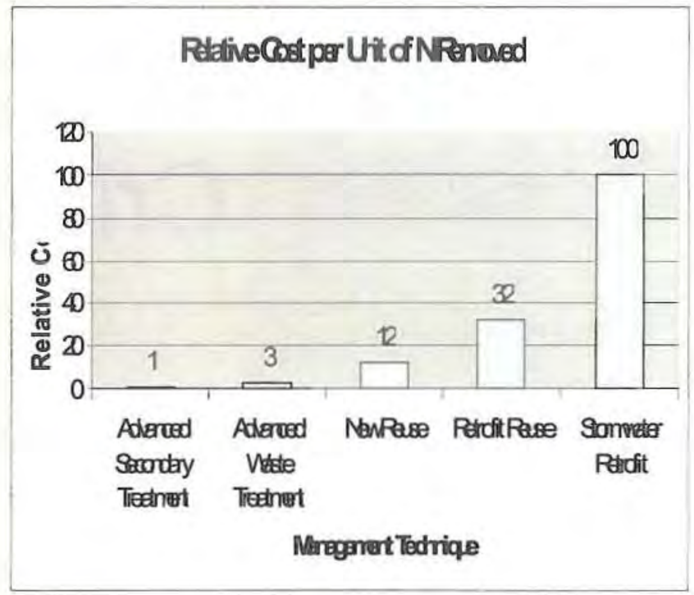
FWEA Utility Council
–
Florida Rural Water Association

Meeting with OMB Regarding Numeric Nutrient Criteria for Florida

January 6, 2010

NE Florida's Lower St. John's River Initiative
Now ...

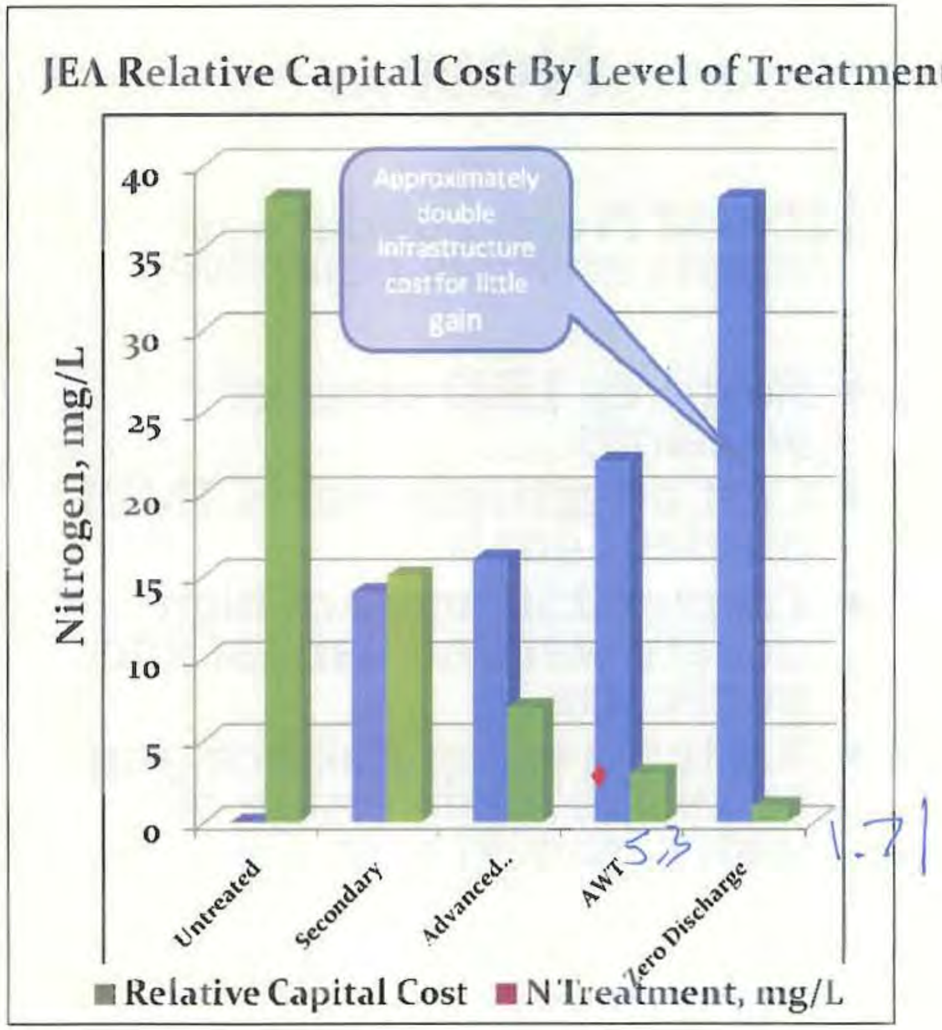
- Stakeholder driven process set scientifically derived nutrient reduction goals to restore river
- Water quality credit trading approach allows efficiencies for Ag/City's/Utilities
  - Allows utilities to optimize projects
  - Allows entities to meet environmental goals for lowest public cost
- ~$500MM in projects planned, underway or completed in basin to restore river



5.3 mg/L N in marine

NE Florida's Lower St. John's River Initiative
With NNC…

- Arbitrary NNC targets could require each entity to meet limits – removes benefit to environment and public of water quality credit flexibility
- Would approximately double utility cost of compliance with no additional environmental benefit
  - JEA estimates $2 billion to meet NNC criteria



GRU Paynes Prairie Initiative

Now...

$20MM Project underway meets multiple objectives

- Restores 1300 acres of wetlands
- Cost effectively meets TMDL nutrient goals
- Creates 150 acres of high-quality wetland habitat and public use
- Restores water balance due to 1930's Ag diversion of water from Prairie

With NNC ....

NNC limits would derail this project and preclude environmental restoration of the Prarie through this project

- 400% the Capital Cost
- 50% increase in Energy Use

Projected Impact to Ratepayers

- Approximately doubles the typical residential water/sewer bill for most utilities
- Cost of this will be born by our citizens and businesses for dubious environmental benefit
- Diverts public resources

Estimated Capital Costs and Increases in Sewer Rates for Eight Florida Utilities and an Average Florida Case to Construct Facilities to Meet Proposed Numeric Nutrient Limits

	Capital Cost	Monthly (Annual) Sewer Rate Increase per Household
STATE OF FLORIDA[2]	**$24,400,000,000-$50,700,000,000**	**$62.[2] ($740)**
Bay County	$42,000,000	$ 57 ($685)
Broward County	$425,000,000	$ 66 ($793)
Destin Water	$34,000,000	$ 48 ($581)
Escambia County	$275,000,000	$ 49 ($591)
Hollywood	$370,000,000	$ 82 ($996)
Jacksonville	$2,000,000,000	$ 67 ($815)
Point Buena Vista[3]	$2,000,000	$257 ($3,094)
Cross City[3]	$5,800,000	$ 28 ($336)
South Walton[3]	$16,000,000	$ 12 ($147)

Notes:

1.The low end of the range provides the probable opinion of cost assuming only plants with surface water discharges will be required to meet numeric nutrient limits while the high end of the range assume that all plants will need to meet numeric nutrient limits.

2.Estimated average costs for the State of Florida include annual O&M expenses, and are shown for comparative purposes.

3.Assumes 2.5 persons per connection and 150 gpcd.

Costs Will Disproportionately Effect Low Income Citizens

Impact of NNC on Monthly Water and Wastewater Charges

- NNC Requirements Projected to Increase Water/Sewer Rates >100%
- Current Median Monthly Combined Water and Wastewater Bill = $ 56
- Projected Median Monthly Combined Water and Wastewater Bill After NNC = $118

Source: GRU Comparison of 18 Utilities, June 2009

Affordability for Low Income Utility Customers

- 2009 Poverty Threshold for Family of 4 = $22,050
- Water and Wastewater Costs Less Than 4% of Household Income Considered "Affordable"
- Combined Water and Wastewater Monthly Charge of $73.50 Affordable for Florida Families in Poverty
- 12.1% of Florida Citizens At or Below Poverty Level
- Cost to Implement NNC Requirements will make Water and Wastewater Costs Unaffordable for Floridians in Poverty

Source: Water Affordability Programs, AWARF 1998, Margot Saunders, Phyllis Kimmel, Maggie Spade, Nancy Brockway

South Walton Utility Company

- **Late 1980's** – Collection system extended along Cocohatchee Bay to replace septic tanks
- **1991** – Irrigation Reuse Program Implemented
- **2003** – Study to identify additional reclaimed water disposal alternatives
- **2008** – AWT project initiated; will achieve 3 mg/L-TN and 0.5 mg/L-TP
- All collection and treatment system upgrades were paid for by SWUC members

EPA Numeric Nutrient Criteria will...

- Force Florida's regulated public to "recover" water bodies to nutrient levels they would not naturally meet
- Unseat Florida's existing, EPA-approved nutrient TMDLs
- Render moot ongoing environmentally beneficial projects designed to meet EPA-approved nutrient TMDLs
- *Misallocate limited public resources*
- *Impose severe economic burdens on Floridians*

Nitrogen Mass Balance of a Tile-drained Agricultural Watershed in East-Central Illinois

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Simple nitrogen (N) input/output balance calculations in agricultural systems are used to evaluate performance of nutrient management; however, they generally rely on extensive assumptions that do not consider leaching, denitrification, or annual depletion of soil N. We constructed a relatively complete N mass balance for the Big Ditch watershed, an extensively tile-drained agricultural watershed in east-central Illinois. We conducted direct measurements of a wide range of N pools and fluxes for a 2-yr period, including soil N mineralization, soybean N_2 fixation, tile and river N loads, and ground water and in-stream denitrification. Fertilizer N inputs were from a survey of the watershed and yield data from county estimates that were combined with estimated protein contents to obtain grain N. By using maize fertilizer recovery and soybean N_2 fixation to estimate total grain N derived from soil, we calculated the explicit change in soil N storage each year. Overall, fertilizer N and soybean N_2 fixation dominated inputs, and total grain export dominated outputs. Precipitation during 2001 was below average (78 cm), whereas precipitation in 2002 exceeded the 30-yr average of 97 cm; monthly rainfall was above average in April, May, and June of 2002, which flooded fields and produced large tile and riverine N loads. In 2001, watershed inputs were greater than outputs, suggesting that carryover of N to the subsequent year may occur. In 2002, total inputs were less than outputs due to large leaching losses and likely substantial field denitrification. The explicit change in soil storage (67 kg N ha^{-1}) offsets this balance shortfall. Although 2002 was climatically unusual, with current production trends of greater maize grain yields with less fertilizer N, soil N depletion is likely to occur in maize/soybean rotations, especially in years with above-average precipitation or extremely wet spring periods.

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Nitrogen budget calculations in agricultural systems are useful for developing a quantitative understanding of N sources and sinks and assessing overall availability of N to the target crop species as well as efficiency of utilization. These calculations range from simple input/output budgets at the field, watershed, or regional scale to intensive mass balance evaluations at the microplot and small field scale (Watson and Atkinson, 1999). However, accounting for all N fluxes and obtaining a complete N mass balance is extremely challenging due to the inherent complexity of the N cycle and the difficulty in directly measuring various fluxes, particularly denitrification (Davidson and Seitzinger, 2006). Therefore, simple field budgets are more commonly used as performance indicators of nutrient management and as regulatory policy instruments, especially in Europe (Oenema et al., 2003).

In conventional agricultural systems, N budgets generally identify fertilizer N as the major input and N contained in grain as the major output. In the Midwest, where maize (*Zea mays* L.)/soybean (*Glycine max* L.) rotations are the predominant cropping system and tile drainage is extensive, N inputs often include an estimate of soybean N_2 fixation, and N outputs include N leaching from tiles (McIsaac et al., 2002). A mass balance, on the other hand, implies a more rigorous investigation into N pools and fluxes throughout the plant/soil system and often involves applying ^{15}N to microplots (Stevens et al., 2005). Regardless of the experimental rigor, N accounting and budgeting has been used to evaluate the potential negative impact of agricultural production on water quality and more recently on the soil resource (Jaynes and Karlen, 2008).

Numerous studies in the Midwest have presented field N budgets to evaluate the effects of agricultural practices on N leaching losses (Kladivko et al., 1991; Gentry et al., 1998; Karlen et al., 1998; Andraski et al., 2000; Jaynes et al., 2001; Webb et al., 2004). These studies show that leaching losses can be substantial and are largely dependent on the rate of fertilization, soil type, and precipitation. There are a variety of cultural practices that can improve field N balances and decrease N loss, including timing of N application, variable rate technology, use of nitrification inhibitors and slow release fertilizers,

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Abbreviations: NNI, net nitrogen input.

and cover crops (Randall and Vetsch, 2005; Mamo et al., 2003; Snapp et al., 2005). There have also been N budget calculations on the watershed scale (David et al., 1997; Burkart and James, 1999; McIsaac et al., 2002; Libra et al., 2004). Collectively, these studies have established a clear link between agricultural production and riverine N loads but do not account for all N pools (amount of stored N) and fluxes (movement of N).

Our goal in this study was to directly measure as many of the major N inputs and outputs as possible, supplemented with estimates where needed, to construct a relatively complete N mass balance in an extensively tile-drained agricultural watershed under a maize/soybean production system. We conducted direct measurements of a wide range of N pools and fluxes for a 2-yr period including soil N mineralization, soybean N_2 fixation, tile and river N loads, and ground water and in-stream denitrification. Due to the lack of animal production and municipal wastewater discharge, this agriculturally dominated watershed was well suited to evaluate the linkage between terrestrial N cycling and riverine N load.

Materials and Methods

Site Description

The Big Ditch watershed (101 km^2) is a relatively flat and extensively tile-drained area dominated by row crop agriculture (89%) with approximately an equal mixture of maize and soybean planted annually, typical of east-central Illinois watersheds (previously described by Borah et al., 2003; Schaller et al., 2004; Royer et al., 2006; Mehnert et al., 2007). In conjunction with this study, Mehnert et al. (2007) provide a detailed map of the watershed. Our measurements of N pools and fluxes in the Big Ditch watershed were conducted in 2001 and 2002 (riverine and tile N concentrations and loads were calculated on a water-year basis: October 2000 through September 2002). We estimated land area annually planted to maize and soybean throughout the watershed based on the ratio of these two crops in Champaign County, Illinois (Illinois Agricultural Statistics). Local precipitation was estimated by averaging available daily observations from the three closest weather stations (Rantoul, Fisher, and Mahomet), although there were occasional missing values for these stations. Monthly mean precipitation values were also obtained from the eastern Illinois Climate Division operated by the National Oceanic and Atmospheric Administration (National Climate Data Center, 2008). The water-year average precipitation in the climate division during 1971 to 2000 was 97 cm, with a range of 65 cm in 1988 to 141 cm in 1993.

Nitrogen Inputs

Values of atmospheric N deposition in the Big Ditch watershed were from the National Atmospheric Deposition Program/National Trends Network site at Bondville, IL (NADP, 2008), which was located just outside the watershed boundary. Fertilizer N rate for maize was estimated at 184 kg ha^{-1} based on a farmer survey conducted in the Big Ditch watershed in 2000 (von Holle, 2005). This survey found that about 50% of maize fields received fall application of anhydrous ammonia. Soybean N_2 fixation rates were determined by the difference method, subtracting the amount of above-ground N accumulation of non-nodulated soybean from the N accumulation of nodulated soybean and dividing by the N accumulation of the nodulated soybean (Vasilas and Ham, 1984; Gentry et al., 2001). Values for soybean N fixation in the watershed were determined by multiplying N fixation rate by soybean plant N.

Following the experimental design, plot size, and cultural practices by Gentry et al. (2001) and Bergerou et al. (2004), two adjacent tile-drained fields separated by a small tributary of the Big Ditch that were alternately cropped to maize or soybean in a maize/soybean rotation were selected for microplot study. During the soybean phase of the rotation in these fields, microplots of nodulating and non-nodulating isolines of Williams 82 were established in a randomized block design with four replicates on the predominant silty clay loam soil type (Drummer/Flanagan silty clay loam, fine-silty, mixed, superactive, mesic Typic Endoaquolls) in the watershed. Soybean plants in a 1-m section of row (0.76-m row spacing) were harvested at the late R6 growth stage before leaf drop, divided into two plant fractions (leaves and stalks, and pods and seeds), and dried to a constant weight at 80°C for biomass determination. Dried samples were ground through a 2-mm mesh and analyzed for total N using a combustion technique (Fisons NA 2000 N Analyzer; Fisons Instruments, Strada Rivoltana, Italy).

Nitrogen Outputs

Grain yields of maize and soybean for Champaign County were used for yield values in the Big Ditch watershed (Illinois Agricultural Statistics, 2000–2001). Grain N content was calculated by multiplying grain yield by grain N concentrations of 1.44% for maize and 6.4% for soybean. Grain N concentrations were calculated using an average grain protein concentration of 9% for maize and 40% for soybean (University of Illinois, 2008) and dividing by the average mass ratio of N to grain protein (1:6.25). Total plant N was calculated by dividing grain N by the N harvest index of 0.70 for maize and 0.80 for soybean (David et al., 1997).

Daily river N loads were determined by multiplying daily discharge by inorganic N (including nitrate-N [NO_3–N] and ammonium-N [NH_4–N]) and by total N concentrations. A total of 241 river samples were analyzed during the 2001 and 2002 water years. Linear interpolation was used to estimate N concentrations between sampling dates using SAS 8.2. Water samples were collected on a weekly basis and supplemented with an automated water sampler for periods of rapid change in discharge (ISCO 2900; ISCO, Lincoln, NE). Filtered water samples (0.45 μm) were analyzed for NO_3–N on an ion chromatograph (Dionex, Sunnyvale, CA) and for NH_4–N on a Lachat Quikchem8000 (Lachat, Loveland, CO) flow injection analyzer (American Public Health Association, 1998). For total N, unfiltered aliquots underwent persulfate digestion and were analyzed for NO_3–N by Cd reduction on a Lachat Quikchem8000 (American Public Health Association, 1998).

Shallow ground water and riverine (in-stream) denitrification were determined on the Big Ditch watershed as part of this project and have been previously published (Mehnert et al., 2007; Royer et al., 2004; Schaller et al., 2004). A brief summary is given here; details are available in the publications cited. Mehnert et al. (2007) measured shallow ground water denitrification by monitoring 11 wells installed throughout the watershed. Isotopic ratios of N and O in the nitrate ion were used to suggest the extent of denitrification. Push-pull tests were conducted to determine in situ NO_3–N reduction rates. The software GFLOW was used to create a two-dimensional ground water model (Mehnert et al., 2007).

In-stream denitrification was determined on ditch sediments and associated aquatic plants using the chloramphenicol-amended acetylene inhibition procedure (Royer et al., 2004; Schaller et al., 2004). Measurements were made throughout the year within the stream system of the watershed.

A reliable technique for determining field denitrification was not available (Groffman et al., 2006); therefore, we used weather patterns and our knowledge of N budgets to make some general assumptions. Due to dry conditions in 2001, we assumed field denitrification was not an important watershed output. However, with several large precipitation events in April, May, and June of 2002, where rainfall exceeded infiltration rates and soils were saturated for several days, we believe conditions were conducive for field denitrification. We estimated field denitrification in 2002 by difference using the complete watershed N mass balance, assuming total inputs plus grain N derived from soil (explicit change in soil storage) equaled total outputs and solving for missing N.

Grain Nitrogen Derived from Soil

Estimates of maize fertilizer N recovery and soybean N_2 fixation were used to calculate a value for grain N derived from soil. Fertilizer N recovery in maize was determined by measuring the difference between fertilized and unfertilized above-ground plant N accumulation and dividing by the fertilizer N rate. Unfertilized maize N accumulation can also be used as a proxy for net soil N mineralization during the growing season (Gentry et al., 2001). Four plots of unfertilized maize were established in the alternate field adjacent to the soybean microplots during the maize phase of the rotation. Four plant samples from a 6.1-m row length (0.76-m row spacing) of unfertilized maize were harvested at physiological maturity, divided into three plant fractions (leaves and stalks; tassel, husk, and cob; and grain), and dried to a constant weight at 80°C for biomass determination (Gentry et al., 1998). Dried samples were ground through a 2-mm mesh and analyzed for total N using a combustion technique (Fisons NA 2000 N Analyzer). Above-ground plant N accumulation of fertilized maize was based on county estimates of grain N divided by N harvest index. Using the percent fertilizer recovery, we calculated maize grain N derived from fertilizer and assumed the remainder was from soil. For soybean, we calculated grain N derived from fixation and assumed the remainder was from soil.

Tile Drainage

Three agricultural drainage tiles along the Big Ditch were monitored during 2001–2002. These tiles cumulatively drained 25.5 ha, with the majority of the effective drainage area planted to soybean in 2001 and maize in 2002. The effective drainage area of each tile was determined by assuming the ratio of river discharge to precipitation for the watershed is the same for tiles, dividing annual tile volume by annual precipitation, solving for area, and averaging effective tile drainage area over the 2 yr. Tile discharge was gauged using a Sigma 900 MAX (Hach Co., Loveland, CO) area velocity sampler, and water samples were collected on a flow proportional basis using an automated water sampler (ISCO 2900). Water samples were analyzed for NO_3–N, NH_4–N, and total N as described previously. Tile water flow-weighted mean N concentrations and loads were determined to compare and contrast to riverine N.

Nitrogen Balance Calculations

We calculated simple field N balances for maize (fertilizer N minus grain N) and for soybean (N_2 fixation minus grain N). We calculated simple watershed N balances as inputs (deposition, fertilizer N, soybean N_2 fixation) minus outputs (maize and soybean grain N), comparing riverine N loads with these watershed balances. Finally, we calculated the overall watershed N mass balances as the inputs (deposition, fertilizer N, soybean N_2 fixation) minus the outputs (maize and soybean grain N, stream N load, in-stream and ground water denitrification, field denitrification).

Results and Discussion

Precipitation and Crop Yield

Weather patterns and annual precipitation in Champaign County varied greatly during 2001 and 2002; however, crop yields were similar in both years. The 2001 water year was particularly dry (78 cm measured in local rain gages and an average of 91 cm measured in the climate division), and crop yields (0% moisture) were 8.27 and 2.68 Mg ha^{-1} for maize and soybean, respectively. Although annual precipitation was low in 2001, rainfall occurred at timely intervals during the growing season that resulted in crop yields that were within 5% of the 1997–2000 averages. The 2002 water-year precipitation was 110 cm in local rain gages and 112 cm for the climate division, which was the third wettest water year since 1971. The county average maize yield declined by 7% to 7.7 Mg ha^{-1}, whereas soybean yield increased by 17% to 3.2 Mg ha^{-1}. The 2002 growing season began with a wet, cool April and May but became hot and dry during late June and early July, which negatively affected maize production; however, the soybean crop benefited from rainfall in mid-August. Overall, crop yields in Champaign County in 2001 and 2002 were similar to adjacent counties and were above the state average.

Maize Fertilizer Nitrogen Recovery and Soybean N_2 Fixation

Fertilizer N recovery values were 51 and 38% in 2001 and 2002, respectively (Table 1). Net soil N mineralization as indicated by unfertilized maize N accumulation was 77 and 90 kg ha^{-1} in 2001 and 2002, respectively. These estimates suggest that the drier conditions of 2001 limited soil N mineralization. Based on the difference in N accumulation of nodulated and non-nodulated soybean isolines, we determined N_2 fixation rates to be 77 and 60% of the total N accumulation in the above-ground biomass in 2001 and 2002, respectively (Table 1). By multiplying fixation rate and total above-ground N accumulation (grain N divided by N harvest index), we estimated soybean N_2 fixation in the watershed to be 163 and 150 kg N ha^{-1}

in 2001 and 2002, respectively. Nitrogen accumulation of the non-nodulating soybean isoline can also be used as an indication of net soil N mineralization and was less for 2001 than 2002 (32 and 63 kg ha^{-1}, respectively). Compared with unfertilized maize, N accumulation by the non-nodulating soybean was less in both years (Table 1). This may in part be due to differences in growing period, root architecture, and N absorption patterns of maize and soybean; however, maize has been shown to stimulate soil N mineralization by as much as 50% (Sanchez et al., 2002). Overall, the drier growing season of 2001 created conditions that increased maize N fertilizer recovery and soybean N_2 fixation.

Field Nitrogen Balance

For simple field N balances, we used only fertilizer N or N_2 fixation for inputs and grain N for output; we did not consider atmospheric N deposition here. Subtracting maize grain N from the fertilizer N rate of 184 kg N ha^{-1}, we found field N balances to be positive, indicating a net gain of 65 and 73 kg N ha^{-1} for maize fields in 2001 and 2002, respectively. Maize yields would need to be >13 Mg ha^{-1} (assuming 1.44% grain N) to remove more N than was supplied at this fertilization rate. For soybean, field N balances were negative for both years because N from fixation was less than grain N output. Subtracting soybean grain N from plant N_2 fixation (grain N divided by N harvest index multiplied by N_2 fixation rate), we found net removal of N in soybean fields to be 7 and 51 kg ha^{-1} in 2001 and 2002, respectively. Although a soybean crop is often given a N credit when preceding maize (Gentry et al., 2001), studies report a negative balance in soybean fields (Heichel and Barnes, 1984; Zapata et al., 1987).

Net Nitrogen Input

Simple watershed N balances have been used to compare riverine N loads with net nitrogen input (NNI); however, basin size, intensity of agricultural production, and extent of artificial drainage influence the relationship. For example, Howarth et al. (1996) found riverine N load to be about 22% of NNI for the entire Mississippi River basin. David and Gentry (2000) estimated the combined riverine N load for the major rivers of Illinois to be 51% of the NNI. In watersheds within Illinois, McIsaac and Hu (2004) found large differences in riverine N load to NNI, based on the presence or absence of tile drainage. Riverine N load represented 25 to 37% of NNI in non-tile drained watersheds, whereas riverine N load was 100% of NNI for watersheds containing extensive tile drainage. These results suggest that the value for NNI cannot account for both N leaching and denitrification in tile-drained regions.

For the Big Ditch watershed, N fertilizer provided 83 and 85 kg N ha^{-1} in 2001 and 2002, respectively (Table 2). Although we used the same fertilizer N rate for both years of the study, there was a slight increase in maize acres in the watershed in 2002. Soybean N_2 fixation contributed 71 and 64 kg N ha^{-1} to the entire watershed in 2001 and 2002, respectively (Table 2). By summing atmospheric N deposition, fertilizer N, and N_2 fixation and subtracting total grain N, we calculated NNI for the Big Ditch watershed to be 30 and 17 kg N ha^{-1} in 2001 and 2002. For this relatively small and extensively drained agricultural watershed, riverine N loads represent 70 and 294% of NNI for the 2 yr. In accordance with McIsaac and Hu (2004), our calculation of NNI could not account for both riverine N flux and denitrification during the wet year of 2002.

Riverine and Tile Nitrogen Load

Discharge and N loads exiting the Big Ditch watershed varied greatly between the two water years. Total discharge was 19 and 34 million m^3 (19 and 34 cm) for the 2001 and 2002 water years, respectively, which represented 24 and 31% of the annual precipitation. During the 1994–2003 water years, annual discharge for this stream ranged from 8 to 38 cm, with an average value of 26.5 cm (Royer et al., 2006).

Based on the entire watershed area, riverine total N loads were 21 and 50 kg N ha^{-1} for the 2001 and 2002 water years, respectively (>90% of the total N was NO_3–N). Flow-weighted mean NO_3–N concentrations for the Big Ditch were 10.2 and 14.8 mg N L^{-1} for the 2001 and 2002 water years, respectively. Although precipitation was low in 2001, there were two large discharge events in February (Fig. 1). These precipitation events generated overland runoff as indicated by the dilution of riverine NO_3–N concentrations during peak discharge, producing relatively small total N loads. During 2002, numerous flow events occurred, and riverine NO_3–N concentrations tended to increase through May (Fig. 1). With 50% of the fertilizer N applied in the fall in this watershed, we speculate that this was an important source of river and tile NO_3–N during the wet spring.

In 2002, N load and annual flow-weighted mean NO_3–N concentrations in the Big Ditch were the highest recorded during the 10 yr from 1994 to 2003 (Royer et al., 2006). During this period, NO_3–N flux was highly correlated with water yield ($r^2 = 0.72$), but two years were outliers: in 1994 the observed NO_3–N flux was 12 kg N ha^{-1} less than the trend line, and in 2002 the observed NO_3–N flux was 13 kg N ha^{-1} greater. Precipitation throughout the region in 1993 was the greatest on record, as were river flows, and this appeared to flush NO_3–N out of the soil and ground water so that NO_3–N concentrations tended to be lower in 1994. Precipitation and flows during 1999–2001 were below average, allowing accumulation of NO_3–N in soil and shallow ground water, which appeared to have been mobilized during the high flows of 2002.

Total N load per unit area for three tiles located in the Big Ditch watershed were similar to river loads for both years. Based on the total effective drainage area for all three tiles, cumulative N loads were 22.7 and 59.9 kg ha^{-1}, and flow-weighted mean NO_3–N concentrations were 11.7 and 19.2 mg L^{-1} for the 2001 and 2002 water years, respectively. Similar to the Big Ditch, tile NO_3–N concentration decreased during the large discharge events in February of 2001 and tended to increase with discharge for each successive flow event in 2002. After tile flow cessation, river NO_3–N concentration quickly decreased below detection limits for both years. This similar pattern of river and tile NO_3–N suggests that tiles were the major source of riverine N.

Denitrification (In-Stream and Shallow Ground Water)

In-stream denitrification was estimated to be no more than 1 kg N ha^{-1} yr^{-1} (Table 2). Although in-stream denitrification

Table 1. Maize and soybean crop parameters used to calculate nitrogen balances in the Big Ditch watershed during 2001 and 2002.

Maize crop	2001	2002
	— kg N ha^{-1} —	
Fertilizer N rate	184	184
Grain yield, kg ha^{-1}	8270	7730
Grain N	119	111
Grain N derived from soil	59	69
Fertilized plant N	170	159
Unfertilized plant N	77	90
Fertilizer N recovery	51	38
Soybean crop		
Fertilizer N rate	0	0
Grain yield, kg ha^{-1}	2680	3160
Grain N	172	202
Grain N derived from soil	36	74
Nodulated plant N	136	156
Non-nodulated plant N	32	63
N_2 fixation rate	77	60

rates have been shown to be substantial during the summer months in east-central Illinois (Royer et al., 2004; Opdyke et al., 2006), tile drainage has generally ceased at this time (David et al., 1997). In contrast, the majority of river NO_3–N was exported when tile drainage was occurring, stream water residence time was short, and temperatures were cool (Royer et al., 2004). These factors combined to make in-stream denitrification a negligible N output from the Big Ditch watershed.

Denitrification in shallow ground water was greater than in-stream denitrification; however, it was a minor output from the watershed. Mehnert et al. (2007) found that 1.8 and 5.7 kg ha^{-1} of N were denitrified from ground water in this watershed during the 2001 and 2002 water years, respectively, representing 6 and 34% of NNI. These estimates were considered minimum values because the hydrologic model only accounted for steady state ground water flow and ignored transient flow events, such as flow from precipitation events. It is likely that the existence of extensive tile drainage decreases the potential for shallow ground water denitrification in this watershed.

Nitrogen Mass Balance

The overall annual mass balances of the Big Ditch watershed are shown in Fig. 2 and 3. Assuming field denitrification was <1 kg ha^{-1} in 2001, N inputs (158 kg N ha^{-1}) were greater than outputs (152 kg N ha^{-1}), indicating a positive N mass balance of 6 kg N ha^{-1}. Although we calculated a value of −44 kg N ha^{-1} for grain N derived from soil, the positive N balance indicates that net soil N depletion did not occur in 2001. It is likely that dry years with moderate grain yields and small leaching and dentrification losses create surplus N, allowing carryover of N to the subsequent year (David et al., 1997).

Gaseous N losses from soils are considered the most difficult measurements to conduct on a large spatial scale and were not directly measured in this study. Although the 2001 watershed N mass balance suggests that as much as 6 kg N ha^{-1} could be lost from soils via processes such as dentrification and nitrification, soils were not inundated when temperatures were favorable for dentrification. Nitrification of ammoniacal fertilizers (especially at fertilization rates greater than sufficient) has been shown to produce gaseous N loss; however, fertilizer N rates in the Big Ditch watershed were not considered excessive, which would minimize the importance of this N output in our mass balance calculations (McSwiney and Robertson, 2005). In general, the extensive tile drainage that exists throughout east-central Illinois is thought to decrease the occurrence of field denitrification (McIsaac and Hu, 2004). Therefore, in the drier year of 2001, we believe field denitrification was not likely an important watershed output.

Table 2. Cropland area, inputs, outputs, and explicit change in soil storage for the Big Ditch watershed in 2001 and 2002.

	2001	2002
Cropland area		
Maize, ha	4547	4658
Soybean, ha	4381	4269
Inputs	—kg N ha^{-1}—	
Deposition	4	5
N fertilizer	83	85
Soybean N fixation	71	64
Total	158	154
Outputs		
Maize grain N	54	51
Soybean grain N	75	86
Big Ditch total N load	21	50
In-stream denitrification	<1	1
Ground water denitrification	2	6
Field denitrification	<1	27
Total	152	221
Explicit change in soil storage		
Maize grain N derived from soil	−26	−32
Soybean grain N derived from soil	−18	−35
Total	−44	−67

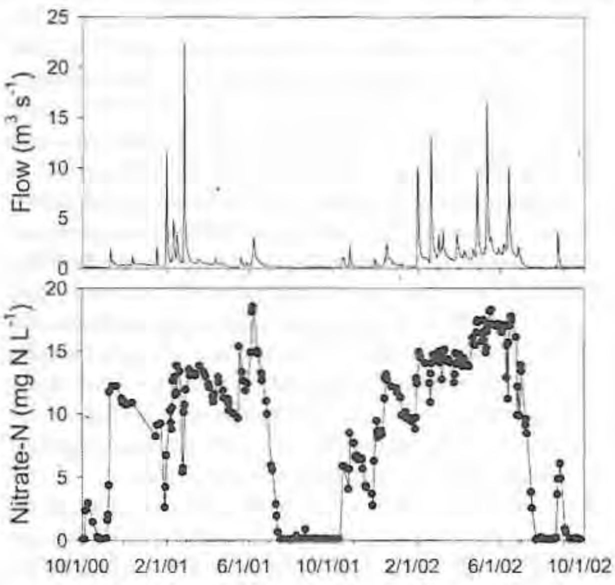

Fig. 1. Big Ditch discharge and NO_3–N concentrations.

In 2002, with large N leaching losses and likely substantial field denitrification, total inputs (154 kg N ha^{-1}) were less than all measured outputs (194 kg N ha^{-1}) (Table 2). This negative balance suggests a reduction in stored soil N in 2002. Here we treat explicit change in soil storage (−67 kg ha^{-1}) as an input, offsetting the balance shortfall and solving for field dentrification. By assuming that N input plus the absolute value for explicit change in soil storage is

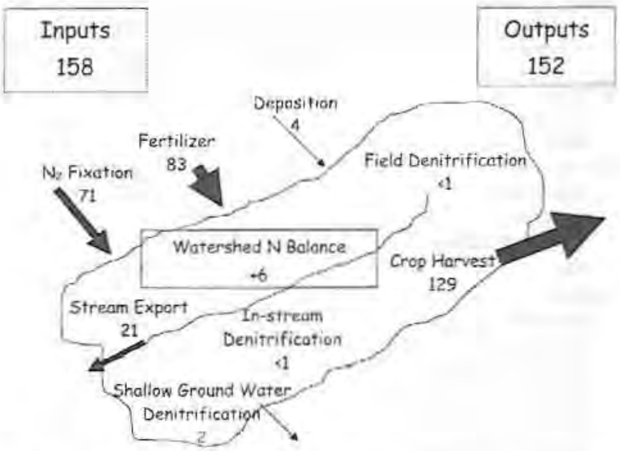

Fig. 2. Nitrogen mass balance for the Big Ditch watershed in 2001, showing summed inputs and outputs as well all major measured and estimated fluxes. All units are kg N ha^{-1} yr^{-1}, and arrows are proportional to fluxes.

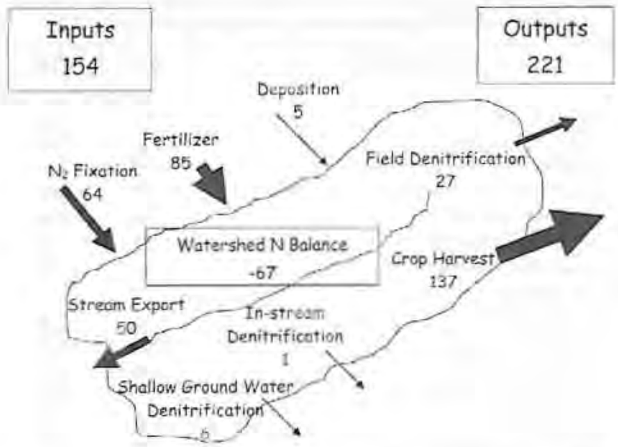

Fig. 3. Nitrogen mass balance for the Big Ditch watershed in 2002, showing summed inputs and outputs as well all major measured and estimated fluxes. All units are kg N ha^{-1} yr^{-1}, and arrows are proportional to fluxes.

equal to total N outputs (due to the extensive flushing of N from soil and shallow ground water), we estimated that 27 kg N ha^{-1} was lost from the watershed via field denitrification in 2002 (Table 2). Although few studies have quantified field denitrification, a study in east-central Illinois (Torbert et al., 1992, 1993) found that when Drummer soil was artificially flooded for more than five consecutive days, nearly 50% of the fertilizer N applied was lost via denitrification. As indicated by the numerous river and tile discharge events during May and June of 2002, frequent precipitation at that time created saturated soil conditions in the Big Ditch watershed for extended periods at temperatures favorable for dentrification. In extensively tile-drained regions, only small amounts of N enter shallow ground water because tile and stream networks quickly transport N downstream and out of the watershed (Mehnert et al., 2007; Royer et al., 2006). Therefore, we believe that denitrification in the upper soils was likely the most important source of gaseous N loss during the wet year of 2002.

David et al. (2009) compared five models that simulate the N cycle in agricultural systems and predicted denitrification (SWAT, DAYCENT, DRAINMOD-N II, EPIC, and DNDC) for the Embarras River watershed in Champaign County, directly south of the Big Ditch watershed. The Embarras River watershed has similar soils, cropping patterns, fertilizer N use, and riverine N exports as in the Big Ditch watershed (Royer et al., 2006). The models predicted an average denitrification flux for 2002 of 13.5 kg N ha^{-1}. For the agronomic-based models SWAT, DRAINMOD-N II, and EPIC, field denitrification rates were estimated to be 22, 24, and 14 kg N ha^{-1} yr^{-1}, respectively, which is similar to our estimate of 27 kg N ha^{-1} yr^{-1}; the two biogeochemistry models DAYCENT and DNDC had estimates of 3.5 to 4.2 kg N ha^{-1} yr^{-1}, respectively. Using our estimate for field denitrification, we find a gross loss of N via leaching and denitrification (field, shallow ground water, and in-stream) of 84 kg N ha^{-1} for the Big Ditch watershed in 2002.

Overall, our watershed mass balance analysis indicates that N fertilizer is the largest input, that grain N is the largest output, and that total outputs are greater than total inputs. During the past 20 yr, US fertilizer N sales have remained relatively constant, whereas crop yields and N harvested (especially maize since the introduction of GMO traits) have increased (USEPA, 2007). Given this perceived increase in maize N utilization efficiency, it would be expected that riverine N loads in the Mississippi River watershed would be declining. Surprisingly, riverine loads in tile-drained regions have not declined much, if at all, during this period (USEPA, 2007). In our analysis of the Big Ditch watershed, the N mass balance could not be closed without considering the explicit change in soil storage. Therefore, our results suggest that a maize/soybean rotation depletes soil N in this extensively tile-drained watershed, especially during an extremely wet year. In addition, tile drainage losses can be substantial (>50 kg N ha^{-1} yr^{-1}), even with a favorable crop N balance, as indicated by a NNI of 17 kg ha^{-1} in 2002. Finally, there is little doubt that NO_3–N leaching, largely mediated through tile drainage networks, is the major source of N in surface waters in east-central Illinois, contributing to local water quality problems and nutrient loading in the Gulf of Mexico.

Conclusions

Our comparison of N cycling in the Big Ditch watershed was conducted during 2 yr of differing N leaching patterns driven by precipitation. The watershed N balance calculations indicate that N inputs were greater than outputs (+6 kg ha^{-1}) in the drier year (2001) but were much less than outputs (−67 kg ha^{-1}) in the wetter year (2002), indicating soil N depletion. In years with modest leaching losses and minimal denitrification, N may accumulate and carry over to the next year, thus partly offsetting net depletion of soil N. Our analysis suggests that soil N depletion can occur in maize/soybean rotations in years with above-average precipitation or extremely wet spring periods. With current production trends of higher grain yields with flat or even declining fertilizer N rates, these data suggest the likelihood that soil N depletion may be exacerbated.

Acknowledgments

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TECHNOLOGIES TO MEET NUMERIC NUTRIENT CRITERIA AT FLORIDA'S DOMESTIC WATER RECLAMATION FACILITIES

November 18, 2009

Table of Contents

TECHNOLOGIES TO MEET NUMERIC NUTRIENT CRITERIA AT FLORIDA'S DOMESTIC WATER RECLAMATION FACILITIES

EXECUTIVE SUMMARY

The consent decree between the US Environmental Protection Agency (EPA) and a coalition of environmental groups over the need to set numeric nutrient criteria (NNC) for surface waters in the State of Florida will have severe consequences for Florida's wastewater utilities and the customers they serve. If the expected NNC to be proposed by EPA are implemented in the state's water quality regulations, all Florida wastewater facilities that discharge reclaimed water to rivers, streams, lakes and estuaries will likely be required to meet new, very low limits on the discharge of total nitrogen (TN) and total phosphorus (TP). These new limits are anticipated to require a reduction in the current discharge of TN by 40 to 80% and TP by 90 to 95%.

Should the proposed numeric nutrient criteria become incorporated in the discharge permits for these treatment plants, Florida utilities will spend an estimated $24.4 to $50.7 billion in capital costs for additional treatment facilities, and will incur an estimated $0.4 to $1.3 billion dollars per year more in increased operating costs. This corresponds to tens of billions of dollars in costs passed down to approximately three quarters of the individual citizens and businesses in the State of Florida. In addition, the treatment technologies needed to achieve these limits will potentially use almost 5 percent of the total electric power generating capacity in the State of Florida and contribute over 17.4 million tons of carbon dioxide (CO_2) emissions per year.

According to the Florida Department of Environmental Protection (FDEP), there are approximately 2,100 permitted domestic wastewater facilities in the State with a total treatment capacity of about 2.4 billion gallons per day, approximately half of which discharge effluent to surface waters. FDEP reported in 2005 that approximately 64% of Florida's population is served by central sewer systems that feed these treatment facilities. Florida's total population is in excess of 18 million residents according to the US Census Bureau. Based on these figures, approximately 12 million citizens of Florida are served by permitted wastewater treatment facilities that will be impacted by the proposed NNC.

Required facility upgrades to treat the 2.4 billion gallons of wastewater per day to the new standards would result in a cost of approximately $1.6 to $3.3 billion dollars per year over the next 30 years in addition to current treatment costs. Considering the interest paid on bonds required to fund these improvements, the State's wastewater utilities are facing a $47.6 to $98.7 billion dollar price tag to meet the proposed nutrient criteria over the next 30 years. These figures do not account for the corresponding increase in operating costs. We estimate that annual user fees for customers served by the impacted facilities will increase by an average of $673 to $726 per year in total. Table 1 summarizes the potential cost to Florida and the average individual household.

This paper addresses the anticipated costs for Florida utilities to be in compliance with EPA-promulgated numeric nutrient criteria; the treatment technologies available to achieve the proposed criteria; and lastly, the additional $673 to $726 in yearly costs to customers should these regulations be adopted.

Table 1 Summary of Estimated Project Costs and the Estimated Average Increase in Annual Sewer Rates for the State of Florida to Implement Numeric Nutrient Criteria.

	Project Cost[1]	Annual Debt Service[2]	Increase in Annual Operating Costs[3]	Yearly Sewer Rate Increase per Customer[4]
Florida Facilities with NPDES Permits[5]	$24,400,000,000	$1,600,000,000	$433,000,000	$673
All Florida Facilities[5]	$50,700,000,000	$3,300,000,000	$1,330,000,000	$726

Notes:

1. Project costs include estimated construction costs plus contingencies, administrative, legal, engineering and financing costs. Estimates also assume all private facilities must be upgraded to provide advanced wastewater treatment, microfiltration, and reverse osmosis with concentrate drying with landfill disposal. Assumes public facilities with deep wells will construct new deep wells for concentrate disposal. All other public facilities assumed to construct facilities for concentrate drying.
2. Annual debt service based on 30-year amortization schedule and 5% interest.
3. Assumes current flow is 50% of design flow, and that this is representative of "actual" usage of the plant. Capital upgrades will be required for the entire design flow, while operating costs are based on actual usage.
4. Average cost for public and private systems. Customer size estimated at 2.1 persons per household based on US Census information.
5. The range of project costs was based on two scenarios. The lower project cost assumes that only surface water dischargers (e.g. assumed to be plants with NPDES permits) would be required to comply with the proposed nutrient criteria. The higher project cost assumes that all Florida plants will be required to comply with the proposed nutrient criteria.

EXISTING AND PROPOSED NUTRIENT STANDARDS

Florida's water reclamation plants already operate within a complex system of water quality regulations that are among the most stringent in the world. Currently, the highest level of treatment provided by Florida wastewater treatment plants is Advanced Wastewater Treatment (AWT). In Florida, the term "AWT" is often used to refer to wastewater treatment that produces a reclaimed water achieving standards set out in section 403.086(4), Florida Statutes, i.e. reclaimed water containing no more than 5 mg/ L of carbonaceous five-day biochemical oxygen demand ($cBOD_5$), 5 mg/L total suspended solids (TSS), 3 mg/L TN, and 1 mg/L TP.

According to the proposed consent decree between EPA and a coalition of environmental groups, EPA is to propose NNC for fresh waters in Florida in January 2010 and marine waters by January 2011. NNC proposed by FDEP in June 2008 would drop the TN limit from 3.0 mg/L to 0.82 – 1.73 mg/L and TP limits from 1.0 mg/L down to 0.069 – 0.415 mg/L. Based on review comments from EPA on the FDEP NNC, the actual values for EPA's NNC are expected to be lower than the NNC proposed by the FDEP. While these proposed changes might seem small, the new NNC are at (for TP) or below (for TN) the concentrations achievable with proven nutrient removal technologies now in full-scale use at municipal wastewater plants. Table 2 allows a comparison of the current standards for secondary and AWT standards with the proposed NNC standards.

The very low concentration criteria on TN and TP being proposed can be met, although at significant cost, by adding additional treatment processes at the end of a Florida advanced wastewater treatment (AWT) plant. AWT as currently practiced will not provide consistent compliance with the proposed NNC, and additional technologies must be employed.

Table 2 Comparison of Maximum Pollutant Concentrations Allowed by Water Reclamation Standards

Pollutant	Secondary Limits[1]	AWT Limits[2]	Proposed Numeric Nutrient Limits[3]
$cBOD_5$, mg/L	20-30	5	-
TSS, mg/L	20-30	5	-
TN, mg/L	No limit	3	0.82 – 1.73
TP, mg/L	No limit	1	0.069 – 0.415

Notes:

1. Nationwide technology based standards required by the Clean Water Act for all wastewater treatment plants.
2. Florida standards required by state law for discharge to specific nutrient sensitive water bodies.
3. Proposed in-stream water quality standards that would become end-of-pipe limits unless a facility obtains a mixing zone or site-specific alternative criteria.

CURRENT NUTRIENT REMOVAL TECHNOLOGY

Today more than 65 Florida plants with a total treatment capacity of over 500 million gallons per day (mgd) use various AWT technologies to produce high quality reclaimed water. However, these plants represent only a relatively small fraction of the plants in Florida. In order to achieve current AWT water quality limits, a plant generally must remove about 96% to 98% of TN and 85 to 88% of TP from raw sewage. At nearly all AWT plants in Florida, reclaimed water meeting AWT limits is achieved using a combination of a biological nutrient removal (BNR) treatment technology plus some form of filtration. BNR processes grow naturally occurring bacteria to remove the oxygen demanding organic pollutants, nitrogen, and phosphorus from raw sewage. Most of the nitrogen is released to the atmosphere as nitrogen gas while the phosphorus is removed with biosolids wasted from the plant.

The TN in reclaimed water consists of inorganic nitrogen (ammonia and nitrates) and organic nitrogen. The ammonia and nitrates can be removed down to low levels by conventional BNR processes – nitrification and denitrification with supplemental carbon (e.g. methanol) if necessary. The presence in reclaimed water of soluble nitrogen and phosphorus compounds that are not biodegradable (also known as refractory compounds) ultimately sets the lowest concentrations possible at treatment plants that rely on biological treatment methods. Researchers call the refractory dissolved organic nitrogen (RDON). RDON is present in many raw water supplies and thus is present in the influent to water reclamation facilities. RDON is also a byproduct of biological treatment and will be found in reclaimed water even if not found in raw sewage.

The effluent TN and TP concentrations reported by Florida AWT plants are generally consistent with the lowest effluent concentrations of TN and TP typically reported by researchers as achievable with current BNR technologies. In other words, Florida AWT plants are already removing nutrients as well or better than most BNR plants.

ADDITIONAL TECHNOLOGIES TO MEET PROPOSED NUMERIC NUTRIENT CRITERIA

Water treatment technologies do exist to produce reclaimed water with any desired water quality such as the ultra pure water required for electronics manufacturing and high-pressure boiler feed water. Since most technologies that can achieve these limits are typically used for applications other than removing nutrients from reclaimed water, little data exists on their performance in removing nitrogen and phosphorus to meet NNC.

The additional treatment technologies that might be used to meet numeric nutrient criteria include high-pressure membranes (reverse osmosis and nanofiltration), adsorption (activated carbon and reactive filtration), oxidation (ozone, UV, peroxide), chemical coagulation and precipitation, and ion exchange.

Of the aforementioned additional treatment technologies, reverse osmosis (RO) is a leading candidate technology when considering very low nutrient limits because of its ability remove nearly all constituents in the water to very low concentrations. In simple terms, RO is a selectively permeable barrier that rejects nearly all contaminants larger than a specific size, while letting water molecules through. A number of large water reclamation plants currently use RO to produce very high quality reclaimed water. These include the West Basin plant in El Segundo, California (22.5 mgd), the Scottsdale, Arizona Water Campus (14 mgd), the GWRS (70 mgd) plant in Orange County California, and four NEWater plants in Singapore (Bedok, 8.4 mgd; Kranji, 10.6 mgd; Seletar, 5.0 mgd; and Ulu Pandan, 39.1 mgd) among others. All of these facilities have successfully used membrane technologies to produce exceptionally high quality water for other purposes than to remove nutrients. Specifically, the West Basin plant in El Segundo and the GWRS plant use MF/RO treated reclaimed water to create saltwater intrusion barriers and for indirect potable reuse. The Scottsdale plant treats to drinking water standards and then injects MF/RO treated water into an aquifer to supplement drinking water supplies, and the four NEWater plants in Singapore use MF/RO or UF/RO to create high quality industrial water supplies for electronics manufacturing and to provide supplemental water to a reservoir used as a drinking water supply.

Figure 1 provides a schematic representation of the sequence of treatment zones in the typical Florida water reclamation plant designed to meet AWT limits using BNR along with the selected additional technology of reverse osmosis membranes and associated concentrate disposal.

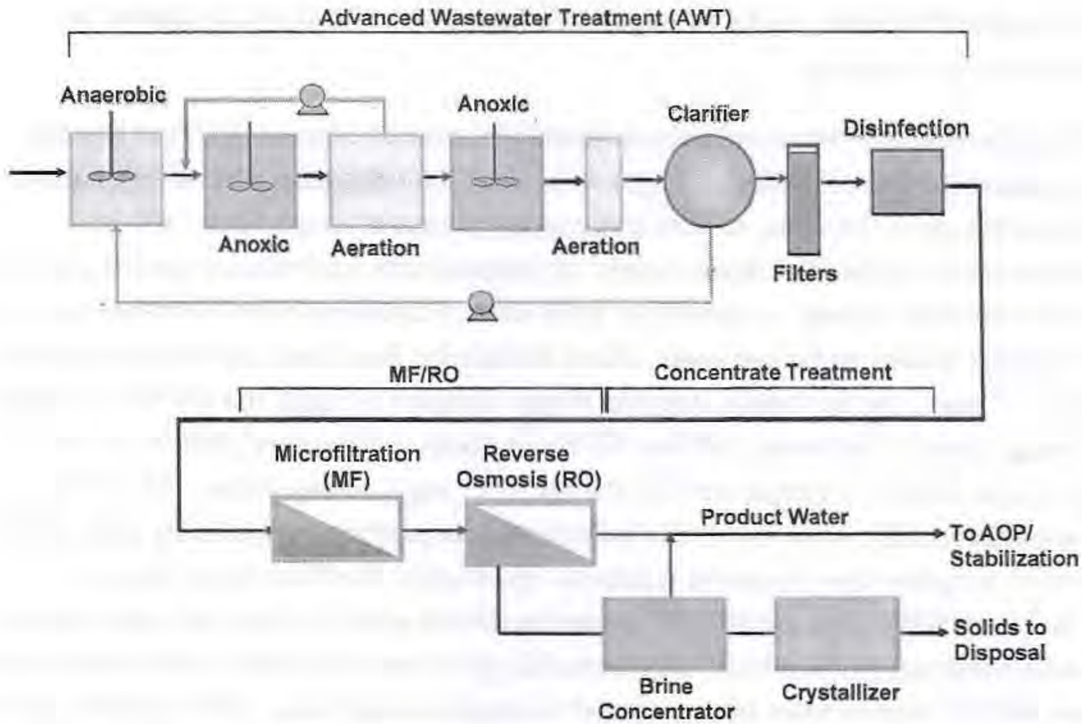

Figure 1. Schematic illustration of the type of processes likely required for Florida wastewater plants to comply with the proposed numeric nutrient criteria.

CHALLENGES ASSOCIATED WITH REVERSE OSMOSIS SYSTEMS

RO systems create several significant challenges. First, since membranes simply separate various constituents from water, RO creates a "concentrate" stream, typically 15 to 20% of the feed water, that contains concentrated levels of the rejected constituents. For example, if the mineral calcium was in the untreated water at 100 mg/L, it would be treated to nearly 0 mg/L in the treated stream with a concentration of about 500 mg/L in the concentrate stream that then needs to be disposed of. Finding economical methods that will comply with regulatory and public perception requirements for reuse or dispose of concentrate can be a challenge. In areas with suitable geology, deep well disposal is an option. Concentrate cannot be recycled upstream of an RO system as the salt concentration very quickly builds to intolerable levels. Concentrate can also be evaporated, dried, and landfilled; however, both the capital and operating costs of drying concentrate are on the order of $3.00 per 1000 gallons treated more expensive than other disposal methods.

Another significant drawback to RO treatment is the high power cost required to operate an RO system. Adding RO to the typical water reclamation plant will increase power consumption for an already energy-intensive system by at least 90%. Power requirements for RO systems treating reclaimed water range from about 1,400 to 2,400 kWh per mgal treated, compared with the average power consumption for existing water reclamation facilities of about 1,500 kWh/mgal. RO membranes also tend to foul quickly in reclaimed water applications, requiring pretreatment of feed water and regular chemical cleaning of the membranes, both of which increase the cost and complexity of building and operating an RO system. At present most reclaimed water RO systems use microfiltration (MF) for pretreatment.

Finally, RO treatment still needs to be preceded by a BNR process. RO alone will not remove ammonia and nitrate to the degree required due to their similar size and molecular characteristics to water. A typical RO membrane will reject about 90% of the nitrate and 95% of the ammonia in the feedwater. While phosphates are better removed by RO membranes (rejection $\geq$ 99.8%), too much phosphorus going to an RO system will also foul membranes as a result of the precipitation of insoluble phosphate compounds.

COSTS TO MEET THE NUMERIC NUTRIENT CRITERIA

A Specific Case Study

To illustrate the costs to implement additional treatment to meet the proposed numerical nutrient criteria, consider an existing 10 mgd Florida water reclamation plant that meets advanced secondary water quality standards for public access reuse. To meet the numeric nutrient criteria, the biological portion of this plant must be upgraded to meet AWT limits, and then additional tertiary physical-chemical treatment processes must be added. For this example, assume that conventional BNR and MF/RO processes will be used.

Based on a study of 50 BNR upgrade projects, the expected cost to upgrade a 10 mgd plant from secondary to AWT standards is on average about \$82 million. Based on bid-data available from eleven wastewater treatment plants using MF-RO, adding MF-RO treatment to an existing water reclamation plant is estimated to cost between \$54 million to \$81 million depending on the concentrate disposal method. These costs could be significantly higher at existing sites because of site constraints. The total capital cost to upgrade a 10-mgd water reclamation facility not already meeting AWT limits could be in the range of \$140 – \$160 million, and operating costs would increase by at least \$1.00 per thousand gallons treated (about \$3.6 million per year), not including concentrate disposal costs which can range from an additional \$1.10 per thousand gallons for a deep well up to \$3.00 per thousand gallons for a brine concentrator system (these numbers have been normalized to reflect design flow and not concentrate flow). These additional operating costs are in addition to the estimated \$1.00 to \$1.50 per thousand gallons being treated already being spent for typical Florida treatment facilities producing water for reuse. Table 3 summarizes the incremental capital costs required to meet the proposed NNC.

Further, meeting low yearly average nutrient limits requires highly consistent performance by the treatment process, which will likely result in significant conservatism in the sizing of treatment facilities, provision of redundant systems, and other provisions to increase reliability that ultimately add additional costs. These additional costs are not accounted for in the estimates provided here.

Table 3 Summary of Incremental Capital Costs ($M) for BNR and MF/RO Treatment to Upgrade a 10-mgd Water Reclamation Facility for Three Different Concentrate Disposal Options.

	Deep Well (10,000 ft)[3]	Deep Well (2,500 ft)[3]	Zero Liquid Discharge[4]
BNR Upgrade	80	80	80
MF-RO Systems[1]	50	50	50
Concentrate Disposal[2]	11	4	31
Total	141	134	161

Notes:
1. Assumed recoveries: MF = 95%; RO = 85%.
2. Concentrate disposal costs are estimated from a combination of bid-data, vendor quotes, and a report published by the AWWA Membrane Residuals Management Subcommittee.
3. Excluding any pretreatment and standby disposal system.
4. With a brine concentrator and brine crystallizer; excluding cost for solids disposal; brine concentrator recovery assumed = 95%.

The Big Picture

In order to further quantify the effect of the proposed NNC on the State, the case study variables described above were applied to all publicly owned wastewater treatment plants documented by FDEP. The plants were broken down into plants that already have AWT and ones that do not. AWT technology upgrades were assumed and projected to cost $8.20 per design gallon to upgrade. Subsequently, all plants would be required to be upgraded by the addition of an advanced tertiary technology following AWT. Reverse osmosis was assumed at an average cost of $5.00 per gallon per day of design treatment capacity. An additional cost of $1.10 per gallon of design treatment capacity was applied for concentrate disposal for facilities that have an existing deep injection well, as they were assumed to be able to construct an additional deep injection well for RO concentrate disposal. All other facilities included a $3.10 per gallon capital cost for a brine concentrator, as these facilities were assumed to not have access to a deep injection well. Lastly, the estimated increase in annual operating costs was calculated as follows: $1.00 per 1,000-gallons treated for AWT and MF/RO treatment, and $3.00 per 1,000 gallons for facilities that must treat RO concentrate with a brine concentrator. Applying the "dollar" per gallon unit capital costs to the FDEP database provided a lump sum capital expenditure for the state of $24.4 to $50.7 billion dollars. This cost was converted to an annual debt payment over an assumed 30 years with 5% interest, which amounts \$1.6 to \$3.3 billion dollars plus an additional \$0.4 to \$1.3 billion dollars per year in increased operating costs.

Much of the increase in operating costs can be attributed to power consumption. We anticipate that the addition of these facilities will increase overall power demand by almost 26 million megawatt-hours/year. The estimated increase in connected electrical load requires about 5 percent of the total power generation capacity in Florida. Moreover, based on data published by the Department of Energy on CO_2 emissions by power plants, the additional power consumption will increase CO_2 emissions in Florida by over 17.4 million tons per year.

These costs would be passed along to customers through increased wastewater rates. To estimate the increase in rates, facility upgrade costs and population information were used to calculate the average monthly cost per household. On average, rates would increase by an average of \$673 to \$726 per year per household.

Sample of Utilities

A sampling of utilities was conducted to support the general effort of analyzing the cost impacts to all of the Florida utilities described in the "Big Picture" approach. These utilities have based their estimated capital costs on various technical alternatives to meet the NNC rule that range from the addition of RO, to implementing 100% reuse via infrastructure improvements, to deep well injection of effluent for aquifer recharge. The utilities sampled have wastewater treatment facilities that range in size from 6 to 42 mgd, and are located in various areas of the state. Each utility reported their present average monthly residential customer wastewater charge and a preliminary estimate of the anticipated increase in rates to meet the proposed NNC. These percentages were translated to approximate rate increases. Table 4 summarizes the results of this analysis. The average rate increase reported across these 8 utilities (\$618/yr) compares well with the independent conceptual statewide estimate prepared by the independent engineer (\$673/month to \$726/month).

Table 4 Summary of Estimated Capital Costs and Increases in Sewer Rates for Eight Florida Utilities to Construct Facilities to Meet Proposed Numeric Nutrient Criteria.		
	Capital Cost	**Yearly Sewer Rate Increase per Household**
STATE OF FLORIDA[1]	*$24,400,000,000-$50,700,000,000*[2]	*$673-$726*
Bay County	$42,000,000	$684.44
Broward County	$425,000,000	$793.32
Destin Water	$34,000,000	$581.16
Escambia County	$275,000,000	$590.88
Hollywood	$370,000,000	$995.88
Jacksonville	$2,000,000,000	$815.04
Cross City[3]	$5,800,000	$336.48
South Walton[3]	$16,000,000	$147.00

Notes:

1. Estimated average costs for the State of Florida including annual O&M expenses and are shown for comparative purposes.
2. The low end of the range provides the probable opinion of cost assuming only plants with surface water discharges will be required to meet numeric nutrient criteria while the high end of the range assume that all plants will need to meet numeric nutrient criteria.
3. Assumes 2.5 persons per connection and 150 gpcd.

CONCLUSIONS

Implementation of the proposed numeric nutrient criteria will initiate widespread and dramatic changes in the way Florida wastewater utilities treat and reuse reclaimed water. The necessary changes will likely require decades to fully implement. The major impacts expected for Florida consumers include the following:

1. Wastewater utility rates can be expected to at least double for those utilities required to meet the proposed NNC criteria.

2. The total municipal wastewater treatment cost to the people of the State of Florida will range from $47.6 to $98.7 billion dollars over the next 30 years. The actual cost within this anticipated range will depend upon how many utilities would be required -- besides those discharging *directly* to surface water streams -- to meet these criteria.

3. Energy requirements for treatment could increase by almost 26 million megawatt-hours/year.

4. Emission of CO_2 and other greenhouse gases could increase by over 17.4 million tons per year of CO_2.

FLORIDA DEPARTMENT OF ENVIRONMENTAL PROTECTION

Division of Water Resource Management, Bureau of Watershed Management

NORTHEAST DISTRICT • LOWER ST. JOHNS BASIN

TMDL Report

Total Maximum Daily Load for Nutrients for the Lower St. Johns River

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Table of Contents

given year in the freshwater zone. In addition, reductions were based on meeting the target within all five WBIDs in the freshwater zone. As such, the "worst-case" WBID controlled the amount of reduction needed. Finally, point source flows and loads used in the WLA for the freshwater zone were based on existing flows and loads with an allowance for growth rather than assuming permitted limits. This approach provides an implicit MOS because it is extremely unlikely that all of the point sources would simultaneously discharge at their full WLA.

Conservative assumptions were also part of the development of the TMDL for the oligohaline/ mesohaline portion of the river. As in the freshwater zone, four different years were simulated. However, in this case, the worst-case year (1999) was used to establish necessary nitrogen load reductions in the oligohaline/mesohaline zone because the controlling factor, DO, can result in impairment in shorter time frames than increased algal biomass. In 1999, there were reduced rainfall and increased residence times, which resulted in reduced DO levels and a large fish kill. As in the freshwater zone, the percent reduction needed for the oligohaline/mesohaline zone was based on ensuring that the target was met in all of the WBIDs in these zones.

Another conservative assumption involved the methodology used to establish the DO SSAC in the marine portion of the river. For example, a minimum DO of 4.0 mg/L was specified and certain conservative assumptions were made regarding larval recruitment and growth in the development of the SSAC.

Finally, point source flows and loads used in the WLA for the oligohaline/mesohaline zones were based on existing flows and loads with an allowance for growth rather than assuming permitted limits. As noted previously, this approach provides an implicit MOS because it is extremely unlikely that all of the point sources would simultaneously discharge at their full WLA.

6.6 Seasonal Variability

Seasonal variability was assessed during the development of this TMDL as part of the development of the site-specific water quality targets and the determination of assimilative capacity. The site-specific targets developed for the freshwater and oligohaline/mesohaline zones account for the seasonal cycles in algal growth. In the freshwater zone, the critical period occurred from April through August, when excessive algal growth led to imbalances in the algal community structure (dominance by only a few species) and impacts to the food web (undesirable prey for zooplankton and fish species). The chlorophyll *a* target for the freshwater zone (40 µg/L not to be exceeded more than 10 percent of the time) was specifically designed to prevent algal blooms of sufficient duration to cause these imbalances in flora and fauna in the future.

The TMDL for the oligohaline/mesohaline zone also accounted for seasonal variability. As discussed earlier in the MOS section, the summer of 1999 was a critical period, during which DO was below 4.0 mg/L at levels and for durations that could adversely impact the aquatic fauna in the oligohaline/mesohaline zones. The method used to develop the DO target accounts for these critical, seasonal (and diurnal) periods and ensures that excursions of DO levels below the chronic threshold will not occur at a magnitude or duration that would result in impacts to aquatic fauna.

complex for municipalities that wanted to aggregate their wastewater and MS4 allocations, because the allocations to MS4s were expressed as percent reductions. The approach was to simply convert the percent reduction back into a load using the loading in the allocation spreadsheet.

This approach clearly works for the TMDL for the freshwater portion of the river, which is based on a long-term average condition (based on the chlorophyll *a* target of not exceeding 40 ug/L more than 10 percent of the time). This approach also works in the marine portion of the river, even though the TMDL is based on a dry year (1999 was the worst-case year for DO, when tributary flows were low, nutrients were concentrated in the river due to less dilution, and residence times were longer). Model runs[4] indicate that the percent reductions needed in other model years are about half of the reductions in 1999 (a 15 percent reduction required in 1996 and 1997, compared with a 28.5 percent reduction required in 1999), while the urban stormwater loads for these years are less than twice the 1999 load. As such, it is adequately protective to use the 1999 load for aggregation purposes.

For the aggregate allocations, the Department plans to issue watershed permits that will require compliance with the aggregate WLA. These permits will be in addition to the facilities' current permits, and will focus on compliance with the WLA.

This approach of converting the percent reduction back into the allowable load for 1999 is also applicable if MS4s decide to meet their required reductions through water quality credit trading. The WLAs given to point sources can be modified via trading as long as the overall load does not exceed the TMDL. The combined WLA (both total and facility-specific) is provided to allow flexibility so that reductions from one discharger can be shifted to another as long as the net allocation achieves the TMDL. The Department plans to address the permitting process and requirements for water quality credit trading, including trading factors, in the Basin Management Action Plan (BMAP) for the TMDL.

6.5 Margin of Safety

Consistent with the recommendations of the Allocation Technical Advisory Committee (Department, 2001), an implicit MOS was assumed in the development of this TMDL. An implicit MOS was provided by the conservative decisions associated with a number of modeling assumptions, the development of site-specific alternative water quality targets, and the development of assimilative capacity.

In the freshwater zone, multiple years of phytoplankton and zooplankton field measurements were evaluated to establish the site-specific chlorophyll *a* level beyond which zooplankton abundance and diversity started to decline. Hydrodynamic/water quality simulations over four different years were then evaluated to determine the appropriate long-term average TN and TP load reductions necessary to meet the chlorophyll *a* target. These four years represent flows that were slightly drier than average conditions and, given that the effects of nutrient impairment are more prominent in dry conditions, this long-term, yet dry period is considered conservative.

The expression of the TMDL also provided an implicit MOS because equal percent reductions of both TN and TP were required, even though both nutrients may not be the limiting factor for a

[4] These model runs were evaluated because the Department was concerned that the amount of load aggregated, if based on the dry year loading, could conceivably be inadequately protective during wetter years when MS4 loads would be higher, **depending on the percent reduction required for the wetter years.**