three months in post-BMP years failed to reject the null hypothesis of no difference. The pattern of the differences was such that for 2006 and 2007, the mean daily loading was smaller than that in 2005 but the difference was small relative to the amount of variability that was observed in the data and so the results were not significant.

Comparisons of loadings for each month (August, September, and October) in 2005 versus the same month in subsequent years indicated that there were some differences between individual months (Table 7). The mean daily loadings of TP at outflow (F3) adjusted to the mean loading at inflow (combined loadings at F1 and F2) differed significantly between 2005 and 2008 for the months of August (t_{363} = -3.64, p = 0.0003) and October (t_{363} = 5.72, p = < 0.0001) and between 2005 and 2007 for the month of October (t_{363} = 2.17, p = 0.0306). The loading was higher in August, 2008 (post-BMP) than in August, 2005 (Pre-BMP) but for the other two significant tests (between October in 2005 and 2007, and between October in 2005 and 2008), the loadings were lower during post-BMP than pre-BMP period (Table 7).

Table 7. Comparisons of mean daily loadings of Total Phosphorus for a particular month in 2005 versus the same month in subsequent years.

Month	Comparison between	Estimated difference	Standard error	t value	Pr > |t|
August	2005 and 2006	-1.23	0.67	-1.83	0.0687
August	2005 and 2007	-3.81	2.35	-1.62	0.1059
August	2005 and 2008	-6.97	1.92	-3.64	0.0003
September	2005 and 2006	-0.21	3.34	-0.06	0.9488
September	2005 and 2007	3.53	5.64	0.63	0.5323
September	2005 and 2008	1.31	3.32	0.40	0.6926
October	2005 and 2006	4.07	5.08	0.80	0.4244
October	2005 and 2007	2.44	1.12	2.17	0.0306
October	2005 and 2008	2.92	0.51	5.72	<0.0001

3.2.1.2. Total Nitrogen loadings

The estimated mean daily TN loadings averaged over the three months at outflow (F3) for each year ranged from 2.82 kg in 2006 to a high of 10.83 kg in 2008. Tests comparing the annual mean daily loading over the three wet months of 2005 versus the same three months in post-BMP years failed to reject the null hypothesis of no difference. The pattern of the differences was such that for 2006 and 2007, the mean daily loading was smaller than that in 2005 but the difference was small relative to the amount of variability that was observed in the data and so the results were not significant. Comparison of TN loadings for a particular month in 2005 (Pre-BMP) versus the same month in subsequent years (post-BMP) indicated that there were some differences between individual months (Table 8). For August, the three post-BMP years had higher mean daily loadings of TN and the differences were statistically significant for 2006 ($t_{363} = -3.04$, $p = 0.0025$) and 2008 ($t_{363} = -3.07$, $p = 0.0023$). There were large drops in the mean daily loadings of TN at outflow for October between 2005 (pre-BMP) and subsequent years (post-BMP), and the difference was significant for 2007 ($t_{363} = 3.52$, $p = 0.0005$) and 2008 ($t_{363} = 2.41$, $p = 0.0163$).

Table 8. Comparisons of mean daily loadings of Total Nitrogen for a particular month in 2005 versus the same month in subsequent years.

Month	Comparison between	Estimated difference	Standard error	t value	Pr > |t|
August	2005 and 2006	-3.12	1.03	-3.04	0.0025
August	2005 and 2007	-4.17	6.36	-0.65	0.5132
August	2005 and 2008	-17.81	5.80	-3.07	0.0023
September	2005 and 2006	0.07	1.77	0.04	0.9693
September	2005 and 2007	5.12	6.74	0.76	0.4482
September	2005 and 2008	0.70	1.73	0.41	0.6856
October	2005 and 2006	15.37	18.00	0.85	0.3937
October	2005 and 2007	12.02	3.44	3.50	0.0005
October	2005 and 2008	7.85	3.25	2.41	0.0163

The results from the ANCOVA model suggested that the loadings of TP at outflow were high for October, 2005 and August, 2008 and low for all the three months in 2006 and 2007. During October 2005 and August 2008, the site received heavy rainfall due to Hurricane Wilma and Tropical Storm Fay, respectively resulting in high loadings. This is indicative of the fact that high flow generated high P and N loadings at the study site. The ANCOVA model for TN suggested uncertainty in N loadings for the pre- and post-BMP periods. This might be due to the combined effects of denitrification, uptake/release of N by macrophytes, and flow variability for the study periods.

Therefore, statistical analyses based on the P and N concentrations for the time-specific samples were conducted to determine if there was a change (increase or decrease) in mean P and N concentrations at outflow relative to inflow at specific times.

3.2.2. Nitrogen and Phosphorus concentrations

The results of the paired t test for the time-specific samples indicated that there was a significant increase in the mean TP concentration at outflow in comparison to that at inflow during pre-BMP period (t = 2.45, p = 0.0261). There were significant decreases in mean TP concentrations at outflow compared to those at inflow point during post-BMP1 (t = -3.51, p = 0.0014) and post-BMP3 (t = -2.19, p = 0.0379) periods. For post-BMP2, there was a significant increase in TP concentrations at outflow point as compared to that at inflow point (t = 2.47, p = 0.0379). The results based on P concentrations from the paired t test were consistent with the results from the P loading analyses for the pre- and post-BMP periods (Table 3).

There were increases in mean TN concentrations at outflow during the pre-BMP (t = 0.09, p = 0.9276), post-BMP1 (t = 1.06, p = 0.2959), post-BMP2 (t = 0.43, p = 0.6700) periods, and the increases were not significant. There was a decrease in mean concentration of TN during post-BMP3 period (t = -0.92, p = 0.3661), and the decrease was not significant. Given the variability in flow and rainfall, and due to denitrification and uptake/release of N by the macrophytes, the effects of the BMP on N might have been masked.

3.3. Scenarios of post-BMP P and N reductions

Due to the variability in rainfall and flow, P and N loadings at inflow and outflow points varied greatly for the three post-BMP periods. Post-BMP1 had moderate flow and nutrient (P and N) loadings and there were some reductions of P and N at the BMP site during that period. Flow and loadings were the lowest during post-BMP2, and nutrients (P and N) were added at the BMP site during that period. Rainfall and flow for post-BMP3 were comparable to those during the pre-BMP period. Considering these variability, three scenarios for P and N reductions during the post-BMP periods can be proposed: conservative, moderate and liberal. The conservative estimates of P and N reductions would be to consider the average P and N reductions for all the three post-BMP periods. The moderate estimates would be to eliminate post-BMP2, and consider the average loadings for only post-BMP1 and post-BMP3. The liberal estimates of P and N reductions would be to consider only post-BMP3 when the reductions of P and N were the greatest. According to the conservative estimate, the upstream and downstream TP loadings were 294.53 and 263.64 kg, respectively indicating a reduction of 30.89 kg (10% reduction). Upstream and downstream TN loadings from the BMP site were 674.65 and 600.85 kg indicating a reduction of 73.80 kg (11% reduction) of TN at the BMP site. The conservative, moderate, and liberal estimates of TP reductions during the post-BMP periods were 0.20, 0.34, and 0.57 kg/day, respectively (Table 9). The conservative, moderate, and liberal estimates of TN reductions during the post-BMP periods were 0.48, 0.73, and 1.38 kg/day, respectively.

Table 9. Conservative, moderate, and liberal estimates of P and N reductions for post-BMP periods.

Estimate	Dissolved Organic Nitrogen (kg)			Total Nitrogen (kg)			Total Phosphorus (kg)		
	Inflow	Outflow	Reduc*	Inflow	Outflow	Reduc*	Inflow	Outflow	Reduc*
Conservative	541.86	487.99	-53.87	674.65	600.85	-73.80	294.53	263.64	-30.89
Moderate	796.44	714.38	-82.06	992.36	880.93	-111.44	432.32	379.65	-52.67
Liberal	1513.00	1355.15	-157.85	1882.00	1671.38	-210.62	810.32	722.29	-88.03

*reduction

The above analyses suggested that there were increases in TP loads and concentrations at outflow point as compared to those at inflow point during pre-BMP period, and there were decreases in TP loadings and concentrations during post-BMP1 and post-BMP3. The dry conditions during post-BMP2 might have added P at the BMP site resulting in the increase in TP loads and concentrations at outflow point. Due to the possible effects of denitrification and uptake/release of N by the macrophytes, the effects of the BMP on N might have been masked considerably.

The results indicating the reductions of P at the ditch section during the post-BMP periods were consistent based on the analyses of P loadings, flow-weighted concentrations, and mean concentrations (for time-specific samples). This might be indicative of the fact that the BMP was effective in reducing P at outflow point. The effects of the BMP on N might have been masked due to combined effects of denitrification, macrophytal activity, and variability of rainfall and flow.

Many of the cattle exclusion studies in the past had high uncertainty associated with BMP effectiveness that were caused by the variability in one or more factors: rainfall, flow, seasons of the year considered for pre- and post-BMP periods, fertilizer application, and livestock stocking rate, etc. between pre- and post-BMP periods. This variability may introduce bias in the conclusion about the effectiveness of the BMPs. Some of these factors (e.g. differences in rainfall, flow, and number of storms) are beyond human control. But there are some factors (e.g. seasons of the year used for pre- and post-BMP periods, fertilizer application, livestock stocking rate) that may be controlled. It is desirable that there is minimum variability in these factors between pre- and post-BMP periods to minimize the masking effects of these variables on the BMP results. In this study, the fertilizer application rates were the same for pre- and post-BMP periods. Same period (June-October) of the year was used for pre- and post-BMP periods. Furthermore, pre-BMP1 and post-BMP3 had similar rainfall and flow volumes reducing the masking effects of the external factors. The animal stocking rates during post-BMP periods were higher than that during pre-BMP period making the BMP results conservative. Even though post-BMP1 and post-BMP2 periods experienced lower rainfall, wet conditions prior to post-BMP1 resulted in higher flow during post-BMP1 than post-BMP2.

3.4. Economic Analysis of BMP

Structural improvements made at the study site in support of the BMPs included installation of fencing, gates and culvert crossings to exclude cattle from the natural water bodies, along with provision of aboveground water tanks supplied from wells for watering animals. Capital costs totaled $20,245 for the waterway exclusion BMP, and amortized annual cost of structural improvements were $1,625. Capital costs per hectare treated were $76.6, and average annual capital costs per animal unit averaged of $10.2, which represented 3.9% of the annual feed and forage improvement expenses per animal. These annual expenses represented a similar share (3.6%) of the southeast U.S. region average annual cost per cow ($282), as reported by Cattle Fax (2006). Based on the amortized cost per year, the average unit cost of nutrient removal ranged from $3.23 to $9.23 per kg of nitrogen and $$7.74 to $22.05 per kg of phosphorous, under the liberal and conservate estimates of nutrient removal rates, respectively (Table 10).

Table 10. Nutrient load reduction and unit cost of removal for waterway exclusion BMP during the post-treatment period (2006-08).

Estimate	Dissolved Organic Nitrogen		Total Nitrogen		Total Phosphorous	
	Average Nutrient Removal Rate (kg/day)	Average Nutrient Removal Cost ($/kg)	Average Nutrient Removal Rate (kg/day)	Average Nutrient Removal Cost ($/kg)	Average Nutrient Removal Rate (kg/day)	Average Nutrient Removal Cost ($/kg)

Conservative	0.352	12.64	0.482	9.23	0.202	22.05
Moderate	0.536	8.30	0.728	6.11	0.344	12.93
Liberal	1.032	4.32	1.377	3.23	0.575	7.74

Data reflects June-October monitoring period (153 days).

Amortized capital cost of stream fencing improvements was $1,625 per year (for 20 years @ 5% interest).

4. Summary and Conclusion

Performance of DFCC BMP was evaluated at a cattle ranch by comparing P and N concentrations and loadings at inflow and outflow points for the BMP site during pre- and post-BMP periods. This study indicated that the BMP was effective in reducing P loads. During pre-BMP period, TP load increased by 35% at outflow and decreased by 32 and 11%, respectively during post-BMP1 and post-BMP3, in comparison to inflow. Possible loss of N by denitrification and N uptake/release by macrophytes might have masked the effects of the BMP on N. Post-BMP2 period experienced severe dry conditions resulting in decomposition of macrophytes and subsequent release of N and P to the BMP site. This masked the effects of the BMP during post-BMP2 period. Overall, this study shows that the BMP was effective in reducing the P loading from BMP site at the cow-calf ranch.

Economic analyses of DFCC BMP indicated that the DFCC BMP has rather modest capital costs when amortized over an expected useful life of 20 years, representing only 3 percent of total operating costs per animal in the region. Therefore, it is expected that this BMP would not be a financial burden for the ranch owner to implement. This BMP was also shown to be an effective and low cost method for surface water nutrient reduction. For phosphorous, the nutrient of critical importance in the Lake Okeechobee/Everglades water basin, conservative estimates of costs for P removal averaged $22.05 per kg. This cost is less than storm water treatment areas ($442 to $1,109 per kg) as reported by Sano et al (2005).Ranch managers indicated that the BMPs resulted in no changes in ranch operations management, overhead expenses, or general herd health. The view was expressed that excluding cattle from natural streams and providing water supply in above-ground tanks improved the quality of drinking water, although a definite value could not be placed on this benefit.

One of the uncertainties associated with the quantificaiotn of this BMP was the use of one pre-BMP period data with three post-BMP period data. The monitoring at this site is continuing to better quantify the effectiveness of this BMP.

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APPENDIX D

Effects of Wetland Water Retention Best Management Practice (BMP) on Nitrogen and Phosphorus Transport from a Beef Cattle Ranch in the Lake Okeechobee Basin

Introduction

A project was started in 2004 to evaluate the wetland water retention (WWR) best management practice (BMP) in the Lake Okeechobee basin. Two wetland sites (wetlands 1 and 4, Figure 1) within a beef cattle ranch were selected for evaluating WWR BMP. The areas of the two wetland sites are 81.7 and 24.4 ha, respectively.

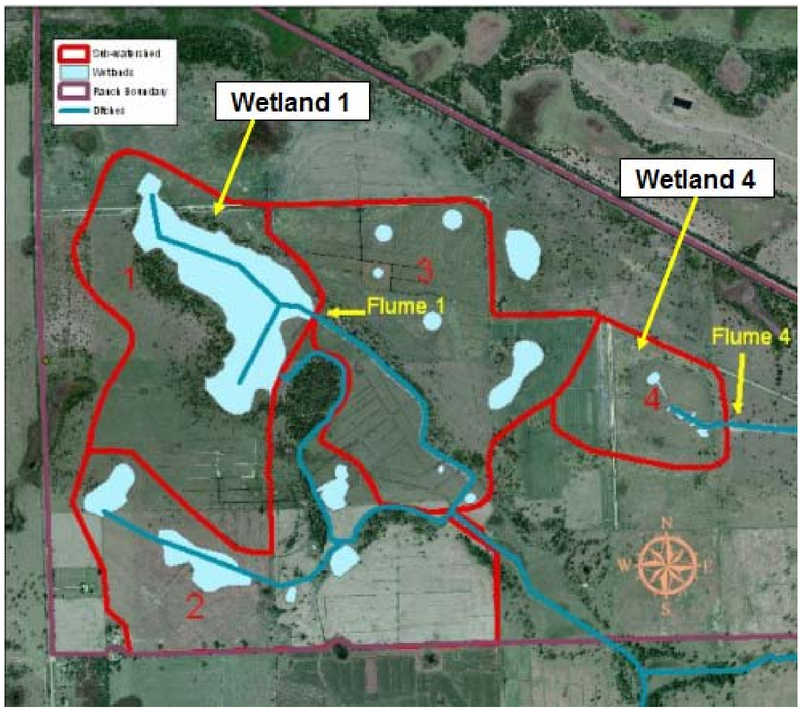

Figure 1. Wetland sites at Pelaez ranch.

Wetland 1 contains two drainage ditches. The two ditch segments are 669 and 332 m in length and converge and flow together for 147 m before exiting the wetland. Soil types within wetland 1 are Myakka (7.5 ha), Immokalee (40.1 ha), Basinger (21.5 ha), Floridana (6.0 ha), and Valkaria (5.0 ha). Wetland 4 contains a drainage ditch which flows in south-easterly direction for about 282 m before exiting the ranch. The soil types in wetland 4 include Basinger (13.4 ha), and Immokalee (11.0 ha). The land use within the wetland drainage areas includes improved pastures and forested areas. The two major forage types within the drainage areas of the two wetlands are Bahia and Floralta.

Materials and Methods

The WWR BMP entails the installation of a culvert at the wetland outlet equipped with a flash-board riser at the invert of the culvert (Figure 2). The flashboard riser is used to allow water to flow only when the water level exceeds the top elevation of the flashboard. The goal of this BMP is to retain more water upstream of the flashboard riser keeping the wetland area hydrated for a longer period of time.

Figure 2. Wetland water retention BMP (flashboard riser) at wetland 4.

Trapezoidal flumes fitted with pressure transducers were installed downstream of the flashboard risers to measure the flow. Since 2007, an Acoustic Doppler Velocimeter (ADV) was installed to monitor the flows. A monitoring well was installed near each culvert to measure the surface water level at the culvert using a pressure transducer. Auto-samplers were installed at the flume sites to collect water samples that were analyzed for Total Phosphorus (TP), Total Kjeldahl Nitrogen (TKN), Ammonia (NH_3-N), and Nitrate (NO_3-N).

A pre- and post-BMP monitoring design was used to evaluate the effectiveness of the WWR BMP. By comparing the nutrient (N and P) loading downstream from the flashboard riser for pre- and post-BMP periods, the effectiveness of the BMP can be evaluated. The pre-and post-BMP periods for the two wetlands were different and are shown in Table 1.

Table 1. Pre and post-BMP periods for the wetland water retention BMP study.

Location	Pre-BMP period	Post-BMP period
Wetland 1	June 1, 2005 – May 31, 2006 (Pre-BMP1) June 1, 2006 – May 31, 2007 (Pre-BMP2)	June 1, 2007 –May 31, 2008 (Post-BMP1) June 1, 2008 – May 31, 2009 (Post-BMP2)
Wetland 4	June 1, 2005 – May 31, 2006 (Pre-BMP1)	June 1, 2006 – May 31, 2007 (Post-BMP1) June 1, 2007 – May 31, 2008 (Post-BMP2) June 1, 2008 – May 31, 2009 (Post-BMP3)

Results and Discussions

Nutrient Loadings and Concentrations Analyses

The rainfall and runoff depths for the two wetland sites have been presented in Tables 2 and 3, respectively.

Table 2. Rainfall and runoff depths for the pre- and post-BMP periods for wetland 1.

Period	Rainfall (cm)	Runoff depth (cm)	Runoff depth per unit rainfall
Pre-BMP1	147.3	15.7	0.107
Pre-BMP2	91.4	0.5	0.006
Post-BMP1	109.2	0.3	0.002
Post-BMP2	157.5	23.9	0.152

Table 3. Rainfall and runoff depths for the pre- and post-BMP periods for wetland 4.

Period	Rainfall (cm)	Runoff depth (cm)	Runoff depth per unit rainfall
Pre-BMP	147.32	44.2	0.300
Post-BMP1	91.44	1.8	0.019
Post-BMP2	109.22	1.8	0.017
Post-BMP3	157.48	42.4	0.269

A large variability in rainfall amounts was observed during the entire monitoring period (June 2005 – May 2009). Years 2006 and 2007 had below average rainfall in South Florida resulting in low flow volumes from the wetland sites. Years 2005 and 2008 received similar amounts ofrainfall; therefore, comparing these periods (corresponding to pre-BMP1 and post-BMP2 for wetland 1, and pre-BMP and post-BMP3 for wetland 4) provides a better assessment of the BMP's effectiveness.

Using the periods mentioned above for wetland1, there were increases in N and P loads during post-BMP2 compared to pre-BMP1 (Table 4). Mean and flow-weighted N and P concentrations also increased during post-BMP2 compared to pre-BMP1 (Table 4). Runoff depth was also high during post-BMP2 (Table 2). Based on the data for these two periods, it appears that the BMP was not effective in improving water quality at wetland 1.

Table 4. Nitrogen and Phosphorus loads and concentrations for the pre- and post-BMP periods at wetland 1.

Period	Loads (kg)		Mean Conc. (mg/L)		Flow-weighted Conc. (mg/L)	
	Total Nitrogen	Total Phosphorus	Total Nitrogen	Total Phosphorus	Total Nitrogen	Total Phosphorus
Pre-BMP1	312.0	88.80	2.93	0.59	2.48	0.70
Pre-BMP2	11.4	5.10	2.57	1.00	2.2	0.99
Post-BMP1	5.8	2.00	3.41	1.25	3.51	1.18
Post-BMP2	602.8	183.40	4.32	1.38	3.15	0.96

Using the same comparison periods for wetland 4loadings of TN and TP were less for post-BMP3 compared to the pre-BMP (Table 5). This indicates the likely effectiveness of WWR BMP in reducing nutrient loadings. The flow weighted concentrations of N and P were also less for post-BMP3 compared to pre-BMP (Table 5).

Table 5. Nitrogen and Phosphorus loads and concentrations for the pre- and post-BMP periods at wetland 4.

Period	Loads (kg)		Mean conc. (mg/L)		Flow-weighted conc. (mg/L)	
	TN	TP	TN	TP	TN	TP
Pre-BMP	319.3	182.4	3.78	1.53	2.96	1.69
Post-BMP1	7.2	5.1	2.02	1.16	1.56	1.11
Post-BMP2	51.9	22.6	8.00	3.94	11.35	4.95
Post-BMP3	233.2	142.3	3.58	1.82	2.26	1.38

Groundwater Concentration Analyses

Groundwater samples were collected once a month from the groundwater wells installed in the upland areas surrounding the wetland perimeter. The average concentrations of the N and P species for all the wells in the respective wetlands were calculated. For wetland 1, average groundwater TP concentration for post-BMP periods (506 μg/L) were 319% higher than that during pre-BMP periods (121 μg/L) (Table 6). Higher groundwater TP concentrations during post-BMP periods may be due to the drought conditions experienced during 2006 and 2007. Drought conditions resulted in lower water table depths creating a 'dilution effect', i.e., a similar mass of nutrients in a reduced volume of groundwater resulting in higher nutrient concentrations. The drought also created suitable conditions for the mineralization of soil and plant P. This mineralized P was then transported to the groundwater via infiltration during post-drought rainfall events

Table 6. Average groundwater TP concentrations (μg/L) during pre- and post-BMP periods for wetland 1.

Wells	Pre-BMP period	Post-BMP period
	Average TP conc. (μg/L)	Average TP conc. (μg/L)
Well 21	33	221
Well 23	172	445
Well 24	205	1207
Well 25	71	151
Average conc.	121	506

For wetland 4, the average groundwater TP concentration for the entire post-BMP period (28 μg/L) was 20%less than that for pre-BMP period (35 μg/L) (Table 7). Even though a detailed wetland vegetation study has not been undertaken, basic observations and aerial photography show that wetland 4 has less wetland vegetation than wetland 1. Wetland 4 is principally populated by drought-tolerant pasture species with small sections of wetland vegetation bordering the central drainage ditch. In contrast, wetland 1 is dominated by wetland species that produce large amounts of organic matter prone to mineralization during periods of drought. During the 2006-2007 drought, less P mineralization took place due to the factors described above resulting in less available and mobile P and N. These differences in groundwater quality between the two wetlands provide a potential explanation for the disparity in BMP effectiveness between the two wetlands.

Table 7. Average groundwater TP concentrations (μg/L) during pre- and post-BMP periods for wetland 4.

Wells	Pre-BMP period	Post-BMP period
	Average TP conc. (μg/L)	Average TP conc. (μg/L)
Well 11	40	16
Well 13	31	19
Well 15	17	9
Well 17	27	15
Well 19	59	83
Average conc.	35	28

Soil Phosphorus Dynamics

To better understand the P contributions from the wetlands, soil samples were collected to determine the Soil Phosphorus Storage Capacity (SPSC). A positive SPSC (Soil Phosphorus Storage Capacity) indicates a P sink and a negative SPSC indicates a P source. Both sinks and sources of P were observed at the two wetlands based on the SPSC values. At both sites, there are some locations in which the SPSC values were very low indicating ‘hotspots’ for P. Those areas are prone to release P if inundated. Figures 3 and 4 show the soil sampling locations and the corresponding SPSC values for wetland sites 1 and 4, respectively.

Figure 3. Soil sampling and groundwater well locations at wetland 1. (Average groundwater phosphorus concentration. (µg/L in green) and Soil Phosphorus Storage Capacity (SPSC in orange) values are shown).

Figure 4. Soil sampling and groundwater well locations at wetland 4 (Average phosphorus concentration (μg/L in green) and Soil Phosphorus Storage Capacity (SPSC in orange) values are shown).

For the entire monitoring period, the highest water levels (Tables 8 and 9) for each year were used to determine the inundated areas at both wetland sites using Light Detection and Ranging (LIDAR) topographic data in a Geographic Information System (GIS). It was observed that for wetland 1, the hotspot area was inundated only once during the entire monitoring period (Figure 5). This hotspot inundation took place during tropical storm Fay (20 August, 2008) during which 50% of the annual P loadings for the post-BMP2 period (2008) occurred. For wetland 4, the P hotspot was never inundated during the monitoring period (Figure 6). This might further explain the high P loadings and concentrations during post-BMP2 at wetland 1 and the lower loadings and concentrations measured at wetland 4.

Table 8. Highest water levels during the monitoring years at wetland 1.

Period	Highest water level (m)	Date
Pre-BMP 1	9.56	26 October, 2005
Pre-BMP2	9.30	30 August, 2006
Post-BMP1	8.96	7 October, 2007
Post-BMP2	10.18	20 August, 2008

Table 9. Highest water levels during the monitoring years at wetland 4.

Period	Highest water level (m)	Date
Pre-BMP	9.79	24 October, 2005
Post-BMP1	9.49	26 August, 2006
Post-BMP2	9.58	2 October, 2007
Post-BMP3	10.28	20 August, 2008

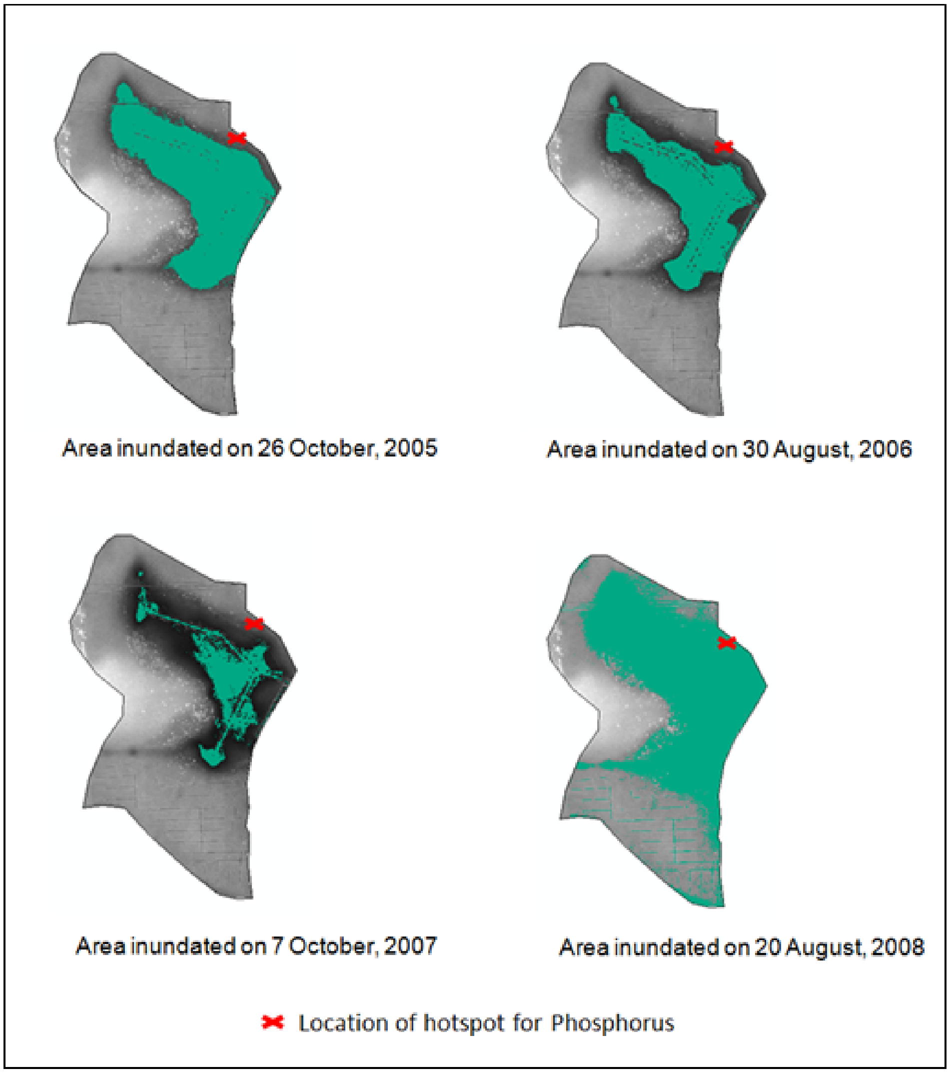

Figure 5. Area inundated during the highest water level for each year at wetland 1.

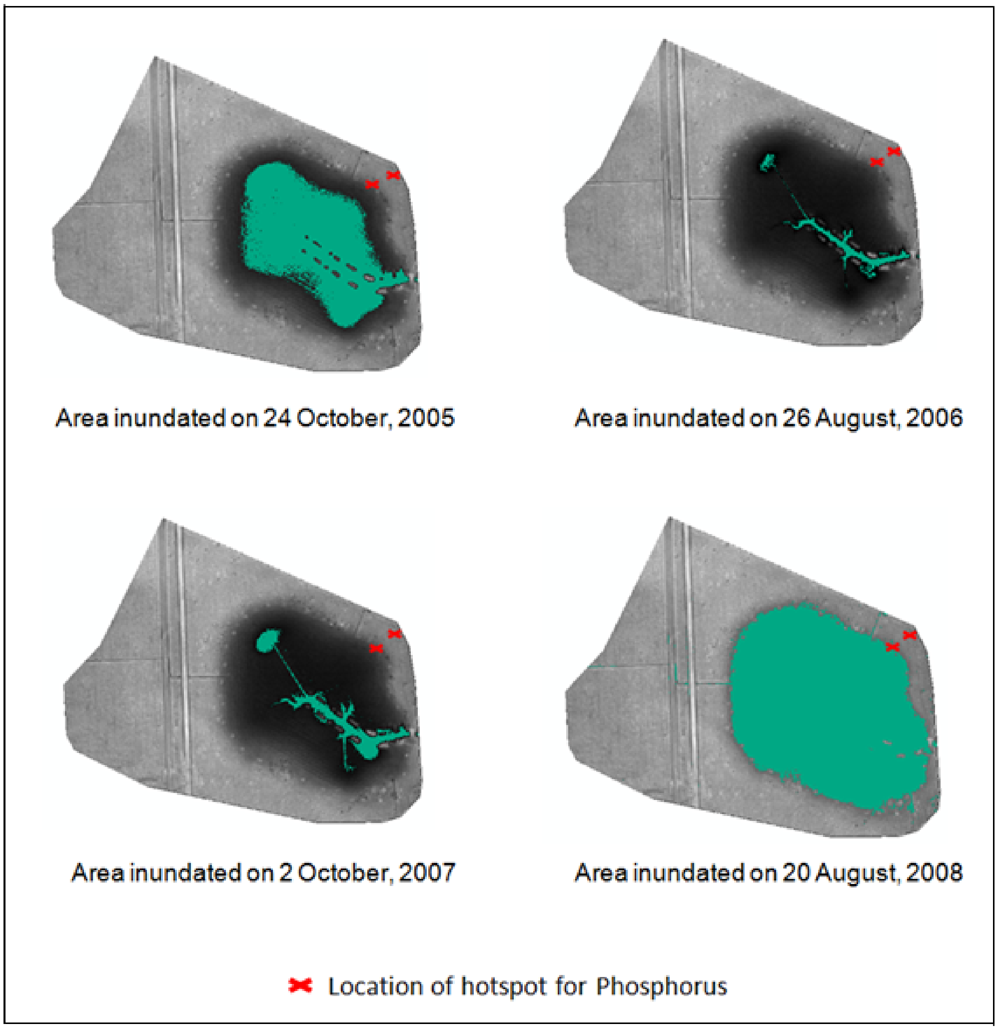

Figure 6. Area inundated during the highest water level for each year at wetland 4.

Hydrologically Active Areas for Implementation of the BMP

Hydrologically active areas within a ranch could be a result of several factors such as topography, soil type and location. The spatial variability in the soil characteristics (e.g., presence or absence of low conductivity subsurface layers such as spodic layer, thickness of the spodic layer, etc.) can result in differences in the water table rise in response to rainfall. Differential water table response in different areas within a ranch can result in spatially variable ponding depth and runoff volume. Variations in topography and drainage network within the ranch can also result in spatially variable flows (runoff and subsurface drainage) from different areas within the ranch. Groundwater depths (2005-2008) monitored at the three wells at the two wetland sites were analyzed to evaluate the presence of differences in hydrologic response in different areas of the ranch. Two of the wetland 1 wells (Wells 21 and 23) were at the border of the wetland while one well (Well 24) was in an upland area (Figure 7). At wetland 4, three deep wells (Wells 11, 15, and 19, Figure 8) were in the border areas of the wetland. To evaluate the variability in the differential hydrologic response, the water table depth data for several storm events that resulted in water table reaching the ground surface were analyzed.

<u>Wetland 1</u>

At wetland 1 during Hurricane Wilma (October, 2005), water level at Well 24 reached above ground level and this condition continued for two days after which the water level receded to below ground level. This means that the water flowed over land as runoff from the area around and near Well 24. In the other two wells (Wells 21 and 23), the water levels were below the land surface.

Tropical storm Fay (August, 2008) was another large storm which resulted in high flows in the Okeechobee basin. For this storm, water level in Well 21 reached above ground level and this condition continued for 3 days while in other two wells (Wells 23 and 24) the water level was below the ground level. Overall, areas near Wells 21 and 24 seem to be more hydrologically active than Well 23. The water table depth data for additional wells are currently being examined and will be included in the final project report.

<u>Wetland 4</u>

During Hurricane Wilma (October, 2005), the water levels at all the three wells (Wells 11, 15, and 19) were higher than the ground level even though the levels of inundation were different at each well. Well 11 experienced the highest ponding depth followed by wells 19 and 15. In contrast to wetland 1, water levels at all wells reached the ground surface level at the same time. The water level at Well 15 remained above ground surface level for only 5 hr 30 min. This analysis shows that the runoff generation potential at Well 11 area is higher than the other two areas. Figure 9 shows the mean daily water table depths at wetland site 4 for the entire monitoring period (2005-08).

During tropical storm Fay (August, 2008), water level at Well 15 exceeded the ground level earlier than Wells 11 and 19. Water near Wells 11 and 15 receded almost at the same time but water at Well 19 receded more quickly than those at Wells 11 and 15. Overall, it seems that area near Well 11 is hydrologically more sensitive than the areas near the other two wells at wetland 4. If the amount of P stored in the soil and its potential to move is higher at Well 11, it may result in transport of much higher P through overland flow than the other two well sites. The groundwater table depth data at additional wells at wetland 4 will be examined to identify the hydrologically active areas for the final project report.

Figure 7. Well locations at wetland 1.

Figure 8. Well locations at wetland 4.

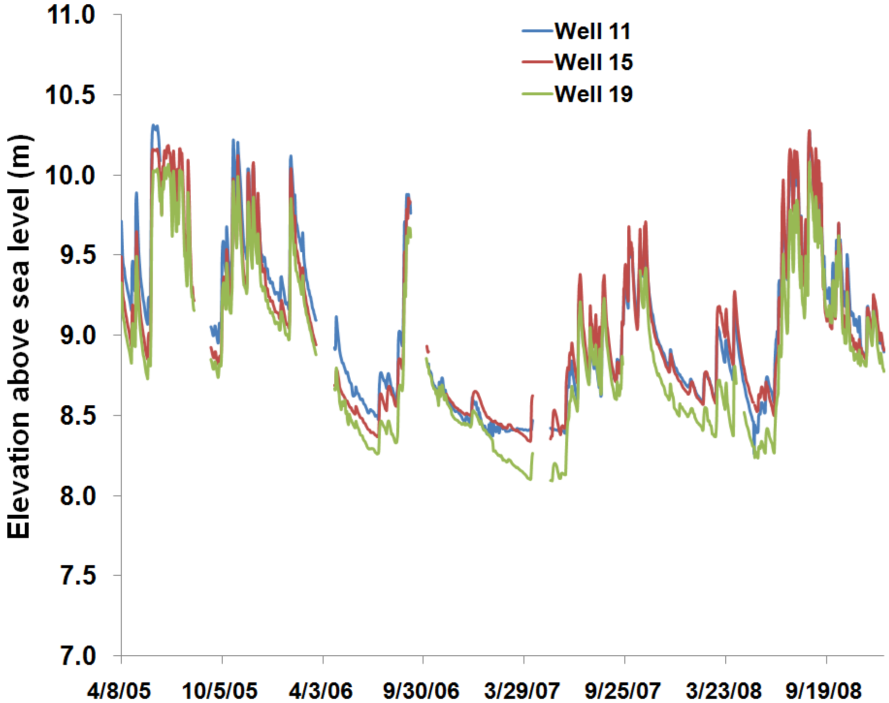

Figure 9. Mean daily water table depths (meter above sea level) at wetland site 4 for the entire monitoring period (2005-08)

Evaluation of Flow Measurements for Quantifying BMP Effects

Acoustic Doppler Velocimeters (ADV) were installed at all flume sites in the ranch to accurately measure the flow rates. This has resulted in improving the accuracy in estimating the N and P loadings for the two BMP evaluations. To compare the flow rates measured by the ADV and the flume, the flow rates (Figure 10) measured by both ADV and flume at wetland 4 (flume 4) outlet were selected. It was observed that the flow rates measured by the ADV and the flume were very similar (r^2 = 0.97). The installation of the ADVs has increased the accuracy in flow measurements and BMP analyses.

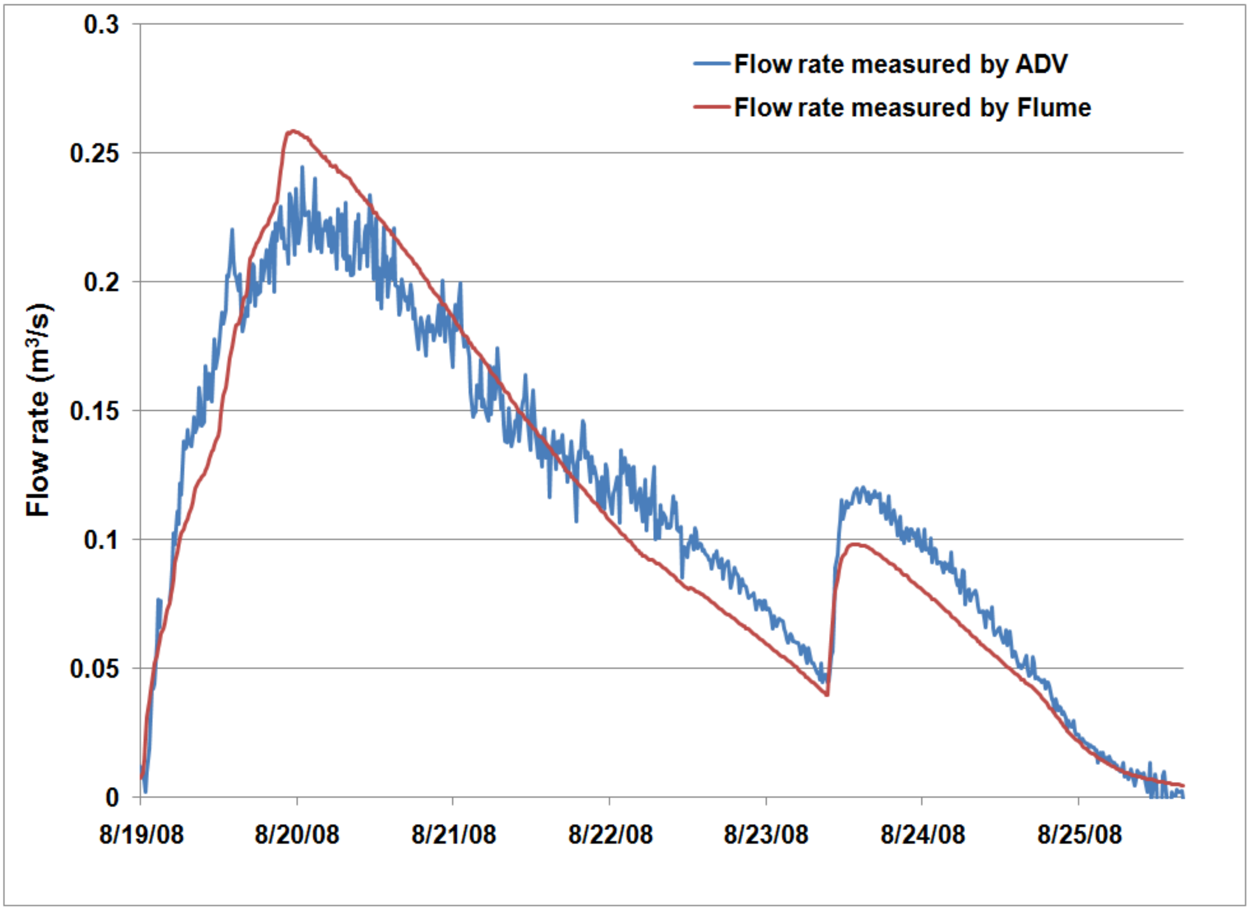

Figure 10. Comparison of flow rates measured by acoustic Doppler velocimeter (ADV) and flume (wetland site 4).

Economic Analysis of the WWR BMP Installation

BMPs were also assessed regarding their economic effectiveness for nutrient removal. Potential impacts of BMP's on cow herd health and performance and other non-market values were also evaluated through interviews with ranch personnel. For the WWR BMP, N and P loads flowing out of the two wetlands were compared between the pre- and post-BMP periods. Nutrient loads were reduced during post-BMP periods at wetland 4 but not at wetland 1. Average TN and TP loadings (combined for both wetlands) were calculated based on average loadings from all pre- and post-BMP periods for the two sites. This showed net reductions of N and P during the post-BMP period. There were three pre-BMP periods and five post-BMP periods in total for the entire monitoring period for the two wetlands. On average there were 16% reduction in TN loadings (Pre-BMP: 214 kg and post-BMP: 180 kg), and 23% reduction in TP loadings (pre-BMP: 92 kg and post-BMP: 71 kg) during the post-BMP periods. Based on the amortized cost of $1,070 per year for the WWR BMP, the average unit cost nutrient removal was $31.40 per kg of N and $50.90 per kg of P (Table 10).

Table 10. Nitrogen and phosphorus load reduction and unit cost of removal for wetland water retention BMP

Nutrient	Pre-BMP Average Load (kg)	Post-BMP Average Load (kg)	Loading Change (%)	Average Nutrient Removal Rate (kg/year)	Average Nutrient Removal Cost ($/kg)
Total Phosphorous	92.1	71.1	-22.8%	21.0	50.9
Total Nitrogen	214.2	180.2	-15.9%	34.1	31.4

Summary and Conclusion

Performance of WWR BMP was evaluated in two wetlands (wetlands 1 and 4) within a beef cattle ranch (Pelaez & Sons) by comparing N and P loads and concentrations between pre- and post-BMP periods. For wetland 4, the BMP seemed to be effective since the average TN and TP loads for the three post-BMP periods were lower than the TN and TP loads for the pre-BMP period. Also the TN and TP loadings and flow-weighted concentrations during post-BMP3 were lower than those for the pre-BMP period (pre-BMP and post-BMP3 periods had similar rainfall).For wetland 1, the average TN and TP loads for the two post-BMP periods were higher than the average TN and TP loads for the two pre-BMP periods. The BMP did not seem to be effective in reducing the N and P loads at wetland 1. Factors such as inundation of P hot spots during large rainfall events, different groundwater contribution of N and P during pre-BMP and post-BMP periods may have resulted in increased P loadings during post-BMP periods at wetland 1. Hydrologic and water quality monitoring is continuing at this site for another year and soil P storage capacity of the soils within the wetland drainage area will be quantified to better evaluate the WWR BMP at the Pelaez ranch.

APPENDIX E

Groundwater Contribution of Flow and Nutrients to a Ditch in a Beef-cattle Ranch in Florida

ABSTRACT

Cattle ranches are identified as one of the non-point sources of phosphorus (P) and nitrogen (N) for Lake Okeechobee in Florida (USA). A study was conducted to estimate groundwater flow, and N and P contributions to a ditch section of 170 m in a beef-cattle ranch in the Lake Okeechobee basin for 2006-2008. Ten wells were installed on both sides of the ditch section to monitor heads in the wells and to sample water for determining N and P concentrations. The Dupuit equation was used to estimate the groundwater flow to and from the ditch. Total Nitrogen (TN) and Total Phosphorus (TP) loads were calculated based on flow and nutrients concentration data. The average annual groundwater flow to the ditch section for the three years was 1098 m^3. Average annual TN and TP loads for the three years were 2.98 and 0.25 kg, respectively. It was observed that the N and P loadings were influenced by the groundwater flow volume.

1. Introduction

Lake Okeechobee is the largest freshwater lake in the southeastern United States. The lake has been threatened by eutrophication (Harper, 1992) as a result of increased phosphorus (P) loading from surrounding watersheds (Reddy et al., 1995). Phosphorus loads originate from agricultural non-point sources, predominantly beef-cattle ranches and dairy farms (MacGill et al., 1976; Boggess et al., 1995). Ranchers use a mix of improved and unimproved pastures in the ranches. Pasture improvement includes drainage, improved forage grasses, and application of inorganic fertilizers. In the ranches, the drainage water is routed through ditches to the tributaries downstream. The ditches receive Nitrogen (N) and P loads primarily with runoff and groundwater flow. Very few studies are available which have investigated the groundwater contributions of N and P to the ditches. There were many studies (e.g. Boggess et al., 1995; Hiscock et al., 2003.) conducted in the Lake Okeechobee watershed on the estimation of P loadings by surface water but there seems to be no study available on quantification of groundwater flow and N and P loads to the ditches which contribute flow and nutrients to the tributaries which eventually flow to Lake Okeechobee.
The Dupuit equation is extensively used to calculate groundwater flow in unconfined aquifers (Delleur, 2007; Rosenberry and LaBaugh, 2008). It can be used to study ditch-aquifer interactions and to estimate groundwater contribution to the ditches. The objective of this study was to estimate the groundwater flow to a ditch section situated in a beef-cattle ranch in the Lake Okeechobee watershed using the Dupuit equation, and combine the groundwater flow with N and P concentrations data to estimate the groundwater loadings of N and P to the ditch section.

2. Materials and Methods

2.1. Study area

This study was conducted in a beef-cattle ranch in the Okeechobee watershed in southwest Florida. The ranch is dominated by improved pastures laced with shallow ditches for drainage. Native habitats in the ranch include seasonal wetlands, sawgrass marshes, palm-oak hammocks and live oak woodlands. In the ranch, a ditch section (170 m) was chosen for this study. This ditch section drains an estimated area of 0.4 ha in the ranch. This ditch section is a part of a watershed of 250.7 ha within the ranch. The downstream point of the ditch section coincides with the outlet of the watershed. The ditch section was chosen such that there was no runoff contributing to the ditch section. A topographic survey assured that and therefore, groundwater (baseflow) was the only source of flow to the ditch section. The two major forage types near the ditch section are Floralta and Stargrass.

In southwest Florida, 70% of the annual rainfall occurs during June-October (Shukla et al., 2009). Therefore June-October would be referred to as wet period and the rest of the year (January-May and

November-December) would be referred to as dry period. The average annual rainfall in the Lake Okeechobee watershed is 1130 mm (wet period: 720 mm, dry period: 410 mm (Ali and Abtew, 1999).

2.2. Hydrologic and water quality data collection

Ten wells were installed on both sides (five wells on each side) of the ditch section for water table monitoring and water sample collection (Figure 1). Out of these ten wells, six wells (well no. A, B, C, F, G, and H) were installed on a transect vertical to the ditch section with three wells on either side (east and west) of the ditch. Other four wells were installed near the transect wells close to the ditch (wells D, E, I, J). Pressure transducers were installed in the wells to monitor heads at the wells. Water samples were collected from the wells once a month for determining groundwater N and P concentrations. Rainfall data were available from a weather station installed in the ranch.

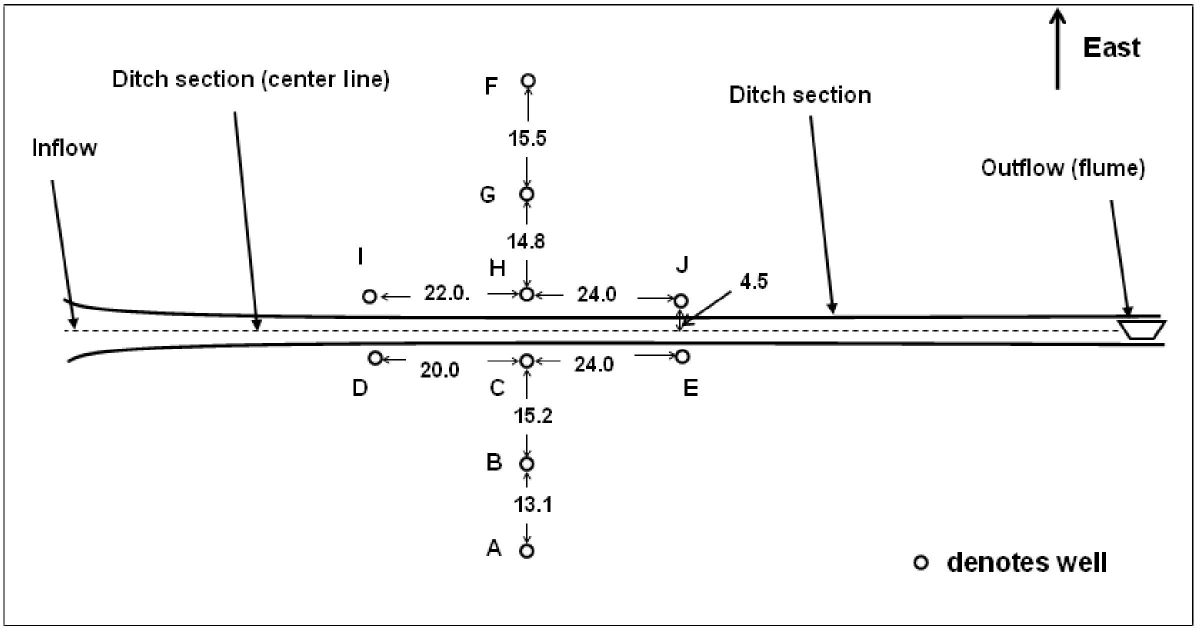

Figure 1. Experimental lay-out for the study. All distances are in meters.

Head data and water samples were collected at the site from January 2006 to December 2008. Water samples were sent to the Analytical Research Laboratory (ARL) at the University of Florida, Gainesville, FL where they were analyzed for Total Phosphorus (TP), Total Kjeldahl Nitrogen (TKN), and Nitrate-N (NO_3-N). In this study, Total Nitrogen (TN), and Total Phosphorus (TP) loads contributed by groundwater were estimated. Total Nitrogen is the combination of TKN and NO_3-N.

2.3. Groundwater flow estimation

The Dupuit equation was used to estimate the groundwater flow (baseflow) contributed per unit length of the ditch section. In some unconfined flows with a free surface, the vertical component of flow can be neglected (Freeze and Cherry, 1979). This approximation pioneered by Dupuit and utilized later by Forchheimer is know as the Dupuit-Forchheimer assumptions. The assumptions are: (a) there are no hydraulic gradients in the vertical dimension, and (b) the hydraulic gradient in the horizontal dimension equals the slope of the water table (Reddi, 2003).

The horizontal groundwater flux into the ditch section were calculated at the transect using the heads at the wells and the water level in the ditch. The Dupuit equation is given by:

$$q = \frac{1}{2} K (\frac{h_1^{\;2} - h_2^{\;2}}{L}) \tag{1}$$

where,
q = flow per unit length ($m^2\ s^{-1}$) of the stream
K = hydraulic conductivity ($m\ s^{-1}$)
h_1 = head at the well (m)
h_2 = head at the ditch (m)
L = distance between well and the ditch (m)

Groundwater flow per unit length of the ditch section multiplied by the length of the ditch section yielded the total groundwater flow contributed by the entire ditch section assuming an average depth of the ditch section in this flat topography. The groundwater flow for each side of the ditch section was calculated separately and the flow volumes from both sides were added to get the total groundwater volume. For the east and west sides, heads in wells F and A, respectively were used in the Dupuit equation. This equation gives reasonable results when the depth of the unconfined flow is shallow and the slope of the free surface is small (Delleur, 2007; Tsubo et al., 2007). Delleur (2007) reported that the shape of the water table may not be predicted correctly by the Dupuit assumptions but the equation for the flux (groundwater flow) is nevertheless exact.

Sieyes et al. (2008) used the Dupuit assumptions to estimate baseflow in an aquifer between two wells where hydraulic gradients were low. In this study also, the hydraulic gradients between the wells and the ditch were low. The average difference (absolute value) between the well head and the ditch water level for 2006 were 15 (west side) and 17 cm (east side). The average gradients for 2007 were 12 (west side) and 23 cm (east side), and for 2008, the average gradients were 12 (west side) and 21 cm (east side). The ditch section was shallow in depth; the average depth from the land surface was 1.40 m. The vertical component of groundwater flow was ignored using Dupuit Forchhemeir assumptions. The bottom elevation for the ditch section was assumed to be the reference line from which the heads in the wells and ditch water levels were measured.

Soil at the experimental site is predominantly Basinger. The saturated hydraulic conductivity for Basinger soil varies from 4.23×10^{-5} to $1.41 \times 10^{-4}\ m\ s^{-1}$ (NRCS, 2003). Therefore, an average saturated hydraulic conductivity value ($9.17 \times 10^{-5}\ m\ s^{-1}$) was used in the Dupuit equation for groundwater flow estimation. When the water level in the ditch is lower than the heads in the wells, the groundwater moves toward the ditch. This direction of flow will be referred to as positive flow. When the water level in the ditch is higher than the heads at the wells, the water moves from the ditch to the sides. This direction of flow will be referred to as negative flow. It was observed that the groundwater flowed in and out of the ditch at different times during the entire study period.

2.4. Groundwater load estimation

The loads of TN and TP were calculated by multiplying the groundwater flow ($L\ s^{-1}$) with the corresponding TN and TP concentrations ($mg\ L^{-1}$) in the water samples collected from the wells. When groundwater flow was positive, the average concentrations of TN and TP for all the five wells in one side of the ditch were used to calculate the loadings from that particular side. But when the groundwater flow was negative, the concentrations of TN and TP species at the wells nearest to the ditch were used for load estimation for the respective sides (wells C, D, E for west side and wells H, I, J for east side) assuming that the N and P concentrations in these wells were similar to the ditch water concentrations due to the close proximity of these six wells to the ditch. The load was positive or negative using the same convention as for groundwater flow. Groundwater flow, and TN and TP loadings were calculated on yearly basis and also separately for dry (January to May, and November-December) and wet (June-October) periods of the year.

2.5. Ranch management data

Ranch management data were collected from the ranch owner for the entire study period. The data on ranch management included pasture type, fertilizer application rates, and animal (cattle) stocking rate. Equal amounts of NPK fertilizer (20-5-5) were applied during each year in September at the rate of 392 kg ha^{-1}. The heads of cattle at the site were similar for all years. Therefore, there were no major variability in terms of fertilizer application and animal stocking rates.

2.6. Surface flow and water quality monitoring

A trapezoidal flume was installed at the downstream point of the ditch section which is also the outlet of the watershed, to measure the surface flow volume that exited the watershed (ranch). An auto-sampler was installed near the flume to sample water to determine TN and TP concentrations in the surface water exiting the ranch. The objective was to study the flow and nutrients (TN and TP) contributed by the ditch section in relation to those contributed by the entire watershed.

3. Results and Discussion

During the study period, there were yearly variations in rainfall and surface flow. Year 2006 was moderately wet, 2007 was a dry year, and 2008 was a wet year at the study site in terms of surface flow and wetness of soil. The average groundwater flow to the ditch section for the three years was 1098 m^3 per year. During 2006 and 2008, the annual groundwater flows contributed by the ditch section were similar (Table 1). Even though there was a difference in rainfall amounts for these two years, relatively wetter season in 2005 due to hurricane Wilma that occurred in October 2005 made the first part (January-May) of 2006 wet even though June-October was relatively dry. For the year 2008, the first part (January-May) was relatively dry, but June-October was wet due to tropical storm Fay (August, 2008) resulting in high groundwater contribution during the wet period.

Table 1. Groundwater flow volumes contributed by the ditch section for the study period (2006-2008).

Year	Annual		Wet period		Dry period	
	Rainfall (mm)	Baseflow (m^3)	Rainfall (mm)	Baseflow (m^3)	Rainfall (mm)	Baseflow (m^3)
2006	937	1409	633	955	302	454
2007	983	320	755	569	226	-248
2008	1494	1566	1124	1978	371	-413
Average	1138	1098	836	1167	300	-69

'-'sign denotes that the flow is negative (flowing from the ditch to the sides).

The first part (January-May) of 2008 had the lowest groundwater flow among all years due to the extreme dry condition that continued since 2007. Lake Okeechobee experienced the lowest water level in 2007 based on historical data collected by the South Florida Water Management District (SFWMD, 2009). But the wet period (June-October) of 2008 contributed high amounts of flow as a result of very wet condition caused by tropical storm Fay. During the dry periods (January-May, and November-December) of 2007 and 2008, the net groundwater flow volumes to the ditch section were negative due to low amounts of rain (226 and 371 mm, respectively) for those periods. For January-May of 2006, the net groundwater flow to the ditch was positive because the heads at the wells were higher than the ditch water level for most instances. Even though, the rainfall amount during the dry period in 2006 was only 302 mm, the high rainfall during the last part of 2005 created a wet condition during January-May (2006) resulting in larger groundwater flow volume.

Table 2 shows the annual surface flow volume measured at the flume and contributed by the entire watershed, and groundwater contributed by the stream section on per unit area (ha) basis. Since there was no runoff contribution within the ditch section, the only source of flow was through groundwater within the ditch section. It was observed that the annual flow volume per ha was larger for the experimental site (ditch section) as compared to the flow contributed by the entire watershed area for all years.

Table 2. Groundwater flow volumes per unit area (ha) for the study period (2006-2008).

Year	Entire watershed		Experimental site (ditch section)	
	Surface flow measured at flume (m^3)	Flow per unit area $m^3\,ha^{-1}$	Total flow (m^3)	Flow per unit area $m^3\,ha^{-1}$
2006	88,194	352	1409	3523
2007	14,933	60	320	800
2008	514,019	2051	1566	3915
Average	205,715	821	1098	2746

There are many factors that might have contributed to the generation of lower overall flow generation per unit area of the watershed as compared to that generated at the experimental site (ditch section). Out of the 250.7 ha of the total watershed area, 80 ha was encompassed by a wetland and a wetland retains flow and nutrients (Reddy et al., 1999; Saunders and Kalff, 2001) reducing flow downstream. Moreover, the entire watershed encompassed many large and small ditches. The ditch in the experimental site is one of the largest and deepest ditches in the watershed and therefore, was expected to receive more flow.

Average annual TN and TP loads for the three years were 2.98 and 0.25 kg, respectively (Table 3). Figure 2 represents the groundwater flow and loads (TN and TP) for the study period. During the year 2006, there were positive net contributions of TN and TP during both dry (January-May and November-December) and wet (June-October) periods. During the dry periods (January-May and November-December) of 2007 and 2008, the net contributions of TN and TP were negative. The linear regressions between flow and TN load, and between flow and TP load were performed for dry and wet periods separately to determine the correlation (if any) between groundwater flow and nutrients loads. For wet periods, the coefficients of determination (r^2) between groundwater flow and TN load, and between groundwater flow and TP were 0.99 and 0.84, respectively. For the dry period, the coefficients of determination (r^2) for TN and TP with flow were 0.99 and 0.93, respectively. This indicates that strong correlations exist between flow volumes and nutrient (TN and TP) loads.

Table 3. Total nitrogen (TN) and total phosphorus (TP) loads contributed by the ditch section for the study period (2006-2008).

Year	Total Nitrogen (kg)			Total Phosphorus (kg)		
	Annual	Wet period	Dry period	Annual	Wet period	Dry period
2006	3.448	2.265	1.183	0.160	0.134	0.026
2007	0.885	1.43	-0.545	0.162	0.189	-0.027
2008	4.601	5.629	-1.028	0.422	0.49	-0.068
Average	2.978	3.108	-0.130	0.248	0.271	-0.023

'-'sign denotes that the load was contributed by the ditch section to groundwater on both sides.

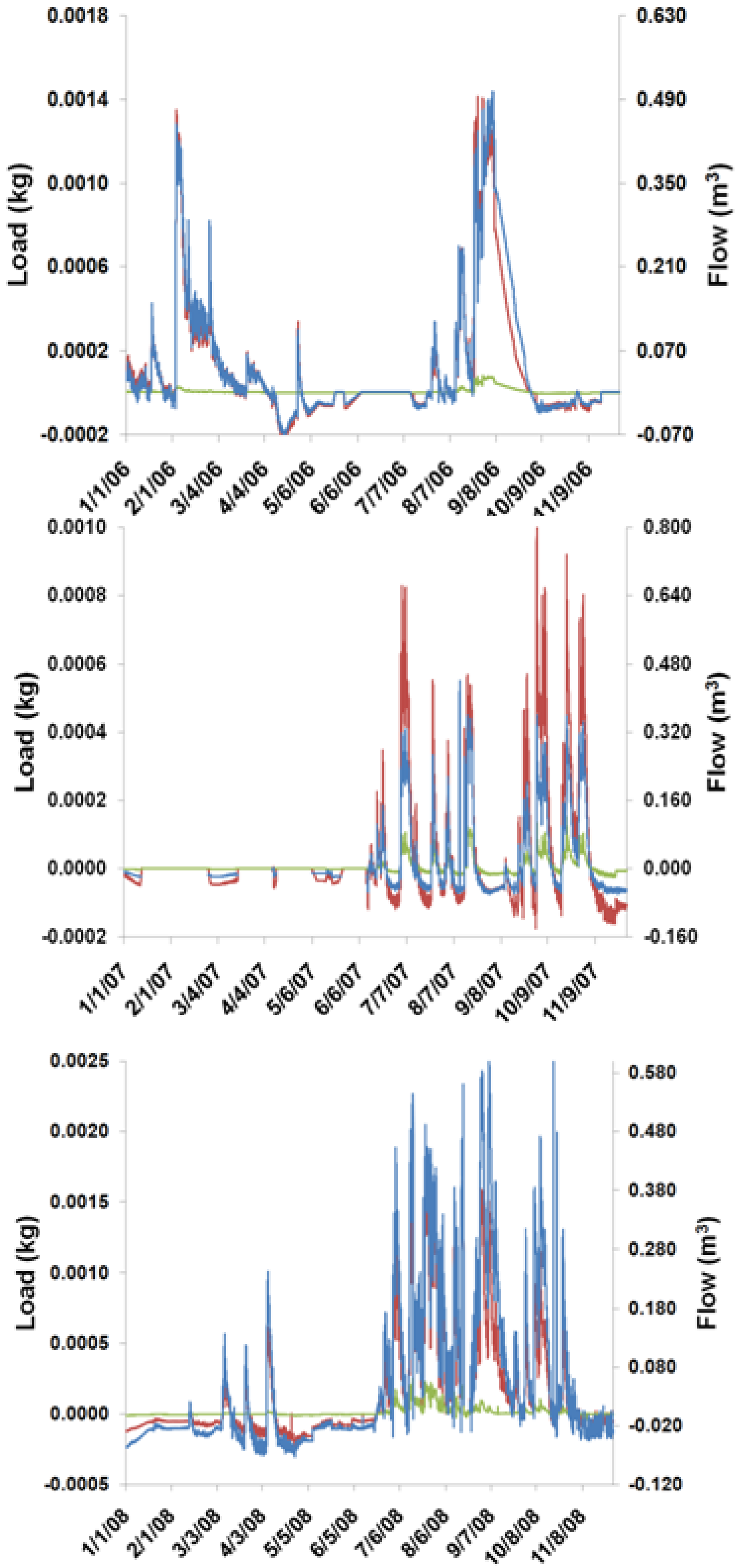

Figure 2. Plots showing the groundwater flow volume (blue line), Total Nitrogen (red line), and Total Phosphorus (green line) loads for the monitoring period 2006-2008.

Table 4 shows the annual TN and TP loads contributed by the entire watershed and by the experiential site (ditch section) alone. The overall contributions of TN and TP by the watershed were lower than those contributed by the ditch section, expect for TP in 2008 when TP contribution by the ditch section was lower than the watershed-scale contribution of TP. The reasons for lower TN and TP contribution by the entire watershed might be the same as those for lower groundwater flow contributed by the watershed compared to the ditch section (possible role played by wetland, size of ditch).

Table 4. Total Nitrogen (TN) and Total Phosphorus (TP) loads per unit area (ha) for the study period (2006-2008).

Year	Entire watershed				Experimental site (Ditch section)			
	Total Nitrogen (kg)		Total phosphorus (kg)		Total Nitrogen (kg)		Total phosphorus (kg)	
kg	Total	Per ha	Total	Per ha	Total	Per ha	Total	Per ha
2006	206.19	0.82	75.47	0.30	3.45	8.62	0.16	0.40
2007	41.59	0.17	31.96	0.13	0.89	2.21	0.16	0.41
2008	1685.00	6.72	751.32	2.99	4.60	11.50	0.42	1.06
Average	644.26	2.57	286.25	1.14	2.98	7.44	0.25	0.62

There are a number of ranches in southwest Florida which contribute flow and nutrients to the Lake Okeechobee watershed. This type of study may be helpful in formulating the water quality best management practices suitable for these sites.

4. Conclusion

This study was conducted to estimate the flow and nutrients (N and P) contributed to a ditch section through sub-surface environment within a beef-cattle ranch for three years (2006-08). The ditch section was located at the downstream end of a small watershed within the ranch. The yearly average groundwater flow contributed to the ditch section was 1098 m^3. The wet periods generated larger groundwater flow as compared to the dry periods. The N and P contributions were influenced by the groundwater volume. Larger groundwater flow contributed larger N and P loads to the ditch. In general, the flow, and N and P loads contributed by the entire watershed were lower than those contributed to the ditch section under study. This might be due to the larger size of the ditch section under study which was expected to receive more flow and nutrients (N and P), and the presence of a wetland at the upstream part of the watershed which was expected to retain flow and nutrients (N and P).

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Appendix F

Preliminary results on Hydrologic Model Evaluation

Calibration and evaluation of Watershed Assessment Model (WAM) are ongoing using hydrologic component (runoff) for both wetlands for the pre-BMP periods (Jan'05-Jan'07 for wetland 1, Jan'05-Feb'06 for wetland 4). Calibration and evaluation of the water quality component (phosphorus loadings) for both wetlands and evaluation of the wetland water retention (WWR) BMP are underway in cooperation with Soil and Water Engineering Technology (SWET), Gainesville, FL.

Calibration and evaluation of Agricultural Catchment Research Unit (ACRU) model using hydrologic and water quality data (2005-08) are continuing for both wetlands. Model calibration and evaluation are being carried out using two statistical criteria, the root mean squared error (RMSE) and the index of agreement (*d*). Preliminary results for WAM and ACRU are provided in Figures 1 and 2, and Tables 1, 2, 3, and 4. Results show that ACRU over-predicted the runoff for both wetlands and WAM slightly under-predicted the runoff for wetland 1. Detailed discussion of the model evaluation results will be provided in the final report.

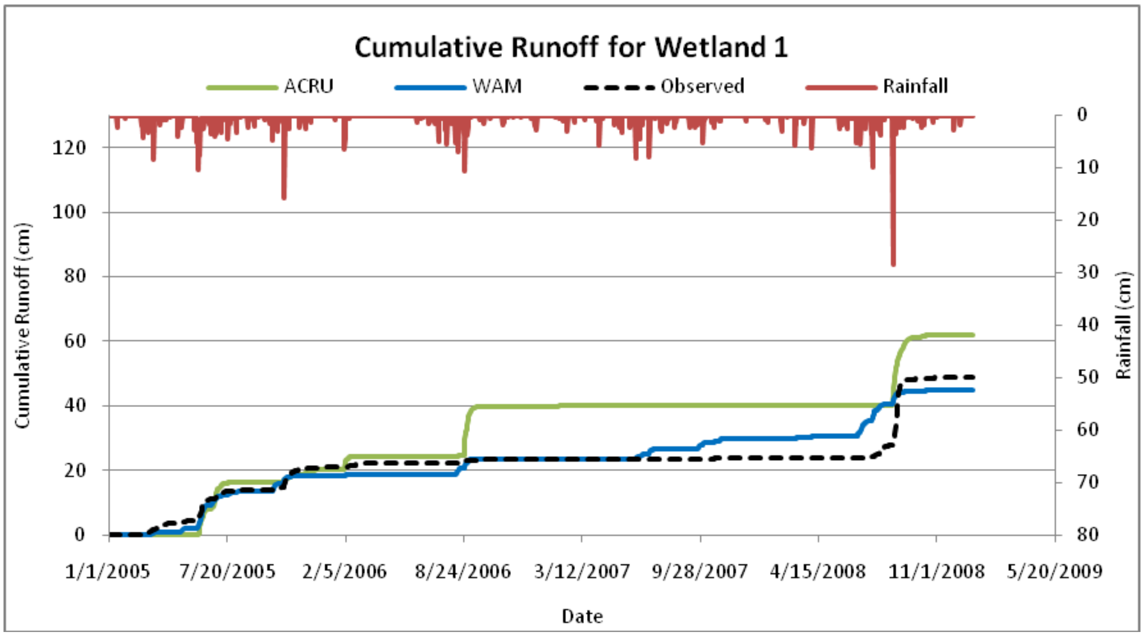

Figure 1. Rainfall, observed cumulative runoff, and WAM and ACRU simulated cumulative runoff for wetland 1 for the pre-BMP period (Jan '05-Jan '07) and post-BMP period (Feb '07-Dec '08).

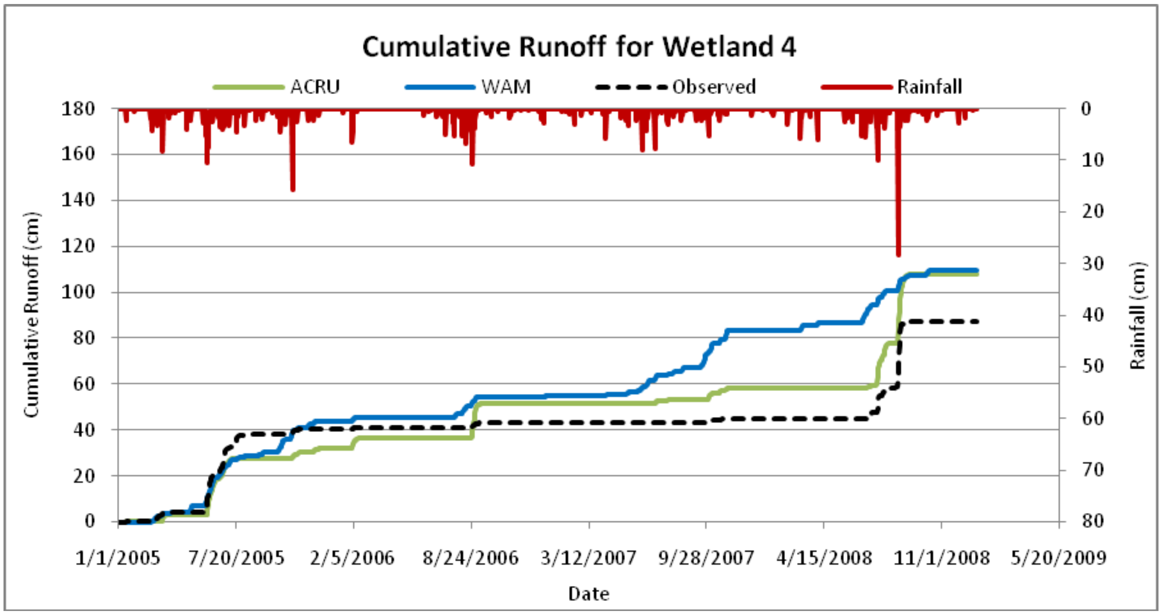

Figure 2. Rainfall, observed cumulative runoff, and WAM and ACRU simulated cumulative runoff for wetland 4 for the pre-BMP period (Jan ’05-Feb ‘06) and post-BMP period (March ’06-Dec ’08).

Table 1. Statistic criteria (Mean, standard deviation, root mean square error (RMSE) and index of agreement (*d*)) analysis of ACRU for wetland 1.

Mean				Standard Deviation				RMSE		d	
Calibration		Evaluation		Calibration		Evaluation					
Obs[a]	Sim[b]	Obs	Sim	Obs	Sim	Obs	Sim	Cali[c]	Eval[d]	Cali	Eval
0.031	0.052	0.036	0.032	0.086	0.241	0.218	0.209	0.22	0.19	0.42	0.76

[a]Observed, [b]Simulated, [c]calibration, [d]evaluation.

Table 2. Statistic criteria (Mean, standard deviation, root mean square error (RMSE) and index of agreement (*d*)) analysis of WAM for wetland 1.

Mean				Standard Deviation				RMSE		d	
Calibration		Evaluation		Calibration		Evaluation					
Obs	Sim	Obs	Sim	Obs	Sim	Obs	Sim	Cali	Eval	Cali	Eval
0.031	0.032	0.036	0.032	0.086	0.101	0.218	0.099	0.08	0.21	0.80	0.35

Table 3. Statistic criteria (Mean, standard deviation, root mean square error (RMSE) and index of agreement (*d*)) analysis of ACRU for wetland 4.

Mean				Standard Deviation				RMSE		d	
Calibration		Evaluation		Calibration		Evaluation					
Obs	Sim	Obs	Sim	Obs	Sim	Obs	Sim	Cali	Eval	Cali	Eval
0.097	0.086	0.044	0.069	0.337	0.241	0.393	0.378	0.25	0.24	0.79	0.89

Table 4. Statistic criteria (Mean, standard deviation, root mean square error (RMSE) and index of agreement (*d*)) analysis of WAM for wetland 4.

Mean				Standard Deviation				RMSE		d	
Calibration		Evaluation		Calibration		Evaluation					
Obs	Sim	Obs	Sim	Obs	Sim	Obs	Sim	Cali	Eval	Cali	Eval
0.097	0.107	0.044	0.049	0.337	0.212	0.393	0.142	0.29	0.35	0.65	0.48