Evaluation of Cow-calf Best Management Practices With Regards to Nutrient Discharges in the Lake Okeechobee Basin

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Contents

Project Summary

A long-term project was started in 2003 to demonstrate and evaluate the water quality effectiveness of the cow-calf BMPs within the Okeechobee basin. Ditch fencing and culvert crossing (DFCC) and wetland water retention (WWR) were the two BMPs evaluated in this study with regards to water quality (N and P concentration and load) and economics. The two BMPs were evaluated at a 275 hectare ranch (Pelaez Ranch, Figure 1) located in Okeechobee County, FL. Pre- and post-BMP monitoring data collection and analyses were used to evaluate the two BMPs. The DFCC BMP, aimed at reducing the direct input of nutrients by cattle exclusion from the ditch, was implemented along the main drainage ditch, which spans from the junction of Q2 and Q3, to Q5 (Figure 1). Most of the ranch drains through this ditch with Q5 being located at the end of the ranch. The main drainage ditch receives flows from two ditches with outlets at Q2 and Q3. The WWR BMP, aimed at increasing the wetland water and nutrient retention, was evaluated at the outflow points of two wetlands located at Q1 and Q4 (Figure 1). This report summarizes the hydrology and water quality data collected between June 2009 to May 2010 as part of a 2009 project funded by the FDCAS and SFWMD and uses this data in conjunction with earlier data (June 2005 – May 2009) to evaluate the two BMPs. The task for this project were: collect land use, hydrologic, and water quality data to evaluate the BMPs *(Task 1)*; collect and analyze soils data to evaluate the role of soils in affecting the BMP Effectiveness (*Task 2*); disseminate the study results (*Task 3*); evaluate the accuracy of ADV and head-based flow measurements at flumes (*Task 4*); and evaluate the effectiveness of the two BMPs (*Task 5*). The project was amended to collect limited flow and water quality data for the June 2010-December 2010 period and combine with earlier data to evaluate the WWR BMP. Results for the above tasks are summarized below with details of specific Tasks presented in Appendices A-1, A-2, A-3, B-1, B-2, B-3, C, D, E, and F.

Evaluation of the BMPs (Tasks 1, 2, and 5)

The DFCC BMP was evaluated using the data for two pre-BMP periods (pre-BMP1 and pre-BMP2, June 2005-May 2006 and June 2009-May 2010) and three post-BMP periods (post-BMP1, post-BMP2, and post-BMP3, June-May of 2006-2007, 2007-2008, 2008-2009, respectively). For the WWR BMP evaluation at Wetland 1, June-May of 2005-2006 (pre-BMP1) and 2006-2007 (pre-BMP2) were the two pre-BMP periods while June-May of 2007-2008 (post-

BMP1), 2008-2009 (post-BMP2), and 2009-2010 (post-BMP3) were the three post-BMP periods. For Wetland 2, June-May of 2005-2006 was the only pre-BMP period, while June-May of 2006-2007 (post-BMP1), 2007-2008 (post-BMP2), 2008-2009 (post-BMP3), and 2009-2010 (post-BMP4) were the four post-BMP periods. June-December 2010 was considered the post-BMP4 period for Wetland 1, and post-BMP5 period for wetlands 2. Although this period was shorter than other pre- and post-BMP periods, for completeness sake it was assumed to represent one year of hydrologic and water quality data.

Ditch Fencing and Culvert Crossing

The DFCC was evaluated with regards to reductions in P and N concentration and load and economic feasibility at a 170 m section of the drainage ditch section between Q5 (downstream) and junction of Q2 and Q3 (upstream). Details of DFCC BMP analysis for the wet season and annual (June-May) periods are presented in Appendices A-1 and A-2, respectively. Groundwater fluxes of N and P to the ditch section was also analyzed to examine whether the year to year variation in N and P loads to the drainage ditch could be a source of uncertainty in the evaluation of the DFCC.

Average pre-BMP upstream total P (TP) load was 435 kg and was lower than the downstream P load of 539 kg. Phosphorus load increase of 24% at the downstream location was mainly due to deposition of feces and urine from cattle traffic. Reverse TP load trends were found for the three post-BMP period with downstream average of 268 kg against upstream average of 300 kg showing 11% load reduction after cattle exclusion. For total N (TN) loads, mixed results were observed. Nitrogen loads at the downstream site were lower than upstream during the pre-BMP1 (10%), post-BMP1 (13%), and post-BMP3 (11%) periods while they were higher than the upstream TN loads during the pre-BMP2 (17%), and post-BMP2 (2%) periods. The ditch was a sink for N for all periods except the pre-BMP2 and post-BMP2 periods. Unusual dry conditions during the post-BMP2 (2007-2008) resulted in the addition of P and N at the BMP site, probably due to the release of P and N from soil and plants. Average groundwater contributions of TP and TN loads to the ditch section ranged from 0.080 to 0.174 and 2.053 to 4.481 kg, respectively. Since the TP and TN contributions from groundwater were relatively small they can be ignored (details presented in Appendix A-3).

Reductions in total P concentrations during the wet period at the downstream location were statistically significant during the post-BMP periods. The downstream minus upstream TP load, a measure of P addition due to cattle traffic, for the wet season post-BMP3 period were statistically less than the pre-BMP1 period for the months of July, August and September. For post-BMP1 period, statistically significant results were observed for September. Overall, considering similar rainfall for the pre-BMP1 and post-BMP3 periods and significant reductions in TP contribution during the post-BMP3 period (three of the four wet season months compared), it was concluded that the BMP was effective in reducing the TP loads and concentrations.

To consider potential P contributions from the soil and plant, two scenarios, conservative and liberal were considered to estimate P load reductions due to the DFCC. For the conservative scenario, P contribution from soil and plant was considered while for liberal it was not. Reductions in P loads due to DFCC for the conservative and liberal scenarios were 0.26 and 0.29 kg/day, respectively. Phosphorus removal cost for the conservative scenario was $17.02 /kg of P which is considerably less than the cost of other P reduction strategies in the basin. *Overall, results show that the DFCC is a BMP and can reduce TP concentration and loads from ranches without causing adverse impact on cattle production.*

Wetland (and Pasture) Water Retention

Wetland water retention (WWR) BMP was implemented at two wetland (Wetlands 1 and 2) sites within the ranch (Figure 1). The WWR was implemented at the two wetlands by installing a flashboard riser structure at the ditch that drained the wetland as well as the connected upland pasture areas. Boards were added to heights of 1.10 and 0.52 m above the ditch bottom at Wetland 1 and Wetland 2, respectively. At Wetland 1, June-May periods of 2005-2006 (pre-BMP1) and 2006-2007 (pre-BMP2) were the two pre-BMP periods while June-May of 2007-2008 (post-BMP1), 2008-2009 (post-BMP2), and 2009-2010 (post-BMP3) were the three post-BMP periods. For Wetland 2, June-May of 2005-2006 was the only pre-BMP period and June-May of 2006-2007, 2007-2008, 2008-2009, and 2009-2010 were the four post-BMP periods (post-BMP1, post-BMP2, post-BMP3, and post-BMP4 periods, respectively). Due to incomplete data for the water year, June-December (2010) period was considered as the post-BMP4 period for Wetland 1, and post-BMP5 period for Wetland 2. Loads and concentrations of TP and TN for the pre- and post-BMP periods were compared to evaluate the effectiveness of the WWR.

Compared to pre-BMP1, higher runoff, runoff per unit rainfall, surface storage, and inundated area, and longer hydroperiods were observed at Wetland 1 during post-BMP2 and 3 periods. Rainfalls during these three periods were similar. Shallower groundwater and increased ponding and surface water connectivity of upland areas (e.g. upland pastures and high P-impacted cattle feeding areas) to the wetland may have increased the runoff depth from Wetland 1 area. Although TP and TN loads for post-BMP1 period were less than those during the two pre-BMP periods, these reductions were mainly due to record drought conditions during this period. For post-BMP2 period, TP and TN loads were more than twice than those during pre-BMP1 period. Post-BMP3 TN load was higher than the pre-BMP1 period while TP loads were similar for these two periods. Total P and TN loads during post-BMP4 period were low due to low flow (due to lower than average rainfall) during this period. The average TP and TN loads, and mean concentrations for the post-BMP periods were higher than those for the two pre-BMP periods. Statistical analyses of the TP and TN concentrations and loads revealed significantly higher values for the post-BMP periods. Both mean daily annual and wet season TP and TN concentrations for Wetland 1 increased while increased mean daily TP loads was observed for post-BMP3 compared to pre-BMP1 period. When the statistical analyses of mean daily TP loads was limited to only wet season, post-BMP TP loads became significantly higher than the pre-BMP loads. Increased TP concentration (annual and wet season) and loads (wet season) for the post-BMP periods were likely due to combination of increased rainfall and runoff, inundated areas, and surface water connectivity of areas with no soil P retention capacity (e.g. high P-impacted areas located at the cattle feeding areas). Overall, there exists some evidence of increase in runoff, and TP and TN concentrations and loads from the Wetland 1 area. These increases may have been caused by the WWR, however given the unequal numbers of pre-BMP and post-BMP periods with varying rainfall depths combined with surface and groundwater interactions, long-term data is needed to accurately quantify the effects of WWR on TP loads.

Compared to Wetland 1, results for Wetland 2 were mixed with regards to the effects on runoff and water quality. Similar to Wetland 1, longer hydroperiods were observed at Wetland 2. However unlike Wetland 1, the runoff at Wetland 2 decreased for the post-BMP3 and 4 periods, compared to the pre-BMP period. These three periods had similar rainfall depths. The mean TP and TN post-BMP concentrations were higher than the mean concentrations during the pre-BMP period. Numerically, TP loads for all the five post-BMP periods were lower than the pre-BMP period. Total N loads for all post-BMP periods except post-BMP4, were lower than pre-BMP loads. Results from statistical analyses showed no significant difference between the mean daily pre-BMP and post-BMP TP and TN concentrations. For the statistical comparisons of mean daily TP loads during the wet period, there was some evidence that post-BMP wet period TP loads were higher than the pre-BMP wet period TP loads ($p=0.0962$). In summary, statistical analyses indicated limited to no evidence that post-BMP TP and TN concentrations or loads were significantly different than the pre-BMP period. Overall, some level of water retention was observed at the Wetland 2 site but TN and TP retention benefits of WWR could be not be established.

The two wetlands responded differently to the BMP. Accurate estimation of evapotranspiration (ET) (Appendix E) and further analyses of subsurface flow dynamics during post-BMP periods may improve the ability to explain the fate of retained water at the two wetland sites. This will improve the ability to quantify the effects of WWR on water retention. Average TP loads increased at Wetland 1, and decreased at Wetland 2 during all post-BMP periods. Soil Phosphorus Storage Capacity (SPSC) analysis for the soil at Wetland 1 indicated that a large fraction of the Wetland 1 area has no P retention capacity left in the surface horizon. These areas can become source of P upon flooding. Rise in water table and inundation of areas with no P retention capacity (including the P hotspots at the cattle feeding areas) due to increased inundation resulting from the WWR may have increased the higher TP concentration and loads observed for some of the post-BMP periods. In contrast to Wetland 1, most of the Wetland 2 area still has some P retention potential left at the surface soils and does not have P hotspots. This may be the reason for lower post-BMP TP concentration and loads compared to the pre-BMP period.

Based on this study, the following observations can be made:

1. Although water retention on ranchlands using a combination of wetland and pasture as water storage areas has been assumed to be a promising BMP, both with regards to water and nutrient retention, data from this project do not necessarily support these assumptions.

2. WWR involves interaction of surface and subsurface water and nutrient processes which when combined with natural climatic variability makes it difficult to attribute the observed changes in water and P dynamics to WWR alone.

3. Long-term data comprising multiple years of pre- and post-BMP periods is necessary to detect statistically significant changes due to WWR implementation. Although six years of monitoring data was used to quantify WWR effects, the unequal number of pre- and post-BMP periods (e.g. only one pre-BMP period for Wetland 2) and climatic variability makes it difficult to draw conclusions on the effects of WWR.

4. The evaluation of WWR performed in this study is interim at best. The results of this study show that there is likelihood of increasing TP concentration and loads from implementation of WWR, especially if soil/hydrologic conditions favor both increased overland flow and soil P release. Inundation of high P-impacted areas as well as areas with low soil P retention may result in increasing the TP load. The ability to reduce the TP load will depend on the balance between relative reductions in runoff and increased TP concentrations due to inundation of areas with low soil P retention capacity (e.g. Wetland 1).

5. The WWR monitoring study at Pelaez should continue for at least three years to capture additional data under pre-BMP conditions. Advantage of additional pre-BMP data in improving the quantification of BMP effects was clearly seen for the DFCC BMP. However, DFCC mainly involved surface (soil, plant, and water) processes while both surface and subsurface processes are important for WWR which may result in the effects of WWR on TP loads not detectable before several years after WWR implementation. This further necessitates the need for long-term data for evaluating this important proposed BMP.

6. Use of hydrologic models for evaluating water aspect of WWR with acceptable accuracy is possible especially with the use of eddy based ET estimates and accurate representation of subsurface processes in the model. However, evaluating effects on P concentration and loads may not be so easily achievable through the use of models.

7. The two wetlands considered in this study capture a partial range of soil, topographic, hydrologic, and plant variability present in the wetland/upland ecosystems located at ranchlands within the northern Everglades basin. Results indicated differences in response to the WWR implementation such that runoff and TP loads seemed to have increased at Wetland 1 while they decreased at Wetland 2.

Details of WWR evaluation are presented in Appendix B-1. Details on the effect of WWR on hydroperiods and water storage are presented in Appendix B-2. The soil P retention characteristics and their effects on observed TP loads for the two wetland sites are presented in Appendix B-3.

Performance, Reliability, Accuracy of the Acoustic Doppler Velocimeters and Transducer Methods for Flow Measurements (Task 4)

Two flow measurement techniques were used to estimate surface water flows at the ranch and were evaluated for their performance, reliability, advantages and disadvantages. The two methods included (a) the use of Acoustic Doppler Velocimeters (ADVs) to measure water velocity combined with area-velocity technique for flow estimation, and (b) flow rating equation using flume structure and hydraulic heads to estimate flow.

The ADV area-velocity technique was considered to be the standard due to its proven accuracy and reliability under both lab and field conditions. The ADV performed well under all flow conditions, but did experience data loss when vegetation or other material obstructed the sensor path. The flume-based flow measurement agreed well with the ADV-based flow when free-flowing conditions existed, but over-predicted flow under submerged conditions. Despite the lack of agreement between the two results, when compared to each other, the coefficients of determination (R^2) were relatively high for the two flumes evaluated (0.79 and 0.90). Detailed results of these analyses are presented in Appendix C.

Dissemination of the Project Results (Task 3)

Results from the BMP studies have been disseminated to the ranchers, state agencies, and other stakeholders. Results from the study have been presented at various state and/or national conferences. Details of these presentations are presented in Appendix D.

Eddy Flux Data to Improve Evapotranspiration Estimates in BMP Evaluation (Part of Tasks 1 and 5)

Results for this sub-task are presented in Appendix E.

Use of Hydrologic Models to Evaluate Different Water Retention Scenarios (Part of Task 5)

Results for this sub-task are presented in Appendix F.

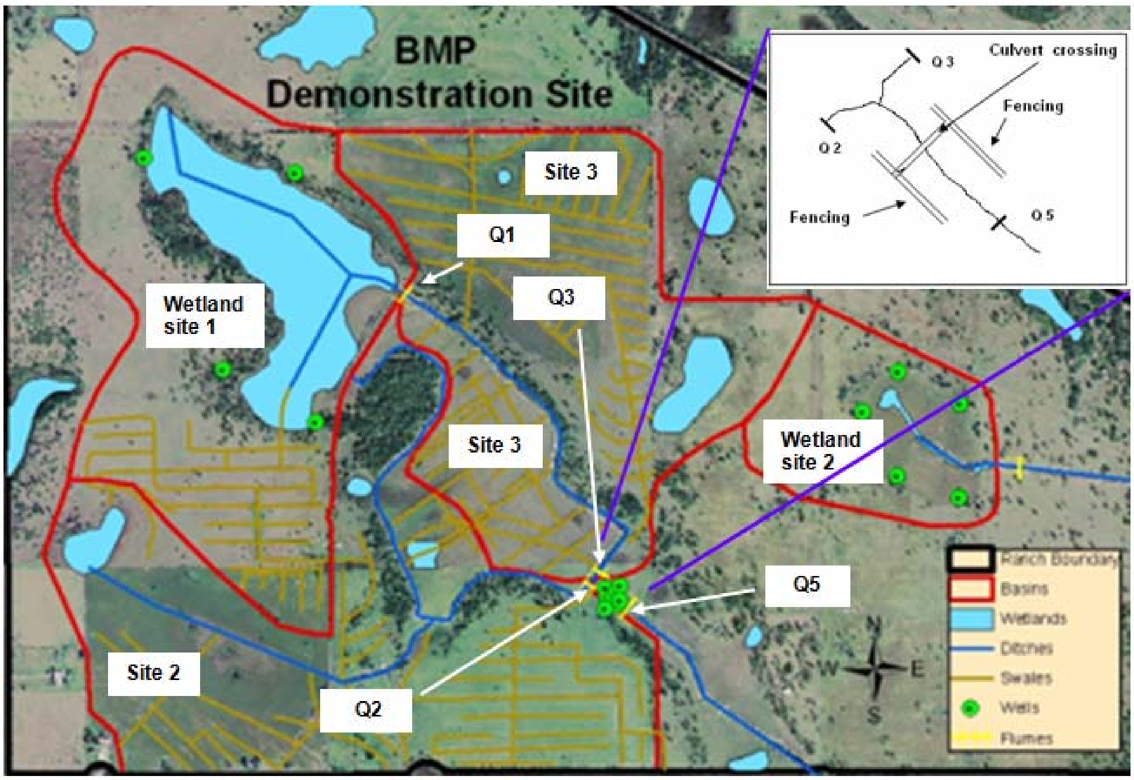

Figure 1. Ditch fencing and culvert crossing (DFCC) and wetland water retention (WWR) BMP sites at the Pelaez ranch.

Appendix A-1

Water quality effectiveness of ditch fencing and culvert crossing in the Lake Okeechobee basin using wet season (June-May) pre- BMP and post-BMP data

Abstract

Ditch fencing and culvert cattle crossing Best Management Practice (BMP) was evaluated in this study with regards to phosphorus (P) and nitrogen (N) load reductions and economic feasibility in the Lake Okeechobee (LO) basin. The BMP was implemented at a 170 m section of a drainage ditch within a ranch in the LO basin and flow and concentration (N and P) data at the upstream and downstream of the ditch were collected for one pre-BMP (June-October, 2005) and three post-BMP (June-October, 2006-2008) periods. During the pre-BMP period, downstream total P (TP) load was 20% (67.0 kg) higher than the upstream, indicating the cattle crossing to be a source of P. Downstream loads of TP in 2006 and 2008 (post-BMP periods) became 26% (14.7 kg) and 11% (85.9 kg) lower than the upstream loads, respectively indicating that the BMP reduced the P loads. The site was a sink for N for all periods except the 2007. Unusual 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 plants. Average of three post-BMP period loads showed a 10% reduction of TP loads at the downstream (251.8 kg) compared to the upstream (281.0 kg) location. To consider potential P contributions from the soil and plant, two scenarios, conservative and liberal were considered to estimate P load reductions due to the BMP. For the conservative scenario, P contribution from soil and plant was considered while for liberal it was not. Reductions in P loads for conservative and liberal scenarios were 0.35 and 0.44 kg/day, respectively. Phosphorus removal cost for the conservative scenario was $12.61 /kg of P which is considerably less than the cost of other P reduction strategies in the basin. Overall, results show that the BMP can reduce P concentration and loads from ranches without causing an adverse impact on cattle production.

Introduction

Lake Okeechobee (LO) 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 problems facing the lake (Bottcher et al., 1995; Nair et al., 2007; Tweel and Bohlen, 2008). Cow-calf operations, the single largest land use in the Lake Okeechobee watershed, are the most important source of external P loadings to the lake (Capece et al., 2007). The Florida Cattlemen's Association (FCA) in cooperation with the University of Florida (UF), Florida Department of Agriculture and Consumer Services (FDACS), Florida Department of Environmental Protection (FDEP), and South Florida Water Management District (SFWMD) has developed a variety of Best Management Practices (BMPs) to control the P discharges from the cow-calf operations (FDACS, 2008). 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) for the lake. Keeping cattle out of waterways with ditch fencing and culvert crossing within a ranch is one such BMP which is promising with regards to reducing P and Nitrogen (N) loads. The term '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 to avoid/eliminate the direct deposition of cattle feces and urine into ditches, streams, creeks, or rivers.

Grazing cattle in pastures with unfenced streams and drainage ditches contributes significant loads of P and N to surface waters (Byers et al., 2005). Several studies (Sheffield et al., 1997; Line et al., 2000; Galeone et al., 2006) have been conducted to study the effects of cattle exclusion from waterways on water quality. Sheffield et al. (1997) evaluated alternative water source as an option to reduce cattle entry to a stream in Virginia by comparing the pre-BMP (August, 1994 - April, 1995) and post-BMP (April, 1995 - October, 1995) monitoring data. They concluded that providing off-stream water source reduced the in-stream cattle traffic which eventually reduced Total Suspended Solids (TSS), Total Nitrogen (TN), and Total Phosphorus (TP) concentrations by 90, 54, and 81%, respectively. Line et al. (2000) evaluated effects of fencing a 335 m long riparian corridor in North Carolina by comparing the pre-BMP (81 weeks) and post-BMP (137 weeks) P and N loads and found that post-BMP mean weekly loadings of Total Kjeldahl Nitrogen (TKN), TP, TSS, and Total Solids (TS) were reduced by 79, 76, 82, and 83%, respectively. A study was conducted by Meals (2001) to evaluate the combined effectiveness of livestock exclusion along with stream bank restoration, and riparian zone protection in Vermont. Results indicated reductions in TP concentration (25%) and load (42%) in the treated watershed compared to a control watershed. In a paired-watershed study, Galeone et al. (2006) evaluated the effects of stream bank fencing in Pennsylvania and found reductions in Nitrate (NO_3-N) , Nitrite (NO_2-N), Ammonia (NH_3-N) , TKN, and TP loads.

In a review of cattle exclusion studies, Dillaha (2007) noted several limitations in some studies. For example, in the study conducted by Sheffield et al., (1997), the pre-BMP period included winter, while the post-BMP period included summer. The post-BMP rainfall was 54% higher than the pre-BMP rainfall which might have introduced bias in the BMP results. For the Galeone et al. (2006) study, Dillaha (2007) noted that several factors other than the BMP might have resulted in reductions in flows and P concentrations: P and N fertilizer applications decreased by 27-30% during the post-BMP period; number of livestock decreased by 50% during the post-BMP period; and the pre-BMP rainfall was 11% higher than post-BMP rainfall which may have resulted in reduced post-BMP stream flows. In the Line et al. (2000) study, there were four major storms during the pre-BMP period while no similar storms occurred during the post-BMP period. The larger storms during the pre-BMP period were expected to produce larger nutrient loads. Variability in factors such as climate, fertilizer, and cattle density observed in these studies has limited the ability to attribute the water quality improvements to cattle exclusion.

In light of the above limitations of cattle exclusion studies and regional differences in weather, soil, topographic, and hydrologic factors, conclusions drawn from one soil-hydrologic region may not be applicable to another region. For example, soils, topography, 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 (NRCS, 2003). 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 the relatively flat topography present in Florida, flow rates in the ditches are typically slow except during major storm events. Additionally, the flows in the ditches mainly occur during the wet period (June-October) which receives 70% of the total annual rainfall in this region (Shukla et al., 2010). Given such differences in soils, climate, hydrology, and landscape, cattle exclusion studies conducted elsewhere in the U.S. have limited applicability to south Florida. Moreover, no study till date has incorporated economic analyses as part of their cattle exclusion BMP evaluation. Economic analysis is an important component of the BMP effectiveness for facilitating and planning a BMP implementation at the basin scale. The goal of this study was to evaluate the effectiveness of DFCC BMP in a ranch located within the LO basin in south Florida with regards to P and N loadings and economic effects.

Materials and methods

Site description

The study site is a commercial 250 ha cow-calf ranch located in south-west Okeechobee County. Soils in the area are typically poorly drained and highly sandy (NRCS, 2003). The ranch is dominated by improved pastures laced with shallow ditches for drainage. The three major forage types in the ranch are Bahia (*Paspalum notatum)*, Floralta (*Hemarthria altissima*), and Stargrass (*Cynodon nlemfuensis*). 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 the lake. The ranch can be divided into two sub-watersheds, termed as 1 and 2. Discharges from the two sub-watersheds (Flume 1 and Flume 2) combine and flow through a main drainage ditch (length = 170 m) before exiting the ranch (Flume 3) (). The DFCC BMP was evaluated at the main drainage ditch.

Ditch fencing and culvert crossing BMP

A pre- and post-BMP monitoring design was used to evaluate the DFCC BMP at the main drainage ditch (Figure A-1). The BMP involved installation of a culvert crossing and fencing in January 2006 to route the cattle over the ditch instead of through it (Figure A-1). Before BMP implementation, there was a cattle crossing pathway almost midway in the ditch, which over the years resulted in erosion of the ditch bank that formed an elevated section of land in the middle of the ditch. The average depth of the drainage ditch is 1.40 m which results in the availability of water in the ditch for most of the year due to shallow groundwater in the region. The ditch served as the drinking water source as well as a temperature control area for the cattle.

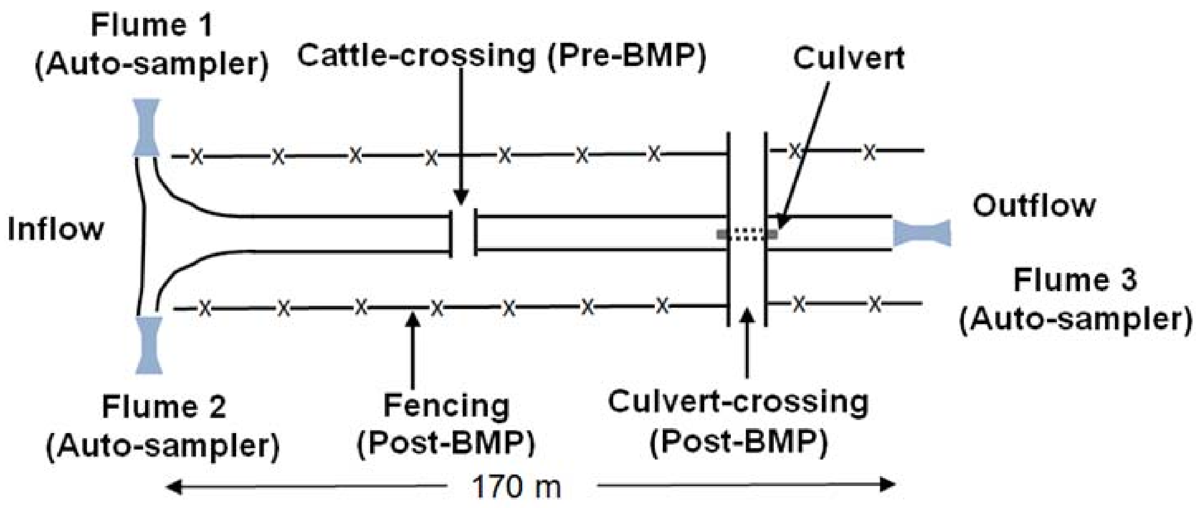

Figure A-1. Pre-BMP and post-BMP cattle crossing and flow and water quality monitoring at the study site.

Hydrologic and water quality monitoring

Three trapezoidal flumes (Flumes 1, 2, and 3) equipped with digital shaft encoders (to monitor the hydraulic head), were installed at the inflow (Flumes 1 and 2) and outflow (ranch outlet, Flume 3) locations of the selected ditch to measure the flows until August 2007 and afterwards using Acoustic Doppler Velocimeters (ADVs). By comparing P and N loadings at inflow and outflow, P and N addition/removal at the BMP site were quantified. The wet season (June-October) of 2005 was 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).

Auto-samplers were installed at each flume site to collect surface water samples to determine TP, TKN, NH_3-N, and NO_x-N (nitrate-nitrite) concentrations. Nutrient concentration data were combined with the flows to estimate the N and P loads using the linear interpolation (between two consecutive samples) method (Kronvang and Bruhn, 1996). A weather station was installed near Flume 2 to collect rainfall and other climatic data.

Land use management

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 rates, and cattle stocking rate. Equal amounts (392 kg/ha) of NPK fertilizer (20-5-5) were applied at the site each year during the study period. The average cattle weight is about 527 kg. The age of the cattle ranges from less than a year to over two years. The cattle density is about 1.5 cows per ha of pasture which is typical for beef-cattle ranches in Florida (Tweel and Bohlen, 2008). The cattle are grazed on rotational basis in the pastures within the ranch. Above-ground water tanks were used for providing drinking water to cattle. There were more cattle present at the study site during post-BMP periods (average 66 cattle) as compared to the pre-BMP period (59 cattle). Therefore, the estimates of the reductions in P and N loadings in this study should be conservative.

Comparison of time-specific phosphorus and nitrogen concentrations

To evaluate the BMP effects, both concentrations and loads for the pre-BMP and post-BMP periods were compared. To compare the concentrations of P and N species between the inflow and outflow points, 30 water samples collected from each of the three flume sites (Flume 1, Flume 2, and Flume 3) at similar sampling times (time-specific samples) were selected and the respective P and N concentrations were compared.

Statistical analyses

Two types of statistical analyses, using SAS v9.2 (SAS Institute, Cary, NC), were conducted to evaluate the BMP effectiveness. First, TP and TN (TKN+NO_3-N) loads for the pre- and post-BMP periods were compared using an Analysis of Variance (ANOVA) model. Second, the pre- and post-BMP time-specific TP and TN concentrations were compared using paired t tests.

TP and TN loads for the pre- and post-BMP periods were compared to model the difference in total daily loads (inflow – outflow) at the BMP site as a function of year and month. We modeled difference to capture the nutrient contribution from cattle traffic or other sources. The months compared were July, August, September and October in 2005 (pre-BMP) versus 2006 and 2008 (post-BMP). There was insufficient load data for June at least in one of these years. Data for 2007 were excluded from the analysis because it was a drought year. A two-factor model was run to obtain mean TP and TN loads at the BMP site for each month and year combination and the residuals were checked to determine if the assumptions of the model were met. There was no evidence of auto-correlation in the daily data (p-value ≈ 1), so subsequent analyses assumed that the data were independent. The residuals from the initial model had a strongly skewed distribution and therefore, natural logarithms of the data were used for further analyses. To determine the effect of the BMP, the mean difference (inflow – outflow) in monthly load for each post-BMP year was compared to those for the pre-BMP year. Dunnett's adjustment was used to control the experiment-wise error rate.

Economic analysis

The DFCC BMP was evaluated for capital costs of implementation and cost-effectiveness of nutrient (P and N) removal. Expenses for construction of structural improvements, obtained from the rancher cooperator, included materials, labor, and contract services. 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 ditch section, along with provision of above-ground drinking water tanks for the cattle. Capital costs were amortized over assumed useful life of 20 years at 5% annual interest rate. Production records were provided by the ranch owner for animal stocking, forage fertilization, and supplemental feeding. The BMP was assessed for its economic effectiveness for N and P removal. Potential impacts of the BMP on cattle health and performance and other non-market values were also assessed through interviews with ranch personnel.

Results and discussion

Phosphorus and nitrogen loads and concentrations

The groundwater contributions of P and N to the ditch section, estimated using the Dupuit equation, were used in conjunction with inflow and outflow loads to determine the loads contributed to or removed from the BMP site. However, the average groundwater TP (pre-BMP = 0.8 kg and post-BMP = 0.1 kg) and TN (pre-BMP = 7.9 kg and post-BMP = 2.5 kg) contributions were small compared to the inflow and outflow loads and therefore are not likely to mask the BMP effects. The TP loads at outflow were 20% higher compared to inflow during the pre-BMP period (Table A-1) indicating addition of TP due to the cattle traffic and/or other sources. The organic nitrogen (ON = TKN – NH_3-N) and TN loads at outflow were 17 and 15% lower compared to inflow. During post-BMP1 and 3 periods, TP loads at outflow became 26 and 11% lower compared to inflow indicating the effectiveness of the BMP in reducing the TP loads. For post-BMP1 and 3 periods, the outflow loads of ON and TN were lower than inflow indicating no contribution of organic and inorganic N but rather a reduction at the ditch section. Reductions in TN loads after passing through the ditch section could be attributed to uptake by aquatic vegetation and denitrification at the BMP site. Reduction in TP loads can mainly be attributed to plant uptake and soil adsorption. For post-BMP2, P and N loadings at outflow were higher than the inflow loadings indicating addition of P and N at the BMP site. South Florida experienced severe drought conditions in 2007 (post-BMP2 period), resulting in reduced flow and potential release of P and N from dead aquatic vegetation which increased the outflow P and N loads. This masking effect limited our ability to attribute the loading changes to the BMP during post-BMP2 period. However, the average inflow and outflow TP loadings for the three post-BMP periods were 281.0 and 251.8 kg, respectively, indicating a long-term average reduction of 10% for TP at outflow due to the removal of cattle traffic. Average inflow and outflow TN loadings were 583.6 and 497.6 kg, respectively for the three post-BMP periods with 15% reduction of TN at outflow. The flow-weighted concentration (Table A-1) showed similar trends as those shown for the P and N loadings.

There was high variability in rainfall for the monitoring periods (June-October: 2005-2008) at the study site. Pre-BMP and post-BMP3 had similar rainfall amounts (1116 and 1124 mm, respectively) that were higher than those for the post-BMP1 and 2 periods (633 and 755 mm, respectively). Similar rainfall amounts for pre-BMP and post-BMP3 periods made these two periods suitable for BMP evaluation. 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 a historically low water level in July 2007 (SFWMD, 2009). The drought conditions resulted in no water in the ditch, low moisture content in the ditch soil, and decomposition of aquatic vegetation (macrophytes). Phosphorus and N are released to the water when the macrophytes decompose (Reddy et al., 1995; Chimney and Pietro, 2006). Additions of TP (13.0 kg), and TN (2.8 kg) at the BMP site during post-BMP2 period can therefore be attributed to release from decomposed vegetation. Small loads generated due to low flow and relatively high contribution of P and N by the macrophytes potentially masked any reductions in P and N due to the BMP during the post-BMP2 period.

Table A-1. **Inflow and outflow organic nitrogen (ON), total nitrogen (TN), and total phosphorus (TP) loads (kg), flow weighted concentrations (FWC, mg/l), time-specific mean concentrations (TMC, mg/l), and % change in respective values along with the p values (t-test for TMC at inflow and outflow) for one pre (June 2005-October 2005) - and three post-BMP (post-BMP1 - June 2006 to October 2006, post-BMP2 - June 2007 to October 2007, and post-BMP3 - June 2008 to October 2008) periods.**

P and N Loads and Conc.		pre-BMP			post-BMP1			post-BMP2			post-BMP3		
		In	Out	% Change (p value)	In	Out	% Change (p value)	In	Out	% Change (p value)	In	Out	% Change (p value)
ON	Loads (kg)	1021.0	852.1	-16.5	83.4	75.1	-10.0	32.8	36.2	10.4	1312.8	1115.6	-15.0
	FWC (mg/l)	2.25	1.92	-14.7	2.12	1.94	-8.5	2.33	2.53	8.6	2.54	2.23	-12.2
TN	Loads (kg)	1387.1	1179.3	-15.0	105.2	92.7	-11.9	39.4	42.2	7.1	1606.1	1357.9	-15.5
	FWC (mg/l)	3.06	2.66	-13.1	2.67	2.39	-10.5	2.80	2.95	5.4	3.11	2.72	-12.5
	TMC (mg/l)	3.50	3.55	1.4 (0.928)	3.01	3.25	8.0 (0.296)	3.88	4.01	3.4 (0.670)	4.24	3.85	-9.2 (0.366)
TP	Loads (kg)	334.1	401.1	20.1	56.6	41.9	-26.0	19.2	32.2	67.7	767.3	681.4	-11.2
	FWC (mg/l)	0.74	0.91	23.0	1.44	1.08	-25.0	1.36	2.25	65.4	1.49	1.36	-8.7
	TMC (mg/l)	0.54	0.66	22.2 (0.026)	2.28	1.77	-22.4 (0.001)	1.51	1.86	23.2 (0.038)	1.69	1.56	-7.7 (0.038)

During pre-BMP, post-BMP1, and post-BMP3 periods, there were reductions in N loadings at outflow compared to inflow. Reddy et al. (1989 and 1995) observed that a major portion of N added to streams was lost from the system by denitrification. Gordon et al. (1986) reported highest denitrification in south Florida between temperatures of 26 to 32 ^{o}C. This temperature range existed at the ranch for most of the daytime hours during the study period. South Florida environment is conducive to denitrification due to saturated soils (water table close to the surface) and high temperature during summer (Martin and Reddy; 1997). Denitrification is the likely cause of considerable N losses during the pre-BMP, post-BMP1 and post-BMP3 periods. Low ditch water levels during post-BMP2 (2007) might have reduced denitrification and increased N mineralization, resulting in net addition of N at the BMP site (Table A-1).

During pre-BMP period, mean TP concentration for the time-specific samples at outflow was higher than that at inflow (Table 1). For post-BMP1 and 3 periods, the trends were reversed (outflow < inflow) indicating elimination of P input from cattle traffic at the BMP site. During post-BMP2, time-specific mean TP concentration at outflow was higher than that at inflow, likely due to the contribution from the macrophytes and in-ditch soil (Table A-1). There were increases in TN concentrations at outflow in the time-specific samples for all but post-BMP3 period.

Statistical analyses

ANOVA models for phosphorus and nitrogen loadings

The ANOVA model suggested that there was significant reduction in TP loads at the BMP site in July ($p < 0.002$), August ($p < 0.001$) and September ($p < 0.001$) during the post-BMP3 period compared to the pre-BMP period. For the post-BMP1 period, the model showed significant TP load reduction ($p = 0.001$) only in the month of September. Conversely, the TP load trend reversed for July during the post-BMP1 period. Almost no flow and TP loads during the first two weeks of July of the post-BMP1 (inflow – outflow = < 0.01 kg) period compared to the pre-BMP period (inflow – outflow = -26.56 kg) resulted in introducing a bias in the load comparison. For TN loads, significant reductions ($p = 0.002$) were observed only in August during the post-BMP3 period compared to the pre-BMP period. Overall, considering similar rainfall for the pre-BMP and post-BMP3 periods and significant reductions in TP contribution during the post-BMP3 period (three of the four months compared), we conclude that the BMP was effective in reducing the TP loads.

Phosphorus and nitrogen concentrations

The results of the paired t-test for the time-specific samples (Table A-1) indicated that TP concentrations at outflow were significantly higher ($p = 0.026$) than inflow during the pre-BMP period. In contrast, outflow TP concentrations were significantly lower than inflow during the post-BMP1($p = 0.001$) and 3 periods ($p = 0.038$). For post-BMP2, TP concentrations at outflow were significantly higher than inflow. For TN, no statistically significant differences were observed in any of the four periods (Table A-1). Overall, the results indicated that the BMP decreased the TP concentrations in the ditch.

Scenarios of post-BMP phosphorus reductions

Due to the variability in rainfall and flow, P and N loadings at inflow and outflow varied greatly for the three post-BMP periods. During post-BMP periods 1 and 3, the P load data showed no addition of P at the ditch section indicating the effectiveness of the BMP in eliminating P loads due to cattle traffic. The addition of P at the BMP site during post-BMP2 period was likely due to the contribution of P by the decomposed aquatic vegetation. During the pre-BMP period, there was an addition of 67.0 kg of P (outflow load of 401.1 minus inflow load of 334.1, Table A-1) at the BMP site. There was a net release of 13 kg P (outflow load of 32.2 minus inflow load of 19.2, Table A-1) from the ditch section during the post-BMP2 period. Although this release was mainly influenced by the drought conditions during 2007, a reduced level of P release from ditch may occur during average rainfall years due to dry conditions that exist prior to the wet season (June-October). To consider this likely source of P addition during average rainfall years, two scenarios of P reductions, moderate and conservative, were considered. These two scenarios were formulated based on the likelihood of P release from the decomposed vegetation at the ditch section. Although the 13 kg of P addition occurred during the drought year of 2007, for the conservative scenario, it was assumed that this P contribution occurred each year including the pre-BMP period. Therefore, the P contribution from cattle traffic was adjusted to 54.0 kg of P (67.0 kg minus 13.0 kg) for the conservative scenario. For the liberal scenario, it was assumed that all of the 67.0 kg of P addition observed during the pre-BMP period was derived from the cattle traffic. Since the DFCC prevented the cattle traffic through the ditch, P reductions due to the BMP for the two scenarios were 67.0 (liberal) and 54.0 (conservative) kg. These reductions translated to 0.44 and 0.35 kg/day of P load reductions for the liberal and conservative scenarios, respectively.

Economic analysis

Capital costs for the DFCC BMP totaled $20,245.00, and amortized annual cost of structural improvements was $1625.0 (for 20 years @ 5% interest). 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 ($282.0), as reported by Cattle Fax (2006). Therefore, it is expected that this BMP would not be a financial burden for the ranch owner to implement. Ranch manager indicated that the BMP resulted in no changes in ranch operations management, overhead expenses, or general herd health. Based on the amortized cost per year and the liberal (67 kg) and conservative (54 kg) scenarios' P reductions, the average costs of P removal were $10.2 and $12.6 per kg of P, respectively. This cost is more than order of magnitude less than storm water treatment areas ($442.0 to $1109.0 per kg) for treating P before discharging to the Florida Everglades (Sano et al., 2005).

Many of the cattle exclusion studies in the past experienced high uncertainty associated with the BMP effectiveness analyses caused by the variability in one or more factors: rainfall, flow, seasons/months of the year considered for pre- and post-BMP periods, fertilizer application rate, and livestock stocking rate between pre- and post-BMP periods. The variability combined with measurement errors can introduce bias in the BMP effectiveness results. Some of these factors (e.g. differences in rainfall and flow) are beyond human control and were also present in this study. But there are some other factors (e.g. seasons/months of the year used for pre- and post-BMP periods, fertilizer application, livestock stocking rate) that can be controlled. It is desirable that there is minimum variability in these factors between pre- and post-BMP periods to minimize the possible masking effect on the BMP results. In this study, the fertilizer application rates were same for pre- and post-BMP periods. Data for the same period (June-October) in different years were used for comparing pre- and post-BMP nutrient loads and concentrations. Furthermore, pre-BMP1 and post-BMP3 periods 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.

Summary and conclusions

The DFCC BMP was evaluated by comparing P and N concentrations and loadings between inflow and outflow in a ditch section for one pre- and three post-BMP periods. Reductions in P concentration and load were observed during the two post-BMP periods (post-BMP1 and post-BMP3) compared to the pre-BMP period. During post-BMP2 period, extreme dry conditions resulted in the release of P from the BMP site due to release of P from plant and soil. Results for BMP effects on N discharges were mixed mainly due to the extensive denitrification. Nitrogen is not the limiting nutrient for Lake Okeechobee and is therefore, not the nutrient of main concern. Due to the variability in rainfall and its effects on P dynamics and loadings, liberal and conservative scenarios were considered for estimating the P load reductions due to the BMP. An economic analysis of the BMP was conducted which considered the costs of per unit (kg) of P removal for the two scenarios. Following conclusions can be drawn from this study:

1) DFCC BMP evaluated in this study reduced the P loadings by 0.35 and 0.44 kg/day, for conservative and liberal scenarios, respectively.

2) Given the climatic variability and uncertainty in measuring BMP effects, conservative estimates of P load reduction can be used for basin-scale BMP adoption.
3) The BMP did not cause any adverse impact on cattle production. The P removal cost for the conservative scenario was $12.61/kg of P which is much less than the cost of other P reduction strategies such as storm water treatment areas ($442.00 to $1109.00 per kg).
4) DFCC seems to be an economically feasible BMP that can be implemented to reduce P loads from cattle ranches in the Lake Okeechobee basin, especially if it is supported by the state and federal cost-share funding.

This is the first cattle exclusion BMP study in the U.S. that has been evaluated with regards to water quality as well as economic feasibility. Similar evaluations are needed for other ranchland BMPs. Economic feasibility of DFCC BMP can then be compared with other BMPs to better design the P load reduction strategies for the watersheds containing large acreage of ranchland.

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Appendix A-2

Water quality effectiveness of ditch fencing and culvert crossing in the Lake Okeechobee basin using annual (June-May) pre-BMP and post-BMP data

In addition to evaluating the effectiveness of the ditch fencing and culvert crossing **(**DFCC) by comparing one wet season (June-Oct) as pre-BMP period with three wet seasons as the post-BMP periods, the DFCC was also evaluated using annual data. Water quality data for two pre-BMP years (June 05-May 06: pre-BMP1, and June 09-May 10: pre-BMP2) were compared with three post-BMP years (June 06-May 07: post-BMP1 June, 07-May 08: post-BMP2, and June 2008-May 2009: post-BMP3) to quantify BMP effects.

During pre-BMP1 period, outflow P load (526 kg) was 23% higher than the inflow P load (426 kg). Also pre-BMP2 outflow P load (552 kg) was 25% higher than the inflow P load (443 kg). On the average, there was 24 % increase in P load at outflow (539 kg) compared to the inflow (435 kg) for the two pre-BMP periods.

During post-BMP1 period, there was a 31% reduction in P load at downstream (37 kg) compared to the upstream (54 kg). During post-BMP3 period there was 11% reduction in P load at downstream (733 kg) compared to upstream (823 kg). P load increased at downstream (33 kg) compared to the upstream (24 kg) by 38% during post-BMP2 period. The year 2007 was a drought year in south Florida resulting in lower than average flow in the ditches. Most of this P addition at the BMP site was due to the contribution of P from the decomposition of macrophytes at the ditch. On the average, there was 11 % reduction in P load at downstream (268 kg) compared to the upstream (300 kg) for the three post-BMP periods. The BMP was effective in reducing P loads at the DFCC site.

There were reductions in N loads at the BMP site during pre-BMP1 (10%), post-BMP1 (13%), and post-BMP3 (11%) periods while there were additions of N loads at the BMP during pre-BMP2 (17%), and post-BMP2 (2%) periods. Reductions in N loads after passing through the ditch section could be attributed to uptake of N by aquatic vegetation and denitrification at the BMP site. Average inflow N load was 1584 kg against an outflow N load of 1624 kg for the two pre-BMP periods with 3% load increase at outflow. Average inflow N load for the three post-BMP periods was 682 kg against an outflow N load of 609 kg with 11% N load reduction at outflow. The effect of the BMP on N loads could not be clearly observed due to excessive denitrification at the site. Table A-2 shows the pre- and post-BMP TP and TN loads at the DFCC BMP site.

Table A-2. Pre- and post-BMP Total Phosphorus (TP) and Total Nitrogen (TN) loadings at the ditch fencing and culvert crossing (DFCC) site in the beef cattle ranch.

Period	Rainfall (cm)	Total Nitrogen (kg)		Total Phosphorus (kg)	
		Inflow	Outflow	Inflow	Outflow
pre-BMP1	147	1733	1568	426	526
pre-BMP2	160	1434	1679	443	552
post-BMP1	91	103	90	54	37
post-BMP2	109	41	42	24	33
post-BMP3	157	1902	1696	823	733

Scenarios of Phosphorus reduction at the BMP site

During the two pre-BMP periods, there was an average addition of 104.5 kg of P (average outflow load of 539.0 kg minus average inflow load of 434.5 kg) at the BMP site. There was a net release of 9.0 kg P (outflow load of 33.0 kg minus inflow load of 24.0 kg) from the ditch section during the post-BMP2 period. Although this release was mainly influenced by the drought conditions during 2007, a reduced level of P release from ditch may occur during average rainfall years due to dry conditions that exist prior to the wet season (June-October). To consider this likely source of P addition during average rainfall years, two scenarios of P reductions, moderate and conservative, were considered. These two scenarios were formulated based on the likelihood of P release from the decomposed vegetation at the ditch section. Although the 9.0 kg of P addition occurred during the drought year of 2007, for the conservative scenario, it was assumed that this P contribution occurred each year including the pre-BMP periods. Therefore, the P contribution from cattle traffic was adjusted to 95.5 kg of P (104.5 kg minus 9.0 kg) for the conservative scenario. For the liberal scenario, it was assumed that all of the 104.5 kg of P addition observed during the pre-BMP period was derived from the cattle traffic. Since the DFCC prevented the cattle traffic through the ditch, P reductions due to the BMP for the two scenarios were 104.5 (liberal) and 95.5 kg (conservative). These reductions translated to 0.29 and 0.26 kg/day of P load reductions for the liberal and conservative scenarios, respectively.

Economic analysis of the DFCC BMP effectiveness

Capital costs for the DFCC BMP totaled $20,245.00, and amortized annual cost of structural improvements was $1625.0 (for 20 years @ 5% interest). Capital costs per hectare treated were $76.6, and annual capital costs per animal unit averaged $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 ($282.0), as reported by Cattle Fax (2006). Therefore, it is expected that this BMP would not be a financial burden for the ranch owner to implement. Ranch manager indicated that the BMP resulted in no changes in ranch operations management, overhead expenses, or general herd health. Based on the amortized cost per year and the liberal (104.5 kg) and conservative (95.5 kg) scenarios' P reductions, the average costs of P removal were $15.55 (liberal) and $17.02 (conservative) per kg of P, respectively. This cost is more than order of magnitude less than storm water treatment areas ($442.0 to $1109.0 per kg) for treating P before discharging to the Florida Everglades (Sano et al., 2005).

Appendix A-3

Flow and nutrients contributions from groundwater to a drainage ditch in a beef cattle ranch in the Lake Okeechobee Basin, Florida

Lake Okeechobee located in south-central Florida is the second largest freshwater lake contained wholly within the continental 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. The pasture accounts for a large fraction of P contribution in the Lake Okeechobee basin (Hiscock et al., 2003). To lower the water table in the ranches for pasture development, the drainage ditches are used. Drainage accounts for major flows and nutrients to Lake Okeechobee. The ditches receive P and nitrogen (N) loads primarily from groundwater flow and surface runoff. There have been many studies (e.g. Boggess et al., 1995; Hiscock et al., 2003) conducted in the Lake Okeechobee basin on the estimation of P loadings by surface water, but quantification of groundwater flow, and P and N loads to the ditches which contribute flow and nutrients to Lake Okeechobee has not been reported in the literature. In central and south Florida, excessive infiltration in the sandy soil reduces the runoff potential (Shukla et al., 2010), and therefore, groundwater is a prominent flow component. The information on the fraction of total P and N loadings contributed by groundwater is needed for developing nutrient loading control programs for the lake.

The objective of this study was to estimate the flow, P and N contributions from groundwater to a typical drainage ditch located in a beef cattle ranch. The groundwater flow was estimated using the Dupuit equation. The Dupuit equation is extensively used to calculate groundwater flow in unconfined aquifers (Delleur, 2007; Tsubo et al., 2007; Rosenberry and LaBaugh, 2008). The assumptions (Dupuit-Forchheimer assumptions) for groundwater flow estimations using the Dupuit equation 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).

These assumptions are valid for mild sloping land and shallow water depth (Delleur, 2007; Tsubo et al., 2007). In essence, these assumptions neglect the vertical flow components (Freeze and Cherry, 1979).

Materials and Methods

Study area

The study site is located in a 250.7 ha beef cattle ranch (Okeechobee County) in the Lake Okeechobee watershed which has been identified as the key area that contributes to the eutrophication of Lake Okeechobee (Davis and Marshall, 1975; Federico et al., 1981). The ranch is dominated by improved pastures with shallow ditches for drainage. A ditch section (170 m) within the ranch was chosen for this study. The maximum depth from the land surface to the ditch bottom within the study site is 1.40 m with average depth of 1.28 m. The ditch section is located near the main water outlet from the ranch watershed and conveys the drainage water from the ranch to downstream. There was no surface runoff contribution to the ditch section from the groundwater contributing areas from both sides (east and west) of the ditch section (fig. 1). This region is under surficial aquifer system. The thickness of the surficial aquifer is typically less than 15 m (FDEP, 2011).

The two major forage types near the ditch section are Floralta and Stargrass. Each year, equal amounts of NPK fertilizer (20-5-5) were applied at the study site in September during 2005-2008 at the rate of 392 kg/ha. Soil at the study site is predominantly Basinger fine sand. Following the USDA soil taxonomic system, this soil is siliceous, hyperthermic spodic psammaquents (Basinger series) (NRCS, 2003). In general, the soils in this region come under flatwood soils which dominate the Florida landscape (Bottcher et al., 1999). Flatwood soils are typically spodosols, which are nearly level, poorly drained, sandy soils that have an organic Bh horizon. The saturated hydraulic conductivity (K_{sat}) values for Basinger fine sand for this study were obtained from the Florida Soil Characterization Data Retrieval System (FSCDRS) (http://flsoils.ifas.ufl.edu) for the Okeechobee-Everglades watershed. The FSCDRS provides access to a comprehensive soil dataset including soil profile descriptions, soil taxonomic information, and physical and chemical soil properties. To address the uncertainty caused due to the variability in K_{sat} values, the groundwater flow, and N and P loads were quantified using the highest and lowest K_{sat} values for Basinger fine sand reported in the FSCDRS. Depth weighted average K_{sat} (equivalent K_{sat}) values (Das, 2009) calculated from K_{sat} values and the corresponding soil profile depths varied from 3.84 x 10^{-5} (high value) to 8.38x 10^{-5} m/s (low value). These two extreme K_{sat} values were used to estimate a range (high and low values) of flow, and P and N loadings data. Table A-3 shows the highest and lowest K_{sat} values with the corresponding soil depths for Basinger fine sand soil for the Lake Okeechobee-Everglades watershed. Flow and nutrient loadings were estimated for four years: June 2005-May 2006 (year 1), June 2006-May 2007 (year 2), June 2007-May 2008 (year 3), and June 2008-May 2009 (year 4). The average annual rainfall in the Lake Okeechobee basin is 113 cm (wet season, June-October = 72 cm, dry season, November-May = 41 cm) (Ali and Abtew, 1999; Guardo, 1999).

Table A-3. The highest and lowest saturated hydraulic conductivity (K_{sat}) values for Basinger fine sand soil in the Lake Okeechobee-Everglades watershed. (Procured from Florida Soil Characterization Data Retrieval System at http://flsoils.ifas.ufl.edu)

Highest K_{sat}		Lowest K_{sat}	
Soil depth (cm)	Ksat (m/s)	Soil depth (cm)	Ksat (m/s)
0-15	6.82E-05	0-10	1.11E-04
15-41	6.57E-05	10-36	1.12E-04
41-53	5.65E-05	36-64	1.13E-04
53-76	3.35E-05	64-74	4.56E-05
76-132	1.77E-05	74-91	3.75E-05
132-140	2.59E-05	91-112	5.19E-05

Hydrologic and water quality data

Ten wells were installed near the ditch section which flow in the north-south direction (Figure A-2). Six wells (well no. A, B, C, F, G, and H) were installed along a main transect vertical to the ditch section with three wells on each groundwater contributing area (east and west). The other four wells (wells I, D, J, E) were installed along two transects (two wells in each transect) parallel to the main transect. The wells were constructed with 5.08 cm, schedule 40 PVC tubes and the boreholes were backfilled with sand. The wells were 3.05 m in length and screened between 1.2 to 2.1 m from the ground surface. Pressure transducers (Solinst Canada Ltd., LT model F15/M5, accuracy: ± 0.3 cm) were installed in the six wells along the main transect to monitor heads in the wells at 15-min frequency. Rainfall data were collected from a weather station installed in the ranch.

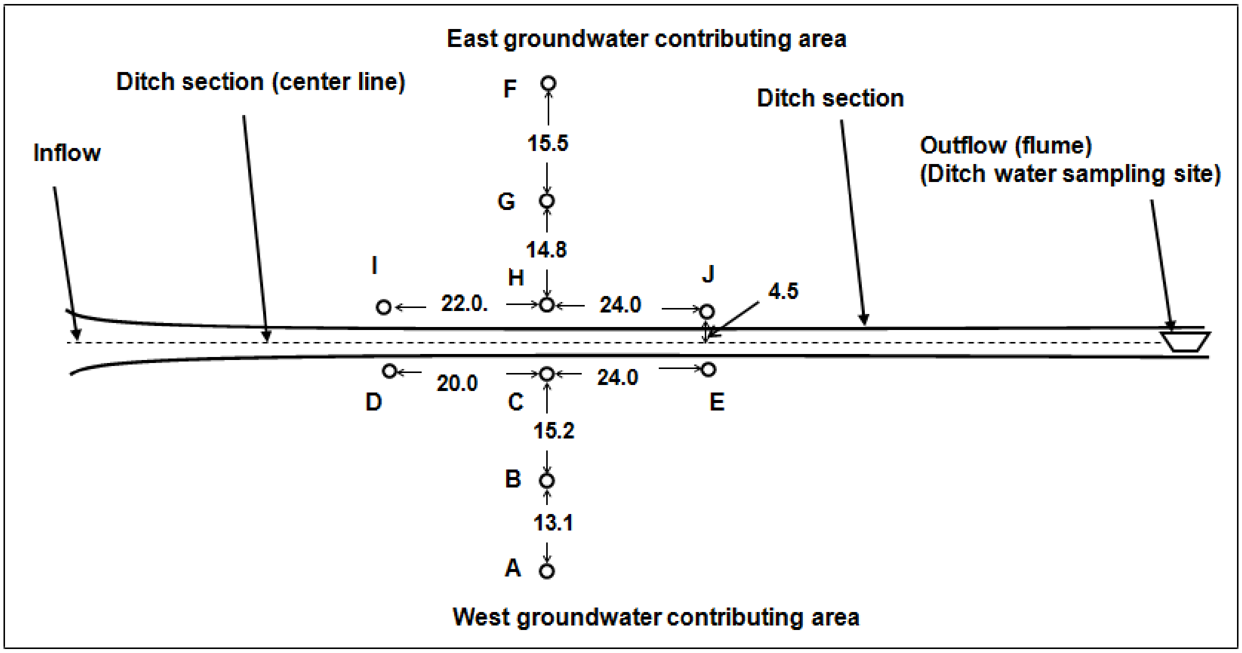

Figure A-2. Experimental layout for the study. All distances are in meters.

Water samples were collected from all ten wells on a monthly basis during June 2005-May 2009 period. Ditch water samples were collected using an auto-sampler at a flume located at the downstream end of the 170 m ditch section (Figure A-2). To collect water samples from the ditch, a head based sampling strategy was used. The auto-sampler collected a water sample whenever there was a change of 5 cm ditch head. Grab samples were also collected once a week, subject to the availability of water in the ditch. Water samples were analyzed for Total Phosphorus (TP), Total Kjeldahl Nitrogen (TKN), Ammonia (NH_3-N), and Nitrate (NO_3-N). Total Nitrogen (TN) was determined by summing TKN and NO_3-N.

Groundwater flow

The Dupuit equation was used to estimate the groundwater flow contributed by per unit length of the ditch section. The horizontal groundwater flux into the ditch section were calculated at the main transect using the heads at the farthest wells (wells A and F) and the ditch. The Dupuit equation is given by $q = 0.5 \times K_{sat} \times \left(\left(h_1^2 - h_2^2\right)/ L\right)$ where q = flow per unit ditch length (m^2/s), K_s = saturated hydraulic conductivity (m/s), h_1 = head at the well (m), h_2 = head at the ditch (m), and L = distance between the well and the ditch (m). The Dupuit equation does not take the vertical flow component into account. The vertical flow component of the groundwater flow was assumed to be negligible and was therefore not considered in this study. At the study site, a clay-rich confining layer (hard pan) exists approximately at the ditch bottom level. In addition, due to the deposition of organic matter and finer sediments at the ditch bottom over the years, the ditch bottom has lower hydraulic conductivity. Clay and organic matter were observed approximately at 1.5 m below ground surface while installing the groundwater wells. This justifies the use of Dupuit equation to estimate groundwater flow at this site. Similar approach has been used for hydrologic modeling studies in south Florida. Min et al. (2010) observed a clay-rich confining layer in their study site (a beef cattle ranch in Okeechobee County) in Basinger series soils. Due to the presence of clay and organic matter rich sub-surface horizons, they used modified Dupuit's equation to estimate groundwater flow and neglected the vertical groundwater flow component. Ouyang (2009) assigned a no-flow boundary at the bottom face in the model domain (MODFLOW/MT3DMS model) due to the presence of a shallow hard pan layer to estimate groundwater flow in flatwoods soils in Florida.

The head data from the six wells along the main transect and the ditch water level showed a gradual decrease in head from wells F and A (farthest wells) towards the ditch during storms. Moreover, there is no other ditch in close vicinity to intercept the seepage. Therefore, it is reasonable to assume that the flow was orthogonal to the ditch and there exists no groundwater-divide between the farthest well and the ditch. It should be noted that the primary purpose of the ditches on the ranchlands of south Florida is to drain the shallow groundwater to make the land suitable for agricultural production (Bohlen and Gathumbi, 2007).

Groundwater flow per unit ditch length multiplied by the length of the ditch section (170 m) provided an estimate of the total groundwater flow contributed by the entire ditch section assuming same ditch water and groundwater levels and K_{sat} for the entire ditch section. These assumptions might result in uncertainty in flow and P and N load estimations to some extent. The groundwater flow from each groundwater contributing area (east and west) was calculated separately and summed to get the total groundwater flow. For the east and west groundwater contributing areas, heads in wells F and A, respectively were used in the Dupuit equation (Figure A-2). The average bottom elevation (7.92 m above sea level, North American Vertical Datum 88) for the ditch section was used as the reference line from which the heads in the wells and ditch water levels were calculated. When the head in the ditch is lower than the heads in the wells, the groundwater moves toward the ditch and this flow would be referred to as positive flow. When the head in the ditch is higher than the heads at the wells, water moves from the ditch to the water table aquifers. This flow would be referred to as negative flow.

Groundwater nutrient loading

The TP and TN loadings to and from the ditch were calculated by multiplying the groundwater flow (L/s) by the corresponding TP and TN concentrations (mg/L) in the water samples. When the groundwater flow was positive, the average TP and TN concentrations for all the five wells on one groundwater contributing area were used to calculate the loads from that groundwater contributing area. But when the groundwater flow was negative, TP and TN concentrations in the ditch water were used to calculate the TP and TN loads. The load was positive or negative using the same convention that was used for groundwater flow. To calculate groundwater TP and TN loads, the concentration data need to be multiplied by the groundwater flow data. Groundwater flow data were collected at 15-min intervals while groundwater samples were collected once a month. Therefore, a linear interpolation between two monthly data points was used to estimate the groundwater TP and TN concentrations at 15-min interval (Kronvang and Bruhn, 1996). Summing the loads for all 15-min intervals would provide the total load for the entire period. Use of monthly groundwater concentration data may introduce uncertainty in estimating groundwater N and P loads. But in the absence groundwater flow, and P and N loads data in the literature for the south Florida beef cattle ranches, the results from this study may be used as a reference.

Results and discussion

The average net groundwater flow to the ditch section for the four years were 765 and 1,729 m^3 per year for the low (3.84×10^{-5} m/s) and high (8.38×10^{-5} m/s) K_{sat} values, respectively (Table A-4). During the study period, there were large variations in annual rainfall (Table A-5). Years 1 and 4 were wet, year 2 was moderately wet, and year 3 was dry. During years 1 and 4, total rainfall amounts were similar but groundwater flow during year 1 (low = 1,794 m^3 and high = 4,052 m^3) was higher than that during year 4 (low = 738 m^3 and high = 1,667 m^3) (Table A-4). The five months (January-May 2005) prior to year 1 had above average rainfall (51 cm). Moreover, four hurricanes namely Charley, Frances, Ivan, and Jeanne hit Florida in August-September, 2004 (total rainfall = 65 cm within 6 weeks) which created excessively wet condition and high groundwater levels in the watershed during early part of year 1. In addition to that, Hurricane Wilma (rainfall=19.3 cm) occurred in October 2005 resulting in high groundwater flow during year 1. Hurricane Wilma created wet condition for the remaining part of year 1 resulting in overall high groundwater flow during year 1. Figure A-3 shows groundwater and ditch water levels, and groundwater flow volumes for years 1-4. Well head data for 16 August-15 September 2005 were not available. For these days, average daily flow and loadings for that year (year 1) were used. The average daily flow gradient (well head minus ditch head) between well A (west groundwater contributing area) and the ditch, and between well F (east groundwater contributing area) and the ditch were 0.127 and 0.132 m (year 1), 0.013 and 0.082 m (year 2), -0.069 and 0.003 m (year 3), and 0.010 and 0.051 m (year 4). A negative average gradient indicates predominance of negative flow.

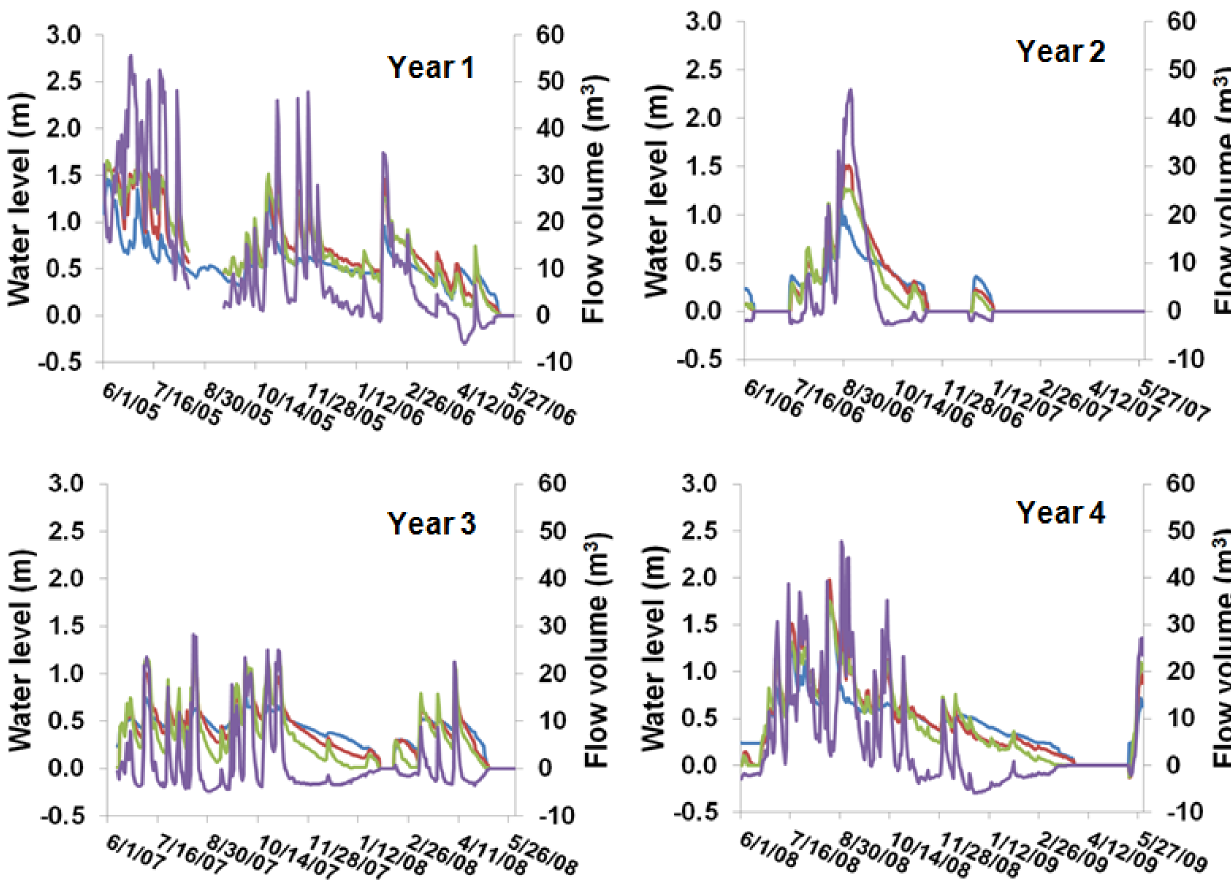

Figure A-3. Groundwater (well A red line, and well F green line) and ditch water (blue line) levels, and groundwater volumes (violet line) for years 1-4 using 8.38x 10^{-5} m/s (upper value) as saturated hydraulic conductivity.

Table A-4. Net groundwater flow volumes (high, low, and average values) contributed to the ditch section using high (8.38 x 10^{-5} m/s) and low (3.84 x 10^{-5} m/s) saturated hydraulic conductivity during the years 1-4. A'-'sign denotes that the net flow is negative (flowing from the ditch to the water table aquifers).

Period	Groundwater flow (m^3)				
	Flow estimation	Year 1	Year 2	Year 3	Year 4
Annual	High	4,052	829	366	1,667
	Low	1,794	367	162	738
	Average	2,923	598	264	1,203
Wet season	High	2,752	873	521	1,839
	Low	1,218	386	231	814
	Average	1,985	630	376	1,327
Dry season	High	1,300	-44	-155	-172
	Low	576	-19	-69	-76
	Average	938	-32	-112	-124

Table A-5. Annual, wet season (June-October), and dry season (November-May) rainfall amounts for the years 1-4 at the study site.

Period	Rainfall (cm)		
	Annual	Wet season	Dry season
Year 1	147	111	36
Year 2	91	64	27
Year 3	109	76	33
Year 4	157	112	45

Although year 2 experienced lower rainfall than year 3, there was higher groundwater flow during year 2 compared to year 3. Year 3 was a record drought season in south-central Florida, and Lake Okeechobee experienced historically lowest water level in July 2007 (during year 3) (SFWMD, 2009). The wet conditions during year 1 created higher antecedent groundwater storage for year 2 generating more groundwater flow. The term antecedent groundwater storage is used here to indicate the groundwater level immediately before a year or a season. Table A-6 shows the average groundwater depths (above mean sea level) at the study site a day prior to the beginning of the wet season (June 1-October 31) and dry season (November 1-May 31). These groundwater levels in the wells were reflective of the antecedent groundwater storage for wet and dry seasons. The wet season for year 1 had the largest antecedent groundwater storage contributing high groundwater flow to the ditch. Year 2 had larger antecedent groundwater storage than that for year 3 which was reflected by higher groundwater level on 31 May 2006 than on 31 May 2007.

Table A-6. Groundwater levels (North American Vertical Datum 88) above mean sea level at the experimental site prior to the wet season (June-October) on 31 May, and prior to the dry season (November-May) on 31 October.

Year	Groundwater level (m)	
	31 May	31 October

2005	9.37	9.56
2006	7.99	8.42
2007	7.60	8.86
2008	7.91	8.49

Out of the four dry seasons (November-May), only during year 1 there was net positive flow contribution to the ditch because year 1 was an overall wet year (Table A-5) with Hurricane Wilma occurring on 24 October, 2005, one week before the beginning of dry season for year 1. For the years 2, 3, and 4, there were net negative flows (Table A-4, Figure A-3). Table A-7 shows the number of days when the ditch section gained (net positive daily flow) and lost (net negative daily flow) groundwater flow. Year 1 had the highest number of days with positive flow while year 2 had the lowest number of days with positive flow and highest number of days with no flow.

Table A-7. Number of days when positive and negative flows occurred at the site during years 1-4.

Year	Days with positive flow	Days with negative flow	Days with no flow
Year 1	305	46	14
Year 2	62	95	208
Year 3	95	225	46
Year 4	153	158	54

Table A-8 shows the yearly, wet season, and dry season TP, TN, TKN and NO_3-N loads (high, low, and average) for years 1-4. Average net annual TP and TN loads (high, and low values) to the ditch section for the four years were 0.080-0.174 and 2.053-4.481 kg, respectively. The net annual groundwater contributions of TP and TN for all years were positive with the exception of drought year 3. TKN loads were higher than NO_3-N loads for all years. Figure A-4 shows the daily TP and TN loadings (high values) for the study period.

The average groundwater TKN, and NO_3-N concentrations for the study period were 2.705 and 0.084 mg/L, respectively. The average ditch water TKN and NO_3-N concentrations for the study period were 3.650 and 0.058 mg/L, respectively. The average TKN concentrations in both groundwater and ditch water were higher than the average NO_3-N concentrations primarily due to excessive denitrification in south and central Florida (Martin and Reddy; 1997). Bohlen (2009) also observed higher ditch water TKN concentrations (average concentration = 3.090 mg/L) compared to NO_3-N concentrations (average concentration = 0.040 mg/L) in a pasture water management study in Buck Island Ranch (Okeechobee County) in the Lake Okeechobee watershed during 2005-08.

Average groundwater NH_3-N and Dissolved Organic Nitrogen (DON = TKN – NH_3-N) concentrations for the study period were 1.035 and 1.670 mg/L, respectively while ditch water NH_3-N and DON concentrations were 0.670 and 2.980 mg/L, respectively. Higher ON and NH_3 -N concentrations in the water were due to cow defecation and urination.

Table A-8. High, low and average net Total Phosphorus, Total Nitrogen, Total Kjeldahl Nitrogen, and Nitrate-N loads contributed by the ditch section using 8.38 x 10^{-5} and 3.84 x 10^{-5} m/s as saturated hydraulic conductivity. A '-'sign denotes that the net nutrient load was from the ditch section to the water table aquifers (negative flow).

Period	Load estimation	Total Phosphorus (kg)			Total Nitrogen (kg)			Total Kjeldahl Nitrogen (kg)			Nitrate-N (kg)		
		Total	Wet season	Dry season	Total	Wet season	Dry season	Total	Wet season	Dry season	Total	Wet season	se
Year 1	High	0.972	0.858	0.114	12.445	8.645	3.800	11.985	8.294	3.690	0.460	0.351	0
	Low	0.445	0.393	0.052	5.703	3.962	1.741	5.492	3.801	1.691	0.211	0.161	0
	Average	0.709	0.626	0.083	9.074	6.304	2.771	8.739	6.048	2.691	0.336	0.256	0
Year 2	High	0.046	0.054	-0.008	1.920	1.997	-0.077	1.909	1.982	-0.073	0.011	0.015	-0
	Low	0.021	0.025	-0.004	0.880	0.915	-0.035	0.875	0.909	-0.034	0.006	0.007	-0
	Average	0.034	0.040	-0.006	1.400	1.456	-0.056	1.392	1.446	-0.054	0.009	0.011	-0
Year 3	High	-0.713	-0.530	-0.183	-0.837	0.806	-1.644	-0.832	0.780	-1.612	-0.005	0.027	-0
	Low	-0.327	-0.243	-0.084	-0.384	0.369	-0.753	-0.382	0.357	-0.739	-0.003	0.012	-0
	Average	-0.520	-0.387	-0.134	-0.611	0.588	-1.199	-0.607	0.569	-1.176	-0.004	0.020	-0
Year 4	High	0.390	0.448	-0.058	4.396	5.144	-0.748	4.296	5.058	-0.762	0.100	0.086	0
	Low	0.178	0.205	-0.027	2.014	2.357	-0.343	1.969	2.318	-0.349	0.045	0.039	0
	Average	0.284	0.327	-0.043	3.205	3.751	-0.546	3.133	3.688	-0.556	0.073	0.063	0

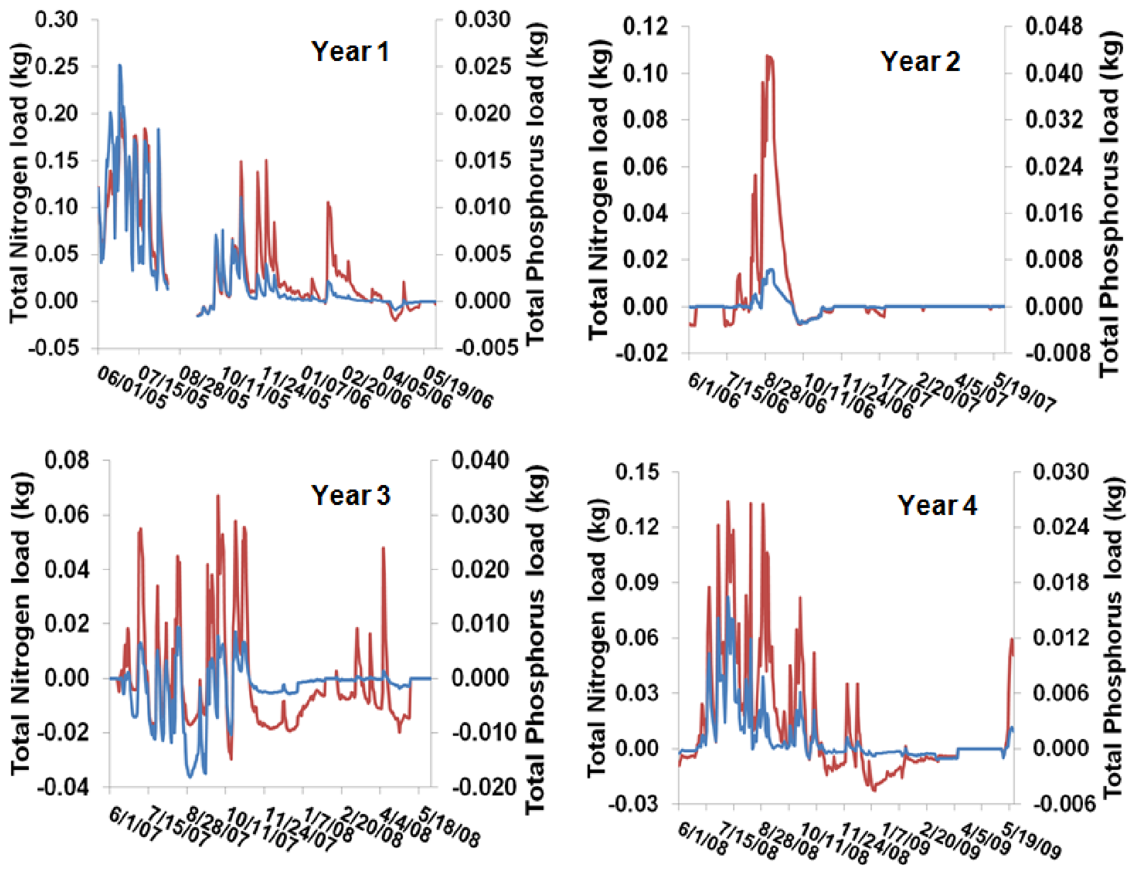

Figure A-4. Daily Total Phosphorus (blue line) and Total Nitrogen (red line) loads for years 1-4 using saturated hydraulic conductivity 8.38 x 10^{-5} m/s (high value).

The average ditch water TP and TN concentrations were higher than the average groundwater TP and TN concentrations (Figure A-5 and Figure A-6). Both increase and decrease (dilution) of TP and TN concentrations were observed during the study period depending on rainfall and fertilizer application time. Figure A-7 and Figure A-8show the daily average heads for well A (west water table aquifer) and well F (east water table aquifer) and groundwater and surface water TP and TN concentrations during years 1-4.

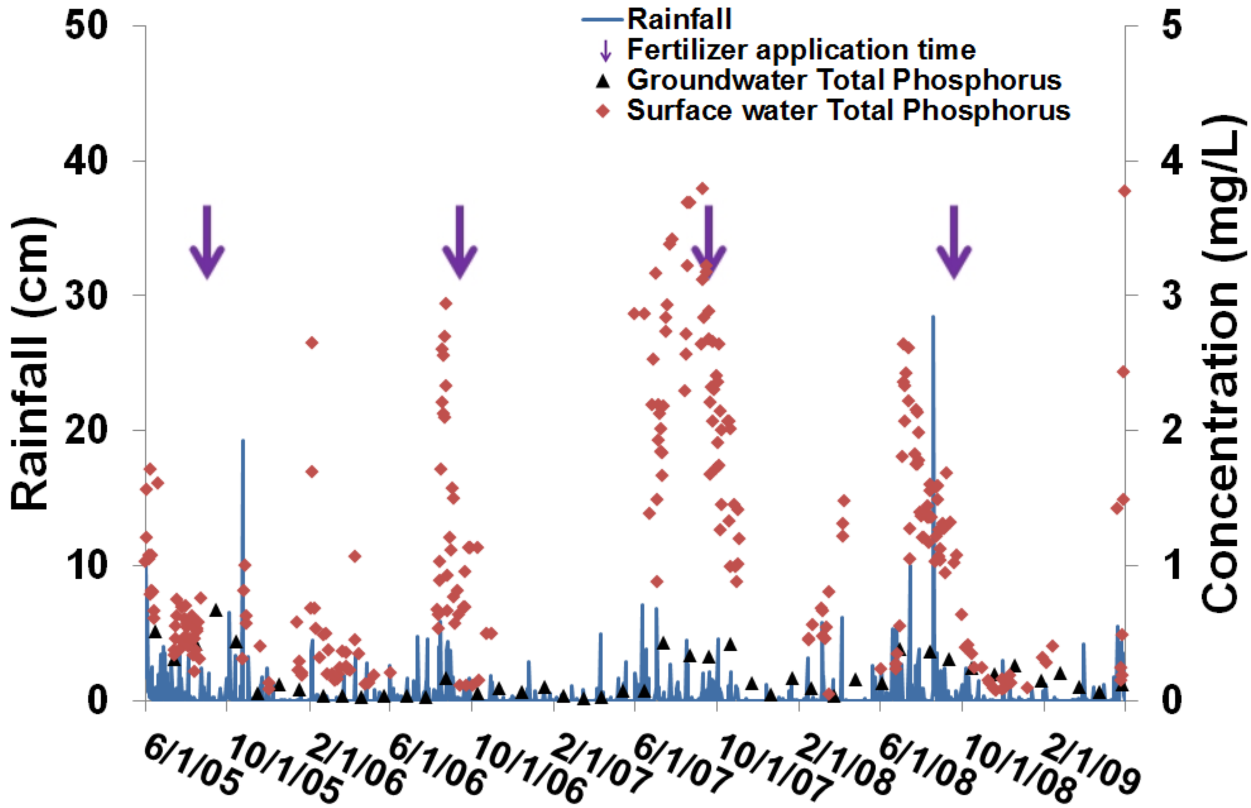

Figure A-5. Fertilizer application time, daily rainfall, and daily average groundwater and surface water Total Phosphorus concentrations during years 1-4 (June 2005-May 2009).

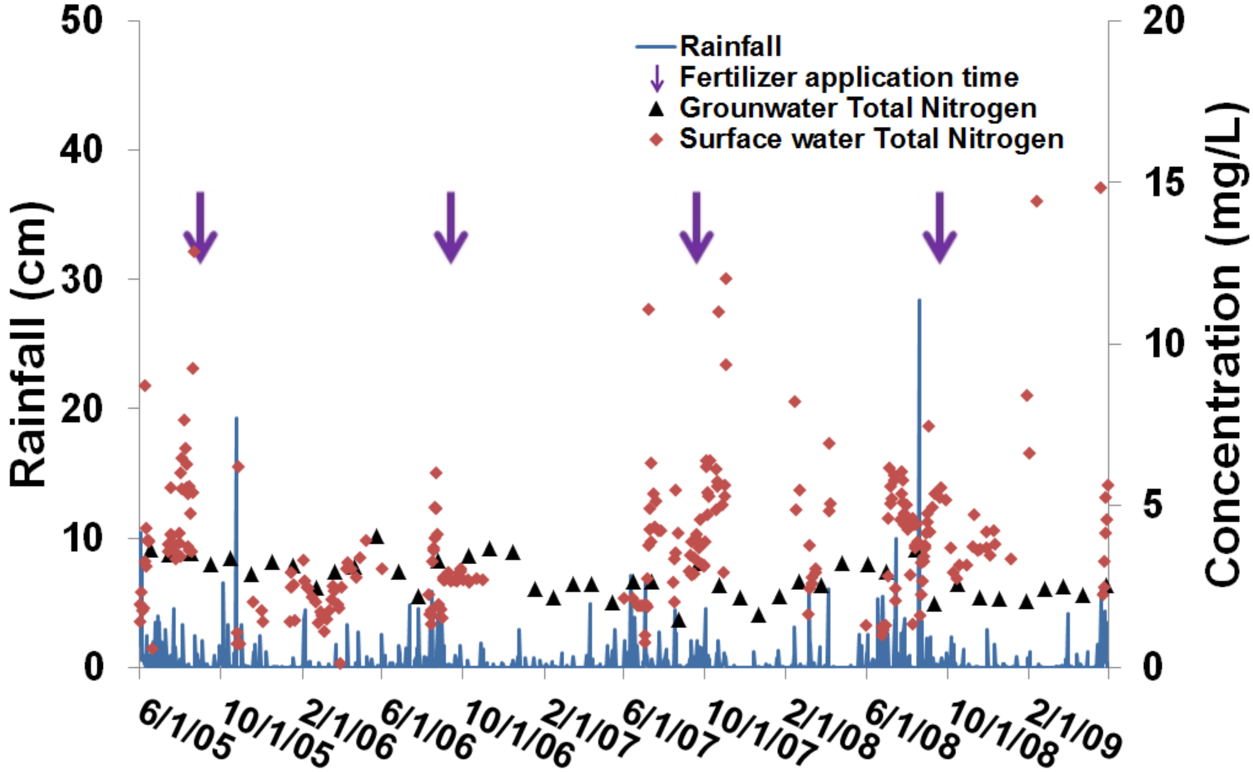

Figure A-6. Fertilizer application time, daily rainfall, and daily average groundwater and surface water Total Nitrogen concentrations during years 1-4 (June 2005-May 2009).

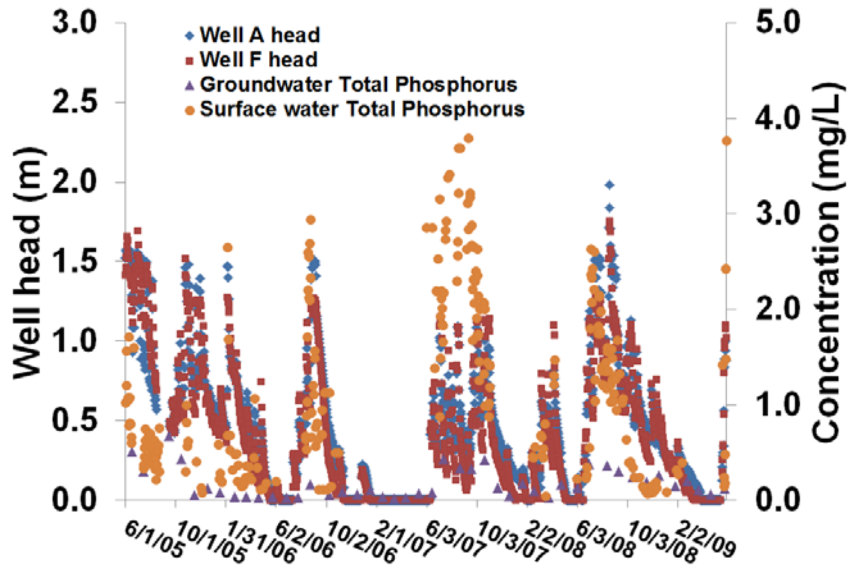

Figure A-7. Daily average heads for well A (west water table aquifer) and well F (east water table aquifer) and average groundwater and surface water Total Phosphorus concentrations during years 1-4 (June 2005-May 2009).

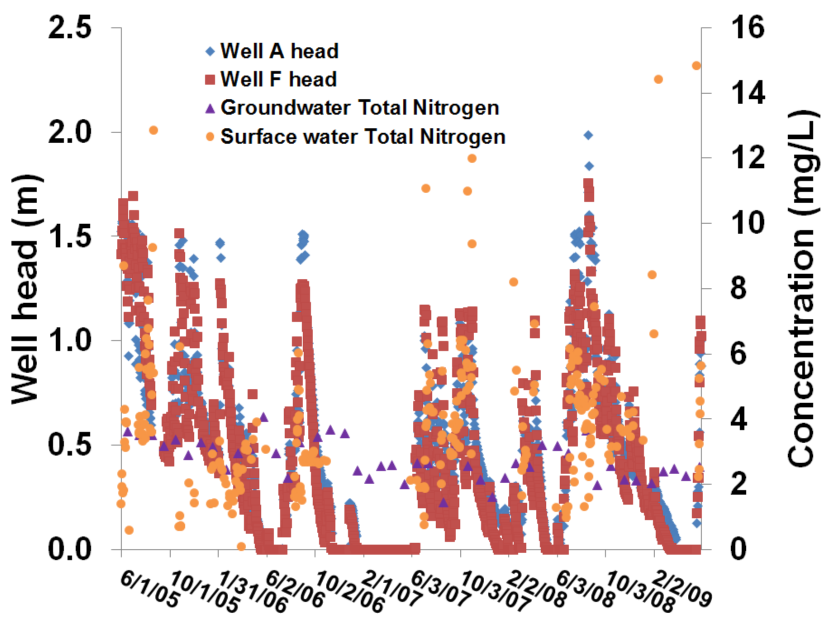

Figure A-8. Daily average heads for well A (west water table aquifer) and well F (east water table aquifer) and average groundwater and surface water Total Nitrogen concentrations during years 1-4 (June 2005-May 2009).

The total area of the Lake Okeechobee watershed is about 1,392,852 ha (SFWMD, 2011). From our study, the average annual TP and TN loads (low and high values) to the ditch section were 0.080-0.174 and 2.053-4.481 kg, respectively. The total area of the study site (ditch section, and east and west groundwater contributing areas) is approximately 0.4 ha. By scaling up the TP and TN loads from this study site to the entire Lake Okeechobee watershed, the extrapolated TP and TN loads became 278,570-605,891 (low and high values) and 7,148,814-15,603,428 kg (low and high values), respectively. The total estimated TP and TN loads (runoff and groundwater contributions) to Lake Okeechobee from the Lake Okeechobee watershed during water year 2010 (May 1, 2009-April 30) were 443,000 and 5,092,000 kg, respectively (SFWMD, 2011). The estimated P load (443,000 kg) to Lake Okeechobee is within the extrapolated low and high groundwater contributions (278,570-605,891 kg). This analysis shows that groundwater is an important pathway for P and N to Lake Okeechobee.

Movement of nutrients out of the ditch to the neighboring groundwater observed in this study indicates that part of the nutrient moved to the groundwater would be retained by the soil before it moves back into ditch. Such retention is beneficial with regards to reducing the P loads from a ranch. This observation is relevant to some Best Management Practices (BMPs) such as water retention using flashboard riser structures (FDACS, 2008). Use of flashboard riser at the outflow point in the ranch can increase the ditch water level which in turn can reverse the flow direction from the ditch to the water table aquifers on many occasions when the flow would have been otherwise from the water table aquifers to the ditch. Reversing the flow direction to facilitate more negative flow will result in increased residence time and therefore increase the opportunities for P retention through sorption on to the soil and plant uptake of P with the end result of reducing the net nutrient loadings from the ditches in ranches to Lake Okeechobee.

Conclusion

This study provided an insight into flow, and nutrient (P and N) loadings contributed by groundwater to a typical ditch section within a beef cattle ranch in the Lake Okeechobee basin. The average net groundwater flow to the ditch section for the four years were 765 (low value) to 1,729 m^3 (high value). Average net annual TP and TN loads to the ditch section for the four years were 0.080-0.174 (low and high values) and 2.053-4.481 kg (low and high values), respectively. In this study, two extreme values (high and low) of K_{sat} for similar soil from FSCDRS were used to estimate groundwater flow. Ditch water and groundwater levels were assumed to be same for the entire ditch section at any specific time. Limited numbers of groundwater P and N concentration data were used in this study. These might have generated uncertainty in flow and load estimations. There are no data available in the literature related to groundwater flow volume, and TP and TN loadings for drainage ditches within the beef cattle ranches in Florida to compare with the data derived from this study. The future studies on groundwater flow and load estimations in the beef cattle ranches should involve on-site hydraulic conductivity estimations, groundwater flow estimations along several transects vertical to the ditch, and more frequent groundwater and surface water sampling. In the absence of data in the literature on groundwater contribution of flow and P and N loads to drainage ditches in the beef cattle ranches in south Florida, the results from this study may be used as a reference for all practical purpose.

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Appendix B-1

Evaluation of Wetland Water Retention (WWR) BMP

The WWR BMP evaluation component of the cow-calf BMP project was started in 2003 at the Pelaez Ranch located within the Lake Okeechobee basin. Two wetland sites (wetlands 1 and 2, Figure B-1) located within this beef cattle ranch were selected for quantifying WWR effects on hydrology and water quality. Both wetlands contain a ditch that was used to drain the wetlands as well as the associated drainage areas containing pastures. The term wetland area mentioned here includes the actual wetland as well as the associated pasture areas that are drained through the ditch. The total areas of the two wetland sites (wetland plus upland areas) are 81.7 and 24.4 ha, respectively.

The WWR BMP was implemented at both wetlands by installing a flashboard riser structure at a ditch that drained the wetland. Boards were added to heights of 1.10 and 0.52 m above the ditch bottom at wetlands 1 and 2, respectively. The WWR was evaluated with regards to water and nutrient retention upstream of the flashboard riser. The WWR was also evaluated for its effects on wetland hydroperiod, water storage, and inundated area.

The water retention levels for the two wetlands were set based on the comfort level of the rancher cooperator. For Wetland 1, there were two pre-BMP periods and four post-BMP periods for evaluating the BMP effectiveness (Table B-1). For Wetland 2, there was only one pre-BMP period and five post-BMP periods. June-December 2010 was considered the post-BMP4 period for Wetland 1, and post-BMP5 period for Wetland 2. Although this period was shorter than other pre- and post-BMP periods, for completeness sake it was assumed to represent one year of hydrologic and water quality data.

Table B-1. Pre- and post-BMP periods for wetland water retention BMP evaluation at wetlands 1 and 2.

Wetland 1	Wetland 2	Period
pre-BMP1	pre-BMP	June, 2005-May, 2006
pre-BMP2	post-BMP1	June, 2006-May, 2007
post-BMP1	post-BMP2	June, 2007-May, 2008
post-BMP2	post-BMP3	June, 2008-May, 2009

post-BMP3	post-BMP4	June, 2009-May, 2010
post-BMP4	post-BMP5	June, 2010-Dec 2010

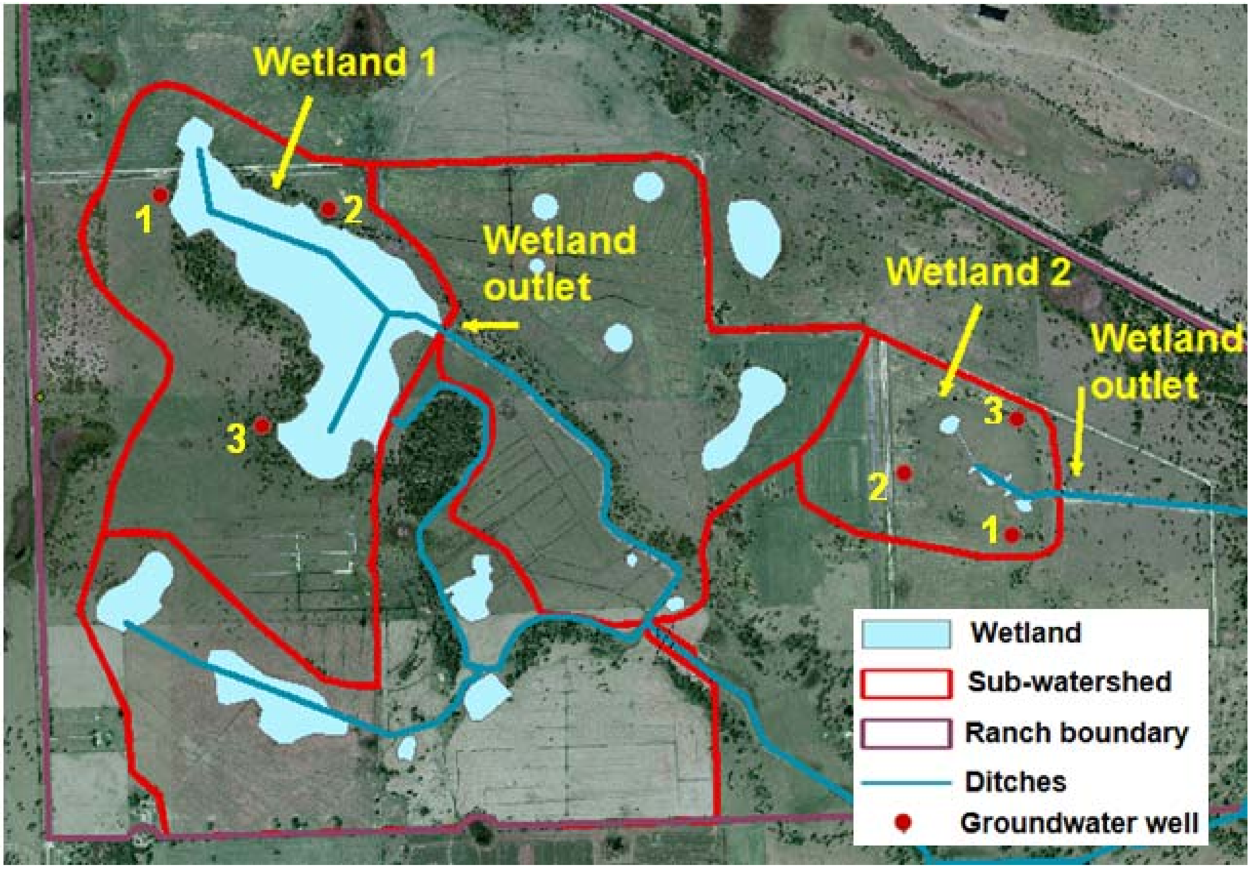

Figure B-1. Locations of wetlands 1 and 2 within the beef cattle ranch. Red lines denote drainage sub-basins for the two wetland sites (wetlands plus upland contributing areas) within the ranch.

Wetland 1 contains two main drainage ditches that drain the wetland as well as upland pasture areas. Drainage from two ditch segments (669 and 332 m in length) converges and flows together for 147 m before exiting the wetland. Soil types within Wetland 1 area are Myakka fine sand (7.5 ha), Immokalee fine sand (40.1 ha), Basinger fine sand (21.5 ha), Floridana sand (6.0 ha), and Valkaria sand (5.0 ha). Wetland 2 contains a single drainage ditch which flows in a south-easterly direction (Figure B-1). The soil series within Wetland 2 include Basinger (13.4 ha) and Immokalee fine sands (11.0 ha). The wetland drainage areas include improved pasture and forested areas for both wetlands. The two major forage types within the drainage areas of the two wetlands are Bahia (*Paspalum notatum)* and Floralta (*Hemarthria altissima*).

Materials and Methods

The WWR was implemented by installing a flashboard riser structure at the end of each wetland's drainage ditch (Figure B-2). The flashboard riser is used to allow water to flow only when the water level exceeds the top elevation of the flashboard. The bottom elevations of Wetland 1 and 2 were 7.95 and 8.97 m AMSL, respectively (NAVD 88). The elevations of the flashboard riser tops at wetlands 1 and 2 were at 9.06 and 9.47 m, respectively.

Figure B-2. Wetland water retention BMP (flashboard riser with boards) at Wetland 2.

Trapezoidal flumes fitted with pressure transducers were installed downstream of the flashboard riser and culvert structure to measure flow. In July, 2008 an Acoustic Doppler Velocimeter (ADV) was installed at the flume to measure flow velocity and rate. Wetland stages were monitored at each wetland using pressure transducers. Auto-samplers were installed at the flume sites to collect water samples that were analyzed for TP, Total Kjeldahl Nitrogen (TKN), Ammonia (NH_3-N), and Nitrate (NO_3-N). Total N was estimated as TKN + NO_3-N.

Evaluation of the WWR at the two wetland sites was based on numerical and statistical comparisons of the P and N concentrations and loads for pre- and post-BMP periods. Statistical analyses were conducted using Proc GLIMMIX in SAS v9.2 (SAS Institute, Cary, NC) with heterogeneous variance models to compare TP and TN loads and concentrations for the pre- and post-BMP periods.

Results and Discussions

Wetland 1

Water Dynamics

Pre-BMP1, post-BMP2, and post-BMP3 periods had comparable rainfall, and therefore use of data for these periods may batter evaluate the BMP effects (Table B-2). The average annual rainfall for the region is 125 cm. Pre-BMP 2 had lower than average rainfall which combined with yet another lower than average rainfall for post-BMP1 resulted in extreme drought conditions in the latter period. Both these years had very low actual runoff as well as rainfall adjusted runoff (rainfall normalized) which clearly impacted the TP and TN loads. Compared to pre-BMP1 period, post-BMP2 period had only 7% higher rainfall, but runoff depth in the wetland was 52% higher. Unusually high runoff depth for the post-BMP2 period was mainly due to Tropical Storm Fay that resulted in 30 cm rainfall (August 19 – 28.5 cm and August 20 – 1.5 cm) and resulted in highest inundation area during the entire monitoring period. This rainfall event combined with other smaller rainfall (August 19 – September 4, 2008, total rainfall = 39.4 cm) resulted in continuous flow till September 4, 2008 accounting for 81% of the annual flow and 69 and 65 % of annual TP and TN loads, respectively. Post-BMP3 period had 9% higher rainfall compared to pre-BMP1, but had only 1% higher runoff depth mainly due to the absence of extreme rainfall event. Pre-BMP 1 period also had an extreme rainfall event on October 24, 2005 (Hurricane Wilma, total rainfall = 19.3 cm). This event combined with an additional rainfall of 4.1 cm resulted in continuous flow till November 12, 2005 and accounted for 31% of total annual flow and 20% TP load and 17% TN load.

The average runoff for the four post-BMP periods was 11.8 cm compared to the average runoff of 8.3 cm for the two pre-BMP periods indicating that, on average, there was higher runoff and less water retention during the post-BMP periods compared to the pre-BMP periods. It should be noted that water retention here is defined as the reduction in surface flow (runoff) due to the WWR. During post-BMP2 and post-BMP3 periods, total runoff and rainfall adjusted runoff depths were higher compared to the pre-BMP period. This was mainly due to very high flow from Wetland 1 resulting from above average rainfall as well as rainfall intensity (high intensity rainfall for post-BMP2) and distribution (above average rainfall for dry period of post-BMP3) for these two post-BMP periods. For the post-BMP2 period, high rainfall during July (32.7 cm) and August (40.6 cm) caused the relatively higher annual runoff. The post-BMP3 period experienced higher than average rainfall during the dry period which increased the dry period runoff as well as the annual runoff. Higher than average rainfall during March (19.3 cm) and April (27.3 cm) of the post-BMP3 combined with the relatively higher antecedent water table depths for these two months resulted in increased runoff during these two months thereby increasing the annual runoff depths for this period. Wetland water retention may have resulted in shallower water table depths in the entire wetland area. Shallower water table depths in the wetland and upland areas can increase the potential for water table rise to the surface which then results in increased ponding and runoff generation. Wetland hydroperiod, water storage, and inundated areas for the post-BMP2 and 3 periods were higher than the pre-BMP period indicating the increased volume as well as distribution of inundated areas. Details of the hydroperiod, storage, and inundation for the pre and post-BMP periods are presented in Appendix B-2. Figure B-3 shows the maximum daily inundated areas observed for the pre-BMP1, post-BMP2 and post-BMP3 periods. For the pre-BMP1 period, inundated area observed a day after Hurricane Wilma (October 24, 2005, rainfall = 19 cm) was lower than the maximum inundation observed for the post-BMP2 (Tropical Storm Fay on August 18-19, 2008; rainfall = 30 cm) and post-BMP3 (March 21-28, 2010; rainfall = 8 cm) periods.

In summary, unusually higher rainfall and/or the relatively shallower water table depths caused by the WWR may have increased the connectivity of the ponded areas which resulted in higher than normal runoff for the post-BMP2 and 3 periods compared to the pre-BMP periods.

Table B-2. Rainfall and runoff depths for the pre- and post-BMP periods (June-May) at Wetland 1.

Period	Rainfall (cm)	Runoff depth (cm)	Runoff depth per unit rainfall
pre-BMP1	147	15.9	0.108
pre-BMP2	91	0.6	0.007
post-BMP1	109	0.2	0.002
post-BMP2	157	24.2	0.154
post-BMP3	160	17.5	0.109
post-BMP4*	53	5.3	0.099

*covers June-Oct, 2010

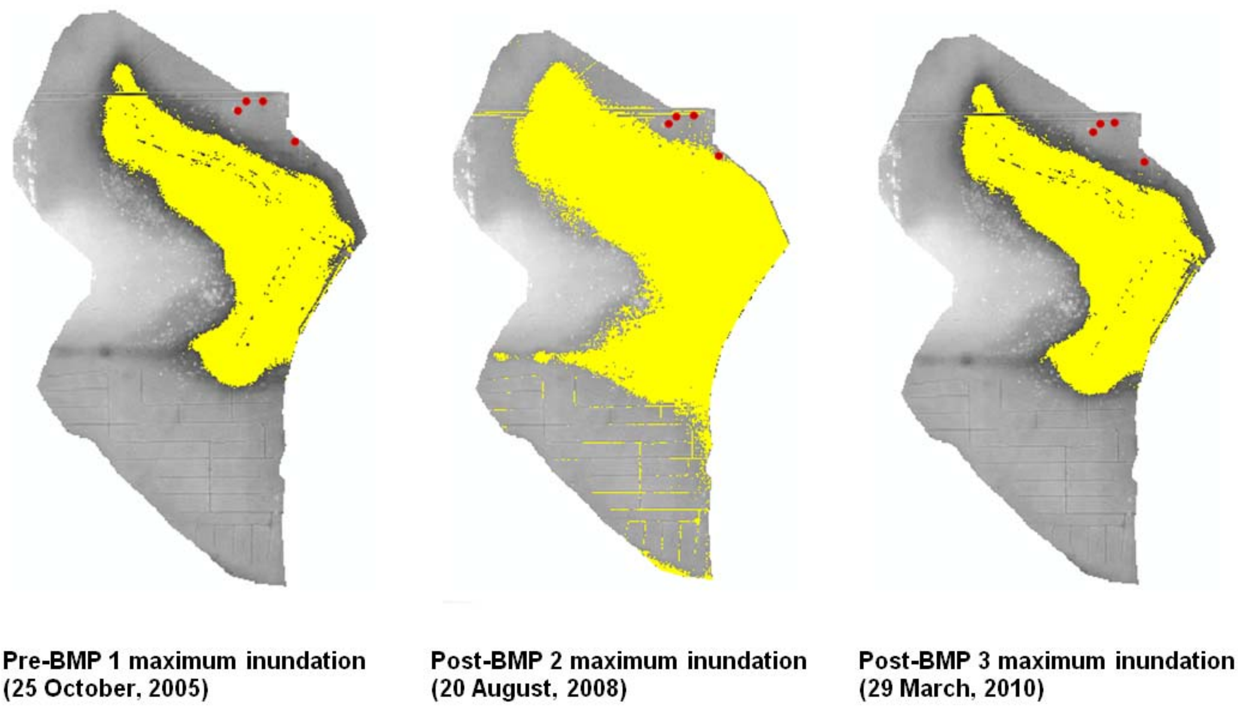

Figure B-3. Maximum inundations during pre-BMP1, post-BMP2, and post-BMP3 periods at Wetland 1. The red dots indcate P hotspots with high nagative SPSC values.

Phosphorus and Nitrogen Concentration and Load

Post-BMP2 TP load, mean measured concentration, and mean flow-weighted concentrations increased by 106, 134, and 55% compared to those for the pre-BMP1 period (Table B-3 and Table B-4). Post-BMP2 had the highest TP load during the entire monitoring period. Large increases in the TP load during the post-BMP2 period resulted in increasing the average TP load for the entire post-BMP period. The mean measured and flow-weighted TP concentrations during the post-BMP3 increased by 63 and 20%, respectively compared to those for the pre-BMP1 period. Although TP load reductions were observed during post-BMP1 and post-BMP4 periods, these reductions were partly caused by the lower than average rainfall and flow due to relatively dry conditions that existed in the Lake Okeechobee basin during this period. The post-BMP1 (June 2007-May 2008) period experienced extreme drought conditions. Similarly, the rainfall between June 2010 and December 2010 was lower than average in the region for this period. Overall, there was 55% increase in average post-BMP period TP load (73 kg) compared to the average pre-BMP period TP load (47 kg).

Increased TN loads and concentrations were also observed during the post-BMP periods. Post-BMP2 TN load, mean measured and mean flow-weighted concentrations were 93, 47, and 39% higher than the pre-BMP1 period (Table B-3 and Table B-4). Post-BMP3 TN load, mean measured and mean flow-weighted concentrations also increased by 37, 76, and 53% compared to the pre-BMP1 period. Similar to TP, TN load reductions were also observed during the post-BMP1 and post-BMP4 periods. However, as stated before, these reductions were mainly due to the low flow resulting from lower than average rainfall. Overall, there was 80% increase in average post-BMP period TN load (290 kg) compared to the average pre-BMP period TN load (162 kg). Following the analysis of TP loads on annual basis was a statistical analysis of daily TP and TN concentrations to determine whether the changes in concentrations and loads were statistically significant.

Results from Proc GLIMMIX indicated that mean daily annual and wet season TP concentrations for the post-BMP periods were significantly higher than those for the pre-BMP period ($p<0.0001$). Due toyear-to-year variability in both rainfall and the number of days with zero flows /TP loads for the pre-BMP and post-BMP periods, analyses of annual TP load was limited to year to year comparisons (instead of comparing aggregate pre-BMP periods with aggregate post-BMP periods). Mixed results were obtained when individual pre-BMP and post-BMP years were compared. For all the pre-BMP and post-BMP comparisons, statistically significant TP load changes were observed only for the following combinations: pre-BMP1 > post-BMP1 ($p < 0.001$), post-BMP3 > pre-BMP1 ($p = 0.0063$), and pre-BMP1 > post-BMP4 ($p < .0001$). Decreases in TP loads during the post-BMP1 and post-BMP4 were mainly caused by the lower than average rainfall and flows. Given that pre-BMP1 and post-BMP3 had similar rainfall, it is likely that WWR BMP may increase the TP loads.

One of the reasons for not observing statistically significant differences in mean annual daily TP load comparisons was the presence of long periods of zero flows during the dry season. To reduce the bias due to non-flow days and the fact that most of the annual TP load occurs during the wet season, the wet season pre-BMP and post-BMP loads were also compared. Overall, wet season post-BMP TP loads were higher than the pre-BMP TP loads ($p< 0.001$). When specific pre-BMP and post-BMP wet season loads were compared, the following comparisons were statistically significant: post-BMP2 > pre-BMP1 ($p < 0.001$), post-BMP3 > pre-BMP1 ($p <$

0.001), post-BMP2 > pre-BMP2 (p < 0.001), post-BMP3 > pre-BMP2 (p < 0.001), and post-BMP4 > pre-BMP2 (p = 0.0021). In summary, the results from the statistical analyses indicated that both annual and wet season TP concentrations increased during the post-BMP period. There was some evidence for the increased annual TP loads for some of the post-BMP periods while there was more convincing evidence that the wet season post-BMP TP loads were higher than the pre-BMP periods.

Results from statistical analyses for TN concentrations and loads were similar to those observed for TP. There were significant increases in mean daily post-BMP annual ($p<0.001$) and wet season ($p< 0.001$) TN concentrations. There was significant increase in mean daily TN load during post-BMP3 period ($p<0.0001$) while post-BMP1 and post-BMP4 TN loads were significantly lower compared to pre-BMP1 loads ($p<0.0001$). Mean daily TN loads for the post-BMP wet periods were significantly higher than the pre-BMP periods ($p<0.0001$).

Increased loads during post-BMP2 (both TP and TN) and post-BMP3 (TN only) periods were partly due to the higher runoff than the pre-BMP periods. Higher runoff was mainly due to the occurrence of a higher number of large rainfall events, higher water storage and shallower water depths which resulted in enhanced connectivity of the ponded areas within the entire drainage area. Part of the increased TP and TN loads may also have been due to the enhanced losses of surface and subsurface soil P stored in the drainage area. Soil TP storage and retention capacity were analyzed to examine the role of soils in affecting the TP loads for the pre-BMP and post-BMP periods.

Table B-3. Total Phosphorus (TP) and Total Nitrogen (TN) mean, and mean flow-weighted concentrations for the pre- and post-BMP periods at Wetland 1.

Period	Mean concentration (mg/L)		Mean flow-weighted concentration (mg/L)	
	Total Phosphorus	Total Nitrogen	Total Phosphorus	Total Nitrogen
pre-BMP1	0.59	2.93	0.71	3.10
pre-BMP2	1.00	2.57	0.79	1.63
pre-BMP Average	**0.80**	**2.75**	**0.75**	**2.37**
post-BMP1	1.25	3.41	1.27	3.72
post-BMP2	1.38	4.32	1.10	4.31
post-BMP3	0.96	5.15	0.85	4.75
post-BMP4	0.93	7.28	0.54	5.04

post-BMP Average	1.13	5.04	0.94	4.26

Table B-4. Annual and seasonal (wet and dry) total Phosphorus (TP) and total Nitrogen (TN) loads for the pre- and post-BMP periods at Wetland 1.

Period	Rainfall (cm)	Total Phosphorus (kg)			Total Nitrogen (kg)		
		Total	Wet period	Dry period	Total	Wet period	Dry period
pre-BMP1	147	88.8	81.1	7.7	312.0	282.6	29.4
pre-BMP2	91	5.1	5.1	0.0	11.4	11.4	0.0
pre-BMP Average	**119**	**47.0**	**43.1**	**3.9**	**161.7**	**147.0**	**14.7**
post-BMP1	109	2.0	0.9	1.1	6.0	3.1	2.9
post-BMP2	157	183.4	183.4	0.0	602.8	602.8	0.0
post-BMP3	160	87.5	43.2	44.3	426.4	252.2	174.2
post-BMP4	53	18.1	18.1	0.0	126.3	126.2	0.1
post-BMP Average	**120**	**72.8**	**61.4**	**11.4**	**290.4**	**246.1**	**59.0**

Wetland 2

Water Dynamics

Although pre-BMP, post-BMP3, and post-BMP4 periods had comparable rainfall, post-BMP3 and post-BMP4 runoff depths were 4 and 33% lower than that for pre-BMP period. This indicates that there was more water retention during the post-BMP periods (post-BMP3 and post-BMP4) compared to the pre-BMP period. Increases in hydroperiod, water storage, and inundated areas at Wetland 2 site for the post-BMP3 and post-BMP4 periods were also observed. Since the wetland is located at a higher elevation compared to the surrounding areas, the retained water is likely to be lost to the surrounding low lying areas through sub-surface pathways which unlike Wetland 1, is likely to increase the available surface and subsurface storage areas for subsequent rainfall events. Figure B-4 shows the maximum daily inundated areas observed for the pre-BMP, post-BMP3 and post-BMP4 periods. For the pre-BMP1 period, the inundated area observed a day after Hurricane Wilma (October 24, 2005, rainfall = 19 cm) was lower than the maximum inundation observed for the post-BMP2 (Tropical Storm Fay on August 18-19, 2008; rainfall = 30 cm) and post-BMP3 (March 21-28, 2010; rainfall = 8 cm) periods. Overall, WWR resulted in higher retention of water and larger inundated areas during the post-BMP3 and post-BMP4 periods compared to the pre-BMP period.

Low water retentions during post-BMP1, post-BMP2, and post-BMP5 periods were caused by low flow resulting from dry conditions that prevailed during these periods. Average runoff depth

(16.5 cm) for all the post-BMP periods was lower than the runoff depth (44.2 cm) for the pre-BMP period (Table B-5).

Table B-5. Rainfall and runoff depths for the pre- and post-BMP periods at Wetland 2.

Period	Rainfall (cm)	Runoff depth (cm)	Runoff depth per unit rainfall
pre-BMP	147	44.2	0.300
post-BMP1	91	1.8	0.020
post-BMP2	109	1.8	0.016
post-BMP3	157	42.4	0.269
post-BMP4	160	29.43	0.209
post-BMP5*	53	7.30	0.151

*covers June-Oct, 2010

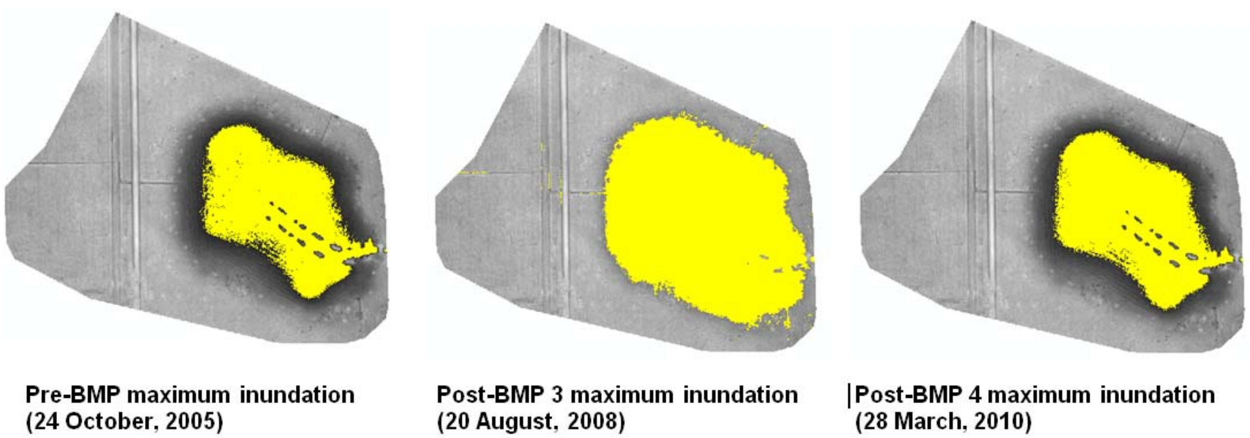

Figure B-4. Maximum inundations during pre-BMP, post-BMP3, and post-BMP4 periods at Wetland 2.

Phosphorus and Nitrogen Concentration and Load

Mean measured and mean flow-weighted TP concentrations during the post-BMP3 period increased by 19 and 2%, respectively, compared to the pre-BMP period (Table B-7). Mean measured and mean flow-weighted TP concentrations also increased by 43 and 23%, respectively, for the post-BMP4 period. Annual TP loads for the post-BMP3 (22%) and post-BMP4 (8%) periods were lower than the pre-BMP period (Table B-6). Even though TP loads decreased during all the post-BMP periods compared to the pre-BMP period, average TP concentrations were higher for the post-BMP periods (2.13 mg/L) compared to that for the pre-BMP period (1.53 mg/L). Less than average rainfall and runoff for post-BMP2 period was partly responsible for unusually high concentrations for this period and increased the average post-BMP TP concentration.

Mean measured and mean flow-weighted TP concentrations during post-BMP3 period increased by 19 and 2%, respectively compared to the pre-BMP period (Table B-7). Also during post-BMP4 period, mean measured and mean flow-weighted TP concentrations increased by 43 and 23%, respectively, compared to the pre-BMP period. Total P loads were lower during post-BMP3 (22%) and post-BMP4 (8%) periods compared to the pre-BMP period (Table B-7). Even though TP loads decreased during all the post-BMP periods compared to the pre-BMP period, average TP concentrations were higher for the post-BMP periods (2.13 mg/L) compared to those of the pre-BMP period (1.53 mg/L).

Similarly, TN load reductions were observed during post-BMP3 (21%) and post-BMP4 (8%) periods compared to the pre-BMP period. TN load reductions during the rest of the post-BMP periods (post-BMP1, post-BMP2, and post-BMP5) were mainly due to the low flow resulting from lower than average rainfall. Average TN load (129.7 kg) for the post-BMP periods was lower compared to that for the pre-BMP period (319.1 kg). Similar to TP, mean measured and mean flow-weighted TN concentrations for the five post-BMP periods were higher than those for the pre-BMP period. Total P and TN loads depend on the concentrations and flow volume. Reduced outflow from the wetland resulted in reduced TP and TN loads during post-BMP periods compared to the pre-BMP period (except TN load in post-BMP4). High TP and TN concentrations during some of the post-BMP periods could result from low flow volume due to WWR BMP. Due to the reduction in flow volumes during the post-BMP periods, the TP and TN concentrations could not have been sufficiently lowered by dilution to have lower than pre-BMP concentrations. Following the analysis of TP loads on annual basis was a statistical analysis of daily TP and TN concentrations to determine whether the changes in concentrations and loads were statistically significant.

Results from Proc GLIMMIX indicated that there was no statistical difference in the mean daily TP and TN concentrations for the pre-BMP and post-BMP periods. When mean daily TP loads were compared, the only statistically different result was that post-BMP4 TP load was higher than the pre-BMP TP load ($p<0.0001$). However, when only wet season data was analyzed, there was some evidence that post-BMP wet period mean daily TP loads were higher than the pre-BMP TP loads ($p=0.0962$). Post-BMP3 period could not be used in this analysis due to lack of flow and load data during that period's dry season. Similar to TP loads, there was significant increase in TN load during post-BMP4 period compared to pre-BMP period ($p<0.0001$). There was no evidence that mean daily post-BMP wet season TN loads were statistically different than the pre-BMP wet season TN load ($p=0.5103$). Overall, there was limited to no evidence for effect of WWR BMP on TP concentration and loads. Long-term data, especially the data for additional pre-BMP years, is needed before the effects of WWR can be quantified at Wetland 2. Only one year of pre-BMP data precludes us from accurately quantifying the WWR BMP effects.

Table B-6. Total Phosphorus (TP) and Total Nitrogen (TN) mean, and mean flow-weighted concentrations for the pre- and post-BMP periods at Wetland 2.

Period	Mean concentration (mg/L)		Mean flow-weighted concentration (mg/L)	
	Total Phosphorus	Total Nitrogen	Total Phosphorus	Total Nitrogen
pre-BMP	**1.53**	**3.78**	**1.87**	**2.95**
post-BMP1	1.16	2.02	0.97	1.00
post-BMP2	3.94	8.00	3.47	6.59
post-BMP3	1.82	3.58	1.90	3.96
post-BMP4	2.19	4.42	2.30	4.33
post-BMP5	1.47	2.92	1.76	1.96
post-BMP Average	**2.12**	**4.19**	**2.08**	**3.97**

Table B-7. Annual and seasonal (wet and dry) total Phosphorus (TP) and total Nitrogen (TN) loads for the pre- and post-BMP periods at Wetland 2.

Period	Rainfall (cm)	Total Phosphorus (kg)			Total Nitrogen (kg)		
		Total	Wet period	Dry period	Total	Wet period	Dry period
pre-BMP	**147**	**182.4**	**173.5**	**8.9**	**319.1**	**311.8**	**7.3**
post-BMP1	91	5.1	5.1	0.0	7.2	7.2	0.0
post-BMP2	109	22.6	22.6	22.6	51.9	49.1	2.8
post-BMP3	157	143.6	143.6	0.0	239.3	239.3	0.0
post-BMP4	160	166.9	82.2	84.7	319.3	170.0	149.2
post-BMP5	53	25.0	0.0	25.0	30.7	30.7	0.0
post-BMP average	**114**	**72.6**	**50.7**	**26.5**	**129.7**	**99.3**	**38.0**

Soil Phosphorus Storage Capacity

Soil samples were collected at 45 wetland and upland locations at the two wetland sites for different soil depths. The soil samples were analyzed for Soil Phosphorus Storage Capacity (SPSC). Soil with a negative SPSC value is a source of P while soil with a positive SPSC value is a sink for P. There are certain locations at Wetland 1 which are considered to be strong sources (termed 'hotspots') of P. These locations are characterized by soil with very low (negative) SPSC values. Most of the high negative SPSC soil profiles are located at and close to the cattle feeding area of Wetland 1. These locations are heavily impacted with P from manure-accumulation due to cattle congregations. Many of these hotspots were inundated during Tropical Storm Fay during post-BMP2 period. Even though some hotspots were not inundated during Fay, these were connected by ditches and/or swales to the wetland. Even a water table that is high yet doesn't reach the surface at those locations would likely result in P being dissolved and lost via runoff resulting in P distribution across surface soils. This explains the high N and P loads during post-BMP2 period at Wetland 1.

WWR BMP may result in increased connectivity of the ponded areas in the upland to the wetland which can favor transport of P and N available on surface through overland flow rather than subsurface flows. Such change in the hydrology has the potential of increasing the P and N loads contributed by a wetland to downstream.

Wetland 2 does not have high P hotspots as determined by the SPSC analyses. Additionally, the absence of swales in the vicinity of Wetland 2 resulted in reduced connectivity of the wetland to the upland area during storm periods. Figure B-3 and Figure B-4 show the inundated areas for the highest wetland water levels at Wetland 1 (pre-BMP1, post-BMP2, and post-BMP3 periods) and Wetland 2 (pre-BMP, post-BMP3, and post-BMP4 periods), respectively.

Evaluation of Wetland and Pasture Retention using Hydrologic Models

Although WWR may increase the surface and subsurface storage in the wetland drainage area, part of this storage is likely to move out through subsurface pathways. Enhancing subsurface flows, though a desirable outcome from the point of view of reducing peak flow, the true water retention is only achieved through increased ET due to higher areal extent of ponded water and near saturation soil moisture. However, verification of such increases in ET requires accurate quantification of ET. The most common approach to estimating ET is to use one of the available

ET models such as Priestley-Taylor, Penman-Monteith and Blaney-Criddle. Equally important is the accurate quantification of subsurface flow gains and losses from water retention areas.

Hydrologic models are useful for evaluating a variety of scenarios that are not otherwise feasible to be tested using field-collected data. Given the above stated importance of ET and subsurface flow components, use of any model for scenario evaluation must ensure that the model accurately predicts both of them. Most hydrologic models use potential ET data as input and combine it with a user supplied crop (vegetation) coefficient. Data on vegetation coefficients is limited for wetland systems. Furthermore, using vegetation coefficient developed for one hydro-climatic region and applying it for another inherently introduces an unknown level of uncertainty. If region-specific, field-measured ET data is available it can be used to better calibrate the hydrologic models for improved simulation. In-situ measurement of wetland ET using the traditional approach of lysimetry is very difficult, if not impossible. Eddy covariance technique offers a suitable alternative for obtaining relatively accurate ET estimates. An eddy covariance station was installed at Wetland 1 to measure the required micro-meteorological parameters for estimating ET. Eddy based ET estimates for July 2009 to June 2010 was 1140 mm while the estimates from the FAO Penman-Monteith method (using literature vegetation coefficient values) was 936 mm. Such difference between the two estimates clearly shows the importance of accurate ET quantification with regards to retention quantification of both field-measured and modeled WWR scenarios. Details of eddy covariance ET and its comparison to Penman-Monteith are presented in Appendix E.

Watershed Assessment Model (WAM) was used in conjunction with hydrologic data from Wetlands 1 and 2 to simulate various water retention scenarios. These scenarios included 1) increase current board height by 15 cm; 2) increase current board height by 30 cm in the dry season (November-May) and 15 cm in the wet season (June-October); 3) no board (pre-BMP) condition; 4) and increase current board height by 30 cm. Unlike the conclusions we were able to obtain from the field-measured data, the results from the WAM simulations showed consistent increase in water retention (reduced runoff) as board heights were increased. However, due to some of the limitations mentioned above regarding ET estimates used in the model and the lack of comprehensive groundwater component in WAM, results from WAM should be used with caution. For example, WAM predictions of runoff for Tropical Storm Fay were 76 % less than the observed runoff. As stated above, Fay accounted for 81% of the annual runoff from Wetland 1 for the post-BMP2 period while WAM predictions for the same storm were only 23% of the annual flow. Details of WAM predictions for the above scenarios are presented in Appendix F.

Efforts should be made to use a hydrologic model with strong surface and groundwater components such as Mike-SHE/Mike 11 (DHI, 2011) or a modified WAM that can accurately simulate ET as well as surface and subsurface fluxes.

Conclusions

Based on this study, the following observations can be made:

1. Although water retention on ranchlands using a combination of wetland and pasture as water storage areas was a promising BMP, both with regards to water and nutrient retention, data collected from this project do not necessarily support these assumptions.

2. WWR involves interaction of surface and subsurface water and nutrient processes which when combined with natural climatic variability makes it difficult to attribute the observed changes in water and P dynamics to this proposed BMP alone.

3. Long-term data comprising multiple years of pre- and post-BMP periods is necessary to detect statistically significant changes due to WWR implementation. Although six years of monitoring data was used to quantify WWR effects, the unequal number of pre- and post-BMP periods and climatic variability makes it difficult to draw conclusions on the effects of WWR.

4. The evaluation of WWR performed in this study is interim at best. In fact, the results of this study show that there is likelihood of increasing TP concentration and loads from implementation of WWR, especially if soil/hydrologic conditions that favor both increased overland flow and soil P release.

5. The WWR monitoring study at Pelaez should continue for at least three years to capture additional data under pre-BMP conditions. Advantage of additional pre-BMP data in improving the quantification of BMP effects was clearly seen for the DFCC BMP. However, DFCC mainly involved surface soil and water processes while both surface and subsurface processes are important for WWR which may result in the effects of WWR on

TP loads not detectable before several years after WWR implementation. This further necessitates the need for long-term data for evaluating this important proposed BMP.

6. Use of hydrologic models for evaluating water aspect of WWR with acceptable accuracy is possible especially with the use of eddy based ET estimates and accurate representation of subsurface processes in the model. However, effects on P concentration and loads are not so easily achievable through the use of models.

7. The two wetlands considered in this study capture a partial range of the physical (soil, topographic, hydrologic, plant) variability present in the wetland/upland systems located at ranchlands in the northern Everglades basin. Results indicated differences in response to the WWR implementation such that runoff and TP loads seemed to have increased at Wetland 1 while decreasing at Wetland 2.

Appendix B-2

Effects of Passive Hydration on Wetland Hydroperiod, Water Storage and Ponded Area in the Lake Okeechobee Basin

Introduction

The Northern Lake Okeechobee watershed in Florida comprises an area of about 5,160 km^2 and contributes nutrient-enriched runoff to Lake Okeechobee (LO), the second largest freshwater body located wholly within the continental USA (Hiscock et al. 2003). The watershed has been impacted by a variety of human activities. An extensive system of canals and levees, constructed primarily during the first half of the last century, has disrupted the natural flow and altered the hydrology of the watershed (DeBusk et al. 1994). Large areas in the watershed have been drained for agriculture and urban development. The drainage has resulted in rapid fluctuations (unnaturally high and low) in the LO water level and high phosphorus (P) loads. Damaging discharge to LO during high peak flow wet season (June-October) is released to the Caloosahatchee and St. Lucie estuaries in the west and east coasts of Florida, respectively. This freshwater has adversely affected the salinity and subsequently the flora and fauna of the estuarine ecosystem.

Cattle ranching is the dominant land use (36% by area) in the LO watershed and wetlands on ranches are considered important landscape features for storing water and nutrients in the LO watershed. Wetlands account for 15% of total land area in the LO watershed (Tweel and Bohlen, 2008). Wetlands in the cow-calf ranches with improved pastures (i.e. pastures planted with forage species, fertilized and maintained for high productivity) contribute substantially to water and soil P storage in this watershed (Dunne et al. 2007). However, a large fraction of these wetlands have been partially or completely drained. Rehydration of these on-ranch wetlands can reduce the flow and P loads to LO by increasing wetland water storage.

The Comprehensive Everglades Restoration Program (CERP) which started in the year 2000 aims at addressing the complex hydrologic, environmental, and societal issues of water control in the greater Everglades Ecosystem of south Florida and the interconnected Kissimmee River-Lake Okeechobee-Everglades Ecosystem (Marshall III et al. 2009) (Figure B-5). The primary goal of the CERP is to restore the timing, quantity, quality, and distribution of freshwater within the greater Everglades ecosystem which includes LO and its tributary areas, so that the ecosystem conditions approximate pre-development conditions as closely as possible (Wolfert-Lohmann et al. 2008). Passive hydration (Means and Means, 2008) of on-ranch wetlands can be used to increase surface storage in the Northern Everglades watershed. Wetland passive hydration is a method to rehydrate a wetland by controlling water outflow from the wetland. Passive hydration of wetlands located in the ranchland is promoted by the state agencies as a BMP to achieve water and P retention. It is also one of the objectives of the CERP to increase water storage in the Northern Everglades watershed (Evergladesplan 2010). However, the effectiveness of passive hydration on wetland hydroperiod and water storage has not been quantified.

Wetland hydroperiod is defined as the period of time (days) a wetland holds water during an annual hydrological cycle (Snodgrass et al. 2000). Hydroperiod is the major factor influencing the unique vegetation patterns of the wetlands, and is a strong determinant of community composition, standing stock, and survival of fish and invertebrate species (Bruno et al. 2001). The ecological characteristics of a wetland are controlled by hydroperiods and increases in hydroperiod and water level in a wetland could also enhance its habitat value (Ewel, 1990). The hydroperiod varies from year to year and from wetland to wetland depending on the underlying geology, soil characteristics, depth, and size. Precipitation has been shown to be an important controlling factor for hydroperiods of the wetlands in South Carolina (Lide et al. 1995, Kennamer 2001), Minnesota (Palik et al. 2001), Maine (Joyal et al. 2001), and Florida (Mansell et al. 2000) (Brooks, 2004). Assessing and understanding wetland hydroperiod are essential for making management decisions in minimizing or avoiding loss or degradation of wetlands that provide breeding habitat for amphibian/marine life (Tarr and Babbitt 2009). Snodgrass et al. (2000) in their study in the Savannah River Site, a U.S. Department of Energy facility on the upper coastal plain of South Carolina, observed that short hydroperiods occur predominantly among smaller wetlands such as those found in south Florida ranchland, indicating that preservation of smaller wetlands is essential for the conservation of many wetland-associated species. They recommended that wetland regulations include wetland hydroperiod as a criterion for assessing wetland function.

Estimation of hydroperiods in a wetland involves the assessment of the number of days when there is water in the wetland. This can be carried out by monitoring surface water levels in the wetland. Brooks and Hayashi (2002) calculated an index as the ratio of the number of wetland visits when the wetland held standing water to the total number of visits to the wetland to study hydroperiods. They did not use surface water wells to monitor wetland water levels. The hydroperiod indices were calculated for 34 wetlands during 1998-2000 in central Massachusetts. The hydroperiod index did not indicate the full length of hydroperiod, as the wetlands were only visited for part of each year. However, as all wetlands were visited for the same length of time each year, the index provided a rough but unbiased indication of the variation in hydroperiod in the wetlands. Brooks (2004) used a multiple regression model to estimate hydroperiods with potential evapotranspiration and precipitation as variables. This model is a simplification of the climate-water budget equation (Lide et al. 1995).

Passive hydration may be used to increase wetland hydroperiod, water storage (volume of water contained in the wetland as surface water) and ponded (inundated) area. In Florida, state agencies like Water Management Districts (Means and Means, 2008) are encouraging adoption of passive hydration for wetland augmentation. But its effects on hydroperiod, water storage, and ponded area have not been quantified. There is a need to quantify the effects of passive hydration for basin-wide adoption of this approach in the LO watershed. Wetland hydroperiod can be estimated using the surface water levels at the wetland using surface water wells. To accurately estimate hydroperiod, water storage, and ponded area in a wetland using wetland water depths , accurate information related to the topography of the wetland and the upland flow contributing area is essential.

Digital Elevation Model (DEM) topographic data along with water depth can be used to derive wetland water storage and ponded area. Commonly, the elevation data available from the National Elevation Dataset (NED) of the United States Geological Survey (USGS) is used for characterizing the topography and hydrological attributes of wetlands. These USGS topographic survey information are of insufficient resolutions (usually 30 m) to demarcate depressions like wetlands and ditches very accurately (Colson et al 2006; Trettin et al. 2009). Trettin et al. (2009) in their study in U.S. Forest Service Santee Experimental Forest (SEF) in South Carolina observed that the USGS DEMs could not represent the stream lengths and the sinuosity (a ratio that describes whether a channel is straight or meandering) very accurately. James and Hunt (2010) observed that the USGS DEMs could not delineate the flow channels properly in a small watershed in South Carolina. The landscape in south Florida is characterized by relatively flat topography and the use of low resolution USGS DEMs to delineate the wetlands and ditches may lead to inaccuracy in estimating the surface water storage and ponding areas. Currently, many local jurisdictions have acquired Light Detection and Ranging (LIDAR) data, an increasingly important data source for high resolution DEMs. Resolutions of LIDAR data are, in general, higher than other DEM sources (Li and Wong 2010). DEM quality and resolution affect the accuracy of any extracted hydrological features including wetland hydroperiod and water storage capacity (Kenward et al. 2000). The high quality DEMs lead to detailed terrain and hydrological attributes with high accuracy. Water resource management and hydrological modeling require high quality DEMs (Zhao et al. 2010). LIDAR technology allows rapid and inexpensive measurements of topography over large areas (Zhang et al 2003). This technology is becoming a primary method for generating high-resolution topographic data that are essential to numerous applications such as flood modeling and landslide prediction. Even though LIDAR technology is more expensive than some other technologies for remote sensing at present, sometimes it provides the only appropriate option to delineate flood mapping in the flat plains (Sanyal and Lu 2004) of south Florida. It is widely recognized that a national LIDAR dataset will benefit many state and federal agencies such as the Federal Emergency Management Agency's (FEMA) flood map modernization effort (NRC 2007). The National LIDAR Mapping Initiative, also known as the Elevation for the Nation (EFTN) program, is an effort to collect accurate, seamless, high-resolution elevation data for the 50 states through implementation of medium-altitude, advanced technology (James and Hunt 2010).

Original LIDAR data offer a single high resolution DEM, from which multiple lower resolution data can be derived by resampling it. By resampling, a DEM with a specific resolution can be converted to one with a different resolution (Li and Wong 2010). To compare two DEMs spatially, the DEMs have to have the same spatial resolution. This can be accomplished by resampling one of the DEMs so that they have the same resolution (Wang and Zheng 2005).

There is a need to quantify the effect of wetland passive hydration on water storage and ponded area and evaluate the reliability of topographic data in estimating the effects of passive hydration. The objectives of this study were: 1) to quantify the effects of passive hydration of historically drained on-ranch wetlands in south Florida on hydroperiod, water storage and ponded area; and 2) evaluate the topographic data related errors introduced in evaluating the passive hydration effects by comparing the storage, hydroperiod, and ponded area estimates from the commonly used USGS DEM and LIDAR DEMs. To achieve objective 1, high resolution LIDAR (1 m) DEMs for the two wetlands were used while for objective 2, 30 m USGS and 30 m LIDAR DEMs (resampled from 1 m LIDAR DEMs) were used.

Figure B-5.. Central and south Florida showing the interconnected Kissimmee River-Lake Okeechobee-Everglades Ecosystem (Courtesy: South Florida Water Management District).

Materials and Methods

Passive hydration was implemented at two wetlands (wetlands 1 and 2) (Figure B-6) in a beef-cattle ranch in the LO basin. The areas of the wetland sites are 0.80 and 0.24 km^2, respectively. These areas include the wetland areas in addition to the flow contributing upland areas. The wetland and the flow contributing upland area together will be referred to as drainage area hereafter. Wetland 1 includes two drainage ditches which are 669 and 332 m in length and they converge and flow together for 147 m before exiting the wetland. The average width and depth of the ditches at Wetland 1 are 4.60 and 0.53 m, respectively. Average elevations for wetlands 1 and 2 are 8.54 m and 9.31 m above mean sea level, respectively. All elevations in this study were measured using North American Vertical Datum of 1988 (NAVD 88). Wetland 2 contains a single drainage ditch which flows in south-easterly direction for about 282 m before exiting the ranch. The average width and depth of the ditches at Wetland 2 are 4.15 and 0.21 m, respectively.

Passive hydration was provided by installing a riser board (flashboard riser) structure at the outlet (culvert) of the wetland and adding boards until the desired water retention level was achieved. Since the hydration of wetland can cause in the inundation of improved pasture areas in the upland and result in economic loss to the rancher, the water retention level was decided based on the comfort level of the rancher. The goal of using rancher-based retention approach is that for wetland hydration to be successful in the LO watershed, the wetland hydration must coexist with cattle ranching. The flashboard riser allows water to flow only when the water level exceeds the top elevation of the flashboard, thereby retaining additional water in the wetland. Figure B-7 shows Wetland 1 during pre- and post-hydration periods. The elevations of the flashboard riser tops at wetlands 1 and 2 were set at 9.06 and 9.47 m. The bottom elevations of the two wetlands were 7.95 and 8.97 m, respectively. Two surface water wells (one in the middle of the wetland and the other near the wetland outlet) were continuously (at 15-min interval) monitoring wetland water levels using pressure transducers for the entire monitoring period (May 2005-April 2010). Three groundwater wells were installed in the upland area close to each wetland to monitor groundwater levels continuously (at 15-min interval) using pressure transducers (Figure B-6). Surface and groundwater level data were collected every 15 minutes. To quantify soil water storage in the wetland-upland areas (vadoze and saturated zones), Enviroscan sensors (Sentek Technologies, Australia), capacitance-based soil moisture sensors, were installed in the wetlands. Ten such sensors were placed 10-20 cm apart in a PVC tube to cover a soil depth of 150 cm.

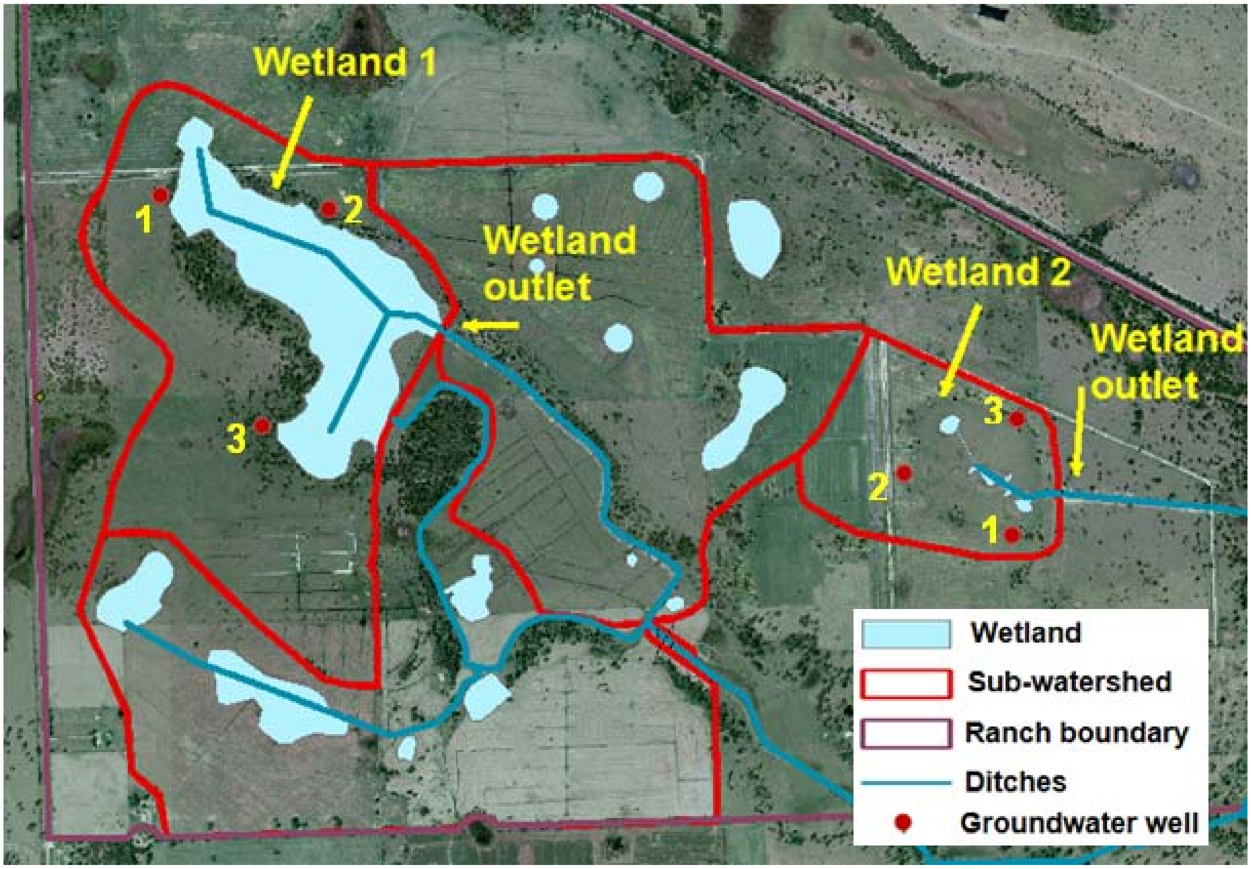

Figure B-6. The beef-cattle ranch showing the two wetland sites, wetland outlets, and the location of groundwater wells.

Hydroperiods, water storages (volumes of surface water stored in the wetland drainage area) and ponded (inundated areas in the wetland drainage areas) areas for the two wetlands were compared for the pre- and post-installation periods of passive hydration structures to study the effectiveness of passive hydration. Pre- and post-hydration periods for the two wetlands are shown in Table B-8. There was large rainfall variability between the monitoring periods during 2005-10 at the wetland sites. The years 2006 and 2007 were dry periods in Florida with very little water in the wetlands and ditches, and therefore, these two periods were excluded from the analysis. LO experienced the lowest water level in 2007 based on historical data collected by the South Florida Water Management District (SFWMD 2010).

Table B-8. Pre- and post-hydration periods for wetlands 1 and 2.

Period	Duration
Pre-hydration period	May 1, 2005 - April 30, 2006 (Pre-hydration)
Post-hydration period	May 1, 2008 - April 30, 2009 (Post-hydration 1) May 1, 2009 - April 30, 2010 (Post-hydration 2)

Figure B-7. Wetland 1 outlet with only the culvert during the pre-hydration period (left) and with the culvert and flashboard riser during the post-hydration period (right).

LIDAR DEMs (1m) for the wetlands sites were used to calculate the hydroperiods, water storages and ponded areas in the two wetlands. Hydroperiod for a wetland was estimated as the total number of days for which the wetland water level was at least 15 cm. This was the criterion used by Bruno et al. (2001) to calculate hydroperiods in seasonal wetlands of Everglades National Park, Florida, USA. A macro was written in ArcGIS v.9.3.1 (ESRI, Redlands, California) to estimate wetland water storage and ponded areas. The macro uses wetland water depths (measured at the surface water wells) and the topographic data (DEM) to estimate wetland water storages and ponded areas. To estimate the average daily water storage and ponded area for a period, first the daily water storage and ponded area were estimated based on daily (average of all 15-min data) wetland depth for the hydroperiod constituting days and then averaged them for the days constituting the hydroperiod.

To identify the wetland bankfull stage is important because it is the water level at which the flow fills the wetland and begins to inundate the upland (floodplain) (Johnson and Padmanabhan 2010). Bankfull stage approximates the water level at which the wetland and the surrounding upland area become connected through surface flows. The bankfull stages for the two wetlands were analyzed to study the interaction of flow between wetland and the surrounding upland areas that contributed surface and subsurface flows to the wetland. Passive hydration may help in achieving the bankfull stage and when it occurs, this may reduce flow to downstream outlet. Bankfull stage also results in larger ponded area which enhances water loss by evapotranspiration reducing downstream flow. There are several methods available in literature for estimating bankfull stage. These estimations are based on the sediment size location, aerial photographs, and type of vegetation etc, but these methods have their own limitations (Williams 1978; Jurmu and Andrle 1997) and therefore, may not be very accurate. In this study, the bankfull stages for the two wetlands were estimated using LIDAR topographic data.

Hydroperiods, water storages, and ponded areas for the two wetland sites were estimated using the original LIDAR (1m), resampled LIDAR (30 m), and the publically available non-LIDAR USGS (30 m) DEMs for comparing the hydroperiods, water storages, and inundated areas derived from LIDAR and non-LIDAR DEMs.

Results and Discussion

Passive Hydration Effects on Hydroperiod, Water Storage and Ponded Area

Wetland 1

At Wetland 1, the maximum and average daily depths (for all days constituting the hydroperiod) of water during the post-hydration 1 period were 38 and 31% higher than those during the pre-hydration period (Table B-9). The maximum and average daily depths of water during the post-hydration 2 period were 1 and 30% higher than those during the pre-hydration period. Average daily water stored at the Wetland 1 site increased by 185 and 121% during post-hydration 1 and post-hydration 2 periods compared to the pre-hydration period. Passive hydration also increased the ponded areas. Maximum and average daily ponded areas during post-hydration 1 period were 55 and 136% larger than the pre-hydration period. Maximum and average daily ponded areas during post-hydration 2 period were 2 and 130% larger than the pre-hydration period.

The hydroperiods for the post-hydration 1 and post-hydration 2 periods were longer than the hydroperiod for the pre-hydration period. The hydroperiod is affected by the rainfall distribution patterns (Figure B-8). If equal amounts of rainfall occur during two years, the year in which the rainfall is more evenly distributed over the entire year is expected to have longer hydroperiod. The Pre-hydration and post-hydration 2 periods experienced similar amounts of rainfall and the rainfall was more evenly distributed over the respective periods compared to pre-hydration period (Figure B-8). Also post-hydration 1 period rainfall was lower compared to the post-hydration periods. Despite that, post-hydration 1 period experienced longer hydroperiod (above 15 cm wetland depth) compared to the pre-hydration period. This longer hydroperiod was attributed to the passive water retention at the wetland. In spite of the lowest rainfall, average water depths were the highest during post-hydration 1 period, even though the hydroperiod was not the longest during that period.

Table B-9. Wetland 1 average daily hydroperiod, water storage, and ponded area.

Period	Rainfall (cm)	Hydro-period (day)	Maximum depth (m)	Average depth (m)*	Average water storage (m^3)*	Max. ponded area (m^2)	Average ponded area (m^2) *
Pre-hydration	163.4	314	4.856	2.36	5,468	203,702	26,884
Post-hydration 1	135.1	320	6.695	3.08	15,594	316,312	63,370
Post-hydration 2	165.6	342	4.910	3.06	12,092	208,490	61,747

* for the days constituting the hydroperiods

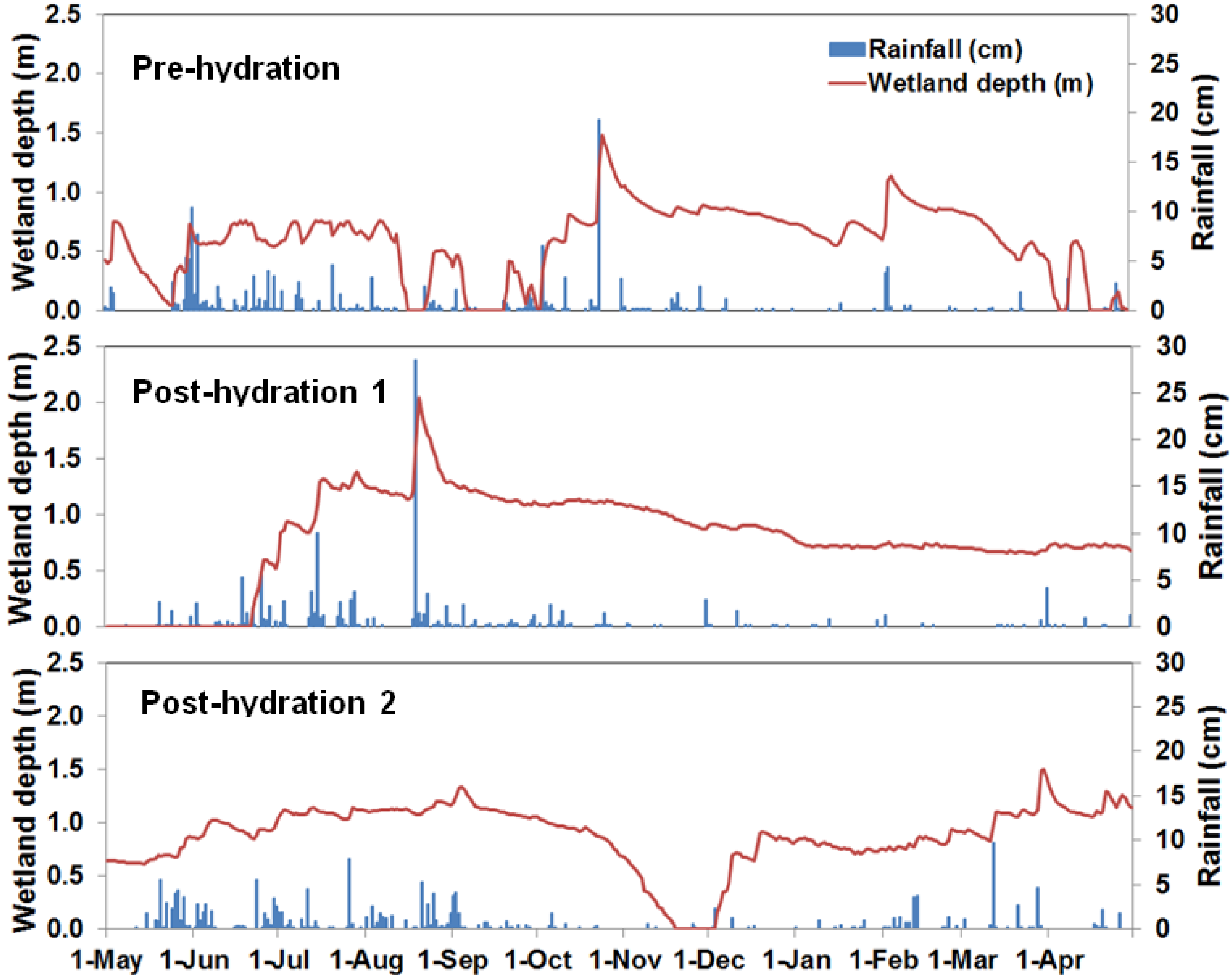

Figure B-8. Wetland 1 water depths and rainfall for the pre-hydration, post-hydration 1 and post-hydration 2 periods.

Wetland 2

The maximum and average daily depths of water during post-hydration 1 period were 123 and 118% higher than those for the pre-hydration period (Table B-10). The maximum and average daily depths of water during post-hydration 2 period were 43 and 120% higher respectively, than those for the pre-hydration period. The average daily water storages for post-hydration 1 and post-hydration 2 periods were 520 and 146% higher than the storage for the pre-hydration period. The maximum and average daily ponded (inundated) areas for the post-hydration 1 period were 1058 and 366% larger than those for the pre-hydration period. The maximum and average daily ponded areas for the post-hydration 2 period were 419 and 137% larger than those for the pre-hydration period. Results show that passive hydration was successful in increasing water storage and ponded areas at Wetland 2.

Table B-10. Wetland 2 average daily hydroperiod, water storage, and ponded area.

Period	Rainfall (cm)	Hydro-period (day)	Maximum depth (m)	Average depth (m)*	Average water storage (m^3)*	Max. ponded area (m^2)	Average ponded area (m^2) *
Pre-hydration	163.4	130	1.780	0.513	155	6,912	1,202
Post-hydration 1	135.1	126	3.971	1.117	961	80,071	5,605
Post-hydration 2	165.6	223	2.540	1.130	382	35,870	2,849

* for the days constituting the hydroperiods

At Wetland 2, the hydroperiod for the post-hydration 1 period was shorter than that for the pre-hydration period (Table B-10). Post-hydration 1 period had less rainfall compared to the pre-hydration period, and the rainfall was less evenly distributed over the entire period compared to the pre-hydration period (Figure B-9). This resulted in some dry periods (months) within the period (post-hydration 1). The dry conditions which started in the year 2007 continued until May 2008 resulting in very low groundwater level, and ponding in the wetland started only in the last week of June 2008 (post-hydration 1 period). Post-BMP2 hydroperiod was 72% longer compared to pre-BMP period even though rainfall amounts were similar for both periods.

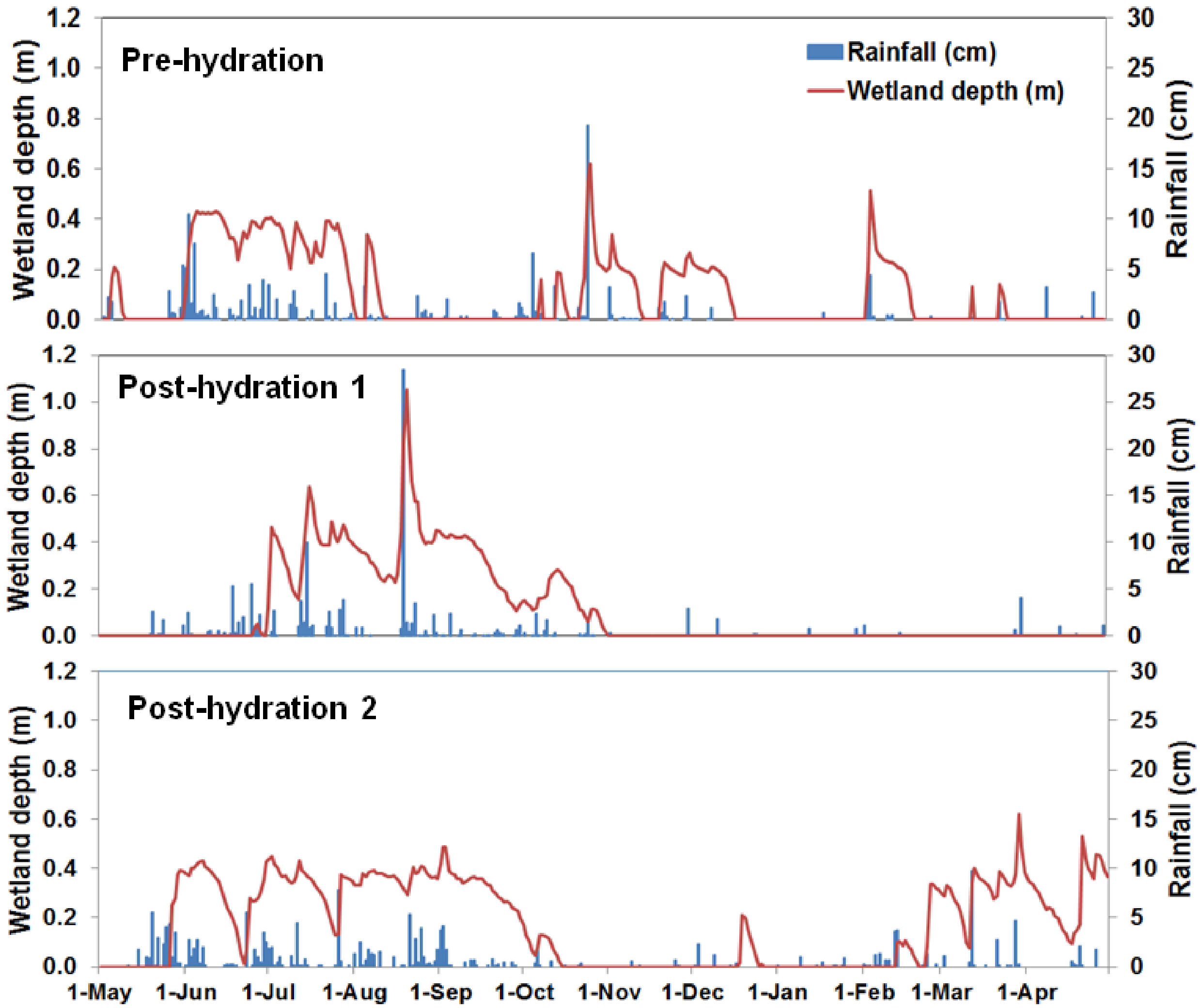

Figure B-9. Wetland 2 water depths and rainfall for the pre-hydration, post-hydration 1 and post-hydration 2 periods.

Effect of Water Retention on Bankfull Stage for the Two Wetland sites

At bankfull stage, the wetland contributes surface flow to the upland area and that reduces flow to downstream outlet. Water retention due to passive hydration aids in achieving bankfull stage. Larger ponded area due to bankfull stage would also result in higher evapotranspiration losses from the wetland, thereby further reducing downstream flow from the wetland. There are variations in the bankfull stages for the two wetland sites because the wetland bottom elevations and depths were different for the wetlands. The wetland bottom elevation and the bankfull stage for Wetland 1 are 7.95 and 9.80 m, respectively. The water depth at Wetland 1 needs to be about 1.75 m to reach the bankfull stage. The bankfull stage inundates 0.26 km2 (35.10 %) of Wetland 1 drainage area as determined using LIDAR topographic data in ArcGIS (Figure B-10).

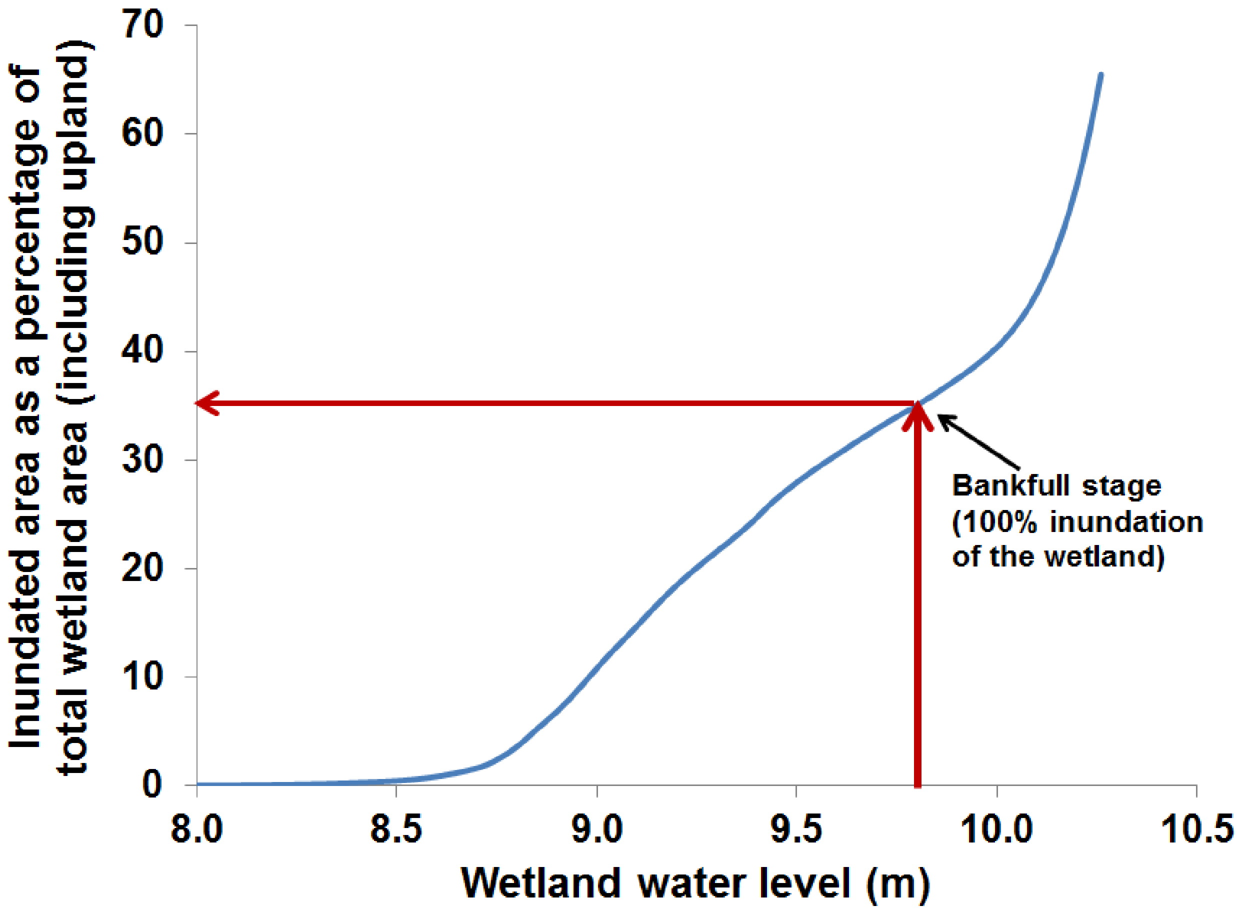

Figure B-10. Inundated area at Wetland 1 site as percentage of total wetland drainge area vs. wetland water level.

Wetland 1 reached bankfull stage on 19-24 August 2008 during tropical storm Fay. During this period, the wetland contributed flow to the upland area. The upland also contributed flow to the wetland during this period through subsurface environment which but that is a slower process than surface flow contribution by the wetland to the upland. Figure B-11 shows the wetland water and groundwater levels at Wetland 1 drainage area during tropical storm Fay. At the start of the storm on 18 August, the wetland water level was lower than the groundwater level. As time elapsed, the wetland water level became higher than the groundwater level. Wetland water retention aided in attaining the bankfull stage. Passive hydration resulted in high wetland water level during a storm that occurred in July 2008 (one month prior to Fay) (Figure B-8) and this water retention enhanced the water retention during Fay resulting in the bankfull stage.

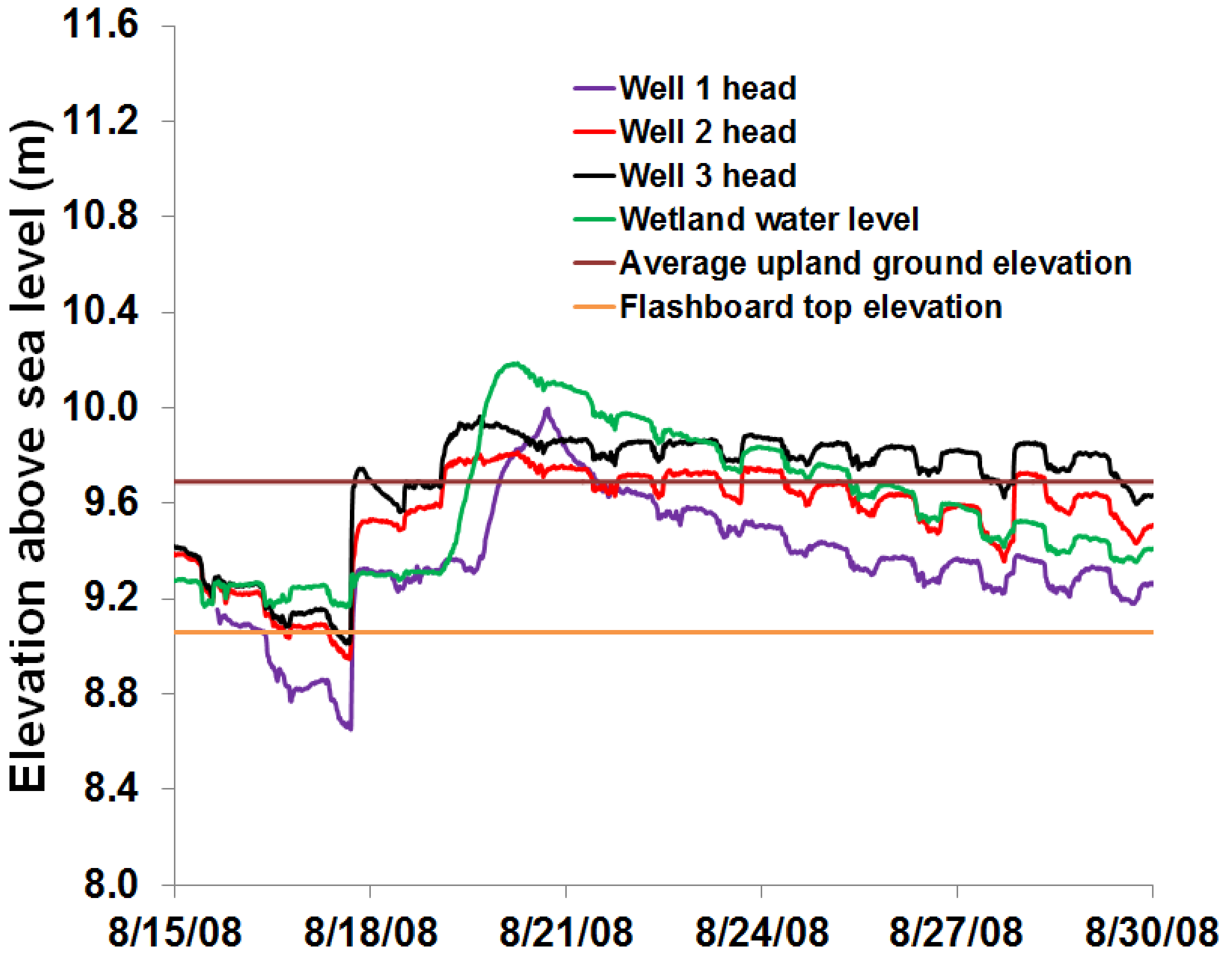

Figure B-11. Surface water and groundwater levels at Wetland 1 on 15-30 August 2008 (post-hydration 1) which included the bankfull stage,

Figure B-12 shows the inundated areas (in green color) at Wetland 1 for water levels at culvert bottom (pre-hydration period), flashboard riser top (post-hydration period) and bankfull stage. It can be seen that without passive hydration, natural storage capacity of the wetland is low. Wetland 1 was in bankfull stage for 3 days (post-hydration 1 period). At Wetland 1, the water was flowing over the flashboard riser for 91(hydroperiod = 320 days) and 73 (hydroperiod = 342 days) days during post-hydration 1 and post-hydration 2 periods, respectively.

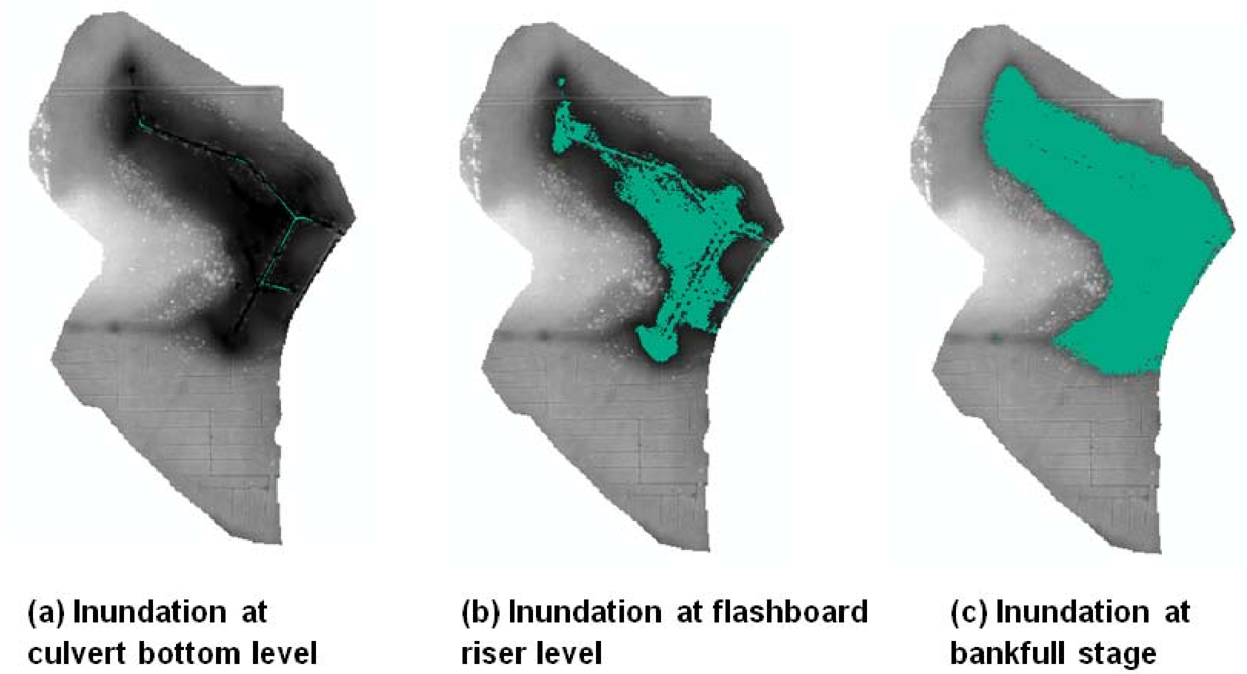

Figure B-12. Inundated areas (green color) at Wetland 1 for water (a) at culvert bottom level (b) flashboard riser top level, and (c) bankfull stage.

The wetland bottom elevation and bankfull stage for Wetland 2 were 8.97 and 10.10 m, respectively. The water level at the wetland needs to be about 1.13 m to reach the bankfull stage. For Wetland 2, 0.07 km^2 (30.3% of the wetland area) of the total wetland area gets inundated at bankfull stage (Figure B-13).

At Wetland 2, water level was at the bankfull stage on 19-21 August 2008 during tropical storm Fay. The flashboard aided in achieving the bankfull stage. During this period, wetland water level was higher than that the groundwater level in the upland area as recorded in at least one groundwater well (Figure B-14). Surface connectivity was established between the wetland and portion of upland areas and there was surface water interaction between wetland and upland.

At the beginning of the storm (18 August 2008), upland groundwater levels were higher than wetland water levels and then the upland started to contribute to the wetland through surface (ditches) and subsurface flows, and only then the wetland water level attained bankfull stage.

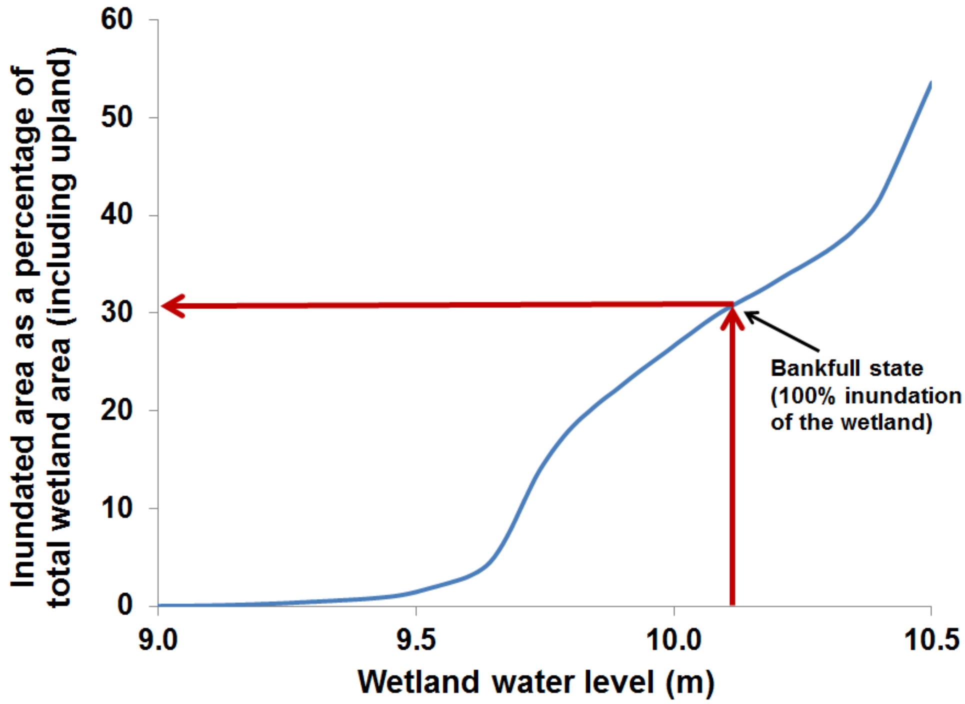

Figure B-13. Inundated area at Wetland 2 site as percentage of total wetland drainge area vs. wetland water level.

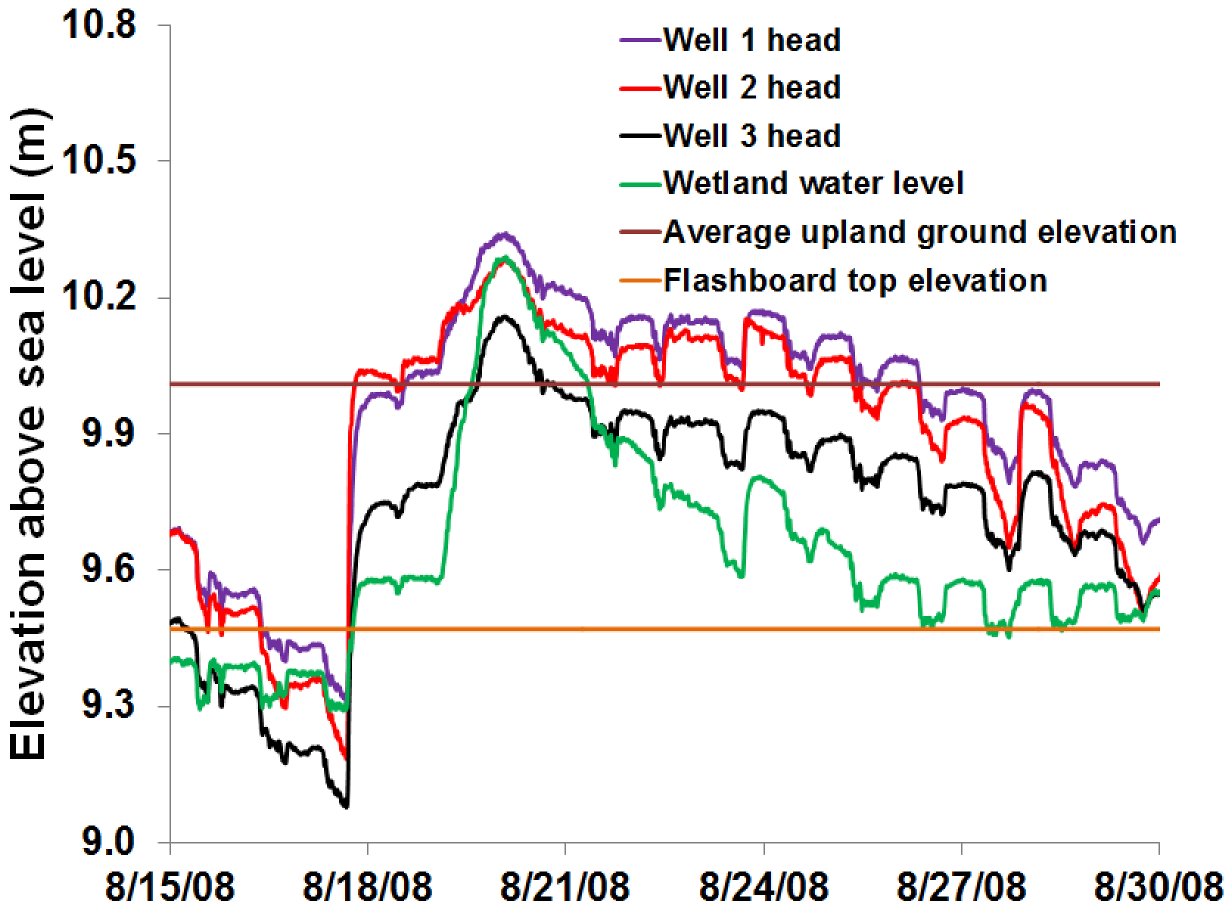

Figure B-14. Surface water and groundwater levels at Wetland 2 on 15-30 August 2008 (post-hydration 1) which included the bankfull stage.

Figure B-15 shows the inundated areas (in green color) at Wetland 2 for water levels at culvert bottom (pre-hydration period), flashboard riser top (pre-hydration period) and bankfull stage. Wetland 2 was in bankfull stage for 2 days (post-hydration 1 period). The water was flowing over the flashboard riser for 13 (hydroperiod =126 days) and 104 (hydroperiod = 223 days) days during post-hydration 1 and post-hydration 2 periods, respectively. It can be seen that without passive hydration, natural storage capacity of the wetland is low.

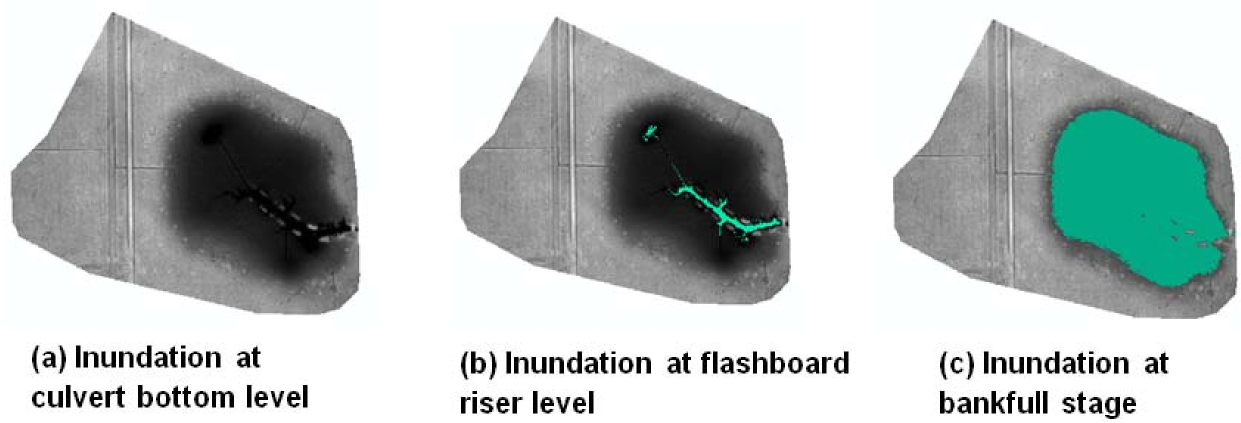

Figure B-15. Inundated areas (green color) at Wetland 1 for water (a) at culvert bottom level (b) flashboard board top level, and (c) bankfull stage.

Wetland Passive Hydration Effectiveness Per Unit Rainfall

In Florida, the water-year starts in May 1 (SFWMD 2009) which coincided with the beginning of a period (pre- and post-hydration) for this study. Higher subsurface and surface water in the wetland drainage area at the beginning of the water-year results in faster filling up of the available soil pore space and reduces the time to start surface flows from wetlands at the beginning of the water-year. Therefore, in addition to rainfall amounts, the antecedent soil water storage (water/moisture amounts in the sub-soil before the start of a water-year) has an important role to play in hydroperiod length and surface water storage in the wetland. The sub-soil moisture amount at the beginning of each period was quantified and it was added to the total rainfall for that particular period (pre- or post-hydration period) to determine the total moisture/water amount (referred to as adjusted rainfall hereafter) for that particular period. The moisture amounts recorded by the enviroscan sensors in the vadoze and saturated zones were used to determine the antecedent sub-soil water storage (moisture/water). The average depth to the groundwater from the ground surface at the two wetland sites on 1 May for the three years was approximately 150 cm. To account for the variability in antecedent soil water storage, the antecedent soil water storages for a 150 cm soil column for 1 May of 2005, 2008, and 2009 were added to the rainfall amounts for the respective periods. Average daily water storages (for the days constituting the hydroperiod) for pre- and post-hydration periods were normalized using this adjusted rainfall amounts for both wetlands (Table B-11). At Wetland 1 site, the water storages per cm of rainfall were 240 and 128% higher for post-hydration 1 and 2 periods, respectively than the storage for the pre-hydration period. At Wetland 2 site, water storages per cm rainfall were 682 and 157% higher for post-hydration 1 and 2 periods, respectively than the storage for the pre-hydration period. Wetland 1 has larger area compared to Wetland 2 and therefore, has larger storage per unit rainfall than Wetland 2.

Table B-11. Adjusted rainfall-normalized surface water storages (storage per cm of rainfall) at wetland sites 1 and 2.

Period	Normalized water storage (m^3)*	
	Wetland 1	Wetland 2
Pre-hydration	25.8	0.84
Post-hydration 1	87.6	6.57
Post-hydration 2	58.8	2.16

*For the days constituting the hydroperiods

Comparison of Wetland Hydroperiods, Water Storages, and Ponded Areas Derived From LIDAR and Non-LIDAR DEMs

To compare the hydroperiods, water storages and ponded areas, these parameters derived from resampled LIDAR (30 m) and non-LIDAR USGS (30 m) DEMs were used. For comparison purpose, the LIDAR and non-LIDAR DEMs should have same resolution (Liu et al 2005). For Wetland 1, USGS (30 m) DEM represented a smaller and shallower footprint for Wetland 1 compared to LIDAR DEMs (both 1 and 30 m) which depicted the topography of the wetland site well. The elevation ranges (difference between highest and lowest elevation points) for Wetland 1 drainage area based on the original LIDAR, resampled LIDAR and USGS DEMs were 3.77, 2.82, and 0.29 m, respectively (Figure B-16). For Wetland 2 site also, the USGS DEM represented a smaller and shallower footprint for the wetland compared to LIDAR DEMs (both 1 and 30 m). The elevation ranges for Wetland 2 drainage area based on the original LIDAR, resampled LIDAR and USGS DEMs were 2.06, 1.65 and 0.42 m, respectively (Figure B-17). Both original and resampled LIDAR DEMs captured the depressions (ditch and wetland) at the wetland sites better compared to the USGS DEMs at both wetlands.

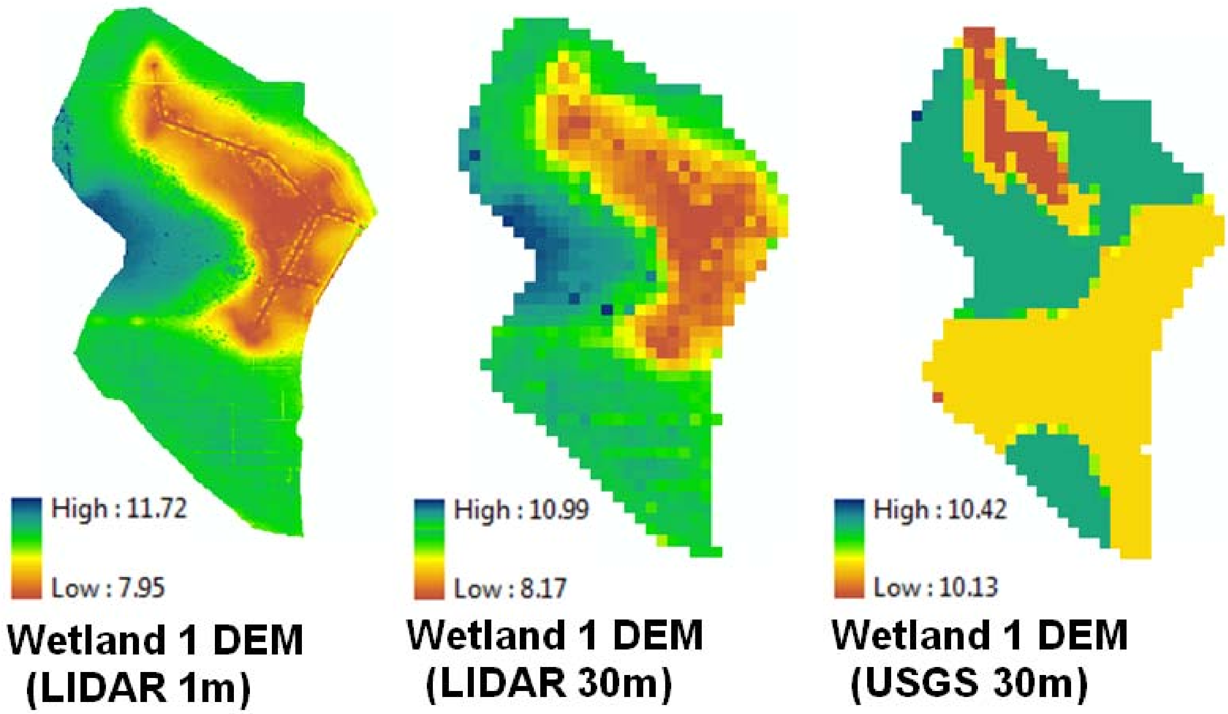

Figure B-16. LIDAR (1 and 30 m) and USGS (30 m) DEMs for Wetland 1.

The LIDAR (30 m) and USGS DEMs showed the bottom elevation of Wetland 1 to be 8.17 and 10.13 m, respectively (Figure B-16). The maximum daily surface water levels for pre-hydration, post-hydration 1, and post-hydration 2 periods based on surface well data were 9.44, 9.99, and 9.45 m, respectively which were below the bottom elevation (10.13 m) of Wetland 1 determined by the USGS DEM. Therefore, use of USGS DEM to estimate hydroperiods, surface water storage and ponded areas yielded no values at Wetland 1. Table B-12 shows the hydroperiods, average daily water storages, maximum and average ponded areas at Wetland 1 drainage area for pre-hydration and post-hydration periods using LIDAR and USGS DEMs.

Table B-12. Hydroperiod, average daily water storage, maximum and daily average ponded areas using LIDAR (30 m) and USGS DEMs for Wetland 1 site.

Period	Hydroperiod (day)		Average water storage (m^3)*		Max. ponded area (m^2)*		Average ponded area (m^2)*	
	LIDAR	USGS	LIDAR	USGS	LIDAR	USGS	LIDAR	USGS
Pre-hydration	292	0	2,089	0	198,381	0	15,068	0
Post-hydration 1	309	0	12,625	0	297,071	0	58,022	0
Post-	342	0	9,027	0	202,024	0	56,817	0

hydration 2								

*For the days constituting the hydroperiods

Wetland 2 bottom elevations based on LIDAR (30 m) and USGS DEMs were 9.11 and 9.93 m, respectively (Figure B-17). The maximum daily water levels for pre-hydration, post-hydration 1, and post-hydration 2 periods were 9.59, 10.19, and 9.74 m, respectively based on the surface well data. Table B-13 shows the hydroperiods, average daily water storages, maximum and average daily ponded areas at Wetland 2 for pre-hydration and post-hydration periods using LIDAR (30 m) and USGS DEMs. The wetland water levels measured by the surface water wells for pre-hydration and post-hydration 2 periods were lower than the wetland bottom elevation for USGS DEM. For post-hydration 1 period, the average daily surface water storage and average daily ponded area estimated by USGS DEM were larger compared to those estimated by LIDAR DEM. The USGS DEM recognized only three days during tropical storm Fay to constitute the hydroperiod for the post-hydration 1 period. Since the wetland water levels were high during those three days, the average daily surface water storage and average daily ponded area for those three days were larger compared to those estimated using LIDAR DEM which recognized 108 days as the hydroperiod. These 108 days included many days with very low wetland water levels. Therefore, the average daily water storage and average daily ponded area estimated by LIDAR were smaller compared to USGS DEM.

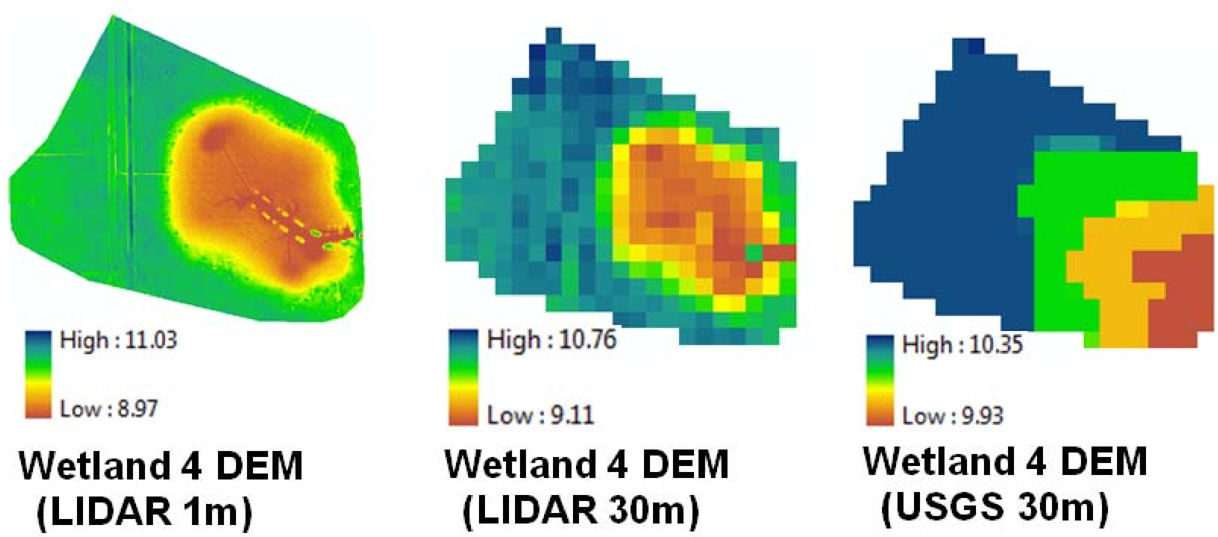

Figure B-17. LIDAR (1 and 30 m) and USGS (30 m) DEMs for Wetland 2.

Table B-13. Hydroperiod, average daily water storage, maximum and daily average ponded areas using LIDAR (30 m) and USGS DEMs for Wetland 2 site.

Period	Hydroperiod (day)		Average water storage (m^3)*		Max. ponded area (m^2)*		Average ponded area (m^2)*	
	LIDAR	USGS	LIDAR	USGS	LIDAR	USGS	LIDAR	USGS
Pre-hydration	56	0	23	0	4,554	0	317	0
Post-hydration 1	108	3	686	1,369	78,940	51,376	4,650	17,140
Post-hydration 2	187	0	111	0	33,231	0	1,233	0

* For the days constituting the hydroperiods

The maximum water levels at wetlands 1 and 2 during the study period were 9.99 and 10.19 m, respectively during tropical storm Fay on 20 August 2008. The surface water storages and ponded areas for these water levels estimated using the LIDAR (30 m) and USGS DEMs are shown in Table B-14. At Wetland 1 site, the use of USGS DEM did not yield any surface water storage and ponded area for the observed maximum water level because the bottom elevation of the wetland (10.33) based on the USGS DEM was higher than the wetland water level (9.99). Figure B-18 shows the inundated areas for 9.99 m elevation of wetland water level at Wetland 1 during tropical storm Fay for LIDAR (1 and 30 m) and USGS DEMs. For Wetland 2, USGS DEM underestimated the surface water storage and ponded surface area by 84 and 34%, respectively compared to LIDAR (30 m) DEM (Table B-14, Figure B-19).

Table B-14. Surface water storages and ponded areas for wetland sites 1 and 2 for the maximum water levels of 9.99 and 10.19 m, respectively that occurred during tropical storm Fay in August 2008 estimated using LIDAR (30 m) and USGS DEMs.

Wetland	LIDAR (30 m) DEM		USGS DEM	
	Water storage (m^3)	Ponded area (m^2)	Water storage (m^3)	Ponded area (m^2)
Wetland 1	209,517	295,755	0	0
Wetland 2	28,017	79,501	4,431	52,266

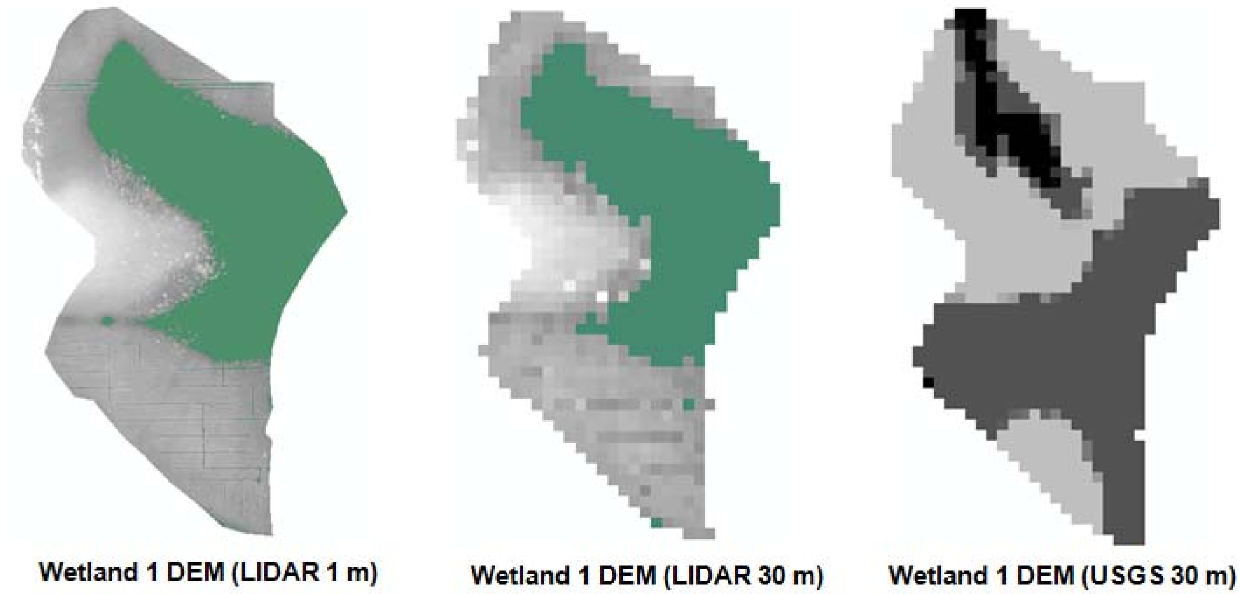

Figure B-18. Ponded area (green color) for Wetland 1 maximum water level of 9.99 m for LIDAR (1 and 30 m), and USGS (30 m) DEMs during tropical storm Fay.

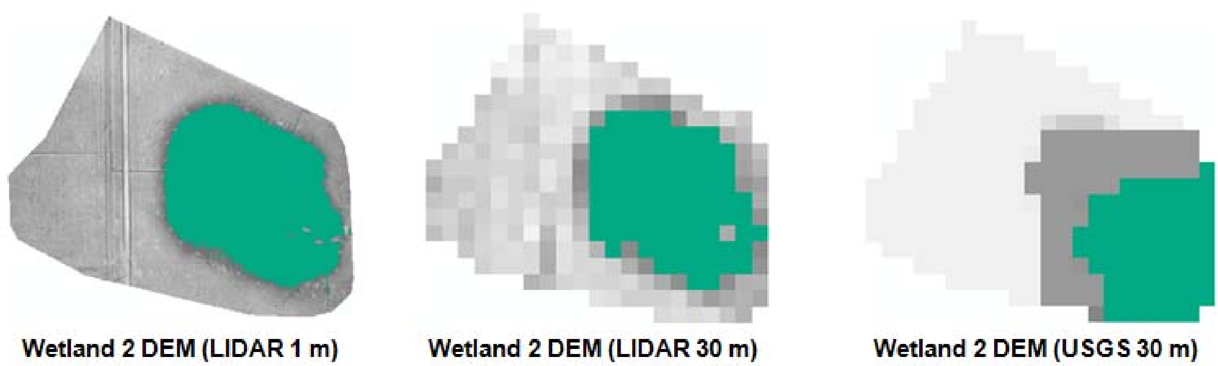

Figure B-19. Ponded area (green color) for Wetland 2 maximum water level of 10.19 m for LIDAR (1 and 30 m) and USGS (30 m) DEMs.

This study showed that the wetland passive hydration was effective in increasing the water storage and ponded areas in the two wetlands within the beef-cattle ranch. Hydroperiod was dependent on water storage, and therefore increase in water storage resulted in the increase in the hydroperiod for all periods except during post-hydration 1 period at Wetland 2. Low and less evenly distributed rainfall during post-hydration 1 period were responsible for shorter hydroperiod at Wetland 2.

To better quantify water storage and ponded area, accurate representation of the topography is highly essential. LIDAR data can provide accurate representation of the topography. This is particularly important for Florida (specially south Florida) with its relatively flat topography which has subtle variations in terrain elevations. Only LIDAR can delineate the small elevation differences between two adjacent locations in the flat terrains of this region. In this study, the use of LIDAR topographic data ensured correct estimations of hydroperiods, water storages and ponded areas at the two wetland sites.

Summary and Conclusion

Effects of passive hydration on hydroperiod, water storage, and wetland ponded area were studied at two wetland sites within a beef-cattle ranch in the LO basin by comparing one pre-hydration period (May 2005-April 2006) with two post-hydration periods (post-hydration 1: May 2008-April 2009 and post-hydration 2: May 2009-April 2010). Passive hydration of the wetlands was carried out by installing flashboard risers at the outlet of the wetlands. Similar amounts of rainfall fell during the pre-hydration and post-hydration 2 periods. The two post-hydration periods experienced larger storages and ponded areas than the pre-hydration period. Post-hydration hydroperiods were longer for both wetlands except for post-hydration 1 hydroperiod at Wetland 2. Low and less evenly distributed rainfall over post-hydration 1 period played an important role in shorter hydroperiod at Wetland 2. The main reason for this was that the year 2007 was a drought year and the dry conditions continued until May 2008. As a result, ponding started to occur only in the last part of June 2008 reducing the hydroperiod length .

Following conclusions may be drawn from this study:

1) Passive hydroperiod was effective in increasing wetland water storage and ponded areas in the two wetlands.
2) Hydroperiod depends on water retention as well as rainfall amount and distribution pattern within the study period.
3) Accuracy in the estimations of hydroperiod, water storage and ponded area in the wetlands depends on the accuracy of the topographic data used in the estimation of these parameters. Use of high resolution LIDAR DEMs coupled with accurate measurement of water depths in the wetlands can result in accurate estimation of these wetland parameters.
4) Non-LIDAR low resolution DEMs can not estimate the hydroperiod, water storage and ponded area in the wetlands accurately.

5) Inaccurate estimation on water storage in the Okeechobee watershed would lead to inaccurate estimation or prediction of water flow to LO. Accurate estimation of these parameters would result in proper water management strategies and policy decisions.
6) The benefits derived from the use of LIDAR data for various purposes like flood mapping, land slide prediction, and modeling outweigh the cost of acquiring them.
7) Various state agencies in the state of Florida are encouraging adoption of passive hydration and to facilitate basin-wide adoption of this approach. The State agencies should offer cost-share programs to reduce the financial burden on the users for using these methods.

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Appendix B-3

Role of Soils in Phosphorus Retention and Release in the two wetland sites

Soil samples were collected from the Ranch at 45 locations at least at two depths: 0-10 cm and 10-20 cm. To evaluate the role of soils in storing P deeper in the soil profile, a few of the samples were taken to a lower depth to include the Bh horizon when that horizon existed within a sampling depth of ~100 cm. All soils were analyzed for total P (TP), Total C by the loss on ignition (LOI) procedure, water soluble P (WSP) and Mehlich 1-P, Fe and Al. The Mehlich 1 values were used to calculate the soil P storage capacity (SPSC) using a generalized equation:

$SPSC = (0.1 - Soil\ PSR_{M1})*Mehlich\ 1\text{-}extractable\ (Fe + Al)*31*1.3$ (mg/kg)

where

$Soil\ PSR_{M1} = Mehlich\ 1\text{-}P / Mehlich\ 1\ (Fe + Al)$, *and* P, Fe and Al are expressed in moles (Nair et al., 2010).

Soil characteristics data including TP, LOI, WSP, Mehlich 1-P, Fe and Al along with PSR and SPSC calculations are given in Table B-15 (Wetland 1) and Table B-16 (Wetland 2). Locations of sampling at Wetland 1 and Wetland 2 are shown in Figure B-20 and Figure B-21. When SPSC values at the 10-20 cm depth were lower than at the surface 0-10cm, it appeared likely that P has moved down the soil profile. Therefore we examined soils to a depth greater than 20 cm. The SPSC values of two such profiles – Profile 39 in Wetland 1 and Profile 40, a ditch in the Wetland – were examined in detail (Figure B-22 and Figure B-23). The spodic horizon at the wetland occurred at 125 cm while it was at 65 cm depth in the ditch soil profile.

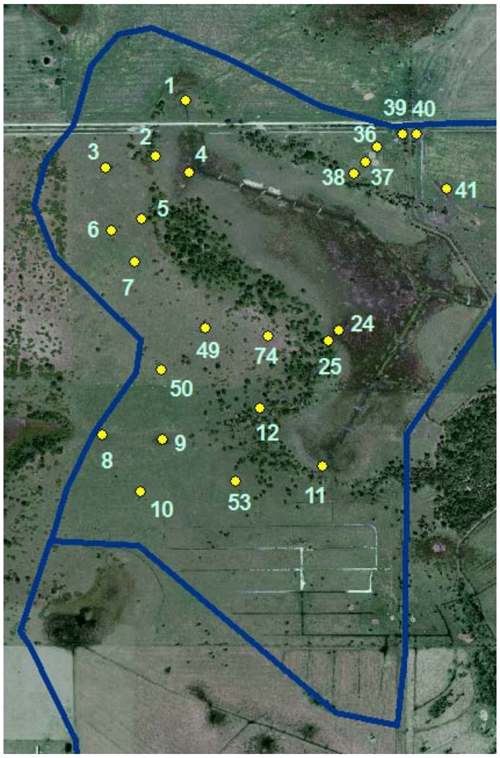

Figure B-20. Locations of sampling sites in Wetland 1. Profile 39 is within the wetland; Profile 40 is a ditch sample near Profile 39. The wetland and ditch locations are close to a heavy P-impacted barn.

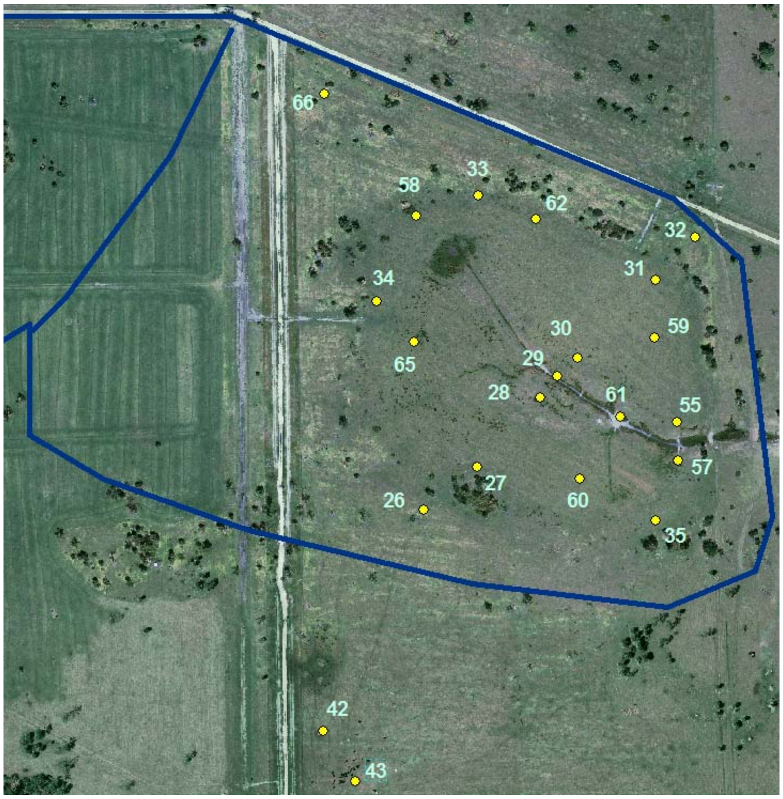

Figure B-21. Soil sampling locations at Wetland 2.

Following general observations can be made based on the soil characteristics data presented in Table B-15 and Table B-16.

1. Wetland 1 is the more heavily impacted wetland with several soil profiles with negative SPSC values.
2. Most of the negative SPSC soil profiles, in the uplands, wetlands and ditches of Wetland 1 are close to the cattle feeding area suggesting that these locations are heavily impacted with P from manure-accumulation.
3. Raised water table and increased connectivity of the ponded areas due to WWR at Wetland 1would likely in P being dissolved and lost via runoff resulting in P distribution across surface soils and increased P loads. An example of such connectivity of ponded area occurred during the post-BMP2 period after Tropical Storm Fay (rainfall= 30 cm) which is the likely the reason for observing higher than expected TP loads.
4. Low TP values at depths below the surface horizon just indicates that this is a "transition zone", likely an E horizon of a Spodosol which has little P retentive capacity. Phosphorus moves through the soil profile and often TP shows accumulation at lower depths, e.g. profiles 37, 38, 39, and 40 for Wetland 1(Table B-15). The SPSC values will depend on the Fe and Al at a given depth
5. High TP at 0-10 and 10-20 cm depths does not necessarily mean that the soil is a P risk. For example, soil profiles 24 and 25 for example have high TP values, but they still have the capacity to retain additional P. At these locations, high Al values result in positive SPSC values.
6. Soil profile 39 is a sample in Wetland 1 that is heavily P-impacted at the surface (Figure B-22). The P continues to move downwards from the surface and accumulates below the spodic horizon as indicated by the high TP at lower depths. This soil profile is a P source until it reaches the spodic horizon where the soil has sufficient capacity to sorb additional P. However, this soil will likely lose a substantial amount of P via subsurface flow above the spodic horizon. Furthermore, for periods that experience increased water table depth due to WWR is likely to increase the overland flow from these heavily P-impacted sites which will result in increased dissolved P losses.

7. Profile 37 is of a similar situation as profile 39, however, in this profile the spodic horizon occurring at 110 cm depth is also a P source as in the case of ditch sample 40 discussed below.
8. Ditch sample 40 is heavily P impacted. As in the case of the soil profile at 39, P moves down the profile with high TP values at the lower depths (Figure B-23). At this location, the Bh horizon is shallower than at location 39. P has moved through the spodic exhausting the capacity of the horizon to receive any additional P.
9. Soil samples 1-12, 24, and 25 do not appear to be heavily P impacted as also samples 49. 50, 53, and 74. Most of these soils show typical characteristics of sandy surface soils with little or no retention capacity (see section on isotherms); absolute SPSC values either positive or negative for these soils are less reliable because of the division of available P by a very small Fe+Al value leading to unavoidable computational errors.
10. Soil samples at, or west of the tree line is less influenced by the heavy manure P-impacted areas near the barn; consequently most samples described in (9) above are generally not a major P source. Trees are known (besides acting as buffers) to remove excess P from the soil profile leaving higher SPSC in such land-uses compared to tree-less systems (Michel et al., 2007; Nair et al., 2007).
11. Almost all samples of Wetland 2 (Table B-16) have positive SPSC and this wetland is likely to release minimal P upon flooding. Positive SPSC combined with less frequent ponding in the wetland as well as upland areas, even after WWR implementation, was the likely reason for reduced TP loads from this site (See Appendix B-1).

Table B-15. Water soluble P (WSP), total P (TP), Mehlich 1- P (M1-P), Fe (M1-Fe) and Al (M1-Al) at Wetland 1. The phosphorus saturation ratio (PSR) and soil P storage capacity (SPSC) are calculated values.

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%	mg/kg						mg/kg
1	0-10	14.64	4.75	479	21.0	405	41	0.04	36.1
	10-20	12.35	3.79	455	44.8	646	43	0.06	41.2
2	0-10	14.71	2.59	392	3.8	264	40	0.01	37.3
	10-20	2.56	0.91	49	0.5	90	86	0.00	19.0
3	0-10	13.11	20.35	201	18.1	15	0	1.08	-21.4
	10-20	3.18	1.87	ND	0.0	20	11	0.00	3.7
4	0-10	42.74	12.43	721	10.1	267	11	0.03	27.5

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%	mg/kg						mg/kg
	10-20	33.17	2.59	454	3.1	548	36	0.00	80.4
5	0-10	4.86	11.95	106	4.0	41	26	0.07	2.7
	10-20	3.27	5.71	65	10.4	72	102	0.07	4.7
6	0-10	9.01	7.15	121	5.0	49	9	0.08	1.5
	10-20	5.88	2.11	39	0.0	88	36	0.00	15.8
7	0-10	12.33	15.31	127	4.5	32	4	0.11	-0.7
	10-20	5.78	3.07	26	0.0	33	5	0.00	5.3
8	0-10	18.53	17.23	225	5.9	67	13	0.07	3.2
	10-20	5.82	3.79	51	1.7	83	11	0.02	11.1
9	0-10	6.4	10.51	177	17.0	106	8	0.13	-5.6
	10-20	6.37	12.19	202	110.7	531	21	0.18	-63.2
10	0-10	12.73	12.19	170	2.6	68	2	0.03	6.9
	10-20	4.61	2.59	22	0.0	33	2	0.00	5.0
11	0-10	27.71	3.07	636	0.9	214	43	0.00	33.8
	10-20	6.09	0.67	43	0.2	745	33	0.00	113.3
12	0-10	8.36	7.63	ND	2.4	31	4	0.06	1.9
	10-20	3.02	1.15	ND	0.0	15	5	0.00	2.5
24	0-10	82.18	23.54	1218	15.3	373	1	0.04	35.8
	10-20	55.34	2.35	415	5.2	2669	9	0.00	392.3
25	0-10	26.51	4.27	452	2.1	3002	35	0.00	447.9
	10-20	15.77	1.15	151	0.2	2377	23	0.00	356.2
36	0-10	20.7	136.29	875	176.6	21	3	7.03	-226.3
	10-20	17.57	59.52	361	63.6	38	4	1.40	-76.8
	20-50	1.02	15.31	27	17.7	3	1	4.32	-22.4
	50-130	0.98	3.79	ND	3.5	232	7	0.01	30.5
	>130	1.47	5.47	46	31.5	9	1	2.85	-39.6
37	0-10	10.1	13.63	123	10.8	35	6	0.25	-8.5
	10-20	5.56	7.87	34	4.5	35	6	0.10	-0.1
	20-50	1.57	8.59	20	8.7	4	0	2.17	-10.8
	50-110	1.57	2.83	ND	2.1	9	0	0.21	-1.4
	>110	5.42	16.75	206	134.2	1006	11	0.12	-23.6
38	0-10	16.97	19.87	231	10.6	66	9	0.13	-3.2
	10-20	4.56	6.43	23	4.7	47	39	0.06	3.7
	20-50	3.68	6.43	61	23.1	168	81	0.10	1.0
	50-140	4.48	0.91	62	22.8	325	34	0.06	21.2
	140-190	3.05	0.43	109	20.5	328	14	0.05	23.4

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%	mg/kg						mg/kg
39	0-10	6.73	45.13	328	124.8	57	6	1.80	-153.2
	10-20	4.41	93.11	327	327.0	70	8	3.86	-414.3
	20-50	1.84	35.53	64	39.3	16	3	1.91	-48.4
	50-100	1.55	9.31	31	21.0	7	2	2.29	-26.1
	125-170	3.11	ND	173	16.7	1142	9	0.01	149.3
40	0-10	10.94	59.52	213	73.0	37	2	1.67	-89.3
	10-20	13.36	59.52	255	219.0	68	3	2.73	-274.2
	20-50	0.43	8.11	37	26.6	13	2	1.69	-32.5
	50-65	1.46	12.19	27	16.7	6	2	2.14	-20.7
	65-105	3.49	153.08	394	331.9	582	2	0.50	-344.4
41	0-10	5.1	193.86	210	247.2	30	3	6.87	-316.7
	10-20	8.7	57.12	180	73.0	61	8	0.98	-85.2
	20-50	3.57	30.73	66	37.9	24	3	1.27	-45.4
	50-100	0.64	2.11	33	28.7	10	2	2.38	-35.8
	100-120	3.73	4.99	63	30.6	361	6	0.07	14.5
42	0-10	11.57	150.68	608	280.1	19	3	12.08	-361.2
	10-20	4.15	21.14	72	22.8	57	12	0.32	-20.3
	20-50	3.15	7.63	66	22.1	140	11	0.13	-7.1
	50-80	2.03	2.59	35	13.2	126	5	0.09	2.0
	80-95	2.3	1.15	44	15.8	199	9	0.07	9.9
49	0-10	7.86	8.22	35	6.7	12	39	0.19	-5.4
	10-20	4.01	3.99	28	2.1	6.	32	0.08	0.7
	20-50	1.59	0.68	10	0.1	0.1	5	0.03	1.0
	50-80	1.20	0.22	7	0.2	0.3	3	0.09	0.1
50	0-10	7.88	8.22	65	9.3	5.7	28	0.42	-32.4
	10-20	1.79	2.28	19	1.9	2.4	7	0.29	-5.7
	20-50	1.20	0.68	12	1.4	0.8	3	0.54	-8.2
	50-80	67.1	0.22	3	0.1	0	3	0.53	0.2
53	0-10	4.2	8.45	24	5.9	11.0	47	0.15	-2.1
	10-20	3.2	0.45	13	0.3	4.4	24	0.18	1.6
	20-50	2.0	0.45	10	0.4	0.9	14	0.04	0.9
	50-80	2.4	0.22	8	0.1	0.3	7	0.02	0.9

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%	mg/kg						mg/kg
74	0-10	6.3	4.11	55	2.1	4.9	42	0.07	2.3
	10-20	2.8	0.9	14	0.6	2.7	11	0.06	0.8
	20-50	1.42	0.22	17	0.7	1.0	5	0.17	-1.8
	50-80	1.67	0.22	10	0.2	0	3	0.15	-0.6
	80-110	1.41	0.22	8	0.1	0	2	0.07	0.6

ND = Not detectable
N/A = Not Available

Table B-16. Water soluble P (WSP), total P (TP), Mehlich 1- P (M1-P), Fe (M1-Fe) and Al (M1-Al) at Wetland 2. The phosphorus saturation ratio (PSR) and soil P storage capacity (SPSC) are calculated values.

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%	mg/kg						mg/kg
26	0-10	9.16	7.63	56	0.9	37	8	0.02	4.9
	10-20	5.49	2.11	23	0.0	13	10	0.00	2.7
	20-50	0.45	0.19	ND	0.0	4	2	0.00	0.8
	50-100	0.79	0.19	ND	0.0	3	3	0.00	0.6
	100-125	1.95	0.43	73	33.7	648	12	0.04	53.8
27	0-10	8.95	12.19	120	24.7	81	20	0.24	-18.6
	10-20	9.79	9.55	84	11.1	125	17	0.07	5.4
	20-50	0.88	0.91	22	5.9	45	2	0.11	-0.7
	50-150	1.99	ND	31	0.9	378	11	0.00	55.9
28	0-10	10.8	10.75	143	6.4	63	7	0.08	1.7
	10-20	0.52	1.63	ND	1.7	27	1	0.05	1.9
	20-50	0	ND	ND	0.0	8	1	0.00	1.3
	50-140	0.5	ND	ND	0.0	47	5	0.00	7.4
29	0-10	28.99	18.91	329	6.6	144	21	0.04	14.5
	10-20	23.08	3.55	68	3.3	484	13	0.01	68.9
	20-50	5.48	ND	42	0.0	290	5	0.00	43.6
	50-100	2.46	ND	ND	0.0	114	4	0.00	17.4
	100-130	0.57	0.67	ND	2.1	40	1	0.05	3.4
30	0-10	12.24	18.19	195	15.3	75	5	0.17	-8.3
	10-20	3.10	2.35	34	5.9	46	2	0.11	-0.6
	20-50	0.26	1.63	ND	0.9	8	2	0.09	0.1
	50-140		ND	ND	0.0	29	5	0.00	4.7
31	0-10	9.38	8.59	132	12.5	108	6	0.10	0.3

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%			mg/kg				mg/kg
	10-20	3.73	2.83	ND	1.7	54	3	0.03	6.1
	20-50		0.19	ND	0.0	6	1	0.00	0.9
	50-70	0.47	0.43	ND	0.2	7	2	0.03	0.9
	70-120	3.74	ND	58	7.3	616	14	0.01	83.5
32	0-10	9.24	11.23	95	0.9	34	7	0.02	4.4
	10-20	3.81	1.63	ND	0.0	29	4	0.00	4.7
	20-50	0.71	1.15	ND	0.0	6	1	0.00	0.9
	50-100	0.66	0.19	ND	0.0	4	1	0.00	0.7
	>110	4.93	2.11	57	26.6	971	10	0.02	111.0
33	0-10	13.55	18.19	121	18.4	302	12	0.05	22.1
	10-20	2.73	4.27	ND	4.7	71	2	0.06	4.6
	20-50	2.2	5.23	ND	4.5	20	2	0.19	-2.7
	>50	2.34	1.87	ND	2.6	16	4	0.13	-0.7
	>80	1.5	0.19	ND	3.1	192	6	0.01	25.0
34	0-10	10.63	18.43	156	20.0	109	4	0.16	-9.4
	10-20	4.55	7.15	ND	4.6	56	5	0.07	2.8
	20-50	2.46	3.55	31	12.0	90	5	0.11	-1.9
	50-120	1.59	1.15	ND	3.1	24	4	0.10	-0.1
35	0-10	9.64	9.55	87	9.7	117	9	0.07	5.6
	10-20	3.93	3.07	ND	2.4	65	4	0.03	6.8
	20-50	1.22	0.91	ND	1.2	12	1	0.08	0.4
	50-70	0.66	0.67	ND	0.0	10	2	0.00	1.6
	>70	1.07	ND	ND	0.2	124	1	0.00	18.8
55	0-10	13.13	8.35	183	8.0	103	10	0.06	5.6
	10-20	2.69	1.15	35	2.6	66	4	0.03	6.8
	20-50	2.81	0.19	ND	2.1	6	3	0.26	-1.7
	50-80	0	0.43	ND	0.0	5	2	0.00	0.9
57	0-10	6.37	2.83	118	2.4	193	15	0.01	26.9
	10-20	2.9	0.91	33	1.2	232	10	0.00	33.8
	20-50	4.65	0.19	35	0.0	135	6	0.00	20.6
	50-80	4.65	ND	ND	0.0	147	4	0.00	22.2
	>80	6.16	ND	62	0.0	512	17	0.00	77.7
58	0-10	14.94	12.34	217	11.5	7	123	0.15	-16.7
	10-20	3.42	7.31	62	10.7	5	84	0.21	-10.4
	20-50	2.21	4.22	22	6.3	1	31	0.34	-8.4
	50-80	1.61	2.96	24	7.3	2	21	0.54	-16.6
	80-110	2.01	4.79	25	7.0	1	32	0.37	-11.3
59	0-10	6.54	3.79	64	1.7	42	11	0.03	4.8

Profile #	Depth	LOI	WSP	TP	M1-P	M1-Al	M1-Fe	PSR	SPSC
	cm	%			mg/kg				mg/kg
	10-20	0	1.15	ND	0.2	23	2	0.01	3.2
	20-50	0	0.19	ND	0.0	27	2	0.00	4.2
	50-100	0.96	0.43	ND	0.0	13	3	0.00	2.2
	>100	2.64	ND	ND	0.0	64	9	0.00	10.2
60	0-10	6.37	14.62	74	11.6	21	62	0.20	-12.0
	10-20	2.83	3.19	54	5.96	2	48	0.20	-8.8
	20-50	1.18	0.68	15	1.91	1	12	0.27	-3.7
	50-80	1.40	0.22	15	0.24	7	49	0.01	2.1
61	0-10	18.1	18.91	315	8.2	104	13	0.07	5.8
	10-20	5.08	7.39	87	8.2	160	5	0.04	13.5
	20-50	0.9	1.15	ND	0.7	13	2	0.05	1.1
	50-90	3.81	0.19	ND	0.0	14	3	0.00	2.3
	>90	1.22	ND	21	0.0	75	10	0.00	11.9
62	0-10	2.13	3.99	92	31.9	2	80	0.69	-82.2
	10-20	4.70	2.62	43	5.1	3	87	0.10	0.1
	20-50	3.01	0.91	12	2.9	1	18	0.27	-3.1
	50-80	0.99	1.94	11	5.5	1	25	0.36	-4.6
	>80	3.61	0.45	28	14.6	7	467	0.06	1.9
65	0-10	8.95	10.17	103	10	6	27	0.47	-60.1
	10-20	4.69	3.42	36	4	7	34	0.13	-1.9
	20-50	2.01	1.14	13	3	0	15	0.32	-4.4
	50-80	1.81	0.45	10	1	0	14	0.15	-0.7
66	0-10	12.82	7.77	26	5	9	39	0.14	-6.7
	10-20	4.27	4.79	13	4	5	49	0.13	-1.2
	20-50	2.20	1.59	11	1	1	10	0.17	-1.5
	50-80	1.78	2.05	19	4	1	6	1.00	-16.1
	>80	2.36	8.22	55	10	2	68	0.25	-5.1

ND = Not detectable

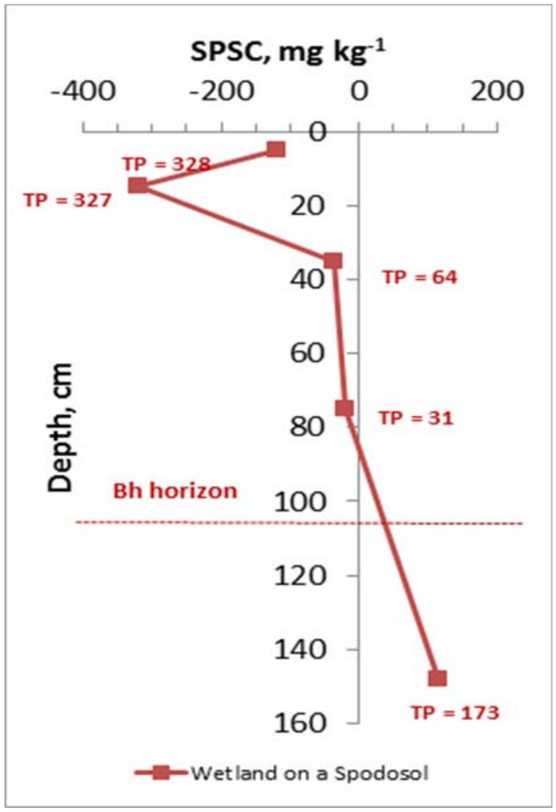

Figure B-22. Soil P storage capacity (SPSC) with depth in a soil profile located in Wetland 1 (Profile 39). The upper boundary of the spodic horizon is at 105 cm. Total P (TP) values corresponding to each SPSC value is also indicated.

Based on the analyses of the TP and SPSC data, the following conclusions can be drawn.

1. Evaluating P risk from a wetland soil cannot be done by P determination of the surface 0-10 cm soils alone
2. Total P values alone are misleading when interpreting the soil's ability to retain additional P. High TP in soils is not always an indicator of a soil at risk for P loss. Low values, on the other hand, do not mean that there is little P risk. Sandy soils of the E horizon, for example may have low TP values, but they form a medium through which P passes through prior to reaching the more retentive spodic horizon.
3. The spodic horizon's capacity to hold additional P may be exhausted (negative SPSC) in some cases. In such cases, the P could continue to move vertically through the soil profile or it can release P to the groundwater that moves laterally above the spodic horizon before it is discharged to the nearest dranage ditch. In both cases, the spodic horizon can be a source of P to the drainage ditch,
4. For the evaluation of P risk from a wetland or a ditch it is necessary to look into the possibility of P loss via surface, subsurface as well as via leaching.

5. Vegetation in wetlands would likely remove only a small amount of P as illustrated by P movement in wetland and ditch soil profiles.

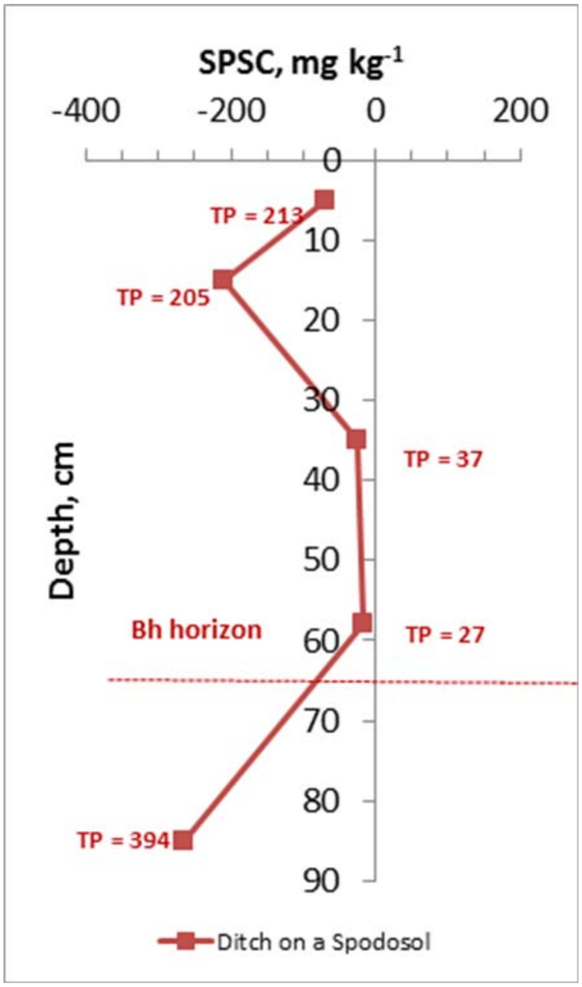

Figure B-23. Soil P storage capacity (SPSC) with depth in a ditch located in Wetland 1 (Profile 40). The upper boundary of the spodic horizon is at 65 cm. Total P (TP) values corresponding to each SPSC value is also indicated.

Isotherm Determination

Twenty-five soil samples were selected to determine traditional isotherms. Phosphate sorption was measured using one gram of an air-dried, homogenized soil treated with 10 mL of 0.01*M* KCl solution containing various levels of P (ranging from 0 to 100 mg P L^{-1}) in 50 mL centrifuge tubes. The tubes were placed on a mechanical shaker for a 24-hour equilibration period. At the end of the period, the soil samples were centrifuged at 6000 rpm for 10 min and the supernatant filtered through a 450 nm membrane filter and the filtrate analyzed for soluble reactive P (Murphy and Riley, 1962) using a Technicon™ Autoanalyzer (EPA 365.1). All extractions and determinations were at room temperature (298 ±3 K).

Langmuir parameters

Adsorption parameters were calculated using the Langmuir adsorption equation:

$$C/S = 1/kS_{max} + C/S_{max}$$

where:

$S = S' + S_o$, the total amount of P sorbed, mg kg^{-1}

S' = P sorbed by the solid phase, mg kg^{-1}

S_o = P originally sorbed on the solid phase, mg kg^{-1}

C = concentration of P after 24 h equilibration, mg L^{-1}

S_{max} = P sorption maximum, mg kg^{-1}

k = a constant related to the bonding energy, L mg^{-1} P

S_o was estimated using a least square fit of S' measured at low (< 10 mg P L^{-1}) equilibrium concentrations, C. At these concentrations, the linear relationship between S' and C can be described by $S' = kC - S_o$ where k is the linear adsorption coefficient (Gale et al., 1994; Graetz and Nair, 1995). The linear portion of the graph used in the calculations had r^2 values of at least 0.95. The method used for calculations of S_o using actual data points was illustrated by Graetz and Nair, 1995.

The Freundlich equation

Although the Freundlich equation, $\log S = \log K_F + n \log C$, does not allow the calculation of a sorption maximum value, its K_F value is a measure of the ratio of P in the solid phase to that in the solution phase; C is the concentration of P remaining in solution after a 24 h equilibration and S, the total amount of P retained.

Results of Isotherm Studies

Although complete isotherms were conducted for all 25 soils, it was not possible to obtain either the Langmuir or Freundlich parameters for several soils. This behavior is not unusual; Nair et al. (1998) could not generate any Langmuir isotherms for surface A and E horizons of Spodosols, although isotherms could be determined for the underlying Bh horizons. Surface horizons of Spodosols have very little Fe and Al and hence no P retention capacity. The SPSC is designed to capture P retention and release from such soils (Nair and Harris, 2004; Nair et al., 2010).

Soil profiles 2 and 11 from Wetland 1 have high TP values in the surface horizons (TP = 392 and 636 mg/kg respectively). They also have high Smax values (526 mg/kg; Table B-17). Water soluble P is low and SPSC is positive, suggesting that in spite of high TP values, the surface soils constitute minimal P risk at present. However, P would likely have moved down these soil profiles. It is suggested that deep soil sampling be conducted at these sites to verify that P has not moved down the soil profile (see Figure B-22 and Figure B-23).

Table B-17. Selected soil characteristics: Water soluble p (WSP) loss on ignition (LOI), soil P storage capacity (SPSC), total (TP), equilibrium P concentration (EPC_0), initial P sorbed to soils (S_0), Langmuir k (the P bonding constant), and the P sorption maximum (S_{max}) for these soils. Also calculated are Freundlich K_F parameters.

Sl no.	Profile	Depth	WSP	LOI	SPSC	TP	EPC0	So	k	Smax	KF (Freundlich)
	no.	cm	(mg/kg)	%	mg/kg	mg/kg	mg/L	mg/kg	L/mg	mg/kg	L/kg
1	2	0-10	3	14.7	37.3	392	0.107	1	0.13	476	140
2	2	10-20	1	2.6	19.0	49	0.09	0.8	0.032	66	12
3	11	0-10	3	27.7	33.8	636	0.488	5.5	0.118	498	131
4	11	10-20	1	6.1	113.3	43	0.01	0.1	0.174	200	61
5	33†	0-10	18	13.6	22.1	121	NA	NA	NA	NA	NA
6	33†	>50	0	1.5	25.0	ND	0.0057	0.3	0.061	142	38
7	34†	0-10	18	10.6	-9.39	156	NA	NA	NA	NA	NA
8	34†	10-20	7	4.5	2.8	ND	NA	NA	NA	NA	NA
9	35†	0-10	10	9.6	5.6	87	NA	NA	NA	NA	NA
10	35†	10-20	3	3.9	6.8	ND	NA	NA	NA	NA	NA
11	36	0-10	136	20.7	-226.3	875	NA	NA	NA	NA	NA
12	36	10-20	60	17.6	-76.8	361	NA	NA	NA	NA	NA
13	36	20-50	15	1.0	-22.4	27	NA	NA	NA	NA	NA
14	36	50-130	4	1.0	30.5	ND	NA	NA	NA	NA	NA
15	36	>130	5	1.5	-39.6	46	0.479	3.5	0.027	52	4
16	38	0-10	20	17.0	-3.9	231	NA	NA	NA	NA	NA
17	38	20-50	6	3.7	0.9	61	0.348	2.9	0.176	93	27
18	38	50-140	1	4.5	21.2	62	0.048	0.5	0.152	189	43
19	40	0-10	60	10.9	-89.3	213	NA	NA	NA	NA	NA
20	40	10-20	60	13.4	-274.2	205	NA	NA	NA	NA	NA

21	40	20-50	8	0.4	-32.5	37	NA	NA	NA	NA	NA
22	40	50-65	12	1.5	-20.7	27	NA	NA	NA	NA	NA
23	40	65-105	153	3.5	-344.4	394	2.34	43	0.042	77	8
24	61†	0-10	19	18.1	5.8	315	NA	NA	NA	NA	NA
25	61†	10-20	7	5.1	13.5	87	NA	NA	NA	NA	NA

† Wetland 2; the other profiles are all from Wetland 1.

It was possible to obtain an isotherm for only the deepest soil horizon in the upland soil profile (Profile 36) in Wetland 1. Although this depth (> 130cm) represents a spodic horizon, negative SPSC and an EPC_0 of 0.479 mg/L suggests that P would likely be released from this horizon as well. The ditch soil profile 38 in Wetland 1 shows some storage capacity at the lower depths which could be an indicator of a shallow spodic. However, deeper soil sampling would indicate if P has moved down the soil profile. Isotherms could not be determined for most horizons of ditch Profile 40 in Wetlands 1. The profile had negative SPSC values for all horizons (Figure 4).

Wetland 2 is likely less P-impacted than Wetland 1. However, it was not possible to obtain isotherms for most of the surface 0 – 10 and 10 – 20 cm depths for soils of this wetland either. The only soil for which isotherms could be determined was at the lower (> 50 cm) depth of profile 33. Note that these soils had SPSC values close to zero with one soil sample (0 – 10 cm depth of profile 34) with a slight negative SPSC value. For a better understanding of the potential of soils of Wetland 2 to release P, more profiles (both number of profiles as well as lower depths within a profile) from this wetland should be analyzed for various parameters.

In a recent report (Nair et al., 2011), it had been shown that EPC_0 values would be minimal when SPSC is positive and that it would begin to increase with negative SPSC. Although there were very few isotherms determined for this project that gave measurable EPC_0 values, we evaluated the SPSC/ EPC_0 relationship (Figure B-24) for these soils. Though linear, this relationship does not capture those soils with potentially high EPC_0 values which could not be assessed due to difficulties in obtaining isotherms. It is suggested that future modeling efforts include more easily determined SPSC rather than the difficult-to-obtain isotherm parameters.

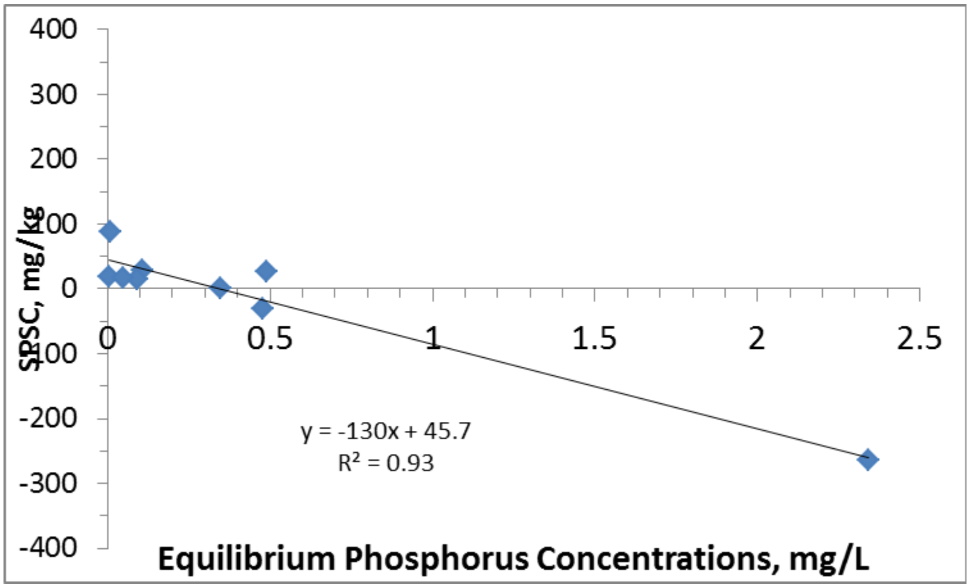

Figure B-24. Relationship of SPSC with the equilibrium P concentrations for those soils for which isotherms could be generated.

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Appendix C

Performance, reliability, accuracy of the Acoustic Doppler Velocimeters and transducers

Measuring the surface water discharge from Pelaez Ranch falls under the category of open-channel flow since the discharges from the ranch are conveyed through natural earthen canals. Measurement of open-channel flows can be achieved through a wide range of methods that vary in their accuracy, ease of use, and infrastructure requirements. Selection of the measurement method or technique is based on the desired accuracy and purpose of flow measurement. To adequately evaluate the effectiveness of the BMPs implemented at Pelaez Ranch, continuous flow measurement and high accuracy were needed at the five hydrologic stations. In order to fulfill those requirements, trapezoidal flumes were selected as the primary measurement device (a hydraulic structure used to create flow conditions that allow stage measurements to be directly related to flow rate). The secondary measurement devices (device used to measure and record stage in the primary measurement device) selected were pressure transducers that were connected to dataloggers which continuously recorded the stage.

Under certain flow conditions that were prevalent during the study period, the accuracy of this flow measurement system was low. This system will provide high accuracy flow estimates under free-flowing conditions, but as soon as submerged conditions prevail, the accuracy of the flume/pressure transducer system begins to decline. The topographic and hydrologic conditions that exist at the study site resulted in submerged conditions more often than anticipated during project design. In order to be able to accurately measure flow under these submerged conditions, additional measurement devices were installed in the flume in August, 2008. Acoustic Doppler Velocimeters (ADVs) were selected and installed in the flumes to provide this higher accuracy flow measurement at the hydrologic stations.

ADVs measure the velocity of particles in the flowing water taking advantage of the Doppler effect and in making water velocity estimates, assume that the water has the same velocity as the particles. This velocity measurement is combined with the cross-sectional area of the water in the trapezoidal section of the flume to generate flow estimates. The ADV area-velocity technique, due to its proven accuracy under most flow conditions, was considered to be the standard measurement technique and was compared to the estimates from the flume/pressure transducer system to determine whether or not there was a correlation between the two techniques under most flow conditions experienced at the study site. Two flumes were considered in this comparison, Flume 3 and Flume 4.

For Flume 3, average daily flow rate estimates derived from the flume/pressure transducer technique and ADV velocity-area technique were compared for the 7/2/08-10/08/10 period. For flow rate comparison purposes, negative flows were removed from both datasets. Additionally, flow velocity values below 0.1 m/sec were removed from the ADV data set since the flumes are not capable of measuring flows in that velocity range. ADV data was also filtered to remove erroneous data caused by interference with the instrument sensor. There was good agreement between the flow rates derived from flume system and ADV velocity data with a coefficient of determination (R^2) of 0.79.

For Flume 4, average daily flow rate estimates derived from the flume/pressure transducer technique and ADV velocity-area technique were compared for 7/25/08-4/30/10 period. Similar data filtering/screening was performed for Flume 4 as described above for Flume 3. There was a good agreement between the flow rates derived from flume system and ADV velocity data with a coefficient of determination (R^2) of 0.90.

The comparisons performed for both flumes showed that despite the fact that the flume rating equation could not accurately estimate the flow under all conditions, there was a strong correlation between the two techniques. The flume rating equation for Flume 4 more accurately predicted the flow while Flume 3 did not perform as well which is more than likely due to the higher prevalence of submerged flow conditions at Flume 3 which are not present at Flume 4. Flume 4 is at a higher elevation in the landscape than Flume 3 and is located in a basin that has better downstream drainage. These two facts are the reason why Flume 4 experienced less interference from the downstream flow conditions resulting in free flow conditions for almost the entire study period while Flume 3 experienced submerged conditions frequently due to downstream flow conditions and regional ground water table interference.

This analysis shows that both techniques have their advantages and disadvantages. A summary of these advantages is included in Table C-1.

Table C-1. Summary of the advantages and disadvantages of two flow estimation techniques (flume rating equation and ADV area-velocity)used and evaluated as part of the BMP evaluation project.

System	Advantages	Disadvantages
Flume/Pressure Transducer (Flume rating equation)	• Lower initial investment • Less maintenance • Lower maintenance repair costs • High accuracy under free-flow conditions	• Accuracy decreases under submerged flow conditions • Calibration is needed to achieve high accuracy for submerged flow conditions
Flume/ADV (ADV area-velocity)	• High accuracy under all flow conditions	• High initial costs • High maintenance requirements

Appendix D

Dissemination of the Project Results

Results from the BMP studies have been disseminated to the ranchers, state agencies, and other stakeholders. Results from the study have been presented at various state and/or national conferences. A summary of the presentations is presented in Table D-1.

Table D-1. List of presentations made as part of the BMP evaluation project to disseminate the results.

Date	Presentation Title	Authors	Event	Audience
June 2009 and 2010	Meetings with Pelaez Ranch owner for ranch management data and presentation of results.	S. Shukla, D. Goswami, A. Hodges, J.M. Knowles	N/A	Rancher.
June 2009	Effects of cattle fencing and wetland water retention BMPs on phosphorus loadings.	S. Shukla, A. Hodges.	Lake Okeechobee Inter-Agency Meeting, Okeechobee, FL.	Ranchers, FDACS, FDEP, SFWMD, UF/IFAS researchers, land use planners and managers.
Summer 2009	Site visit with SFWMD personnel.	S. Shukla.	N/A	SFWMD staff
June 2009	Effects of water retention on nitrogen and phosphorus loadings from two drained wetlands in the Lake Okeechobee Basin.	D. Goswami, S. Shukla, J.M. Knowles, W.D. Graham.	ASABE Annual International Meeting, Reno, NV.	Scientists and Engineers
February 2010	Effects of cattle exclusion best management practice on Phosphorus and Nitrogen discharges in the Lake Okeechobee basin.	D. Goswami, S. Shukla. W.D. Graham, A. Hodges, M. Christman	Second Sustainable Water Resources Symposium, University of Florida Water Institute,	University researchers, state and federal agencies, engineering consultants, agricultural landowners,

			Gainesville, FL.	and land use planners and managers.
February 2010	Water quality effectiveness of the water retention BMP at the two isolated wetlands in the Lake Okeechobee Basin.	S. Shukla, D. Goswami, W. D. Graham, A. Hodges, V. Nair.	Second Sustainable Water Resources Symposium, University of Florida Water Institute, Gainesville, FL.	University researchers, state and federal agencies, engineering consultants, agricultural landowners, and land use planners and managers.
May 2010	Presentation of preliminary results of cow-calf BMP evaluation.	S. Shukla	Southwest Florida Research and Education Center, Immokalee, FL	SFWMD staff
June 2010	Effects of cattle fencing and wetland water retention BMPs on phosphorus loadings.	S. Shukla, A. Hodges.	Lake Okeechobee Inter-Agency Meeting, Okeechobee, FL.	Ranchers, FDACS, FDEP, SFWMD, UF/IFAS researchers, Environmental Organizations, and land use planners and managers.
June 2010	Effect of wetland water retention on water storage and hydro-period in two isolated wetlands in the Lake Okeechobee basin, Florida.	D. Goswami, S. Shukla, W.D. Graham.	ASABE Annual International Meeting, Pittsburgh, PA	Scientists and Engineers

Appendix E

Quantification of Wetland Area ET Using Eddy-Covariance Method

Introduction

The recent emergence of Eddy Covariance (EC) method has significantly increased the accuracy of evapotranspiration (ET) estimates. The EC method is a conceptually simple, one-dimensional approach for measuring the turbulent fluxes of vapor and heat above a surface. Eddy Covariance sensors measure the vertical motions and admixtures (such as flux of heat, water vapor) between the surface and the atmosphere. The EC method has been used successfully to estimate ET in Florida (Bidlake et al, 1993; Knowles, 1996; Sumner 1996).

The EC technique offers several advantages to alternative water budget approaches (lysimeter or regional water budget) by providing better areal integration and less site disruption than lysimeters and eliminating the need to quantify other terms of a water budget (precipitation, deep percolation, runoff, and storage) (Sumner, 2001). It avoids soil surface heterogeneity issues by placing the sensors above the crop canopy, which makes it better suited for measuring ET for various types of vegetation including those found in wetlands (Sumner, 1996, 2001; Gholz and Clark, 2002; Sumner and Jacobs, 2005; Jia et al., 2007).

A project was started in 2003 to evaluate the cow-calf Best Management Practices (BMPs) at a beef cattle ranch within the Lake Okeechobee (LO) basin. One of the BMPs evaluated was wetland and pasture water retention (WWR). There exists uncertainty in quantifying the WWR effects on water retention due to the year-to-year climatic variability. Use of EC technique will help us better quantify ET from the wetlands. This will help answer the question of whether the water retained in the wetland and pastures reduces the net outflow from the ranch.

Materials and methods

An EC station was installed at the center of the actual wetland within the Wetland 1 area to provide adequate fetch in all directions. The EC system consists of a CSI (Campbell Scientific Inc., Logan, UT) CSAT3 three-dimensional sonic anemometer and a KH20 krypton hygrometer. The anemometer measures fluctuations in wind speed and virtual temperature using three pairs of non-orthogonal sonic transducers, and the hygrometer measures the fluctuations of water vapor density. The temperature and relative humidity were measured using a CSI HMP45C sensor. Wind speed and temperature fluctuation were measured using the CSI CSAT-3 anemometer. Water vapor density fluctuations were measured using the CSI KH20 krypton hygrometer. The instrument output (sampling frequency of 10 Hz) were processed and stored in a CSI CR1000 datalogger. Average fluxes were calculated and stored for every 30 minutes for the July 2009-July 2010 period.

Using the measured covariance between vertical wind speed and water vapor density, the latent heat flux (λE) can be calculated as:

$$\lambda E = \lambda \overline{\rho_v' w'} \qquad (1)$$

where λE is latent heat flux, in watts per square meter; λ is the latent heat of vaporization, in joules per gram; ρ_v' is the fluctuation in the water vapor density computed using the hygrometer (10 Hz measurements), in grams per cubic meter; w' is the fluctuation in the vertical wind speed computed using the anemometer 10 Hz measurements, in meters per second. Using the measured covariance between the vertical wind speed and the air temperature, the sensible heat flux (H) can be calculated as:

$$H = \rho_a C_p T' w' \qquad (2)$$

where ρ_a is the density of air, in grams per cubic meter; C_p is the specific heat of moist air, in joules per gram per degree Celsius, and T' is the fluctuation in the air temperature, in degrees Celsius.

The 30-minute latent heat flux data computed from Equation (1) was corrected for temperature-induced fluctuations in air density (Webb et al., 1980) and for the hygrometer sensitivity to oxygen (Tanner and Green, 1989). The sensible heat flux data was corrected for differences between the virtual temperature and the actual air temperature (Schotanus et al., 1983). Both the sensible and latent heat fluxes were corrected for misalignment with respect to the natural wind coordinate system (Baldocchi et al., 1988). The latent heat flux can be modified as:

$$\lambda E = \lambda\left(\overline{\rho_v' w'} + \frac{\rho_v H}{\rho_a C_p (T + 273.3)} + \frac{F K_o H}{K_w (T + 273.3)}\right) \quad (3)$$

where F is a factor that accounts for molecular weights of air and atmospheric abundance of oxygen, equal to 0.229 gram degree Celsius per joule; T is air temperature, in degrees Celsius; K_o is extinction coefficient of hygrometer for oxygen, estimated as 0.0045 cubic meters per gram per centimeter; K_w is extinction coefficient of hygrometer for water and is equal to the manufacturer-calibrated value, in cubic meters per gram per centimeter. The latent heat flux obtained from Equation (3) was corrected to ensure the energy balance closure. This correction was done using the Bowen ratio (B) which is given by:

$$B = H / \lambda E \quad (4)$$

Using the B value for each 30-minute time step, the λE λE was calculated using the following equation:

$$\lambda E = \frac{R_n - G}{1 + B} \quad (5)$$

where R_n is the net radiation, in watts per square meter; and G is the soil heat flux, in watts per square meter. Finally, the ET rate $ET_{c_EC}$for each 30-min period was calculated using the following equation:

$$E = \frac{\lambda E}{1000 \times \lambda \times 1800 \times \rho_w} \quad (6)$$

where E is ET_{C_EC} ET_{c_EC}in millimeters.

During certain periods, such as at nights, early mornings with dew formations, and after rainfall, the hygrometer measurements were not available due to the water on the KH20 lenses. Therefore, the data analysis was conducted only for daytime measurements. Daytime periods after rainfall were also excluded from the analysis. For period with missing data, fluxes were estimated from a modified Priestley-Taylor model (Priestley and Taylor, 1972).

The ET_{C_EC} was compared to the estimates from the FAO Penman-Monteith (PM) method, a method commonly used throughout the world for estimating ET for a variety of crops and natural vegetation. By comparing ET_{C_EC} and ET from the FAO Penman-Monteith (PM) method, uncertainty in quantifying ET could be evaluated. The meteorological data (air temperature, solar radiation, net radiation, wind speed and humidity) collected at the weather station at the ranch were used to calculate reference ET (ET_0) by using FAO Penman-Monteith (PM) method (Allen et al., 1998).

The FAO PM equation is represented as:

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$$ET_0 = \frac{1}{\lambda}\frac{0.408 \times \Delta(R_n - G) + \gamma \frac{900}{T + 273} U_2 (e_a - e_d)}{\Delta + \gamma(1 + 0.34U_2)} \quad (7)$$

where Δ is the slope of the saturated vapor pressure curve, in kilopascals per degree Celsius; T is mean daily air temperature at 2 m height, in Celsius; U_2 is wind speed at 2 m height, meter per second; e_a is saturated vapor pressure, in kilopascals; e_d is actual vapor pressure, in kilopascals; e_a-e_d is vapor pressure deficit, in kilopascals; and γ is a psychrometric constant, in kilopascals per degree Celsius. Using the FAO PM method to calculate ET requires the specific crop coefficient values from the literature. Mao et al. (2002) conducted a lysimeter experiment to estimate crop coefficients for different wetland environments in Fort Drum Marsh Conservation Area located approximately 60 km from Pelaez ranch. By using the crop coefficients of cattail and open water estimated by Mao et al. (2002), FAO-based ET (ET_{C_R}) for actual wetland at Wetland 1 site were estimated using the following equation:

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$$ET_{C\_R} = \frac{ET_0 \times K_{CO} \times \text{open water area} + ET_0 \times K_{CT} \times \text{vegetated area}}{\text{Wetland area}} \quad (8)$$

where K_{CO} is crop coefficient for open water; K_{CT} is the coefficient for cattail. The wetland area was delineated by using aerial photo and LIDAR topographic data. The open water area was estimated from the measured surface water levels and topographic data, and the vegetated area was calculated as wetland area minus open water area. Daily ETC_R values were calculated for the July 2009 to June 2010 period. To evaluate the uncertainty in the commonly used FAO PM method in estimating ET through the use of crop coefficients obtained from literature, the literature crop coefficient values were multiplied with ET_0 and resulting ET value, ET_{C_R}, was compared with the ET_{C_EC}. The average monthly crop coefficient values of cattail and open water from Mao et al. (2002) were used to calculate ET_{C_R}. The ET_{C_EC} was divided by ET_0 to calculate the monthly wetland crop coefficient (K_{cw}) for its use for similar wetlands in the LO basin.

Results and discussion

The daily average ET_{C_EC} and ET_0 values along with the rainfall are shown in Figure E-1. The total ET_0 for the July 2009-July 2010 period was higher than ET_{C_EC}. However, there were several days during the wet season (June to October) and the beginning of the dry season when the ET_{C_EC} was higher than ET_0. Larger inundated areas combined with the fact that this is the period of maximum growth of wetland vegetation resulted in ET_{C_EC} being higher than ET_0 (Figure E-1). Among the 12 months, the lowest average daily ET_{C_EC} value (1.02 mm/d) occurred in January 2010, and the highest value (4.94 mm/d) occurred in May 2010 (Table E-1).

Daily wetland vegetation (crop) coefficient (K_{CW}) values calculated as the ratios of ET_{C_EC} to ET_0 are presented in Table E-1. The monthly K_{CW} values ranged from 0.46 to 1.18. The highest and lowest K_{CW} values were observed for October 2009 and February 2010 (Figure E-2), respectively. Relatively higher K_{CW} values observed for the March-May 2010 (Figure E-2) may partly due to above average rainfall for this period.

From the July 2009-June 2010 period, total ET_{C_R} was 936 mm while the ET_{C_EC} was 1140 mm. Assuming that ET_{C_EC} represents the actual ET, results show that use of commonly used ET estimation method could result in underestimating ET by 18 %. Daily ET for the two methods (EC-based and FAO-PM based) shown in Figure E-3 clearly shows that ET_{C_EC} was higher than ET_{C_R} for almost the entire monitoring period.

Results from the EC-based ET data show that use of the commonly used FAO-PM method can result in significant underestimation of ET losses from wetlands located at ranchlands within the LO basin. These errors are likely to result in errors in estimating the effect of WWR BMP on water retention. Use of EC-based ET in conjunction with a hydrologic model is likely to improve the evaluation of WWR with regards to surface and subsurface storages and flow. The fate of increased storage as a result of WWR and the available storage for successive storms is mainly influenced by the subsurface water fluxes, and up to some extent the ET losses. Therefore, models used for evaluating the WWR should have the ability to capture the strong surface and groundwater interactions that exist in the LO basin. While increased storage of rainfall in the wetland drainage area may reduce the flow volume and rate for the period when significant surface and subsurface storage is available, it may increase the flow during the periods when the available storage is already occupied due to past rainfall events. Given the high hydraulic conductivity of the soils in the region, a significant portion of the additional storage created by the WWR may move through subsurface pathways and appear as surface flow downstream of the WWR structure. The only "true" retention due to WWR will be the increased ET due to increased inundation and soil wetness in the drainage area. Results from the study indicate that the ability to quantify this "true" retention can be improved by using the long-term eddy data for the wetland as well as upland areas along with the groundwater monitoring data within a hydrologic model that has the ability to effectively simulate the surface and groundwater interactions. Given the current focus on increasing the distributed storage in the LO basin, future efforts should focus on designing ranch-specific strategies that will result in actually reducing the flows from the ranchlands of south Florida.

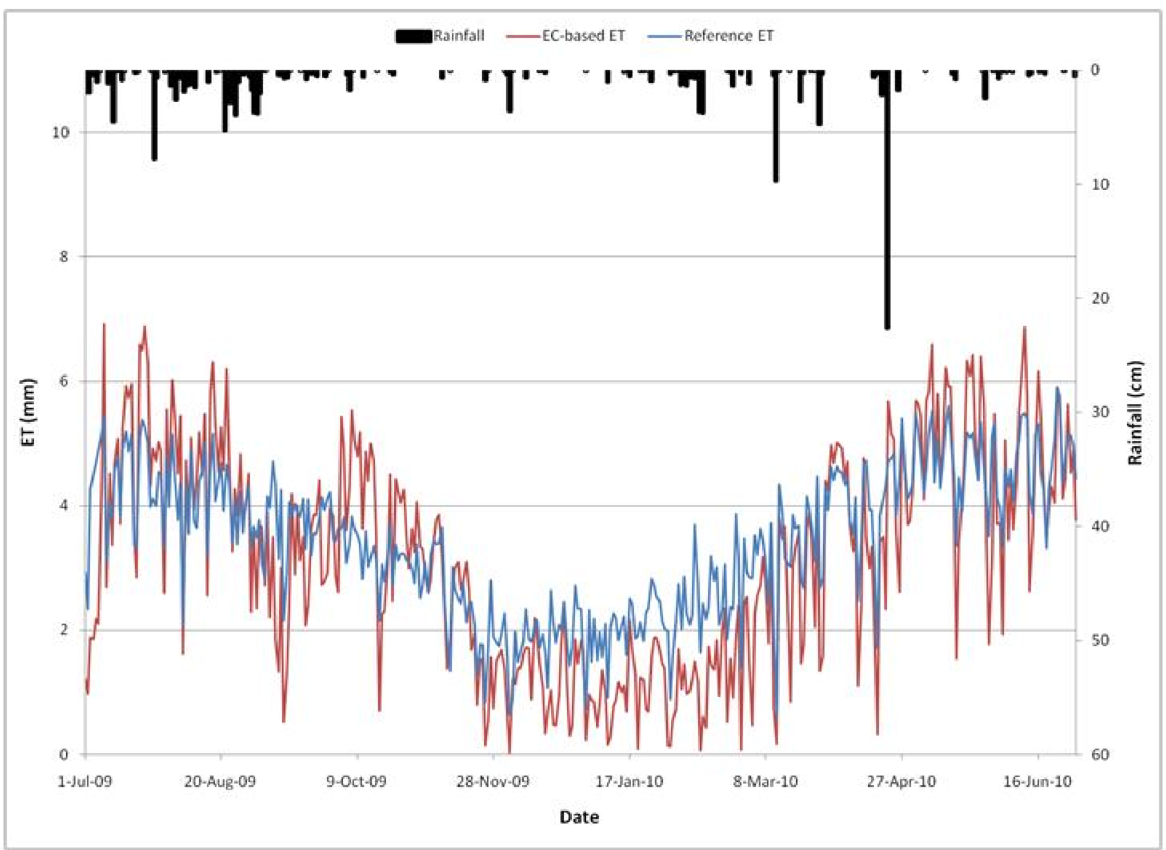

Figure E-1. Rainfall, Eddy Covariance (EC) based ET, and FAO PM reference ET for wetland 1 for the July 2009-June 2010 period.

Table E-1. Average daily Eddy Covariance (EC) ET (ET_{c_EC}, mm/d), FAO-PM PET (ET_0, mm/d), and wetland (crop) coefficient (K_{CW}) for 12 months for the July 2009-June 2010 period.

Year	Month	ET_{C_EC} (mm/d)	ET_O (mm/d)	K_{CW}
2009	7	4.29	4.36	0.96
	8	4.52	4.11	1.09
	9	3.01	3.72	0.81
	10	3.88	3.23	1.18
	11	2.28	2.39	0.92
	12	1.23	1.86	0.66
2010	1	1.02	2.03	0.49
	2	1.22	2.46	0.46
	3	2.60	3.28	0.75
	4	3.80	4.14	0.89
	5	4.94	4.75	1.03
	6	4.54	4.68	0.96

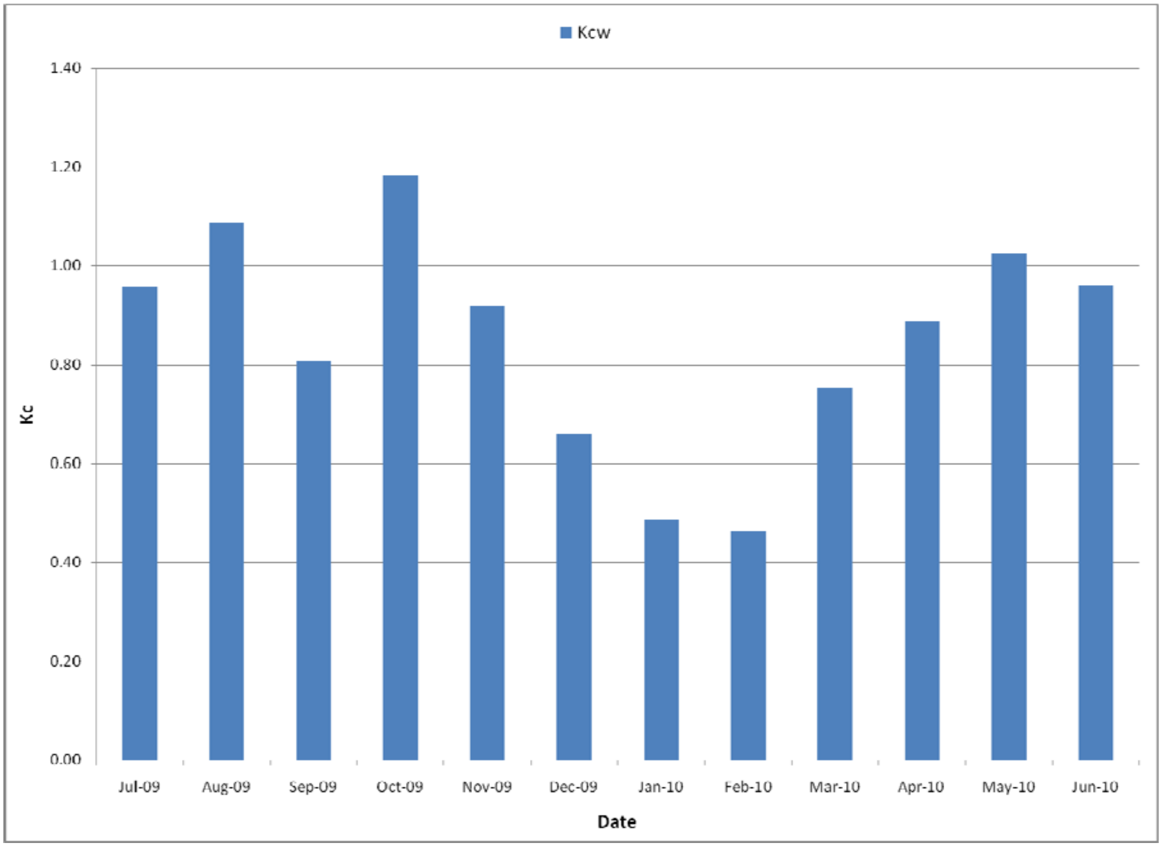

Figure E-2. Monthly wetland (crop) coefficients (K_{CW}). The K_{CW} values were calculated from the Eddy Covariance (EC) ET (ET_{c_EC}, mm/d) and FAO-PM PET (ET_0, mm/d).

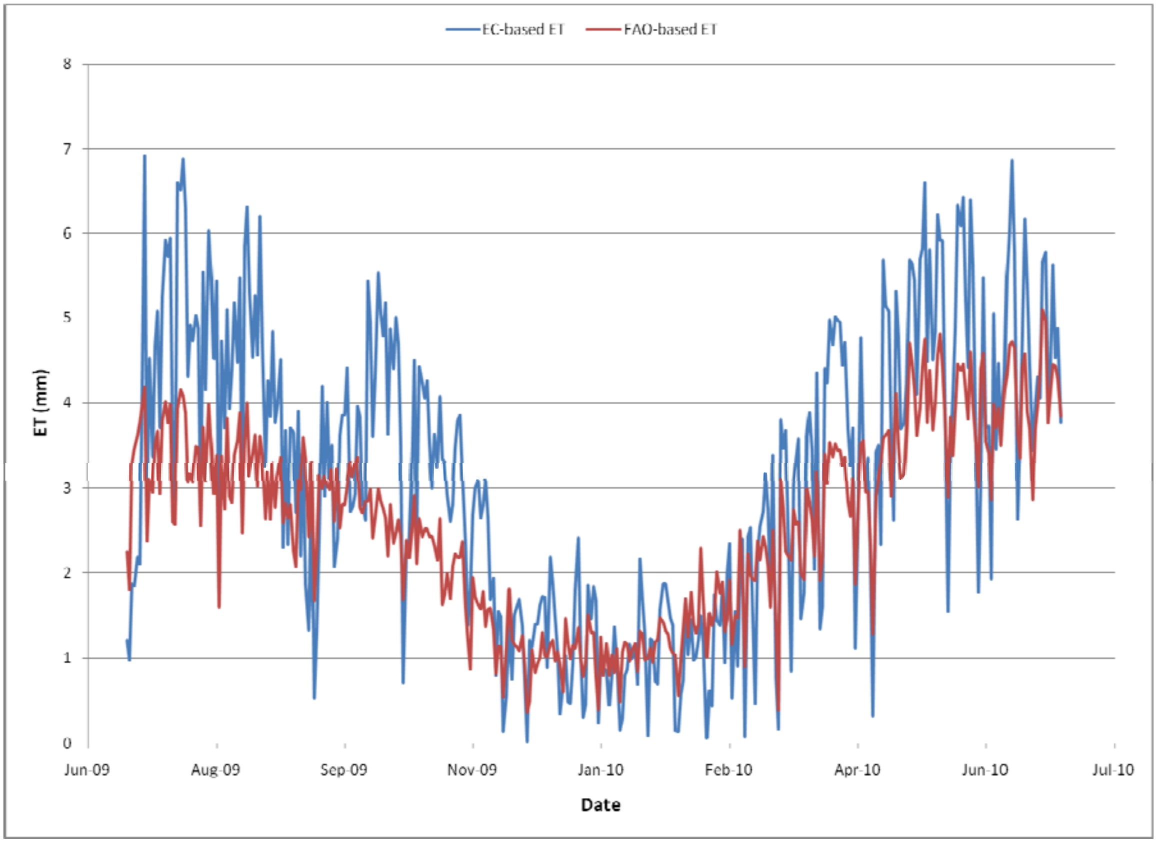

Figure E-3. Daily ET from the Eddy Covariance (EC) and the FAO-PM methods for the July 2009-June 2010 period.

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Appendix F

Evaluation of Water Retention Scenarios Using WAM

Introduction

Lake Okeechobee (LO) is a large, multi-functional lake located at the center of the Kissimmee-Okeechobee-Everglades aquatic ecosystem in Florida. It is also a source of drinking water for lakeside cities and towns, and irrigation water for the expansive Everglades Agricultural Area (EAA). Lake Okeechobee receives most of its inflow from central Florida via the flows of the Kissimmee River. During the 20th Century, much of the land around LO was converted to agricultural use (Harvey and Havens, 1999). Changes in land use and the excavation of agricultural drainage ditches have increased the capacity of the landscape to drain more water to LO and have caused decreased storage in the watershed. Associated with the land use changes, were large increases in the rate of nutrient (e.g. Nitrogen and Phosphorus) inputs to the lake which resulted in degradation of lake water quality.

Excessive phosphorus (P) loading is one of the serious problems facing the lake (Boggess et al., 1995; Rice et al., 2002). Nonpoint source pollution from cow-calf operation is a matter of concern in the LO basin. P loadings in this basin have exceeded the Total Maximum Daily Load (TMDL) for LO. Therefore, a cow-calf BMP manual was developed by the Florida Cattlemen's Association in 1999 in cooperation with Florida Department of Agriculture and Consumer Services (FDACS), Florida Department of Environmental Protection (FDEP), and South Florida Water Management District (SFWMD). The purpose of this manual was to educate and encourage adoption 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.

A project was started in 2003 to evaluate effectiveness of two BMPs, namely ditch fencing and culvert crossing (DFCC), and wetland water retention (WWR) in a cow-calf ranch in the LO basin. The goal of this subtask is to evaluate the different water retention scenarios by using Watershed Assessment Model (WAM) (Bottcher et al., 1998) in conjunction with the hydrologic data collected for the cow-calf BMP evaluation study.

Model Description

Watershed Assessment Model (WAM)

Watershed Assessment Model (WAM) is a comprehensive Geographic Information System (GIS) based software developed by Soil and Water Engineering Technology, Inc. (SWET). It allows planners to interactively simulate and assess the environmental effects of changing land uses. WAM originated from the Basin New Zealand (BNZ) model. To create WAM, the BNZ model was modified by adding a new routing component, incorporating the Groundwater Loading Effects of Agricultural Management System (GLEAMS) model (Leonard et al., 1987; Knisel et al., 1993), and by creating simple sub-models for wetland and urban areas. WAM has been customized for use by various water management districts across Florida (Bottcher et al., 1998). The WAM interface was written for a previous version of ARC/INFO (Environmental Systems Research Institute (ESRI), CA) which used the ARC/INFO Macro Language (AML). With the introduction of ArcView 3.0 (ESRI, CA), the model was upgraded to WAMView. The model is now capable of simulating primary processes that are necessary for studying watershed hydrologic and pollutant transport and has therefore been extensively used in Florida. The GIS interface of the modeling system represents the watershed in a grid cell based format and requires systematic input of spatial data. WAM is capable of routing surface water and groundwater flows from the cells throughout the watershed.

Model Set-up

WAM, a GIS-interfaced model, was set up for the entire ranch using a grid size of 10 m by 10 m. The ditch network, represented as a stream feature in WAM, was created by examining aerial photos in conjunction with field surveys. The GIS interface also created stream profile files which were created by using the stage-volume relationship derived from LIDAR topographic data. The setup of sub-basins and main ditches is shown in Figure F-1. The data on stocking rate, grazing schedule, fertilizer application and supplemental feed collected from the rancher was incorporated in the models.

In WAM manual, it is recommended that the simulations have a startup-period of at least 3 years to give the soils time to reach equilibrium. However, the weather station started collecting weather data from the second quarter of 2004 at Pelaez ranch. In order to supplement the model with additional weather data, a dataset (2002-2004) from the MacArthur Agro-Ecology Research Center (MAERC) at Buck Island Ranch was utilized. The Buck Island Ranch is located 15 miles from Pelaez Ranch.

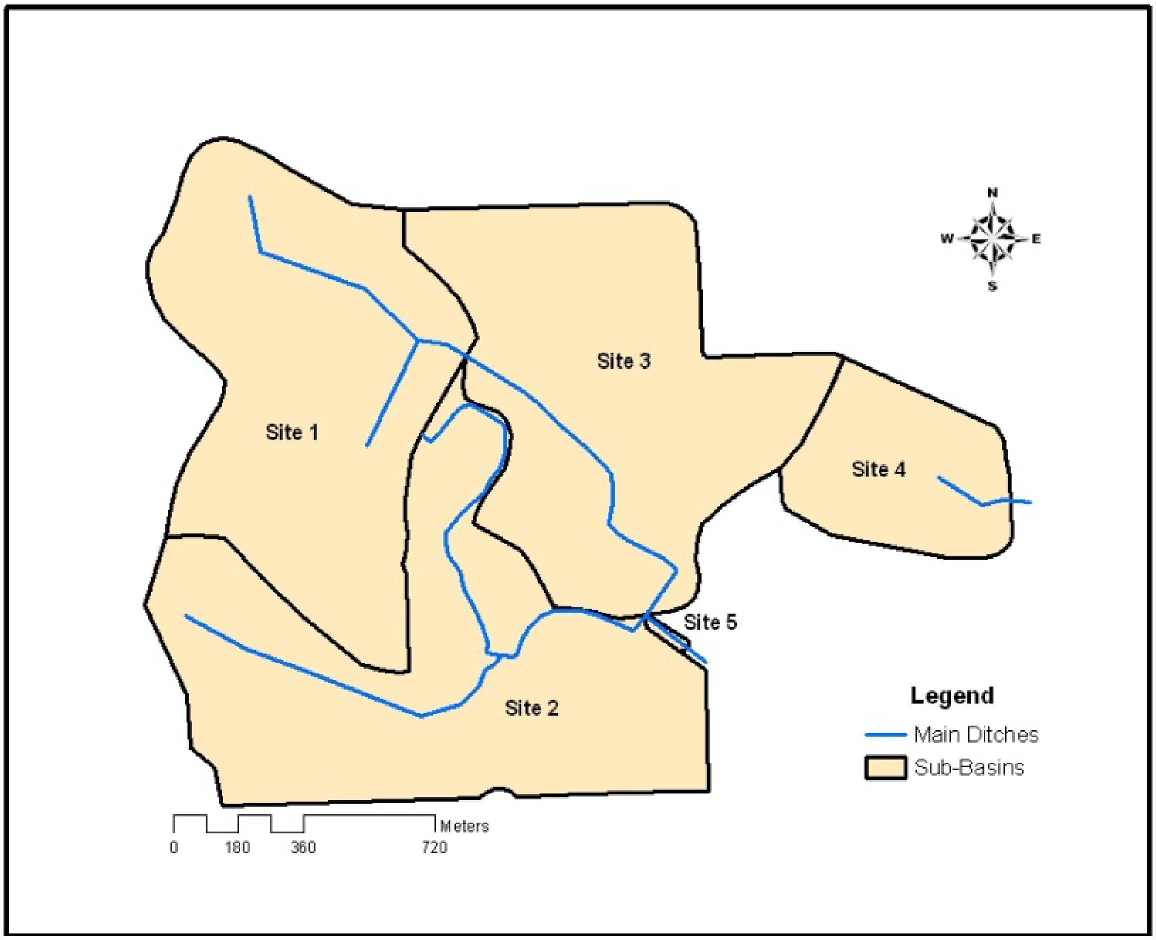

Figure F-1. The setup of sub-basins and stream features in WAM.

Model Calibration and Validation

Calibration is the process by which model parameters are adjusted to obtain the best fit between simulated results and observed data. Pre-BMP period data was used for model calibration. Pre-BMP periods for Site 1 and Site 4 are January 2005-January 2007 and January 2005-February 2006, respectively. After calibration, the model needs to be validated for a period different from the calibration period. The purpose of validation is to determine if the model is sufficiently accurate for its application of the simulation study. Post-BMP data was used for model validation. Post-BMP periods for Site 1 and Site 4 are February 2007-December 2008 and March 2006-December 2008, respectively.

The evaluation of hydrologic model behavior and performance is commonly made and reported through comparisons of simulated and observed variables. Efficiency criteria are used by hydrologists to provide an objective assessment of the "closeness" of the simulated behavior to the observed measurements (Krause et al., 2005). The most fundamental approach to assessing model performance in terms of behaviors is through visual inspection of the simulated and observed graphics. In addition to graphical analysis, other commonly used criteria in the literature are the root mean square error (RMSE) and the index of agreement (*d*). RMSE and *d* are expressed as follows:

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}((O_i - S_i)^2)}$$

$$d = 1 - \frac{\sum_1^n (O_i - S_i)^2}{\sum_1^n (|S_i - \bar{O}| + |O_i - \bar{O}|)^2}$$

where O and S are the observed and simulated values, respectively; $\bar{O}$ is the mean observed value; and n is the total number of observations. The *d* is a dimensionless term that varies from 0 to 1, with 1 representing a perfect agreement between the model and the observed data.

Water Retention Scenarios

After WAM was calibrated and evaluated, WAM was used to quantify the volume of water retained for different WWR scenarios. These scenarios are as follows: 1) increase current board height by 15 cm; 2) increase current board height by 30 cm in the dry season (November-May) and 15 cm in the wet season (June-October); 3) no board condition; 4) increase current board height by 30 cm. These scenarios were evaluated for the January 2008-December 2010 period.

Results and Discussions

Preliminary results show that the index of agreement (d) for Site 1 for the calibration and evaluation periods was 0.65 and 0.42, respectively (Table F-1) while it was 0.67 and 0.49, respectively for Site 4 (Table F-2). WAM performed better for the pre-BMP period for both Site 1 and Site 4 than the post-BMP period. The simulated runoff was closer to the measured runoff for the pre-BMP period but not for the post-BMP period (Figure F-2, Figure F-3, Table F-1, and Table F-2).

The inability of WAM to simulate the observed runoff may be due to an issue about the scale and resolution of the GIS data used in the model. The elevation of the low and the upland area could be averaged when these two areas are delineated into a single grid. Furthermore, the model uses 3-day unit hydrograph and 90-day unit hydrograph for simulating surface and groundwater flow to the stream feature that resulted in underestimating peak flow when there is a high rainfall storm. In WAM, the ET rate was calculated by using the Penman-Monteith method in conjunction with crop coefficients from the literature. However, WAM only uses one value for the whole year which may not represent the vegetation status on a seasonal basis and the increased ET losses due to the wetland water retention.

The results of evaluated scenarios were presented in Table F-3. Results show that when the board height at Site 1 was increased by 15 cm and 30 cm, the total runoff was reduced by 17% and 34%, respectively. When the board height at Site 4 was increased by 15 cm and 30 cm, the total runoff was reduced by 3% and 17%, respectively. Results from WAM show that the total runoff for the January 2008-December 2010 period can be reduced by 24% by implementing the current level of WWR implementation at Site 1 while it was reduced by 6% at Site 4. This is mainly due to the difference in topography and the board height which results in higher storage (below the top of the board) created at Site 1 compared to Site 2. Although WAM results show that the WWR can reduce the runoff from the site, given the less than satisfactory performance of the model, further investigation of specific processes such as ET, surface and groundwater interactions, and runoff generation are needed before results from this model can be used for evaluating the effects of ranchland water retention. Consider for example, the ET predictions for

WAM for the July 2009-June2010 period. The Eddy-based ET estimates for the actual wetland plus the FAO-PM-based estimates for the upland for this period was 93 cm for the Wetland 1 area while the WAM-predicted value was 191 cm (105% higher). Furthermore, the runoff predictions at Site 1 for the August 19, 2008-September 4, 2008 period which resulted in high flow mainly due to the Tropical Storm Fay (rainfall = 30 cm), were 4.6 cm while the observed runoff was almost four times higher (18.8 cm). As noted in Appendix B-1, observed runoff data from Site 1 (Wetland 1) showed that this period accounted for almost 80% of the total flow for the post-BMP 2 period. Similarly, the predicted runoff for Site 4 (Wetland 2) for the same period was 5.9 cm as compared to the observed runoff of 27.8 cm (4.7 times higher than simulated value). These underpredictions could partly be due to the inadequate representation of susbsurface processes as well as the process of runoff generation and routing in WAM. Given the strong interaction of surface water and groundwater in the region, efforts are needed to better represent these processes in WAM.

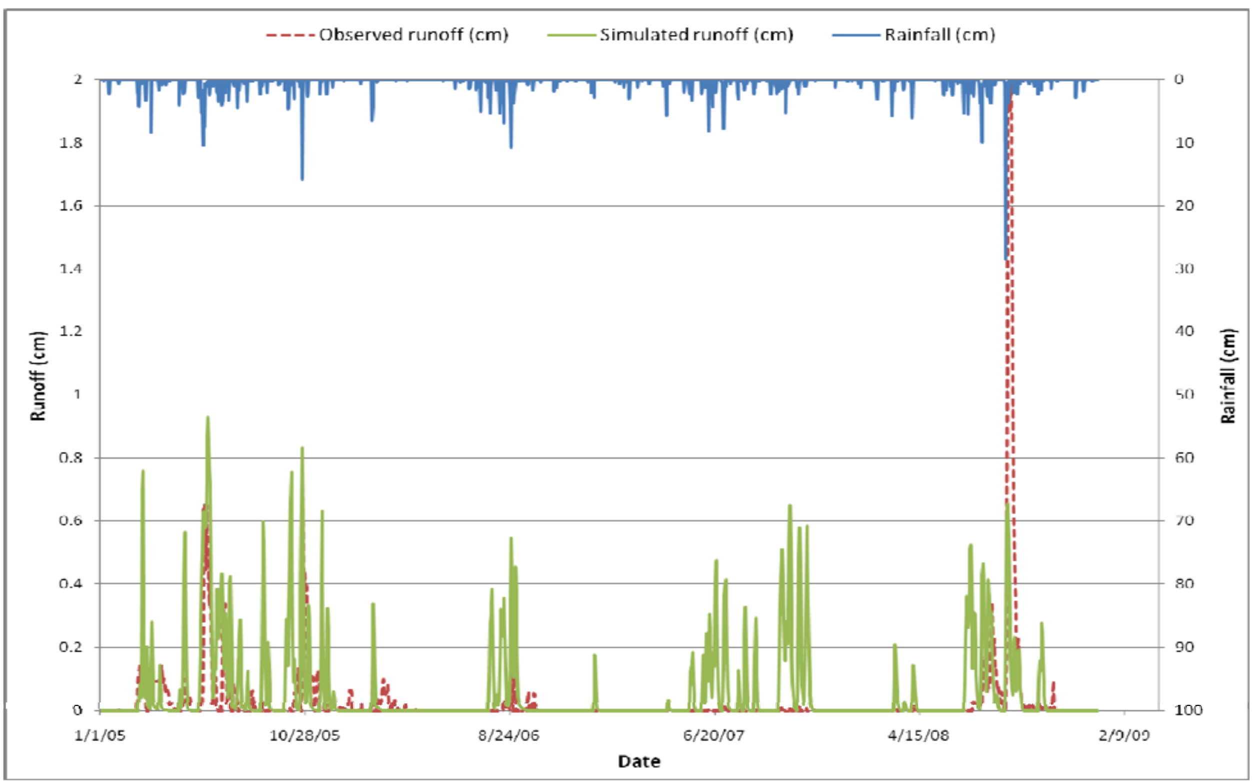

Figure F-2. Rainfall, observed and simulated runoff for Site 1.

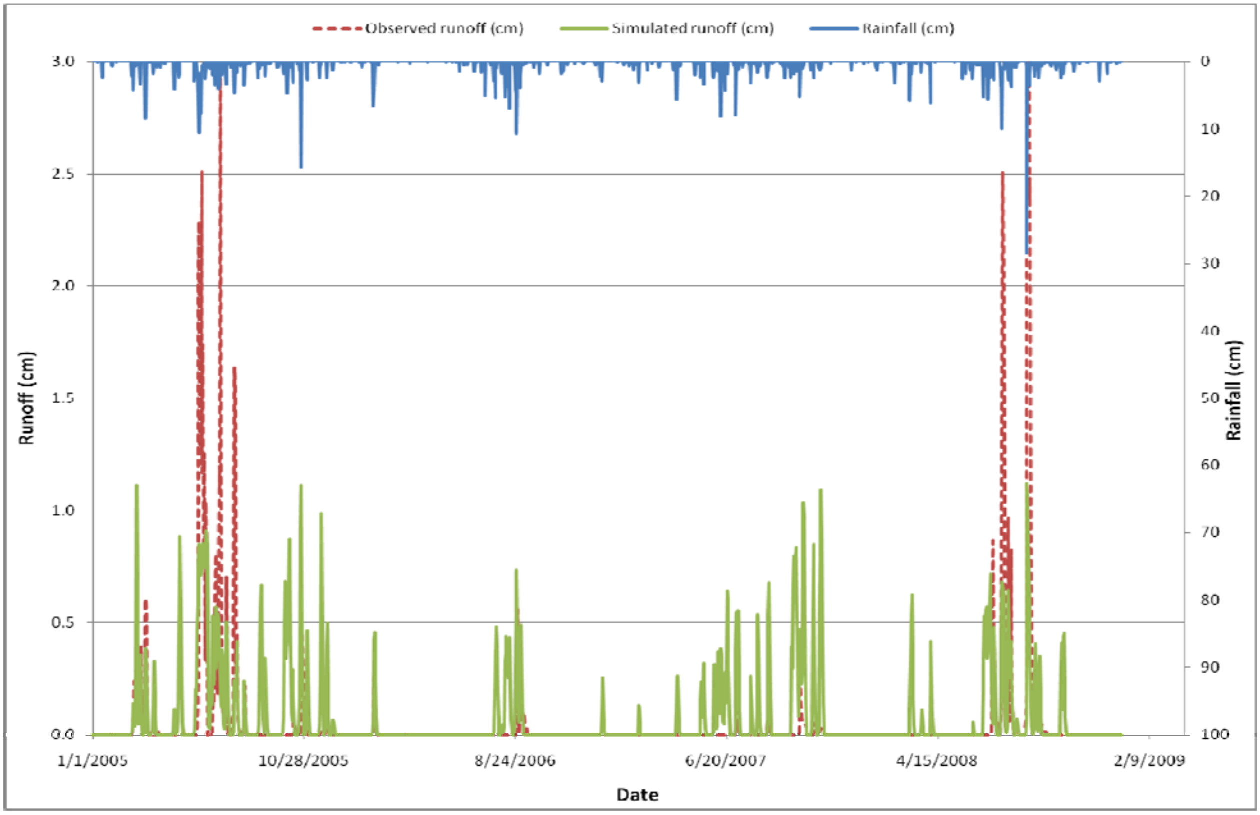

Figure F-3. Rainfall, observed and simulated runoff for Site 4.

Table F-1. Statistical analyses results (Mean, standard deviation, root mean square error (RMSE) and index of agreement (d)) of WAM performance for runoff simulations at Site 1.

Mean (cm)				Standard Deviation				RMSE		*d*	
Calibration		Evaluation		Calibration		Evaluation					
Obs[a]	Sim[b]	Obs	Sim	Obs	Sim	Obs	Sim	Cali[c]	Eval[d]	Cali	Eval
0.03	0.06	0.03	0.06	0.09	0.15	0.22	0.12	0.13	0.22	0.65	0.43

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

Table F-2. Statistical analyses results (Mean, standard deviation, root mean square error (RMSE) and index of agreement (d)) of WAM performance for runoff simulations at Site 4.

Mean (cm)				Standard Deviation				RMSE		*d*	
Calibration		Evaluation		Calibration		Evaluation					
Obs	Sim	Obs	Sim	Obs	Sim	Obs	Sim	Cali	Eval	Cali	Eval
0.10	0.12	0.04	0.07	0.34	0.22	0.39	0.17	0.31	0.36	0.67	0.49

Table F-3. Comparison of observed and simulated total runoff for the present condition and the water retention scenarios.

	Site 1		Site 4	
	Total runoff (cm)	% of runoff change	Total runoff (cm)	% of runoff change
Present condition	46.6		79.3	
Simulated present condition	51.6		74.3	
Scenario 1	42.9	-17	72.3	-3
Scenario 2	41.7	-19	68.8	-7
Scenario 3	63.9	24	78.5	6
Scenario 4	34.3	-34	61.6	-17

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