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ST. JOHNS RIVER WATER MANAGEMENT DISTRICT

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POLLUTANT LOAD REDUCTION GOALS FOR SEVEN MAJOR LAKES IN THE UPPER OCKLAWAHA RIVER BASIN

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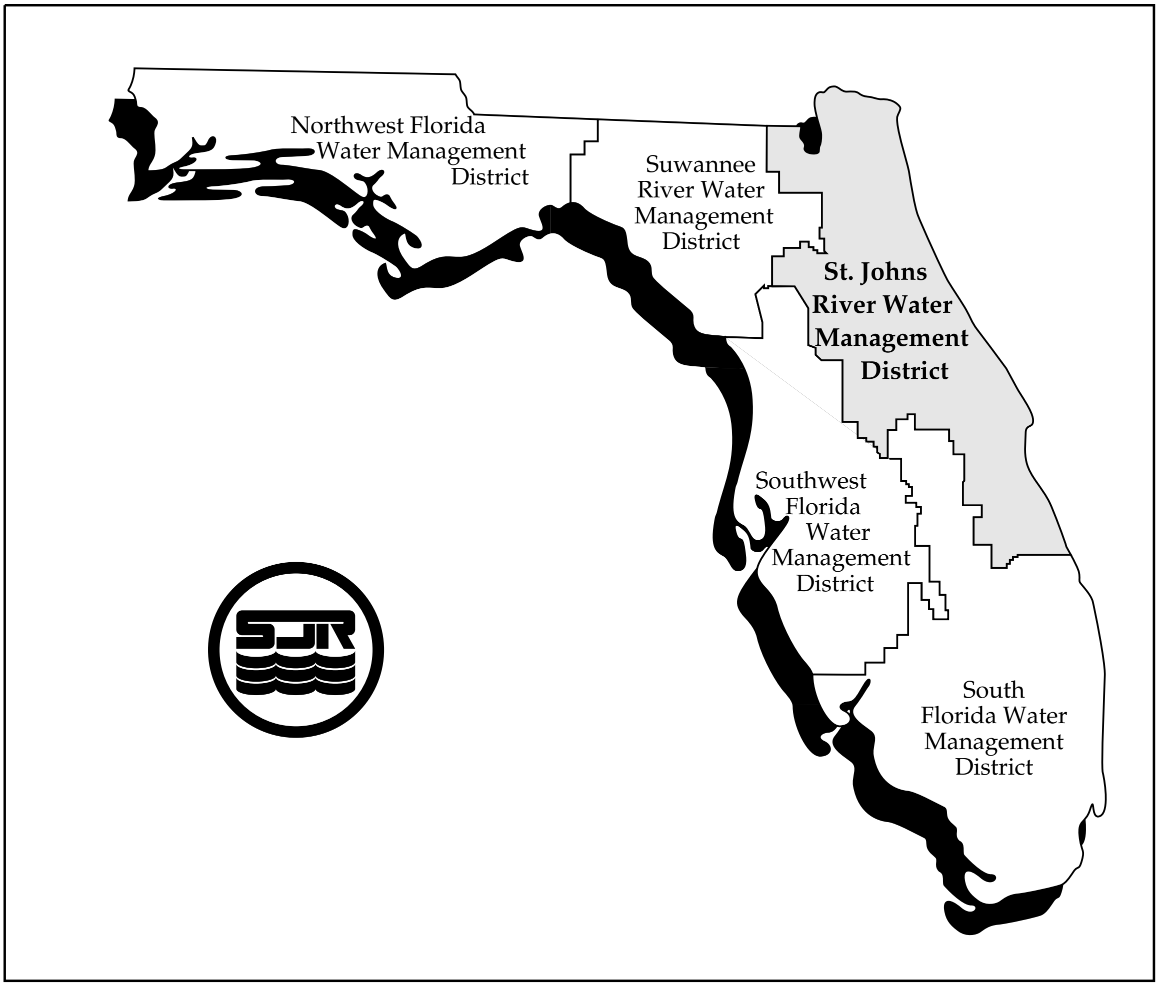

The St. Johns River Water Management District (SJRWMD) was created by the Florida Legislature in 1972 to be one of five water management districts in Florida. It includes all or part of 18 counties in northeast Florida. The mission of SJRWMD is to ensure the sustainable use and protection of water resources for the benefit of the people of the District and the state of Florida. SJRWMD accomplishes its mission through regulation; applied research; assistance to federal, state, and local governments; operation and maintenance of water control works; and land acquisition and management.

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EXECUTIVE SUMMARY

Pollutant load reduction goals (PLRGs) are defined as estimated numeric reductions in pollutant loadings needed to preserve or restore designated uses of receiving bodies of water and maintain water quality consistent with applicable state water quality standards.

The general process for development of PLRGs is to

1. Identify the critical pollutant(s).
2. Estimate the existing pollutant load.
3. Determine the desired concentration for restoration or compliance with state water quality standards.
4. Determine the allowable pollutant load to reach the desired concentration.
5. Determine the necessary load reductions (PLRGs).

One priority area for PLRG determinations is the impaired water bodies in the Upper Ocklawaha River Basin (UORB). The UORB was selected for one of five Surface Water Improvement and Management (SWIM) programs within the St. Johns River Water Management District (SJRWMD). Surface waters within the UORB are naturally productive. However, increases in nutrient loading from intensive agriculture and urbanization have severely degraded water quality. This report develops revised PLRGs for phosphorus for the seven major lakes in the UORB: Beauclair, Dora, Harris-Little Harris, Eustis, Griffin, Yale, and Weir. The recommendations for phosphorus PLRGs for these lakes are the difference between the current loads and the reduced loads needed to restore water quality to levels more similar to natural background conditions.

Interim PLRGs for the UORB lakes were recommended by SJRWMD in May 2003 (Fulton et al. 2003). The interim PLRGs have been used by the Florida Department of Environmental Protection to establish total maximum daily loads for the lakes. This report includes additional analyses and modeling to develop revised PLRG recommendations for these lakes.

Existing external phosphorus loads for the seven lakes were estimated for the 10-year period 1991–2000. The natural background phosphorus concentration for the lakes was determined through a combination of existing concentrations in reference lakes and modeling of natural background conditions in the basin. Data collected from the UORB lakes between 1998 and 2001 were analyzed to determine the relationship between phosphorus concentrations and water transparency (compensation point). Target phosphorus concentrations were established by allowing a 10% degradation from natural background water quality as outlined in Chapter 62-302.530, *Florida Administrative Code*. Water quality modeling for the UORB lakes supported a predicted proportional relationship between phosphorus concentrations in the lakes and external phosphorus loading.

The estimated current and recommended external total phosphorus (TP) loads for the UORB lakes are as follows:

Lake	External Phosphorus Load (metric tons/year)			Percent Reduction
	Current TP load	Target TP load	Load Reduction Goal	
Beauclair	21.2	3.2	18.0	85
Dora	18.2	6.2	12.0	66
Harris	12.7	8.7	4.0	31
Eustis	16.2	10.4	5.8	36
Griffin	35.6	11.9	23.7	67
Yale	1.52	1.37	0.15	10
Weir	1.23	1.23	0	0

These PLRGs were prepared in response to a request from the Florida Department of Environmental Protection and the U.S. Environmental Protection Agency, for their use in development of total maximum daily loads for these water bodies.

SUMMARY OF CHANGES FROM INTERIM PLRG RECOMMENDATIONS (MAY 2003)

There were several changes in the methods for developing external nutrient budgets.

There were a few updates to the estimated nutrient concentrations in stormwater runoff.

Nutrient loading calculations in the interim PLRGs assumed that there was treatment of storm water from all lands developed after 1987, which was estimated from cases in which areas in the 1995 land use map exceeded those in the 1987 land use map for all land use categories. In this report, stormwater treatment was not assumed for cases in which there were increases in area of undeveloped land use categories (forest/rangeland, wetlands, and water) in the 1995 land use map.

The stormwater treatment function was deleted from the estimates of discharges from municipal and industrial sprayfields because the reported nutrient concentrations are apparently for discharges leaving the sites after any stormwater treatment.

The estimates of nutrient losses in transport in the interim PLRGs assumed that 50% of phosphorus and nitrogen in stormwater runoff was sediment-associated and 50% was dissolved for all land uses except muck farms and restoration areas. In this report, land-use specific fractions of sediment-associated and dissolved phosphorus and nitrogen were used in estimates of transport losses. Similar procedures were used to estimate transport losses for point source discharges.

These changes to the nutrient budgets tended to raise estimates of nutrient loading from stormwater runoff for most land uses and from point sources, although the overall effects on nutrient budgets were small. The largest changes were about a 6% increase in the estimated total phosphorus loading and a 13% increase in the estimated total nitrogen loading for Lake Yale. Increases in estimated total phosphorus loading were about 4% for Lake Harris and 1% or less for the other lakes.

Other changes and additions:

- Additional information was included on the lakes and the watershed, including morphometric data for the lakes, estimates of hydraulic detention times, nutrient retention and sedimentation in the lakes, and land use in contributing subbasins.
- Additional text clarification and new appendices summarizing model results for 1986–90 and estimated natural background water quality from reference lakes data sets were added in response to questions about the May 2003 report.
- Changes to estimates of hydraulic detention times for the lakes resulted in changes in the estimate of natural background phosphorus concentrations from hydrologically and morphologically similar lakes, and this in turn

resulted in slight changes in the target phosphorus concentrations for the lakes.

- A preliminary assessment was incorporated of the effects of the target phosphorus concentrations on water quality (chlorophyll-*a*, Secchi depth, and compliance with the Florida Impaired Waters Rule) and colonization of submersed aquatic vegetation.
- Water quality modeling was conducted to evaluate the assumption in the interim PLRGs of a directly proportional relationship between external nutrient load and in-lake nutrient concentrations.
- The report includes a preliminary evaluation of the feasibility of attaining the proposed PLRGs for the UORB lakes by reducing discharges from the upstream Lake Apopka Basin and from SJRWMD wetland restoration projects in the UORB.

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INTRODUCTION

The Florida Legislature, through the Water Resources Implementation Act (Chapter 62-40, *Florida Administrative Code* [*F.A.C.*]), requires the water management districts to develop pollutant load reduction goals (PLRGs) for problem constituents in priority water bodies within their boundaries. The priority water bodies are identified by the Florida Department of Environmental Protection (FDEP) as impaired in the Florida 305b report to the U.S. Environmental Protection Agency (EPA) and are listed on the 303d list as requiring a total maximum daily load (TMDL) to be developed to allow the water bodies to achieve their designated uses.

PLRGs are defined as estimated numeric reductions in pollutant loadings needed to preserve or restore designated uses of receiving bodies of water and to maintain water quality consistent with applicable state water quality standards.

The general process for development of PLRGs is to

1. Identify the critical pollutant(s)
2. Estimate the existing pollutant load
3. Determine the desired concentration for restoration or compliance with state water quality standards
4. Determine the allowable pollutant load to reach the desired concentration
5. Determine the necessary load reductions (PLRGs)

PLRGs reflect the long-term reductions needed to achieve designated uses and are expressed as average annual or seasonal loads or concentrations. PLRGs are implemented through watershed management plans or appropriate regulations. PLRGs may primarily be achieved through an emphasis on the development and implementation of best management practices (BMPs). A number of other management techniques may also be utilized, including purchase and restoration of muck farms, construction of agricultural recycling reservoirs, utilization of water management areas, and regulatory enforcement of point sources.

One priority area for PLRG determinations is the impaired water bodies in the Upper Ocklawaha River Basin.

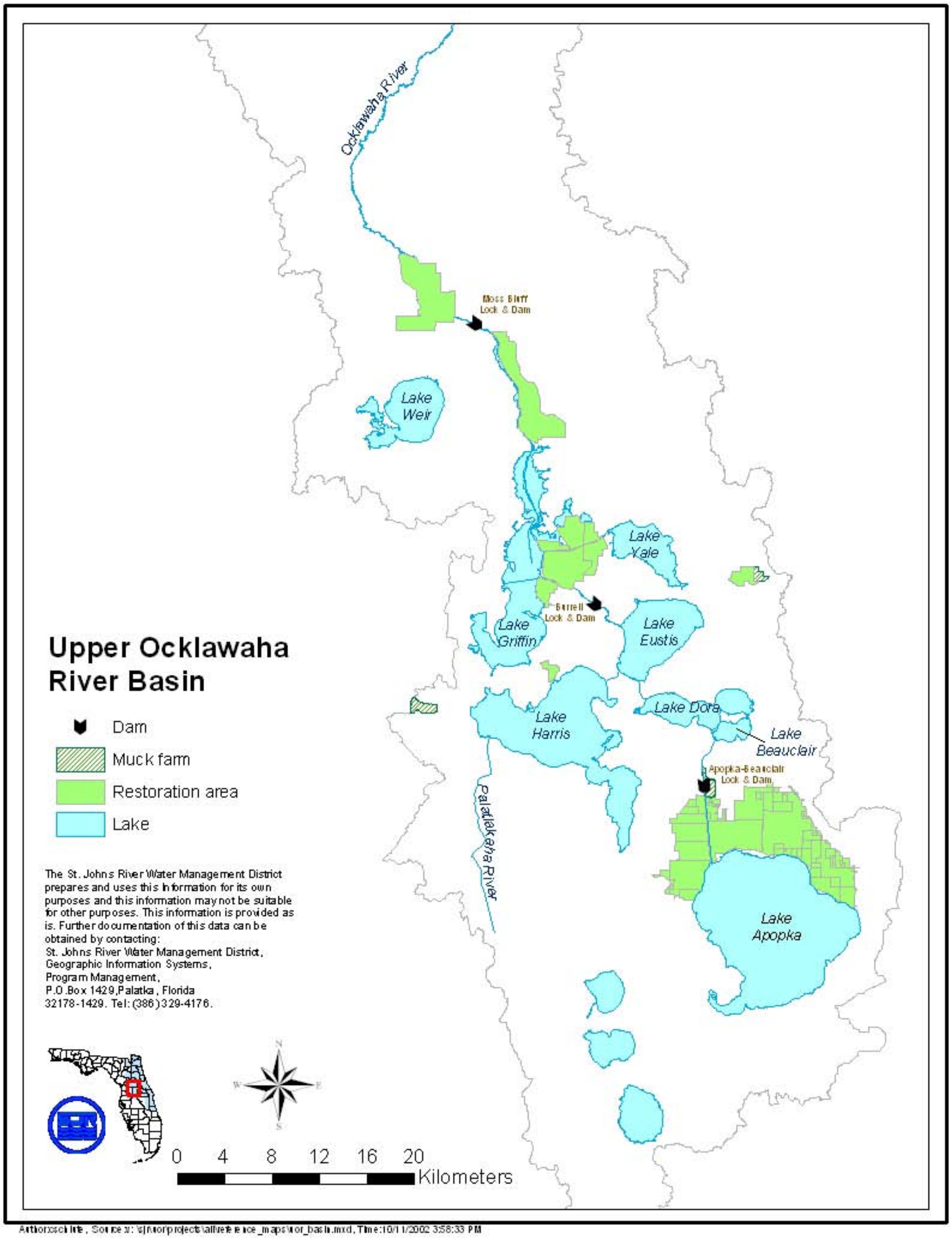

Figure 1. The Upper Ocklawaha River Basin

THE UPPER OCKLAWAHA RIVER BASIN

The Upper Ocklawaha River Basin (UORB) was identified by Lowe et al. (1988) as having a high priority for restoration. The UORB was subsequently selected for development as one of five Surface Water Improvement and Management (SWIM) programs within the St. Johns River Water Management District (SJRWMD). A major goal of the SWIM program for the UORB is the reduction of nutrient levels to attain water quality necessary to restore and maintain healthy and productive natural systems and to meet FDEP Class III water quality standards (Fulton 1995b).

The UORB is located in Marion, Lake, Orange, and Sumter counties of central peninsular Florida (Figure 1). The drainage basin encompasses 1,652 square kilometers (km^2), extending from the Apopka-Beauclair water control structure north of Lake Apopka to State Road 40 near Ocala. The southern region includes several interconnected lakes which comprise the Ocklawaha chain of lakes. Flow into the basin originates from the Palatlakaha River subbasin, which enters into Lake Harris, and the Lake Apopka subbasin, which drains into Lake Beauclair through the Apopka-Beauclair Canal. Lake Beauclair drains directly into Lake Dora, which drains into Lake Eustis through the Dora Canal. Lake Harris-Little Lake Harris also connects with Lake Eustis through the Dead River. Lake Eustis is connected to Lake Griffin by Haines Creek. Lake Yale is connected to Lake Griffin by the Yale-Griffin Canal, and also has a culvert connection with Lake Ella. The Ocklawaha River starts at the north end of Lake Griffin. Lake Weir also drains into the Ocklawaha River.

Virtually all the surface water flow is regulated by water control structures. These structures have altered the natural periodic fluctuations in lake stages and stream discharges. As a result, the lakes function hydrologically as managed reservoirs rather than as natural water bodies. Flow from the Palatlakaha River is controlled by a series of structures operated by the Lake County Water Authority. The Apopka-Beauclair Lock and Dam is operated by SJRWMD to regulate water levels in Lake Apopka. Burrell Lock and Dam on Haines Creek is operated by SJRWMD to maintain water levels in Lakes Eustis, Harris, Dora, and Beauclair. SJRWMD operates the Moss Bluff Lock and Dam on the Ocklawaha River as the local sponsor for the Four River Basins Project in accordance with regulations prescribed by the U.S. Army Corps of Engineers (USACE) to maintain water levels in Lake Griffin; this structure also influences water levels in Lake Yale. Lakes Yale and Weir are partially controlled by fixed crest weirs, which allow outflow through their outlet canals only when lake levels exceed the weir crest elevation.

During the late 1800s, resources in the UORB were developed for tourism, and for agricultural and commercial industry, as barge and steamship traffic increased. Visitors were attracted to the region for its outstanding fishing and other aquatic-related recreation. The construction of water control structures and channelization of the river to facilitate navigation began as early as 1893. The present configuration of locks and dams was completed in 1974.

Much of the agricultural land around the major lakes and the Ocklawaha River consisted of drained wetlands. About 6,000 hectares of floodplain wetlands in the UORB were drained for agriculture. Interior drainage ditches, pump stations, and perimeter levees often drained these muck farms, with drainage water pumped into adjacent water bodies.

Water quality and aquatic and wetland habitats in the basin declined dramatically over the last century. The Ocklawaha River was dredged for navigation, and 24 km of the upper river channel were abandoned, floodplain wetlands were drained for agriculture, three dams stabilized water levels, and urban growth became a major factor in the basin. The impacts of urban and agricultural development on water quality within the basin were first documented during the late 1940s. Pollutants from upstream Lake Apopka and in stormwater and wastewater discharges promoted algae growth in the basin lakes, dead algae accumulated as deep organic sediments on lake bottoms, and aquatic plants died because sunlight could not penetrate the murky waters. Stabilized water levels and reduced flows contributed to further degradation of water quality.

Surface waters within the UORB are naturally productive (Canfield 1981). However, increases in nutrient loading from intensive agriculture and urbanization have degraded water quality to levels that severely impact the ecological, aesthetic, recreational, and commercial benefits of these aquatic resources. Most of the major lakes in the basin have been characterized as eutrophic, including Lakes Beauclair, Dora, Harris-Little Harris, Eustis, and Griffin. Only Lakes Weir and Yale have been classified in the less productive mesotrophic category (Shannon and Brezonik 1972; Canfield 1981).

SJRWMD has acquired more than 4,000 hectares of former muck farms in the UORB and has begun restoration of wetland habitat on these lands. SJRWMD restoration efforts focus on reducing nutrients and other pollutants in basin water bodies, reestablishing more natural water level fluctuations and flows, restoring the original Ocklawaha River channel, and restoring aquatic and wetland habitats at former muck farms.

IDENTIFICATION OF PHOSPHORUS AS THE PRIMARY POLLUTANT OF CONCERN

Studies of lake sediments in the UORB have shown substantial historical increases in phosphorus accumulation rates, indicative of increases in external loading of phosphorus to the lakes (Schelske 1998; Schelske et al. 1999, 2001). In four of the lakes for which nitrogen accumulation rates were also reported, the increases in phosphorus accumulation rates exceeded those for nitrogen (Schelske et al. 2001).

Ratios of total nitrogen (TN) to total phosphorus (TP) in lake waters suggest that algal production is potentially limited by phosphorus availability, except in lakes where excessive phosphorus loading has led to potential nitrogen limitation or mixed phosphorus and nitrogen limitation (i.e., Lakes Beauclair and Griffin) (Fulton 1995a). Recent bioassay experiments in Lake Griffin have indicated that phytoplankton production was generally limited by nitrogen, although phosphorus limitation tended to occur during periods of lowest apparent external nutrient loading and low or declining phytoplankton abundance (Phlips and Schelske 2002). We interpret these results as indicating that excessive phosphorus loading has decreased the N/P ratios, leading to secondary nitrogen limitation. Phytoplankton assemblages in the UORB lakes have been dominated by nitrogen-fixing cyanobacteria (Phlips and Schelske 2002; SJRWMD unpublished data), which can convert abundant supplies of nitrogen gas to bioavailable forms of nitrogen when sufficient supplies of phosphorus and other nutrients are available. Therefore, we believe that phosphorus is the primary nutrient of concern for eutrophication of these lakes, and at this time we will recommend a PLRG only for phosphorus. Actions taken to reduce phosphorus loading to the lakes can also be expected to reduce nitrogen loading. For example, stormwater treatment systems are effective in removing both phosphorus and nitrogen (Appendix B; England 2001). Also, lowered phosphorus levels in the lakes will contribute indirectly to reduced nitrogen levels by limiting nitrogen-fixing cyanobacteria (Paerl et al. 2001).

Phosphorus load reduction also can be expected to reduce levels of un-ionized ammonia in basin water bodies. Un-ionized ammonia is one form of nitrogen found in aquatic systems. It is a naturally occurring compound, but human activities that contribute to eutrophication of water bodies can increase concentrations of un-ionized ammonia. Ammonia is formed primarily by decomposition of organic compounds containing nitrogen. Elevated ammonia levels result from decomposition of algal blooms stimulated by excess levels of phosphorus and nitrogen. Ammonia occurs in two forms, un-ionized ammonia and ammonium ion. Un-ionized ammonia is more toxic to aquatic animals than is ammonium ion. The amount of ammonia that takes the form of un-ionized ammonia depends primarily on the acid-base balance of the water (the pH). Higher pH leads to a greater proportion of un-ionized ammonia (FDEP 2001). High pH is also a result of eutrophication-induced algal blooms. So for several reasons (increased nitrogen fixation, decomposition of algal blooms, high pH), un-ionized ammonia is a consequence of eutrophication.

METHODS

Four steps were taken in development of PLRGs for the lakes in the UORB:

1. To develop a water budget for the UORB lakes
2. To determine water quality and external loading of phosphorus for existing conditions
3. To establish the desired water quality from natural background conditions and management goals
4. To conduct lake water quality modeling to establish phosphorus loading limits to meet the water quality goals

WATER BUDGET FOR THE UORB LAKES

Lake water budgets were developed for the years 1991–2000 for four major hydrologic basins within the UORB:

- Burrell Basin, which includes the four lakes upstream of the Burrell gauge and downstream of the gauges on the Apopka-Beauclair Canal and Palatlakaha River (Lakes Beauclair, Dora, Harris, and Eustis)
- Lake Griffin Basin
- Lake Yale Basin
- Lake Weir Basin

The lake water budgets were developed using the following equation:

$$\Delta V = P - E - L - O + I + S_i + R + Er \qquad \textbf{(1)}$$

where

ΔV = change in lake storage volume
P = rainfall on lake
E = lake evaporation
L = leakage from lake to underlying aquifer
O = stream discharge from lake
I = stream inflow into lake
S_i = spring discharge into lake
R = runoff from watershed to the lake
Er = error term

Ideally, the sum of the water budget components equals zero. However, there is usually a residual because of random and bias errors in the measurements and estimates of the various components. Rather than distribute the error to each of the components on the basis of the component's error, a generic error term (*Er*) has been added to the water budget equation. The long-term sum of errors should approach zero.

Stream inflows (*I* in Equation 1) refer only to the flows in major tributaries from upstream watersheds, such as flows from Lake Apopka, the Palatlakaha River, or upstream basins within the UORB. The upstream basins are outside of the contributing subbasins for the lakes in question (see Figure 2). These flows are measured at gauges or estimated as discussed later. All other inflow terms in Equation 1, particularly *R* (watershed runoff), originate from within the contributing subbasins for the lakes in question. For example, there are nine contributing subbasins for Lake Eustis (Figure 2). Tributary inflows to Lake Eustis are those that come from the Lake Harris subbasin through the Dead River and from the Lake Dora subbasin through the Dora Canal. Watershed runoff to Lake Eustis represents that coming only from the nine contributing subbasins (some of which may enter the lake from streams originating within those subbasins).

Lake Volume

Lake volume changes (ΔV) were determined from observed daily water elevations in each of the lakes. Elevation data for Lake Eustis and Lake Dora were used for the Burrell Basin. Lake Eustis, Lake Harris, and Little Lake Harris were assumed to have the same elevation, since little head difference is observed between the lakes. There is no stage gauge in Lake Beauclair, so water elevations for that lake were assumed to be the same as for Lake Dora. Lake areas and volumes were determined from water elevations using bathymetric data for the lakes reported in Danek et al. 1991, supplemented with additional topographic data for the lake floodplains and corrections to the benchmark datums used to measure elevations subsequent to the bathymetric survey (Table 1).

Rainfall

Rainfall (*P*) estimates for each of the four UORB basins were developed from a network of SJRWMD and National Oceanic and Atmospheric Administration (NOAA) rainfall stations in the basin. A Thiessen polygon data layer was developed from the rain station locations to define the areal

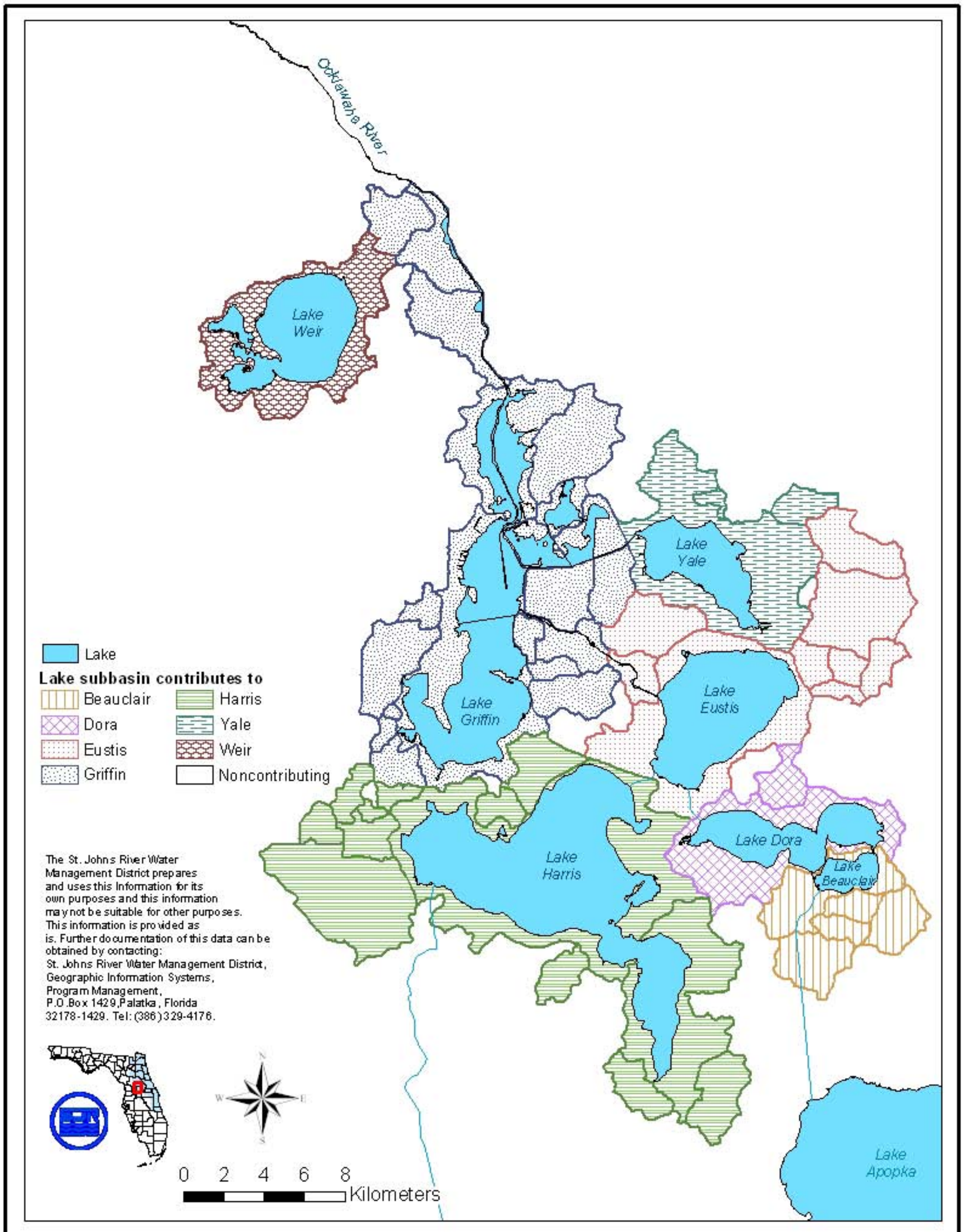

Figure 2. Contributing subbasins for the Upper Ocklawaha River Basin lakes

Table 1. Morphometric data for the major lakes in the Upper Ocklawaha River Basin

Lake	Elevation (ft NGVD)	Surface Area (km^2)	Volume (10^6 m^3)	Mean Depth (m)
Beauclair	63.15	4.39	9.03	2.05
Dora	63.15	17.74	53.28	3.00
Harris	63.17	75.63	277.00	3.66
Eustis	63.14	31.39	108.57	3.46
Griffin	59.29	53.72	106.49	1.98
Yale	59	16.27	60.72	3.73
Weir	57	22.76	131.36	5.77

Note: km = kilometer
m = meter
m^3 = cubic meter
ft NGVD = feet National Geodetic Vertical Datum

extent of each station's rainfall. The number of rain stations varied from year to year, but ranged from 3 to 6 for the Burrell Basin, from 4 to 6 for the Lake Griffin and Lake Yale basins, and from 1 to 2 for the Lake Weir Basin. A composite rainfall estimate was determined each year for each major hydrologic basin by weighting the reported rainfall at the contributing rain stations by the areal extent covered by each station.

Evaporation

Estimates of lake evaporation (*E*) were developed by multiplying measured pan evaporation rates by pan evaporation coefficients. The evaporation estimates used the same stations and monthly varying pan evaporation coefficients as used in the UORB hydrological model developed by the SJRWMD Division of Engineering. For the Burrell Basin, the pan evaporation estimates were weighted 0.67 and 0.33 from the Lisbon and Lake Alfred NOAA stations, respectively, while for the Lakes Griffin, Yale, and Weir basins, only data from the Lisbon station were used. The Lisbon weather station is located within the UORB, while the Lake Alfred station is located south of the UORB, in Polk County.

Leakage

Leakage (L) losses from the lakes were estimated by a procedure similar to that used by Deevey (1988). Deevey started from a general water balance for a seepage lake:

$\Delta H=(P\text{-}E\text{-}L+R)$, where ΔH is change in lake level and the other parameters are as defined above. He reasoned that leakage is most likely to be observed in low rainfall months, when R is near zero and net precipitation (P-E) is negative. Therefore, in those months, L~=net precipitation-ΔH. Leakage for a lake was estimated as the geometric mean of (net precipitation-ΔH) for the months in which net precipitation was negative and the lake level fell by an amount exceeding the net precipitation deficiency. This procedure assumes that leakage occurs throughout the year at near constant rates, driven by a large hydrostatic head difference between the lake surface and the underlying aquifer. The head difference was assumed to be large enough that lake level fluctuations would only slightly change the head. Using this procedure, Deevey's leakage estimates for 18 Florida lakes ranged from 28.4 to 50.9 centimeters per year (cm / yr) (the estimate for Lake Dora was 42.7 cm).

Deevey's methodology was modified because most of the lakes in this basin have significant inflow and outflow, which must be accounted for in the water balance. ΔH was calculated as $(O - I - S_i + \Delta V) / LA$, where LA is lake area and the other parameters are as defined above. Leakage was then calculated using Deevey's method. This resulted in average leakage estimates for 1991–2000 of 21.0 cm / yr for the Burrell Basin lakes, 40.0 cm / yr for Lake Griffin, 28.6 cm / yr for Lake Yale, and 31.0 cm / yr for Lake Weir.

Tributary Discharge

Gauged Tributary

Tributary flows in (I) and out (O) of the lake basins were computed from U.S. Geological Survey (USGS) measured data for the Apopka-Beauclair Lock and Dam, Palatlakaha River (M-1 Structure), Burrell Lock and Dam, and Moss Bluff Lock and Dam. These data were used as currently calculated by USGS except for the period between January 1, 1997, and July 31, 1997, at the Burrell Lock and Dam. Previous modeling efforts for Lake Griffin and Lake Eustis indicated that USGS overestimated discharge during this period. USGS did not take any actual measurements of discharge during this period. Therefore, SJRWMD estimates of discharge from the SJRWMD operation of the Burrell Lock and Dam were used during this period.

Ungauged Tributary

Discharges from Lake Yale to Lake Griffin and Lake Yale to Lake Ella were estimated from rating curves developed using the HEC-RAS model of USACE. The assumptions used for developing these rating curves are as follows:

- Lake Yale to Lake Ella: 91 cm CMP with an invert of 18.0 meters
- Lake Yale to Lake Griffin: 91 cm CMP with an invert of 18.2 meters

Note: CMP = corrugated metal pipe.

Discharge volumes on ungauged tributaries connecting lakes within the Burrell Basin were estimated on a monthly basis using a water budget procedure described in Fulton 1995a. As a first step, we developed estimates of areal net water yield for the Burrell Basin on a monthly basis:

$$\Delta W_B = \frac{O - I - S_i + \Delta V}{A} \quad \textbf{(2)}$$

where

ΔW_B = areal net water yield for the Burrell Basin
A = contributing watershed area (includes surface areas of lakes in the basin), and the other parameters are as defined in Equation 1

The estimated areal net water yields were then used to estimate flows on tributaries without discharge gauges. For example, the estimate of discharge from Lake Harris-Little Lake Harris into Lake Eustis (O_H) was calculated using

$$O_H = I_P + S_i + (A_{Hw} x \Delta W_B) - \Delta V_H \quad \textbf{(3)}$$

where

I_P = inflows to Lake Harris in Palatlakaha River discharge
S_i = spring discharges into Lake Harris-Little Lake Harris
A_{Hw} = Lake Harris-Little Lake Harris contributing watershed area (including lake surface area)
ΔV_H = change in storage volume for Lake Harris-Little Lake Harris

Flow reversals are known to occur on some of the ungauged tributaries connecting the lakes in the Burrell Basin, particularly in the Dead River connecting Lake Harris with Lake Eustis. Fulton (1995a) assumed that discharges from the 'upstream' lake were zero in months during 1984–90 in which it was estimated that there were net negative discharges from that lake. In this report, we assumed that occurrences of net negative discharges were incidents of net flow reversal. From May 1993 through September 1996, USGS estimated discharges on the Dead River, although the accuracy of those estimates was considered to be poor (USGS 1996). Excluding two questionable outliers in the USGS measurements, our estimates from the water budgets of monthly discharges between Lakes Harris and Eustis were strongly correlated with the USGS estimates ($r=0.74$, $p<0.01$), and the average discharge estimated from the water budget was about 75% of the reported average discharge. Even including the outliers, the correlation would still be significant ($r=0.61$, $p<0.01$). Thus, we believe that the water budgets give estimates of net flows between the lakes on ungauged tributaries that are reasonably consistent with available flow information.

Spring Discharge

Spring discharges (S_i) were estimated from the four known springs discharging into Lake Harris. The owner of Bugg Spring, Mr. Joe Branham, estimates discharges and collects samples for analysis by Lakewatch on a monthly basis; monthly volume discharges for this spring were estimated from these measurements. More limited discharge data are available for the other springs (no more than twice annually). Regression relationships between measured discharge for each spring and the head difference between Lake Harris and a well site in Winter Gardens (L-0199) near Lake Apopka were used to predict discharge volumes from the other three springs (minimum $r^2 = 0.734$). If there were any unknown spring discharges to the lakes, they would act to reduce the estimates of leakage losses and increase the error term in the overall water budget.

Stormwater Runoff

Stormwater runoff (R) was calculated using Soil Conversation Service (SCS) methodologies as outlined in Suphunvorranop 1985 and incorporated into the water budget. The SCS method is based on Equation 4:

$$Q = \frac{(P - 0.2 * S)^2}{P + 0.8 * S} \tag{4}$$

where

Q = runoff (cm)
P = rain (cm)
S = potential storage (cm)

The potential storage S is expressed as in Equation 5:

$$S = \frac{2540}{CN} - 25.4 \quad \textbf{(5)}$$

where

CN = curve number

Curve number (CN) is a dimensionless number ranging from 0 to 100 and is a function of soil type, land use, and antecedent soil moisture.

The drainage basins used for runoff estimates were slightly modified from a delineation previously produced by the Engineering Division of SJRWMD for use in developing a hydrologic model for the UORB. The drainage basin for each lake was divided into a limited number of contributing secondary subbasins (ranging from one subbasin for Lake Weir to 20 subbasins for Lake Griffin) (Figure 2).

The soil survey geographic (SSURGO) database developed by the Natural Resources Conservation Service was used to classify the hydrologic soil groups (A, B, C, D, W, and X). The primary land use data layer was derived from aerial photography taken in 1994–95. A secondary land use layer was developed from aerial photography taken primarily in 1987. Land use categories were aggregated into the general categories shown in Table 2. Summaries of land uses in contributing subbasins are included in Appendix A. Curve numbers were developed for 19 land uses and the soil types previously described, based on average antecedent moisture conditions (AMC II).

An areal-weighted CN was developed for each lake system using Equation 6, based on the CNs in Table 2 and watershed areas. A weighted CN based on total impervious area underestimates runoff for smaller storm events (2.5–5 cm) when used with the standard SCS runoff equation. Therefore, the equation for predicting direct runoff (Q) was expanded to account for directly and non-directly connected impervious areas (DCIA and NDCIA, respectively). The percentages of DCIA and NDCIA were adjusted from literature values based on field observation and review of aerial photography.

Table 2. Curve numbers* by soil type and land use

Land Use	Curve Numbers Soil Type			
	A	B	C	D
Low density residential	51	68	79	84
Medium density residential	57	72	81	86
High density residential	77	85	90	92
Low density commercial	77	85	90	92
High density commercial	89	92	94	95
Industrial	81	88	91	95
Mining	32	58	72	79
Openland/recreational	49	69	79	84
Pasture	47	67	81	88
Cropland	64	75	82	84
Tree crops	32	58	72	79
Feeding operations	59	74	82	86
Other agriculture	59	74	82	86
Forest/rangeland	36	60	73	79
Water	98	98	98	98
Wetlands	89	89	89	89
Spray fields	59	74	82	86
Muck farm/restoration areas	70	81	86	90
Lakes	98	98	98	98

*AMC II (AMC = antecedent moisture condition)

Source: Suphunvorranop 1985

Table 3 shows typical values used in the UORB. A major adjustment in the impervious percentages was made for DCIA and associated pervious values in the water and wetland land use categories to account for closed lake and wetland systems, as shown in Table 4. It was our opinion that lakes and wetlands which did have not well-defined outfalls should have their

Table 3. Percentages of impervious area by land use, for all four basins

Land Use	DCIA	NDCIA	Pervious	Sum of NDCIA and Pervious
Low density residential	5	10	85	95
Medium density residential	15	20	65	85
High density residential	25	40	35	75
Low density commercial	40	40	20	60
High density commercial	45	35	20	55
Industrial	50	30	20	50
Mining	1	1	98	99
Openland/recreational	1	1	98	99
Pasture	1	1	98	99
Cropland	1	1	98	99
Tree crops	1	1	98	99
Feeding operations	2	1	97	98
Other agriculture	1	1	98	99
Forest/rangeland	1	1	98	99
Water*	85	15	0	15
Wetlands*	75	0	25	25
Spray fields	2	2	96	98
Muck farm/restoration areas†	2	2	96	98
Lakes	100	0	0	0

Note: DCIA = directly connected impervious area
NDCIA = non-directly connected impervious area

*Subsequently modified (see Table 4)
†Subsequently modified (see Table 5)

Source: Camp Dresser and McKee 1994

Table 4. Basin-specific impervious and pervious percentages for water and wetlands

Land Use	Burrell Basin		Lake Griffin Basin		Lake Yale Basin		Lake Weir Basin	
	DCIA	Pervious	DCIA	Pervious	DCIA	Pervious	DCIA	Pervious
Water	50	50	35	65	65	35	85	15
Wetlands	60	40	70	30	40	60	75	25

Note: DCIA = directly connected impervious area discharges reduced. Our method of reduction was to change DCIA, NDCIA, and pervious percentages. These changes were based on staff knowledge of the area and comparison of closed basins with open basins.

Adjustments were also made to the DCIA and pervious values for the muck farm / restoration area land use category, based on areas of upland and muck farm or wetland restoration and the operational history (Table 5). This land

Table 5. Muck farm/restoration area-specific impervious and pervious percentages

Lake Basin	Farm or Restoration Area	Upland Area* (hectares)	Farm/Wetland Area (hectares)	DCIA (%)	Pervious (%)
Beauclair	Hurley	10	170	50	50
Harris	JA-Mar	28	162	50	50
	Lake Harris Conservation Area	0	171	50	50
Eustis	Pine Meadows	0	271	60	40
	Springhill	0.5	92	60	40
Griffin†	Griffin Flow-Way/ Knight-South-Mathews	192	817	40	60
	Long	80	263	40	60
	Knight-North	25	150	60	40
	Walker Ranch	283	157	40	60
	Eustis	0	243	40	60
	Lowrie Brown	18	253	60	40

Note: DCIA = directly connected impervious area

*Percentages of impervious area for upland area are shown in Table 3
†An area-weighted average of the uplands and the farms were used in the Lake Griffin Basin

use category includes operating muck farms developed in historic wetland areas and wetland restoration projects on former muck farms. The DCIA estimation for the muck farms / restoration areas (Table 5) is a starting point for PLRG development and is based on very limited data. The functions and operations of these properties create hydrologic differences from typical land uses in runoff calculations. There are periods when there is no discharge from the sites and backflow enters the areas from the lakes, making accurate estimations of runoff coefficients impossible. The sites should be treated as point sources. Currently, there are no adequate data to make these calculations. For the interim, we have left the calculations as they are, with the understanding that in the future, the sites will be treated as point sources when enough data are available. As can be seen from Tables 4 and 5, the estimated DCIA for these sites falls within the DCIA ranges used for water and wetlands.

As expected, the runoff coefficients are also approximately the same for the muck farm / restoration areas, water, and wetlands. The estimation of nutrient discharges from the restoration areas addresses uncertainties in runoff by shutting off flow from the sites during periods in which water levels in the restoration areas were below lake levels or below the invert elevations of connecting culverts.

The DCIA, NDCIA, and pervious areas were calculated by applying the percentages in Tables 3–5 to the land use areas by soil type, and summing the totals for the three categories. The weighted watershed CN (Equation 6) can also be expressed as Equation 7, and this formula can be rearranged to solve for pervious area CN (Equation 8). The CN for the DCIA was assumed to be 98. With the CN for DCIA and the weighted watershed CN (Equation 6), an average AMC II CN for the pervious areas (NDCIA and pervious) in each watershed could be determined using Equation 8.

$$CN_{watershed} = \frac{\sum (Area * CN)}{TotalArea} \tag{6}$$

$$CN_{watershed} = \frac{((CN_{DCIA} * Area_{DCIA}) + (CN_{pervious} * Area_{pervious}))}{TotalArea} \tag{7}$$

$$CN_{pervious} = \frac{((CN_{watershed} * TotalArea) - (CN_{DCIA} * Area_{DCIA}))}{Area_{pervious}} \tag{8}$$

AMC I and III were accounted for in applying the CNs to the direct runoff equations by analyzing the previous 5-day rainfall totals using the criteria in Table 6 and adjusting the calculated CN up or down based on an AMC of I or III (Table 7). This adjustment was made based on dormant and growing seasons. If the rainfall event produced less rain than the soil storage volume, only the runoff from the DCIA was calculated.

Equation 4 is the runoff equation from the SCS methodologies. Soil storage (*S*) for pervious area is calculated by Equation 9, with $CN_{pervious}$ determined as described in the previous section. Developing runoff, which accounts for DCIA, requires the segregation of pervious area from DCIA (Equation 10).

Equation 11 and Equation 12 are used for the pervious area and DCIA direct runoff calculations, respectively. Precipitation for the pervious area is increased by the NDCIA relationship to the pervious area to account for the NDCIA discharging to the pervious area using Equation 13.

Table 6. Classifications of antecedent moisture conditions (AMC)

Total 5-Day Antecedent Rainfall (centimeters)		
AMC	Dormant Season	Growing Season
I	<1.3	<3.6
II	1.3–2.8	3.6–5.3
III	>2.8	>5.3

Table 7. Equivalent curve numbers going from AMC II to either lower (AMC I) or higher (AMC III) antecedent moisture conditions

AMC II	AMC I	AMC III
0	0	0
5	2	17
10	4	26
15	7	33
20	9	39
25	12	45
30	15	50
35	19	55
40	23	60
45	27	65
50	31	70
55	35	75
60	40	79
65	45	83
70	51	87
75	57	91
80	63	94
85	70	97
90	78	98
95	87	99
100	100	100

$$S = \frac{2{,}540}{CN_{pervious}} - 25.4 \quad \textbf{(9)}$$

$$Q = Q_{pervious} + Q_{DCIA} \quad \textbf{(10)}$$

$$Q_{pervious} = \frac{(P' - 0.2 * S)^2}{P' + 0.8 * S}\left(PerviousArea \big/ TotalArea\right) \quad \textbf{(11)}$$

$$Q_{impervious} = P * 0.9 * \left(DCIA \big/ TotalArea\right) \quad \textbf{(12)}$$

$$P' = \left[1 + \left(NDCIA \big/ PerviousArea\right)\right] * P \quad \textbf{(13)}$$

where

Q, *P*, *S*, and *CN* are as described above, and

P' = rain on pervious areas adjusted for NDCIA volume (cm)

Runoff (*Q*) was estimated for every rainfall event for the 10-year period using the above equations. The *Q* was multiplied by the watershed area to produce a volume of runoff in the water budget. The daily volumes of runoff were totaled and divided by the total rainfall for the 10-year period to determine a long-term basin weighted runoff coefficient for each watershed (Table 8).

Table 8. Upper Ocklawaha River Basin weighted runoff coefficients

Basin			
Lake Weir	Lake Griffin	Lake Yale	Burrell
0.16	0.17	0.18	0.24

Annual Storm Runoff Coefficients

Estimation of phosphorus loading in stormwater runoff required the development of individual annual storm runoff coefficients (ASRC) for each land use/soil type, based on the stormwater runoff estimates from the water budgets. As a first step, the basin-weighted runoff coefficients from the water budgets ($ASRC_{wb}$, Table 8) were used to calculate a weighted runoff coefficient (*WRc*) for the pervious areas (NDCIA + pervious) in each basin. This was accomplished by setting a standard method of weighting a runoff coefficient equation equal to the water budget ASRC (Equation 14) and solving for a weighted runoff coefficient (*WRc*) for the pervious areas (Equation 15).

$$ASRC_{wb} = \frac{(DCIA * 0.9) + (PerviousArea * WRc_{pervious})}{TotalArea} \quad \textbf{(14)}$$

$$WRc_{pervious} = \frac{(ASRC_{wb} * TotalArea) - (DCIA * 0.9)}{PerviousArea} \quad \textbf{(15)}$$

The pervious weighted runoff coefficient ($WRc_{pervious}$) for the watershed was then proportioned to the soil types based on soil storage ratios according to typical depth to the groundwater. Based on typical depth to groundwater and the resultant soil storage, it was assumed that D soils would give four times the runoff as compared to A soils. The proportional runoff coefficient (PRC) was computed using the following equation:

$$WRc_{pervious} = \frac{(PRC * Area_{Asoils}) + (2 * PRC * Area_{Bsoils}) + (3 * PRC * Area_{Csoils}) + (4 * PRC * Area_{Dsoils})}{Total\ Pervious\ Area}$$

which was re-arranged to yield

$$PRC = \frac{Total\ Pervious\ Area * WRc_{pervious}}{Area_{Asoils} + (2 * Area_{Bsoils}) + (3 * Area_{Csoils}) + (4 * Area_{Dsoils})} \quad \textbf{(16)}$$

Soil Type	Depth to Groundwater (meters)	Ratio	Soil Type Coefficient
A	>1.2	1	PRC
B	0.9	2	2*PRC
C	0.6	3	3*PRC
D	0.3	4	4*PRC

For example, using the following information in Equations 14, 15, and 16, we can calculate the PRC for the Lake Weir basin:

$ASRC_{wb}$ = 0.156
DCIA (hectares) = 491
Pervious area (hectares = 2,466

Total watershed area (hectares) = 2,958
Area A soils (hectares) = 2,032
Area B soils (hectares) = 159
Area C soils (hectares) = 175
Area D soils (hectares) = 100

Equation 14:

$$0.156 = \frac{(491 * 0.9) + (2{,}466 * WRc_{pervious})}{2{,}958}$$

Equation 15:

$$WRC_{pervious} = \frac{(0.156 * 2{,}958) - (491 * 0.9)}{2{,}466} = 0.0078$$

Equation 16:

$$PRC = \frac{2{,}466 * 0.0078}{2{,}032 + (2*159) + (3*175) + (4*100)} = 0.005846$$

A soils	Pervious coefficient	PRC∗1 = 0.005846
B soils	Pervious coefficient	PRC∗2 = 0.011692
C soils	Pervious coefficient	PRC∗3 = 0.017537
D soils	Pervious coefficient	PRC∗4 = 0.023383

Finally, the pervious area soil-specific runoff coefficients were combined with the runoff coefficients for the DCIA to calculate an average area-weighted runoff coefficient for each land use by soil type. The D-soils pervious coefficient was assigned to soil classifications W and X. For example, Table 9 shows the calculation of soil-specific ASRCs for low density residential land use for the Lake Weir watershed.

Table 10 shows the computed ASRCs for Type B soils for the UORB basins, while Table 11 shows how the ASRCs change with soil type for the Lake Griffin Basin. The UORB ASRCs are generally lower than runoff coefficients used in previous SJRWMD studies, with the exceptions of wetland and water habitats (Fulton 1995a, Table 10; Mundy and Bergman 1998; Hendrickson and Konwinski 1998). However, the UORB ASRCs tend to be similar to or higher than those developed by Pandit and Gopalakrishnan (1996) for the Orlando area, with the exceptions of cropland and feeding operations (Table 10).

Table 9. Lake Weir watershed calculation of annual storm runoff coefficients (ASRC) for low density residential land use by soil type

A	B	C	D	E	F
Soil Type	DCIA Area (hectares)	Pervious Area (hectares)	DCIA Area *x* 0.9	Pervious Area *x* Soil PRC	Weighted ASRC (D+E) / (B+C)
A	22.24	422.59	20.02	2.47	0.051
B	0.81	15.40	0.73	0.18	0.056
C	2.55	48.46	2.30	0.85	0.062
D	0.41	7.73	0.37	0.18	0.067
W	0.07	1.29	0.06	0.03	0.067

Note: DCIA = directly connected impervious area
PRC = proportional runoff coefficient

WATER QUALITY AND EXTERNAL LOADING OF PHOSPHORUS FOR EXISTING CONDITIONS

External loading of phosphorus and nitrogen and in-lake water quality were determined for the 10-year period 1991–2000. Most water chemistry data were collected by SJRWMD monitoring programs. However, for some sites in which insufficient SJRWMD data were available, we used supplementary data from FDEP, the Florida Fish and Wildlife Conservation Commission, Lake County Environmental Services, and Lakewatch.

Methods used in developing external loading estimates for phosphorus and nitrogen were similar to those used in Fulton 1995a. This section briefly summarizes the methods, highlighting changes from the procedures used in Fulton 1995a. See Fulton 1995a for further details on methodology.

Nutrient loading was estimated using the following general equation:

$$L = L_{Pr} + L_{Sp} + L_{Tr} + L_{Ro} + L_{Mf} + L_{ST} + L_{PSI} \tag{17}$$

where

L = total mass loading of TP or TN
L_{Pr} = loading from atmospheric deposition (rainfall and dry deposition)
L_{Sp} = loading from spring discharges
L_{Tr} = loading from tributary inflows

Table 10. Upper Ocklawaha River Basin annual storm runoff coefficients for Type B soils by land use

Land Use	Annual Storm Runoff Coefficients								General Runoff Coefficients
	Lake Beauclair Basin	Lake Dora Basin	Lake Harris Basin	Lake Eustis Basin	Lake Griffin Basin	Lake Yale Basin	Lake Weir Basin	Pandit (1996)	Fulton (1995a)
Low density residential	0.122	0.075	0.064	0.056	0.058	0.066	0.056	0.078	0.367
Medium density residential	0.204	0.162	0.152	0.145	0.147	0.154	0.145	0.113	0.367
High density residential	—	0.249	0.240	0.234	0.236	0.242	—	0.185	0.567
Low density commercial	0.409	0.379	0.372	0.367	0.368	0.373	0.367	0.329	0.700
High density commercial	0.450	0.423	0.416	0.411	0.413	0.417	0.411	—	0.700
Industrial	0.491	0.466	0.460	0.456	0.457	0.461	—	0.256	0.500
Mining	—	—	0.029	—	0.023	—	—	—	0.500
Openland/recreational	0.089	0.041	0.029	0.020	0.023	0.031	0.021	0.026	0.167
Pasture	0.089	0.041	0.029	0.020	0.023	0.031	0.021	0.026	0.233
Cropland	0.089	—	0.029	0.020	0.023	0.031	0.021	0.102	0.233
Tree crops	0.089	0.041	0.029	0.020	0.023	0.031	0.021	0.023	0.233
Feeding operations	—	—	0.038	0.029	—	0.040	—	0.231	0.420
Other agriculture	0.089	0.041	0.029	0.020	0.032	0.031	—	—	0.233
Forest/rangeland	0.089	0.041	0.029	0.020	0.023	0.031	0.021	0.026	0.167
Water	0.491	0.466	0.460	0.456	0.324	0.593	0.767	—	0.353
Wetlands	0.572	0.553	0.548	0.545	0.368	0.373	0.678	—	0.353
Spray fields	—	—	0.038	—	0.032	0.040	—	—	—
Muck farm/restoration areas	0.466	—	0.425	0.545	0.324	—	—	—	—
Lakes	0.900	0.900	0.900	0.900	0.900	0.900	0.900	—	—

Note: — = no data available; no land uses in that category

Table 11. Annual storm runoff coefficients for the Lake Griffin Basin by land use and hydrologic soil group

Land Use	Annual Storm Runoff Coefficients			
	Soil Type			
	A	B	C	D
Low density residential	0.052	0.058	0.065	0.072
Medium density residential	0.141	0.147	0.153	0.159
High density residential	0.230	0.236	0.241	0.246
Low density commercial	0.364	0.368	0.373	0.377
High density commercial	0.409	0.413	0.417	0.420
Industrial	0.454	0.457	0.461	0.464
Mining	0.016	0.023	0.030	0.037
Openland/recreational	0.016	0.023	0.030	0.037
Pasture	0.016	0.023	0.030	0.037
Cropland	0.016	0.023	0.030	0.037
Tree crops	0.016	0.023	0.030	0.037
Feeding operations	0.025	—	—	—
Other agriculture	0.025	0.032	0.039	0.046
Forest/rangeland	0.016	0.023	0.030	0.037
Water	0.320	0.324	0.329	0.333
Wetlands	0.364	0.368	0.373	0.377
Spray fields	0.025	0.032	0.039	0.046
Muck farm/restoration areas	0.320	0.324	0.329	0.333
Lakes	0.900	0.900	0.900	0.900

Note: — = no data available; no land uses in that category

L_{Ro} = loading from nonpoint runoff in the drainage basin
L_{MF} = loading from muck farm or restoration area discharges
L_{ST} = loading from septic tank seepage
L_{PSI} = loading from point source discharges

Atmospheric Deposition

Data on nutrient concentrations in rainfall and dry deposition were taken from a wet / dry deposition collector operated by SJRWMD at the Lake Apopka Marsh Flow-Way, located just south of the UORB. The rainfall and dry deposition data were highly skewed, with occasional very high measurements (possibly contaminated samples) substantially affecting mean values. Therefore, annual median nutrient concentrations in rainfall and rates of dry deposition (which are not as affected by occasional outliers) were determined and used in calculations of direct atmospheric deposition to the lake surface, using equations described in Fulton 1995a. Rainfall estimates from the basin water budgets were multiplied by annual median nutrient concentrations to estimate wet deposition.

Spring Discharges

Four known springs discharge into Lake Harris. Nutrient loading from Bugg Spring was estimated by multiplying monthly discharge by monthly nutrient concentration. More limited discharge and water quality data are available for the other springs (no more than twice annually). Spring discharge volumes estimated as part of the basin water budget were multiplied by average annual nutrient concentration in the spring discharges to estimate nutrient loading from these springs.

Tributary Discharges

For most of the tributaries, nutrient discharges were determined using annual flow-weighted mean nutrient concentrations multiplied by annual discharge volumes (Galat 1990). In cases in which tributary discharges were infrequent and intermittent, nutrient discharges were estimated by multiplying monthly discharge by mean nutrient concentration (includes discharges from Lake Yale and flow reversals in the Burrell Basin).

Stormwater Runoff

Nutrient loads generated from stormwater runoff from land uses other than muck farms, restoration areas, or point source waste disposal areas were calculated by estimating the volume of potential runoff from contributing drainage areas and multiplying by a nutrient concentration. Runoff volumes were estimated by multiplying annual rainfall by the land use / soil-specific ASRCs developed in the previous section. Nutrient concentrations used for stormwater runoff (Table 12) were developed primarily from a compilation of Florida studies by Harper (1994). Other literature sources used in development of runoff concentrations included CH2M HILL 1978, Izuno et al. 1991, Hendrickson and Konwinski 1998, Fonyo et al. 1991, Rushton and Dye 1993, Goldstein and Ulevich 1981, and Ritter and Allen 1982. Runoff

Table 12. Nutrient concentrations and fraction dissolved nutrients used for estimating nutrient loading from stormwater runoff from land use classes

Land Use	Total Phosphorus		Total Nitrogen	
	Concentration (mg/L)	Fraction Dissolved	Concentration (mg/L)	Fraction Dissolved
Low density residential	0.177	0.501	1.77	0.753
Medium density residential	0.3	0.501	2.29	0.753
High density residential	0.49	0.501	2.42	0.753
Low density commercial	0.195	0.414	1.22	0.657
High density commercial	0.43	0.767	2.83	0.767
Industrial	0.339	0.761	1.98	0.761
Mining	0.15	0.467	1.18	0.657
Pasture	0.387	0.722	2.48	0.908
Tree crops	0.14	0.629	2.05	0.908
Cropland	0.666	0.600	4.56	0.908
Other agriculture	0.492	0.687	2.83	0.908
Feeding operations	6.532	0.583	78.23	0.908
Openland/recreational	0.057	0.501	1.25	0.753
Forest/rangeland	0.057	0.501	1.25	0.753
Wetlands	0.057	0.507	1.25	0.775
Water*	0.013/0.025	0.118	0.493/0.716	0.413

Note: mg/L = milligrams per liter

*Lake Weir Basin/Harris Chain-of-Lakes Basin

coefficients reported in these Florida studies are similar in magnitude to other compilations (e.g., Reckhow et al. 1990). Muck farms, restoration areas, and point source waste disposal areas were removed from the general categorization for separate calculation of discharges from those areas. The nutrient concentrations for runoff from wetlands and water habitats presented in Harper 1994 appeared to be from areas that were impacted by runoff from adjacent uplands, and therefore would not reflect the independent contribution from these habitats. Therefore, for wetland habitat, we used the same nutrient concentrations as developed by Harper (1994) for undisturbed upland habitat. For runoff from water habitat, nutrient concentrations used were those estimated for natural background conditions for the Lake Weir and Harris Chain-of-Lakes basins (see “Establishment of desired water quality from natural background conditions and management goals”).

As noted earlier, the primary land use data layer used for stormwater runoff estimates was derived from aerial photography taken in 1994–95. A secondary land use layer was developed from aerial photography taken primarily in 1987. Differences between the 1987 and 1995 land use maps were used to determine development that occurred in the watershed after 1987. It was assumed that there was no stormwater treatment for lands already developed in 1987, but there was treatment of stormwater from lands developed after 1987. We developed summaries by contributing subbasin (Figure 2) of areas of each land use-soil class for both the 1987 and 1995 land use maps. Whenever the area for a land use-soil class within a contributing subbasin was greater in the 1995 map than in the 1987 map, we assumed that the increase was due to new development and applied the stormwater reduction term to runoff from that new development. If the area for a land use-soil class within a contributing subbasin in the 1995 map was less than or equal to that in the 1987 map, then no stormwater treatment was applied. Based on the average treatment performance from 13 studies of Florida stormwater systems (Appendix B), it was assumed that 63% of the phosphorus load and 42% of the nitrogen load was removed by stormwater treatment. We assumed no stormwater treatment for cases in which there were increases in area of undeveloped land use categories (forest/rangeland, wetlands, and water) in the 1995 land use map.

Phosphorus and nitrogen generated in stormwater runoff were calculated using the equation

$$L_{LS} = A_{LS} \times Pr_G \times ASRC_{LS} \times C_{RL} \times ST \qquad \textbf{(18)}$$

where

L_{LS} = stormwater nutrients-generated loading from land use/soil combination
A_{LS} = area of land use/soil combination
Pr_G = annual rainfall
$ASRC_{LS}$ = runoff coefficient for land use/soil combination
C_{RL} = nutrient concentration for runoff from land use
ST = stormwater treatment (if applicable)

Losses of nutrients in transport between the runoff source and the receiving water bodies were estimated using a relationship developed by Reckhow et al. (1989), which estimates losses of sediment-associated nutrients as a function of distance between the runoff source and the receiving water body. The function developed by Reckhow et al. (1989) was based on re-analysis of data presented by Maner (1958), who measured long-term sediment accumulation in 25 reservoirs and related that to sediment losses in the surrounding watersheds. For the UORB watershed, flow paths between the centroid of contributing subbasins and the receiving lakes or major connecting tributaries were determined from 5-foot-contour coverages of the floodplains by developing a digital elevation model using Arcview 3.2 and additional extensions, including “Spatial Analyst” and “3D Analyst” (ESRI, www.esri.com), “Radiating Lines” and “Surface Analysis Tools” (Jenness Enterprises, www.jennessent.com), and “Add XY Centroid” (Mark Cederholm, script available at www.esri.com). It was assumed that there were no nutrient losses in the major connecting tributaries (Apopka-Beauclair Canal, Haines Creek, Ocklawaha River) because continuous flows and periodic high flows would limit sediment accumulation.

To apply the Reckhow et al. (1989) function to estimate transport losses requires estimates of the sediment-associated fraction of nutrients in stormwater runoff. Many of the literature sources used in development of TP and TN concentrations in runoff also included measurements of orthophosphate (PO4). A smaller number of Florida studies were located that also measured dissolved TP (TDP) and TN (TDN) in stormwater runoff (Hendrickson 1987; Fall and Hendrickson 1988; German 1989; Fall 1990; Dierberg 1991; Izuno et al. 1991; Harper and Miracle 1993a; and unpublished data for Lakes Carlton and Denham in the UORB watershed and for wetlands in St. Johns River Water Management District). In this smaller set of studies, TDP generally exceeded PO4 (13 of 16 sites, mean ratio TDP / PO4=1.6), the fraction TDN always exceeded the fraction TDP (20 sites, mean ratio fraction TDN / fraction TDP=1.4), and the fraction TDN always exceeded the fraction PO4 (16 sites, mean ratio fraction TDN / fraction PO4=2.3). So, although the most literature data are available for PO4, it appears that the fraction PO4 often substantially underestimates both fraction TDP and fraction TDN in stormwater runoff.

There were fewer land uses represented in the smaller study set and some of the studies were for drainage basins with mixed land uses, so the fractions TDP and TDN had to be averaged across similar land uses. For all land uses except water, to estimate fraction TDP, we used the maximum of the fraction PO4 from the larger set of studies or the fraction TDP from the smaller study set. The rationale for this procedure is that TDP is expected to be higher than PO4, but when there are limited or no TDP data for some land use categories, the higher fraction of PO4 should be a better estimate of TDP. Similarly, for all land uses except water, to estimate fraction TDN, we used the maximum of the fraction PO4 from the larger set of studies or the fraction TDN from the smaller study set. The rationale for this procedure is again that TDN is expected to be higher than PO4, but when there are limited or no TDN data for some land use categories, the higher fraction of PO4 should be a better estimate of TDN. For water bodies in the watershed, we used fractions TDP and TDN from SJRWMD data for Lakes Carlton and Denham. The estimated land use-specific fractions of dissolved TDP and TDN (Table 12) are generally consistent with ranges given for a limited number of land uses by Wanielista and Yousef (1992).

No transport losses were assumed for the dissolved fraction of nutrients in runoff, but the Reckhow et al. (1989) relationship was applied to the sediment-associated fraction (converse of the fractions in Table 12). Appendix C shows the estimated sediment and nutrient delivery ratios (fraction of sediments or nutrients in runoff transported to receiving lake) for each of the UORB contributing subbasins. The mean delivery ratios (sediments, 0.22; phosphorus, 0.66; and nitrogen, 0.79) are quite similar to delivery ratios reported in City of Tallahassee and ERD 2002 (sediments, 0.2; phosphorus, 0.6; and nitrogen, 0.85), which were reportedly based on previous experience by ERD in other parts of Florida.

Muck Farms and Restoration Areas

Several muck farms were operating in the early 1990s before acquisition by SJRWMD, and three continue to operate (Hurley Farm in the Lake Beauclair watershed, Ja-Mar Farm in the Lake Harris watershed, and Springhill Farm in the Lake Eustis watershed). Discharges from operating muck farms were estimated by using a multiple regression equation developed in Fulton 1995a, which relates discharge volumes to area in production, rainfall, and evaporation. Permit records for the Ja-Mar Farm included information on pump discharges during 1994–96. During that period, the estimated discharges from pump records exceeded those calculated by the multiple regression equation by about 21%, so the regression discharge estimates for that farm were adjusted upward by that percentage. In most cases, nutrient concentrations in farm discharges were taken from monitoring data included in permit records. An exception was made for several of the former farms operating in the Emeralda Marsh Conservation Area (EMCA). There were no permit records covering the periods these farms operated after 1991, and nutrient concentrations found on the sites after SJRWMD acquisition were substantially higher than reported in farm permit records for the 1980s. Therefore, for periods the EMCA farms operated after 1991, we used the average nutrient concentrations found in the first 2 years of SJRWMD monitoring after acquisition of the properties.

Different methods were used to estimate discharges from restoration areas, depending on how the sites were operated. There were nine known periods in which water was pumped out of the Lake Harris Conservation Area (LHCA) into Lake Harris, and one period in which water was pumped from the Knight-South restoration area into Lake Griffin prior to its operation as the Lake Griffin flow-way. We estimated the volumes discharged during these pump events from records of water levels, topographic information for the sites, and estimates of rainfall and evaporation during the period of discharge. Water quality data collected concurrently with the pump discharges were used to estimate nutrient discharges. Some pump discharges from the LHCA occurred prior to 1994, but there was insufficient information to estimate discharges during that period. Therefore, we estimated nutrient discharges from the LHCA during 1991–93 as the average annual discharge determined for the years 1994–2000.

Net phosphorus discharges from the Lake Griffin flow-way were estimated from frequent measurements of intake and discharge TP concentration and discharge volumes. During flow-way operational periods, intake volumes were assumed to be equal to discharge volumes.

Other restoration areas in the EMCA and the Pine Meadows Restoration Area did not have pump discharges, but for some periods of time did have open connections with adjacent water bodies through which runoff could occur. Discharges from these areas were estimated similarly to estimates of stormwater runoff from other land uses, with a few modifications. First, since we had water quality data for these sites, we used those data for discharge concentrations rather than the concentrations listed in Table 12. Beginning in 1996, the Springhill muck farm discharged into Pine Meadows, so the total discharge volumes from Pine Meadows included both the pump discharges from Springhill and the estimated runoff from Pine Meadows. It was assumed that there were no discharges from the EMCA properties to Lake Griffin during the initial period after SJRWMD acquisition until the sites were reconnected to the lake and water levels on the sites increased to lake level. After reconnection, it was assumed that water levels in the EMCA sites fluctuated in parallel with water levels in Lake Griffin. For some time periods, some of the EMCA properties were disconnected from Lake Griffin because water elevations were below the invert elevations of the connecting culverts.

During these periods, it was assumed that there were no discharges to Lake Griffin.

The same procedures used to estimate nutrient losses in transport of stormwater runoff were applied to estimate losses of nutrients in transport from the muck farms and restoration areas. The only changes were to the fraction of the nutrient discharge that was sediment-associated. For the restoration areas, we used the average percentages of particulate and dissolved phosphorus and nitrogen from SJRWMD water quality data. Except for one site that had formerly been used for pasture rather than for row crop agriculture, the percentage of particulate phosphorus was about 20%, and we used this percentage for estimating transport losses in discharges from the operating muck farms.

Septic Systems

Nutrient loading from septic systems was estimated similarly to methods used by Fulton (1995a). Only septic systems located within 200 meters of the lakes, lakeshore wetlands, or canals connecting to the lakes were assumed to contribute nutrients to the lakes. Counts of structures within this zone were made from 1995 digital ortho quads and older aerial photos, excluding areas known to be served by municipal sewage treatment plants. Also, several package sewage treatment plants were located within the 200-meter buffer zone, and it was assumed that effluents from these plants also reached the lakes, so counts were made of houses within the service areas of those plants. Loading estimates from septic tanks and package treatment plants used the same per capita nutrient load and soil retention of nutrients as used by Fulton (1995a).

Point Sources

Nutrient loading was estimated for several point sources, including surface (weak waste) discharges from citrus processing plants, spills from citrus processing plants, runoff from waste disposal areas for citrus processing plants and municipal waste treatment plants, and municipal waste spills. Information on these sources was obtained primarily from FDEP permit files. These records are incomplete, so it was assumed that discharges during periods for which there were no records were the same as for periods in which discharges were reported.

Three citrus processing plants occur in the UORB. Cutrale Citrus discharges weak wastes and has a sprayfield for strong wastes discharging to Lake Griffin. Golden Gem discharges weak wastes to Lake Yale, but its sprayfield for disposal of strong wastes is located outside of the contributing subbasins for the UORB lakes. Silver Springs Citrus has one of its two sprayfields within the contributing basin for Lake Harris, but its weak waste discharges are located outside of the contributing subbasins for the UORB lakes. Municipal waste treatment plants in the basin include those for Leesburg, Eustis, Tavares, Mount Dora, and Umatilla, and dispose of treated wastes to sprayfields, rapid infiltration basins, or percolation ponds.

Nutrient loading from surface discharges from each citrus processing plant was estimated by multiplying the nutrient concentration in weak waste discharges by the reported discharge volume, using averages for periods for which no data were reported.

Methods for estimating runoff from waste disposal areas for citrus processing plants and municipal waste treatment plants were similar to those used for other stormwater runoff. Areas for waste disposal were mapped. Total flow to the disposal areas was determined as the sum of reported discharges and estimated rainfall to the sites. Runoff coefficients were taken from the most similar land use in Table 12. The only citrus processing plant that reported nutrient concentrations for runoff from sprayfields was Cutrale Citrus, so those concentrations were also used for the other citrus plants. The only municipal waste treatment plant that reported nutrient concentrations for runoff from sprayfields was Leesburg, so those concentrations were also used for the other municipal plants. No stormwater treatment was applied to the runoff from municipal or industrial sprayfields because the reported nutrient concentrations are apparently for discharges leaving the sites, after any stormwater treatment. The only reported data on discharges from waste disposal areas during 1991–2000 were for discharges from the Leesburg sprayfield during part of 1998–99. These reported discharges averaged 34,000 cubic meters per month and were very similar to our estimates for discharges from the sprayfield during that period using the above methods (36,000 cubic meters per month). We conclude that our methods provide a reasonable estimate of runoff discharges from waste disposal areas.

The reported locations of municipal and industrial spills were mapped from descriptions given in permit records. In a few cases, the spill locations could not be determined, but these spills represented a negligible volume and phosphorus load. Also, in several cases, the volume spilled was not reported, so no estimates of discharges could be made for those spills. Almost all of the municipal spills appeared to be of untreated wastes, so nutrient concentrations in those spills were estimated from typical concentrations given for untreated domestic wastewater in Tchobanoglous and Burton 1991. There were a limited number of reported industrial spills, although the only ones that included estimates of spill volumes were for Cutrale Citrus. For estimates of discharges from these spills, we used the nutrient concentrations reported for one of the spills.

The same procedures used to estimate nutrient losses in transport of stormwater runoff were applied to estimate losses of nutrients in transport from the point sources. Estimates of fraction of dissolved nutrients in weak waste discharges and sprayfield runoff were taken from the most similar land use in Table 12. No transport losses were assumed for spills from Cutrale Citrus since they were reported to discharge directly into Lake Griffin. For domestic wastewater spills, the fractions of dissolved nutrients were estimated from fractions of inorganic phosphorus (0.68) and nitrogen (0.6) reported in Tchobanoglous and Burton 1991.

ESTABLISHMENT OF DESIRED WATER QUALITY FROM NATURAL BACKGROUND CONDITIONS AND MANAGEMENT GOALS

Two approaches were examined to estimate natural background phosphorus concentrations in the UORB lakes:

- Modeling of external loading and water quality under natural background conditions
- Inferences of reference conditions from regional lakes

We also examined estimates of historic phosphorus concentrations in the UORB lakes made from diatom microfossils in the lake sediments (Schelske 1998; Schelske et al. 1999, 2001). However, for four of the seven lakes, the estimates from diatom microfossils of historical phosphorus concentrations were similar to or exceeded existing phosphorus concentrations. These estimates of historical phosphorus concentrations are inconsistent with substantial historical increases in phosphorus accumulation rates found in the same studies, which are indicative of increases in external loading of phosphorus and of phosphorus concentrations in the lakes (Schelske 1998; Schelske et al. 1999, 2001). Therefore, we decided that the diatom-based estimates of historical phosphorus concentrations were unreliable and did not use them in estimating natural background water quality.

Modeling of External Loading and Water Quality Under Natural Background Conditions

Model predictions of natural background phosphorus concentrations for the UORB lakes were slightly revised from those reported in Fulton 1995a. The modeling used variations of a mass-balance model (Vollenweider 1969), which treats a lake as a continuously stirred tank reactor, with phosphorus inflows, outflows, and losses to the sediments, and predicts concentrations of phosphorus at steady-state. Four variations of this model were used, which differ in how the key model parameter, the sedimentation coefficient, is determined. Models developed by Reckhow (1991) and Canfield and Bachmann (1981) predict the sedimentation coefficient as partially a function of the external phosphorus load, while the models developed by Larsen and Mercier (1976) and Salas and Martino (1991) predict the sedimentation coefficient as a function of hydraulic detention time.

Each of these four models was applied to predict phosphorus concentrations for the existing hydrology and nutrient loading for the 5-year period from 1986–90. We then selected the model that best predicted the reported phosphorus concentrations for that period and applied that model to estimates of phosphorus loading under natural background conditions. This application of the model predictions to natural background conditions assumes that the natural background conditions were similar to existing conditions, except for nutrient loading. Natural background phosphorus loading was estimated by

- Using a phosphorus concentration for discharges into the basin from upstream Lake Apopka of 40 (µg/L), the midpoint of the most probable range of antecedent conditions determined for that lake (Lowe et al. 1999)
- Converting all existing land uses in the basin to either forest/rangeland or wetlands for estimating stormwater runoff from within the watershed
- Eliminating all point source and septic tank discharges

Finally, we adjusted the model predictions to account for errors in prediction of reported phosphorus during 1986–90. For example, if the model underestimated reported phosphorus concentrations by 20%, it was assumed that it would also underestimate natural background phosphorus concentrations by 20%. Percent errors in prediction of 1986–90 phosphorus concentrations tended to be greater in lakes with lower concentrations (Table 13, Appendix D). This is in part because a small absolute error represents a larger percentage error if actual concentrations are low. For example, if the existing phosphorus concentration was 15 µg/L, an overestimate of 5 µg/L would be a 33% error, while if the existing concentration was 200 µg/L, a similar overestimate of 5 µg/L would be only a 2.5% error. The substantial overestimate of phosphorus concentration in Lake Yale may have been because the lake was experiencing a significant *Hydrilla* infestation at that time (Hestand et al. 1991). Utilization of nutrients by the substantial amount of submersed aquatic vegetation in Lake Yale may have resulted in lower nutrient water column concentrations than predicted by the trophic state models (Canfield et al. 1983). In support of this explanation, phosphorus concentrations in Lake Yale nearly doubled during the 1990s, as aquatic vegetation was eliminated by introduced grass carp, increasing from an average of 17 µg/L during 1986–90 to 32 µg/L during 1997–2000.

Table 13. Errors in model predictions of Upper Ocklawaha River Basin lake total phosphorus (TP) concentrations

Lake	Mean TP 1986–90 (µg/L)	Mean Percent Error in Prediction of 1986–90 TP	Best-Fitting Model
Beauclair	224	+2.4	Larsen and Mercier (1976)
Dora	124	–5.9	Larsen and Mercier (1976)
Eustis	52	+0.1	Reckhow (1991)
Harris	35	–23.7	Reckhow (1991)
Griffin	87	+19.2	Canfield and Bachmann (1981)
Yale	17	+105.3	Reckhow (1991)
Weir	12	–37.8	Reckhow (1991)

Note: µg/L = micrograms per liter

Inferences of Reference Conditions From Regional Lakes

EPA has defined reference conditions as the natural, least impacted, or best attainable conditions, and if a known population of true reference lakes has not been established, recommends that reference conditions be determined as the 25th percentile of a regional population of lakes (EPA 2000a; 2000b). We used three estimates of phosphorus concentrations under reference conditions for the UORB lakes. None of these reference data sets remove lakes that may have been impacted by eutrophication and thus do not represent a population of true reference lakes, so we followed the EPA recommendation that the 25th percentile is the best estimate of the reference phosphorus concentration.

Two of the estimates of reference phosphorus concentrations were based on lake regions determined to be geologically or geographically similar to the UORB lakes.

- The first estimate used a detailed ecoregional map developed for Florida lakes by EPA and the Florida Lakewatch program (Griffith et al. 1997). The UORB lakes were included in two ecoregional groupings. Lake Weir was included in group 75-14, Lake Weir-Leesburg upland lakes (including 11 lakes), and the remaining UORB lakes were included in group 75-08, Central Valley lakes (including 40 lakes). Data files with statistical summaries of water quality data from lakes in these ecoregions were obtained from Florida Lakewatch. We used the 25th percentile of these ecoregion groups as the estimate of the reference phosphorus concentration.

- The second estimate used a data set for ecoregions within SJRWMD assembled by John Hendrickson (n.d.). The UORB lakes were included in two ecoregional groupings. Lake Weir was included in the group Plio-Pleistocene Ridge seepage lakes (including 149 lakes), and the remaining UORB lakes were included in the group Exposed Miocene lakes (including 54 lakes). We again used the 25th percentile of these ecoregion groups as the estimate of the reference phosphorus concentration.

The third estimate of phosphorus concentrations under reference conditions selected lakes with similar morphology and hydrology to the UORB lakes, using a data set of Florida lakes compiled by Huber et al. (1982). The morphological and hydrological characteristics selected are among the primary determinants of lake water quality. Using procedures similar to those used by Lowe et al. (1999), subsets of lakes were selected that met four criteria:

1. Ratio of drainage area/surface area no more than 10 times and no less than one-tenth that of the lake of interest
2. Average depths no more than two times and no less than one-half that of the lake of interest

3. Surface area no less than one-tenth and no more than 10 times that of the lake of interest
4. Hydraulic detention time no less than one-fifth and no more than five times that of the lake of interest

Application of these criteria resulted in lake subsets ranging in number from 8 (Lake Weir) to 58 (Lake Dora). We again used the 25th percentile of these lake subsets as the estimate of the reference phosphorus concentration.

Although we did not attempt to model other water quality parameters under natural background conditions, we did use the reference lakes approaches to estimate natural background TN and chlorophyll-*a* concentrations and Secchi depths for the UORB lakes.

Integration of Different Estimates of Natural Background Phosphorus Concentrations

The two ecoregional approaches were averaged to provide an estimate of natural background phosphorus concentrations from geologically and geographically similar lakes. An overall estimate of natural background phosphorus concentrations for the UORB lakes was then determined by averaging the estimates from geologically and geographically similar lakes, hydrologically and morphologically similar lakes, and modeling of natural background basin conditions.

Selection of Phosphorus Criteria for the UORB Lakes

For development of interim PLRGs for the UORB lakes, we elected to use an existing water quality standard for water transparency published in Chapter 62-302.530, *F.A.C.*: “Depth of the compensation point for photosynthetic activity shall not be reduced by more than 10% as compared to the natural background value.”

Having used the above methods to estimate natural background phosphorus concentrations in the UORB lakes, in order to apply the water quality standard, we required a method to relate phosphorus concentration to water transparency. Water transparency was measured in the UORB lakes between 1998 and 2001 using a LiCor-1000 meter, which measures photosynthetically active radiation (PAR) using a combination of an on-deck sensor and an underwater spherical sensor. Extinction coefficients were calculated from a vertical profile of underwater measurements taken at 10-cm-depth intervals from the surface to the bottom of the water column.

We conducted a regression analysis of data on phosphorus and water transparency (measured as the light extinction coefficient) (Figure 3). The regression analysis used only samples with total phosphorus concentrations

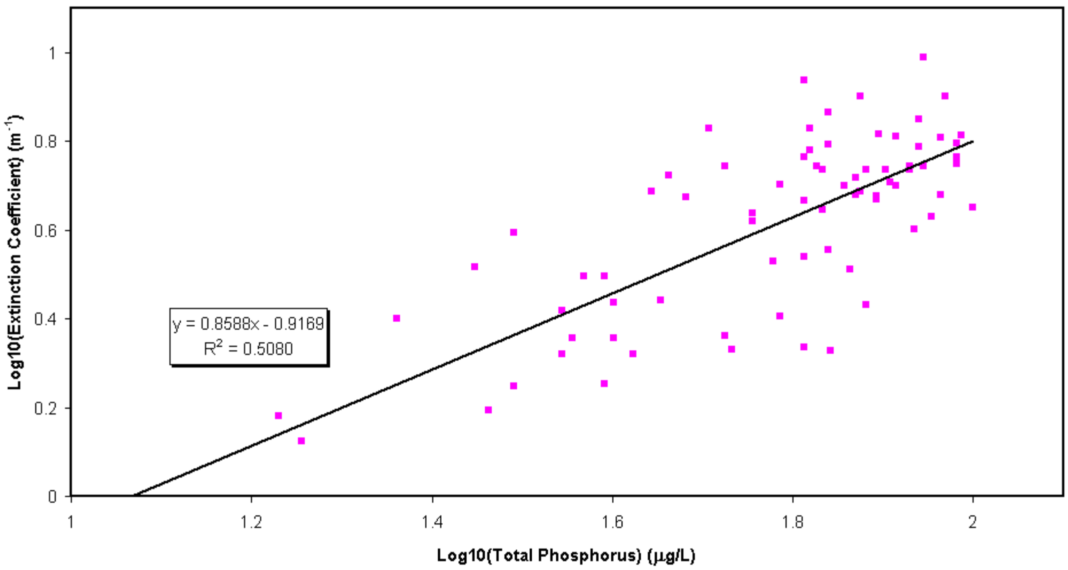

Figure 3. Upper Ocklawaha River Basin lakes, relationship between log10(Extinction Coefficient) and log10(Total Phosphorus, TP<100 µg/L)

less than 100 µg/L, because of evidence that phosphorus is not limiting to phytoplankton growth at concentrations above that level (Brown et al. 2000). The measurements in Figure 3 include a wide range in water quality. However, we have no light extinction measurements for Lake Weir, and the lowest TP measurement included in Figure 3 is 17 µg/L, which is slightly above the average concentration in Lake Weir. We would prefer to have more observations at low TP levels, but we used the best data set available. Presumably, water color and inorganic turbidity would be relatively more important in influencing transparency when chlorophyll-*a* is very low than they would be at higher chlorophyll-*a* concentrations. However, this fact would not influence the slope of the relationship in Figure 3 unless the magnitude of water color or inorganic turbidity changed systematically as chlorophyll-*a* declined.

The compensation depth was determined from the extinction coefficient using the Lambert-Beer equation. Using these procedures, a 10% reduction in the compensation depth corresponds with a 13% increase in phosphorus concentration. Therefore, the recommended phosphorus criterion for interim PLRGs for the UORB lakes is 13% higher than the estimated natural background concentration in the lakes.

LAKE WATER QUALITY MODELING TO ESTABLISH PHOSPHORUS LOADING LIMITS TO MEET THE WATER QUALITY GOALS

In order to predict lake trophic responses to nutrient loading, lake morphometric and hydrologic data are required, including lake surface area, volume or mean depth, and hydraulic detention (residence) time. The lake areas, volumes, and mean depths reported in Table 1 are for the reference elevation. In calculation of detention times, volumes used were determined from mean lake elevations by quadratic regressions fit to hypsographic data for the lakes.

Hydraulic detention times can be calculated as mean lake volume divided by estimated water losses or inflows (Reckhow and Chapra 1983). Detention times were calculated for all of the lakes using water losses (tributary discharges plus leakage losses), because these were felt to be generally more accurate than the estimates of water inflows (net precipitation, tributary inflows, spring discharges, and watershed runoff, adjusted for changes in lake volume). For Lakes Yale and Weir, water losses are dominated by leakage losses, which were estimated as a long-term average, with unknown accuracy; so for these lakes, we also calculated detention times using estimated water inflows.

From the nutrient budgets, nutrient retention coefficients and net sedimentation coefficients were calculated for the UORB lakes. The difference between external nutrient loading and losses through downstream discharge and leakage is the net retention / export within the lakes. The nutrient retention coefficient is the net retention / export of nutrients divided by the external nutrient loading. The net sedimentation coefficient (σ) was calculated as the mean net deposition of nutrients to the sediments divided by the mean lake water mass of nutrients:

$$\sigma = \frac{Nutrient_{load} - Nutrient_{loss} - \Delta NutrientStorage_{water}}{MeanNutrientMass_{water}} \qquad \textbf{(19)}$$

The change in nutrient storage was determined from estimates of nutrient masses in lake water at the beginning and end of the time period of interest.

For the interim PLRG recommendations (Fulton et al. 2003), we made the simplifying assumption that phosphorus concentration in the lakes is directly proportional to external phosphorus loading. Mathematically, this is equivalent to a relationship between external load and lake nutrient concentration that is linear, with an intercept of zero (i.e., when the external load is zero, then the lake concentration would also be zero). In terms of the Vollenweider (1969) mass-balance model, this is equivalent to assuming that there will be no change in the sedimentation coefficient from existing conditions. This is a common assumption in cases where the sedimentation coefficient is empirically determined, including modeling used for PLRG development in Lake Apopka (Lowe et al. 1999; Coveney 2000) and the Upper St. Johns River Basin lakes (Keenan et al. 2002). This assumption is also used in versions of the model in which the sedimentation coefficient is predicted as a function of hydraulic detention time (e.g., Salas and Martino 1991). This assumption is supported by lake responses to reductions in external phosphorus load, in which the percent reduction in phosphorus concentrations in the lakes was roughly equivalent to the percent reduction in the external load (Sas 1989; Battoe et al. 1999).

We examined the applicability of the assumption of direct proportionality between external loading and in-lake nutrient concentrations for the UORB lakes using variations of the mass-balance model (Vollenweider 1969). The models were used to predict 10-year (or 8-year) average phosphorus or nitrogen concentrations for the lakes. It is most appropriate to use these models to predict long-term means, and not annual or seasonal variations, because the models were developed to predict nutrient concentrations at equilibrium. Water quality modeling was conducted for average external loading and hydrological conditions for the 10-year period 1991–2000, with two exceptions. The 8-year period 1993–2000 was used for Lake Griffin because of concerns about the accuracy of estimated nutrient loadings from the operating Emeralda muck farms during 1991–92. Also, the 8-year period 1991–98 was used for Lake Weir, because the hydrological calculations used in some of the models could not determine a hydraulic detention time for the full 10-year period (see further below).

A variety of models was applied to predict in-lake phosphorus and nitrogen concentrations, some of which assume direct proportionality with external loading and some of which assume nonlinear relationships. These include several models implemented in spreadsheet calculations and several models available in a DOS program, Bathtub (Table 14). Bathtub includes a set of models developed primarily for reservoirs by Walker (1999) for the USACE Waterways Experimental Station. Both the spreadsheet and the Bathtub models are modifications of Vollenweider (1969), in which a lake is considered a continuously stirred system with nutrient inflows, discharges, and losses to the lake sediments. The models differ most fundamentally in the factors that determine the sedimentation coefficient, and secondarily in model coefficients applied to fit the models to lake data sets.

The highlighted models in Table 14 assume that in-lake nutrient concentrations are directly proportional to external loading. The other models predict a nonlinear relationship, either because the sedimentation coefficient is a function of the external nutrient load (P3, P4, P10, N1, and N5) or because of exponents in the equations predicting nutrient concentrations (P2, P8, P9, N3, and N4).

There is a significant difference between the hydrological calculations used between the spreadsheet and Bathtub models. As discussed above, the spreadsheet calculations generally used estimated water losses from the lakes (discharge and leakage) to determine hydraulic detention time, with the exception of Lakes Yale and Weir, for which detention times were calculated using both water losses and inflows. Bathtub takes a different approach in the hydrological calculations. Bathtub estimates a corrected outflow from an input-output budget. Specifically:

$$P + I = O + E + \Delta V + A \qquad \textbf{(20)}$$

$$\text{Total outflows} = O + A \qquad \textbf{(21)}$$

where

P = rainfall to lake surface
I = inflows (tributaries, watershed runoff, and spring discharges in our water budgets)
O = measured or estimated outflows (tributaries and leakage losses)

Table 14. Water quality models applied to estimate nutrient concentrations in the Upper Ocklawaha River Basin lakes

Bold = *model predictions directly proportional to external nutrient loading*

Model Number	Description	Factors Affecting Sedimentation Coefficient
Spreadsheet Phosphorus Models		
P1*	Vollenweider (1969)	Detention time
P2	OECD (Vollenweider and Kerekes 1982)	Detention time
P3	Eutromod Florida lakes model (Reckhow 1991)	Detention time, depth, P load
P4	Canfield and Bachmann lakes model (1981)	Depth, P load
P5	Larsen and Mercier (1976) with coefficients recalculated by Canfield and Bachmann (1981)	Detention time
P6	Salas and Martino (1991)	Detention time
P7	Lake P concentration = inflow P concentration	No net sedimentation
Bathtub Phosphorus Models		
P8	Second-order, available P (default)	Detention time, depth
P9	Second-order	Constant
P10	Canfield and Bachmann reservoir model (1981)	Depth, P load
P11*	Vollenweider (1976)	Detention time
P12	Simple first-order	Constant
P13	First-order settling	Depth
Spreadsheet Nitrogen Models		
N1	Eutromod Florida lakes model (Reckhow 1991)	Detention time, depth, N load
N2	Lake N concentration = inflow N concentration	No net sedimentation
Bathtub Nitrogen Models		
N3	Second-order, available N (default)	Detention time, depth
N4	Second-order	Constant
N5	Bachmann (1980) volumetric load	Depth, N load
N6	Bachmann (1980) flushing rate	Detention time
N7	Simple first-order	Constant
N8	First-order settling	Depth

Note: N = nitrogen
P = phosphorus

*P11 appears to be identical to P1, except for differences in hydrological calculations between the spreadsheet and Bathtub models

E = evaporation
ΔV = change in lake storage volume
A = "advective outflows"—a term introduced to balance outflows with inflows

Thus, Bathtub assumes that outflows are measured least accurately (which is the opposite of the assumption made in the spreadsheet models) and assigns all of the error in the water balance to the outflows. Bathtub then calculates hydraulic detention time (DT) as

$$\text{DT} = \text{lake volume} / (\text{total outflows} + \Delta V) \qquad \textbf{(22)}$$

Thus, the Bathtub calculation of detention time adjusts the outflows for volume change. Since the volume of all of the lakes decreased during the 1991–2000 modeling period, incorporation of a negative volume change in the calculation results in a longer detention time than calculated in the spreadsheet models. For Lake Weir during the 1991–2000 period, ΔV was negative (there was a decrease in lake volume) and the volume decrease exceeded the Bathtub calculation of total outflow, resulting in a negative value for detention time. Therefore, for Lake Weir, the water quality modeling was done for the 8-year period 1991–98, for which the estimated detention time was positive.

RESULTS

WATER BUDGET FOR THE UORB LAKES

For the Burrell Basin lakes and Lake Griffin, the major water sources during 1991–2000 were rainfall and stream inflows, while the major losses of water were evaporation and stream discharges (Table 15). Lakes Yale and Weir lacked significant tributary flows, so the major water budget components were rainfall and evaporation. The error term was negative for three of the lake basins. If the estimates of change in lake storage volume are accurate, a negative error term indicates that either water sources were overestimated or losses were underestimated in the water budgets. However, for all basins, the error term was small relative to the major water budget components.

Table 15. Estimated water budgets, 1991–2000, for the Upper Ocklawaha River Basin, expressed in hectare-meters per day

Source	Burrell Basin	Lake Griffin Basin	Lake Yale Basin	Lake Weir Basin
Change in lake storage volume (ΔV)	–1.3	–0.5	–0.3	–0.4
Rainfall on lake (P)	42.0	18.0	6.9	7.1
Lake evaporation (E)	–41.0	–18.2	–7.0	–7.6
Leakage from lake to aquifer (L)	–7.3	-5.8	–1.6	–1.8
Stream discharge from lake (O)	–44.5	–46.2	–0.1	0
Stream inflow into lake (I)	25.7	44.5	0	0
Spring discharge into lake (S_i)	4.6	0	0	0
Runoff from watershed to lake (R)	22.1	8.9	2.9	1.5
Error term (Er)	–2.9	–1.7	–1.4	0.4

The potential for significant error in the water budgets is greatest with the major components, evaporation, rainfall, and stream discharges. There was large variation between reported annual pan evaporation at the two weather stations, Lisbon (142 cm / yr) and Lake Alfred (192 cm / yr), and there is considerable uncertainty in estimating evapotranspiration losses from lakes and wetlands from pan evaporation data. There was also considerable variation in reported rainfall at basin stations. For example, the coefficient of variation (standard deviation as percent of mean) for annual rainfall at Griffin-Yale Basin stations averaged 13.1%, with a range from 4.9% to 32.1%.

The accuracy of reported discharges at most of the basin structures is considered by USGS to be fair (95% of daily discharges are within 15% of their true values), but is considered to be poor (records do not meet the above criterion) at the Burrell Lock and Dam on Haines Creek.

Estimated mean hydraulic detention times for the UORB lakes for the 1991–2000 period ranged from 0.12 year for Lake Beauclair to 21.79 years for Lake Weir (Table 16). The long detention times for Lakes Yale and Weir are due to the absence of significant tributary inflows or outflows. Estimated mean detention times using water losses were longer for Lake Yale than using water inflows, but shorter for Lake Weir. Detention times could not be calculated using water inflows in some individual years for Lake Weir, because in some years net precipitation was negative and the net precipitation deficit exceeded other estimated inflows, so the estimated detention time became a negative value.

Table 16. Estimated hydraulic detention times for the Upper Ocklawaha River Basin lakes

Mean values were calculated as mean lake volume divided by average annual inflows or losses for the 1991–2000 period. Maximums and minimums are for individual years.

Lake	Detention Time (years)		
	Mean	Maximum	Minimum
Beauclair	0.12	0.44	0.06
Dora	0.65	2.16	0.34
Harris	2.89	7.14	1.07
Eustis	0.60	2.28	0.25
Griffin	0.49	1.88	0.21
Yale—water losses	10.36	13.27	7.88
Yale—water inflows	5.39	14.87	4.15
Weir—water losses	16.85	17.51	15.88
Weir—water inflows	21.79	NA	NA

Note: NA = not available

EXISTING WATER QUALITY

There was nearly a 15-fold variation in phosphorus concentrations in the UORB lakes during 1991–2000, ranging from 209 µg/L in Lake Beauclair to 14 µg/L in Lake Weir (Figure 4). There was an even larger 20-fold variation in chlorophyll-*a* concentrations. Lake Beauclair had a considerably higher mean phosphorus concentration than Lakes Dora and Griffin, but all three lakes had relatively similar mean chlorophyll-a concentrations.

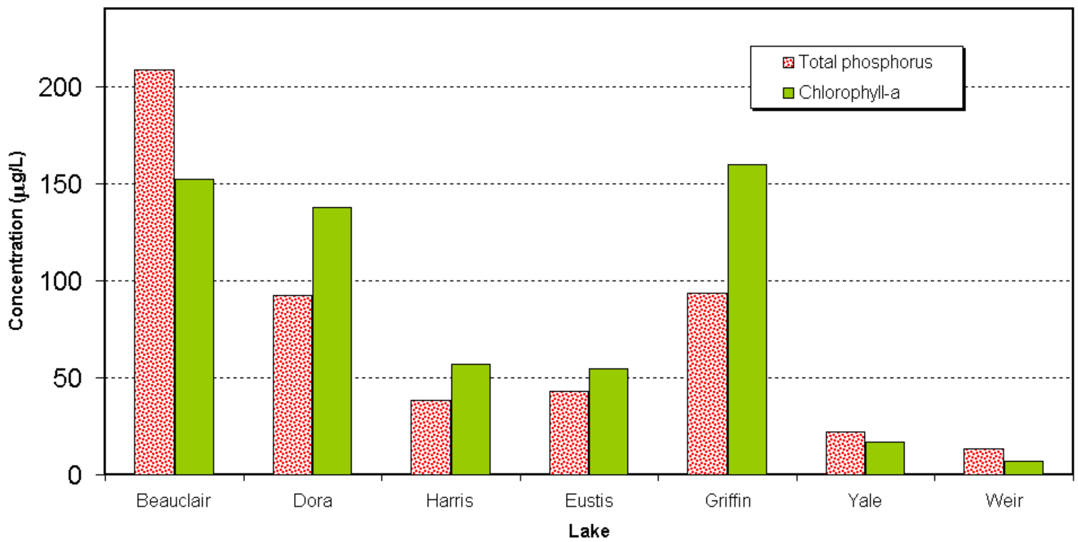

Figure 4. Average total phosphorus and chlorophyll-*a* concentrations in the Upper Ocklawaha River Basin lakes, 1991–2000

Over the period of record for our data sets (since mid-1980s for Lakes Weir and Yale, and since mid-1970s for the other lakes), there have been generally declining or stable trends in phosphorus concentrations, with the exception of Lake Yale (Figure 5). However, chlorophyll-*a* concentrations have increased since the mid-1990s in several of the lakes (Figure 6). There has also been a decrease in Secchi depth transparency in several of the lakes since the mid-1990s (Figure 7). The increased chlorophyll-*a* concentrations and decreased Secchi depths appear to be related to development of blooms of the nitrogen-fixing cyanobacterium *Cylindrospermopsis raciborskii*, a species that was not reported to occur in the lakes prior to the 1990s.

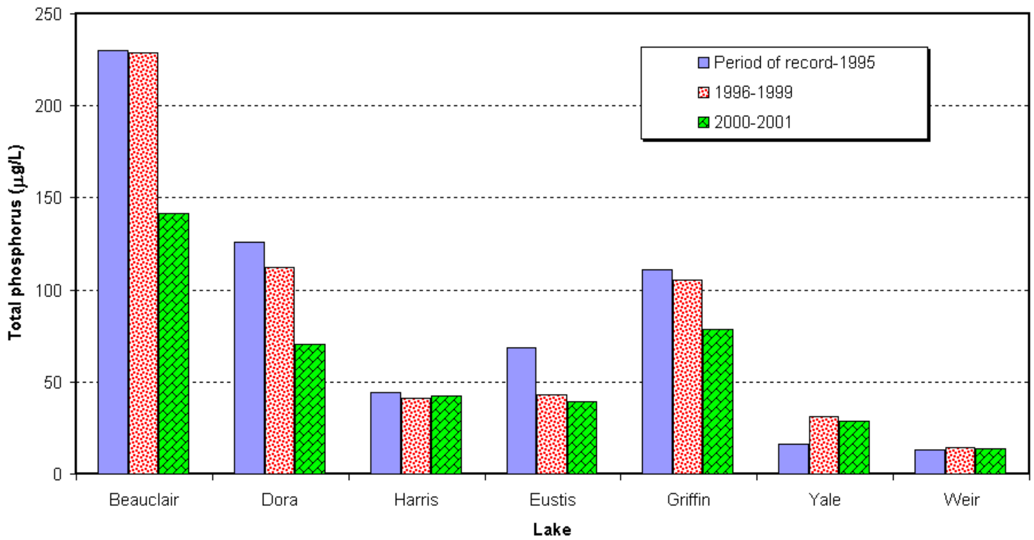

Figure 5. Temporal trends in phosphorus concentrations in the Upper Ocklawaha River Basin lakes

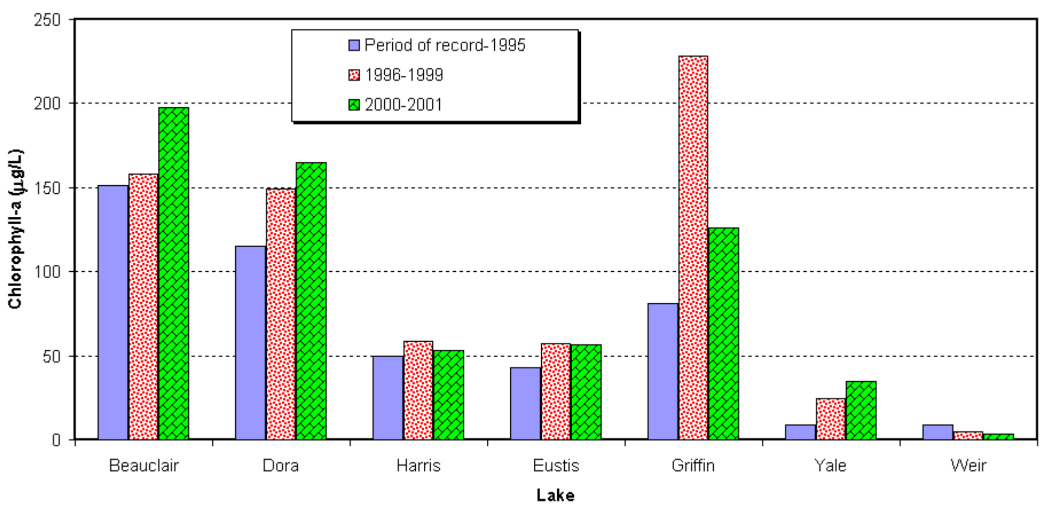

Figure 6. Temporal trends in chlorophyll-*a* concentrations in the Upper Ocklawaha River Basin lakes

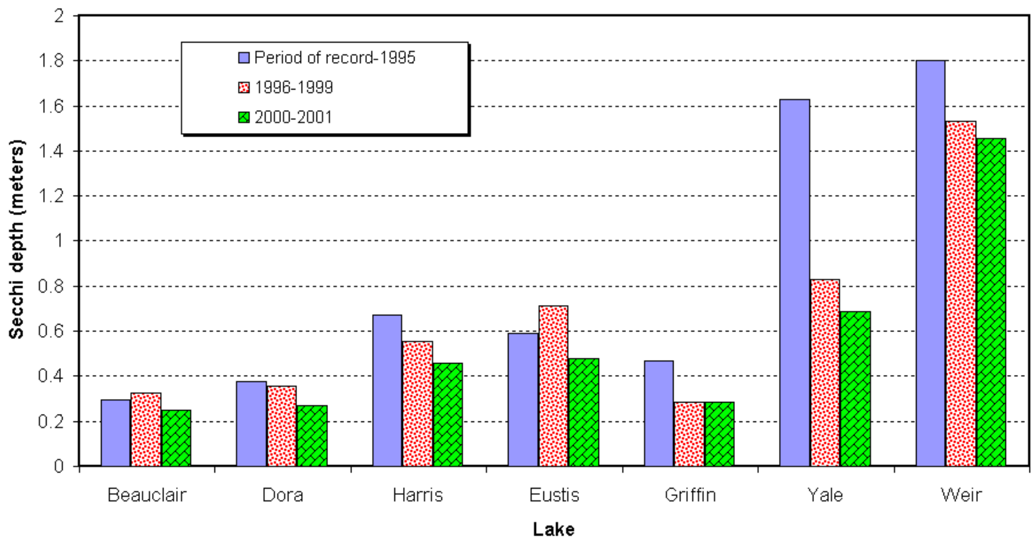

Figure 7. Temporal trends in Secchi depth transparency in the Upper Ocklawaha River Basin lakes

Although variability is high, there appears to be a decreasing trend in chlorophyll-*a* concentrations in the UORB lakes below phosphorus concentrations of about 100 µg/L (Figure 8). This trend is more easily seen from a statistical summary, grouping the data in 10 µg classes up to a phosphorus concentration of 100 µg/L, and in 100 µg classes above that level (Figure 9).

A similar statistical summary of Secchi transparency data indicates a trend toward increasing transparency below phosphorus concentrations of about 80 µg/L (Figure 10).

ESTIMATED EXTERNAL PHOSPHORUS LOADING

Basin Overview

For presentation, in the main text and figures, phosphorus sources are aggregated into a reduced number of categories. More detailed phosphorus and nitrogen budgets are given in Appendix E.

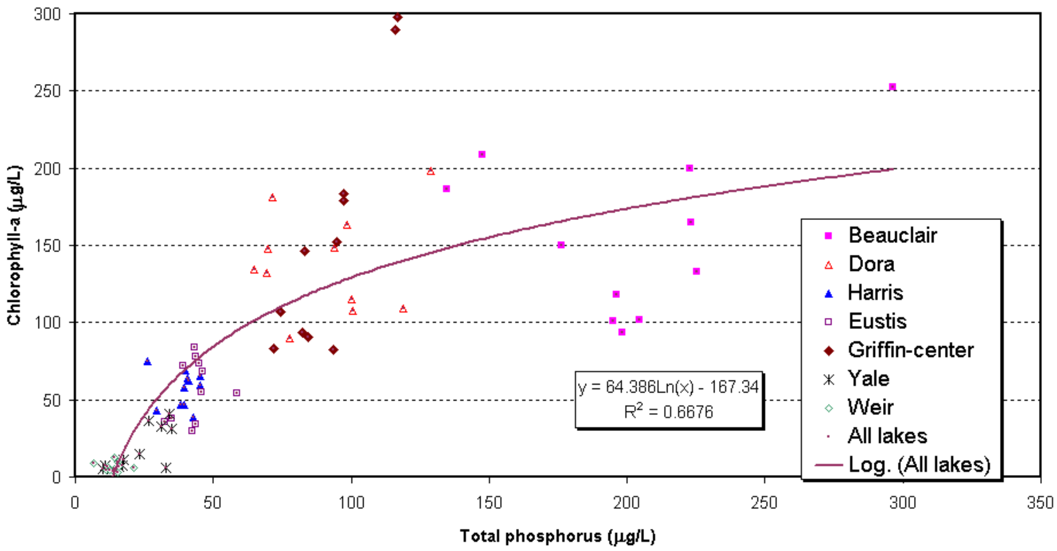

Figure 8. Upper Ocklawaha River Basin lakes, relationship between annual means of total phosphorus and chlorophyll-*a* concentrations

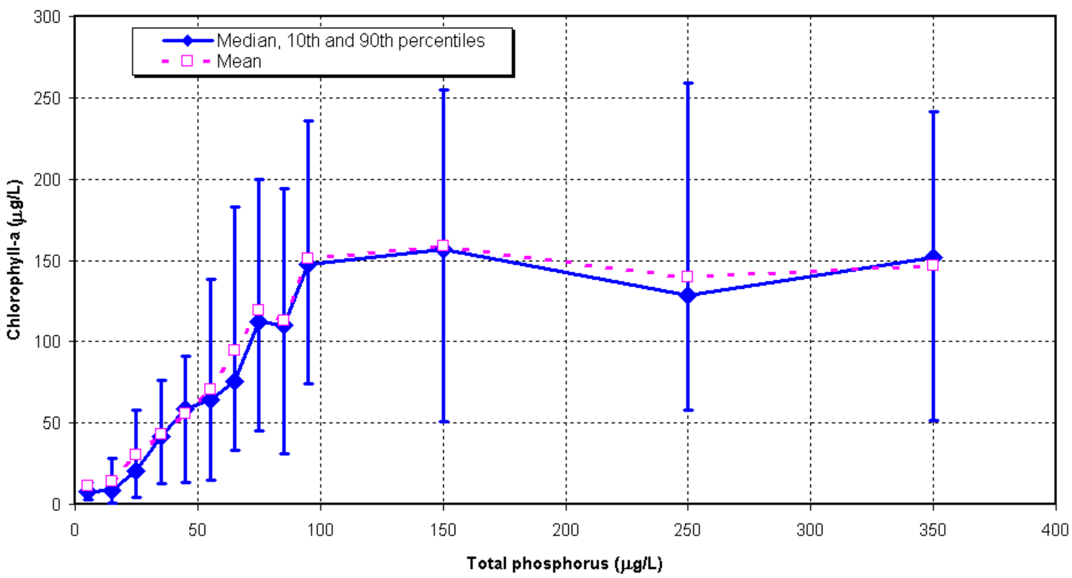

Figure 9. Upper Ocklawaha River Basin lakes, statistical summary of the relationship between total phosphorus and chlorophyll-*a* concentrations

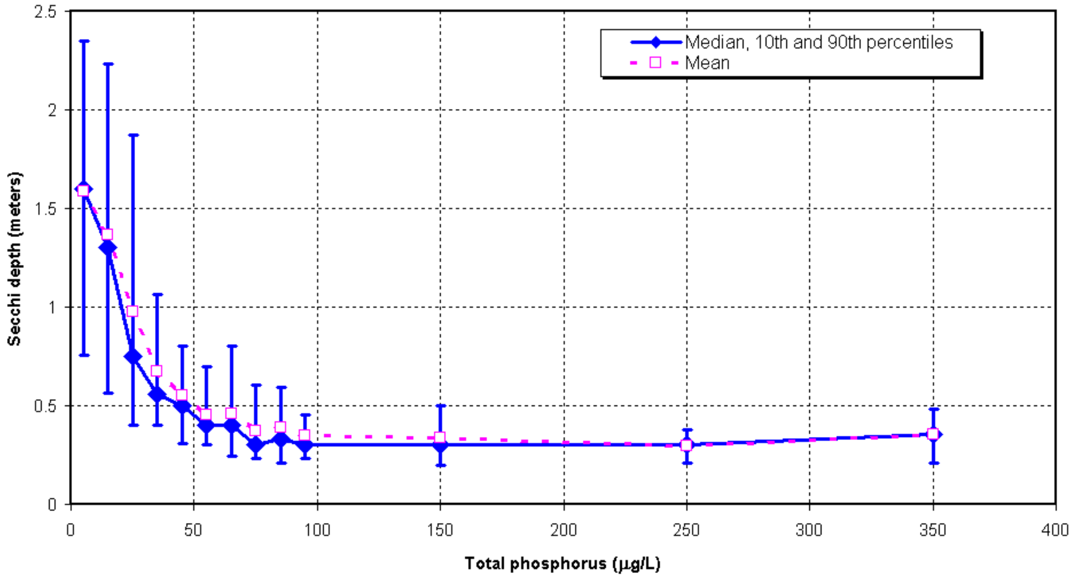

Figure 10. Upper Ocklawaha River Basin lakes, statistical summary of the relationship between total phosphorus and Secchi depth transparency

Tributary flows were the major phosphorus sources for Lakes Beauclair, Dora, and Eustis during the period 1991–2000 (Figure 11). Phosphorus discharges to Lake Griffin were nearly equally divided among the EMCA, agriculture, and tributary flows. Phosphorus loadings for the other lakes were divided among a number of sources, with no single dominant source.

Estimated average annual phosphorus loading was substantially higher for Lake Griffin than for the other lakes (Figure 11, Table 17). However, the ability of a lake to effectively assimilate nutrients is related to the size of the lake and the residence time for water in the lake (hydraulic detention time). Other things being equal, a larger lake can receive a greater external nutrient load without a deleterious impact on water quality.

Conversely, a lake with a longer hydraulic detention time will accumulate nutrients over a longer time period and therefore would be more sensitive to increases in external loading. Water quality modeling incorporates these morphologic and hydrologic factors. Water quality modeling will be discussed further later in the report, but as an initial step, we applied a model developed by Salas and Martino (1991) to predict in-lake phosphorus concentrations from estimates of external loading and hydraulic detention

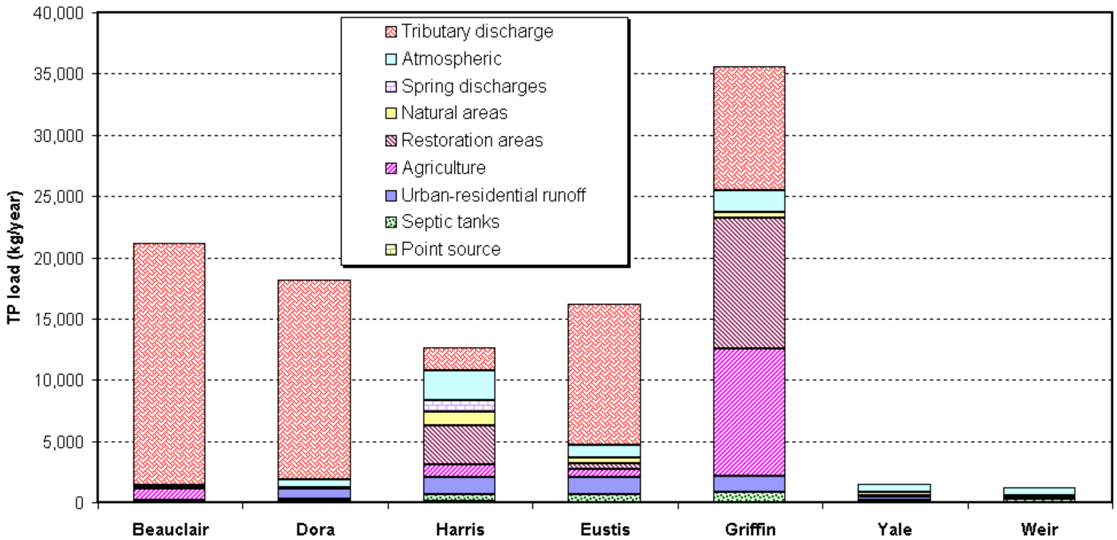

Figure 11. Summary of average annual total phosphorus loadings for the lakes in the Upper Ocklawaha River Basin, 1991–2000

time. The predicted in-lake phosphorus concentrations (Figure 12) are strongly (r=0.99) correlated with average phosphorus concentrations in the lakes (Figure 4).

All of the lakes had net retention of phosphorus during 1991–2000, ranging from 22% of the external load for Lake Beauclair to over 90% for Lakes Yale and Weir (Table 17). Retention and sedimentation coefficients were substantially lower for nitrogen than for phosphorus. There was a net export of nitrogen from Lakes Dora, Harris, and Griffin during this time period. Lakes Dora and Griffin also had negative nitrogen sedimentation coefficients for this time period. The apparent release of nitrogen from the lake sediments could be due to nitrogen fixation in the lakes. Nutrient budgets, retention, and sedimentation are also shown for Lake Griffin for the 1993–2000 time period, because of concern about the accuracy of loading estimates from the operating muck farms during 1991–92. Estimated retention and sedimentation coefficients for Lake Griffin are lower for 1993–2000 than for 1991–2000, which could indicate an overestimate of external loading during 1991–92.

Table 17. Estimated average annual rates of nutrient loading and outflows, retention coefficients, and sedimentation coefficients for the Upper Ocklawaha River Basin lakes, 1991–2000

Lake	TP Loading (kg/year)	TP Outflow (kg/year)	TP Retention Coefficient	TP Sedimentation Coefficient ($year^{-1}$)
Beauclair	21,200	16,500	0.220	2.70
Dora	18,200	9,000	0.504	1.97
Harris	12,700	3,500	0.727	1.01
Eustis	16,200	10,500	0.351	1.33
Griffin (1991–2000)	35,600	18,400	0.484	2.01
Griffin (1993–2000)	29,700	19,600	0.342	1.23
Yale	1,500	130	0.914	0.94
Weir	1,200	90	0.925	0.67
Lake	**TN Loading (kg/year)**	**TN Outflow (kg/year)**	**TN Retention Coefficient**	**TN Sedimentation Coefficient ($year^{-1}$)**
Beauclair	328,000	283,000	0.136	1.24
Dora	306,000	323,000	–0.057	–0.06
Harris	188,000	197,500	–0.051	0.03
Eustis	534,000	457,000	0.144	0.33
Griffin (1991–2000)	530,000	619,000	–0.167	–0.26
Griffin (1993–2000)	563,000	705,000	–0.253	–0.39
Yale	26,200	8,200	0.688	0.21
Weir	23,200	5,900	0.745	0.23

Note: kg = kilogram
TP = total phosphorus
TN = total nitrogen

Lake Beauclair

There was more than a 10-fold year-to-year variation in estimated phosphorus load to Lake Beauclair, due primarily to changes in discharges through the Apopka-Beauclair Lock and Dam (Figure 13). The largest single external phosphorus source for Lake Beauclair was discharges from the Lake Apopka Basin, accounting for about 93% of the estimated load. Agricultural discharges accounted for about 4.4% of the external phosphorus load, most of which came from the Hurley muck farm (3.6%). Urban-residential runoff and septic tank effluents accounted for 0.6% and 0.4% of the estimated load, respectively. Fulton (1995a) had estimated a larger contribution from urban-residential and agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate

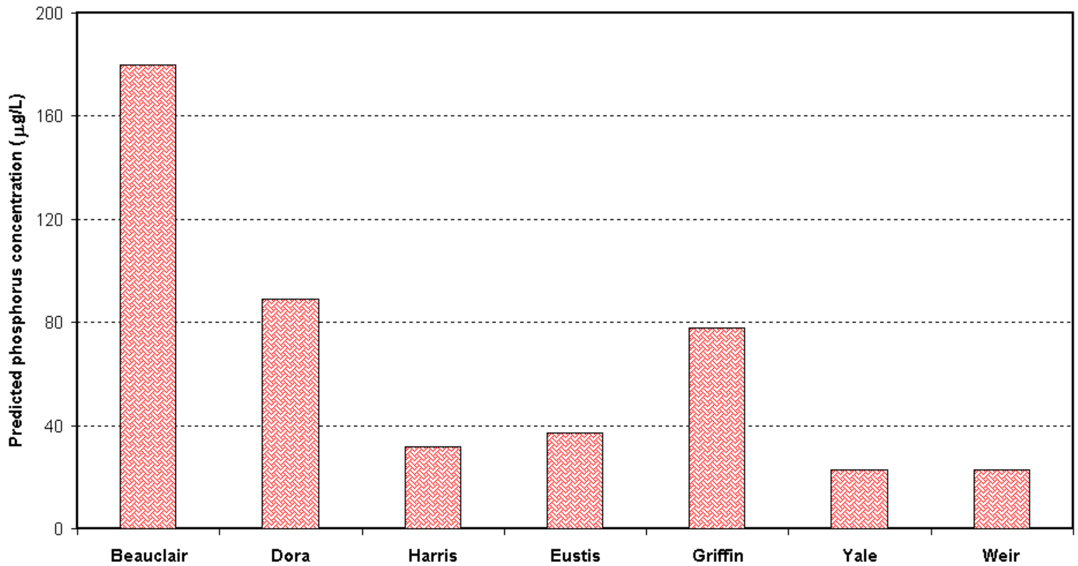

Figure 12. Predicted mean phosphorus concentrations for the Upper Ocklawaha River Basin lakes (1991–2000) using the Salas and Martino (1991) model

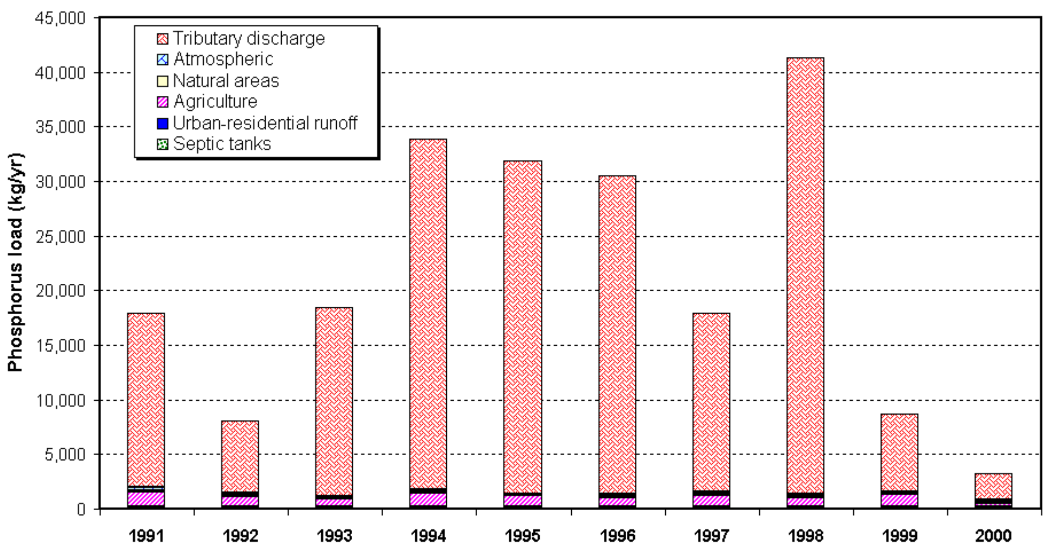

Figure 13. Estimated annual external phosphorus load to Lake Beauclair, 1991–2000

nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets.

Lake Dora

There was more than a 10-fold year-to-year variation in estimated phosphorus load to Lake Dora, due primarily to changes in tributary discharges from Lake Beauclair (Figure 14). The largest single external phosphorus source for Lake Dora was discharges from the Lake Beauclair basin, accounting for about 90% of the estimated load. Urban-residential runoff and septic tank effluents accounted for 4.4% and 1% of the estimated load, respectively. Point sources and agricultural runoff accounted for 0.6% and less than 0.1% of the external load, respectively. Fulton (1995a) had estimated a larger contribution from urban-residential and agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets.

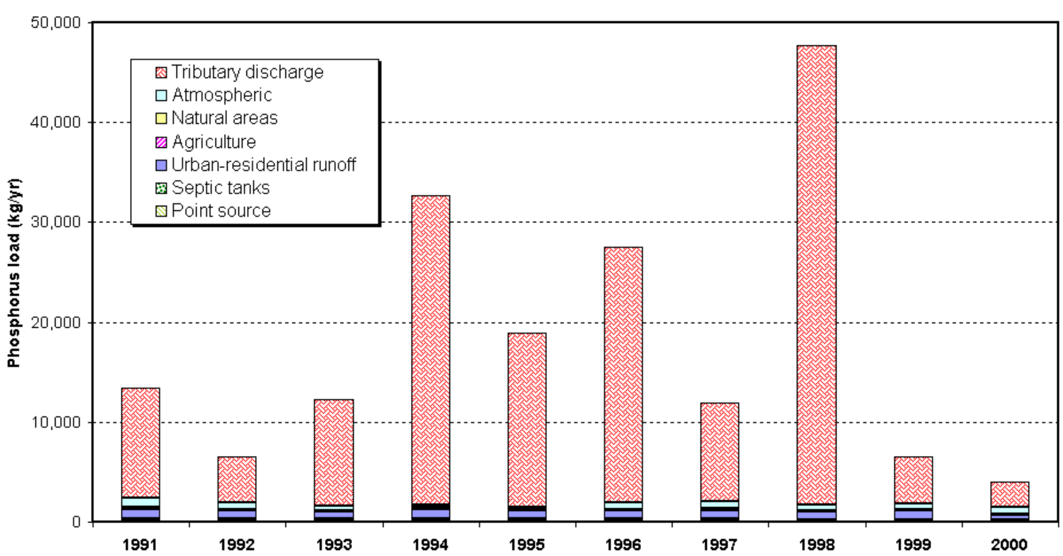

Figure 14. Estimated annual external phosphorus load to Lake Dora, 1991–2000

Lake Harris-Little Harris

There was more than a threefold year-to-year variation in estimated phosphorus load to Lake Harris-Little Harris, due primarily to changes in tributary discharges from the Palatlakaha River and in discharges from the Lake Harris Conservation Area (Figure 15). The largest single external phosphorus source for Lake Harris was discharges from the LHCA, accounting for about 25% of the estimated load. Other phosphorus sources for Lake Harris-Little Harris included atmospheric deposition (20%), tributary discharges (15%), urban-residential runoff (11%), muck farms (7%), septic tank effluents (4.5%), upland agriculture (1.5%), and point sources (0.9%). Fulton (1995a) had estimated a larger contribution from urban-residential and upland agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets.

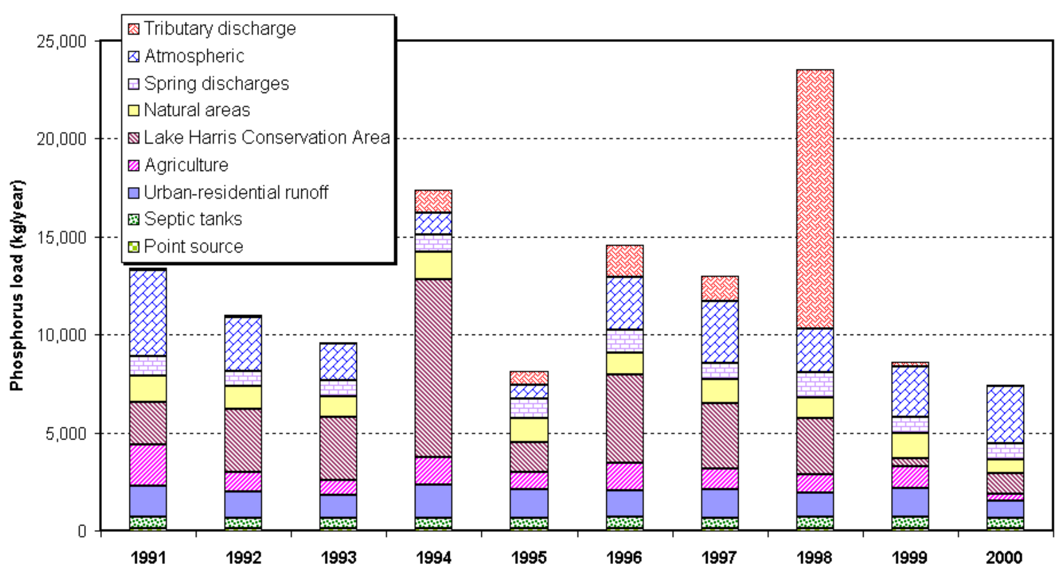

Figure 15. Estimated annual external phosphorus load to Lake Harris-Little Harris, 1991–2000

Lake Eustis

There was nearly a sevenfold year-to-year variation in estimated phosphorus load to Lake Eustis, due primarily to changes in tributary discharges (Figure 16). The largest single external phosphorus sources for Lake Eustis were discharges from Lake Dora and Lake Harris, accounting for about 54% and 18% of the estimated load, respectively. Other phosphorus sources for Lake Eustis included urban-residential runoff (8.5%), septic tank effluents (4.3%), muck farms (3.4%), the Pine Meadows Restoration Area (3.0%), upland agriculture (0.4%), and point sources (0.004%). Fulton (1995a) had estimated a larger contribution from urban-residential and upland agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets.

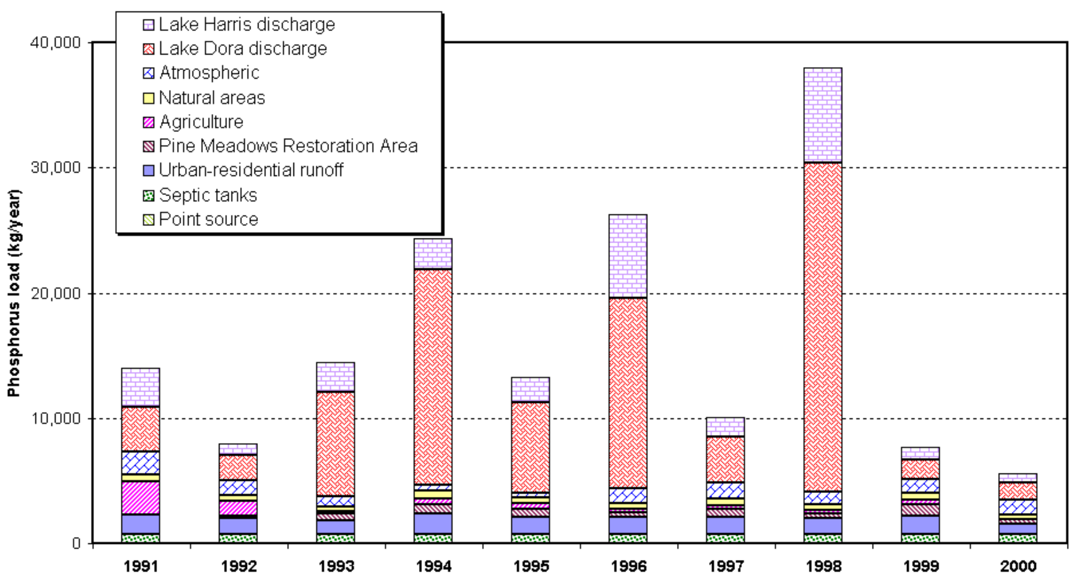

Figure 16. Estimated annual external phosphorus load to Lake Eustis, 1991–2000

Lake Griffin

There was nearly a sevenfold year-to-year variation in estimated phosphorus load to Lake Griffin, due primarily to changes in discharges from the Emeralda muck farms and the EMCA (Figure 17). Over the decade, phosphorus discharges to Lake Griffin were nearly equally divided among the EMCA (30%), agriculture (29%), and tributary flows from Lake Eustis (28%). Almost all of the discharges from agricultural land occurred in 1991–93, from the former Emeralda muck farms before they were purchased by SJRWMD and taken out of operation. Phosphorus discharges from the EMCA increased in late 1994 with the initiation of operations of the Lake Griffin Marsh Flow-Way, but then again decreased in subsequent years. Other phosphorus sources for Lake Griffin included urban-residential runoff (3.5%), septic tank effluents (2.4%), upland agriculture (0.5%), and point sources (0.07%). Fulton (1995a) had estimated a larger contribution from urban-residential and upland agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets.

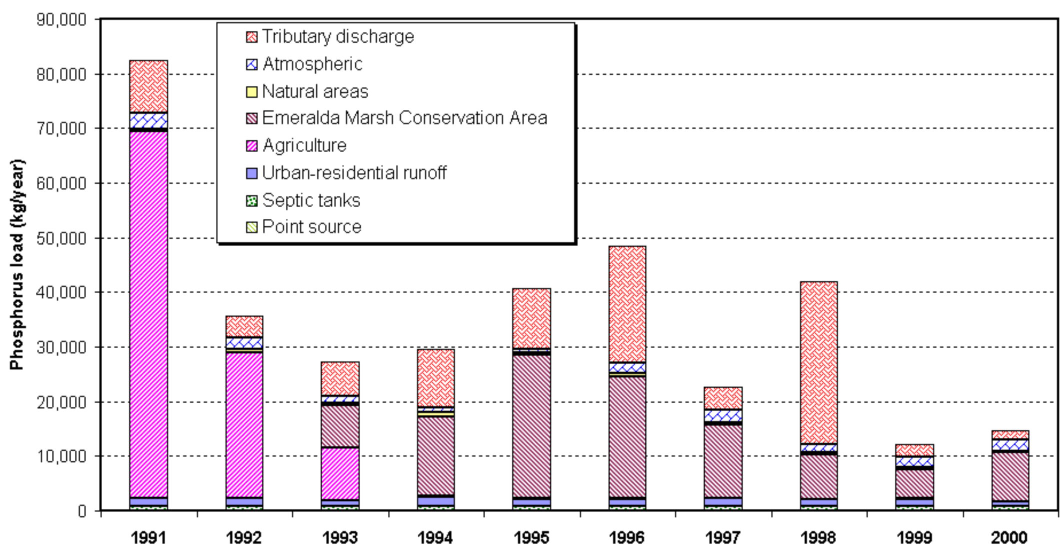

Figure 17. Estimated annual external phosphorus load to Lake Griffin, 1991–2000

Total estimated phosphorus discharges from the Emeralda Marsh agricultural area in 1991 (Figure 18), during which most of the muck farms were still operating, were substantially higher than estimated in Fulton 1995a for the operating muck farms (average 22,400 kilograms phosphorus per year during 1984–90). The higher estimated discharges in 1991 were because we used phosphorus concentrations found on the sites after SJRWMD purchase, which were substantially higher than those reported in the farm permit records. In the early 1990s, most of the estimated phosphorus discharges from the Emeralda Marsh area came from the Long and Eustis properties. There is substantial uncertainty in the estimates of discharges from the operating farms because of the discrepancies in concentrations and absence of information on how actively the farms were operated in the period before acquisition. Discharges from these properties decreased later in the decade after SJRWMD acquisition, due to cessation of agricultural discharges and later to closure of connecting culverts and drought conditions. In the latter years, most of the phosphorus discharges from the EMCA were attributable to the Lake Griffin Marsh Flow-Way.

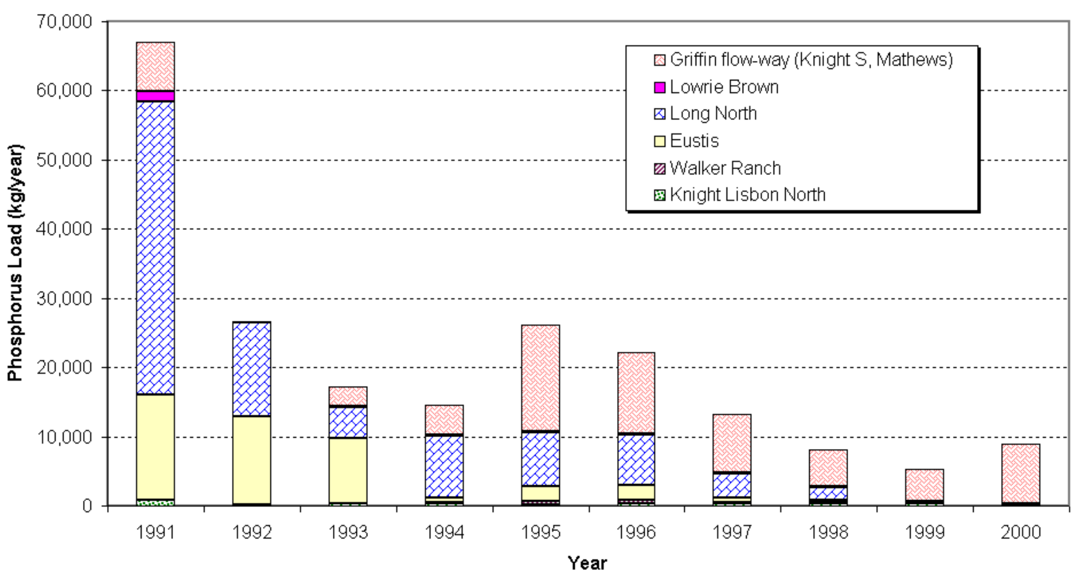

Figure 18. Estimated phosphorus discharges from the Emeralda Marsh properties (primarily agricultural through 1993; afterwards, part of the Emeralda Marsh Conservation Area) to Lake Griffin

Lake Yale

There was a twofold year-to-year variation in estimated phosphorus load to Lake Yale, due primarily to changes in atmospheric deposition (Figure 19). The largest single external phosphorus source for Lake Yale was atmospheric deposition, accounting for about 43% of the estimated load. The largest controllable phosphorus sources were urban-residential runoff (20% of the external load), septic tank effluents (8.7% of the external load), and agricultural runoff (6.2% of the external load). Fulton (1995a) had estimated a larger contribution from urban-residential and agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets. Point sources averaged 5.1% of the external phosphorus load, but substantially declined over the 10-year period (Figure 19), due to a reduction in reported weak waste discharges from Golden Gem citrus plant.

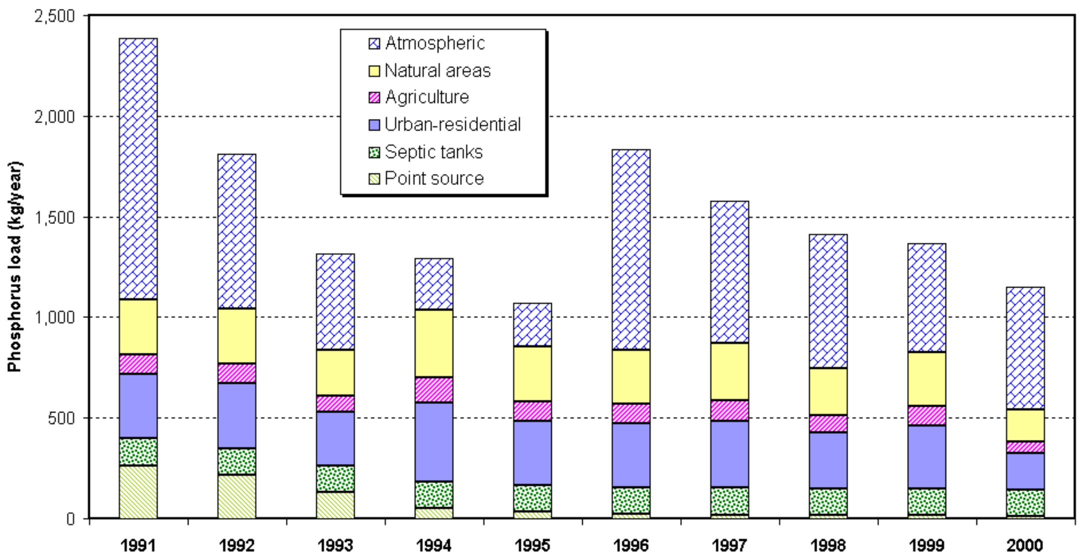

Figure 19. Estimated annual external phosphorus load to Lake Yale, 1991–2000

Lake Weir

There was a nearly twofold year-to-year variation in estimated phosphorus load to Lake Weir, due primarily to changes in atmospheric deposition (Figure 20). The largest single external phosphorus source for Lake Weir was atmospheric deposition, accounting for about 57% of the estimated load. The largest controllable phosphorus sources were septic tank effluents (19% of the external load) and urban-residential runoff (12% of the external load). Fulton (1995a) had estimated a larger contribution from urban-residential and agricultural runoff, due primarily to higher runoff coefficients used in that study. Additionally, a change in methodology to estimate nutrient losses in transport of stormwater runoff has resulted in greater transport losses in the new nutrient budgets.

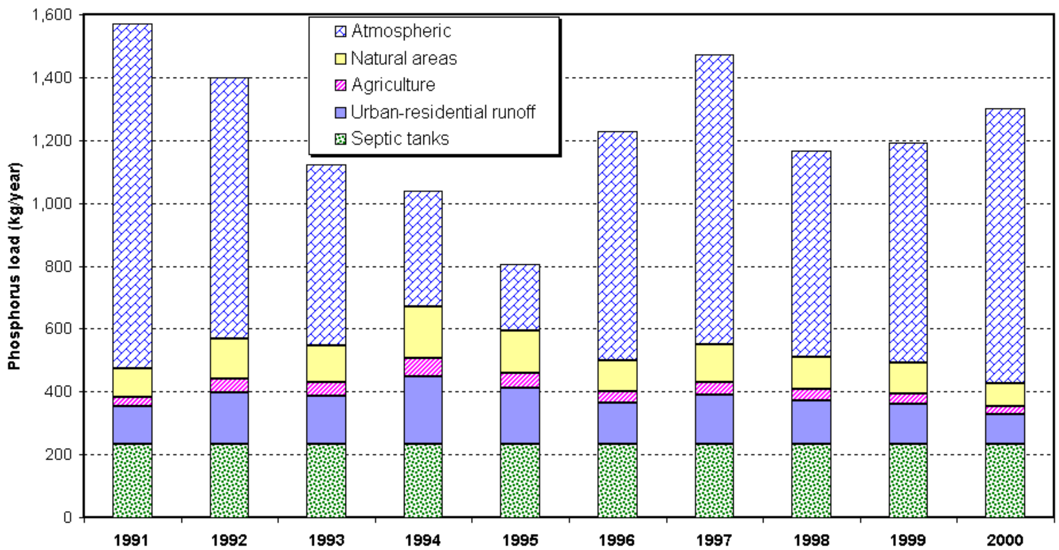

Figure 20. Estimated annual external phosphorus load to Lake Weir, 1991–2000

ESTIMATED NATURAL BACKGROUND AND PROPOSED TARGET PHOSPHORUS CONCENTRATIONS

Modeling of external loading and water quality under natural background conditions resulted in predicted mean natural background TP concentrations for the UORB lakes ranging from 7 to 33 µg/L (Table 18). Estimates of reference conditions determined from the 25th percentile of regional populations of lakes resulted in a similar range of TP concentrations (5–25 µg/L). Estimates of reference conditions determined from the 25th percentile of morphologically and hydrologically similar lakes resulted in a slightly higher range of TP concentrations (22–37 µg/L).

The composite estimates of natural background phosphorus concentrations range from 13 µg/L for Lake Weir to 28 µg/L for Lakes Beauclair, Dora, and Griffin (Table 18). Similar or lower total phosphorus concentrations were reported in single samples collected in 1952 from Lakes Dora (27 µg/L), Eustis (8 µg/L), and Weir (0 µg/L, evidently below the detection limit for the study) (Odum 1953). The same study reported a considerably higher phosphorus concentration in Lake Griffin (152 µg/L), but noted that that site was impacted by sewage discharges.

The proposed target TP concentrations for the UORB lakes range from 14 µg/L for Lake Weir to 32 µg/L for Lake Beauclair (Table 18). These target levels range from 15% of existing concentrations for Lake Beauclair to 100% for Lake Weir. The recommendation may be most uncertain for Lake Weir, since it is based in part on an estimated natural background TP concentration from the subset of morphologically and hydrologically similar Florida lakes that substantially exceeds the existing concentration in the lake (Table 18). This reference estimate may be relatively unreliable because it was based on the smallest subset of similar lakes (eight lakes). As noted above, Odum (1953) reported an undetectable phosphorus concentration for Lake Weir in 1952.

Appendix F summarizes estimates of natural background total nitrogen and chlorophyll-*a* concentrations, and Secchi depths for the UORB lakes, developed from the reference lakes approaches.

Table 18. Summary of existing, natural background, and proposed target total phosphorus (TP) concentrations for the Upper Ocklawaha River Basin lakes

Total Phosphorus Estimate (µg/L)	Lake						
	Beauclair	Dora	Harris	Eustis	Griffin	Yale	Weir
Mean TP 1991–2000	209	92	38	43	94	22	14
Modeled natural background TP	33	28	21	18	23	8	7
EPA-Lakewatch Florida ecoregions, 25th percentile	21						11
SJRWMD ecoregions, 25th percentile	25						5
Morphology and hydrology subsets, 25th percentile	28	32	25	32	37	22	23
Composite natural background TP estimate	28	28	23	24	28	18	13
Proposed TP target	32	31	26	28	31	20	14

PREDICTED EFFECTS OF THE TARGET PHOSPHORUS CONCENTRATIONS ON WATER QUALITY

We evaluated potential effects of the proposed target TP concentrations (Table 18) on chlorophyll-*a* concentrations, Secchi depth transparency, compliance with the Florida Impaired Waters Rule, and colonization of submersed aquatic vegetation.

To estimate chlorophyll-*a* concentrations in the UORB lakes at the target phosphorus concentrations, we conducted a regression analysis of the relationship between mean annual phosphorus and chlorophyll-*a* concentrations for the UORB lakes (Figure 21). Both variables were log-10 transformed, and the regression analysis used only annual means with total phosphorus concentrations less than 100 µg/L, because of evidence that phosphorus is not limiting to phytoplankton growth at concentrations above that level (Brown et al. 2000). The predicted mean chlorophyll-*a* concentrations would range from 8 µg/L for Lake Weir to 30 µg/L for Lake Beauclair (Table 19).

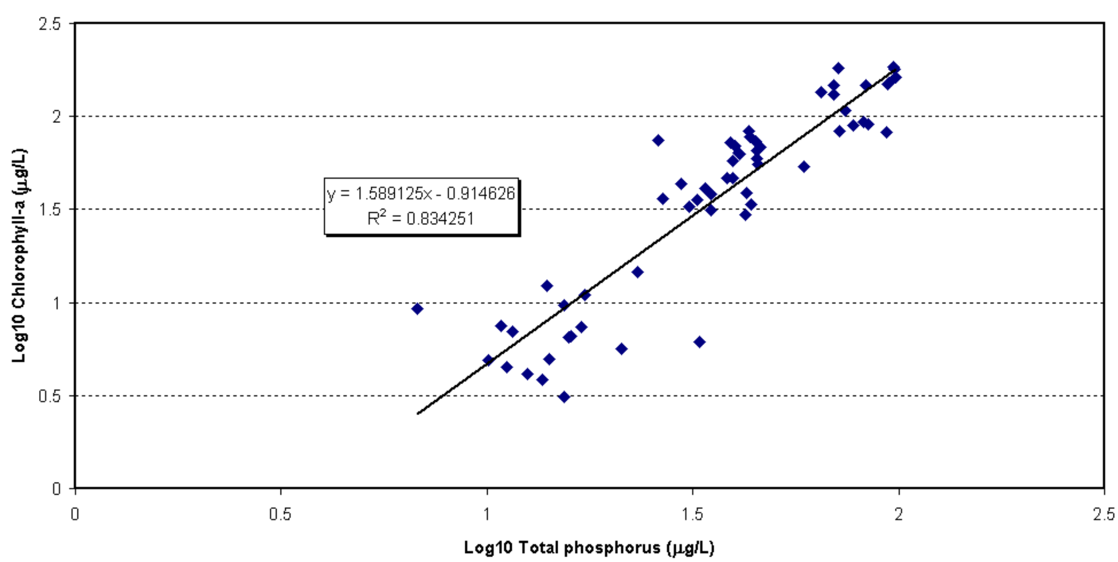

Figure 21. Upper Ocklawaha River Basin lakes, relationship between mean annual total phosphorus and chlorophyll-*a* concentrations (mean TP<100 µg/L)

Table 19. Summary of existing and predicted chlorophyll-*a* concentrations and Secchi transparencies at the recommended total phosphorus concentrations for the Upper Ocklawaha River Basin lakes

Lake	Mean Chlorophyll-*a* 1991–2000 (µg/L)	Mean Secchi Depth 1991–2000 (m)	Predicted Mean Chlorophyll-*a* (µg/L)	Predicted Mean Secchi Depth (m)
Beauclair	152	0.31	30	0.71
Dora	138	0.35	29	0.73
Harris	57	0.56	22	0.85
Eustis	55	0.54	24	0.79
Griffin	160	0.32	29	0.73
Yale	17	1.28	14	1.06
Weir	7	1.63	8	1.43

Note: m = meter
µg/L = micrograms per liter

To evaluate the expected variability in chlorophyll-*a* concentrations at the phosphorus targets, we further examined the variability in reported chlorophyll-*a* concentrations (Figure 9). For a range of phosphorus concentrations from 26 to 32 µg/L, which includes the recommended targets for Lakes Beauclair, Dora, Harris, Eustis, and Griffin, the chlorophyll-*a* statistics were the following: mean—37.6 µg/L; median—30.2 µg/L; 10th percentile—7.0 µg/L; 90th percentile—63.3 µg/L. For a range of phosphorus concentrations from 11 to 20 µg/L, which includes the recommended targets for Lakes Weir and Yale, the chlorophyll-*a* statistics were the following: mean—14.3 µg/L; median—8.4 µg/L; 10th percentile—0.5 µg/L; 90th percentile—28.1 µg/L.

Havens and Walker (2002) evaluated the frequencies of occurrence of chlorophyll-*a* concentrations greater than 40 µg/L (moderate bloom) and 60 µg/L (severe bloom) as function of total phosphorus concentrations in order to specify a phosphorus goal for Lake Okeechobee. A similar analysis for water quality data for the UORB lakes indicates that bloom frequencies reach a plateau at a phosphorus concentration of about 80 µg/L (Figure 22). For a range of phosphorus concentrations from 26 to 32 µg/L, which includes the recommended targets for Lakes Beauclair, Dora, Harris, Eustis, and Griffin, the frequencies were 38% moderate bloom and 11% severe bloom. For a range of phosphorus concentrations from 11 to 20 µg/L, which includes the

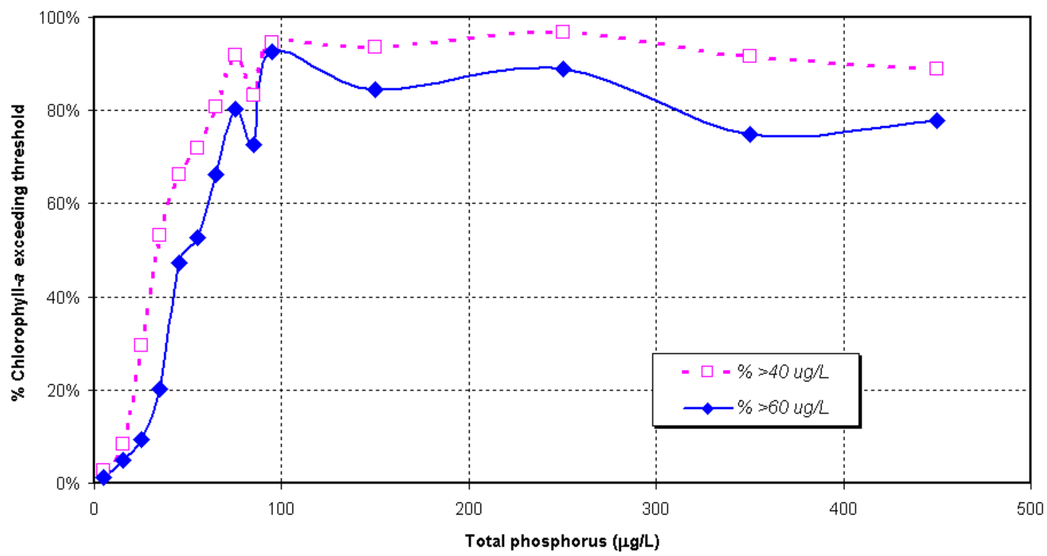

Figure 22. Upper Ocklawaha River Basin lakes, relationship between total phosphorus concentrations and frequencies of moderate (chlorophyll-*a* >40 µg/L) and severe (chlorophyll-*a* >60 µg/L) phytoplankton blooms

recommended target for Lakes Weir and Yale, the frequencies were 8% moderate bloom and 5% severe bloom. These bloom frequencies are greater than those occurring at the selected TMDL phosphorus concentration for Lake Okeechobee (Havens and Walker 2002), but are a substantial decrease from existing bloom frequencies for the UORB lakes, except for Lakes Yale and Weir.

We also conducted a regression analysis of the annual means for the UORB lakes to predict mean Secchi depth transparency at the recommended phosphorus targets (Figure 23). Again, both variables were log-10 transformed, and the regression analysis used only annual means with total phosphorus concentrations less than 100 µg/L. The predicted mean Secchi depths range from 0.71 m to 1.43 m (Table 19), which are about twice the reported levels in the lakes with relatively poor water quality, but are somewhat less than reported mean Secchi depths in Lakes Yale and Weir.

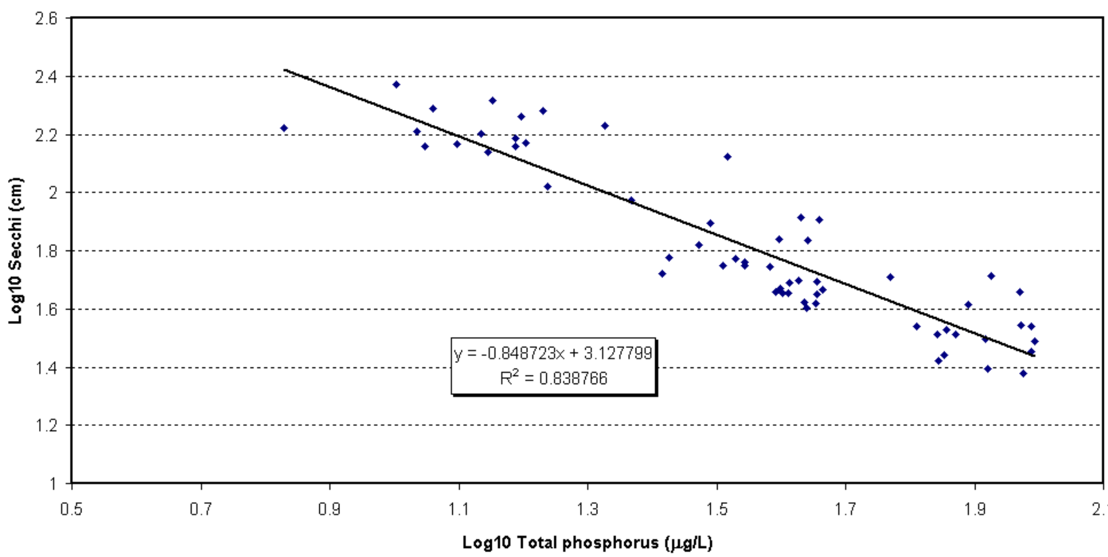

Figure 23. Upper Ocklawaha River Basin lakes, relationship between mean annual total phosphorus concentration and Secchi depth (mean TP<100 µg/L)

Florida's Impaired Waters Rule (62-303, *F.A.C.*) uses the Florida Trophic State Index (TSI) to determine whether lake waters should be assessed further for nutrient impairment. The Florida TSI is developed from chlorophyll-*a* concentrations and nutrient concentrations (TP and TN, with the nutrient equation used depending on the presumed limiting nutrient, as determined by the TN:TP ratio). For lakes, a TSI of 0–59 is considered good, 60–69 is fair, and 70–100 is poor.

According to the Impaired Waters Rule, lakes or lake segments shall be included on the planning list for nutrients if any of the following conditions are met:

1. For lakes with a mean color greater than 40 platinum cobalt units (pcu), the annual mean TSI for the lake exceeds 60, unless paleolimnological information indicates the lake was naturally greater than 60.
2. For lakes with a mean color less than or equal to 40 pcu, the annual mean TSI for the lake exceeds 40, unless paleolimnological information indicates the lake was naturally greater than 40.

3. For any lake, data indicate that annual mean TSIs have increased over the assessment period, as indicated by a positive slope in the means plotted versus time, or the annual mean TSI has increased by more than 10 units over historical values.

Predicted Florida TSIs were calculated for the UORB lakes for the estimated natural background condition and proposed target phosphorus concentrations. In calculating the nutrient TSI, it was assumed that phosphorus would be limiting under both conditions. We used the regression equation from the UORB lakes to predict mean chlorophyll-*a* concentrations for the TSI calculation. Average water color was determined for samples collected in the lakes since 1991.

Lake Beauclair was the only lake with a mean water color exceeding 40 pcu (Table 20). The predicted mean TSI for Lake Beauclair was slightly below the Impaired Waters Rule threshold for the estimated natural background condition, but slightly above the threshold for the proposed target phosphorus concentration. However, the increase in the TSI for Lake Beauclair is less than 10 units over the natural background value.

Table 20. Summary of predicted Trophic State Indices (TSI) for natural background conditions and at the recommended target total phosphorus (TP) concentrations for the Upper Ocklawaha River Basin lakes

Lake	Mean Water Color (pcu)	Predicted Mean TSI	
		Natural Background Condition	Target TP Concentration
Beauclair	56	59	62
Dora	40	58	61
Harris	31	54	57
Eustis	31	56	59
Griffin	38	59	61
Yale	20	48	51
Weir	16	41	43

Note: pcu = platinum cobalt unit

For Lakes Dora, Eustis, Harris, Griffin, Yale, and Weir, the predicted mean TSIs for both the estimated natural background condition and for the proposed target phosphorus concentration exceed the Impaired Waters Rule threshold (Table 20). However, in all of these lakes, the increase in the TSI for the proposed target phosphorus concentration is less than 10 units over the natural background value.

Therefore, in all of the UORB lakes, the predicted mean TSI for the proposed target phosphorus concentration meets the requirement in the Impaired Waters Rule that there be an increase of less than 10 units over the natural background value.

PREDICTED EFFECTS OF THE TARGET PHOSPHORUS CONCENTRATIONS ON COLONIZATION OF SUBMERSED AQUATIC VEGETATION

We estimated the depths to which submersed aquatic vegetation could persist in the UORB lakes under existing conditions and at the proposed target phosphorus concentrations by assuming that the compensation depth for aquatic vegetation (I_c) occurs at 1% of surface illumination (I_o). First, a regression analysis of data on phosphorus and water transparency (measured as the light extinction coefficient) collected from the UORB lakes between 1998 and 2001 (Figure 3) was used to predict the light extinction at existing (mean for 1991–2000) and proposed target phosphorus concentrations. We then used the Lambert-Beer equation to predict the compensation depth for aquatic vegetation and assumed that the compensation depth is equal to the maximum depth of colonization.

Once the predicted maximum depths of aquatic vegetation were determined, we converted them to elevations using average water elevation in the lakes. We then used bathymetric data presented in Danek et al. 1991 to convert the elevations to areas, interpolating between bathymetric contours. The areas of aquatic vegetation were estimated as the difference between the lake areas at average water elevation and the areas at the predicted deepest elevation for aquatic vegetation growth.

Under existing conditions, the five lakes in the Harris Chain have predicted areas of aquatic vegetation of less than or equal to 11% of lake area (Figure 24). Under the proposed target phosphorus concentrations, the predicted vegetation areas for these lakes increase to between 10% and 41% of lake area. The highest estimate, for Lake Beauclair, may be an overestimate of

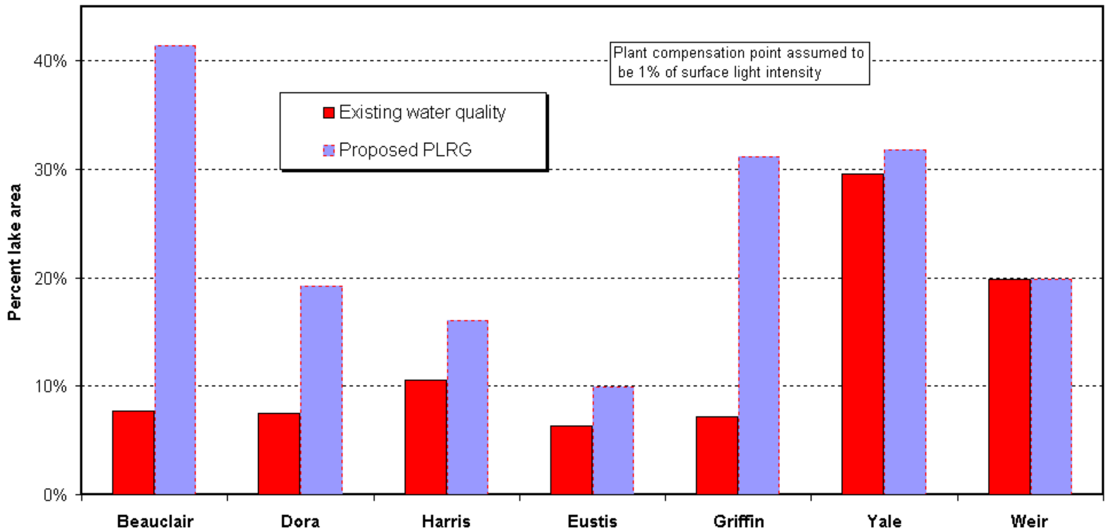

Figure 24. Predicted percent of lake areas supporting aquatic vegetation under existing conditions and at the proposed target phosphorus concentrations

potential vegetated area because this lake has greater water color than the other lakes (Table 20), which would reduce light transmittance in this lake. For Lakes Yale and Weir, the predicted vegetation areas under existing conditions are about 30% and 20% of lake area, respectively, and are not significantly changed at the proposed target phosphorus concentrations.

There are limited available data to evaluate the accuracy of the predictions of vegetation occurrence in the UORB lakes. For 2 years in Lake Yale, predicted and measured occurrence of vegetation corresponded well. Correspondence was poorer for 1 year in Lake Harris with fewer data points. Hestand et al. (1991) reported that Illinois pondweed reached an areal coverage in Lake Yale in March 1987 equivalent to 58% of the lake area. Using the average reported phosphorus concentration for the year prior to March 1987 (eight measurements), the predicted plant cover would be 47% of the lake area. The same study reported that *Hydrilla* reached an areal coverage in Lake Yale at the end of 1991 equivalent to 72% of the lake area. Using the average reported phosphorus concentration during 1991 (five measurements), the predicted plant cover would be 90% of the lake area. Canfield and Hoyer (1992) reported that 27% of the area of Lake Harris supported aquatic vegetation in summer 1987. Using the average reported phosphorus concentration for the year prior to summer 1987 (only two measurements), the predicted plant cover would be 9% of the lake area.

The vegetation coverage reported for Lake Harris by Canfield and Hoyer (1992) seems improbable, following the assumptions used in this analysis. To achieve 27% vegetation coverage in Lake Harris by assuming that the compensation depth for aquatic vegetation (I_c) occurs at 1% of surface illumination would require a phosphorus concentration below 20 µg/L, which is substantially below the reported mean concentration in 1986–87 (50.6 µg/L) and even below the estimated natural background phosphorus concentration for the lake (23 µg/L) (Table 18). Even more improbably, to achieve 27% vegetation coverage in Lake Harris at the reported mean phosphorus concentration in 1986–87 would require that the compensation depth for aquatic vegetation (I_c) occurred at about 0.003% of surface illumination. Factors not considered in this analysis that could increase vegetation coverage include fluctuations in water levels, which may allow plant establishment during periods of low water levels, or occurrence of emergent or floating-leaved vegetation, growth habits which may avoid limitation due to subsurface light extinction. Canfield and Hoyer (1992) did report that emergent and floating-leaved vegetation accounted for 78% of the estimated plant biomass in Lake Harris.

LAKE WATER QUALITY MODELING

The best-fitting water quality models predicted phosphorus concentrations within 10% of the reported averages for all of the lakes except Lake Weir (Table 21). There was substantial variability among lakes in the best-fitting models, but both directly proportional and nonlinear models provided good correspondence with reported phosphorus concentrations (Figure 25). For four of the lakes (Beauclair, Harris, Griffin, and Yale), models in which the predicted phosphorus concentration is directly proportional to the external load showed the best correspondence with reported concentrations. For Lake Dora, there was only a small difference between the best-fitting directly proportional and nonlinear models. For Lakes Eustis and Weir, there were moderate differences between the best-fitting directly proportional and nonlinear models.

Table 21. Correspondence of model predictions with reported phosphorus concentrations

Bold = *model predictions directly proportional to external phosphorus loading. For Lakes Yale and Weir, spreadsheet models using estimated outflows and inflows for detention time estimates are distinguished by (o) and (i), respectively*.

Lake	Model* Predictions Within Specified Percentage of Reported Phosphorus Concentrations				
	<5	5–10	1–20	2–30	3–40
Beauclair	**P5**	**P11**, **P1**, P3	**P6**, P4	**P12**	P2, **P13**, P10, P8
Dora	P4, **P6**		P2, P3, P8	**P5**, P10	**P1**
Harris	**P12**, P2, P4, P10		**P5**, P8, **P6**		**P1**
Eustis	P8, P10	P2, P3, **P5**	P4, **P6**	**P1**, P9, **P11**	
Griffin (1993–2000)		**P11**, **P12**, **P1**, P3	**P5**	P4, **P13**, P8	**P6**, P2, P10
Yale	**P6(i)**, P8, **P12**			P4(i), P10	P9, P4(o), **P5(i)**
Weir (1991–98)			P8	**P12**, P9	**P6(o)**

*Model numbers identified in Table 14

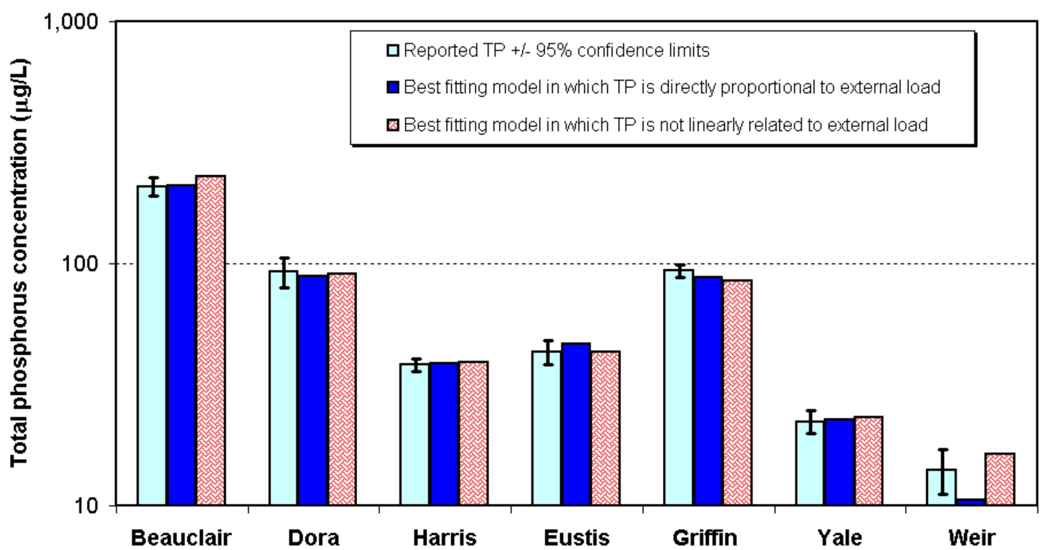

Figure 25. Comparison of reported phosphorus concentrations in the Upper Ocklawaha River Basin lakes with the best-fitting model predictions

For all of the lakes, models in which the predicted nitrogen concentration is directly proportional to the external load more closely corresponded to measured concentrations than did models in which the predicted nitrogen concentration is not directly proportional to the external load (Table 22, Figure 26). The water quality models generally underestimated nitrogen concentrations in the lakes (Figure 26). Nitrogen concentrations in several of the lakes corresponded as or more closely to estimated inflow concentrations ("Model" N2) than to predictions of the models (Table 22). Only in Lakes Yale and Weir were in-lake nitrogen concentrations substantially lower than estimated inflow concentrations. As noted previously, sedimentation coefficients were substantially lower for nitrogen than for phosphorus, with estimated net releases of nitrogen over the study period from the sediments for Lakes Dora and Griffin (Table 17). The water quality models tended to overestimate nitrogen sedimentation, resulting in underestimates of water-column nitrogen concentrations. The apparently low nitrogen sedimentation could be due to substantial underestimates of external loading, but this seems unlikely for lakes in which the nitrogen budget is dominated by large loadings from tributaries (such as Griffin and Dora). Other factors possibly contributing to low net nitrogen sedimentation include nitrogen fixation within the lakes and a large fraction of dissolved nitrogen in the water column (in contrast, water column phosphorus is largely in particulate form).

Table 22. Correspondence of model predictions with reported nitrogen concentrations

Bold = *model predictions directly proportional to external nitrogen loading. For Lakes Yale and Weir, spreadsheet models using estimated outflows and inflows for detention time estimates are distinguished by (o) and (i), respectively.*

Lake	Model* Predictions Within Specified Percentage of Reported Nitrogen Concentrations				
	<5	5–10	10–20	20–30	30–40
Beauclair	**N8**, **N7**	**N2**	**N6**		N1
Dora		**N2**	**N8**		**N6**, **N7**
Harris	**N2**				**N8**, **N6**
Eustis	**N8**		**N2**	**N6**, **N7**	
Griffin (1993–2000)					**N2**
Yale		**N6**, N1(o)			**N8**
Weir (1991–98)			**N8**, N1(i)		N1(o), N5

*Model numbers identified in Table 14

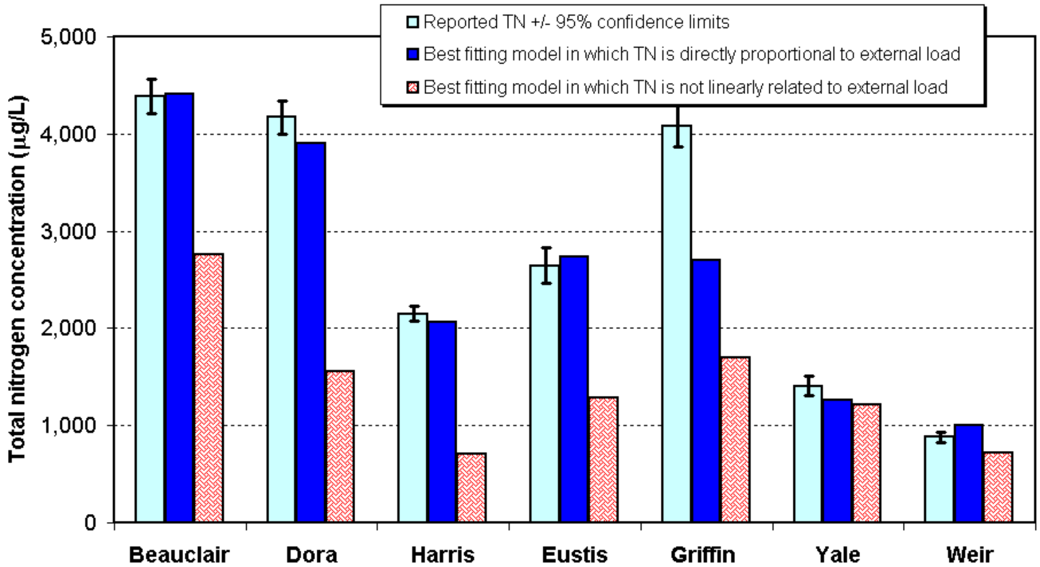

Figure 26. Comparison of reported nitrogen concentrations in the Upper Ocklawaha River Basin lakes with the best-fitting model predictions

Although no single water quality model provided accurate predictions of nutrient concentrations in all of the lakes, models in which predicted nutrient concentrations were directly proportional to the external nutrient load generally were as effective as nonlinear models. This suggests that the assumption of a directly proportional relationship between external loading and in-lake phosphorus concentrations used in the interim PLRG recommendations is reasonable for existing conditions in these lakes.

However, the key question is whether the response of the lakes to nutrient reductions will be directly proportional. In terms of the Vollenweider-type models, a directly proportional response is equivalent to no change in the net sedimentation coefficient. All of the models used in which the sedimentation coefficient is a function of the external load predict a decrease in the coefficient as the external load decreases. This decrease in the sedimentation coefficient results in a predicted decrease in nutrient concentrations that is smaller than the reduction in external load. The models in which the nonlinearity is due to exponents in the equations also predict decreases in nutrient concentrations that are smaller than reductions in the external load.

Conversely, there are reasons to expect that a reduction in external load may result in an increase in net sedimentation coefficients, including reduction in surplus phosphorus, increases in submerged plant biomass, and changes in composition of phytoplankton, macroinvertebrates, and fish communities (Havens and Schelske 2001). Other parts of the SJRWMD restoration programs, including re-establishment of wetland and submersed vegetation habitats and rough fish harvesting, would be expected to increase net sedimentation coefficients. An increase in the sedimentation coefficient would result in a decrease in nutrient concentrations that is greater than the reduction in external load. We do not presently have information to make a quantitative prediction whether sedimentation coefficients would be higher, lower, or similar to present values once they equilibrate to a new lower rate of external phosphorus loading. Given this uncertainty, using a model that assumes a decrease in sedimentation coefficients may prove to be overly conservative, that is, requiring a lower external load than is necessary to meet the in-lake concentration targets. Given that directly proportional models worked best in predicting phosphorus concentrations for four of the six lakes in which reductions in external phosphorus loading are recommended (no loading reductions are recommended for Lake Weir), we believe that continuing to assume direct proportionality is most appropriate at the present time. This assumption should be re-examined as the lakes respond to reductions in external nutrient loading.

PHOSPHORUS LOAD REDUCTION GOALS

The phosphorus load reduction needed to meet the target phosphorus concentrations ranges from 0% in Lake Weir to 85% in Lake Beauclair (Table 23). These phosphorus load reduction goals should be treated as a long-term average annual load. As noted previously, there is substantial year-to-year variation in the phosphorus load to these lakes. The estimated external phosphorus load was lower than the target load in at least one of the years from 1991 to 2000 in all of the lakes, except for Lake Griffin.

Meeting the phosphorus load reduction goals would require substantial reductions in average discharges from upstream Lake Apopka as well as from muck farm restoration areas within the UORB. For a preliminary evaluation of the feasibility of reaching the proposed PLRGs, we calculated hypothetical phosphorus loadings to the UORB lakes for the following phosphorus loading scenario:

Table 23. Existing and target total phosphorus (TP) loads and TP load reduction goals for the Upper Ocklawaha River Basin lakes

Lake	External Phosphorus Load (metric tons/year)			Percent Reduction
	Current TP Load	Target TP Load	Load Reduction Goal	
Beauclair	21.2	3.2	18.0	85
Dora	18.2	6.2	12.0	66
Harris	12.7	8.7	4.0	31
Eustis	16.2	10.4	5.8	36
Griffin	35.6	11.9	23.7	67
Yale	1.52	1.37	0.15	10
Weir	1.23	1.23	0	0

- Phosphorus concentrations in discharges to Lake Beauclair from the Lake Apopka Basin were reduced to 55 µg/L (the established TMDL for that basin). This concentration was multiplied by the mean discharge volume for 1991–2000 to estimate the phosphorus load.
- Discharges from existing restoration areas within the UORB were reduced to 1.1 kilograms of phosphorus per hectare per year. This areal loading is the reduced loading from the North Shore Restoration Area at Lake Apopka necessary to meet that lake's target for total phosphorus. We judge this areal loading to be attainable, albeit with difficulty in the short term.
- Discharges from the LHCA were diverted from Lake Harris to Lake Griffin (which will result from the planned re-establishment of a connection between Lakes Harris and Griffin through the LHCA)
- No other changes to phosphorus loadings to the lakes from the surrounding watersheds

In-lake phosphorus concentrations for the UORB lakes were estimated for this loading scenario by assuming that phosphorus concentration in the lakes is directly proportional to external phosphorus loading (same assumption as made in setting the PLRGs):

$$\text{Hypothetical TP concentrations} = \text{Existing TP concentrations} \times \frac{\text{Hypothetical TP load}}{\text{Existing TP load}} \quad \textbf{(23)}$$

The hypothetical downstream discharge was then estimated as

$$\textit{Hypothetical TP discharge} = \textit{Existing TP discharge} \times \frac{\textit{Hypothetical TP concentration}}{\textit{Existing TP concentration}} \quad \textbf{(24)}$$

The proposed PLRGs could be met under these conditions for three of the lakes in the UORB (Table 24). To meet the target for Lake Beauclair would require a further reduction in the discharge concentration from the Lake Apopka Basin, since at the established TMDL concentration, that source alone (3.5 metric tons per year) would exceed the total PLRG for Lake Beauclair. The PLRGs for Lakes Harris, Griffin, and Yale could potentially be met with reductions in other human-influenced sources (urban-residential runoff, agricultural runoff, septic tanks, point sources), since the additional load reductions required to meet the PLRGs are less than the total load from those sources.

Table 24. Current, target, and hypothetical future external total phosphorus (TP) loads to the Upper Ocklawaha River Basin lakes (metric tons/year)

Future loads were calculated with an assumption of reduction in loads from Lake Apopka and from restoration areas to low levels but no changes in loads from other sources.

Lake	Current TP Load	Target TP Load	Hypothetical TP Load
Beauclair	21.2	3.2	5.0
Dora	18.2	6.2	5.7
Harris	12.7	8.7	9.3
Eustis	16.2	10.4	8.9
Griffin	35.6	11.9	13.0
Yale	1.52	1.37	1.52
Weir	1.23	1.23	1.23

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APPENDIX A—1995 LAND USES IN THE UPPER OCKLAWAHA RIVER BASIN CONTRIBUTING SUBBASINS

Land Use	Contributing Subbasins*						
	Lake Beauclair	Lake Dora	Lake Harris	Lake Eustis	Lake Griffin	Lake Yale	Lake Weir
Low density residential	379.8	367.9	1,029.2	479.0	1,037.6	287.1	521.5
Medium density residential	95.5	414.8	675.2	1,285.6	685.9	104.4	381.1
High density residential	0.0	148.6	324.3	379.8	316.9	111.4	9.3
Low density commercial	20.4	40.0	159.7	113.3	104.1	30.7	23.8
High density commercial	11.6	238.3	449.4	360.8	393.9	42.9	9.2
Industrial	6.1	68.4	110.5	50.8	90.9	45.2	3.0
Mining	0.0	0.1	162.1	8.1	43.3	5.1	9.8
Openland/recreational	34.3	80.8	272.1	197.3	1,277.3	75.8	310.8
Pasture	283.3	48.3	959.3	304.3	844.7	482.5	378.6
Cropland	92.5	15.1	610.5	295.5	904.6	195.3	218.9
Tree crops	440.7	106.1	1,067.9	357.7	214.4	417.9	199.3
Feeding operations	0.0	0.0	12.6	2.8	8.8	2.3	0.0
Other agriculture	56.6	34.1	184.0	148.4	113.7	54.0	6.3
Forest/rangeland	870.7	578.6	2,597.7	1,569.2	3,379.0	1,140.8	404.6
Water	476.8	190.6	461.8	526.5	401.8	253.5	143.1
Wetlands	365.4	495.9	4,144.0	1,702.4	2,781.3	1,326.8	338.4
Spray fields	0.0	2.3	500.6	11.2	58.2	23.8	0.0
Muck farm/restoration areas	180.4	0.0	315.5	344.8	2,433.9	0.0	0.0
Total, contributing subbasins	3,314.1	2,829.8	14,036.3	8,137.3	15,090.3	4,599.5	2,957.8
Lake area	448.0	1,791.1	7,442.2	3,144.7	5,377.0	1,630.3	2,263.0
Total	3,762.2	4,620.9	21,478.5	11,282.0	20,467.3	6,229.8	5,220.8

*Unit of measure is hectares

APPENDIX B—SUMMARY OF TREATMENT EFFICIENCIES (PERCENT REMOVAL) FOR FLORIDA STORMWATER TREATMENT SYSTEMS

Study	Source*	Type	Total Phosphorus Load	Total Nitrogen Load
1	1	Wet detention	65	58.8
2	2	Wet detention/wetland	37	30
3	3	Wetland	70	46
4	4	Wet detention	92.5	67
5	5	Wet detention with filtration	61	–3
6	6	Dry detention	84	86
7	7	Wet detention with filtration	67	
8	8	Wet retention	29	62
9	8	Wet retention	73	19
10	8	Off-line retention/detention	92	85
11	8	Off-line retention/detention	76	30
12	8	Wet detention	91	
13	9	Wet detention	–15.6	–17.8
Mean			63.2	42.1

Note: Blank cell indicates no data

*1, Rushton and Dye 1993; 2, Martin 1988; 3, Carr and Rushton 1995; 4, Sawka et al. 1993; 5, Harper and Miracle 1993b; 6, Harper et al. 1999; 7, Gowan and Watkins 1997; 8, Harper 1995; 9, Stoker 1996

APPENDIX C—ESTIMATED AREAS, FLOW PATH LENGTHS, AND SEDIMENT AND NUTRIENT DELIVERY RATIOS FOR EACH OF THE UPPER OCKLAWAHA RIVER BASIN CONTRIBUTING SUBBASINS

Receiving Lake	Subbasin Number	Area (hectares)	Flow Path Length (meters)	Reckhow et al. (1989) Sediment Delivery Ratio	Phosphorus Delivery Ratio	Nitrogen Delivery Ratio
Beauclair	1	881.7	704.4	0.2954	0.7217	0.8559
Beauclair	2	295.1	2,022.0	0.2064	0.5916	0.7685
Beauclair	3	681.1	3,113.5	0.1782	0.5686	0.7131
Beauclair	4	325.3	797.5	0.2832	0.5124	0.6657
Beauclair	5	1,578.9	474.7	0.3378	0.6904	0.8404
Dora	6	517.2	2,228.9	0.1997	0.6918	0.7490
Dora	7	4,103.8	522.4	0.3270	0.7356	0.8369
Harris	19	568.6	2,599.5	0.1895	0.6618	0.7662
Harris	20	881.3	4,407.4	0.1583	0.6837	0.7728
Harris	23	3,900.8	375.2	0.3659	0.7561	0.8258
Harris	25	455.2	2,295.4	0.1977	0.6944	0.7896
Harris	27	336.0	8,774.0	0.1253	0.7296	0.7987
Harris	28	2,821.3	4,357.8	0.1590	0.6722	0.7671
Harris	30	96.3	9,778.3	0.1208	0.5937	0.7625
Harris	31	330.2	9,588.2	0.1216	0.6951	0.7975
Harris	32	500.3	8,366.0	0.1273	0.6672	0.7605
Harris	33	197.6	2,208.5	0.2003	0.6734	0.7895
Harris	34	338.6	1,379.4	0.2350	0.6948	0.8047
Harris	35	803.2	1,301.1	0.2398	0.7884	0.8058
Harris	36	10,249.1	627.7	0.3072	0.7253	0.8085
Eustis	39	395.1	8,199.1	0.1282	0.6049	0.7160
Eustis	40	962.2	8,409.9	0.1271	0.6337	0.7150
Eustis	41	1,757.8	4,952.4	0.1522	0.6510	0.7534
Eustis	42	267.5	3,163.6	0.1772	0.6443	0.7893
Eustis	43	191.1	3,011.2	0.1802	0.6348	0.7849
Eustis	44	345.2	1,136.9	0.2510	0.6892	0.7775
Eustis	45	1,058.2	2,068.4	0.2048	0.7096	0.7896
Eustis	46	5,841.9	618.4	0.3087	0.7287	0.8175
Eustis	47	463.0	488.3	0.3346	0.6834	0.8116
Griffin	48	651.9	379.7	0.3644	0.7377	0.8573
Griffin	49	248.3	818.6	0.2807	0.6544	0.8390
Griffin	54	706.6	5,997.0	0.1426	0.6129	0.8187

Receiving Lake	Subbasin Number	Area (hectares)	Flow Path Length (meters)	Reckhow et al. (1989) Sediment Delivery Ratio	Phosphorus Delivery Ratio	Nitrogen Delivery Ratio
Griffin	55	662.0	7,280.2	0.1335	0.6314	0.7918
Griffin	56	208.5	1,069.6	0.2563	0.5639	0.7535
Griffin	57	927.7	1,880.1	0.2116	0.5944	0.8018
Griffin	58	217.9	1,442.3	0.2315	0.7216	0.8655
Griffin	59	795.1	2,497.0	0.1921	0.6167	0.8190
Griffin	60	464.7	2,343.8	0.1963	0.7392	0.8040
Griffin	61	136.3	1,111.4	0.2530	0.7889	0.8232
Griffin	62	50.7	2,097.2	0.2038	0.7164	0.7901
Griffin	63	1,148.2	1,930.2	0.2097	0.7009	0.8016
Griffin	64	276.7	1,834.9	0.2133	0.6312	0.8169
Griffin	65	6,478.3	426.7	0.3503	0.7198	0.8407
Griffin	66	480.1	7,482.9	0.1323	0.6375	0.8049
Griffin	67	1,554.5	6,012.3	0.1425	0.6104	0.8142
Griffin	68	2,546.7	897.2	0.2721	0.6583	0.8425
Griffin	70	1,246.2	2,199.4	0.2006	0.6003	0.8051
Griffin	74	768.6	3,184.1	0.1769	0.5673	0.7470
Griffin	75W	898.4	834.0	0.2789	0.6494	0.8288
Yale	51	116.4	3,013.0	0.1802	0.5833	0.7833
Yale	52	429.3	5,284.1	0.1489	0.5041	0.6758
Yale	53	5,684.1	888.1	0.2730	0.6804	0.8203
Weir	86	5,220.8	551.8	0.3210	0.6675	0.8228
Mean				0.2177	0.6626	0.7925

Note: Fraction of sediments and nutrients in runoff transported to receiving lake

APPENDIX D—SUMMARY OF MEAN ANNUAL REPORTED TOTAL PHOSPHORUS CONCENTRATIONS (mg/L) AND MODEL PREDICTIONS, 1986–1990

Year	Mean Reported Total Phosphorus	Model Predictions				Percent Deviations From Reported			
		Larson and Mercier (1976)	Canfield and Bachmann (1981)	Reckhow (1991)	Salas and Martino (1991)	Larson and Mercier (1976)	Canfield and Bachmann (1981)	Reckhow (1991)	Salas and Martino (1991)
Lake Beauclair									
1986	0.195	0.239	0.193	0.257	0.203	22.6	–1.0	31.8	4.1
1987	0.263	0.304	0.243	0.339	0.262	15.6	–7.6	28.9	–0.4
1988	0.242	0.22	0.188	0.24	0.19	–9.1	–22.3	–0.8	–21.5
1989	0.209	0.204	0.164	0.21	0.171	–2.4	–21.5	0.5	–18.2
1990	0.212	0.181	0.147	0.205	0.15	–14.6	–30.7	–3.3	–29.2
Mean	0.224	0.230	0.187	0.250	0.195	2.4	–16.6	11.4	–13.0
Lake Dora									
1986	0.139	0.106	0.084	0.071	0.083	–23.7	–39.6	–48.9	–40.3
1987	0.176	0.122	0.098	0.092	0.097	–30.7	–44.3	–47.7	–44.9
1988	0.146	0.118	0.096	0.087	0.094	–19.2	–34.2	–40.4	–35.6
1989	0.087	0.102	0.079	0.062	0.079	17.2	–9.2	–28.7	–9.2
1990	0.071	0.09	0.071	0.058	0.07	26.8	0.0	–18.3	–1.4
Mean	0.124	0.108	0.086	0.074	0.085	–5.9	–25.5	–36.8	–26.3

Year	Mean Reported Total Phosphorus	Model Predictions				Percent Deviations From Reported			
		Larson and Mercier (1976)	Canfield and Bachmann (1981)	Reckhow (1991)	Salas and Martino (1991)	Larson and Mercier (1976)	Canfield and Bachmann (1981)	Reckhow (1991)	Salas and Martino (1991)
Lake Eustis									
1986	0.04	0.073	0.064	0.047	0.057	82.5	60.0	17.5	42.5
1987	0.073	0.088	0.079	0.069	0.071	20.5	8.2	−5.5	−2.7
1988	0.063	0.074	0.069	0.058	0.06	17.5	9.5	−7.9	−4.8
1989	0.044	0.068	0.06	0.043	0.053	54.5	36.4	−2.3	20.5
1990	0.041	0.071	0.059	0.042	0.055	73.2	43.9	2.4	34.1
Mean	0.052	0.075	0.066	0.052	0.059	49.6	31.6	0.9	17.9
Lake Harris									
1986	0.020	0.071	0.049	0.024	0.053	254.4	145.8	18.1	162.9
1987	0.038	0.054	0.046	0.026	0.041	40.5	19.3	−32.0	6.3
1988	0.044	0.063	0.050	0.028	0.047	41.7	13.8	−37.1	7.0
1989	0.035	0.075	0.051	0.025	0.056	113.8	46.1	−29.8	58.7
1990	0.037	0.074	0.049	0.023	0.055	97.7	30.9	−37.7	46.1
Mean	0.035	0.067	0.049	0.025	0.050	109.6	51.2	−23.7	56.2
Lake Griffin									
1986	0.095	0.142	0.104	0.103	0.112	50.0	9.7	8.9	18.0
1987	0.098	0.112	0.096	0.105	0.091	13.9	−2.0	7.1	−7.6
1988	0.095	0.130	0.107	0.118	0.105	36.7	12.7	23.8	10.7
1989	0.072	0.156	0.109	0.110	0.122	116.1	50.5	52.2	69.2
1990	0.076	0.128	0.095	0.105	0.100	68.4	25.1	38.5	31.9
Mean	0.087	0.134	0.102	0.108	0.106	57.0	19.2	26.1	24.4

Year	Mean Reported Total Phosphorus	Model Predictions: Larson and Mercier (1976)	Model Predictions: Canfield and Bachmann (1981)	Model Predictions: Reckhow (1991)	Model Predictions: Salas and Martino (1991)	Percent Deviations From Reported: Larson and Mercier (1976)	Percent Deviations From Reported: Canfield and Bachmann (1981)	Percent Deviations From Reported: Reckhow (1991)	Percent Deviations From Reported: Salas and Martino (1991)
Lake Yale									
1986	0.019	0.092	0.061	0.031	0.069	380.2	220.6	61.9	259.4
1987	0.027	0.085	0.061	0.033	0.064	211.8	122.0	19.9	134.9
1988	0.013	0.088	0.063	0.033	0.066	572.0	378.4	149.5	407.3
1989	0.015	0.092	0.062	0.032	0.069	521.2	321.1	114.8	366.2
1990	0.011	0.099	0.062	0.030	0.074	826.0	478.2	180.6	589.1
Mean	0.017	0.091	0.062	0.032	0.068	502.2	304.1	105.3	351.4
Lake Weir									
1986	0.015	0.059	0.032	0.007	0.043	289.6	108.0	–55.7	179.7
1987	0.014	0.049	0.031	0.008	0.035	248.5	125.6	–46.0	152.2
1988	0.007	0.046	0.032	0.008	0.034	588.7	371.8	17.4	400.3
1989	0.014	0.055	0.032	0.007	0.040	285.1	123.6	–49.5	177.5
1990	0.010	0.156	0.032	0.005	0.110	1,408.4	214.2	–55.2	964.6
Mean	0.012	0.073	0.032	0.007	0.052	564.1	188.6	–37.8	374.8

APPENDIX E—ESTIMATED AVERAGE ANNUAL TOTAL PHOSPHORUS (TP) AND TOTAL NITROGEN (TN) LOADING TO THE UPPER OCKLAWAHA RIVER BASIN LAKES, 1991–2000

Nutrient Source	Lake Beauclair			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	46.5	0.22	664.1	0.20
Medium density residential	42.2	0.20	421.7	0.13
High density residential	0.0	0.00	0.0	0.00
Low density commercial	4.9	0.02	57.7	0.02
High density commercial	15.2	0.07	115.8	0.04
Industrial	10.0	0.05	59.5	0.02
Mining	0.0	0.00	0.0	0.00
Openland/recreational	1.1	0.01	33.2	0.01
Hurley muck farm	771.8	3.64	4,399.2	1.34
Pasture	59.6	0.28	478.2	0.15
Cropland	49.9	0.24	451.7	0.14
Tree crops	38.5	0.18	734.6	0.22
Feeding operations	0.0	0.00	0.0	0.00
Other agriculture	20.8	0.10	155.6	0.05
Forest/rangeland	29.7	0.14	844.8	0.26
Water	25.1	0.12	1208.3	0.37
Wetlands	97.4	0.46	2762.6	0.84
Septic tanks	87.5	0.41	1,545.1	0.47
Precipitation	58.9	0.28	2,107.2	0.64
Dry deposition	82.2	0.39	612.5	0.19
Apopka-Beauclair Canal discharge	19,744.1	93.17	310,783.9	94.83
Lake Dora discharge	6.8	0.03	278.3	0.08
Total	21,192.3	100.00	327,713.9	100.00

Note: kg/yr = kilograms per year

Nutrient Source	Lake Dora			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	29.8	0.16	404.2	0.13
Medium density residential	153.8	0.85	1,483.0	0.49
High density residential	128.7	0.71	832.8	0.27
Low density commercial	14.9	0.08	135.6	0.04
High density commercial	402.5	2.21	2,706.6	0.89
Industrial	73.8	0.41	496.8	0.16
Mining	0.0	0.00	0.0	0.00
Openland/recreational	1.1	0.01	33.5	0.01
Pasture	4.0	0.02	34.3	0.01
Cropland	2.2	0.01	19.8	0.01
Tree crops	3.5	0.02	64.6	0.02
Feeding operations	0.0	0.00	0.0	0.00
Other agriculture	2.5	0.01	25.4	0.01
Forest/rangeland	9.0	0.05	252.7	0.08
Water	8.9	0.05	434.6	0.14
Wetlands	125.0	0.69	3,536.5	1.16
Septic tanks	187.0	1.03	3,301.7	1.08
Precipitation	239.4	1.32	8,569.7	2.80
Dry deposition	335.0	1.84	2,498.1	0.82
Lake Beauclair discharge	16,333.3	89.89	279,032.5	91.27
Lake Eustis discharge	6.1	0.03	562.0	0.18
Domestic WWTP spills	0.1	0.00	0.6	0.00
Tavares—Caroline St. WWTP runoff	52.1	0.29	625.2	0.20
Mount Dora WWTP runoff	57.3	0.32	687.8	0.22
Total	18,170.3	100.00	305,738.1	100.00

Note: kg/yr = kilograms per year
WWTP = wastewater treatment plant

Nutrient Source	Lake Harris			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	57.6	0.46	853.4	0.45
Medium density residential	228.7	1.83	2,253.2	1.19
High density residential	224.6	1.79	1,569.7	0.83
Low density commercial	48.7	0.39	482.8	0.25
High density commercial	673.0	5.38	4,704.6	2.48
Industrial	126.2	1.01	826.6	0.44
Mining	4.8	0.04	49.7	0.03
Openland/recreational	1.8	0.01	64.1	0.03
Ja-Mar muck farm	828.4	6.62	5,297.2	2.80
Knight-Leesburg muck farm	78.9	0.63	607.6	0.32
Pasture	80.8	0.65	657.3	0.35
Cropland	59.4	0.47	608.7	0.32
Tree crops	25.7	0.20	485.1	0.26
Feeding operations	8.4	0.07	227.1	0.12
Other agriculture	13.5	0.11	109.5	0.06
Forest/rangeland	27.2	0.22	778.3	0.41
Water	22.8	0.18	1,061.4	0.56
Wetlands	967.9	7.73	28,446.0	15.02
Lake Harris Conservation Area	3,132.6	25.03	8,276.4	4.37
Septic tanks	558.6	4.46	9,859.7	5.21
Precipitation	1,024.7	8.19	36,674.5	19.37
Dry deposition	1,434.4	11.46	10,692.4	5.65
Spring discharges	928.3	7.42	43,417.2	22.93
Palatlakaha River discharge	1,765.0	14.10	25,016.1	13.21
Lake Eustis discharge	83.0	0.66	4,941.0	2.61
Silver Springs Citrus sprayfield runoff	3.6	0.03	30.9	0.02
Domestic WWTP spills	3.2	0.03	15.1	0.01
Leesburg WWTP sprayfield runoff	51.8	0.41	711.7	0.38
Tavares—Woodlea Rd. WWTP runoff	54.3	0.43	653.4	0.35
Total	12,517.8	100.00	189,370.7	100.00

Note: kg/yr = kilograms per year
WWTP = wastewater treatment plant

Nutrient Source	Lake Eustis			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	22.0	0.14	345.1	0.06
Medium density residential	381.3	2.37	3,899.5	0.73
High density residential	294.9	1.83	1,950.5	0.37
Low density commercial	46.4	0.29	406.3	0.08
High density commercial	565.7	3.51	3,875.5	0.73
Industrial	60.5	0.38	386.3	0.07
Mining	0.2	0.00	1.8	0.00
Openland/recreational	1.0	0.01	33.4	0.01
Pine Meadows muck farm	293.4	1.82	1,887.4	0.35
Springhill muck farm	248.7	1.54	1,563.5	0.29
Pasture	19.6	0.12	158.9	0.03
Cropland	27.6	0.17	262.1	0.05
Tree crops	6.9	0.04	135.1	0.03
Feeding operations	4.4	0.03	74.6	0.01
Other agriculture	11.3	0.07	91.8	0.02
Forest/rangeland	12.9	0.08	378.8	0.07
Water	20.5	0.13	1,092.5	0.20
Wetlands	392.6	2.44	11,579.3	2.17
Pine Meadows Restoration Area	478.6	2.97	2,971.5	0.56
Septic tanks	691.8	4.29	12,210.8	2.29
Precipitation	425.1	2.64	15,220.4	2.85
Dry deposition	595.6	3.70	4,442.4	0.83
Lake Dora discharge	8,659.0	53.74	307,572.2	57.58
Lake Harris discharge	2,850.6	17.69	163,574.5	30.62
Domestic WWTP spills	0.0	0.00	0.1	0.00
Eustis WWTP runoff	0.7	0.00	10.0	0.00
Total	16,111.3	100.00	534,124.2	100.00

Note: kg/yr = kilograms per year
WWTP = wastewater treatment plant

Nutrient Source	Lake Griffin			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	61.2	0.17	880.1	0.17
Medium density residential	228.2	0.64	2,264.8	0.43
High density residential	180.0	0.51	1,368.9	0.26
Low density commercial	49.1	0.14	410.9	0.08
High density commercial	576.4	1.62	4,099.8	0.77
Industrial	128.5	0.36	780.5	0.15
Mining	0.8	0.00	8.2	0.00
Openland/recreational	9.8	0.03	296.5	0.06
Muck farms	10,298.5	28.95	13,115.3	2.47
Pasture	70.7	0.20	541.1	0.10
Cropland	74.8	0.21	796.1	0.15
Tree crops	4.5	0.01	84.9	0.02
Feeding operations	4.2	0.01	114.5	0.02
Other agriculture	13.5	0.04	105.6	0.02
Forest/rangeland	36.6	0.10	1,055.0	0.20
Water	13.7	0.04	661.1	0.12
Wetlands	459.6	1.29	13,409.2	2.53
Emeralda Marsh Conservation Area	10,619.1	29.85	7,055.6	1.33
Septic tanks	857.4	2.41	15,135.0	2.86
Precipitation	732.7	2.06	26,242.3	4.95
Dry deposition	997.8	2.81	7,449.2	1.41
Haines Creek discharge	10,127.5	28.47	433,903.3	81.86
Lake Yale discharge	1.0	0.00	59.6	0.01
Cutrale Citrus weak waste discharges	8.7	0.02	55.4	0.01
Cutrale Citrus sprayfield runoff	16.3	0.05	135.4	0.03
Cutrale Citrus spills	0.2	0.00	0.1	0.00
Domestic WWTP spills	0.2	0.00	1.1	0.00
Total	35,571.1	100.00	530,029.4	100.00

Note: kg/yr = kilograms per year
WWTP = wastewater treatment plant

Nutrient Source	Lake Yale			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	14.2	0.94	185.5	0.71
Medium density residential	37.2	2.45	368.6	1.41
High density residential	86.4	5.68	549.5	2.10
Low density commercial	16.2	1.06	132.4	0.51
High density commercial	78.3	5.14	515.1	1.97
Industrial	71.8	4.72	419.8	1.60
Mining	0.1	0.00	0.7	0.00
Openland/recreational	0.5	0.03	14.8	0.06
Pasture	46.8	3.07	350.9	1.34
Cropland	32.7	2.15	296.4	1.13
Tree crops	9.1	0.60	172.1	0.66
Feeding operations	1.9	0.13	30.7	0.12
Other agriculture	3.4	0.22	23.6	0.09
Forest/rangeland	17.3	1.14	495.8	1.89
Water	15.0	0.98	740.7	2.83
Wetlands	226.3	14.88	6,558.6	25.06
Septic tanks	132.3	8.69	2,335.1	8.92
Precipitation	282.8	18.59	9,744.6	37.23
Dry deposition	371.5	24.42	2,717.4	10.38
Umatilla WWTP runoff	13.6	0.89	188.5	0.72
Golden Gem weak waste discharge	64.1	4.21	333.1	1.27
Total	1,521.5	100.00	26,173.9	100.00

Note: kg/yr = kilograms per year
WWTP = wastewater treatment plant

Nutrient Source	Lake Weir			
	Mean TP Load 1991–2000		Mean TN Load 1991–2000	
	kg/yr	%	kg/yr	%
Low density residential	37.7	3.07	475.4	2.05
Medium density residential	78.5	6.39	912.6	3.93
High density residential	5.8	0.47	41.1	0.18
Low density commercial	10.8	0.88	89.5	0.39
High density commercial	10.0	0.81	79.7	0.34
Industrial	1.7	0.14	15.8	0.07
Mining	0.2	0.01	1.5	0.01
Openland/recreational	2.3	0.19	65.6	0.28
Pasture	15.9	1.29	134.5	0.58
Cropland	19.5	1.58	171.3	0.74
Tree crops	3.8	0.31	69.8	0.30
Feeding operations	0.0	0.00	0.0	0.00
Other agriculture	0.2	0.02	2.0	0.01
Forest/rangeland	2.9	0.24	81.7	0.35
Water	6.8	0.55	388.4	1.67
Wetlands	102.9	8.37	2,900.3	12.50
Septic tanks	234.2	19.05	4,134.8	17.82
Precipitation	278.4	22.64	10,526.7	45.36
Dry deposition	418.1	34.00	3,118.0	13.43
Total	1,229.7	100.00	23,208.7	100.00

Note: kg/yr = kilograms per year

APPENDIX F—SUMMARY OF ESTIMATED NATURAL BACKGROUND CONDITIONS (TN, CHLOROPHYLL-*a*, AND SECCHI DEPTHS) IN THE UPPER OCKLAWAHA RIVER BASIN LAKES USING REFERENCE LAKES APPROACHES

Appendix F

Summary of Natural Background Total Nitrogen Estimates (mg/L)	Lake							Mean, Harris Chain
	Beauclair	Dora	Eustis	Harris	Griffin	Yale	Weir	
EPA-Lakewatch Florida ecoregions (25th percentile)	0.821	0.821	0.821	0.821	0.821	0.821	0.595	0.821
SJRWMD ecoregions (25th percentile)	0.720	0.720	0.720	0.720	0.720	0.720	0.444	0.720
Morphology and hydrology subsets (25th percentile)	0.820	0.645	0.646	0.606	0.778	0.474	0.466	0.661
Selected natural background TN estimate (weight hydrology and ecoregion estimates equally)	0.795	0.708	0.708	0.688	0.774	0.622	0.493	0.716
Summary of Natural Background Chlorophyll-*a* Estimates (µg/L)	Lake							Mean, Harris Chain
	Beauclair	Dora	Eustis	Harris	Griffin	Yale	Weir	
EPA-Lakewatch Florida ecoregions (25th percentile)	9.72	9.72	9.72	9.72	9.72	9.72	3.03	9.72
SJRWMD ecoregions (25th percentile)	4.53	4.53	4.53	4.53	4.53	4.53	1.40	4.53
Morphology and hydrology subsets (25th percentile)	6.77	5.38	5.78	6.17	6.84	2.57	2.27	5.59
Selected natural background chlorophyll-*a* estimate (weight hydrology and ecoregion estimates equally)	6.95	6.25	6.45	6.65	6.98	4.85	2.25	6.35
Summary of Natural Background Secchi Depth Estimates (m)	Lake							Mean, Harris Chain
	Beauclair	Dora	Eustis	Harris	Griffin	Yale	Weir	
EPA-Lakewatch Florida ecoregions (25th percentile)	1.443	1.443	1.443	1.443	1.443	1.443	2.638	1.443
SJRWMD ecoregions (25th percentile)	—	—	—	—	—	—	—	—
Morphology and hydrology subsets (25th percentile)	1.779	1.735	2.136	1.863	1.445	2.869	3.285	1.971
Selected natural background Secchi depth estimate (weight hydrology and ecoregion estimates equally)	1.611	1.589	1.790	1.653	1.444	2.156	2.961	1.707

Note: m = meter
mg/L = milligrams per liter
µg/L = micrograms per liter
— = no data