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EXTERNAL NUTRIENT BUDGET AND TROPHIC STATE MODELING FOR LAKES IN THE UPPER OCKLAWAHA RIVER BASIN

by

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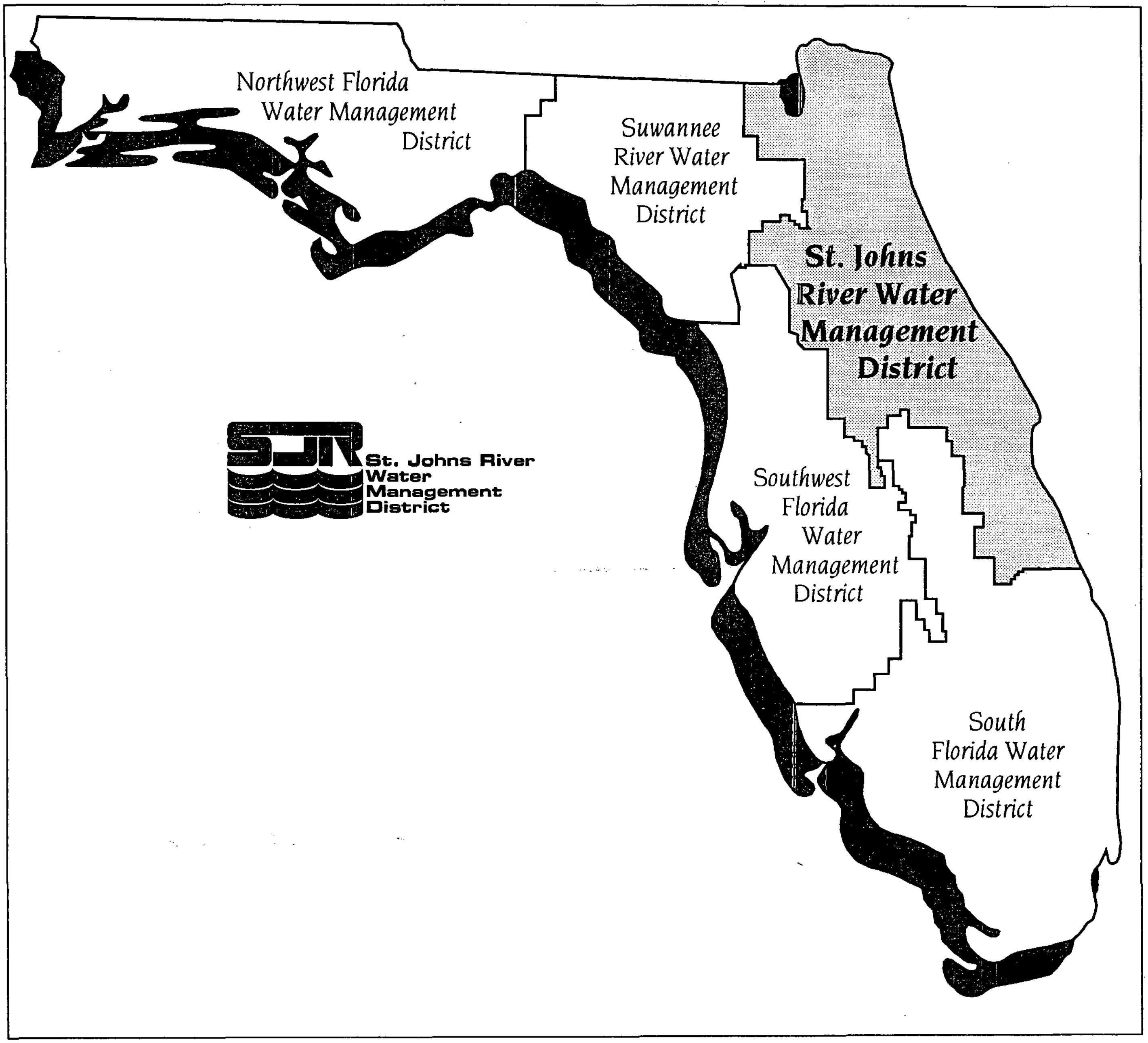

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EXECUTIVE SUMMARY

The upper Ocklawaha River Basin (UORB) has been designated a priority basin for restoration and management under the Surface Water Improvement and Management Program. Water quality in the basin has been degraded by nutrient loading from agricultural and urban development. Several of the lakes in the basin are among the most eutrophic in the state, whereas others are in the less productive mesotrophic category but are potentially threatened by increasing urban development. Water quality in Lakes Beauclair, Dora, and Griffin is rated poor; Lakes Harris-Little Harris and Eustis generally have fair water quality; and Lakes Weir and Yale have good water quality.

The purposes of the present study were (1) to develop preliminary nutrient budgets for the years 1980–90 for the seven major lakes in the basin (Beauclair, Dora, Harris-Little Harris, Eustis, Griffin, Yale, and Weir) and (2) to develop recommendations for further studies and management actions to improve water quality.

POTENTIAL NUTRIENT LIMITATION

Ratios of nitrogen to phosphorus 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).

LAKE NUTRIENT BUDGETS

Estimated average annual total nutrient loadings were highest for Lakes Griffin and Eustis; however, when nutrient loadings are related to lake surface area, Lake Beauclair had substantially higher loading rates than any other lake. Lakes Weir, Yale, and Harris-Little Harris had the lowest areal nutrient loading rates.

- Estimated nutrient loadings to Lakes Beauclair and Dora were heavily dominated by tributary flows from upstream water bodies. Residential stormwater runoff and muck farm discharges within the immediate drainage basins made relatively minor contributions to estimated nutrient loadings of these lakes. Estimated septic tank discharges were negligible sources of nutrients for Lakes Beauclair and Dora.

  Estimated nutrient loading to Lake Harris-Little Lake Harris was divided among a number of sources, with no single one dominant. The most significant estimated nutrient sources included upland agriculture, muck farms, atmospheric deposition, spring discharges, and residential-urban runoff. Septic tank and citrus plant waste discharges were minor nutrient sources for Lake Harris-Little Lake Harris. One of two muck farms in the Lake Harris-Little Lake Harris basin was acquired by the St. Johns River Water Management District in 1991.

- Tributary discharges were estimated to be the major nutrient source for Lake Eustis. Other significant nutrient sources included upland agriculture, muck farm discharges, and residential runoff.

  Muck farm discharges were estimated to be the major source of phosphorus for Lake Griffin during the study period, and tributary discharges were the major source of nitrogen. Other estimated nutrient sources for the lake were relatively minor, of which residential runoff was the most significant. Acquisitions of muck farms by the St. Johns River Water Management District since 1990 have substantially reduced the discharges to Lake Griffin.

- Estimated nutrient loading to Lake Yale was divided among a number of sources, with no single one dominant. Significant estimated nutrient sources included upland agriculture, runoff from natural areas and from residential-urban development, and weak waste discharges from citrus processing plants.

Estimated nutrient loading to Lake Weir also was divided among a number of sources, with no single one dominant. Significant estimated nutrient sources included atmospheric deposition, runoff from residential areas and upland agriculture, and septic tank discharges.

In general, predictions of in-lake phosphorus concentrations by lake trophic state models were similar to observed data for the lakes. These results indicate that the total phosphorus loading estimates are the correct order of magnitude. However, the model predictions for in-lake total nitrogen concentrations consistently underestimated observed concentrations, except for those in Lakes Yale and Weir. These underestimates of in-lake nitrogen concentrations may be due to underestimates in nitrogen loading or inadequacies in the nitrogen trophic state model. A likely source for underestimates of nitrogen loading is ground water seepage.

ASSESSMENT OF EFFECTS OF ALTERNATIVE RESTORATION AND MANAGEMENT ACTIONS

Lake trophic state models were used to predict responses of phosphorus loading and in-lake phosphorus concentrations to several potential restoration and management actions, including muck farm restoration, reduction of phosphorus loadings from within the UORB to "predevelopment" levels (but without reducing loadings from upstream tributaries), reductions in phosphorus concentrations in Apopka-Beauclair Canal discharges, development of a Lake Griffin marsh flow-way project, and combinations of the above alternatives. Responses were predicted for conditions during the period 1980–90 for Lakes Yale and Weir and 1984–90 for the other lakes.

Restoration of muck farms within the UORB is predicted to reduce equilibrium total phosphorus (TP) concentrations by about 50% in Lake Griffin, 20% in Lake Eustis, and no more than 10% in the other lakes in the basin. Attaining predevelopment TP loading is predicted to reduce equilibrium TP concentrations by

35–70% in Lakes Harris-Little Harris, Eustis, Griffin, Yale, and Weir, but would have a minor effect in Lakes Beauclair and Dora.

- Reduction in phosphorus concentrations in Apopka-Beauclair Canal discharges is predicted to reduce equilibrium TP concentrations by 50–70% in Lakes Beauclair and Dora and by about 20% in Lake Eustis, but would have a negligible effect on the other lakes.

- A combination of muck farm restoration and reduction in phosphorus concentrations in Apopka-Beauclair Canal discharges is predicted to reduce equilibrium TP concentrations by 30–72% in Lakes Beauclair, Dora, Eustis, and Griffin, but would have a minor effect on the other lakes.

- A combination of attaining predevelopment TP loading and reduction in phosphorus concentrations in Apopka-Beauclair Canal discharges would be the best approximation of nutrient loading to the lakes prior to the onset of cultural eutrophication. These actions are predicted to reduce in-lake equilibrium TP concentrations in all lakes by 35–77%.

- The trophic state models predict that a Lake Griffin flow-way would have to be operated at high flow rates (250–500 cfs) at the expected retention efficiencies to substantially reduce in-lake TP concentrations beyond that predicted for muck farm restoration. Operation of a flow-way at 500 cfs could achieve predicted in-lake equilibrium TP concentrations similar to levels predicted for attainment of predevelopment TP loading. The equilibrium TP concentrations and the time course of lake response to changes in net TP loading are strongly dependent on the magnitude of the net sedimentation coefficient (internal nutrient loading).

RECOMMENDATIONS

Recommendations for highest priority studies to develop a better understanding of the causes for eutrophication in the basin and for development of final Pollutant Load Reduction Goals (PLRGs) include

- Refinement of estimates of stormwater loading by development of finer-scale maps of stormwater runoff and incorporation of information on existing stormwater management facilities

- Studies of nutrient sedimentation rates and of internal nutrient recycling in the lakes

Recommended interim PLRGs for the UORB are a combination of reduction of muck farm discharges to levels expected from wetland areas with reduction in Apopka-Beauclair Canal TP concentrations to the levels expected under the Lake Apopka PLRG. These actions are expected to reduce equilibrium TP concentrations by 37–74% in the lakes of poorest water quality in the basin (Lakes Beauclair, Dora, Eustis, and Griffin) (based on loading estimates for 1986–90).

The development of final PLRGs for the basin should focus on reductions in pollutant loading from stormwater runoff. Lakes Eustis and Yale are expected to be most responsive to reductions in stormwater runoff. Lakes Harris-Little Harris and Weir are expected to be less affected by reductions in stormwater runoff, but maintenance or improvement in water quality in these lakes requires control of stormwater runoff. Other nutrient sources that should be considered in the development of PLRGs include citrus processing plant discharges (Lake Yale) and septic tank effluents (Lake Weir).

Implementation of a full-scale Lake Griffin flow-way project may significantly reduce nutrient levels in Lake Griffin, particularly if significant reductions in stormwater loading are not feasible.

Studies of sedimentary nutrient stores and internal nutrient cycling are necessary before decisions can be made regarding the necessity and efficacy of expensive restoration actions to reduce internal nutrient loading, such as sediment removal or treatment.

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MEASUREMENT ABBREVIATIONS

cfs	cubic feet per second
cm	centimeter
d	day
ft	feet
gal	gallon
g	gram
ha	hectare
in.	inch
kg	kilogram
km	kilometer
L	liter
m	meter
mg	milligram
mgd	million gallons per day
mo	month
µg	microgram
yd	yard
yr	year

INTRODUCTION

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 four Surface Water Improvement and Management (SWIM) programs within the St. Johns River Water Management District (SJRWMD). The Ocklawaha River is one of 10 major hydrologic units making up SJRWMD (Figure 1). The UORB comprises three of the subunits within the Ocklawaha River hydrologic unit: the Lake Harris unit, the Lake Griffin unit, and the Marshall Swamp unit (Figure 2). The UORB lies downstream from the Lake Apopka and Palatlakaha River units and discharges to the Lake Ocklawaha unit.

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 (Figure 3); only Lakes Weir and Yale have been classified in the less productive mesotrophic category (Shannon and Brezonik 1972; Canfield 1981). A statewide survey characterized Lakes Beauclair, Dora, Harris-Little Harris, Eustis, and Griffin as having trophic-related problems (Huber et al. 1982).

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 Florida Department of Environmental Protection (FDEP) Class III water quality standards (Fulton 1995). Development of effective restoration and regulatory programs to improve water quality in the basin requires a thorough understanding of the nature and causes of eutrophication. One of the most important diagnostic studies to be performed identifies nutrient sources and

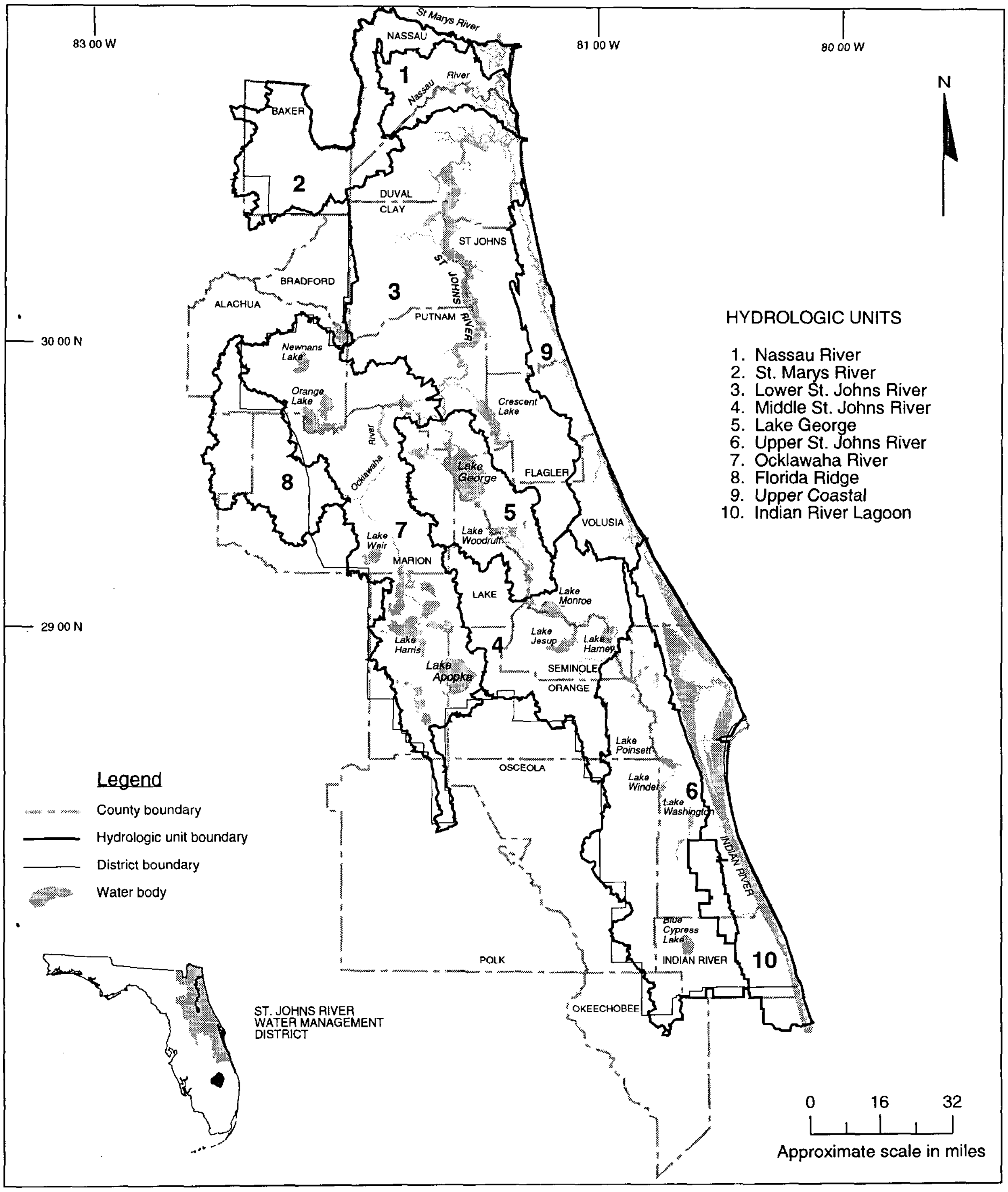

Figure 1. Major hydrologic units of the St. Johns River Water Management District

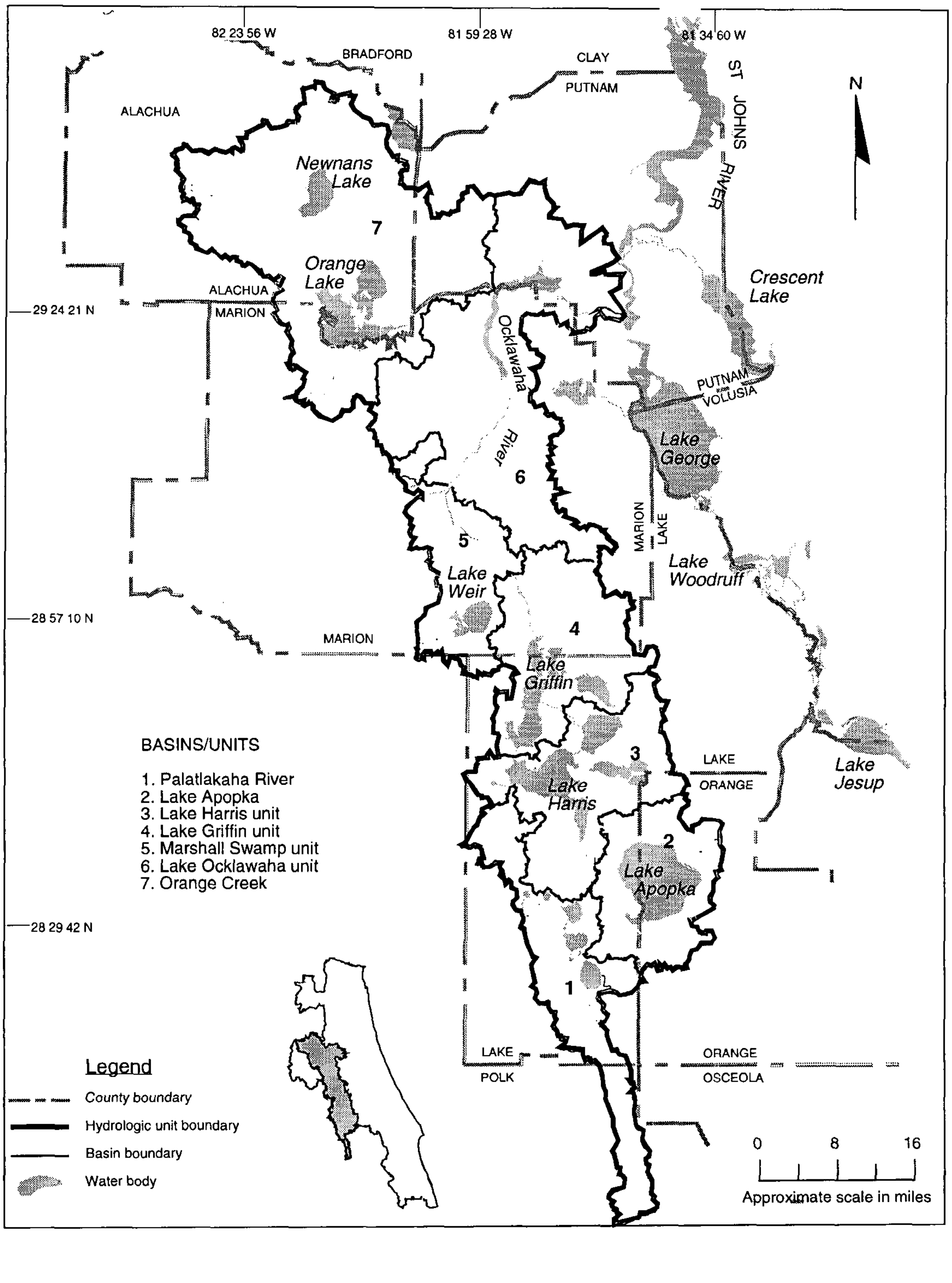

Figure 2. The Ocklawaha River hydrologic unit

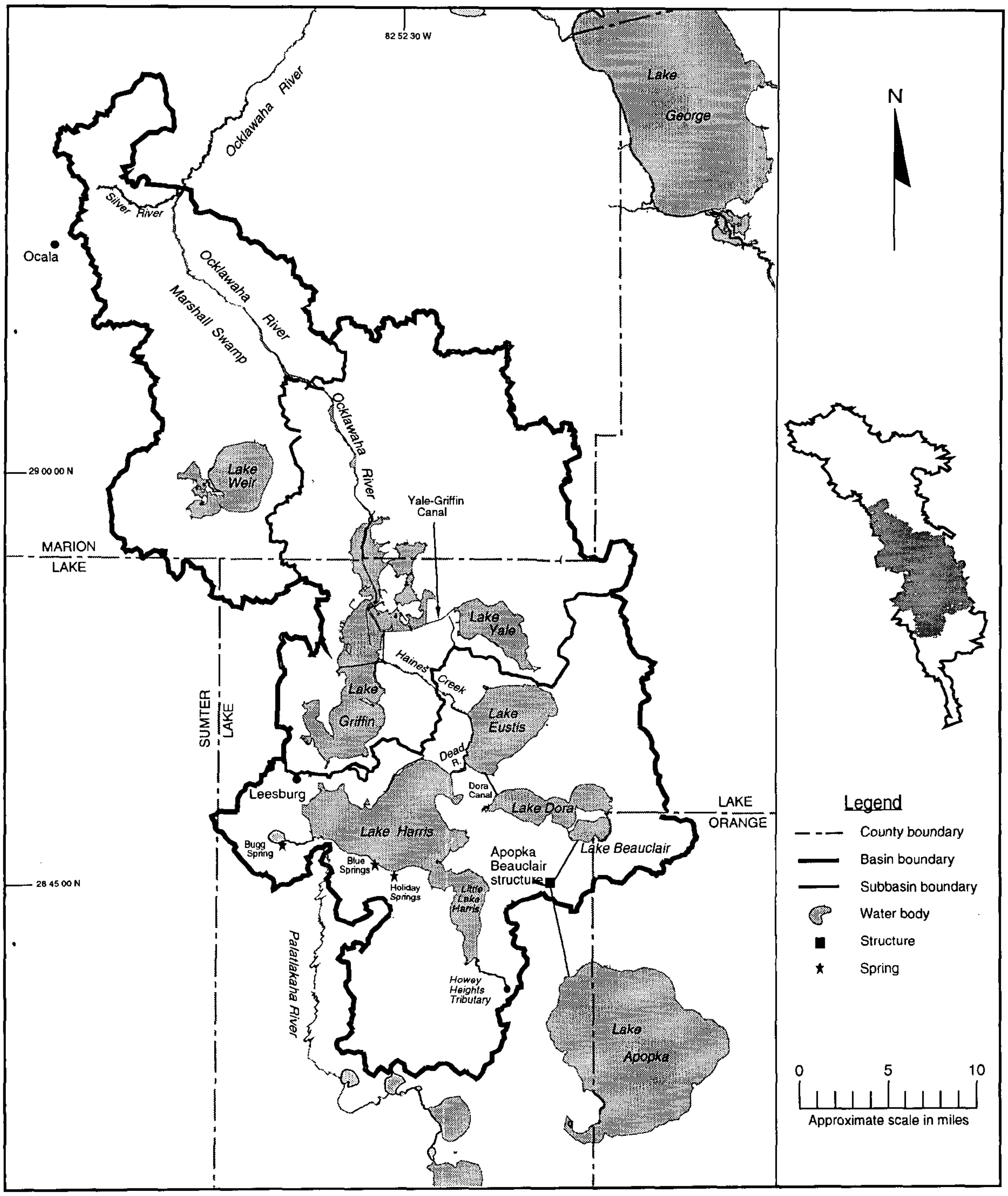

Figure 3. The upper Ocklawaha River Basin

quantifies the loading of nutrients to the lakes in the basin from the surrounding watershed. This "external nutrient budget" will allow development of the most cost-effective restoration and management strategies for water quality improvement.

Phosphorus is generally considered to be the primary nutrient that limits phytoplankton growth in temperate lakes; hence, cultural eutrophication of temperate lakes is primarily due to increases in external phosphorus loading. There have been relatively few controlled experiments to test for nutrient limitation in Florida lakes. Nitrogen was found to be the primary limiting nutrient in experimental assays in Lake Apopka (Schelske et al. 1992; Aldridge et al. 1993). Nutrient enrichment experiments in three other central Florida lakes produced evidence of limitation by both phosphorus and nitrogen, as well as some instances of limitation by iron and trace elements (Cowell and Dawes 1991). There have been no such analyses for lakes in the UORB. Standardized algal bioassays were conducted for Lake Yale as part of the U.S. Environmental Protection Agency's (EPA) National Eutrophication Survey (NES); these tests indicated phosphorus limitation (Baker et al. 1981). Similar tests for Lake Apopka also indicated phosphorus limitation (Baker et al. 1981), contrary to the results of experimental assays for this lake (Schelske et al. 1992; Aldridge et al. 1993).

Objectives of the Study

The external nutrient budget study will be conducted in two phases. Phase I (this report) uses existing land use, hydrologic, and water quality information to develop a preliminary input-output budget. Phase II will use targeted studies to fill major data gaps and refine the estimates of nutrient loading.

The objectives of this Phase I external nutrient budget study are

- To use ratios of nitrogen to phosphorus from water chemistry data to assess potentially limiting nutrients for the seven major lakes in the UORB (Lakes Beauclair, Dora, Harris-Little Harris, Eustis, Griffin, Yale, and Weir)

- To collate and analyze existing information on land use, hydrology, and water quality to develop preliminary phosphorus and nitrogen budgets for the seven major lakes in the UORB

- To use lake trophic state models to estimate effects of alternative restoration and management actions on external phosphorus loading and in-lake phosphorus concentrations

- To develop recommendations for interim pollutant load reduction goals (PLRGs) for the basin

- To identify major information needs and recommend further studies required to develop cost-effective restoration and management programs and formulate final PLRGs for the basin

DEVELOPMENT OF POLLUTANT LOAD REDUCTION GOALS

State water policy requires the development of PLRGs for all water bodies in the state, with the designated SWIM basins having highest priority for PLRG development. PLRGs are defined as estimated reductions in pollutant loadings needed to preserve or restore beneficial uses of receiving waters, with the ultimate primary purpose being that the water quality in receiving waters is restored or maintained consistent with applicable state water quality standards.

PLRGs are expected to be developed in two stages:

1. Interim PLRGs, which are best-judgment estimates of the levels of pollutant load reduction anticipated to result from planned corrective actions. Interim PLRGs are not necessarily intended to be sufficient for achieving and maintaining applicable water quality standards. They generally are based on preliminary estimates of pollutant loadings and represent interim programmatic steps taken until more intensive investigations can be completed.

2. Final PLRGs, which are intended to be sufficient for achieving and maintaining applicable water quality standards and which provide a basis for regulatory action, if necessary. These goals are based on thorough water body investigations, leading to a relatively high degree of confidence in the estimates of pollutant loading and the potential load removal efficiencies of planned corrective actions (FDEP 1993).

The SWIM programs are responsible for developing PLRGs for the SWIM water bodies. Interim PLRGs and schedules for the development of final PLRGs were to be included in updated SWIM plans by December 31, 1994 (FDEP 1993). Interim PLRGs developed in the present study were included in the March 1995 update of the UORB SWIM Plan (Fulton 1995). Recent changes to State Water Policy have eliminated the requirement for development of interim PLRGs, but final PLRGs are still required (C. Fall, pers. com. 1995). Final PLRGs for the UORB are scheduled to be developed by December 31, 1998 (SJRWMD 1995). Final PLRGs will then be incorporated into federal, state, and local permitting programs that regulate discharges into the water bodies.

DESCRIPTION OF THE STUDY AREA

This summary description of the basin is excerpted from the SWIM Plan for the UORB (Fulton 1995), which includes a more complete review of available information on the study area.

THE WATER BODY SYSTEM

The UORB is located in Marion, Lake, Orange, and Sumter counties of central peninsular Florida (Figure 3). The drainage basin encompasses 1,652 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 (Figures 2 and 3), which enters into Lake Harris, and the Lake Apopka subbasin, which drains into Lake Beauclair through the Apopka-Beauclair Canal (Figures 2 and 3). 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 also connected to Lake Griffin by the Yale-Griffin Canal. 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 subbasin 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 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.

HISTORICAL OVERVIEW OF DEVELOPMENT IN THE BASIN

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.

The impacts of urban development within the basin were first documented during the late 1940s. Eutrophication of the surface waters was the result of discharge of domestic, industrial, and agricultural wastes directly to receiving waters; destruction of aquatic habitat; and channelization. Declining regional water quality persists.

Much of the agricultural land around the major lakes and the Ocklawaha River is drained wetlands. These muck farms are often drained by interior drainage ditches, pump stations, and perimeter levees. Upland farms were chiefly developed for citrus groves, and these areas usually require minimal drainage.

The UORB has been affected by a number of events which have led to water quality degradation and loss of aquatic habitat. Table 1 presents a brief chronology of significant events that have affected the major lakes in the basin.

Table 1. Chronology of significant events in the study area

Year(s)	Event
1870–80	The Apopka Canal Company attempts to dredge a canal connecting Lakes Apopka, Beauclair, Dora, and Eustis to the Ocklawaha River to drain farmland and open a transportation route to ship vegetables and citrus
1890	Congress authorizes the Rivers and Harbors Act to provide a 4-ft channel from the mouth of the Ocklawaha River to Leesburg to facilitate navigation
1893	Canal connecting Lake Apopka through Lake Beauclair and Lake Dora to Lake Eustis was completed by the Delta Canal Company
1916	Rivers and Harbors Act includes provisions to construct a lock and dam at Moss Bluff to regulate water levels in Lake Griffin and accept private canals along Ocklawaha River in lieu of natural portions of the river bed
1920s	Direct discharge of primary and secondary sewage effluents and fruit processing wastes to the chain of lakes begins
1925	Construction of Moss Bluff lock and dam and dredging of the Ocklawaha River and Lake Griffin to Leesburg is completed by the U.S. Army Corps of Engineers under the Ocklawaha River Navigation Project
1942	Drainage water discharges from muck farms around Lake Apopka begin
1942–47	Expansion of agricultural activities in Lake Apopka Basin
1947	Hurricane disturbances in Lake Apopka; first algae blooms reported in Lake Apopka
1950	A wooden water control structure was constructed on the Apopka-Beauclair Canal by local interests to stabilize water levels on Lake Apopka and provide optimum levels for agricultural water supply and improved navigation
1956	A permanent water control structure was completed on the Apopka-Beauclair Canal by the Lake Apopka Authority to conserve and protect the water resources of Orange County
1957	Burrell lock and dam, located approximately midway along Haines Creek, was built by the Ocklawaha Basin Recreation and Water Conservation and Control Authority to stabilize water levels on Lakes Griffin, Eustis, Dora, Beauclair, and Harris and to provide optimum levels for agricultural water supply and improved navigation
1962	The Four River Basins Project was authorized by Congress under the Flood Control Act to provide for flood protection and solve water control problems
1967	Lake County Pollution Control is established
1969–74	U.S. Army Corps of Engineers, working on the Four River Basins Project, completes construction on Moss Bluff lock and dam, Lake Griffin to Moss Bluff levee and canal, and Moss Bluff to the north end of Oklawaha Farms agricultural area levee and canal
1969	A no-discharge rule was adopted by Lake County Pollution Control

Table 1—*Continued*

Year(s)	Event
1970s	The discharge of most wastes from sewage treatment, food processing, and industrial facilities ceases
1978	Construction of new Burrell lock and dam water control structure completed
1979	The Lake Griffin Recreational Area receives Outstanding Florida Waters designation
1984	Drawdown of Lake Griffin conducted
1985	Lake Apopka restoration project begins; feasibility and diagnostic studies initiated
1987	The Surface Water Improvement and Management (SWIM) Act becomes law
1988	Consent order with A. Duda & Sons to reduce nutrient loading to Lake Apopka
1989	SWIM plans for the upper Ocklawaha River Basin and Lake Apopka adopted by the SJRWMD Governing Board and approved by the Florida Department of Environmental Regulation
1989	Consent order with Zellwood Drainage & Water Control District to reduce nutrient loading to Lake Apopka
1990	Shad removed from Lake Denham to test for food-chain and nutrient removal effects
1991	Pilot-scale Lake Apopka demonstration marsh flow-way begins operation period to test efficiency of marsh filtration
1991–93	Emeralda Marsh muck farms acquired; flooding and gamefish stocking of properties started
1994	Lake Griffin marsh flow-way pilot project initiated

WATER QUALITY

The principal water quality problem in the UORB is eutrophication, which results from nutrient loading from intensive agriculture and rapid urbanization. In general, those lakes with relatively poor water quality are influenced by flow from hypereutrophic Lake Apopka (Figure 4). Although the discharge from Lake Apopka acts clearly as a point source of pollution for the downstream lakes, the significance of other point and nonpoint pollution sources in the basin is unclear. The water quality of Lake Beauclair and Lake Dora is rated poor. The water quality of Lake Eustis improves slightly to a fair rating. Lake Griffin is generally rated poor; this deterioration from upstream

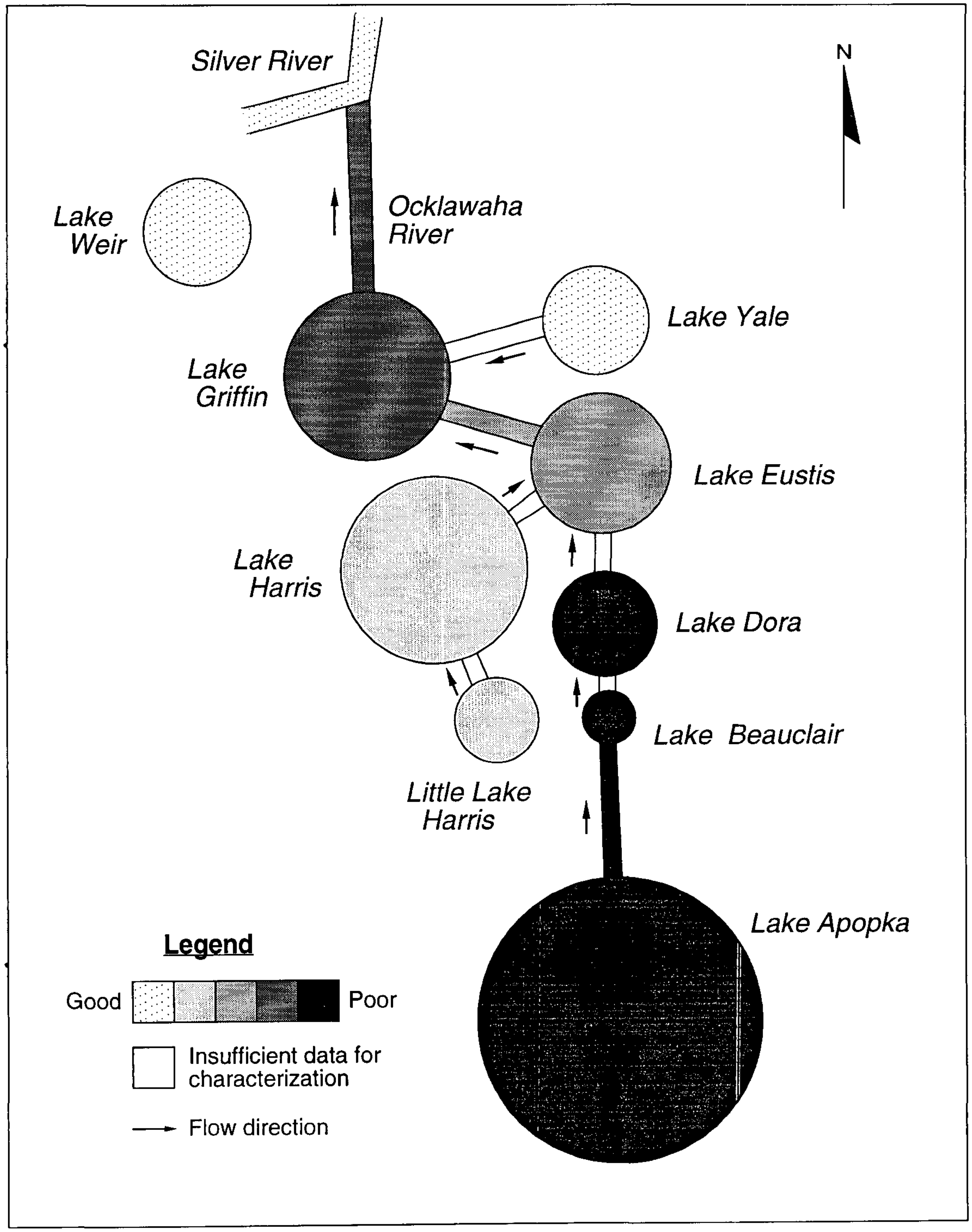

Figure 4. Schematic representation of water quality in the upper Ocklawaha River Basin

waters is perhaps due to discharges from muck farms in the drainage basin or runoff from the City of Leesburg.

Those lakes with relatively good water quality are not influenced by flow from Lake Apopka. Lake Harris-Little Lake Harris, with fair to good water quality, is influenced by high quality water flow from the Palatlakaha River. Lake Yale and Lake Weir, which have no major tributaries, exhibit good water quality.

EXISTING PROGRAMS FOR WATER QUALITY MANAGEMENT

The SWIM Program

The SWIM Act (Chapter 87-97, Laws of Florida) was enacted in July 1987 in response to growing concerns over environmental degradation of Florida's surface waters. The legislature recognized the state's responsibility to protect and enhance environmental and scenic characteristics of surface waters. Passage of the SWIM Act provided the direction and funding necessary to implement a statewide surface water management program. The state's five water management districts were mandated to (1) identify and prioritize significant water bodies in need of restoration or conservation and (2) plan, implement, and coordinate restoration and conservation strategies.

The SWIM legislation identified the Lake Apopka Basin as one of the highest priority basins in the state for development of a SWIM program. As a result of SJRWMD's priority setting process, SJRWMD subsequently also identified the UORB as having a high priority for restoration.

SWIM plans are required to demonstrate a cohesive set of strategies and programs to address the following central concerns of the SWIM Act:

- Point and nonpoint sources of pollution
- Destruction/restoration of the natural systems which purify surface waters and provide habitats

- Correction and prevention of surface water problems
- Research to provide a better scientific understanding of the causes and effects of surface water pollution and of the destruction of natural systems to better manage and improve surface waters and associated natural systems
- Interagency coordination in management
- Public awareness and education

SJRWMD has developed comprehensive plans for improvement and management of the priority SWIM subbasins in the Ocklawaha River Basin (Fulton 1995; Conrow et al. 1993). One of the priority issues to be addressed in the UORB SWIM program is excessive levels of nutrients in the water bodies in the basin. Diagnostic studies to develop a better understanding of the nature and causes of water quality problems in the basin are necessary to develop effective restoration and regulatory programs. As discussed previously, the SWIM programs are responsible for developing PLRGs for the SWIM water bodies.

Other Programs

Point and nonpoint surface discharges are regulated by a variety of local, regional, state, and federal agencies. Lake County prohibited surface discharges from waste treatment facilities in the 1970s. As a result, waste treatment plants in the basin are land application systems with no surface discharge. The only existing point source discharges to the lakes in the UORB are agricultural wastewaters from muck farms and weak wastes from several citrus processing plants.

In 1991, SJRWMD began regulating agricultural discharges under Chapter 40C-44, *Florida Administrative Code* (Agricultural Surface Water Management System rule). These drainage waters typically contain high levels of nutrients and other contaminants. A major focus of the regulatory effort involves the construction of retention pond/recycling systems such that the volume of water discharged off-site is decreased by 60–70%. In cases where the permitted discharges still cause downstream pollution problems, SJRWMD requires additional treatment before the water is discharged.

FDEP regulates the discharges from food processing plants. It is essential that there be a coordinated effort among federal, state, regional, and local regulatory agencies to monitor and upgrade these facilities as required for restoration of the river basin. Wasteload allocations for the four food processing plants in the basin may require re-evaluation following development of PLRGs for the basin.

Nonpoint sources are usually associated with land uses that do not create a discrete surface discharge. Sources of nonpoint pollution include urban and agricultural stormwater runoff, leachate from failed septic systems, contaminants associated with marinas, and leachate from landfills. Each of these nonpoint sources requires a different approach to reduce detrimental effects. Each nonpoint source is currently regulated by at least one agency; consequently, improved coordination of the regulatory effort is important. As with point source pollutants, PLRGs developed for the basin will be incorporated into future permits for nonpoint discharges.

SJRWMD has an active land acquisition program in the UORB, which currently focuses on acquisition of muck farms for habitat restoration and reduction of agricultural discharges (Table 2). To date, SJRWMD has acquired about 16,500 acres in the UORB. In addition, SJRWMD coordinates with active land acquisition programs maintained by county governments and the Lake County Water Authority.

Table 2. Muck farms discharging to the lakes in the upper Ocklawaha River Basin

Farm	Receiving Water Body	Acreage in Production	Treatment System	Active Operations during Study Period	Present Status
Hurley Muck Farm	Apopka-Beauclair Canal	420	10-acre pond, upland spray disposal	1980–90	Valid permit #44-069-0001M
Pine Meadows	Hicks Ditch, Lake Eustis	670	83-acre pond	1980–90	SJRWMD purchase September 1992
Springhill Farms	Hicks Ditch, Lake Eustis	228	--	1980–90	Valid permit #4-069-0281
JA-MAR	Lake Denham, Lake Harris	400	--	1980–90	Valid MSSW permit #4-069-0294
S.N. Knight, Leesburg	Lake Harris	423	47-acre pond	1980–90	SJRWMD purchase June 1991
Lowrie Brown North	Haines Creek	350	32-acre pond with baffle	1980–90	SJRWMD purchase October 1991
Lowrie Brown South	Haines Creek	625	Pasture	1980–90	SJRWMD purchase October 1991
S.N. Knight, Lisbon	Yale-Griffin Canal Lake Griffin Haines Creek	2,040	2 ponds, 71 acres total	1980–90	SJRWMD purchase June 1991
Walker Ranch	Yale-Griffin Canal	388	--	1980–82	SJRWMD purchase June 1991
Paulhamus (Eustis muck farm)	Yale-Griffin Canal Emeralda Marsh	600	75-acre pond	1980–90	SJRWMD purchase December 1993
Long Farms North	Yale-Griffin Canal	650	2 ponds, 46 acres total	1980–90	SJRWMD purchase July 1992

Source: Modica and Associates (n.d.); Fulton (1995); SJRWMD permit records

METHODS

GENERAL PROCEDURES FOR HANDLING WATER CHEMISTRY DATA

Water chemistry data obtained for the period 1977–91 were used in data analyses. The primary sources for water chemistry data were the EPA STORET data base and the SJRWMD data base. Original sources for surface water quality data included monitoring programs of SJRWMD, FDEP, the Florida Game and Fresh Water Fish Commission, the Lake County Environmental Services (LCES), and the U.S. Geological Survey (USGS) and an intensive study of the Ocklawaha chain of lakes conducted by Brezonik et al. (1981). Permit data files of SJRWMD, FDEP, and LCES provided water quality data for muck farm and other point discharges. Supplementary data sources used for other nutrient sources are described in later sections.

Nutrient budgets, and most other analyses, were developed using total phosphorus measured as phosphorus (TP) and total nitrogen measured as nitrogen (TN). Most data sources reported data in the form of TP except the Florida Game and Fresh Water Fish Commission, which reported total phosphorus measured as phosphate (PO_4). These data were converted to TP by multiplying by the ratio of the respective molecular weights of phosphorus to phosphate.

Nitrogen was generally reported as several components of TN. In these cases, TN was calculated using one of the following equations:

$$TN = TKN + NO_3 + NO_2 \qquad (1)$$

or

$$TN = N_{org} + NH_3 + NO_3 + NO_2 \qquad (2)$$

where:

TKN = total Kjeldahl nitrogen
NO_3 = nitrate nitrogen
NO_2 = nitrite nitrogen
N_{org} = organic nitrogen
NH_3 = ammonia nitrogen

Additionally, total inorganic phosphorus (TIP) (estimated using PO_4) and total inorganic nitrogen (TIN) were calculated for assessment of potential limiting nutrients by nitrogen to phosphorus ratios. TIN was calculated using the following equation:

$$TIN=NH_3+NO_3+NO_2 \tag{3}$$

The source data bases used various methods for handling nutrient concentrations below detection limits. Most commonly, the detection limit was reported with a "K" remark code (actual value known to be less than value given). In these cases, the nutrient concentration was assumed to be half the detection limit. In some cases, data below detection limits were apparently given a value of zero. When this occurred, the nutrient concentration was assumed to be half the lowest reported value. Finally, in a few cases, values were reported with a "T" remark code (value reported is less than the criteria of detection); in these cases, the value was used as reported.

A simplified procedure for handling data with values below the detection limit was used for the components of TN in surface water chemistry data. In these cases, inorganic forms of nitrogen were always very small (at least two orders of magnitude smaller) relative to TKN or organic nitrogen and were frequently below detection limits. In cases when the inorganic components of TN were below detection limits or were not reported, they were assumed to be zero in the formulas used to calculate TN. This assumption simplified procedures because summation formulas used in Lotus spreadsheets for calculations automatically treat values with associated remark codes as labels with a value of zero. This simplification was not employed for other nutrient sources or calculation of TIN, in which inorganic forms of nitrogen could be a significant portion of total nitrogen.

Occasionally, other remark codes were encountered in the data. If the remark code simply provided information about the datum (e.g., remark code "A"—value reported is the mean of two or more determinations), then the value was used as reported. Values were rejected if associated remark codes indicated some reason to question the accuracy of the data (e.g., remark code "Q"—sample held beyond normal holding time before analysis).

Total nitrogen data reported by LCES for the period May 1982 to July 1984 appeared to be anomalously low; reported values were consistently lower than data reported by other agencies for the same period or by LCES prior to or subsequent to this period. LCES staff cannot account for the anomalous data. There was a several month hiatus in the reporting of nitrogen data by LCES both before and after this time period, however, which may have allowed for some changes in procedures. LCES TN data for this period were deleted from the data sets used for analyses, except for one station on the Palatlakaha River. All data from the LCES Palatlakaha River station were used because there were limited data for this station and TN was uniformly low for the entire period of record.

Aside from the cases mentioned above, only a few anomalous data points were deleted from the data sets. Two reported values of TP that were more than two standard deviations greater than the mean for the LCES Palatlakaha River station were deleted from the data set. There were limited data for this station, and inclusion of these two points would have increased the average TP concentration by about 44%. Given the generally good water quality at this station and the absence of high values for TN on the same sample dates, it was decided that the two TP data points were likely in error. Similarly, one reported value of TP was deleted from the data set for the Haines Creek station. In this instance, the value was more than six standard deviations greater than the mean.

INFERENCE OF POTENTIAL LIMITING NUTRIENTS THROUGH N:P RATIOS

One method of assessing potential limiting nutrients is through nitrogen to phosphorus (N:P) ratios. Lakes with N:P ratios (by weight) of >30 are often considered to be potentially phosphorus limited, whereas N:P ratios <10 are taken to indicate potential nitrogen limitation, and intermediate ratios ($10 \leq$ N:P ≤ 30) indicate mixed nutrient limitation. This approach has been applied in Florida lakes to ratios of TN:TP (Huber et al. 1982; Hand and Paulic 1992), as well as to ratios of TIN:TIP (Baker et al. 1981; Cowell and Dawes 1991). To assess potential limiting nutrients for the lakes in the UORB, I calculated TN:TP and TIN:TIP ratios for surface water quality data from each of the lakes in the basin.

ESTIMATES OF NUTRIENT LOADING

Nutrient loading was estimated using the following general equation:

$$L = L_{Pr} + L_{Sp} + L_{Tr} + L_{MF} + L_{Ro} + L_{ST} + L_{PSI} \qquad (4)$$

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

These estimates of total mass loading include the major known nutrient sources for the seven study lakes. One potential nutrient source that is not included is ground water inputs other than that attributed to known spring discharges and septic tank seepage. Ground water inputs are expected to be minor, as Deevey (1988) has estimated a net loss of water to the underlying aquifer for a number of lakes in Florida, including Lakes Dora and Weir.

Estimates of loading of TP and TN were determined for Lakes Beauclair, Dora, Harris-Little Harris, Eustis, Griffin, Yale, and Weir. Estimates were calculated for some components of total mass loading for the years 1980–90. However, because of an absence of measurements of flows through the Burrell structure from 1980 to mid-1983, complete nutrient budgets could be calculated only for the years 1984–90 for most of the lakes in the basin.

General Description of EUTROMOD

The calculations of lake nutrient budgets were completed using a modification of EUTROMOD, version 2.50, a spreadsheet program for watershed and lake modeling (Reckhow 1991). EUTROMOD is available from the North American Lake Management Society, Madison, Wisconsin. The program calculates phosphorus and nitrogen loading and employs lake trophic state models to relate nutrient loading to in-lake nutrient concentrations and various other trophic state parameters. These lake trophic state models use empirical data obtained from Florida lakes during the NES. The program calculates annual nutrient loading and predicts average annual lakewide nutrient concentrations; it does not address shorter-term changes in nutrient loading or smaller-scale changes in lake trophic responses. The model also incorporates an uncertainty analysis of model predictions due to hydrologic variability and model error.

EUTROMOD was modified in several respects for the present study:

- The model was originally written to run using the shareware program AsEasyAs. It was modified for the present study to operate under Lotus, version 3.

- The program was modified to increase the number of land use categories and to incorporate information on runoff from different soil types.

- The program was written to calculate total nutrient runoff from some land use categories and separate estimates of dissolved and sediment-attached nutrient runoff for other land use categories. Insufficient information was available to calculate separate estimates of dissolved and sediment-attached nutrient runoff, so the program was modified to calculate total nutrient runoff for all land use categories. Some information on dissolved and sediment-attached nutrient runoff was incorporated into sections of the program that address sediment trapping as a function of watershed area.

- Nutrient runoff from several sources was calculated outside the EUTROMOD program and imported into the spreadsheet for calculation of the complete nutrient budget and lake trophic response.

- As mentioned previously, the program can perform an uncertainty analysis due to hydrologic variability, which is calculated based on the coefficient of variation of annual rainfall. However, rainfall is only one source of hydrologic variability for most of the lakes in the UORB. Another major source is tributary inputs from outside the immediate drainage basin, which were treated essentially as a point source in the nutrient budgets. Because the program's uncertainty analysis includes only a portion of the hydrologic variability for the UORB, I did not use that capability of the model. Instead, I examined the effects of hydrologic variability by calculating nutrient budgets for a period of several years varying substantially in hydrology.

Nutrient Loading from Atmospheric Deposition (L_{P_r})

Nutrient loading due to atmospheric deposition is the sum of nutrients delivered to the lake surface in rainfall and nutrients delivered to the lake surface during nonrainfall periods (dry deposition). There are several rain gauges in or near the basin that can be used to estimate the quantity of rainfall, but there were no measurements of nutrient concentrations in rainfall or of dry deposition for the UORB.

For most of the lakes, rainfall quantity was determined from records at the National Oceanic and Atmospheric Administration (NOAA) weather station at Lisbon, which is centrally located among the Ocklawaha chain of lakes. For Lake Weir, I used nearby SJRWMD rain gauges at Moss Bluff or at Bowers Lake for years that had complete rainfall records. For years in which these nearby stations did not have complete rainfall records, I used rainfall data from the NOAA weather station at Ocala.

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. Average TP and TN concentrations were determined for rainfall samples collected between April 1991 and July 1993, and average dry deposition was determined from samples collected at monthly intervals between May 1991 and May 1993. Although these samples were not collected during the time period for which nutrient budgets were calculated, there is evidence to believe that nutrient concentrations in atmospheric deposition have not significantly changed in recent years. I obtained long-term data sets of rainfall chemistry collected at the University of Central Florida in Orlando (period of record June 1977–December 1987) and the National Atmospheric Deposition Program site near the Kennedy Space Center (period of record August 1983–June 1989) from Thomas W. Dreschel, Bionetics Corporation. Graphical and regression analyses of these data sets indicated no or very weak temporal changes in concentrations of NO_3, NH_3, and PO_4 in these long-term data sets.

Wet and dry deposition were determined using the following equations:

$$L_{Prw} = C_r \times Pr_G \times A_L \tag{5}$$

and

$$L_{Prd} = \frac{C_{de} \times V_{de}}{A_{de} \times t_{de}} \times A_L \tag{6}$$

where:

- L_{Prw} = annual wet deposition of TP or TN
- C_r = average rainfall nutrient concentration from Lake Apopka marsh flow-way rain station
- Pr_G = annual rainfall (gross precipitation)
- A_L = lake surface area
- L_{Prd} = annual dry deposition of TP or TN
- C_{de} = average nutrient concentration in dry deposition samples from Lake Apopka marsh flow-way rain station
- V_{de} = sample volume of dry deposition samples
- A_{de} = surface area of dry deposition sample collection bucket
- t_{de} = exposure time for dry deposition samples

Nutrient Loading from Spring Discharges (L_{Sp})

There was limited information available from the SJRWMD data base or from published sources (Rosenau et al. 1977) on discharge volume or water quality of springs in the UORB, with no data collected since 1971. Known springs discharging into Lake Harris-Little Lake Harris (Figure 3) were sampled 3–5 times during 1991–92 to supplement the limited available data. Water quality samples were collected, and spring discharges were measured downstream of the main spring boil with a Teledyne Gurley flowmeter using ASTM Standard Method D-3858 for open-channel flow measurement (ASTM 1987). Discharge velocities from Bugg Spring were too low to be measured with this method. However, the owner of Bugg Spring, Mr. Joe Branham, regularly estimates discharges by measuring velocities of a neutrally buoyant container at several points across the width of the spring run. Mr. Branham also regularly collects water samples for analysis by the Lakewatch program. Monthly discharge measurements were obtained for Bugg Spring for the period June 1990–March 1993 from Mr. Branham, and monthly water quality data were obtained for the period January 1990–March 1991 from Lakewatch.

For each spring, average annual discharges were determined for each year measurements were available, and an overall mean discharge was determined from the annual averages. Mean TN and TP were determined for each spring from water quality samples collected from 1990 to 1992. For Bugg Spring, separate averages of the SJRWMD and Lakewatch water quality data were calculated, and an overall mean was calculated from the averages of the two data sources. Nutrient loading from each spring (L_{sp}) was assumed to be temporally invariant and estimated using

$$L_{Sp}=C_{Sp}\times S_i \tag{7}$$

where:

C_{Sp} = mean TN or TP concentration in spring discharges
S_i = mean spring discharge

Nutrient loadings from each spring were then summed to obtain overall nutrient loading from spring discharges.

Nutrient Loading from Tributary Inflows (L_{Tr})

Flow Volumes. Daily measurements of discharges were available from USGS gauge stations on several tributaries, including discharges into Lake Beauclair through the Apopka-Beauclair Canal, discharges into Lake Harris-Little Lake Harris from the Palatlakaha River, discharges from Lake Eustis into Lake Griffin through Haines Creek, and discharges from Lake Griffin at Moss Bluff on the Ocklawaha River. Discharge measurements from the Haines Creek gauge site were available from June 1983 to 1990; at the other sites, daily discharge measurements were available during 1980–90.

Discharge measurements are lacking on other tributaries in the basin. As a result, there are no direct estimates of flow into Lake Dora from Lake Beauclair, into Lake Eustis from Lakes Harris-Little Harris and Dora, or into Lake Griffin from Lake Yale. The best approach to estimating ungauged flows from upstream lakes would be to do a complete water budget for each lake, including measured tributary inputs, surface runoff, precipitation, evaporation, seepage, leakage to or from the aquifer, muck farm discharges and withdrawals, and tributary outflows. However, several of these components are not measured and would be difficult to estimate accurately.

A more feasible method of determining flows from upstream lakes was to make use of basinwide net runoff coefficients (*r*), determined from stream discharges at measured gauges and watershed areas:

$$r = \frac{\textit{stream discharge}}{\textit{watershed area}} \tag{8}$$

These net runoff coefficients incorporate all of the sources and losses of water that would be included in a detailed water budget. However, the drainage basin for each stream gauge includes more than one lake subbasin. For example, the Haines Creek Basin includes the subbasins for Lakes Beauclair, Dora, Harris-Little Harris, and Eustis. Therefore, the net runoff coefficients are averaged over the whole drainage basin for the stream gauge, rather than being specific for each lake subbasin.

More specifically, the UORB was divided into two drainage basins:

1. Haines Creek Basin—drainage area downstream of the USGS gauge sites on the Apopka-Beauclair Canal and the Palatlakaha River and upstream of the gauge site on Haines Creek. Major lakes in this basin include Beauclair, Dora,

Harris-Little Harris, and Eustis. The net runoff coefficient for the Haines Creek Basin was determined using the following equation:

$$r_{HC} = \frac{T_{HC} - T_{AB} - T_P - S_i + V_{HC}}{A_{HC}} \tag{9}$$

where:

r_{HC} = net runoff coefficient for Haines Creek Basin (m/mo)
T_{HC} = Haines Creek discharge (m^3/mo)
T_{AB} = Apopka-Beauclair Canal discharge (m^3/mo)
T_P = Palatlakaha River discharge (m^3/mo)
S_i = spring discharges into basin (m^3/mo)
V_{HC} = change in storage volume for lakes in Haines Creek Basin (m^3/mo)
A_{HC} = Haines Creek contributing watershed area (includes surface areas of lakes in the basin) (m^2)

2. Moss Bluff Basin—drainage area downstream of the Haines Creek gauge site and upstream of the Moss Bluff gauge site. Major lakes in this basin include Griffin and Yale. The net runoff coefficient for the Moss Bluff Basin was determined using the following equation:

$$r_{MB} = \frac{T_{MB} - T_{HC} + V_{MB}}{A_{MB}} \tag{10}$$

where:

r_{MB} = net runoff coefficient for Moss Bluff Basin (m/mo)
T_{MB} = Moss Bluff discharge (m^3/mo)
T_{HC} = Haines Creek discharge (m^3/mo)
V_{MB} = change in storage volume for lakes in Moss Bluff Basin (m^3/mo)
A_{MB} = Moss Bluff contributing watershed area (includes surface areas of lakes in the basin) (m^2)

Lake volume changes in the above equations were determined from end-of-month water elevations from the USGS stage gauges in each of the lakes. There was no stage gauge in Lake Beauclair, so it was assumed that water elevations for this lake were the same as in Lake Dora. Water elevations were converted to lake storage volumes by quadratic regression equations that were fit to hypsographic data for the lakes reported in Danek et al. (1991). In all cases, the regressions accounted for more than 99% of the reported variability in lake volume.

Runoff coefficients were calculated monthly based on reported discharges and lake stages. The net runoff coefficients were then used to estimate tributary discharges on tributaries without discharge gauges. For example, the estimate of discharge from Lake Harris-Little Lake Harris into Lake Eustis (T_H) was calculated using

$$T_H = T_P + S_i + (A_{Hw} \times r_{HC}) - V_H \tag{11}$$

where:

T_P = Palatlakaha River discharge (m^3/mo)
S_i = spring discharges into Lake Harris-Little Lake Harris (m^3/mo)
A_{Hw} = Lake Harris-Little Lake Harris contributing watershed area (including lake surface area) (m^2)
r_{HC} = net runoff coefficient for Haines Creek Basin (m/mo)
V_H = change in storage volume for Lake Harris-Little Lake Harris (m^3/mo)

Water Chemistry. Water quality data were available for the USGS gauge sites in the basin on the Apopka-Beauclair Canal, Palatlakaha River, Haines Creek, and Moss Bluff. There were insufficient water quality data for other tributaries, so data from monitoring stations in the upstream lake were used to estimate nutrient loading from tributary inflows. If nutrient concentration data were available for a month, then the average of samples in that month was used to calculate nutrient loading. For periods in which data were not available, a statistical analysis was used to estimate nutrient concentrations.

Nonparametric tests for seasonality and temporal trend were conducted with water chemistry data from each site, using the water quality analysis program, WQStat. Data were first tested for seasonality using a Kruskal-Wallis test for seasonality. If the Kruskal-Wallis test indicated a significant seasonality, then subsequent analyses were conducted using quarterly data. The "quarters" subjected to regression analysis were selected to maximize differences among seasons by inspection of plots of medians and variability of monthly data. As a result, the "quarters" were not necessarily equal in length, nor did they correspond to traditional seasons. For example, the "quarters" used in regression analyses of Lake Dora TP were December–March, April–July, August, and September–November.

The second step in the analysis tested for temporal trend using a seasonal Kendall test. If the Kendall test indicated a significant trend, then a linear regression analysis was conducted to predict nutrient concentration from the sample date. Data from tributary stations also were regressed against reported discharge. If neither regression were statistically significant, then mean values for the overall or seasonal data set were used as an estimate of nutrient concentrations during months with no data. If one or both of the regressions were significant, then the regression that accounted for the largest amount of variability in nutrient concentration was used to estimate nutrient concentrations during months with no data.

Nutrient Loading. Nutrient loading from tributary inflows (L_{Tr}) was estimated on a monthly basis using the following equation:

$$L_{Tr} = T_i \times C_{Tr} \qquad (12)$$

where:

T_i = reported or estimated tributary discharge

C_{Tr} = reported or estimated tributary TN or TP concentration

Nutrient Loading from Muck Farm Discharges (L_{MF})

Although a number of the muck farms have been acquired recently by SJRWMD (Table 2), all were reported to be in operation for the entire period covered by the present study except for Walker Ranch.

Flow Volumes. Limited information on discharges from muck farms in the UORB was available from SJRWMD permit records. Multiple regression was used to estimate discharge volumes during time periods with missing data and for farms lacking discharge information. Discharge data used in the regression analysis included available monthly discharge volume data for UORB muck farms obtained from SJRWMD permit records and discharges reported for A. Duda & Sons, Lake Jem Farm in the Lake Apopka Basin (Applied Technology and Management 1988). Data from A. Duda & Sons (2,350 acres in production) were used to increase the range of farm sizes used in the regression analysis, as UORB farms for which data were available included a narrow range of sizes (350–670 acres).

Independent variables included in the regression analysis were acres in production, rainfall, evaporation, perimeter adjoining wetland or open water areas, perimeter adjoining upland areas, and perimeter adjoining other muck farms. Perimeter lengths were estimated from USGS quad maps. Evaporation was estimated by multiplying pan evaporation measurements from area weather stations by a pan evaporation coefficient of 0.8 (Knochenmus and Hughes 1976).

Some of the muck farm pumps discharge into retention ponds (Table 2). There are no measurements of discharge from these retention ponds into the adjacent water bodies. In these cases, pond discharge was estimated by constructing a simple water budget for the ponds. Pond discharge was estimated as pump discharge into the pond plus direct net precipitation to the pond (this assumes that seepage inflows or losses are negligible):

$$MF_{ponddis}=MF_{pumpdis}+(A_p \times Pr_N) \tag{13}$$

where:

$MF_{ponddis}$ = monthly muck farm pond discharge (gal/mo)
$MF_{pumpdis}$ = monthly muck farm pump discharge (gal/mo)
A_P = retention pond surface area
Pr_N = net precipitation

Net precipitation (Pr_N) was estimated from rainfall and evaporation records for weather stations in the basin using the following equation:

$$Pr_N=Pr_G-(E \times 0.8) \tag{14}$$

where:

Pr_G = rainfall (gross precipitation)
E = pan evaporation
0.8 = pan evaporation coefficient (Knochenmus and Hughes 1976)

Application of these procedures occasionally resulted in negative values for muck farm discharge. Negative values occurred more frequently for the smaller farms. In months with negative estimates for muck farm discharge, discharge volume was assumed to be zero.

Water Chemistry. A limited amount of water chemistry data were available from SJRWMD and LCES permit records for discharges from several of the muck farms. For farms with retention ponds, water chemistry data used were for samples of the pond discharge. Following procedures outlined above for analysis of tributary flows, I combined data from all muck farms and tested the data for seasonality and temporal trends. Next, I tested the water chemistry data for differences among farms. There was a significant heterogeneity of variance (Bartlett's test) that could not be eliminated by log transformation, so I used a nonparametric Kruskal-Wallis test for differences among farms.

Nutrient Loading. Nutrient loading from muck farm discharges (L_{MF}) was estimated on a monthly basis using the following equation:

$$L_{MF}=(MF_{pumpdis} \text{ or } MF_{ponddis})\times C_{MF} \tag{15}$$

where:

$MF_{pumpdis}$ = reported or estimated muck farm pump discharge
$MF_{ponddis}$ = estimated muck farm retention pond discharge
C_{MF} = reported or estimated muck farm discharge TN or TP concentration

Nutrient concentrations (C_{MF}) chosen are discussed in the results section following the statistical analysis of the data.

Nutrient Loading from Stormwater Runoff (L_{Ro})

Nutrient loads from stormwater runoff from land uses other than muck farms were calculated by estimating the volume of potential runoff from contributing drainage areas and multiplying by a nutrient concentration. In the modified EUTROMOD nutrient loading model, stormwater runoff is a function of land use, which affects runoff quantity and quality, and soil type and rainfall, which influence runoff quantity. In addition, EUTROMOD allows for estimation of trapping of sediment-bound nutrients within the watershed during transport from the runoff source area to the mouth of the watershed.

A geographic information system (GIS) was used to estimate areas of land use/soil combinations within the drainage basin for each lake. The drainage basins were taken from a delineation previously produced by the Engineering Division of SJRWMD for use in developing a hydrologic model for the UORB. The USGS basin maps developed in 1947 and field checked in 1948–49 were used as base maps for the Ocklawaha River drainage basin maps. Other sources used to develop the drainage basin maps were USGS 1:24,000 quad maps, maps supplied by Lake County that showed drainage structures and drainage basins for the Upper Ocklawaha chain of lakes, and other available information on current drainage conditions.

The drainage basin for each lake was divided into a limited number of contributing secondary subbasins (ranging from 1 subbasin for Lake Weir to 29 subbasins for Lake Harris-Little Lake Harris (Figure 5). With the exception of Lake Weir, each drainage basin consists of a single subbasin (two subbasins in the case of Lake Harris-Little Lake Harris) which completely surrounds the lake, and one or more peripheral subbasins.

Land Use Data. Two GIS land use maps were used in estimating stormwater runoff. Land uses in the UORB were quantified by SJRWMD by updating 1973 land use maps prepared by the Center for Wetlands, University of Florida, with 1984 color-infrared aerial photography (National High Altitude Photography, USDA). The updating was confined to mapping new developments in the basin subsequent to the 1973 maps. These 1984 land use maps were used to estimate stormwater runoff in the years 1980–85.

Two changes were made in the 1984 land use maps. First, one of the muck farms in the basin (S.N. Knight, Leesburg) was incorrectly identified as wetlands, so an area equal to the reported acreage of the farm was converted from wetlands to muck farm. Second, Walker Ranch was operated as a muck farm only through 1982 (Table 2). Beginning in 1983, the acreage of Walker Ranch was converted to wetlands for use in estimating stormwater runoff.

For the years 1986–90, stormwater runoff was estimated using land use maps developed by Geonex Martel for SJRWMD. Mapping of Lake County was delineated from 1:24,000

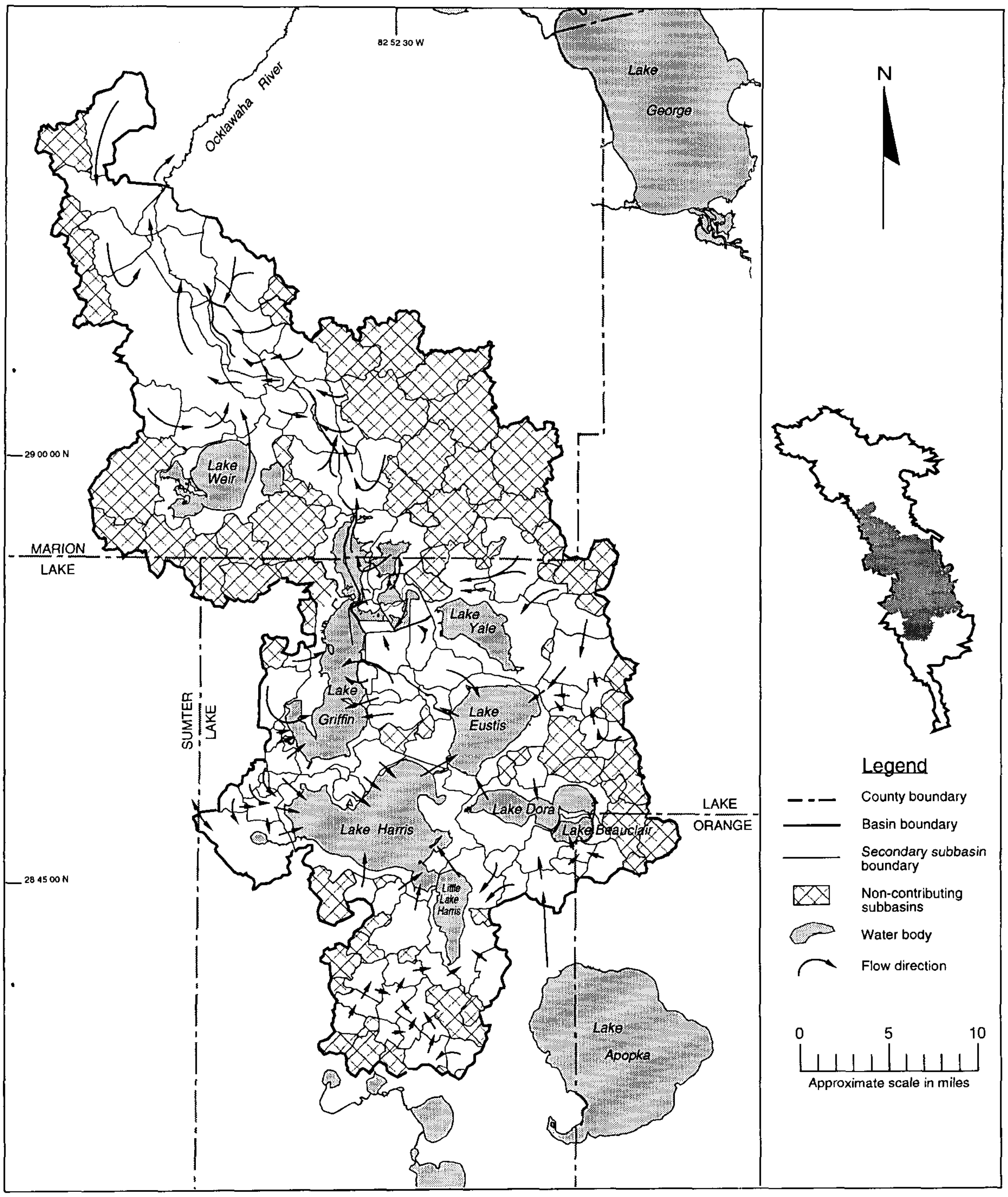

Figure 5. Secondary subbasins and direction of surface water flow in the upper Ocklawaha River Basin

color-infrared photography flown in 1987. Mapping of Marion County was delineated from 1:24,000 black-and-white aerial photography flown in 1989. A small portion of the basin is in Sumter County, outside SJRWMD boundaries. Mapping for the area in Sumter County was obtained from the Southwest Florida Water Management District; this area also was mapped by Geonex Martel.

Both land use maps used the Florida Land Use and Cover Classification Scheme (FLUCCS) (FDOT 1985). To prepare the land use data for use in the model, a simplified land use category coverage was created by aggregating all land cover classes into 12 categories (Table 3 and Appendix A). Muck farms were distinguished as an additional land cover class (Appendix A), but nutrient loading from muck farm areas was estimated by different procedures, as discussed in a previous section.

Soils Data. Soil types were mapped by SJRWMD from county soil surveys (USDA SCS and UF Ag. Exp. Sta. 1975, 1979). The soil maps aggregated individual soils into soil associations. For the purpose of estimating nutrient runoff, the soil associations were further aggregated into the Soil Conservation Service (SCS) hydrologic soil groups A, B, C, and D. Type A soils consist mainly of well-drained sands with high infiltration rates, resulting in a low runoff potential. Type B soils have moderate infiltration rates and consist mainly of moderately fine to moderately coarse textured soils. Type C soils are poorly drained soils with low infiltration rates and a high runoff potential. Type D soils have a very high runoff potential with low infiltration rates and consist mainly of clay soils with a high swelling potential, soils with a permanent high water table, and shallow soils over nearly impervious material. Dual soil classes such as B/D also occur, indicating a Type D soil where drainage improvements may upgrade the hydrologic characteristics to a Type B soil (SCS 1972).

Each soil association consists of a number of different soil types with different soil hydrologic group rankings. I determined a composite hydrologic soil group for each soil association by

Table 3. Nutrient concentrations and runoff coefficients used for estimating nutrient loading from stormwater runoff from land use/soil combinations

Land Use	Total P (mg/L)	Total N (mg/L)	Lower Bound Runoff Coefficient	Upper Bound Runoff Coefficient	Hydrologic Soil Group		
					A Runoff Coefficient	B Runoff Coefficient	D Runoff Coefficient
Residential low-medium density	0.300	2.290	0.30	0.50	0.30	0.367	0.50
Residential high density	0.470	2.220	0.50	0.70	0.50	0.567	0.70
Commercial	0.268	1.886	0.60	0.90	0.60	0.700	0.90
Institutional	0.150	1.180	0.65	0.95	0.65	0.750	0.95
Industrial/mining	0.270	1.638	0.40	0.70	0.40	0.500	0.70
Recreation/open land	0.053	1.250	0.10	0.30	0.10	0.167	0.30
Forest/rangeland	0.053	1.250	0.10	0.30	0.10	0.167	0.30
Open water/wetlands	0.163	1.450	0.33	0.40	0.33	0.353	0.40
Confined feedlots	47.500	127.100	0.31	0.64	0.31	0.420	0.64
Pasture	0.387	2.477	0.15	0.40	0.15	0.233	0.40
Citrus groves	0.140	2.050	0.15	0.40	0.15	0.233	0.40
Other agriculture	0.471	2.680	0.15	0.40	0.15	0.233	0.40

calculating a weighted average for the major soil types in the association, using data on composition of soil associations from county soil surveys (USDA SCS and UF Ag. Exp. Sta. 1975, 1979). No information was available regarding drainage improvements, so dual soil classes were classified as the more impervious class.

Rainfall. Rainfall data used in estimates of stormwater runoff were taken from rain gauges in the basin, as described previously in the section on nutrient loading from atmospheric deposition.

Runoff Volume Coefficients. Rainfall volume was converted to runoff volume by applying a runoff coefficient to the reported annual rainfall. Upper and lower limits for runoff coefficients for the 12 land use categories (Table 3) were derived from standard references (Phillips 1981; Wanielista and Yousef 1992) but agree well with runoff coefficients estimated from Florida studies (Harper 1992).

Because runoff depends on soil type as well as on land use, a runoff coefficient for each land use/soil combination (R_{LS}) (Table 3) was determined using the following equation (SWFWMD 1990; Adamus and Bergman 1993):

$$R_{LS} = LLC + (ULC - LLC) \times X \qquad \textbf{(16)}$$

where:

LLC = lower limit runoff coefficient for a particular land use
ULC = upper limit runoff coefficient for a particular land use
X = 0 for soil Type A
⅓ for soil Type B
⅔ for soil Type C
1 for soil Type D

Nutrient Concentrations. Appendix B presents a summary of literature values on nutrient concentrations in runoff originating from areas with a homogeneous land use. Most of the data are from studies conducted in south and central Florida (except for runoff from confined animal feedlots). Nutrient concentrations used to estimate stormwater runoff in the UORB (Table 3) are averages of the studies listed in Appendix B.

Nutrient Trapping in the Watershed. The movement of nutrients from runoff source area to the receiving water body is a complex process that involves deposition, resuspension, adsorption, chemical reaction, biological uptake, desorption, and decay (Haith and Tubbs 1981; Reckhow et al. 1990, draft). EUTROMOD allows users to divide the drainage area into attenuation zones with differing trapping efficiency resulting from natural or constructed sediment traps.

In watershed nutrient loading models, it is generally assumed that dissolved nutrients are transported without attenuation, whereas transport of sediment-attached nutrients from source areas is determined by a watershed sediment delivery ratio (Haith and Tubbs 1981; Delwiche and Haith 1983, Reckhow et al. 1990, draft). In watershed nutrient loading models, it is further assumed that sediment yield at the mouth of the watershed is equal to the amount delivered from the watershed to the tributary channel (Reckhow et al. 1990, draft).

Sediment delivery ratios have been developed based on distance from the source area to the watercourse or based on watershed area (Reckhow et al. 1990, draft). Although the ratios developed were not based on studies in Florida, they probably represent the best approach to estimate sediment delivery with the limited available information. In the present study, I used an area-based sediment delivery ratio to estimate trapping of sediment-attached nutrients in the watershed. I chose an area-based ratio because subbasin drainage areas were readily available from the GIS drainage basin maps. Sediment delivery ratios that were calculated based on estimated distances from watercourses were very similar to the area-based ratios. The sediment delivery ratio (*SD*) was calculated (Vanoni 1975; Delwiche and Haith 1983) as

$$SD = 0.47 \times A_{AZ}^{-0.125} \tag{17}$$

where:

A_{AZ} = attenuation zone area (km^2)

All of the nutrient concentrations used to estimate stormwater runoff are for total nutrients (Table 3). To make use of sediment delivery ratios, it is necessary to estimate the proportion of total nutrients that are sediment-attached. Table 4 presents a summary of literature estimates of the proportions of total phosphorus that are either dissolved phosphorus or ortho-phosphate. Appendix C presents a more detailed literature survey of dissolved and total phosphorus concentrations in stormwater runoff. I was unable to find similar literature estimates for dissolved and total nitrogen concentrations in stormwater runoff.

Great variability exists among studies, with the Florida sites generally reporting higher proportions of dissolved fractions (Table 4). However, in general, the proportions of dissolved phosphorus fluctuate around 0.5. Therefore, to make use of sediment delivery ratios, I assumed that the proportion of stormwater TP and TN that is sediment-bound is 0.5. Making this assumption, nutrient delivery ratios (*ND*) for attenuation zones were calculated using the following equation:

$$ND = (SD \times 0.5) + (1 \times 0.5) \tag{18}$$

where:

SD = sediment delivery ratio for sediment-attached nutrients
0.5 = proportion of total nutrient concentration that is sediment-attached or dissolved
1 = delivery ratio for dissolved nutrients (i.e., assume no losses of dissolved nutrients during transport from runoff source)

Table 4. Summary of reported studies of particulate and dissolved fractions of phosphorus in stormwater runoff. *Values reported are the proportion of total phosphorus that is either dissolved phosphorus or ortho-phosphate (see Appendix C).*

Land Use	Mean Proportion Dissolved Phosphorus
Agriculture—Florida sites	0.623
Agriculture—non-Florida sites	0.300
Mixed urban-agriculture—Florida sites	0.399
Mixed urban-agriculture—non-Florida sites	0.216
Forest—non-Florida sites	0.444
Land Use	**Mean Proportion Ortho-phosphate**
Agriculture—Florida sites	0.610
Agriculture—non-Florida sites	0.142
Residential/urban—Florida sites	0.354
Forest—Florida sites	0.725
Forest—non-Florida sites	0.097

The drainage basin for each lake was divided into two attenuation zones; one included the subbasin surrounding the lake (Zone 1), and the other included the peripheral contributing subbasins (Zone 2) (Figure 5). For Attenuation Zone 2, the sediment delivery ratio was calculated using the average area for subbasins in the attenuation zone (contributing subbasins border or are crossed by tributaries, so it was assumed that attenuation occurs during transport from the source of runoff to the tributary, but not after runoff enters a tributary). The area of Attenuation Zone 1 is often rather large, which would result in substantial attenuation if an area-based sediment delivery ratio was calculated. In fact, application of the area-based formula would often result in the counterintuitive result that nutrient trapping is greater in Attenuation Zone 1 than in the more distant Attenuation Zone 2. However, area-based ratios do not take into account that the subbasins immediately surrounding the lake are rather narrow bands (Figure 5) that are intersected by a number of tributaries, canals, and stormwater drainage systems. Therefore, I assumed that the sediment delivery ratio for Attenuation Zone 1 was 1 (i.e., no trapping of sediment-attached nutrients).

Nutrient Loading. Annual nutrient loading from each land use/soil hydrologic group combination was calculated using the following equation:

$$L_{LS}=A_{LS}\times Pr_G\times R_{LS}\times C_{RL}\times ND \tag{19}$$

where:

L_{LS} = nutrient loading from land use/soil combination
A_{LS} = area of land use/soil combination
Pr_G = annual rainfall
R_{LS} = runoff coefficient for land use/soil combination
C_{RL} = nutrient concentration for runoff from land use
ND = nutrient delivery ratio for the attenuation zone in which the land use/soil combination occurs

Nutrient loadings were then aggregated by land use to obtain total annual nutrient loading from stormwater runoff from each land use in the drainage basin (L_{Ro}).

Nutrient Loading from Septic Systems (L_{ST})

Nutrient loading from septic systems depends on the number of person-years that septic tank use impacts the lake, the per capita nutrient load to septic systems, and the retention of nutrients by soils in the tile drainage systems.

Capita-years. The number of persons contributing to septic systems that impact a lake is expressed in number of capita-years, which is the product of the number of septic systems and the number of person-years per unit.

Only septic systems from structures within 200 m of the lake shore or within 200 m of canal systems in lakeshore developments were considered to contribute nutrients to the lake (Reckhow et al. 1980; Reckhow and Chapra 1983). Aerial photos (1986–90, 1:200 or 1:400 scale) were used to estimate the number of structures served by septic tanks within 200 m of the lake shore or canals. County and municipal comprehensive plans and additional information provided by Lake County were used to determine service areas of municipal and package treatment plants. It was assumed that all other areas are serviced by septic tanks. This assumption may underestimate septic tank inputs, as there are some septic systems operating within the service areas of municipal waste treatment plants.

County comprehensive plans present information on household size. The Marion County Comprehensive Plan (Marion County 1991) gives an average of 2.18 persons per dwelling unit in 1987. This estimate appears to take into account both permanent and seasonal residents. The Lake County Comprehensive Plan (Lake County 1990) gives an average of 2.51 persons per dwelling unit for the unincorporated part of the county in 1980. However, this estimate considers only permanent residents, whereas the seasonal population averaged 36.5% of the total population in the unincorporated part of the county during the 1980s. Average household size that incorporates seasonal residents for Lake County was estimated by assuming that average household size for seasonal residents was also 2.51 and that seasonal residents spent an average of 0.5 years in residence:

$$\begin{aligned} \textit{Mean Household size} &= (2.51 \times 0.635) + (2.51 \times 0.365 \times 0.5) \\ &= 2.05 \end{aligned} \tag{20}$$

For other structures with septic tanks in the lakeshore areas (two restaurants, two motels with less than 10 rooms, three clubhouses), I assumed an average usage of 10 person-years/unit.

Per Capita Nutrient Loading. Values for per capita nutrient loading were taken from a review of nutrient loads for household water discharged into septic tanks compiled by Reckhow et al. (1980):

TP	1.48 kg/capita/yr (mean of eight studies)
TN	4.75 kg/capita/yr (mean of seven studies)

Soil Nutrient Retention. Generally, phosphorus in septic tank effluents is effectively retained in underlying soils, and only low concentrations will typically be introduced into the ground water (Jones and Lee 1977; Canter and Knox 1985), although there are some reports of elevated phosphorus concentrations at some distance from drain fields (Canter and Knox 1985; Miller 1992). Septic tanks are ineffective in nitrogen removal, primarily converting organic nitrogen to ammonium. Nitrogen in the form of nitrate usually reaches ground water and becomes very mobile because of its solubility and anionic form (Canter and Knox 1985).

Metcalf and Eddy (1979) reported 45% soil retention of nitrogen. Sherwood and Crites (1984) reported nitrogen removal efficiencies of 50–92.5% for a variety of land treatment systems, whereas phosphorus concentrations approach background levels within several hundred feet of the source.

Soil Hydrologic groups in the watershed for the UORB are predominantly Type A or B (Figure 6). There are few restrictions on septic tank use for these soil types. Type A and B soils are generally well drained and are generally acid soils which would tend to increase nutrient retention. However, they are generally sandy soils with rapid permeability which could lead to ground water contamination (USDA SCS and UF Ag. Exp. Sta. 1975, 1979; Reckhow et al. 1980). The Type D hydrologic soil group makes up 1.7–30.8% of residential areas in subbasins immediately surrounding the lakes (Attenuation Zone 1). These latter soil types have severe restrictions on septic tank usage and are poorly to very poorly drained. Although no field surveys of septic tank operation are available for the basin, there have been reports of flooded septic tank drainage fields when lake stages reach near the highest elevation of current regulation schedules. Nitrogen concentrations in spring discharges in the basin are substantially elevated compared to earlier measurements (see Results chapter). These increases could be due to increasing septic tank usage in the basin.

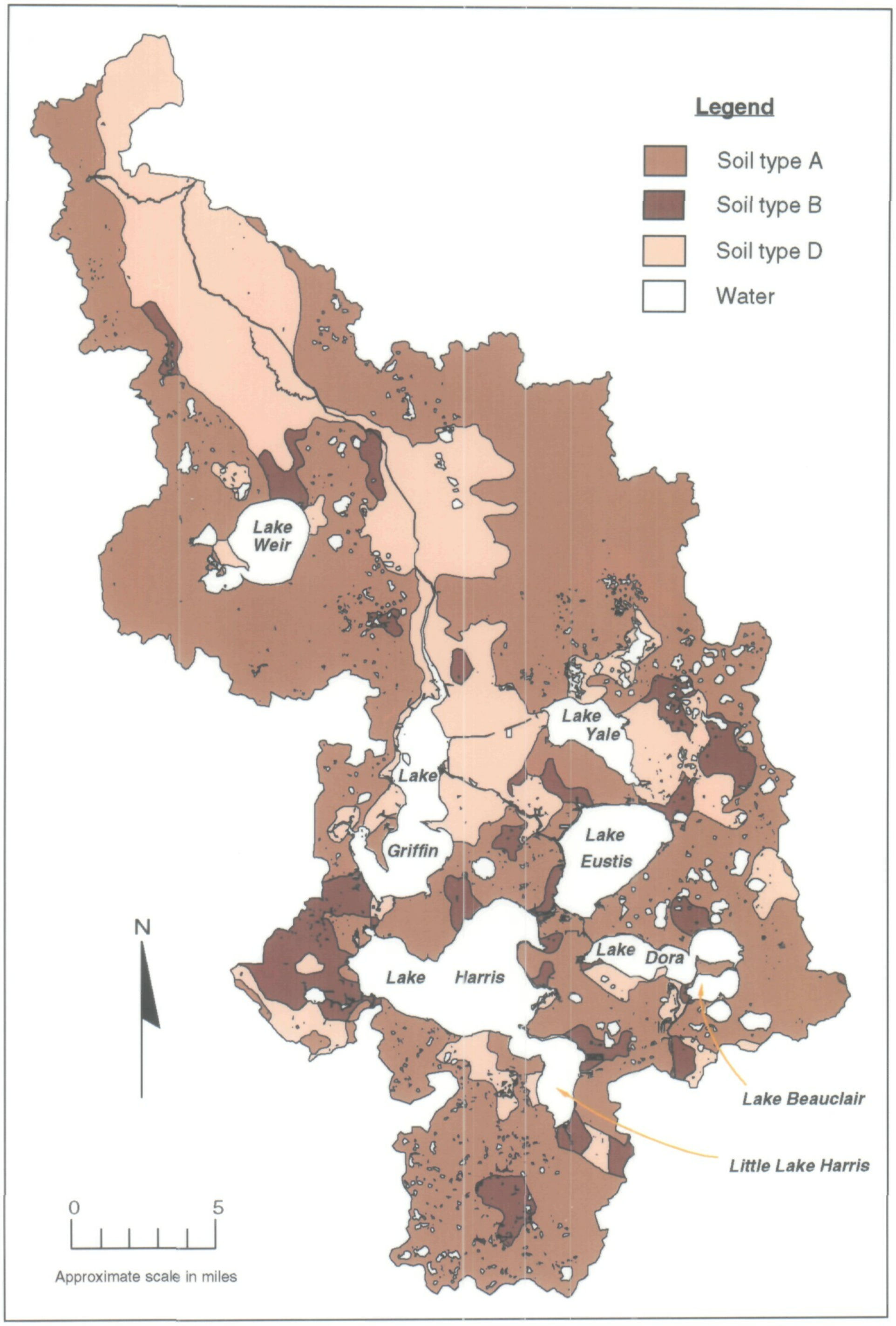

Figure 6. Soil hydrologic group map for the upper Ocklawaha River Basin
(see Table 15 for soil type codes)

Because literature data generally indicate highly effective retention of phosphorus by septic tank soil drainage systems, I assumed a soil retention coefficient of 0.90 (90% retention) for phosphorus. Literature data indicate that septic tank soil drainage systems are less effective in retention of nitrogen, and increases in nitrogen concentrations in spring discharges indicate potential ground water contamination in the basin. Therefore, I assumed a soil retention coefficient for nitrogen on the low end of the reported range, 0.45 (45% retention).

Nutrient Loading. Nutrient loading from septic systems (L_{ST}) is calculated using the following equation:

$$L_{ST}=(L_{Cap}\times CY)\times(1-SR) \qquad \textbf{(21)}$$

where:

L_{Cap} = nutrient load to septic systems per capita-year
CY = number of capita-years in watershed serviced by septic systems impacting the lake
SR = soil retention coefficient

As mentioned previously, there have been no field surveys of septic tank operations in the basin. However, Ritter and Herrera (1994, draft) have estimated the number of septic tank drainfields in the Ocklawaha chain of lakes that may be inundated when lakes are at the highest elevation of the current regulation schedules from ground elevations of houses in the 100-year floodplain. For Lake Yale, which is unregulated, estimates of inundated drainfields used the highest lake stage observed during the period 1980–90. I estimated an upper bound to septic tank nutrient loading, by assuming 0% soil retention of nutrients released by these inundated septic tank drainfields.

Nutrient Loading from Point Source Discharges (L_{PSI})

The only point source discharges in the UORB are weak waste discharges from four citrus processing plants. Weak wastes produced by the citrus plants consist primarily of noncontact cooling water and do not include chemicals involved in the citrus processing. Some information on weak waste discharges was available from LCES and FDEP permit files. Although water chemistry data for weak waste discharges were available during the study period (1980–90), data on discharge volume were available only after the beginning of 1990. Nutrient loading from each citrus processing plant was estimated by multiplying the average nutrient concentration in weak waste discharges by the average reported discharge volume.

LAKE TROPHIC RESPONSES

Lake Morphometry and Hydrology

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. Lake area, volume, and mean depth (Table 5) were taken from a recent bathymetric study of the lakes (Danek et al. 1991).

Table 5. Morphometric data for the major lakes in the upper Ocklawaha River Basin

Lake	Elevation (feet NGVD)	Surface Area (km^2)	Volume* (10^6 m^3)	Mean Depth (m)
Beauclair	63	4.39	9.03	2.05
Dora	63	17.74	53.28	3.00
Harris	63	75.63	277.00	3.66
Eustis	63	31.39	108.57	3.46
Griffin	59	38.09	89.92	2.36
Yale	59	16.27	60.72	3.73
Weir	57	22.76	131.36	5.77

*Volume is for reference elevation. In calculation of detention times, volumes used were determined from year-end lake elevations by quadratic regressions fit to hypsographic data for the lakes reported by Danek et al. (1991).

Source: Danek et al. 1991.

The hydraulic detention time is more difficult to estimate. Hydraulic detention time for a lake can be calculated by dividing the lake volume by the rates of water inflow or outflow. These estimates of water inputs or losses are difficult to determine without complete water budgets, which are not available for the lakes in the UORB.

I estimated detention time for the UORB lakes by two methods. The first method estimated water losses by the sum of tributary discharge (Equation 11) and leakage losses. Deevey (1988) estimated leakage rates to the underlying aquifer for a number of Florida lakes, including Lake Dora. Estimated leakage rates for several central Florida lakes were relatively constant, ranging from 0.3 to 0.5 m/yr. Leakage rates for the UORB lakes were estimated by multiplying the reported leakage rate for Lake Dora (0.427 m/yr) by the lake surface area. For example, the estimate of water losses from Lake Harris-Little Lake Harris (Q_{Ho}) was calculated using

$$Q_{Ho} = T_P + S_i + (A_{Hw} \times r_{HC}) - V_H + (A_{Hl} \times 0.427) \tag{22}$$

where:

T_P = Palatlakaha River discharge (m^3/yr)
S_i = spring discharges into Lake Harris-Little Lake Harris (m^3/yr)
A_{Hw} = Lake Harris contributing watershed area (including lake surface area) (m^2)
r_{HC} = net runoff coefficient for Haines Creek Basin (m/yr)
V_H = change in storage volume for Lake Harris-Little Lake Harris (m^3/yr)
A_{Hl} = Lake Harris-Little Lake Harris surface area

This method is not usable for Lake Weir, as there are no estimates of net runoff coefficients for the lake. Additionally, use of this method is questionable for Lake Yale, because the net runoff coefficient used would be for the Lake Griffin Basin. However, for considerable periods, Lake Yale water levels were below the elevation of the culvert on its outlet canal, so the lake functioned as an isolated drainage basin rather than as part of the Lake Griffin Basin.

A second method to calculate detention time estimates water inputs using runoff coefficients derived from the literature for the land use and soil types occurring in the drainage basin. An area-weighted average runoff coefficient was calculated for the drainage basin using the runoff coefficients tabulated for land use/soil combinations in Table 3. Water inflows (Q_i) were then calculated using the following equation:

$$Q_i = T_i + S_i + (R_{wls} \times Pr_G \times A_w) + (Pr_N \times A_l) \quad \textbf{(23)}$$

where:

T_i = tributary inflows
S_i = spring inflows
R_{wls} = area-weighted average runoff coefficient
Pr_G = gross precipitation
A_w = contributing watershed area (not including lake surface area)
Pr_N = net precipitation
A_l = lake surface area

Nutrient Retention/Export within the Lake Basin

Having calculated nutrient loading from each source as described above, the total annual nutrient loading is calculated using Equation 4. Nutrients exported downstream in tributary discharges are calculated using Equation 12. The difference between nutrient loading and downstream export is the net retention/export within the lake basin. Potential sinks for nutrients retained within the lake basin include the water column in the lake, the bottom sediments, the biota, leakage to the underlying aquifer, and, for nitrogen, denitrification to gaseous forms.

Comparison of Reported In-lake Nutrient Concentrations with Model Predictions

As mentioned previously, EUTROMOD uses empirical equations developed using data obtained from up to 37 Florida lakes during the NES to predict in-lake nutrient concentrations. The equations are based on a general mass-balance model first proposed by Vollenweider (1969). The equations used in EUTROMOD (Reckhow 1991) are

$$\log_{10}(TP) = \log_{10}\left[\frac{TP_{in}}{1 + k_P\tau}\right] \tag{24}$$

standard error = 0.189

where:

$$k_P = 1.71\tau^{-0.21} z^{1.01} TP_{in}^{0.40} \tag{25}$$

$$\log_{10}(TN) = \log_{10}\left[\frac{TN_{in}}{1 + k_N\tau}\right] \tag{26}$$

standard error = 0.136

where:

$$k_N = 0.20\tau^{-0.89} z^{1.56} TN_{in}^{0.33} \tag{27}$$

Symbols:

TP, TN	=	predicted in-lake nutrient concentrations (mg/L)
TP_{in}, TN_{in}	=	average influent nutrient concentrations (mg/L)
k_P, k_N	=	nutrient sedimentation coefficients
τ	=	hydraulic detention time (yr)

z = lake mean depth (m)

I also calculated predicted in-lake TP by two other equations reported in the literature. Although these equations are somewhat different in form, they also are based on the Vollenweider (1969) model and are developed from a data base that includes 290 lakes from North America and Europe (Canfield and Bachmann 1981). The equation developed by Canfield and Bachmann (1981) that best described their data set is

$$TP=\frac{L_a}{z(k_p+\rho)} \qquad k_p=0.162\left(\frac{L_a}{z}\right)^{0.458} \tag{28}$$

where:

L_a = annual TP loading per unit of lake surface area (mg/m^2/yr)
ρ = hydraulic flushing rate = $1/\tau$ (yr^{-1})

Other authors have reformulated the Vollenweider equation, using the phosphorus retention coefficient of a lake rather than the phosphorus sedimentation coefficient. An equation of this form developed by Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, also provided a close fit to Canfield and Bachmann's (1981) data base:

$$TP=\frac{L_a(1-P_{RC})}{q_s} \qquad P_{RC}=\frac{1}{1+0.747\rho^{0.507}} \tag{29}$$

where:

P_{RC} = phosphorus retention coefficient
q_s = annual areal water loading = z/τ (m/yr)

The phosphorus retention coefficient can be estimated independently from the nutrient budget calculations by the difference between annual phosphorus inputs and outputs divided by the annual phosphorus inputs.

To provide an assessment of the accuracy of the nutrient loading estimates, predicted in-lake TP and TN were calculated by the Reckhow (1991), Canfield and Bachmann (1981), and Larsen and Mercier (1976) equations and then compared with observed nutrient concentrations at monitoring stations in the lakes. I also calculated phosphorus retention coefficients from estimates of TP loading and downstream discharge for comparison with coefficients predicted by Larsen and Mercier's (1976) equation. Another use made of model predictions of lake nutrient concentrations was to assess potential lake trophic responses to various lake management and restoration measures (see chapter on Assessment of Potential Strategies for Improving Water Quality).

EUTROMOD also predicts other lake trophic state variables, including chlorophyll-*a*, Secchi depth, and Florida Trophic State Index (Huber et al. 1982). These trophic state variables are calculated as functions of the model predictions of TP and TN, so any errors in predictions of nutrient concentrations are compounded in predictions of other trophic state variables. EUTROMOD tended to underestimate TN concentrations for the UORB lakes (see Results chapter), which resulted in underestimates of other trophic state variables. EUTROMOD predictions of other trophic state variables are not discussed further in the text, but Appendixes D–J include the model predictions.

RESULTS

INFERENCE OF POTENTIAL LIMITING NUTRIENTS THROUGH N:P RATIOS

There are wide variabilities in reported N:P ratios for the lakes in the UORB (Table 6). The extreme values in some samples may

Table 6. Nitrogen to phosphorus ratios for lakes in the upper Ocklawaha River Basin

Lake	Mean TN/TP	Standard Deviation	Range	N*
Beauclair	18.80	9.17	1.66–73.03	100
Dora	38.00	20.31	1.82–128.41	136
Harris	58.99	46.70	19.33–363.86	70
Eustis	48.36	30.74	3.36–220.00	122
Griffin	32.42	17.91	1.42–188.50	260
Yale	117.80	127.40	4.64–524.00	71
Weir	115.09	113.30	12.39–444.00	48
Lake	**Mean TIN/TIP**	**Standard Deviation**	**Range**	**N***
Beauclair	4.08	4.52	0.08–21.00	61
Dora	6.78	9.48	0.07–46.78	84
Harris	4.33	5.69	0.84–18.25	8
Eustis	9.39	12.53	0.65–80.86	63
Griffin	29.87	47.28	0.11–260.48	190
Yale	2.06	1.70	0.36–7.73	28
Weir	4.11	6.83	0.31–40.40	35

*N is the number of observations.

reflect analytical errors. Average TN:TP ratios for the lakes suggest mixed nutrient limitation in Lake Beauclair and possibly in Lake Griffin and potential limitation by phosphorus in the other lakes. TIN:TIP ratios for the lakes in the basin show a different pattern. In general, TIN:TIP ratios are considerably lower than TN:TP ratios and tend to indicate potential nitrogen limitation in all of the lakes except Lake Griffin.

ESTIMATES OF NUTRIENT LOADING

Nutrient Loading from Atmospheric Deposition (L_{Pr})

Table 7 presents average nutrient concentrations in wet and dry deposition samples from the Lake Apopka marsh flow-way site.

Table 7. Average nutrient concentrations from the Lake Apopka marsh flow-way site and estimated loading rates from rainfall and dry deposition

Parameter	Statistic	Rainfall	Dry Deposition
Total phosphorus	Mean (mg/L)	0.0114	0.483
	Standard deviation	0.0115	0.442
	Number of observations	88	22
	Deposition (g/m²/yr)	0.0137*	0.0345
Total nitrogen	Mean (mg/L)	0.551	2.048
	Standard deviation	0.416	0.880
	Number of observations	74	11
	Deposition (g/m²/yr)	0.659*	0.146

*Wet deposition was calculated based on average annual rainfall at the NOAA weather station at Lisbon, 1960–91 (1.196 m). In nutrient budgets, wet deposition was calculated using rainfall in each year studied.

Also presented are areal deposition rates used in nutrient budgets for the UORB lakes. The rainfall nutrient concentrations are rather low in comparison to previous Florida studies, particularly for TP, which had an average concentration of 0.1 mg/L in studies reported by Irwin and Kirkland (1980). However, the previous studies summarized in Irwin and Kirkland are bulk precipitation collections, which combine wet and dry deposition.

As shown in Table 7, dry deposition represents a significant fraction of total atmospheric deposition.

Nutrient Loading from Spring Discharges (L_{Sp})

Nitrate-nitrite concentrations were substantially higher in samples collected for the present study than in samples collected between 1946 and 1972 at three springs discharging into Lake Harris-Little Lake Harris (Table 8). Nitrate-nitrite concentrations were substantially lower in discharges from Bugg Spring than in discharges from the smaller springs, but even with this spring, recent concentrations were nearly twice earlier measurements. The major land uses in the area of the springs are wetlands, agriculture, and residential. Of these three land uses, only residential areas have increased in acreage since earlier mapping in 1973. The area near the springs is not serviced by municipal sewage treatment plants. If the increased nitrate-nitrite concentrations in spring discharges are due to locally occurring events, then the increases seem most likely to be due to increased septic tank discharges or other impacts of residential development. However, because of the mobility of nitrogen in

Table 8. Nitrate-nitrite concentrations in spring discharges in the upper Ocklawaha River Basin

Spring	Year	Mean ± S.D. (N)* (mg/L)
Bugg Spring	1946	0.3 (1)
	1972	0.3 (1)
	1990–91	0.56±0.03 (4)
Blue Springs	1972	0.74 (1)
	1991–92	3.85±0.19 (5)
Holiday Springs	1967	0.1 (1)
	1972	0.61 (1)
	1991–92	3.44±0.18 (5)

*N is the number of observations.

soils, more distant sources, including sprayfields from sewage treatment plants and citrus processing plants, could contribute to elevated concentrations.

Table 9 presents nutrient concentrations and discharge volumes used in calculations of nutrient loading from spring discharges. Because recent nitrogen concentrations were substantially higher than earlier measurements, only nutrient data from samples collected from 1990 to 1992 were used to estimate nutrient loading from spring discharges. Recent measurements of discharge volumes tended to be lower than measurements collected between 1949 and 1972, but there was some overlap, so all measurements were included in calculations of average discharge rate.

Table 9. Nutrient concentrations and flows in spring discharges in the upper Ocklawaha River Basin

Spring	TP Mean ± S.D. (N)* (mg/L)	TN Mean ± S.D. (N)* (mg/L)	Flow (m^3/d)
Bugg Spring			
SJRWMD	0.104±0.021 (4)	0.704±0.104 (4)	
Lakewatch	0.071±0.009 (14)	0.689±0.075 (14)	
Overall	0.088	0.697	30,870
Blue Springs	0.031±0.004 (5)	3.906±0.240 (5)	3,630
Holiday Springs	0.027±0.009 (5)	3.468±0.188 (5)	8,930
Howey Heights Tributary	0.018±0.006 (5)	6.643±0.722 (5)	7,560

*N is the number of observations.

Nutrient Loading from Tributary Inflows (L_{Tr})

Flow Volumes. Estimated monthly net runoff coefficients for the drainage basins were frequently negative. Annual runoff coefficients were negative for 3 or 5 years of the 7-year period (Table 10). The negative runoff coefficients may be related to the extended drought conditions that occurred during the 1980s;

Table 10. Annual net precipitation for the Lisbon NOAA weather station and estimated net runoff coefficients for the Haines Creek and Moss Bluff basins (in cm)

Year	Lisbon Annual Net Precipitation	Haines Creek Runoff Coefficient	Moss Bluff Runoff Coefficient
1984	-9.24	-6.94	47.83
1985	-28.81	-4.31	5.30
1986	-7.35	6.03	-0.12
1987	5.81	16.35	-0.26
1988	13.15	17.95	-1.02
1989	-4.70	4.16	-5.57
1990	-22.40	-3.04	-2.34

annual net precipitation at the Lisbon NOAA weather station was also negative for 5 years of the 7-year period (Table 10). There is a good correlation between annual net precipitation at Lisbon and annual net runoff coefficients for the Haines Creek Basin (r=0.86). However, there is a poor correlation for the Moss Bluff Basin (r=-0.12). The Moss Bluff Basin had an anomalously high runoff estimate in 1984. The high runoff in the 1984 estimate may be related to the drawdown of Lake Griffin that year, although the runoff estimates are adjusted for changes in lake volume. Perhaps the drawdown of surface waters resulted in increased ground water flows. Even if 1984 is omitted from the period of record, there is a negative correlation between annual net precipitation at Lisbon and annual net runoff coefficients for the Moss Bluff Basin (r=-0.41). Although variations in the runoff coefficients for the Moss Bluff Basin are not explained, they are used only to estimate discharges from Lake Yale, which represent a minor nutrient source for Lake Griffin.

Although the net runoff coefficients for the two basins are often negative, this does not mean that stream discharges will be negative, as the net runoff is only one term in the equation used to determine discharge volumes (Equation 11). Monthly estimated lake discharges included negative values only for Lakes

Harris-Little Harris and Yale. The negative discharge estimates could be an indication of flow reversals in the connecting tributaries. But because the net runoff coefficients are basinwide estimates, rather than specific to the subbasin from which discharge is being determined, I felt they were insufficiently sensitive to accurately estimate the low flows that occurred during flow reversals. Therefore, during periods in which estimated lake discharges were negative, it was assumed in the nutrient budgets that downstream nutrient loading was zero. A separate analysis was conducted to determine the effect on the nutrient budgets if the negative discharge estimates were accurate measurements of flow reversals (in Discussion, see section on Assessment of Accuracy of the Nutrient Loading Estimates).

Lake Harris-Little Lake Harris is connected to Lake Eustis by the Dead River. It was assumed that discharges from Lake Harris-Little Lake Harris were zero in months in which it was estimated that downstream discharges were negative (16 of the 84 months from 1984 to 1990).

Lake Yale is connected to Lake Griffin by a culvert in the Yale-Griffin Canal which has an invert elevation of 59 feet National Geodetic Vertical Datum (ft NGVD). Discharges from Lake Yale were assumed to be zero under two circumstances: (1) if the estimated discharges from Lake Yale were negative (33 of the 84 months from 1984 to 1990) or (2) if the Lake Yale water surface elevation was below 59 ft NGVD on both the last day of a month and the last day of the previous month (22 of the 84 months from 1984 to 1990). Due to some overlap between these two circumstances, zero discharge was assumed in 47 of the 84 months from 1984 to 1990.

Water Chemistry. Table 11 summarizes nutrient data for tributaries and lakes in the UORB. Nutrient concentrations were highest in the Apopka-Beauclair Canal and declined downstream, before rising again in Lake Griffin. Nutrient concentrations were lower in water bodies not directly downstream from Lake Apopka, including the Palatlakaha River, Lake Harris, Lake Yale, and Lake Weir. In general, temporal trends and relationships of

Table 11. Nutrient concentrations for tributaries and lakes in the upper Ocklawaha River Basin

Water Body	Period of Record	TP Mean ± S.D. (N)* (mg/L)	TN Mean ± S.D. (N)* (mg/L)
Apopka-Beauclair Canal	Apr 1981–May 1991	0.322±0.160 (79)	4.452±1.267 (58)
Palatlakaha River	Apr 1980–Jan 1991	0.050±0.032 (27)	0.641±0.276 (29)
Haines Creek	Jun 1979–Feb 1991	0.060±0.050 (67)	2.403±0.705 (44)
Ocklawaha River at Moss Bluff	Feb 1980–Aug 1990	0.061±0.031 (77)	2.146±0.762 (71)
Lake Beauclair	Jan 1977–Feb 1991	0.235±0.115 (96)	4.206±1.246 (76)
Lake Dora	Jan 1977–Feb 1991	0.138±0.118 (115)	3.520±0.967 (93)
Lake Harris	Mar 1979–Feb 1991	0.042±0.038 (53)	1.794±0.486 (32)
Lake Eustis	Feb 1977–Oct 1992	0.068±0.065 (117)	2.487±0.794 (97)
Lake Griffin	Feb 1977–Feb 1993	0.119±0.101 (152)	2.990±1.165 (143)
Lake Yale	Mar 1979–Feb 1991	0.025±0.047 (59)	0.950±0.193 (50)
Lake Weir	Mar 1984–Sep 1992	0.015±0.010 (34)	0.714±0.177 (34)

*N is the number of months during which sampling occurred. When there are multiple samples within a month, WQStat calculates monthly mean values prior to further analysis.

nutrient concentrations with discharge were weak. Even when statistically significant, regressions accounted for only a small part of the variability in nutrient concentrations.

Nutrient concentrations in tributary discharges into Lake Beauclair were estimated from a monitoring station at the Apopka-Beauclair Canal lock and dam. A Kruskal-Wallis test indicated a significant seasonality in TP at the Apopka-Beauclair Canal station. For the quarters July–September and October–December, there were no significant temporal trends or relationships of TP with flow, so average values for the quarter were used to estimate nutrient discharges (Jul–Sep

TP=0.436 mg/L, Oct–Dec TP=0.234 mg/L). In the quarters January–March and April–June, there were significant relationships of TP with flow, so the following regression equations were used to estimate TP (mg/L) in these quarters when data were unavailable:

$$\textit{Jan-Mar}\ \ TP = 0.0005702 \times cfs + 0.2486 \tag{30}$$

r^2=0.172

$$\textit{Apr-Jun}\ \ TP = 0.0005820 \times cfs + 0.2383 \tag{31}$$

r^2=0.420

where:

cfs = discharge rate in cubic feet per second

TN concentrations at the Apopka-Beauclair Canal station showed no significant seasonality or temporal trend, but were significantly related to discharge. During time periods when no data were available, TN (mg/L) was estimated by:

$$TN = 0.005846 \times cfs + 4.0907 \tag{32}$$

r^2=0.286

Nutrient concentrations in tributary discharges into Lake Dora were estimated from a monitoring station in Lake Beauclair. At the Lake Beauclair station, TP showed a significant seasonality, but no temporal trend within seasons, so quarterly means were used to estimate TP when data were unavailable: Jan–Mar TP=0.239 mg/L, Apr–May TP=0.208 mg/L, Jun–Aug TP=0.177 mg/L, Sep–Dec TP=0.296 mg/L. TN showed no significant seasonality or temporal trend at the Lake Beauclair station, so in months with no data, TN was estimated by the overall mean for the data set (Table 11).

Nutrient concentrations in tributary discharges into Lake Harris-Little Lake Harris were estimated from the monitoring station on the Palatlakaha River. There were no significant seasonality, temporal trends, or relations with discharge for TP or TN at the Palatlakaha River station, so in months with no data, nutrient concentrations were estimated by the overall means for the data set (Table 11).

Nutrient concentrations in tributary discharges into Lake Eustis from Lake Harris-Little Lake Harris were estimated from monitoring data for the center or north-center of Lake Harris and a limited number of samples from the Dead River, which connects Lakes Harris and Eustis. There were no significant seasonality or temporal trends for TP or TN for the Lake Harris-Dead River stations, so in months with no data, nutrient concentrations were estimated by the overall means for the data set (Table 11).

Nutrient concentrations in tributary discharges into Lake Eustis from Lake Dora were estimated from monitoring data for Lake Dora. Lake Dora TP showed a significant seasonality. There was a significant temporal trend for December–March TP. For other quarters, there was no temporal trend, so quarterly means were used to estimate TP concentrations in months with no data: Apr–Jul TP=0.154 mg/L, Aug TP=0.148 mg/L, Sep–Nov TP=0.103 mg/L. For December–March, TP (mg/L) was estimated using

$$\textit{Dec-Mar TP} = 0.6828 - 0.0000178 \times \textit{L123Date} \tag{33}$$

r^2=0.129

where:

L123Date = Lotus 123 date format, which numbers dates consecutively from January 1, 1900. For example, the L123Date for August 14, 1989, is 32734.

Lake Dora TN also showed a significant seasonality. There was a significant temporal trend for April–June TN. For other quarters, there was no temporal trend, so quarterly means were used to estimate TN concentrations in months with no data: Jan–Mar TN=3.602 mg/L, Jul–Sep TN=2.920 mg/L, Oct–Dec TN=3.714 mg/L. For April–June, TN (mg/L) was estimated using

$$Apr\text{-}Jun\ \ TN = 0.0002809 \times L123Date - 4.905 \tag{34}$$

r^2=0.166

Nutrient concentrations in tributary discharges into Lake Griffin from Lake Eustis were estimated using data from the monitoring station on Haines Creek. For the Haines Creek station, there were no significant seasonality or temporal trends for TP or TN, so in months with no data, nutrient concentrations were estimated by the overall means for the data set (Table 11).

Nutrient concentrations in tributary discharges into Lake Griffin from Lake Yale were estimated using water quality data from Lake Yale. There were no significant seasonality or temporal trends for TN for Lake Yale, so in months with no data, TN concentrations were estimated by the overall means for the data set (Table 11). Lake Yale TP showed no significant seasonality, but did show a significant temporal trend, so in months with no data, TP concentrations (mg/L) were estimated by

$$TP = 0.4489 - 0.00001324 \times L123Date \tag{35}$$

r^2=0.058

Nutrient concentrations in discharges from Lake Griffin were estimated using water quality data from the Ocklawaha River at Moss Bluff. Moss Bluff TP did not show a significant seasonality but did show a significant temporal trend and a significant relationship with discharge. The discharge-TP regression accounted for a larger amount of the variability in TP, so in months with no data, TP concentrations (mg/L) were estimated by

$$TP = 0.05670 + 0.00003137 \times cfs \tag{36}$$

r^2=0.175

Nutrient Loading. Reported or estimated tributary discharges and nutrient discharges are presented in Figures 7 and 8. Because of weak temporal trends and relationships with discharge, nutrient discharges in tributary flows are strongly related to discharge volumes.

Nutrient Loading from Muck Farm Discharges (L_{MF})

Stepwise regression analysis identified three of the independent variables as significant predictors (p<0.15) of monthly pump discharge: acres in production, rainfall, and evaporation. The three-parameter regression equation, accounting for 78.5% of the reported variability in pump discharge, was

$$\begin{aligned} MF_{pumpdis} &= (210{,}209 \times A_{MF}) + (21{,}964{,}400 \times Pr_G) \\ &\quad - (8{,}109{,}250 \times E) - 92{,}521{,}500 \end{aligned} \tag{37}$$

where:

$MF_{pumpdis}$	= monthly muck farm pump discharge (gal/mo)
A_{MF}	= acres in production
Pr_G	= gross precipitation (in./mo)
E	= evaporation (in./mo)

There was no significant seasonality or temporal trend in the muck farm discharge water chemistry data. For both TP and TN, a Kruskal-Wallis test indicated significant differences among farms (p<0.001). A Tukey multiple comparisons test indicated that TP concentration was significantly higher in discharges from Paulhamus Farm than for several other farms (Table 12). However, the multiple comparisons test did not identify significant differences in TN concentration between farms.

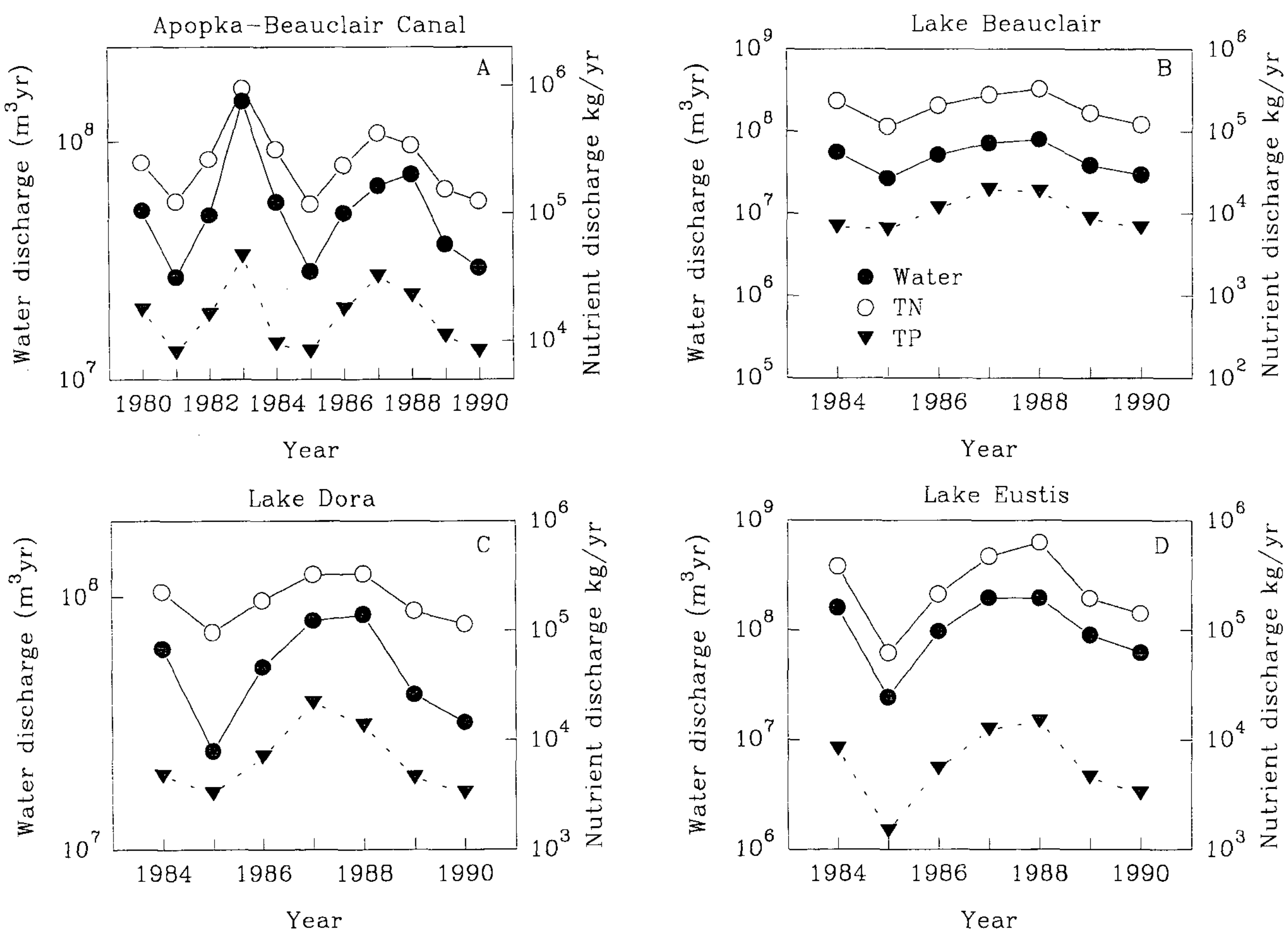

Figure 7. Water and nutrient discharges from the (A) Apopka-Beauclair Canal and Lakes (B) Beauclair, (C) Dora, and (D) Eustis

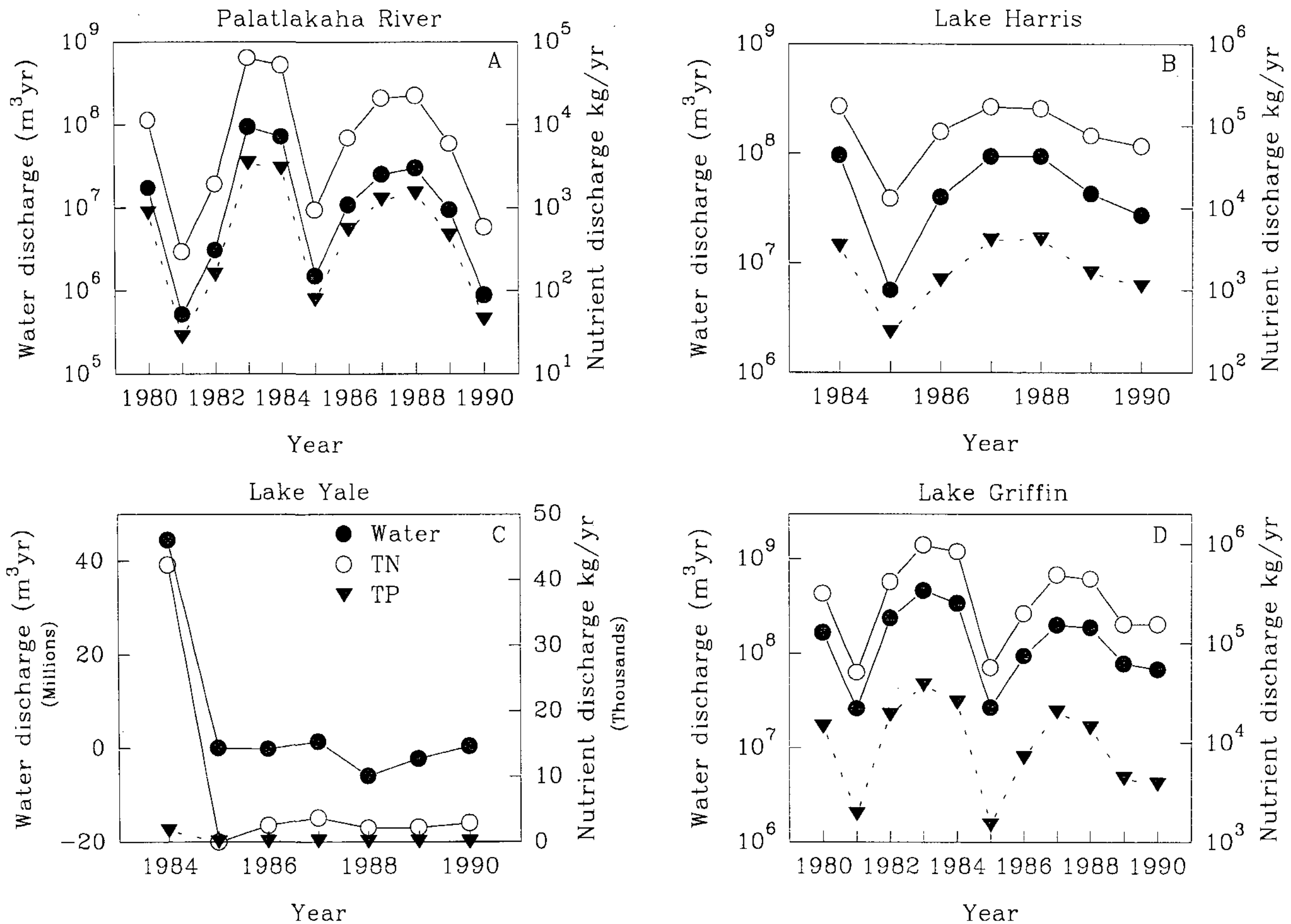

Figure 8. Water and nutrient discharges from the (A) Palatlakaha River and Lakes (B) Harris, (C) Yale, and (D) Griffin

Table 12. Summary water chemistry data for muck farm discharges, upper Ocklawaha River Basin

Farm	Period of Record	TP Mean ± S.D. (N)* (mg/L)	TN Mean ± S.D. (N)* (mg/L)
Hurley Muck Farm	Oct 1983–Feb 1987	0.20±0.07 (8)	2.40 (1)
Pine Meadows	Jan 1989–Apr 1989	0.20±0.09 (2)	3.74±0.1987 (2)
JA-MAR	Apr 1984–Jun 1988	0.27±0.22 (21)	1.69±0.58 (17)
S.N. Knight, Leesburg	Jul 1979–Jun 1992	0.63±0.82 (26)	3.50±2.28 (25)
Lowrie Brown North	Jun 1987–Dec 1988	0.84±0.76 (3)	1.80±0.33 (2)
Lowrie Brown South	Jun 1979–Jul 1990	0.38±0.57 (33)	3.47±3.42 (27)
S.N. Knight, Lisbon (North)	Apr 1984–Nov 1990	0.33±0.22 (7)	2.78±0.62 (5)
S.N. Knight, Lisbon (South)	Apr 1984–Jun 1992	0.86±0.95 (45)	3.18±1.28 (43)
Long Farms North	Oct 1982–Nov 1988	0.78±0.54 (15)	3.12±2.59 (13)
Paulhamus (Eustis muck farm)	May 1989–Apr 1990	1.76±0.62 (7)	3.17±0.86 (7)

*N is the number of months during which sampling occurred. When there are multiple samples within a month, WQStat calculates monthly mean values prior to further analysis.

Source: SJRWMD; LCES permit records

If nutrient concentration data were available for a month for a farm, then the average of samples in that month was used to calculate nutrient loading. If TP data were not available, then the mean TP concentration in discharge samples from Paulhamus Farm (1.76 mg/L) was used to calculate loading from that farm. For estimates of TP discharges from other farms, I used the mean TP concentration for all farms in the basin except Paulhamus Farm (0.58 mg/L). When TN data were not available, I used the mean TN concentration for all farms in the basin (3.07 mg/L).

Estimated mean annual discharges from muck farms in the UORB are given in Table 13.

Table 13. Estimated mean annual water and nutrient discharges from muck farms in the Upper Ocklawaha River Basin

Farm	Receiving Lake	Water Discharge (m^3/yr)	TN Discharge (kg/yr)	TP Discharge (kg/yr)
Hurley Muck Farm	Beauclair	2,119,200	6,480	1,180
Pine Meadows	Eustis	4,383,300	13,800	2,560
Springhill Farms	Eustis	1,108,100	3,400	640
JA-MAR	Harris	2,153,400	6,380	1,130
S.N. Knight, Leesburg	Harris	2,332,800	7,210	1,380
Lowrie Brown North	Griffin	1,721,600	5,350	1,020
Lowrie Brown South	Griffin	4,219,900	13,300	2,250
S.N. Knight, Lisbon	Griffin	16,018,300	49,180	9,720
Walker Ranch*	Griffin	2,128,900	6,530	1,220
Paulhamus (Eustis muck farm)	Griffin	3,699,900	11,710	6,690
Long Farms North	Griffin	4,609,900	14,610	2,880

Walker Ranch in operation only 1980–82

Land Use and Soils Data Used in Calculations of Nutrient Loading from Stormwater Runoff (L_{Ro})

Summaries of land uses delineated for contributing subbasins in the 1984 and 1987–89 land use maps are presented in Table 14, and a map of the 1987–89 land uses is presented in Figure 9. A more detailed breakdown of land uses appears in Appendix K. The major land uses in the basin are agriculture, residential, forest/rangeland, and open water/wetland.

One of the major changes from the 1984 land use maps was a decline in areas of citrus groves in the 1987–89 map. Substantial areas of citrus were abandoned following a series of freezes in the 1980s. These areas were converted to a variety of other land uses, including residential, other urban land uses, other agriculture, and forest/rangeland. Another significant change in land use is a decline in area of confined feedlots in the Lake Eustis subbasin. This change involves a small area but has

Table 14. Land uses in the upper Ocklawaha River Basin contributing subbasins (in ha)

A. 1984 Land Use Maps								
Land Use Group	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin		Lake Yale Subbasin	Lake Weir Subbasin
					1980–82	1983–84		
Residential low-medium density	80.35	243.06	875.39	624.93	441.1	441.11	122.71	869.44
Residential high density	0.00	633.53	164.63	1,535.11	1,053.21	1,053.21	104.13	73.63
Commercial	0.00	0.00	343.62	25.29	62.60	62.60	0.00	0.00
Institutional	0.00	1.62	28.48	42.24	19.91	19.91	0.00	9.83
Industrial/mining	0.00	36.67	144.21	4.11	45.40	45.40	20.45	0.00
Recreation/open land	0.00	13.09	119.60	103.06	30.08	30.08	55.85	320.96
Forest/rangeland	775.69	345.96	1,571.67	552.86	1,330.52	1,330.52	1,121.25	255.58
Open water/wetlands	615.09	816.01	6,706.60	2,215.86	2,205.34	2,454.23	2,406.48	358.11
Confined feeding	0.00	0.00	6.19	72.69	0.00	0.00	0.00	0.00
Pasture	96.99	101.91	3,199.71	1,153.80	1,470.14	1,470.14	939.24	373.50
Citrus groves	1,423.58	681.35	8,422.55	1,687.62	962.64	962.64	1,617.81	678.77
Other agriculture	132.69	0.02	57.97	78.63	53.69	53.69	36.75	9.02
Muck farms	169.97	0.00	333.07	363.42	1,997.77	1,748.89	0.00	0.00
Total drainage area	3,294.36	2,873.22	21,973.69	8,459.62	9,672.41	9,672.42	6,424.67	2,948.84
Lake area	470.42	1,751.08	7,388.54	3,208.30	3,772.81	3,772.81	1,732.70	2,275.63
Total	3,764.78	4,624.30	29,362.23	11,667.92	13,445.22	13,445.23	8,157.37	5,224.47

Table 14—*Continued*

B. 1987–89 Land Use Maps							
Land Use Group	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin	Lake Yale Subbasin	Lake Weir Subbasin
Residential low-medium density	345.20	712.62	1,835.00	1,496.31	1,510.24	270.62	735.26
Residential high density	0.00	153.66	234.53	420.97	181.11	84.21	4.94
Commercial	7.16	221.26	663.90	335.96	297.67	80.23	3.56
Institutional	1.31	17.01	77.12	76.27	100.38	85.17	8.03
Industrial/mining	26.93	34.65	223.15	61.32	124.88	160.20	9.34
Recreation/open land	70.44	118.66	258.15	210.19	132.89	16.98	330.02
Forest/rangeland	604.97	434.11	4,693.73	1,139.06	1,218.93	1,063.34	420.51
Open water/wetlands	858.81	716.90	6,965.42	2,263.94	2,365.77	2,322.01	520.04
Confined feeding	0.00	0.00	0.00	3.22	0.00	0.00	0.00
Pasture	201.96	27.82	1,928.90	590.40	978.24	655.83	202.10
Citrus groves	627.81	181.06	1,919.37	562.45	377.67	376.57	391.94
Other agriculture	386.31	220.80	2,909.13	976.56	654.90	1,272.62	323.10
Muck farms	169.97	0.00	333.07	363.42	1,724.71	0.00	0.00
Total drainage area	3,300.87	2,838.55	22,041.47	8,500.07	9,667.39	6,387.78	2,948.84
Lake area	463.98	1,785.70	7,320.80	3,167.79	3,777.96	1,769.64	2,275.63
Total	3,764.85	4,624.25	29,362.27	11,667.86	13,445.35	8,157.42	5,224.47

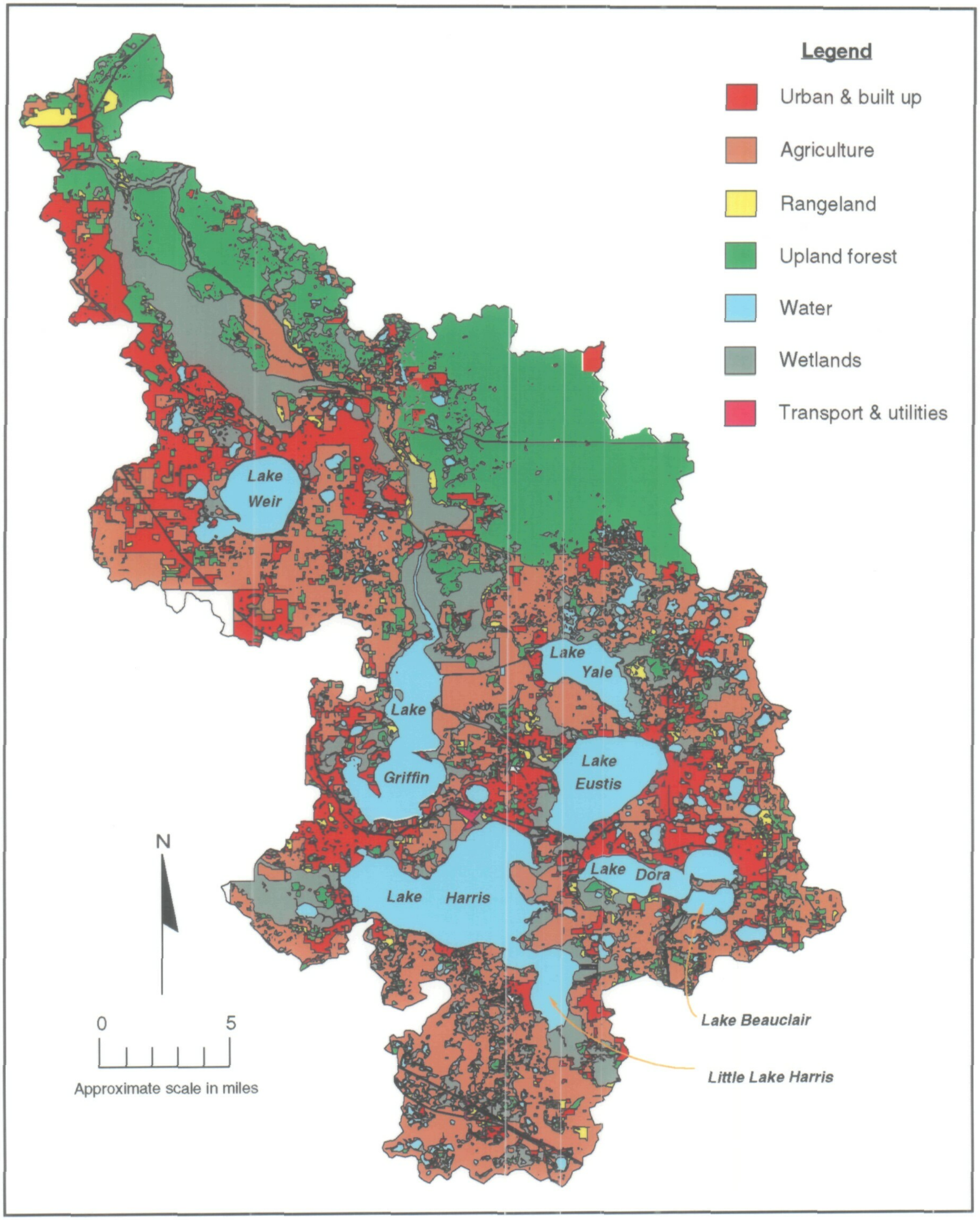

Figure 9. Land use map for the upper Ocklawaha River Basin

substantial effects on estimates of nutrient loading because of the high nutrient concentrations in runoff from confined feedlots (Table 3).

Other differences in land uses between the two maps appear to reflect inconsistencies in the delineations, rather than true changes in land use. For example, the 1984 land use maps for the Lakes Dora, Eustis, and Griffin drainage basins show most of the residential housing to be high density, whereas the 1987 land use maps show residential housing to be mostly low-medium density. This discrepancy does not appear to substantially affect the estimates of nutrient loading because of similar nutrient runoff from these two land use classes (Table 3).

The composite SCS hydrologic soil group rankings for the UORB fell only in classes A, B, and D. Table 15 shows the soil associations in the UORB with the assigned hydrologic soil group ranking. Table 16 shows hydrologic soil group areas for contributing subbasins. The composite hydrologic soil group map for the UORB is illustrated in Figure 6.

Nutrient Loading from Septic Systems (L_{ST})

Nearly all of the septic systems potentially affecting the lakes in the UORB are from residential housing (Table 17). Capita-years of septic tank usage range from 90.2 for Lake Beauclair to 3,530.9 for Lake Harris-Little Lake Harris. Data on ground elevations of houses indicate that only about 1.5% of the septic system drainfields are potentially inundated when the Ocklawaha chain of lakes is at the highest elevations under current regulation schedules. There are no estimates of potential inundation of septic tank drainfields in the Lake Weir subbasin. Water levels in Lake Weir have been declining since 1983, and by 1990 they were near historic low elevations (Crisman et al. 1992). It is thus unlikely that flooding of septic tank drainfields has been a problem in recent years in the Lake Weir subbasin.

Estimated nutrient loadings from septic systems are presented in Table 18. Nutrient loading rates are only slightly increased if it is

Table 15. Summary of soil types for the upper Ocklawaha River Basin

Soil Association	Acreage	Percent of Total
Hydrologic Soil Group A		
Astatula Association	32,200	9.1
Astatula-Apopka & Candler-Apopka Association	194,008	54.6
Arredondo-Gainesville Association	3,500	1.0
Kendrick-Hague-Zuber Association	907	0.3
Hydrologic Soil Group B		
Sparr-Lochloosa-Tavares Association	9,990	2.8
Tavares-Myakka Association	17,305	4.9
Pomello-Paola Association	1,924	0.5
Hydrologic Soil Group D		
Lynne-Pomona-Pompano Association	5,608	1.6
Eureka-Paisley-Eaton Association	16,286	4.6
Myakka-Sellers Association	8,308	2.3
Myakka-Placid-Swamp Association	7,289	2.1
Bluff-Martel Association	6,103	1.7
Anclote-Iberia, var.-Emeralda Association	8,294	2.3
Montverde-Ocoee-Brighton & Okeechobee-Terra Ceia-Tomoka Association	42,893	12.1
Swamp Association	696	0.2
Total	355,311	100.1

assumed that there is 0% soil retention in drainfields that are potentially inundated when the Ocklawaha chain of lakes is at the highest elevations under current regulation schedules.

Table 16. Hydrologic soil group areas (in ha) in the upper Ocklawaha River Basin contributing subbasins

Lake Subbasin	Hydrologic Soil Group			Total
	A	B	D	
Beauclair	1,953.7	361.7	979.0	3,294.4
Dora	1,972.9	275.5	624.8	2,873.2
Harris	13,401.8	5,413.7	3,158.6	21,974.1
Eustis	4,321.8	1,776.3	2,361.5	8,459.6
Griffin	3,394.4	956.2	5,321.8	9,672.4
Yale	2,825.4	546.1	3,053.2	6,424.7
Weir	2,293.6	148.2	507.0	2,948.8

Table 17. Estimated capita-years of septic systems usage affecting the upper Ocklawaha River Basin lakes and estimates of potentially inundated septic system drainfields

Lake	Residences	Other Structures	Capita-years	Inundated Systems
Beauclair	44		90.2	4*
Dora	368		754.4	3
Eustis	777		1,592.8	26
Harris	1,698	2 restaurants 2 motels (<10 rooms) Golf course clubhouse	3,530.9	27
Griffin	1,243	2 marina clubhouses	2,568.2	6
Yale	312		639.6	1
Weir	689		1,502.0	?

*The estimate of inundated septic system drainfields for Lake Beauclair includes systems bordering both Lake Beauclair and adjacent Lake Carlton.

Note: ? = no estimate available

Table 18. Estimated annual nutrient loading to lakes in the upper Ocklawaha River Basin from septic systems and upper bound estimates assuming 0% soil retention of effluents from potentially inundated septic system drainfields (in kg/yr)

Lake	TP Load	TN Load	Upper Bound TP Load	Upper Bound TN Load
Beauclair	13.35	235.65	24.27	253.18
Dora	111.65	1,970.87	119.84	1,984.02
Eustis	235.73	4,161.19	306.74	4,275.25
Harris	522.57	9,224.48	596.30	9,342.79
Griffin	380.09	6,709.42	396.48	6,735.71
Yale	94.66	1,670.96	97.39	1,675.34
Weir	220.30	3,923.98		

Blank cells indicate no estimates of potential inundation of septic tank drainfields in the Lake Weir Basin (see text).

Nutrient Loading from Point Source Discharges (L_{PSI})

Table 19 presents estimated nutrient loading from weak waste discharges by citrus processing plants in the UORB. These may be underestimates of total nutrient loading from citrus processing plants, as three of the plants were reported to discharge strong wastes to poorly maintained sprayfields for much of the 1980s (R. Roof, LCES, pers. com. 1993). Some excess strong wastes ran off the sprayfields into adjacent water bodies, but no estimates are available of discharge volume from sprayfields. These strong wastes contain process wastewater, including detergents, caustic solutions, high strength biochemical oxygen demand, and solids.

Lake Detention Times

Estimates of hydraulic detention times for the UORB lakes based on estimated annual water losses are rather similar to those based on estimates of annual water inflows (Table 20). An exception

Table 19. Nutrient concentrations, flow volumes [mean ± S.D. (N*)], and estimated nutrient loading from weak waste discharges by citrus plants in the upper Ocklawaha River Basin

Citrus Plant	Receiving Lake	TN Concentration (mg/L)	TP Concentration (mg/L)	Flow (m^3/mo)	TN Discharge (kg/yr)	TP Discharge (kg/yr)
Florida Foods	Yale	1.27±0.712 (22)	0.076±0.075 (22)	17,190±8,640 (5)	262.27	15.76
Golden Gem	Yale	1.149±2.628 (67)	0.213±0.304 (81)	387,220±245,680 (28)	5,340.59	991.86
Coca-Cola	Griffin	2.578±4.442 (64)	0.140±0.282 (78)	126,570 (1)	3,915.18	212.05
Silver Springs	Harris	0.688±1.129 (43)	0.296±1.110 (57)	34,490±18,590 (19)	284.56	122.64

*N is the number of observations.

Table 20. Estimates of water losses and water inflows, with corresponding detention times for lakes in the upper Ocklawaha River Basin. *Means are averages for the years 1984–90, except for Lake Weir, which is for the years 1980–90.*

Lake	Annual Water Losses (m^3)			Detention Time Based on Annual Water Losses (year)		
	Mean	Maximum	Minimum	Mean	Maximum	Minimum
Beauclair	52,325,133	80,984,809	28,704,813	0.16	0.28	0.11
Dora	61,372,796	92,285,971	31,968,818	0.82	1.55	0.57
Harris	88,482,075	127,485,116	37,178,677	3.01	7.06	2.04
Eustis	131,368,635	209,849,683	38,020,635	0.79	2.69	0.49
Griffin	157,008,739	352,474,557	42,571,622	0.54	2.03	0.22
Yale	13,536,100	52,415,359	1,861,189	4.45	34.17	1.15
Weir	—	—	—	—	—	—
Lake	**Annual Water Inflows (m^3)**			**Detention Timed Based on Annual Water Inflows (year)**		
	Mean	Maximum	Minimum	Mean	Maximum	Minimum
Beauclair	56,717,514	83,865,317	33,497,261	0.15	0.24	0.11
Dora	59,283,669	93,173,214	29,933,321	0.85	1.65	0.57
Harris	99,165,716	147,410,149	49,564,254	2.69	5.29	1.77
Eustis	138,180,760	216,434,988	45,670,159	0.75	2.24	0.50
Griffin	150,452,680	230,006,483	37,671,984	0.56	2.30	0.33
Yale	20,791,796	28,257,109	11,871,331	2.89	4.86	2.25
Weir	11,616,951	31,587,447	1,045,120	10.76	110.46	4.17

Note: — = no estimate available

was Lake Yale, for which detention time estimates based on water losses were considerably longer and showed high interannual variability. As discussed previously, use of the Lake Griffin net runoff coefficient in calculating water losses for Lake Yale is questionable because Lake Yale is hydraulically isolated from Lake Griffin for considerable periods.

Annual water losses could not be calculated for Lake Weir. Estimated detention times for Lake Weir based on estimates of annual water inflows were considerably larger than those for the other lakes, and highly variable from year to year. The long detention times for Lake Weir reflect the low rainfall experienced during much of the study period and the absence of other water sources for Lake Weir. There was no discharge from Lake Weir for much of the 1980s, because water elevations were below the elevation of the weir on the drainage canal leading from the lake.

In calculations of lake trophic responses to nutrient loading, detention times based on estimates of annual water inflows were used for Lakes Yale and Weir. For the other lakes, detention time estimates based on annual water losses were used in calculations of lake trophic responses.

NUTRIENT BUDGETS FOR LAKES IN THE UORB

Basin Overview

For presentation, nutrient sources often are aggregated into a reduced number of categories in figures. More detailed nutrient budgets are given in Appendixes D–J.

Estimated average annual total nutrient loadings for the lakes in the basin were greatest for Lakes Griffin and Eustis (Figures 10[A] and 11[A], Table 21). However, the ability of a lake to effectively assimilate nutrients is related to the size of the lake. When nutrient loadings are related to lake surface area, Lake Beauclair had nutrient loadings more than four times that of any other lake (Figures 10[B] and 11[B]). Total nutrient loadings were lowest for Lakes Weir and Yale. When expressed as nutrient loading per unit area, Lakes Yale and Harris-Little Harris had similar estimated loading rates.

Tributary flows were the major nutrient sources for Lakes Beauclair and Dora (Figures 10 and 11). The importance of tributary loading diminished further down the chain of lakes in Lakes Eustis and Griffin, although it remained the primary source for nitrogen. Muck farm discharges were the primary phosphorus source for Lake Griffin. Nutrient loadings for the

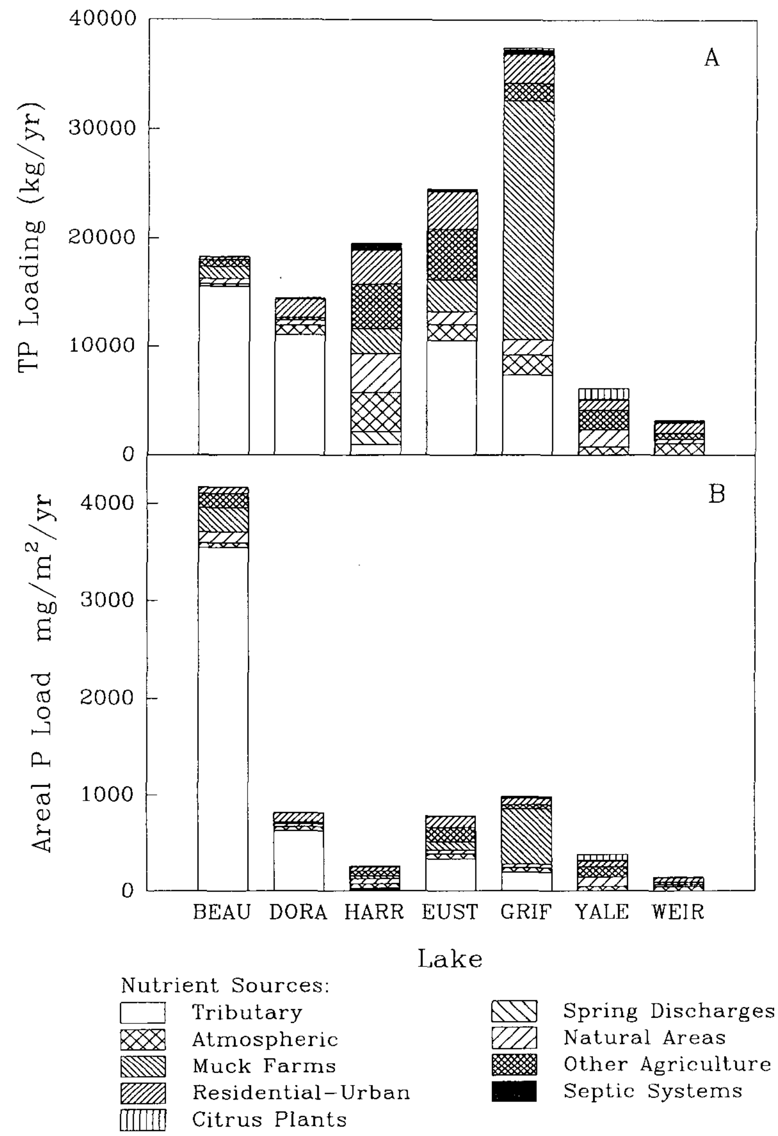

Figure 10. Summary of phosphorus loading estimates for lakes in the upper Ocklawaha River Basin: (A) total phosphorus loading and (B) areal phosphorus loading

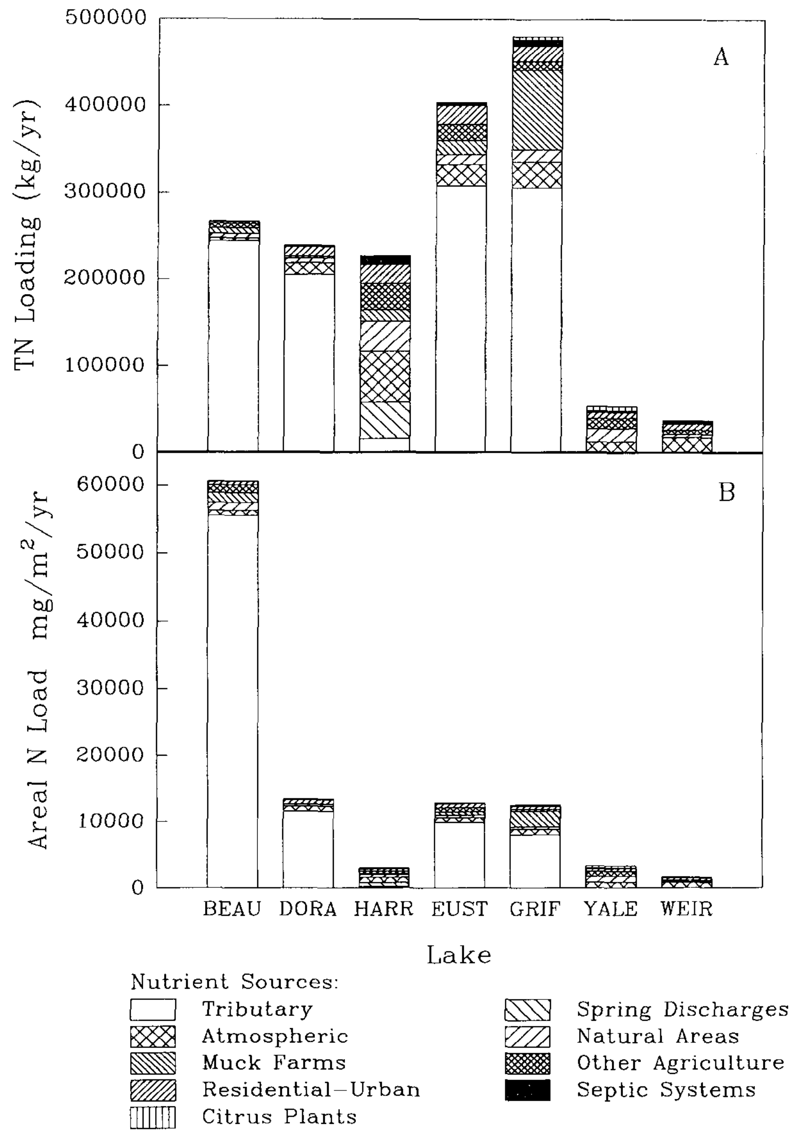

Figure 11. Summary of nitrogen loading estimates for lakes in the upper Ocklawaha River Basin: (A) total nitrogen loading and (B) areal nitrogen loading

Table 21. Upper Ocklawaha River Basin lakes—estimated average annual rates of nutrient loading and discharges, nutrient retention coefficients, and predicted phosphorus retention coefficients

Lake	TP Loading (kg/yr)	TP Outflow (kg/yr)	Mean P_{RC}	Range P_{RC}	Predicted P_{RC}
Beauclair	18,304	11,099	0.390	0.292–0.451	0.354
Dora	14,506	8,273	0.496	0.046–0.665	0.556
Harris	19,781	2,287	0.891	0.807–0.980	0.708
Eustis	24,609	7,117	0.728	0.519–0.946	0.559
Griffin	37,821	10,874	0.739	0.408–0.946	0.519
Yale	6,165	272	0.951	0.687–1.000	0.699
Weir	3,493	?	?	?–1.000	0.837
Lake	**TN Loading (kg/yr)**	**TN Outflow (kg/yr)**	**Mean N_{RC}**	**Range N_{RC}**	**Predicted N_{RC}***
Beauclair	266,490	205,215	0.201	0.064–0.382	
Dora	238,795	202,524	0.181	-0.031–0.324	
Harris	230,360	105,628	0.566	0.297–0.929	
Eustis	404,783	298,035	0.336	-0.061–0.700	
Griffin	481,278	330,591	0.379	-0.385–0.736	
Yale	53,802	7,943	0.849	0.179–1.000	
Weir	41,513	?	?	?–1.000	

*No literature equations are available to predict N_{RC}.

Note: ? = no estimate available

other lakes were divided among a number of sources, with no single dominant source.

Lake Beauclair

Nutrient loading to Lake Beauclair varied more than threefold among years, due to large differences in flows through the Apopka-Beauclair Canal (Figure 12, Appendix D). Over the study period, Apopka-Beauclair Canal discharges accounted for 85.0% of estimated phosphorus loading and 91.5% of estimated

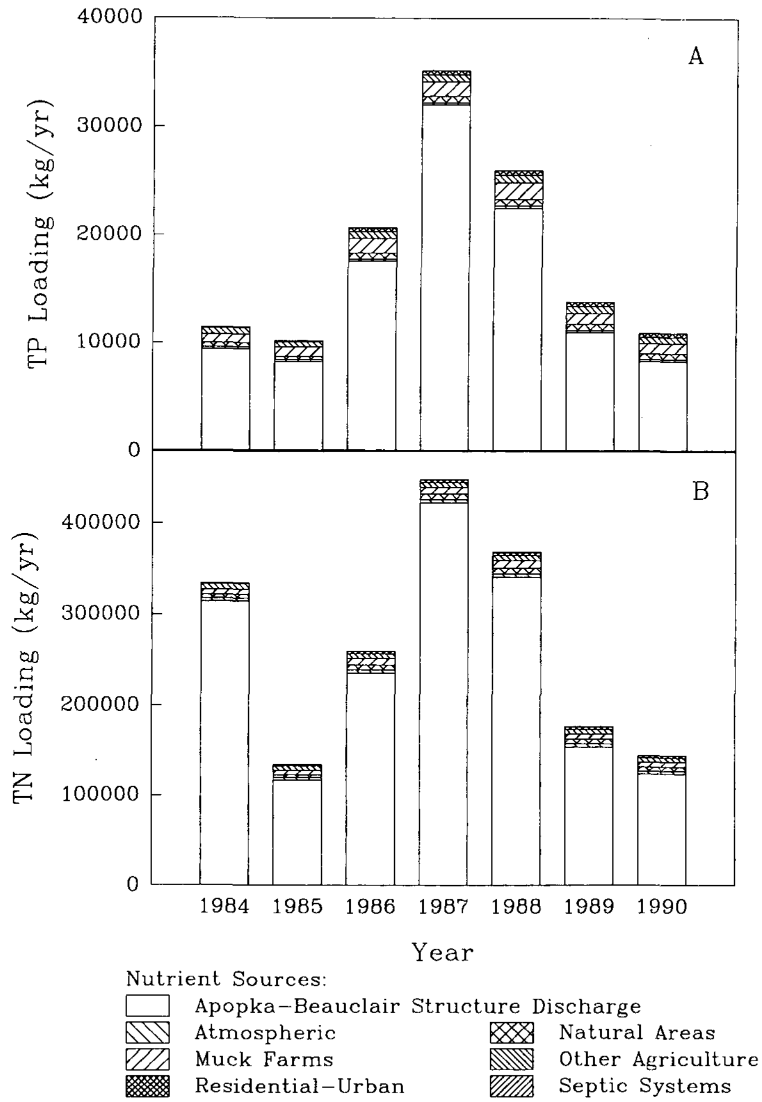

Figure 12. Nutrient loading to Lake Beauclair: (A) total phosphorus loading and (B) total nitrogen loading

nitrogen loading to Lake Beauclair. Muck farm discharges within the Lake Beauclair drainage basin accounted for 6.1% of TP loading and 2.3% of TN loading. It should be noted that muck farm discharges to Lake Apopka or to the Apopka-Beauclair Canal upstream of the Apopka-Beauclair lock and dam likely account for much of the nutrient loading in Apopka-Beauclair Canal discharges.

Other nutrient sources accounted for little of total nutrient loading to Lake Beauclair. Runoff from agriculture other than muck farms accounted for 3.3% of estimated TP loading and 2.0% of TN loading. Runoff from residential-urban land uses accounted for 1.6% of estimated TP loading and 0.8% of estimated TN loading to Lake Beauclair. Septic tank discharges made up less than 0.1% of TP and TN loading. These percentages attributed to septic tank loadings were not significantly increased if upper bounds to estimates of septic tank seepage (Table 18) are used (0.10% of TN loading, 0.13% of TP loading).

An estimated 39% of TP loading to Lake Beauclair was retained within the lake, with the remainder exported downstream to Lake Dora (Table 21). This TP retention efficiency is rather similar to that predicted by the Larsen and Mercier equation (Equation 29). Estimated retention of TN loading within Lake Beauclair was lower, averaging 20.1%.

There was a good correspondence between the trophic state model predictions and monitoring data of Lake Beauclair TP concentrations (Figure 13[A]). Reckhow's model (Equation 24) gave the closest correspondence, with an average 10.5% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 13[A]). It should be noted that the standard error of the Reckhow model predictions represents the variability in the data set used to develop the model and does not include any errors in the estimates of nutrient loading for the lakes in the UORB.

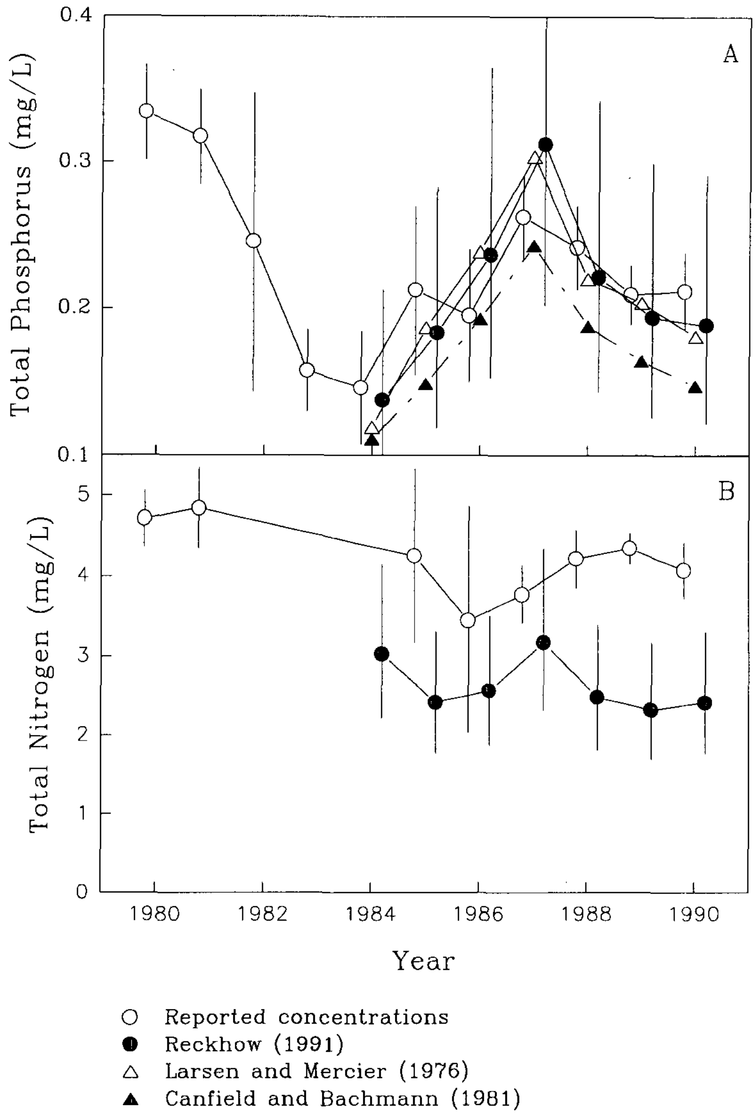

Figure 13. Lake Beauclair—comparison of observed and predicted nutrient concentrations: (A) total phosphorus and (B) total nitrogen. *Error bars are one standard error.*

Table 22. Deviations between reported mean annual nutrient concentrations and model predictions. *Mean values are for the years 1984–90, except for Lake Yale, which is 1980–90.*

Lake	TP % Deviation			TP Absolute Deviation (mg/L)			TN % Deviation	TN Absolute Deviation (mg/L)
	Reckhow	C&B	L&M	Reckhow	C&B	L&M	Reckhow	Reckhow
Beauclair	10.5	19.5	13.5	0.023	0.041	0.028	32.6	1.342
Dora	43.4	32.6	28.1	0.066	0.054	0.044	61.7	2.407
Eustis	12.7	26.7	48.8	0.010	0.013	0.028	53.4	1.316
Harris	31.9	44.4	101.0	0.012	0.013	0.032	53.8	1.039
Griffin	26.3	21.8	56.4	0.027	0.023	0.056	42.0	1.288
Yale	87.9	232.2	382.8	0.022	0.041	0.065	17.5	0.169
Weir	52.6	154.3	538.1	0.008	0.018	0.071	41.3	0.348

Note: Reckhow—Equations 24 (TP) and 26 (TN)
C&B (Canfield & Bachmann)—Equation 28
L&M (Larsen & Mercier)—Equation 29

The Reckhow model (Equation 26) predicted lower TN concentrations for Lake Beauclair than reported monitoring data, with an average 32.6% deviation (Figure 13[B], Table 22). However, reported TN concentrations are within the range of variability of model predictions, as the 95% confidence limits for the model prediction would be approximately two standard errors from the median.

Lake Dora

Discharges from Lake Beauclair were the dominant nutrient sources for Lake Dora, representing 76.5% of estimated TP loading and 85.9% of estimated TN loading (Figure 14, Appendix E). The largest other sources of nutrient loading were runoff from residential areas (9.3% of TP loading and 3.4% of TN loading) and atmospheric deposition (5.8% of TP loading and 5.7% of TN loading). Septic tank discharges made up less than 1% of estimated TP and TN loading to Lake Dora. These percentages attributed to septic tank loadings were not

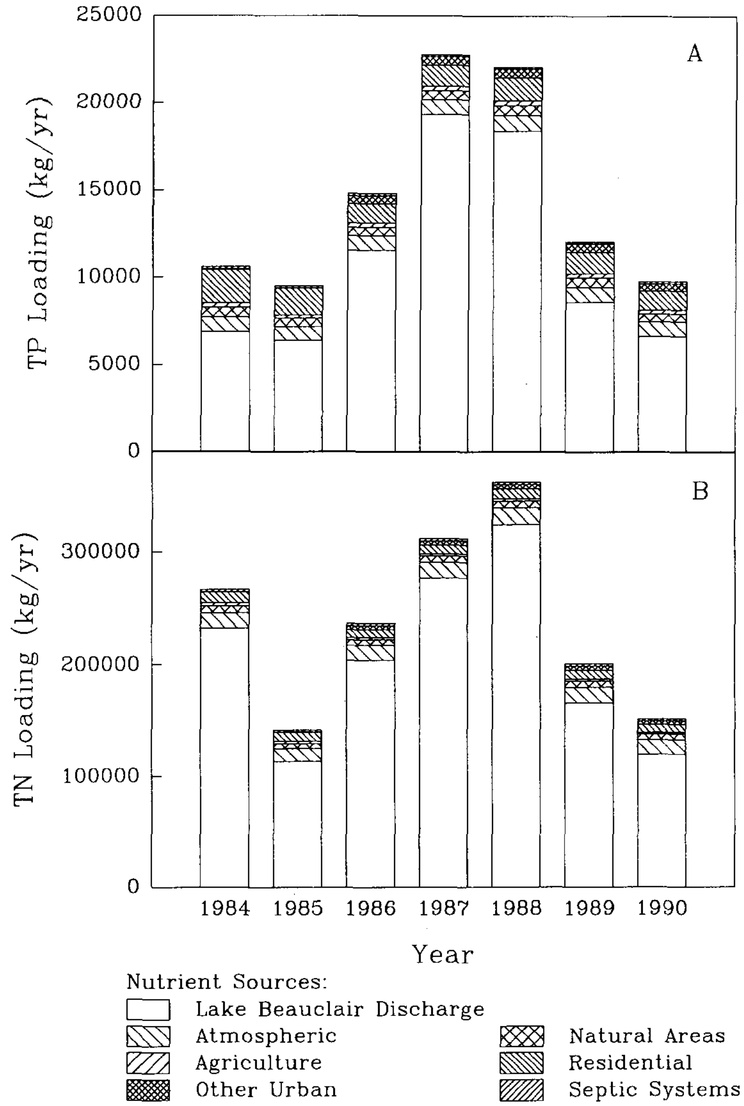

Figure 14. Nutrient loading to Lake Dora: (A) total phosphorus loading and (B) total nitrogen loading

significantly increased if upper bounds to estimates of septic tank seepage (Table 18) are used (still less than 1% of estimated TN and TP loading).

An estimated 49.6% of TP loading to Lake Dora was retained within the lake (Table 21), which is quite similar to that predicted by the Larsen and Mercier equation (Equation 29). Estimated retention of TN loading within Lake Dora was lower, averaging 18.1%, with a net export of TN (i.e., the downstream discharge was greater than the estimated loading) in 1987.

The reported TP concentrations for Lake Dora tended to be higher than the trophic state model predictions (Figure 15[A]). However, there is high variability in the reported monitoring data, so the model predictions are within the range of variability of observed data. The Larsen and Mercier model (Equation 29) predicted the highest TP concentrations for Lake Dora, so it gave the closest correspondence to the reported data (an average 28.1% deviation, Table 22, Figure 15[A]).

The Reckhow model (Equation 26) predicted substantially lower TN concentrations for Lake Dora than reported monitoring data, with an average 61.7% deviation (Figure 15[B], Table 22).

Lake Harris-Little Lake Harris

Nutrient loading to Lake Harris-Little Lake Harris was divided among a number of sources, with no single dominant source (Figure 16, Appendix F). Muck farm discharges represented 12.0% of estimated TP loading and 5.6% of estimated TN loading. One of two muck farms in the Lake Harris-Little Lake Harris subbasin was acquired by SJRWMD in 1991 and is being restored to wetlands. The largest single source of TP loading other than muck farms was runoff from agriculture, which represented 21.0% of estimated TP loading and 13.4% of estimated TN loading. In 1984–85, these agricultural loadings were largely from

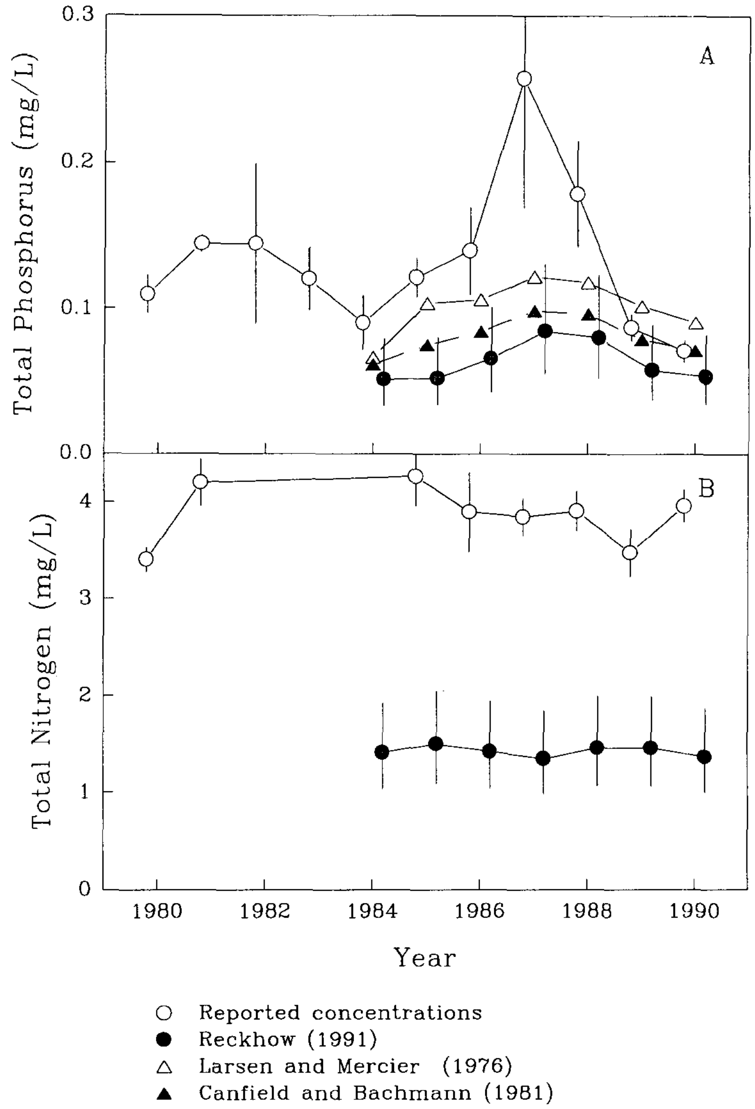

Figure 15. Lake Dora—comparison of observed and predicted nutrient concentrations: (A) total phosphorus and (B) total nitrogen. *Error bars are one standard error.*

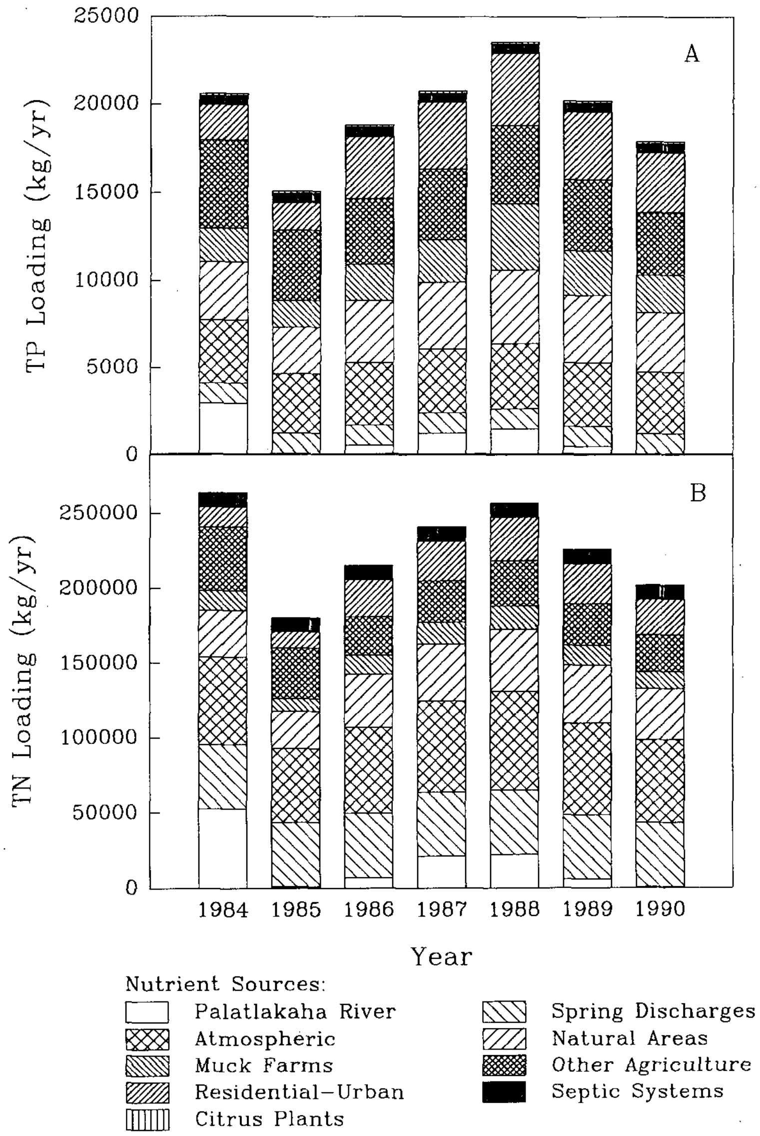

Figure 16. Nutrient loading to Lake Harris-Little Lake Harris: (A) total phosphorus loading and (B) total nitrogen loading

pasture and citrus groves. Nutrient loading from citrus decreased with the decline in citrus acreage following freezes in the mid-1980s, although estimated loadings from other upland agriculture increased. Due to the large surface area of Lake Harris-Little Lake Harris, atmospheric deposition represented a significant nutrient source, accounting for 18.4% of TP loading and 25.8% of TN loading. Also, due to the large amount of undeveloped areas in the Lake Harris-Little Lake Harris watershed, runoff from natural areas (wetlands and forest/rangeland) represented significant portions of total nutrient loading (18.3% of TP loading and 15.4% of TN loading).

Spring discharges represented a significant source of TN loading (18.9%) for Lake Harris-Little Lake Harris, but only 6.0% of TP loading. As discussed earlier, high nitrogen concentrations in spring discharges may be attributable to septic tank usage in the subbasin. However, nutrient loading attributed to septic tank seepage is considerably lower than that from spring discharges (4.1% of TN loading and 2.7% of TP loading). These percentages attributed to septic tank loadings were not significantly increased if upper bounds to estimates of septic tank seepage (Table 18) are used (4.1% of TN loading, 3.0% of TP loading).

Runoff from residential-urban land uses represented 16.1% of TP loading and 9.6% of TN loading to Lake Harris-Little Lake Harris, about two-thirds of which comes from residential housing. Weak wastes from citrus processing plants represented less than 1% of nutrient loading to Lake Harris-Little Lake Harris.

An estimated 89.1% of TP loading to Lake Harris-Little Lake Harris was retained within the lake (Table 21), an amount considerably larger than that predicted by the Larsen and Mercier equation (70.8%). Estimated retention of TN loading within Lake Harris-Little Lake Harris averaged 56.6%.

Reckhow's trophic state model (Equation 24) gave the closest correspondence with monitoring data for Lake Harris, with an average 31.9% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 17[A]). The Larsen

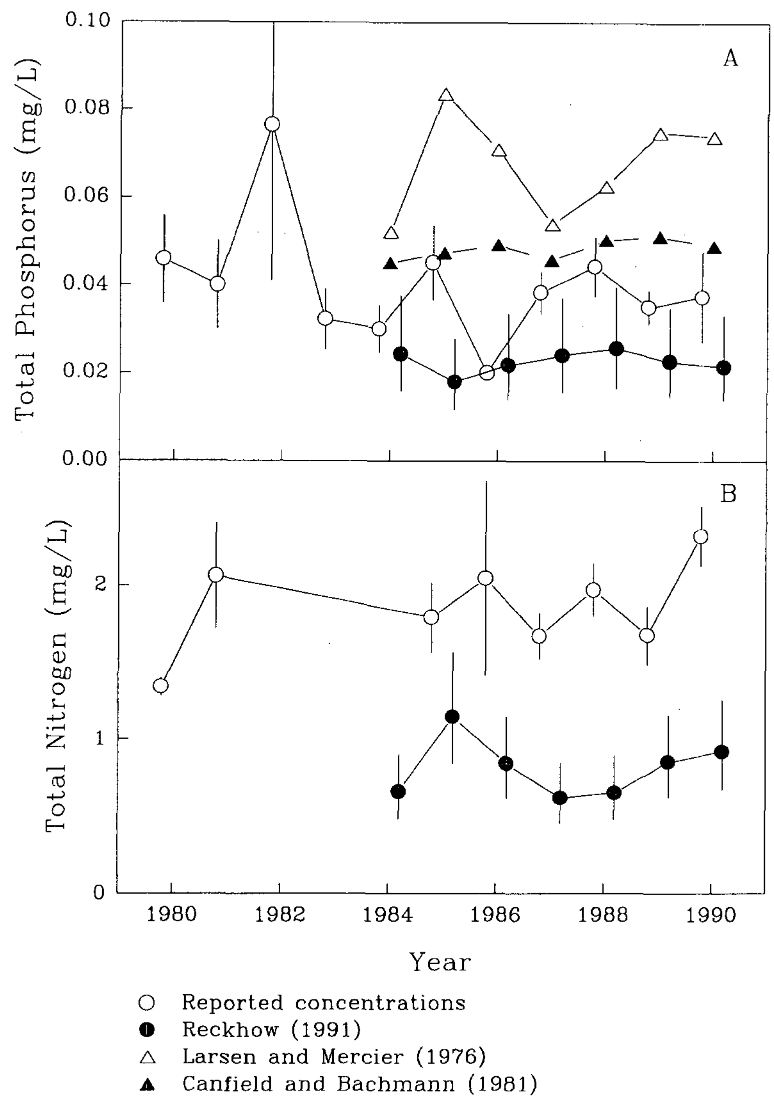

Figure 17. Comparison of observed nutrient concentrations for Lake Harris with predicted nutrient concentrations for Lake Harris-Little Lake Harris: (A) total phosphorus and (B) total nitrogen

and Mercier equation is much more sensitive to changes in detention times than the other models and tends to substantially overestimate TP concentrations in years with long estimated detention times (e.g., 1985 and 1989–90). The percent deviations of model predictions of TP from observed data tended to be higher for Lake Harris and the other less eutrophic lakes (Lakes Yale and Weir) than for more eutrophic lakes in the basin. However, the absolute deviations between predicted and observed TP appeared unrelated to trophic state (Table 22).

The Reckhow model (Equation 26) predicted substantially lower TN concentrations for Lake Harris-Little Lake Harris than reported monitoring data for Lake Harris, with an average 53.8% deviation (Figure 17[B], Table 22).

Lake Eustis

Discharges from Lake Dora represented 33.6% of estimated TP loading to Lake Eustis (Figure 18[A], Appendix G). The significance of TP loading from Lake Dora varied substantially over the years, ranging from 15.1% in 1984 to 57.8% in the high flow year of 1987. Discharges from Lake Dora represented 50.0% of estimated TN loading to Lake Eustis, with less interannual variability (43.1%–55.3%) (Figure 18[B], Appendix G). Discharges from Lake Harris-Little Lake Harris represented 9.3% of estimated TP loading and 26.1% of estimated TN loading to Lake Eustis.

Muck farm discharges represented 12.3% of estimated TP loading and 4.0% of estimated TN loading to Lake Eustis. Aside from Lake Dora discharges, the largest single source of TP loading was runoff from agriculture other than muck farms, which represented 18.9% of estimated TP loading and 4.6% of estimated TN loading. Estimated TP runoff from upland agriculture came primarily from confined animal feedlots (13.2% of TP loading). The reduced acreage of animal feedlots in the 1987 land use maps substantially lowered estimated TP loading (in 1984–85, 38.2% of TP loading was attributed to feedlot runoff, whereas in 1986–90 this was reduced to 2.0%).

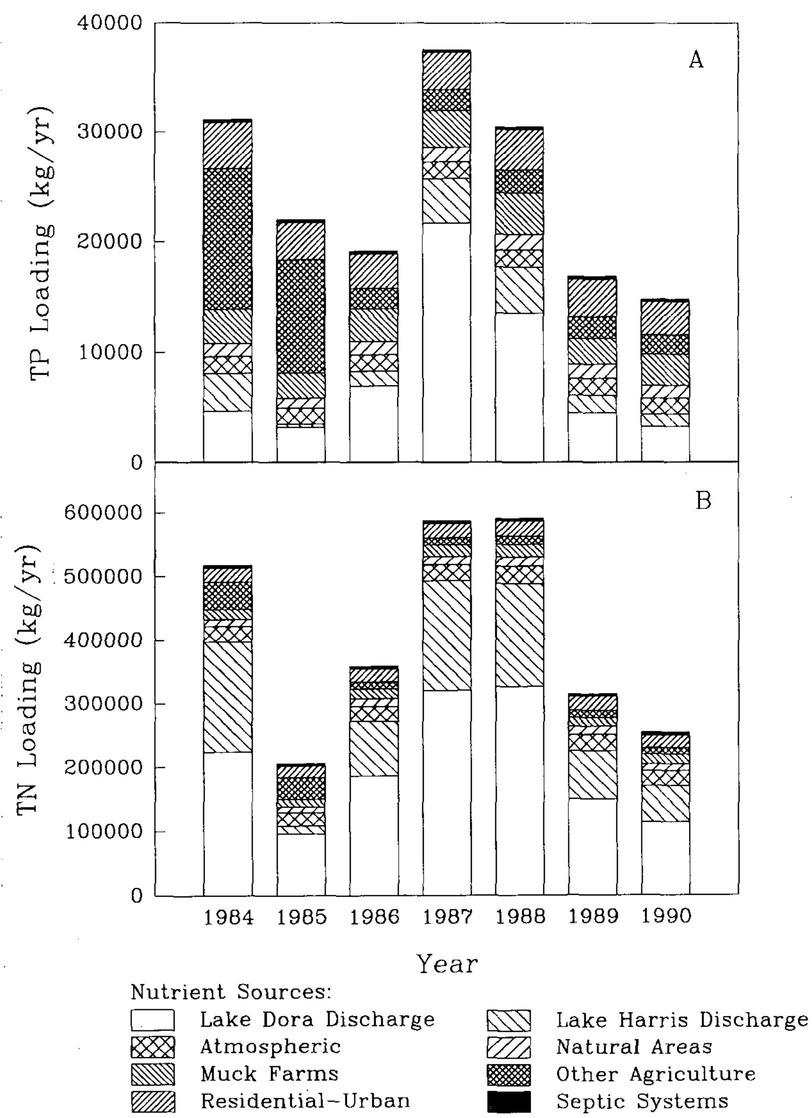

Figure 18. Nutrient loading to Lake Eustis: (A) total phosphorus loading and (B) total nitrogen loading

Runoff from residential land uses represented 11.7% of TP loading and 4.2% of TN loading to Lake Eustis. Septic tank effluents represented only 1% of estimated nutrient loading to Lake Eustis. The percentages attributed to septic tank loadings were not significantly increased if upper bounds to estimates of septic tank seepage (Table 18) are used (1.1% of TN loading, 1.2% of TP loading).

An estimated 72.8% of TP loading to Lake Eustis was retained within the lake (Table 21), which is considerably larger than that predicted by the Larsen and Mercier equation (55.9%). Estimated retention of TN loading within Lake Eustis averaged 33.6%, with a net export of TN in 1988.

There was a good correspondence between the trophic state model predictions and monitoring data of Lake Eustis TP concentrations (Figure 19[A]). Reckhow's model (Equation 24) gave the closest correspondence; with an average 12.7% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 19[A]).

As with most of the lakes, the Reckhow model (Equation 26) predicted lower TN concentrations for Lake Eustis than reported monitoring data, with an average 53.4% deviation (Figure 19B], Table 22).

Lake Griffin

Muck farm discharges represented 59.2% of estimated TP loading to Lake Griffin, with another 18.8% of TP loading attributed to discharges from Lake Eustis through Haines Creek (Figure 20[A], Appendix H). Estimated TN loading to Lake Griffin is nearly the reverse of TP loading, with 19.3% coming from muck farm discharges and 61.9% from Haines Creek discharges (Figure 20[B], Appendix H). Acquisitions of muck farms by SJRWMD since 1990 have substantially reduced their discharges to Lake Griffin.

Other nutrient sources for Lake Griffin are much smaller. Runoff from residential-urban land uses represented 6.9% of estimated

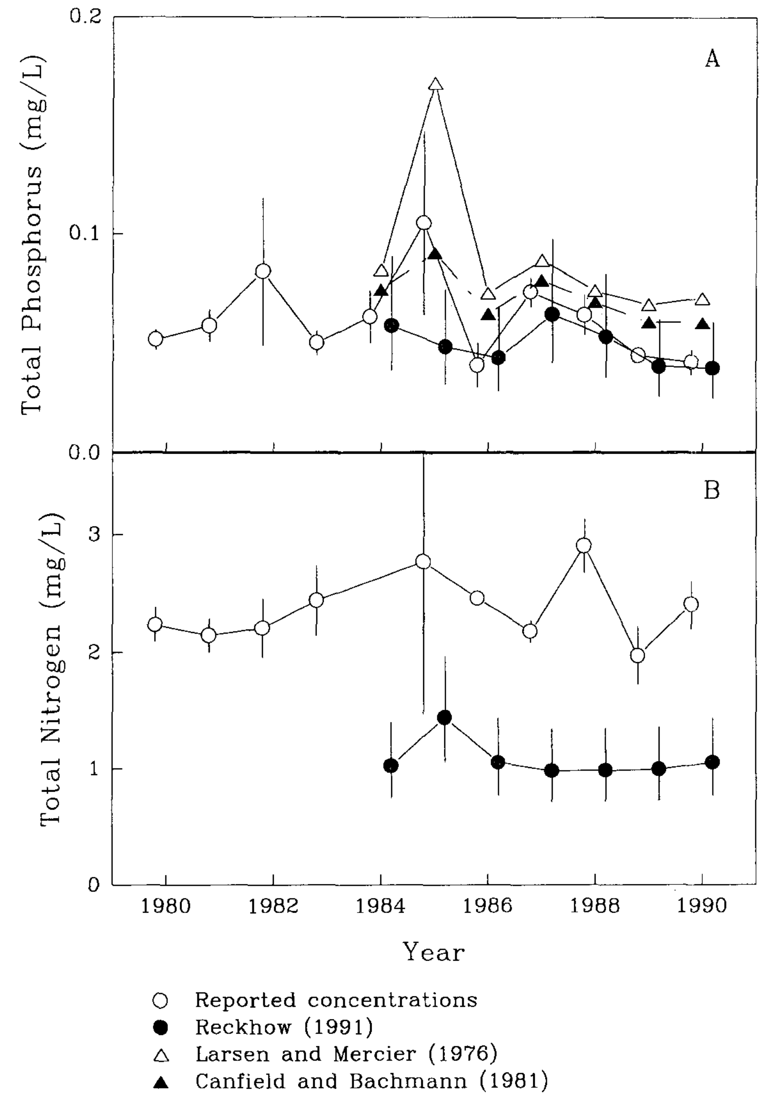

Figure 19. Lake Eustis—comparison of observed and predicted nutrient concentrations: (A) total phosphorus and (B) total nitrogen. *Error bars are one standard error.*

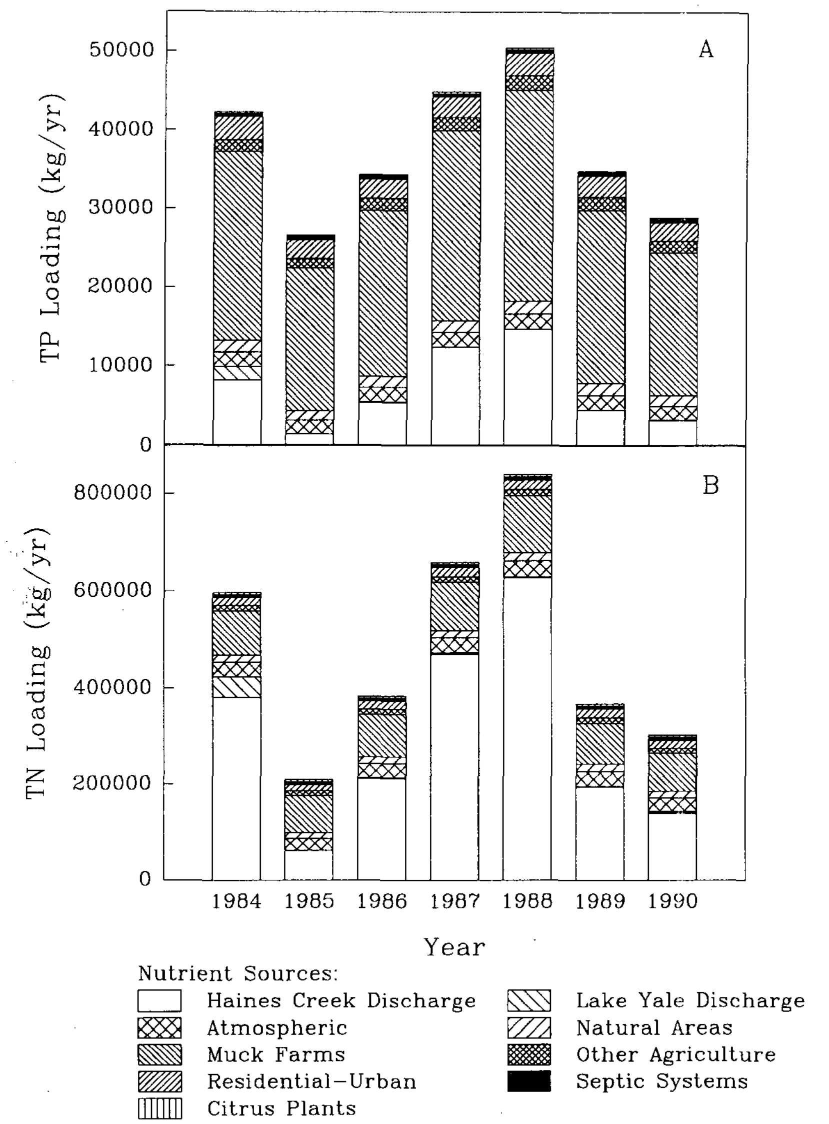

Figure 20. Nutrient loading to Lake Griffin: (A) total phosphorus loading and (B) total nitrogen loading

TP loading and 3.5% of TN loading. Weak waste discharges from citrus processing plants accounted for less than 1% of nutrient loading to Lake Griffin. Septic tank effluents represented 1.0% of TP loading and 1.4% of TN loading. These percentages attributed to septic tank loadings were not significantly increased if upper bounds to estimates of septic tank seepage (Table 18) are used (1.0% of TP loading, 1.4% of TN loading).

An estimated 73.9% of TP loading to Lake Griffin was retained within the lake (Table 21). As with several of the lakes, TP retention was considerably larger than that predicted by the Larsen and Mercier equation (51.9%). The data set used in development of the Larsen and Mercier equation consisted largely of temperate lakes. Greater retention found in the present study may reflect the longer growing season in Florida lakes. Estimated retention of TN loading within Lake Griffin averaged 37.9%. In the drawdown year of 1984, there was a substantial net export of TN and a considerably lower retention of TP than in other years.

There was generally good correspondence between the trophic state model predictions and monitoring data of Lake Griffin TP concentrations (Figure 21[A]). All of the models substantially underestimated TP concentration during the drawdown year of 1984; resuspension of bottom sediments during the drawdown may have increased nutrient concentrations in the lake. Overall, Canfield and Bachmann's model (Equation 28) gave the closest correspondence to the observed data, with an average 21.8% deviation from the reported data. Predictions of Reckhow's model (Equation 24) also closely corresponded with the reported data (Table 22, Figure 21[A]).

The Reckhow model (Equation 26) predicted lower TN concentrations for Lake Griffin than reported monitoring data, particularly during 1984, with an average 42.0% deviation (Figure 21[B], Table 22).

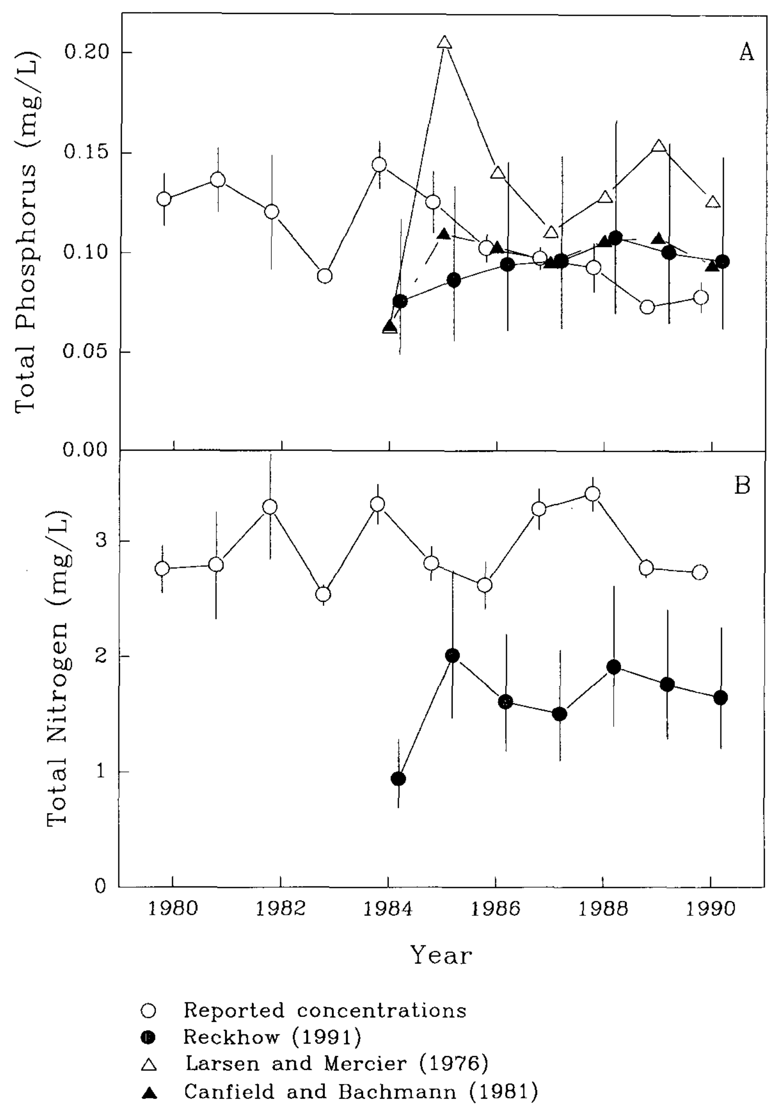

Figure 21. Lake Griffin—comparison of observed and predicted nutrient concentrations: (A) total phosphorus and (B) total nitrogen. *Error bars are one standard error.*

Lake Yale

Nutrient loading to Lake Yale was divided among a number of sources, with no single dominant source (Figure 22, Appendix I). One of the major nutrient sources was runoff from upland agriculture, which represented 28.4% of estimated TP loading and 23.4% of estimated TN loading. In 1984–85, these agricultural loadings were primarily from pasture and citrus groves. Nutrient loading from citrus decreased with the decline in citrus acreage following freezes in the mid-1980s, whereas estimated loadings from other upland agriculture increased.

Another major nutrient source for Lake Yale was runoff from natural areas (26.6% of estimated TP loading and 29.7% of TN loading), resulting from a large area of undeveloped land in the Lake Yale drainage basin. Atmospheric deposition contributed an estimated 13.0% of TP loading and 23.8% of TN loading.

Runoff from residential areas accounted for 9.2% of TP loading and 6.0% of TN loading to Lake Yale. TP loading from runoff from other urban land uses increased from 0.8% of total loading in 1980–85 to 8.1% in 1986–90, which resulted from an increase in area of development in the 1987 land use maps (Table 14). Weak waste discharges from citrus processing plants represented 16.8% of TP loading and 10.3% of TN loading. Septic tank effluents represented an estimated 1.6% of TP loading and 3.1% of TN loading. These percentages attributed to septic tank loadings were not significantly increased if upper bounds to estimates of septic tank seepage (Table 18) are used (1.6% of TP loading, 3.1% of TN loading).

An estimated 95.1% of TP loading to Lake Yale was retained within the lake (Table 21), a percentage considerably larger than that predicted by the Larsen and Mercier equation (69.9%). Estimated retention of TN loading within Lake Yale averaged 84.9%.

Reckhow's trophic state model (Equation 24) gave the closest correspondence with monitoring data for TP in Lake Yale, with

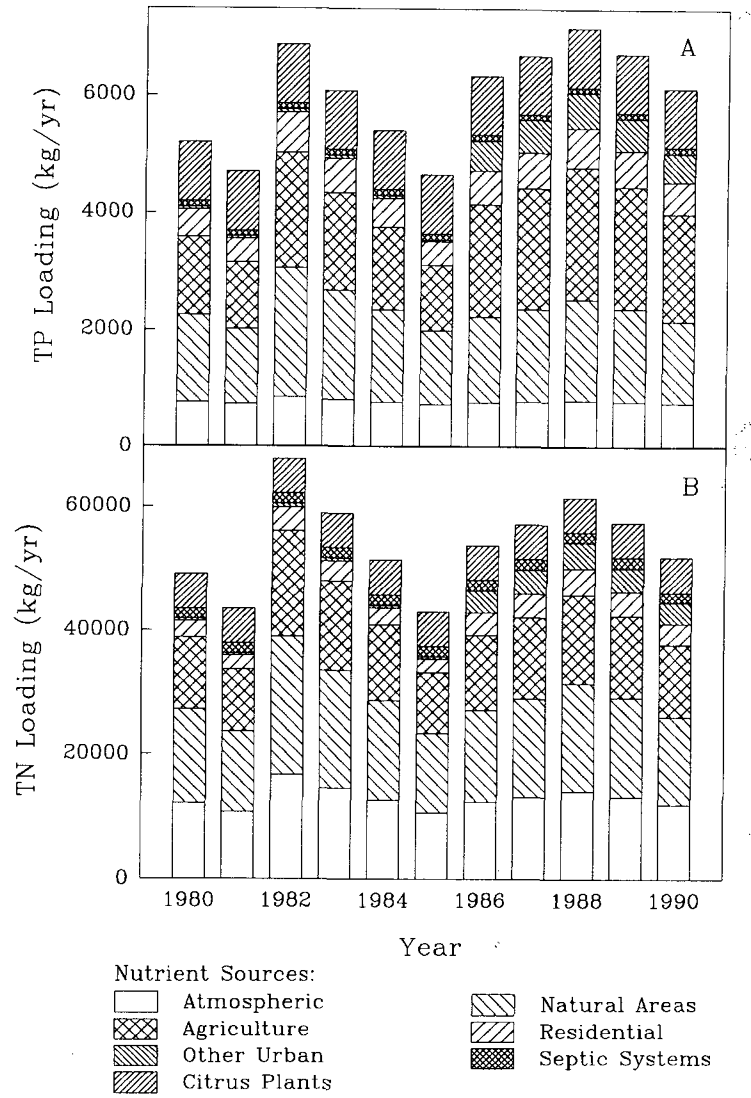

Figure 22. Nutrient loading to Lake Yale: (A) total phosphorus loading and (B) total nitrogen loading

an average 87.9% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 23[A]). The large percent deviations for Lake Yale reflect, in part, the substantial variability in reported monitoring data. Additionally, utilization of nutrients by the substantial amount of submersed aquatic vegetation in Lake Yale may result in lower nutrient water column concentrations than predicted by the trophic state models (Canfield et al. 1983). The models of Canfield and Bachmann (1981) and Larsen and Mercier (1976) would be particularly likely to overestimate nutrient concentrations in lakes with large amounts of aquatic vegetation. These models were developed from data sets consisting primarily of temperate lakes, which typically have lower amounts of aquatic vegetation.

The Reckhow model (Equation 26) predictions of TN in Lake Yale closely corresponded to the observed data, with an average 17.5% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 23[B]).

Lake Weir

Nutrient loading to Lake Weir was divided among a number of sources, with no single dominant source (Figure 24, Appendix J). The largest single nutrient source for Lake Weir was atmospheric deposition, accounting for an estimated 32.2% of TP loading and 47.4% of TN loading to the lake. The other major nutrient source was runoff from residential areas, contributing 32.8% of TP loading and 20.1% of TN loading.

Runoff from agriculture accounted for 16.5% of TP loading and 12.0% of TN loading to Lake Weir. As with several other lakes, in 1984–85 these agricultural loadings were primarily from pasture and citrus groves. Nutrient loading from citrus decreased with the decline in citrus acreage following freezes in the mid-1980s, whereas estimated loadings from other upland agriculture increased.

Septic tank effluents represented an estimated 6.4% of TP loading and 9.4% of TN loading to Lake Weir. As mentioned previously,

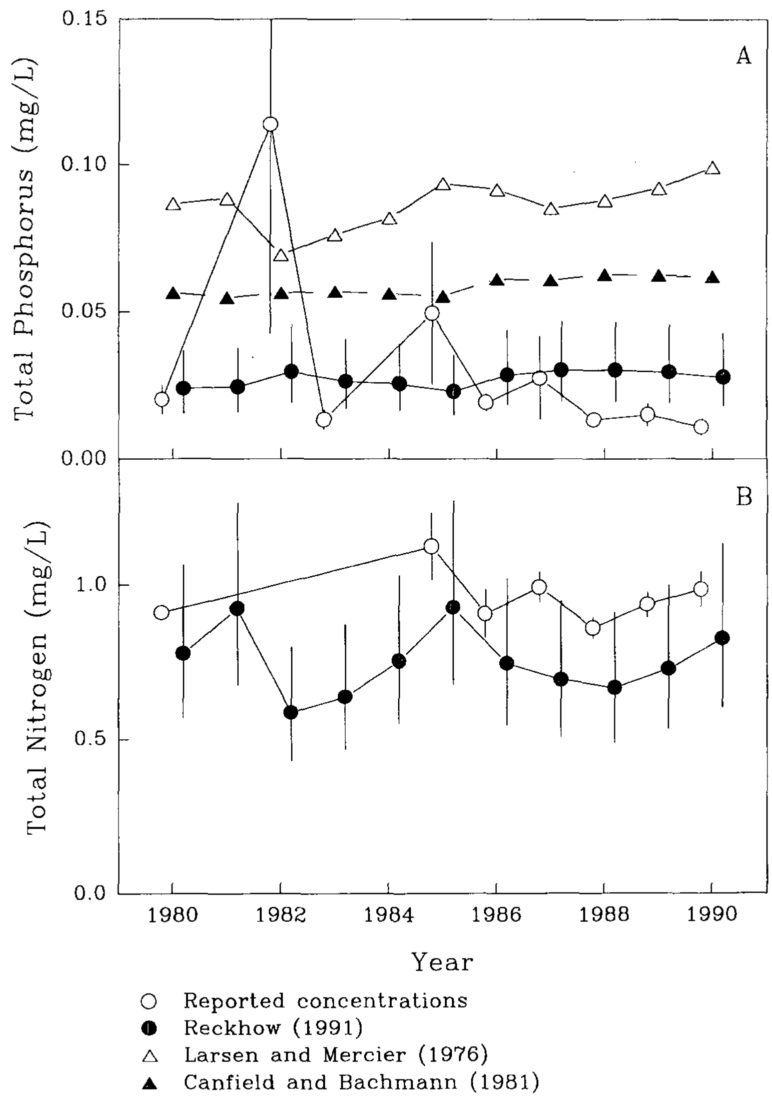

Figure 23. Lake Yale—comparison of observed and predicted nutrient concentrations: (A) total phosphorus and (B) total nitrogen. *Error bars are one standard error.*

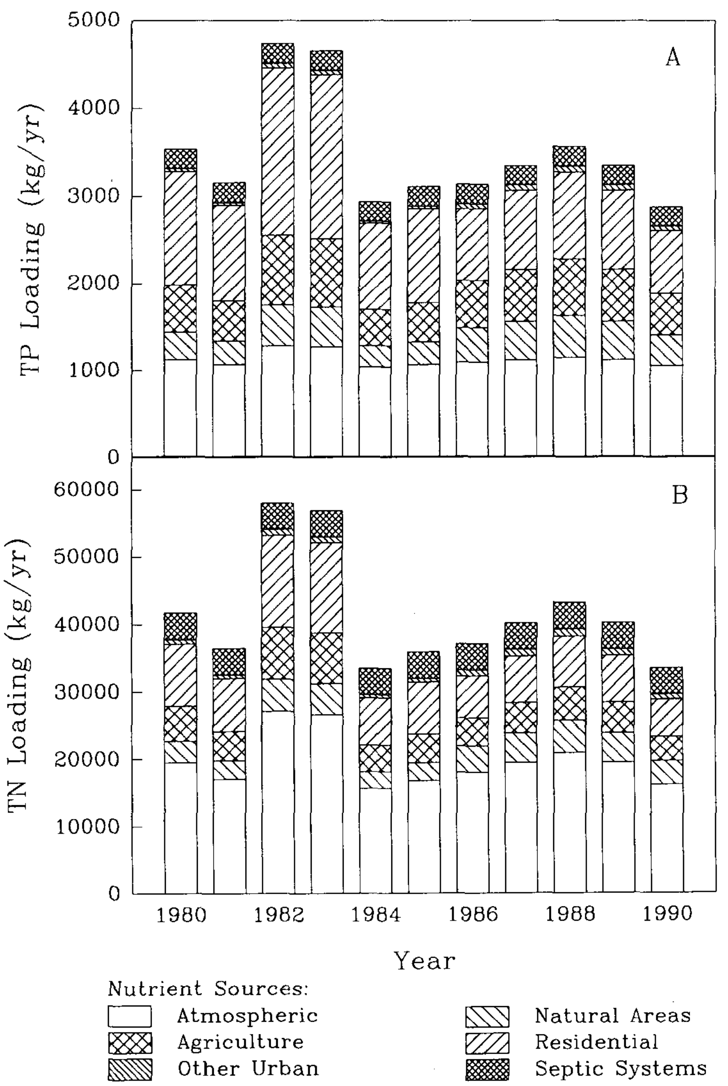

Figure 24. Nutrient loading to Lake Weir: (A) total phosphorus loading and (B) total nitrogen loading

there are no estimates of potential inundation of septic tank drainfields in the Lake Weir subbasin, but low lake levels in recent years make it unlikely that flooding of septic tank drainfields has been a problem.

In most years, water elevations in Lake Weir were below the elevation of the weir on the drainage outlet, so there was no discharge from the lake. Lake elevations exceeded the weir elevation only for portions of 1980, 1982, 1983, and 1984. Estimates of nutrient retention in these years cannot be made because there are no measurements of discharge volume from the lake, but nutrient retention is likely to be very high. The Larsen and Mercier equation predicts an average 83.7% retention of TP loading for Lake Weir (Table 21).

Reckhow's trophic state model (Equation 24) gave the closest correspondence with monitoring data for Lake Weir, with an average 52.6% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 25[A]). The reported TP concentrations in Figure 25(A) are from monitoring of Lake Weir by SJRWMD and are lower than reported by other studies of the lake. Crisman et al. (1992) reported a mean TP of 0.038 mg/L in a study of Lake Weir from 1987–89 and a mean of 0.028 mg/L from earlier studies of the lake. The values reported by Crisman et al. are more similar to the model predictions of the Canfield and Bachmann model (Equation 28). Reasons for the discrepancy between TP reported by SJRWMD and Crisman et al. are not known.

The Reckhow model (Equation 26) predictions of TN in Lake Weir were similar to the observed data, with an average 41.3% deviation from the reported data and strong overlap of standard errors (Table 22, Figure 25[B]).

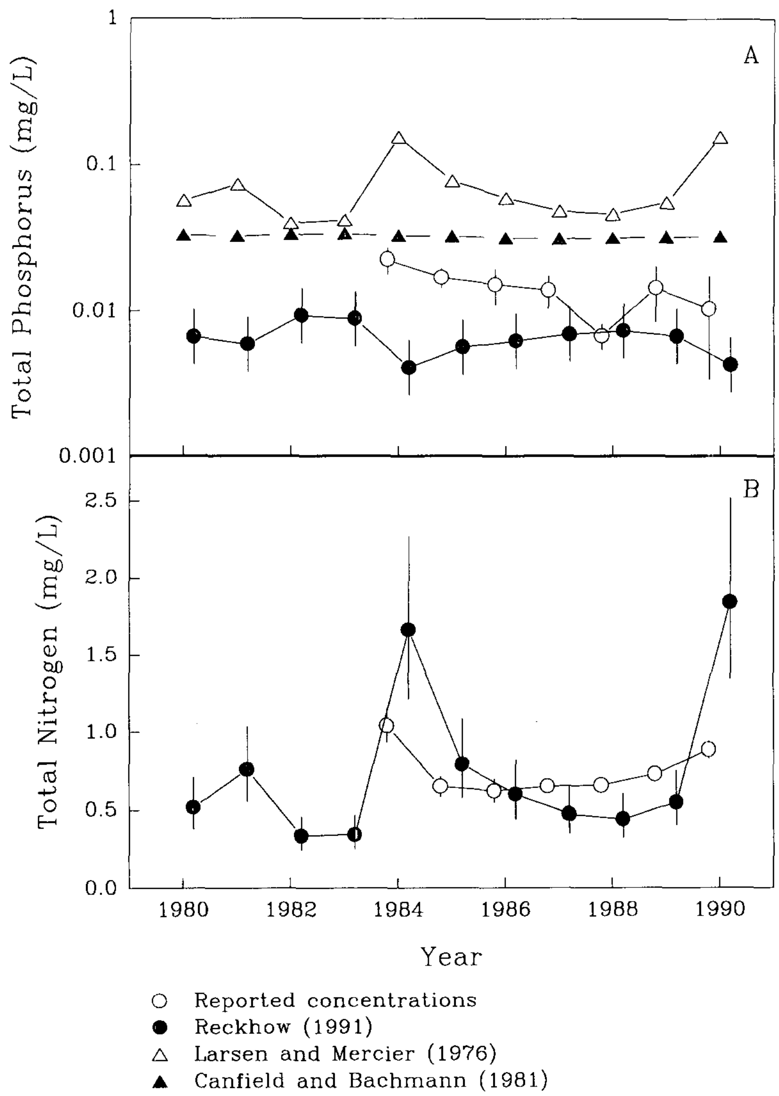

Figure 25. Lake Weir—comparison of observed and predicted nutrient concentrations: (A) total phosphorus and (B) total nitrogen. *Error bars are one standard error.*

DISCUSSION

Assessment of Potential Limiting Nutrients through N:P Ratios

Patterns of potential nutrient limitation indicated by TN:TP ratios (mixed nutrient limitation in Lake Beauclair and possibly Lake Griffin and potential limitation by phosphorus in the other lakes) appear consistent with interpretations of nutrient enrichment experiments in Lake Apopka. Schelske et al. (1992) and Aldridge et al. (1993) attributed primary limitation by nitrogen in Lake Apopka to excessive phosphorus loading from muck farms in the Lake Apopka Basin. Lake Beauclair is immediately downstream from and receives much of its water supply from Lake Apopka, so it may be expected to show similar patterns of nutrient limitation. Lake Griffin also has a number of muck farms which are the primary source of phosphorus for the lake (Figure 20[A]). TN:TP ratios in muck farm discharges in the UORB are low (mean 5.73), which would tend to drive the receiving water body toward nitrogen limitation.

TIN:TIP ratios for the lakes in the UORB are more difficult to interpret. TIN:TIP ratios are generally considerably lower than TN:TP ratios, which tends to indicate potential nitrogen limitation in all of the lakes except Lake Griffin. This pattern is not consistent with the interpretation that phosphorus loading from muck farms pushes receiving water bodies toward nitrogen limitation. However, the reliability of the TIN:TIP ratios is suspect because of frequent occurrences of TIN and TIP concentrations below detection limits. Below-detection-limit concentrations occurred in samples from all lakes and were more common in the lakes with lower nutrient concentrations: Lakes Yale, Weir, and Harris-Little Harris. In some cases, the TIN:TIP ratio becomes simply a ratio of detection limits.

Several factors may complicate the interpretation of N:P ratios. Different algal species are likely to have different nutrient requirements, as implied by the conflicting results from Lake Apopka to standard algal bioassays and experiments with naturally occurring algal assemblages. Nutrients may not be limiting, particularly in eutrophic lakes, due to excessive nutrient loading or high algal concentrations that contribute to light limitation (Paerl and Bowles 1987). TN and TP can include components that are not readily available for phytoplankton growth. Inorganic nutrient concentrations also may not accurately indicate nutrient limitation because of the storage of excess phosphorus in algal cells (see discussion in Schelske et al. 1992 and Aldridge et al. 1993). Because of these potentially confounding factors, nutrient limitation is best determined by experimental nutrient enrichment or nutrient dilution assays. In the absence of such experimental data, the interpretation most consistent with available information is potential phosphorus limitation, except where excessive phosphorus loading has led to potential nitrogen limitation.

ASSESSMENT OF ACCURACY OF THE NUTRIENT LOADING ESTIMATES

Potential Errors in Nutrient Loading Estimates

Many of the nutrient loading estimates are based on indirect sources of information, rather than on direct measurement. The most significant sources of error are probably in the estimates of flows through ungauged tributaries, muck farm discharges, stormwater runoff from other land uses, and ground water seepage.

As mentioned previously, the net runoff coefficients used in determination of ungauged tributary flows are averaged over the entire drainage basin for the stream gauge, rather than being specific for each lake subbasin. Ungauged tributary flows are significant nutrient sources for some of the lakes in the Haines Creek Basin (including Lakes Dora, Harris, and Eustis) (Figures 10 and 11). Errors in estimates of flow volumes between the lakes in this basin, particularly the occurrence of flow reversals, could significantly affect loading estimates.

Flow reversals could be of particular importance between Lakes Harris and Eustis because of the difference in nutrient concentrations between these two lakes (Table 11). There are reports of flow reversals in the Dead River, which connects Lakes Harris-Little Harris and Eustis. USGS measured current velocities on nine occasions between 1971 and 1976. Flow reversals were recorded on two of the nine dates (L. Fayard, USGS, pers. com. 1990). As mentioned previously, it was assumed that discharges from Lake Harris-Little Lake Harris were zero in the 16 of 84 months during 1984–90 in which it was estimated that there were net negative discharges from Lake Harris-Little Lake Harris. Assuming that these negative discharges were true flow reversals in the Dead River would only slightly increase the nutrient loading to Lake Harris-Little Lake Harris (estimated average annual TP loading would increase 0.6%, range 0.05–1.2%, and estimated average annual TN loading would increase 1.9%, range 0.1–3.9%). Assuming that the negative discharges were flow reversals would not change the estimates of nutrient loading for Lake Eustis, but would slightly reduce the retention time for Lake Eustis because the flow reversals would be an additional loss of water from the lake. The average retention time would decrease only 1.4%, from 0.79 years to 0.78 years. This decrease would have negligible effects on the predicted trophic state in Lake Eustis because the models used are quite insensitive to small changes in retention time. Thus, if the estimated net negative discharges from Lake Harris-Little Lake Harris are accurate indications of the extent of flow reversal, then they would have negligible effects on nutrient loading and water quality in Lakes Harris-Little Harris and Eustis.

Flow reversals in the Yale-Griffin Canal may also potentially have significant effects on nutrient loading and water quality. As mentioned previously, it was assumed that discharges from Lake Yale were zero in the 33 of 84 months during 1984–90 in which it was estimated that there were net negative discharges from Lake Yale. Water levels in Lake Yale are typically higher than those in Lake Griffin. However, in 13 months during the study period, water levels were higher on at least one day in Lake Griffin than in Lake Yale (USGS 1981–91). The estimates of net negative discharges do not appear to be accurate indicators of flow reversals into Lake Yale, because they occurred in 20 months in which water levels were always higher in Lake Yale than in Lake Griffin. Nevertheless, I calculated the potential effects on nutrient loading to Lake Yale for months in which net negative discharge estimates occurred and water levels were higher on at least 1 day in Lake Griffin than in Lake Yale. In these calculations, the nutrient concentrations used were the average concentrations in Lake Griffin (Table 11). The use of Lake Griffin nutrient concentrations may underestimate nutrient levels in water entering Lake Yale because three of the muck farms operating at that time discharged directly into the Yale-Griffin Canal (Table 2). Under the assumed conditions, the flow reversals would increase average annual TP loading to Lake Yale by 1.7% (range 0–6.3%) and increase annual average TN loading by 4.9% (range 0–18.5%). Assuming that the negative discharges were flow reversals would not change the estimates of nutrient loading for Lake Griffin, but would slightly reduce the retention time for the lake by 0.6%, from 0.536 years to 0.532 years. Thus, if the estimated net negative discharges from Lake Yale are accurate indications of the extent of flow reversal in months in which Lake Griffin water levels exceed those in Lake Yale, then they could have significant effects on nutrient loading and water quality in Lake Yale in some years, but would have negligible effects on Lake Griffin.

The estimates of muck farm discharges depend in large part on records of pump discharges reported by the farmers. The permit records are very incomplete. I assumed that the pump discharge records were accurate for the months in which data were reported, but it is possible that discharges were underestimated in these records. Although the multiple regression equation accounts for 78.5% of the reported variability in pump discharge, the uncertainty increases in application of this equation to other farms, which may have been operated differently and experienced different rates of seepage and runoff from surrounding areas.

Independent estimates of pump discharges for several of the muck farms were calculated from information on average daily discharges reported in Modica and Associates (n.d.). Estimates of annual pump discharges determined from the multiple regression analysis in this report are compared with estimates developed from data presented by Modica and Associates (Table 23). The

Table 23. Comparison of estimates of annual pump discharges from muck farms in the upper Ocklawaha River Basin developed by the multiple regression analysis in this report with estimates reported in Modica and Associates (n.d.)

Farm	Estimated Water Discharge (m^3/yr)	
	Regression Analysis	Modica and Associates
Pine Meadows	4,487,300	2,316,400
JA-MAR	2,153,400	2,380,700
S.N. Knight, Leesburg	2,332,800	5,331,900
S.N. Knight, Lisbon	16,071,100	18,823,900
Paulhamus (Eustis muck farm)	3,784,800	59,518,400
Long Farms North	4,667,600	4,959,900

estimated pump discharges from the regression analysis shown in Table 23 differ slightly for several of the farms from those reported in Table 13, which are estimated discharges from the farm retention ponds. For most farms, the estimates from the regression analysis and from Modica and Associates are similar, the major discrepancy being for Paulhamus Farm. For Paulhamus Farm, Modica and Associates reported that the Lake County SCS calculated an average discharge of 43.2 mgd. In contrast, SJRWMD permit records for Paulhamus Farm show an average monthly discharge of only 59.2 million gallons (based on 11 months of data; range, 0–287.8 million gallons/month). The estimated discharge volumes for Paulhamus Farm reported in Modica and Associates appear to be much too high.

Excluding Paulhamus Farm, the discharge estimates from the regression analysis tend to be slightly lower than the estimates of

Modica and Associates (n.d.). The only exception is for Pine Meadows, for which the regression analysis estimate is about twice that from Modica and Associates. However, Modica and Associates do mention that their estimate for Pine Meadows is for an average year, and discharges during wet years are considerably higher. The total of estimated discharges reported by Modica and Associates for all of the farms except Paulhamus Farm is about 14% higher than the total from the regression analysis. The similarity of these two estimates indicates that the regression estimates of muck farm discharges are of the correct order of magnitude.

There were no direct measurements of nutrient loading in stormwater runoff from other land uses in the UORB. Stormwater runoff was estimated from runoff coefficients, nutrient concentrations, and attenuation coefficients measured in other areas. The runoff coefficient may be the least precise term of a stormwater loading function. Although it may not be an accurate measure of individual runoff events, it is thought to provide a good description of long-term average runoff estimates (Reckhow et al. 1990, draft). The attenuation coefficients used do not consider any constructed stormwater management facilities (such as retention ponds) that may trap nutrients. Stormwater retention facilities are likely to be lacking in older developments; however, developments built since 1982 have been required to use appropriate best management practices to treat stormwater. Thus, stormwater nutrient loading from recent developments may be overestimated in the loading calculations.

Errors in estimates of ground water seepage may include seepage attributable to septic systems and other contributions to seepage. Nutrient contributions from septic systems within the service areas of municipal waste treatment plants are not considered. Estimates of loading from septic tank effluents do not consider failing systems, other than those with potentially inundated drainfields. Camp, Dresser & McKee (CDM) (1991) reported that annual septic tank failure rates average about 1–2%. In estimating septic system nutrient loading for lakes in the UORB, CDM staff assumed 2–3% annual failure rates and that septic tank systems failures would not be repaired for 5 years. Thus, in an average year, 10–15% of the septic tanks were expected to be failing. It was then assumed that there was 0% soil retention of nutrients from failing septic tanks. Even under these generous assumptions, it was concluded that septic tank failures have only a limited impact on overall nonpoint pollution discharges for the UORB lakes in Lake County. In one respect, the estimates of nutrient loading from septic tanks in the present study may be overestimates, because it was assumed that septic systems on all sides of the lakes contribute to nutrient loading. However, an analysis of ground water flow patterns may show that the direction of flow is away from the lake on some sides of the lake. Overall, it appears unlikely that septic tank nutrient loading is significantly higher than estimated.

As mentioned previously, the nutrient budgets do not include any estimates of seepage other than from septic tanks. Exclusion of other sources of seepage seems reasonable for phosphorus because of its limited mobility in soils (Jones and Lee 1977; Sherwood and Crites 1984; Canter and Knox 1985). However, seepage may be a more important source of nitrogen, due to its mobility in soil and the apparently elevated levels of nitrogen in spring discharges in the basin (Table 8). Potential nitrogen sources for ground water seepage in the basin include a number of sprayfields for municipal and package sewage treatment plants, and citrus processing plant sprayfields, as well as septic tanks.

The close correspondence between TP concentrations predicted by the trophic state models and observed concentrations in the lakes (Table 22, Figures 13[A], 15[A], 17[A], 19[A], 21[A], 23[A], and 25[A]) suggest that the total TP loading estimates are of the correct order of magnitude. However, the model predictions for in-lake TN concentrations consistently underestimate observed concentrations, except for Lakes Yale and Weir. These errors in the model prediction may be due to underestimates in TN loading or inadequacies in the TN trophic state model. As suggested above, a likely source for underestimates of TN loading is ground water seepage. Alternatively, the trophic state models may be less adequate for TN because nitrogen dynamics are more complex than those of TP and there has been less model development for TN. The data for Lakes Beauclair and Dora point toward inadequacies of the TN trophic state model. It is very likely that nearly all of the nitrogen loading to these lakes is from tributary discharges (Figures 12[B] and 14[B]), which should be estimated relatively accurately. However, in-lake TN concentrations are also substantially underestimated by the Reckhow trophic state model for these lakes (Figures 13[B] and 15[B]).

Comparisons with Previous Nutrient Budgets for the UORB Lakes

There have been several previous studies which estimated nutrient loading for lakes in the UORB. The most consistent difference from the present study is that previous studies tended to use larger drainage basins for estimates of stormwater runoff, with no allowance for nutrient trapping in transport from the runoff source to the water body. The causes for the discrepancies in estimates of drainage basin size cannot be determined in most cases because the earlier studies generally did not include maps of the drainage basins. However, one possible cause is that more detailed drainage basin maps used in the present study exclude noncontributing areas from calculations of surface runoff (Figure 5).

East Central Florida Regional Planning Council (ECFRPC) (1971). ECFRPC developed nutrient budgets for Lakes Beauclair, Dora, Eustis, and Griffin and partial budgets for Lakes Harris-Little Harris and Yale. Their discussion of phosphorus refers to "total phosphate," so I converted their numbers to TP by multiplying by the ratio of the molecular weights of phosphorus to phosphate. Their estimates of loading exceed mine by about three times for TP and two times for TN. A major reason for the discrepancies is because the ECFRPC study used considerably larger drainage basins, with no trapping of nutrient runoff within the watershed. Their estimated drainage basin areas ranged from 1.6 to 4.9 times those used in the present study. Also, TP concentrations used by ECFRPC for stormwater runoff

(0.65 mg/L) are higher than that used for all land uses in the present study, except for confined feedlots (Table 3).

Other sources of higher loading estimates in the ECFRPC study included nutrient concentrations in rainwater that were higher than those used in the present study, although they are in the range reported for previous bulk precipitation collections in Florida (Irwin and Kirkland 1980); tributary flow rates that were higher than the averages reported during the period of my study; incorporation of estimates of nutrient loading from seepage (assumed to be 10% of rainfall in their drainage basins); and discharges from several waste treatment plants that no longer have surface discharge.

National Eutrophication Survey (EPA, 1975). As part of the NES, EPA developed a nutrient budget for Lake Griffin, using data collected in 1973–74. Estimates of average TP loading reported by EPA (30,130 kg/yr) are within the range found in the present study, but average TN loading reported by EPA (853,815 kg/yr) approximates the maximum found in the present study (Figure 20). EPA reported higher nutrient loadings in flows from Haines Creek (66.5% of TP loading, 81% of TN loading) than were found in the present study. The higher estimate of nutrient loading from Haines Creek reported by EPA is due to the use of slightly higher nutrient concentrations for Haines Creek water and average flow volumes that were more than twice those reported during the present study.

EPA estimated nutrient loading to Lake Griffin from nonpoint runoff from "minor tributaries and immediate drainage," using a drainage area that was more than three times that used in the present study. The discrepancy in drainage area appears to be partially because EPA included Lake Yale and its watershed in the Lake Griffin drainage area. The estimates of nonpoint runoff include the area of the muck farms, although no direct measurements of muck farm discharge were made. Despite the larger drainage area, the EPA estimates of nonpoint nutrient loading were lower than corresponding estimates from the present study (their estimated average nonpoint TP and TN loadings were only 16% and 90%, respectively, of the average obtained during the present study). This discrepancy may reflect the high areal nutrient loading rates from the muck farms. EPA estimates of nutrient loading from precipitation and septic tanks were lower than those determined in the present study, but in both studies they were minor contributions to the total nutrient loading. Discharges from a waste treatment plant that no longer has surface water discharges accounted for 12.9% of TP loading and 2.1% of TN loading in the EPA nutrient budgets.

EPA reported a much lower retention of TP in Lake Griffin (905 kg/yr) than estimated in the present study (Table 21) and a substantial net export of TN (60,010 kg/yr). Low retention or net export of nutrients seems unlikely and may be due to overestimates of nutrient concentrations and volumes of outflow.

Lake County Environmental Services (Wicks 1983). Another nutrient budget for Lake Griffin was constructed by LCES for the water year 1981. Wicks' estimated total nutrient loading for water year 1981 was 47% of the average TP loading and 21% of the average TN loading determined during the present study. Volume discharges and nutrient loading estimated by Wicks from Lake Eustis through Haines Creek approximated that calculated for 1985, the lowest flow year during the present study (Figure 7[D]). Estimates by Wicks of nutrient loading from muck farms were lower than in the present study, primarily because of lower assumed discharge volumes. Wicks did not estimate nutrient loading that resulted from stormwater runoff or septic tank effluents, but did estimate loading due to ground water seepage. It was unclear how seepage volume estimates were calculated in the Wicks study, but it was concluded that ground water seepage accounted for 7% of water inflows, 27% of TP loading, and 8.5% of TN loading.

Crisman and others (1992). Crisman et al. developed a partial nutrient budget for Lake Weir for 1980 and 1985. The nutrient loading estimates were limited to stormwater runoff and septic tank inputs and did not include atmospheric deposition. Ignoring atmospheric deposition, estimated TP loading to the lake in 1980 was similar in both studies, but the Crisman et al. estimate for 1985 was 60% higher than that in the present study. The Crisman et al. estimates of TN loading were about twice those in the present study for both 1980 and 1985.

The Crisman et al. estimates of nutrient loading from pasture runoff range from 3 to 20 times higher than those in the present study. Their estimate of TN loading from citrus in 1980 is about 10 times higher than estimated TN loading from citrus in the present study, but TN loading estimates for 1985 and TP loading estimates for both years are similar for the two studies. Their estimates of nutrient loading from septic tanks are about twice those in the present study. Conversely, my estimates of nutrient loading from residential runoff are 3 to 6 times those of Crisman et al.

Differences between nutrient loading estimates of Crisman et al. and the present study are primarily due to differences in assumed drainage areas. Crisman et al. used a drainage area of 4,500 ha, about 50% larger than that used in the present study. The drainage basin land use maps used by Crisman et al. assign considerably higher areas to citrus (1980) and pasture (both years) than do those maps used in the present study. For example, Crisman et al. estimated 3,060 ha of pasture in the watershed in 1985, whereas the land use maps used in the present study estimated 374 ha (1984) and 202 ha (1989). Crisman et al. did not measure pasture area, but rather assumed that all areas not identified as residential, citrus, forest, or wetlands were pasture. This assumption partially accounts for the large discrepancy in areas assigned to pasture by Crisman et al. Conversely, the Crisman et al. drainage basin land use maps assign lower areas to residential housing than do those maps used in the present study.

Differences in the choice of export coefficients also contribute to discrepancies in nutrient loading estimates. The nutrient export coefficients used by Crisman et al. are derived from studies conducted prior to 1975, apparently from areas outside of Florida. The present study, however, incorporated data from a number of more recent studies, primarily conducted in Florida. The TN export coefficient for citrus used by Crisman et al. is more than three times the average of studies I used for estimating TN loading (Harper 1992); this high export coefficient contributes to the high estimate of TN loading in 1980 by Crisman et al. Another questionable export coefficient used by Crisman et al. is for TP export from forests, which is nearly six times the average of studies I used for estimating TP loading (Harper 1992).

Estimates of septic tank nutrient loading to Lake Weir by Crisman et al. assume that all septic tanks in the drainage basin contribute to loading, with 25% of TN and 10% of TP septic tank inputs being transported to the lake. These are questionable assumptions, particularly for phosphorus because of its limited mobility in soils (Jones and Lee 1977; Sherwood and Crites 1984; Canter and Knox 1985).

Camp, Dresser & McKee (1991). CDM estimated average annual nonpoint nutrient loads for the lakes in Lake County. Total nutrient loadings were not separated by source, but include stormwater runoff from all land uses and septic tank inputs. There were again discrepancies between drainage basin areas used by CDM and by the present study. CDM defined the drainage basin for Lake Beauclair as about 40% of the size used in the present study; for the other lakes, CDM defined drainage basins up to 75% larger than those used in the present study. Nutrient concentrations by CDM for stormwater runoff were similar to those used in the present study, but it is not clear how runoff volumes were estimated. Differences in nonpoint nutrient loading estimates between CDM and the present study were generally proportional to the differences in assumed drainage basin areas.

Deviations from proportionality to drainage areas between the CDM and the present study probably are due to differences in land use delineations and associated runoff coefficients and to underestimation of nutrient runoff from some sources by CDM (e.g., muck farms in the Lake Griffin subbasin and animal feedlots in the Lake Eustis subbasin).

ASSESSMENT OF POTENTIAL STRATEGIES FOR IMPROVING WATER QUALITY

RESTORATION AND MANAGEMENT ACTIONS EVALUATED

The lake trophic state models were used to assess the effects of potential restoration and management actions on phosphorus loading and equilibrium phosphorus concentrations in the UORB lakes. These restoration and management strategies would also reduce nitrogen levels, but the analysis focused on phosphorus because (1) TN:TP ratios for the lakes and experimental studies of nutrient limitation in Lake Apopka (Schelske et al. 1992; Aldridge et al. 1993) suggest that phosphorus is the primary nutrient contributing to eutrophication in the UORB lakes and (2) the trophic state models were generally more accurate in predicting in-lake phosphorus concentrations than in predicting in-lake nitrogen concentrations (Table 22).

The restoration and management strategies that were considered included

- Restoration of all muck farms in the UORB to wetlands
- Reduction of nutrient loading from within the UORB to levels expected under "predevelopment" conditions, but with no reduction in loading from upstream tributaries
- Reducing the concentration of phosphorus in flows entering the UORB through the Apopka-Beauclair Canal
- A combination of muck farm restoration and reduction in Apopka-Beauclair Canal TP concentrations
- A combination of "predevelopment" nutrient loading from the UORB and reduction in Apopka-Beauclair Canal TP concentrations

This scenario represents the best estimate of levels of nutrient loading and trophic state in the lakes prior to the onset of cultural eutrophication. This state may not be attainable, but it represents the ultimate target if the goal is to truly restore the natural system. As such, it provides an index to judge how effective lesser reductions in nutrient loading or other restoration actions will be in restoring historic water quality.

- Development of a marsh flow-way project on restored muck farmlands in the Lake Griffin subbasin

ASSESSMENT METHODS

The effects of these potential strategies for water quality improvement were evaluated by repeating the trophic state modeling with the phosphorus loadings adjusted to reflect the effects of the restoration and management actions. Nutrient concentrations in tributary flows between lakes within the basin also had to be adjusted. In the modeling of the existing system, nutrient concentrations in tributary flows were developed from analysis of monitoring data for the tributaries or lakes. For the modeling of the restoration alternatives, nutrient concentrations used for tributary flows were the concentrations predicted for the upstream lake by the trophic state models. The trophic state model used for each lake was that which showed the closest match between model predictions for existing phosphorus loading estimates and the reported TP concentrations. Models used were those of Larsen and Mercier (Equation 29) for Lake Dora, Canfield and Bachmann (Equation 28) for Lake Griffin, and Reckhow (Equation 24) for the other lakes (Table 22).

For most of the restoration alternatives, the models were run with the modified nutrient loadings for each year of the study period, (1980–90 for Lakes Yale and Weir, 1984–90 for the other lakes) and the data reported are the average annual loadings and in-lake nutrient concentrations over that period. The procedure for assessment of the effects of a Lake Griffin flow-way was somewhat different. In this case, average conditions for 1984–90 of TP loadings and lake morphology and hydrology were determined first, and then responses to various TP removal rates by the flow-way were modeled.

Most of the muck farms in the UORB have already been acquired, with the intention of wetland restoration (Table 2), and acquisition of the remaining properties is under consideration. For modeling of the effects of muck farm restoration, the estimated TP loadings from muck farm discharges were eliminated and replaced with expected nutrient runoff from wetlands (Table 3).

For modeling of predevelopment nutrient loading conditions, septic tank and citrus plant discharges were eliminated. Stormwater runoff under predevelopment conditions was estimated by converting all land uses in the basin to natural areas; muck farms were converted to open water/wetlands; and other land uses were converted to forest/rangeland. Nutrient concentrations and runoff coefficients for stormwater runoff from the converted land uses were taken from Table 3. No adjustments were made in atmospheric deposition or nutrient loading from tributaries upstream of the UORB (Apopka-Beauclair Canal and Palatlakaha River).

For modeling of reductions in TP concentrations in Apopka-Beauclair Canal inflows, two concentrations were used, 0.1 mg TP/L and 0.05 mg TP/L, which represent expected equilibrium concentrations in Lake Apopka if PLRGs for that basin are partially or fully achieved (M. Coveney, SJRWMD, pers. com. 1994).

The combined restoration scenarios modeled included reduction of TP concentrations in Apopka-Beauclair Canal inflows to 0.05 mg TP/L and either muck farm restoration or predevelopment TP loading conditions within the UORB.

A marsh flow-way project on restored muck farmlands in the Lake Griffin subbasin is presently in a pilot stage. The concept is based on a demonstration marsh flow-way in operation in the

Lake Apopka Basin which uses wetland filtration to remove particulate nutrients from lake water (Lowe et al. 1992). The pilot Lake Griffin flow-way uses a portion of the former S.N. Knight, Lisbon muck farm. The flow-way is operated at approximately lake level, with about 1,500 acres of wetlands and open water. Water from Lake Griffin enters the flow-way through culverts and is pumped back into the lake after passing through the wetland system. Later the flow-way may be expanded to include the former Lowrie Brown South muck farm, which would add about another 500 acres.

The baseline condition for modeling the effects of a Lake Griffin flow-way was average TP loading and predicted in-lake TP concentrations under the scenario of restoration of muck farms in the UORB to wetlands. The effects of the flow-way were modeled by reducing net TP loading to the lake from this baseline by the amount expected to be retained by the flow-way under various operating conditions. Average flow rates through the flow-way modeled included 40 cfs, 80 cfs (similar to flow rates for the pilot project), 120 cfs, 250 cfs, and 500 cfs (potential options for a full-scale project). For each flow rate, two efficiencies of TP retention were modeled: a conservative removal efficiency of 30% (Lowe et al. 1992) and a retention efficiency determined from a quadratic regression equation (Equation 38) developed based on literature studies relating phosphorus retention efficiencies of wetlands to TP loading rates (Figure 26).

$$P_{eff} = 64.909 - (22.795 \times \log_{10} P_A) - [2.7846 \times (\log_{10} P_A)^2] \qquad (38)$$

$r^2=0.71$

where:

P_{eff} = phosphorus retention efficiency (%)
P_A = areal phosphorus loading ($g/m^2/yr$)

This regression equation was developed from studies reported in Nichols (1983) and Davis et al. (1985). Factors other than TP loading rates ($g/m^2/yr$) that may be expected to affect nutrient

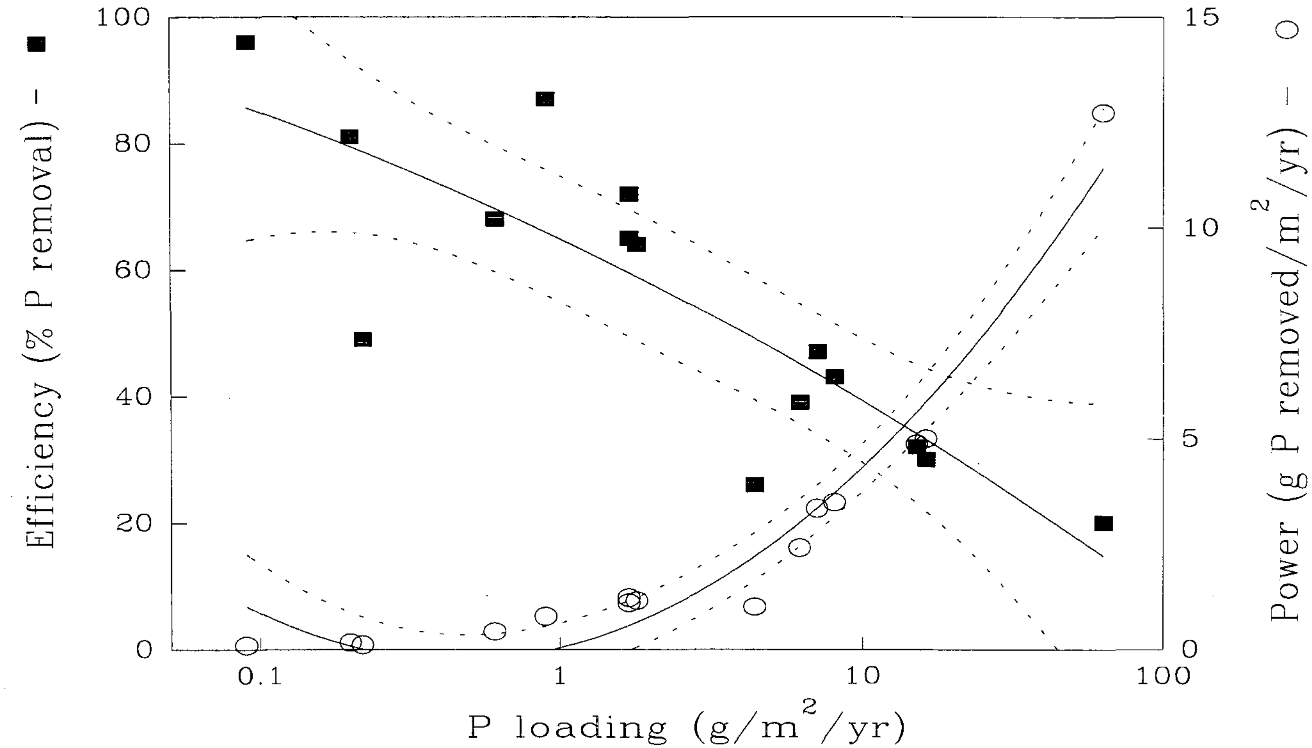

Figure 26. Phosphorus removal by wetland systems. *Efficiency and power of phosphorus removal by wetland systems as functions of phosphorus loading, with fitted quadratic regressions and 95% confidence intervals* (combined data of Nichols 1983 and Davis et al. 1985).

retention (such as nutrient concentrations or current velocities) are not considered in this regression analysis. A variety of wetland types are included in the regression. Many are constructed wetlands used for treatment of wastewater, which may be expected to have substantially higher TP concentrations than Lake Griffin water entering the flow-way. However, seven of the sites included are south Florida marshes that receive primarily agricultural runoff, most of which had influent nutrient concentrations similar to that found in Lake Griffin.

For modeling of the effects of a Lake Griffin flow-way, I assumed that only the Knight Farm property was used for nutrient retention. Addition of the Lowrie Brown property to the treatment area would slightly increase the retention efficiencies predicted by the regression equation. The application of the regression equation to a Lake Griffin flow-way assumes that nutrient retention efficiency will be similar for all inundated areas, regardless of water depth. Substantial areas of the Knight Farm flow-way will be open water areas several feet deep; whether or not these deep areas will have areal nutrient retention similar to shallower vegetated areas is unknown.

PREDICTED RESPONSES TO RESTORATION AND MANAGEMENT ACTIONS

Reductions in TP concentrations of Apopka-Beauclair Canal outflows will have the most significant effects on water quality in Lake Beauclair, reducing predicted TP loading and equilibrium in-lake concentrations by 50–70% (Figure 27). Restoration of muck farms or attainment of predevelopment TP loading in the UORB would only reduce TP loading and in-lake concentrations for Lake Beauclair by 5–10%. Implementing these actions in conjunction with reducing TP concentrations in Apopka-Beauclair Canal outflows reduced predicted TP levels by 70–80% compared to current conditions.

For Lake Dora, reducing TP concentrations in Apopka-Beauclair Canal outflows again is expected to have the most significant effects on water quality, reducing predicted TP loading and equilibrium in-lake TP concentrations by up to 50% (Figure 28). Muck farm restoration would have negligible effects on Lake Dora TP levels, whereas attainment of predevelopment TP loading in the UORB would reduce TP loading and equilibrium in-lake TP concentrations by about 15%. A combination of reducing TP concentrations in Apopka-Beauclair Canal outflows and attainment of predevelopment TP loading would reduce predicted TP loading and equilibrium Lake Dora TP concentrations more than 80%.

Under the present assumption of no significant flow reversals in the Dead River, water quality in Lake Harris-Little Lake Harris will be unaffected by reducing TP concentrations in Apopka-Beauclair Canal outflows. Restoration of muck farms in the UORB is expected to reduce Lake Harris-Little Lake Harris TP loading and equilibrium in-lake TP concentrations by about 10% (Figure 29). Attainment of predevelopment TP loading would reduce predicted TP loading by nearly 50% and equilibrium in-lake TP concentrations by about 35%.

For Lake Eustis, attainment of predevelopment TP loading in the UORB would be expected to have the most significant effects on water quality, reducing predicted TP loading and equilibrium in-lake TP concentrations by more than 60% and 50%, respectively (Figure 30). Muck farm restoration or reducing TP concentrations in Apopka-Beauclair Canal outflows would reduce both predicted TP loading and equilibrium in-lake TP concentrations by about 20%. A combination of attainment of predevelopment TP loading and reduction of TP concentrations in Apopka-Beauclair Canal outflows would reduce predicted TP loading and equilibrium Lake Eustis TP concentrations more than 65%. Combining muck farm restoration with reduced Apopka-Beauclair Canal TP concentrations would reduce equilibrium in-lake TP levels by about 30%.

Muck farm restoration would reduce predicted TP loading and equilibrium phosphorus levels in Lake Griffin by more than 60% and 50%, respectively, whereas attainment of predevelopment TP loading in the UORB would reduce TP loading and in-lake concentrations by about 80% and 70%, respectively (Figure 31). Reducing TP concentrations in Apopka-Beauclair Canal outflows, either alone or in conjunction with the other restoration actions, would have negligible effects on Lake Griffin water quality.

For Lake Yale, attainment of predevelopment TP loading in the UORB would reduce predicted TP loading and in-lake concentrations by about 65% and 40%, respectively (Figure 32).

For Lake Weir, attainment of predevelopment TP loading in the UORB would reduce predicted TP loading and in-lake concentrations by about 50% and 36%, respectively (Figure 33).

The quadratic regression equation developed from studies of wetland phosphorus retention efficiencies predicts that retention efficiencies by a Lake Griffin marsh flow-way would range from 77% at a flow rate of 40 cfs to 58% at the highest flow rate of 500 cfs. As mentioned previously, muck farm restoration in the UORB would reduce predicted phosphorus loading and equilibrium TP levels in Lake Griffin by more than 50% (Figure 31). Substantial flow rates and retention efficiencies by a Lake Griffin marsh flow-way would be required to further reduce phosphorus levels significantly. At the conservative 30% retention efficiency, an average flow rate of more than 120 cfs would be required to reduce equilibrium phosphorus levels by 10% (Figure 34). To achieve about a 30% reduction in net phosphorus loading and in-lake TP concentrations would require a flow rate of 250 cfs at retention efficiencies predicted by the regression equation, or a flow rate of 500 cfs at a 30% retention efficiency. Flow rates of 500 cfs would be required to achieve net TP loading and equilibrium in-lake TP concentrations similar to that predicted if external loading was reduced to predevelopment levels (Figures 31 and 34).

Although these results indicate that the flow rates presently being implemented for the pilot marsh flow-way project (40–80 cfs) are not likely to substantially reduce equilibrium TP concentrations in the lake, it is possible that they may accelerate the rate of

Key to abbreviations used on Figures 27–33

O	- observed TP (total phosphorus) concentration
C	- predicted under current TP loading estimates
MR	- muck farm restoration
PD	- "predevelopment" loading
A1	- Apopka-Beauclair Canal TP = 0.1 mg/L
A2	- Apopka-Beauclair Canal TP = 0.05 mg/L
A2MR	- combined muck farm restoration and Apopka-Beauclair Canal TP = 0.05 mg/L
A2PD	- combined "predevelopment" and Apopka-Beauclair Canal TP = 0.05 mg/L

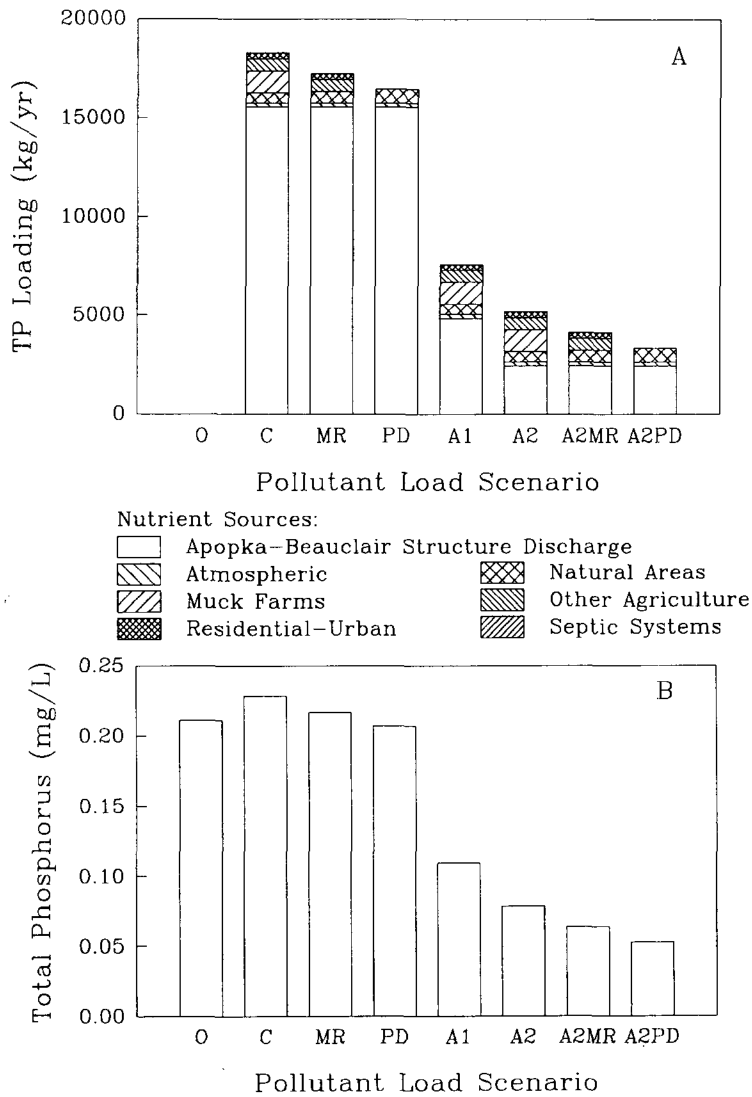

Figure 27. Lake Beauclair restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations; mean values, 1984–90 (see page 124 for key to abbreviations)

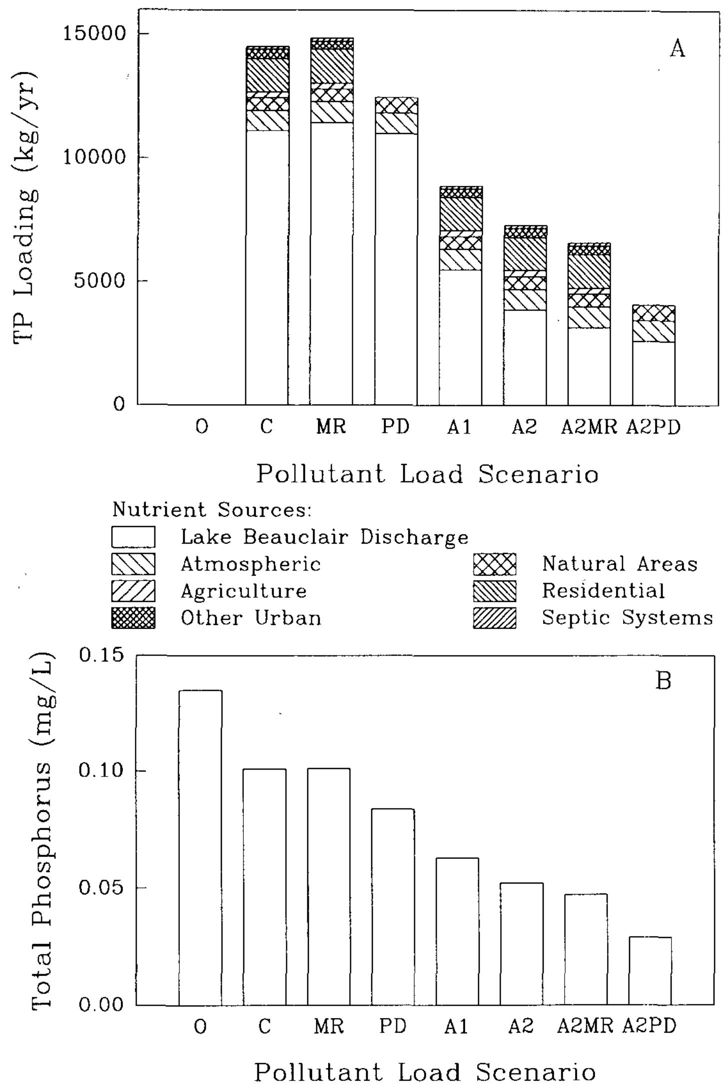

Figure 28. Lake Dora restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations (see page 124 for key to abbreviations)

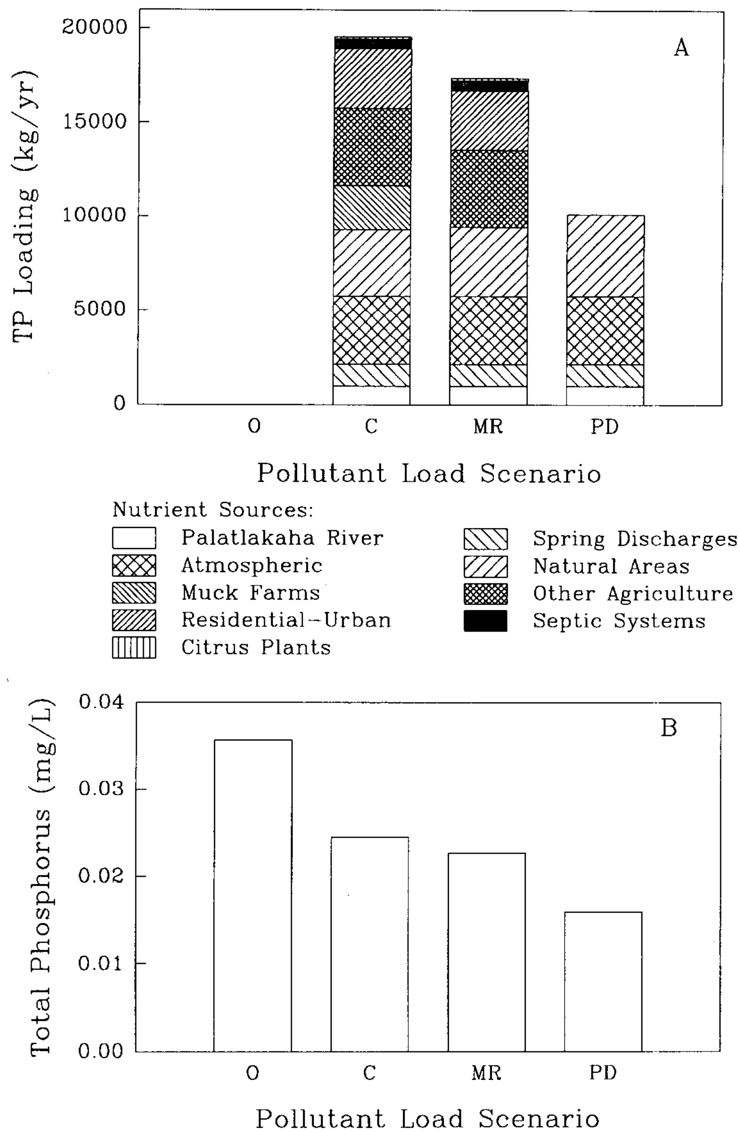

Figure 29. Lake Harris-Little Lake Harris restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations (see page 124 for key to abbreviations)

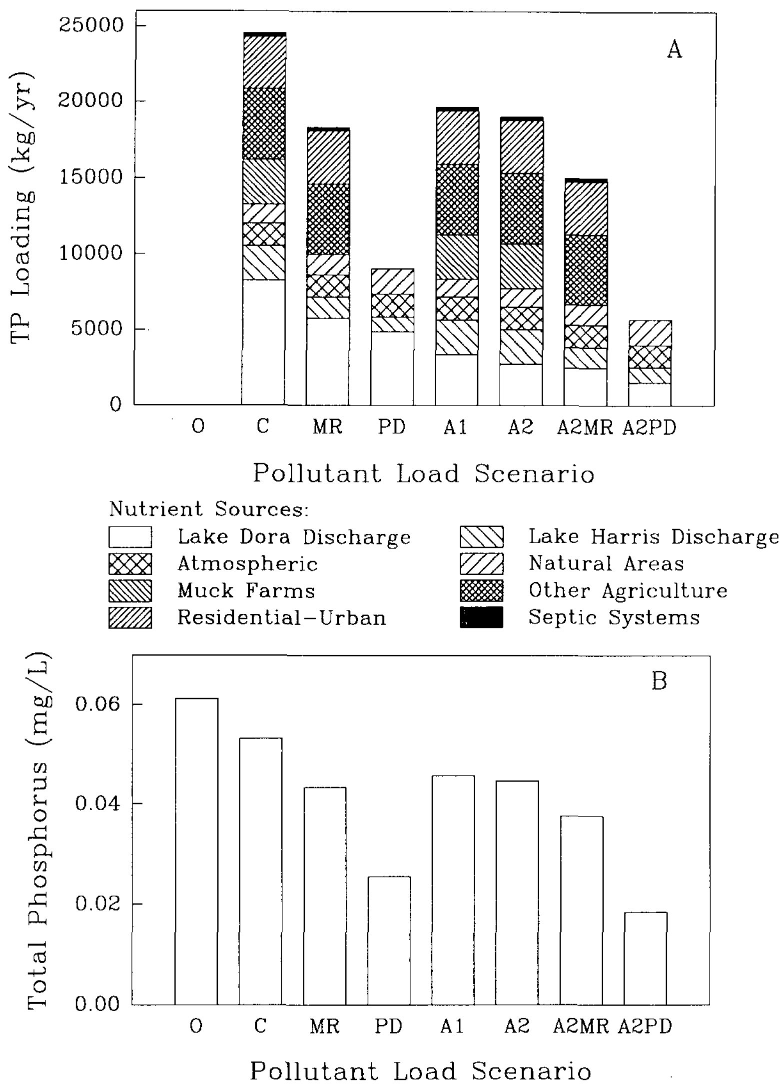

Figure 30. Lake Eustis restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations (see page 124 for key to abbreviations)

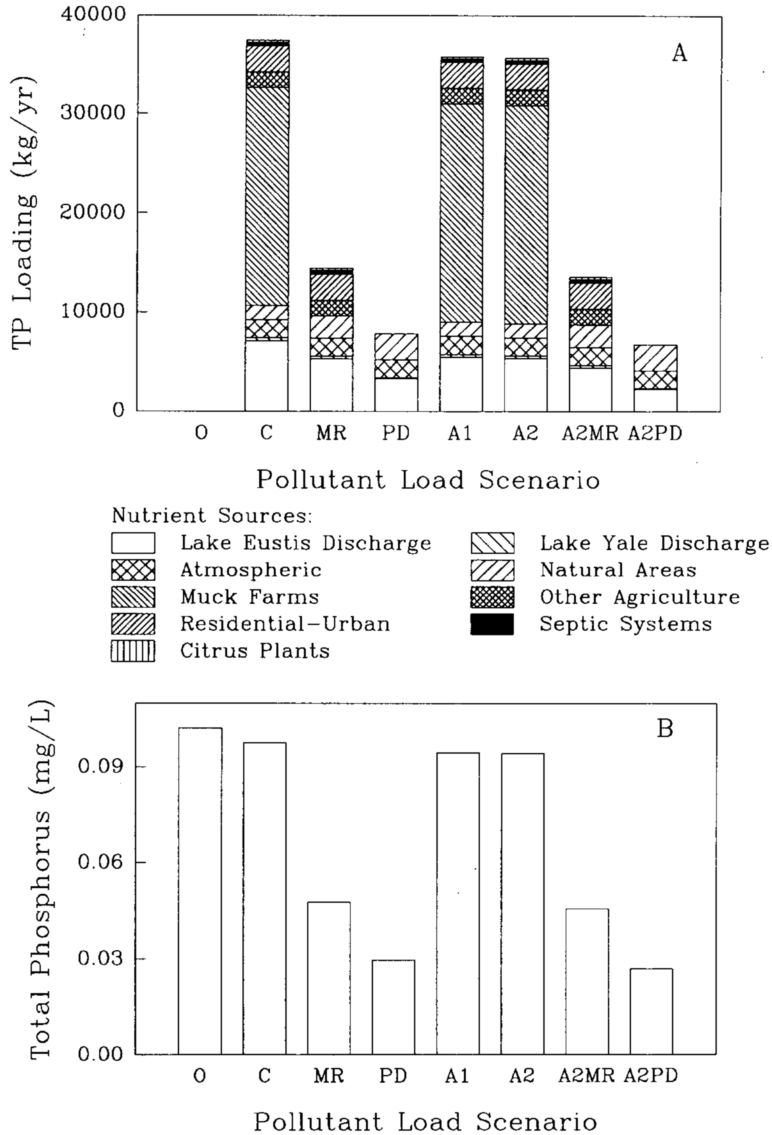

Figure 31. Lake Griffin restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations (see page 124 for key to abbreviations)

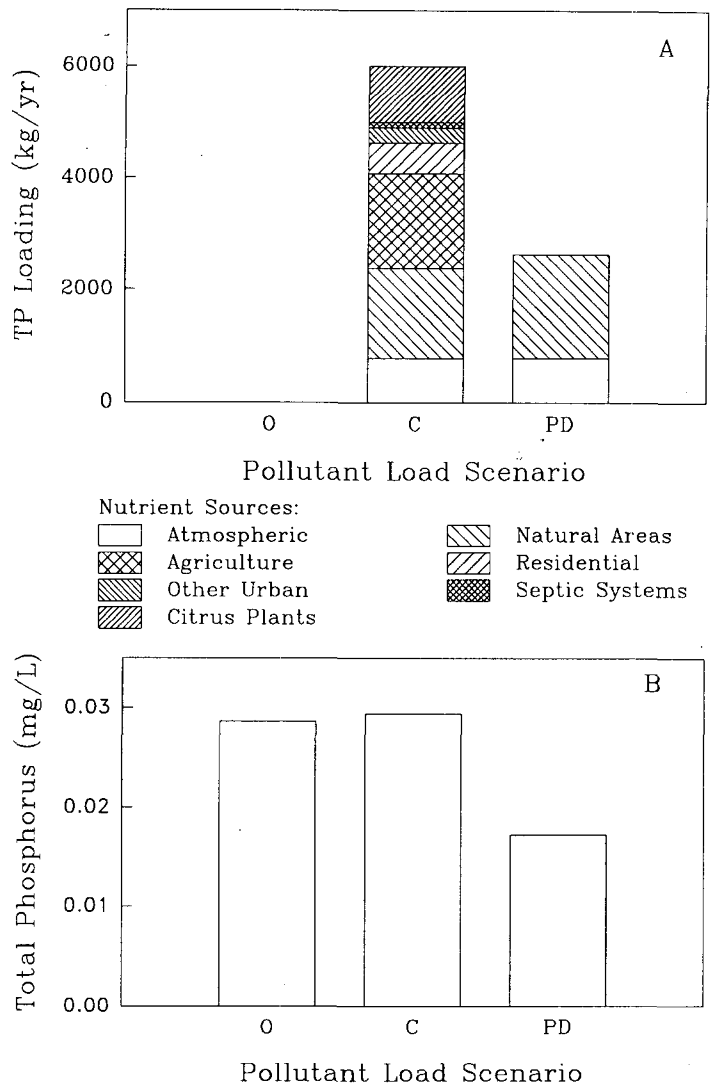

Figure 32. Lake Yale restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations (see page 124 for key to abbreviations)

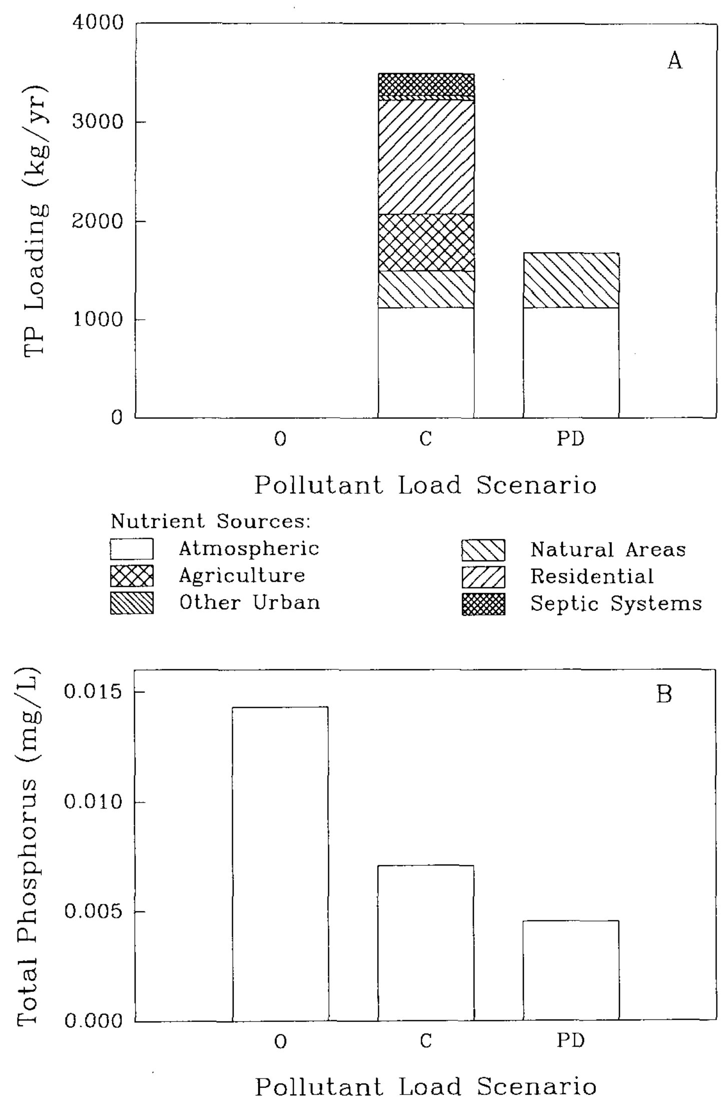

Figure 33. Lake Weir restoration scenarios: (A) estimated phosphorus loading and (B) equilibrium in-lake phosphorus concentrations (see page 124 for key to abbreviations)

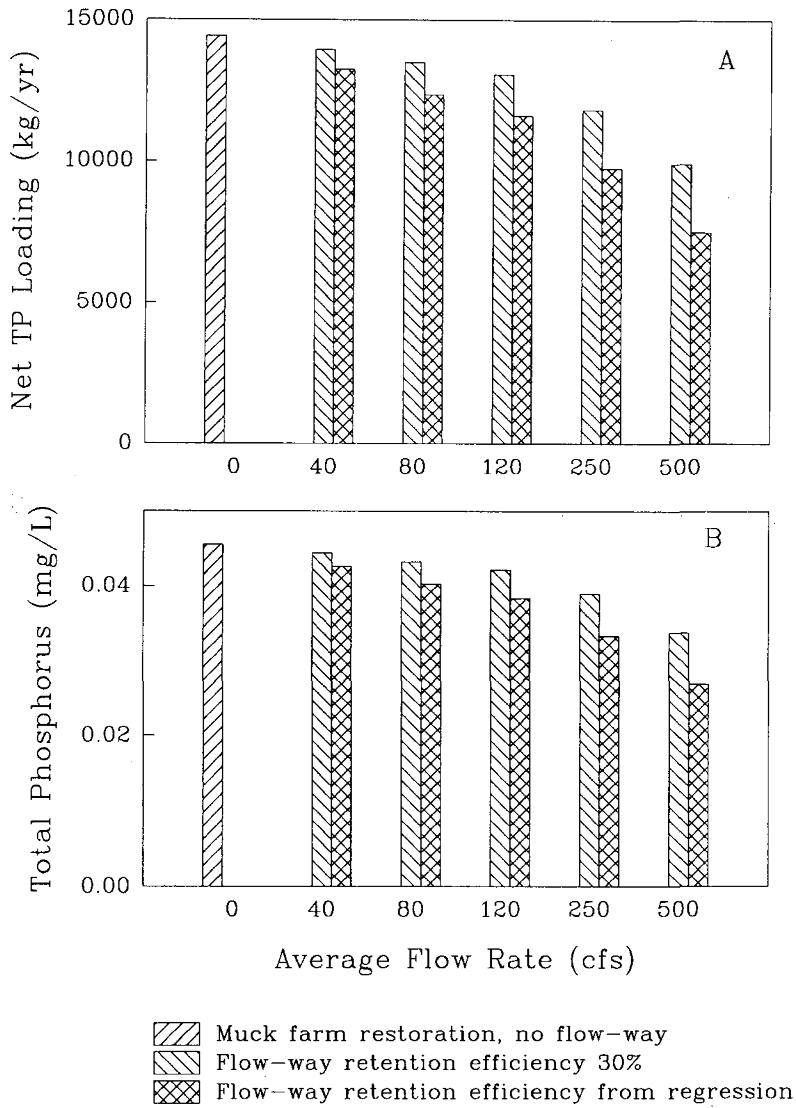

Figure 34. Predicted effects of a marsh flow-way project on (A) estimated net phosphorus loading and (B) equilibrium in-lake phosphorus concentrations for Lake Griffin

recovery from present conditions. For a preliminary assessment of the time frames for attainment of new equilibrium TP concentrations, I used the time-dependent formulation of the Vollenweider (1969) model (Reckhow and Chapra 1983) (as noted previously, the trophic state models used in this report are based on the Vollenweider model):

$$TP_t=\frac{L}{Q_o+k_P V}\{1-\exp[-(\frac{Q_o}{V}+k_P)t]\}+TP_i\exp[-(\frac{Q_o}{V}+k_P)t] \qquad (39)$$

where:

TP_t = TP concentration at time t
L = total annual mass loading of TP to lake
Q_o = annual water losses from lake
k_P = phosphorus net sedimentation coefficient
V = lake volume
TP_i = initial (or previous year's) TP concentration

The model was run for four restoration options for Lake Griffin: muck farm restoration in the UORB, reduction of phosphorus loading within the UORB to predevelopment levels, operation of the Lake Griffin marsh flow-way at an average flow rate of 80 cfs and a 30% retention efficiency, and operation of the Lake Griffin marsh flow-way at an average flow rate of 500 cfs and a 30% retention efficiency.

The time course of lake response to changes in nutrient loading will depend strongly on the net sedimentation coefficient. Over a short term, the probable response to a reduction in TP loading would be a decrease in the sedimentation coefficient caused by a decline in the rate of sedimenting TP with no similar decline in the rate of TP release from the sediments (internal loading) (Lijklema et al. 1986; Lowe et al. 1992). The sedimentation coefficient could even become negative if internal loading exceeds sedimentation rates (Sas 1989). Over time, the sedimentation coefficient would be expected to increase to a new equilibrium as available sedimentary phosphorus becomes depleted. I ran the time-dependent model using two assumed sedimentation coefficients: zero (perhaps more representative of initial conditions) and the equilibrium value predicted by the Canfield and Bachmann model (Equation 28). The predicted equilibrium sedimentation coefficients are muck farm restoration (1.66/yr), predevelopment loading (1.25/yr), operation of the Lake Griffin marsh flow-way at 80 cfs and 30% retention efficiency (1.61/yr), and operation of the Lake Griffin marsh flow-way at 500 cfs and 30% retention efficiency (1.40/yr).

Under the assumption of equilibrium sedimentation coefficients, equilibrium TP levels are reached rapidly, essentially within 1 year of initiation (Figure 35[A]). Operation of the flow-way at 80 cfs had relatively little additional effect on the rate of recovery or equilibrium TP levels beyond that predicted for muck farm restoration. However, operation of the flow-way at a flow rate of 500 cfs and attaining predevelopment loading rates resulted in rapid attainment of a lower equilibrium in-lake TP concentration. Operation of a flow-way at 500 cfs initially had the most rapid decline in TP because high rates of TP retention in the marsh flow-way resulted in the lowest net loading to the lake. However, as lake TP concentrations decline, rates of removal by the flow-way decrease (this is simply because of lower loading rates to the flow-way, as retention efficiency is held constant), resulting in equilibrium net TP loading and in-lake TP concentrations being lowest for the predevelopment loading condition.

Under the assumption of a sedimentation coefficient of zero, equilibrium in-lake TP concentrations are higher and time required to reach equilibrium is somewhat lengthened (Figure 35[B]). Muck farm restoration results in only a 3% decrease in equilibrium TP concentrations from predicted levels under current conditions, whereas TP levels decline 15% with operation of the flow-way at 80 cfs. Operation of the flow-way at a flow rate of 500 cfs and attaining predevelopment loading rates resulted in a 48% decrease in equilibrium in-lake TP concentration. Again, the initial rates of decrease were greatest with the operation of a flow-way at 500 cfs because high rates of

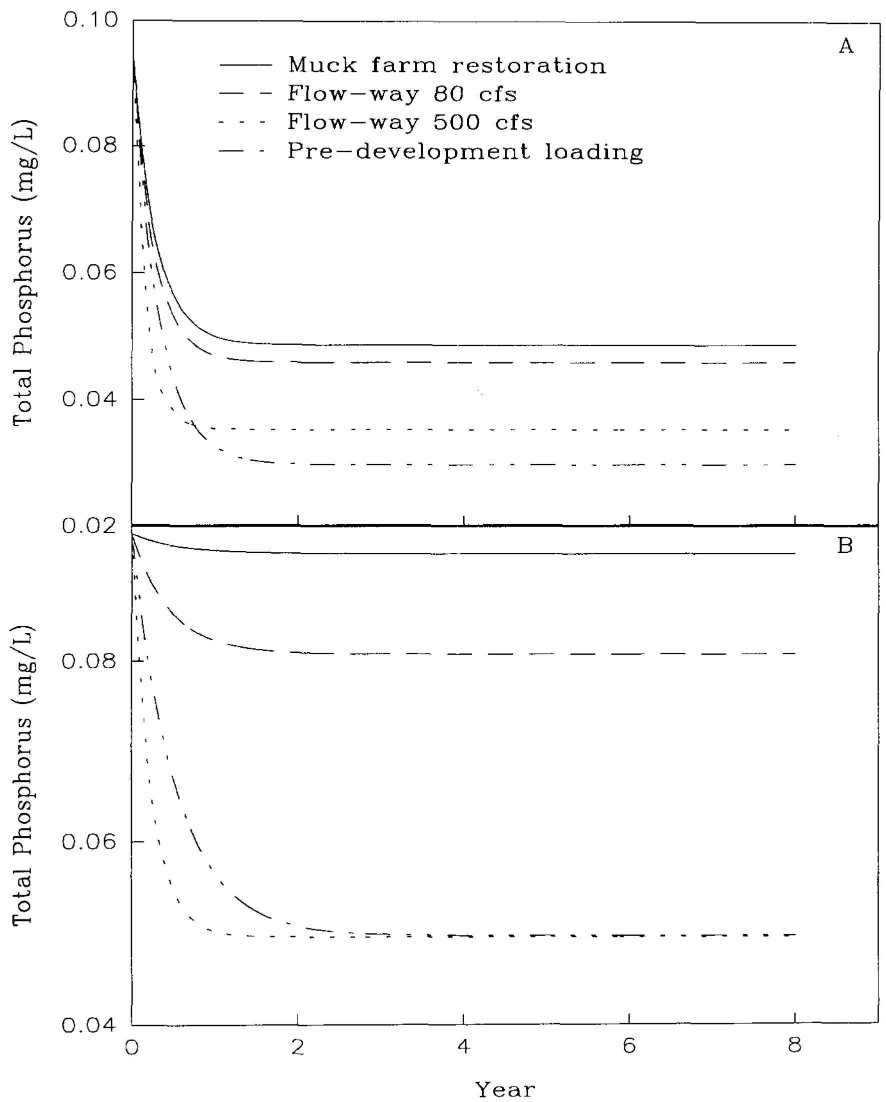

Figure 35. Predicted temporal changes in Lake Griffin phosphorus concentrations under various restoration scenarios. *Sedimentation coefficient for (A) assumed to be equilibrium values predicted by Canfield and Bachmann (1981) (Equation 28); sedimentation coefficient for (B) assumed to be zero.*

TP retention in the marsh flow-way resulted in the lowest net loading to the lake, but marsh retention declines as in-lake TP concentration decreases.

Figure 35 probably represents boundary conditions for the expected response of Lake Griffin to the restoration options. Initially, the net sedimentation coefficient is likely to be low, so the lake response may resemble the changes shown in Figure 35(B). Over time, the sedimentation coefficient may be expected to increase to the equilibrium value used in Figure 35(A), but the time course of this change is unknown. In a number of European lakes, recovery of sedimentation coefficients usually occurred in 1–5 years, although recovery times were longer in lake sediments with high phosphorus content (Sas 1989).

REDUCING INTERNAL NUTRIENT LOADING

Eutrophic lakes often have large accumulations of nutrient-rich organic sediments. Internal nutrient loading from these sedimentary nutrient stores can prevent or delay for many years the recovery of lake water quality after reduction in external nutrient loading (Marsden 1989; Sas 1989).

As is evident from the time-course simulations for Lake Griffin, the equilibrium in-lake nutrient concentrations and the time frames for attainment of equilibrium depend critically on the internal nutrient loading. For example, if internal loading is sufficiently high to reduce net TP sedimentation rates to near zero, then muck farm restoration would have negligible effects on in-lake TP concentrations (Figure 35[B]).

Lakes in the UORB have large volumes of flocculent organic sediments (Danek et al. 1991). These organic sediments are not only a source of internal nutrient loading, but they also limit habitat for reproduction by fish. One potential strategy to accelerate the recovery of lake water quality after reduction in external phosphorus loading would be to limit internal loading by removal or inactivation of sedimentary stores of phosphorus.

Sediment removal can be a very effective lake restoration technique (Cooke et al. 1986; Pollman et al. 1988). However, high cost or disposal problems can limit the feasibility of dredging in many circumstances. Preliminary cost estimates were prepared for the removal of organic sediments from lakes in the UORB. Cost estimates were based on measurements of sediment volume for the lakes (Danek et al. 1991) and unit cost estimates for dredging sediments. I used two cost estimates, one taken from Pollman et al. (1988) and the other from a presentation at SJRWMD given by Larry Madrid of Bromwell & Carrier on January 13, 1992. Pollman et al. estimated dredging costs for Lake Apopka to be \$2.80/m^3 (or \$0.0792/ft^3). The Bromwell & Carrier costs for dredging Banana Lake were reported at about \$1.00/yd^3 (or \$0.0370/ft^3).

These two unit cost estimates are used as upper and lower bounds for dredging cost estimates. It should be noted that neither of these estimates includes costs for land disposal of the sediments. As shown in Table 24, cost estimates range from

Table 24. Preliminary cost estimates for dredging removal of organic sediment accumulations in the lakes of the Upper Ocklawaha River Basin

Lake	Sediment Volume ($ft^3 \times 10^9$)	Upper Bound Dredging Cost	Lower Bound Dredging Cost
Beauclair	0.1836	$14,542,000	$6,800,000
Dora	0.7598	$60,179,000	$28,141,000
Eustis	2.0967	$166,065,000	$77,656,000
Griffin	2.2521	$178,373,000	$83,411,000
Harris-Little Harris	6.7972	$538,359,000	$251,748,000
Yale	0.9632	$76,288,000	$35,674,000
Weir	0.8491	$67,251,000	$31,448,000

Source: Danek et al. 1991; Pollman et al. 1988; L. Madrid, Bromwell & Carrier, pers. com. 1992

$6.8 million for Lake Beauclair to $538 million for Lake Harris-Little Lake Harris. Although these costs seem prohibitive, concentrations of exchangeable phosphorus are likely to decline below the sediment-water interface, so it may be necessary to remove only the surface sediments to significantly reduce internal loading. One major uncertainty in dredging is the potential redistribution of organic sediments within the lake during the project, which may negate much of the benefit of dredging (Pollman et al. 1988). Studies of the distribution of exchangeable phosphorus in lake sediments are necessary before determination of the feasibility and cost-effectiveness of sediment removal can be made.

Inactivation of sedimentary exchangeable phosphorus by treatment with aluminum salts (alum) can be another effective means to reduce internal nutrient loading (Cooke et al. 1986; OCEPD 1987). Preliminary costs were estimated for alum treatment of the lakes in the UORB using an average cost of $300/acre reported by Orange County Environmental Protection Department (1987). The estimated costs of alum application would be substantial, ranging from $243,000 for treatment of only the area with organic sediment accumulations in Lake Beauclair to $5.6 million for treatment of all of Lake Harris-Little Lake Harris (Table 25). Most applications of alum have been in lakes considerably smaller than those in the UORB (Cooke et al. 1986; OCEPD 1987). It is uncertain how effective alum application would be in large lakes with deep accumulations of readily resuspended organic sediments. A potential problem resulting from alum treatment is toxicity from aluminum and lowered pH (Cooke et al. 1986; OCEPD 1987).

Table 25. Preliminary cost estimates for alum treatment of lake sediments in the lakes of the upper Ocklawaha River Basin

Lake	Lake Surface Area (acres)	Lake Treatment Cost	Sediment Surface Area (acres)	Sediment Treatment Cost
Beauclair	1,085	$325,000	810	$243,000
Dora	4,384	$1,315,000	3,726	$1,118,000
Harris-Little Harris	18,688	$5,606,000	18,095	$5,428,000
Eustis	7,756	$2,327,000	7,216	$2,165,000
Griffin	9,412	$2,824,000	9,000	$2,701,000
Yale	4,020	$1,206,000	3,575	$1,072,000
Weir	5,623	$1,687,000	4,124	$1,237,000

Source: Danek et al. 1991; OCEPD 1987

RECOMMENDATIONS

RECOMMENDATIONS FOR FURTHER STUDY

Further study should be focused on those nutrient sources that are continuing or increasing in importance and are subject to the greatest error in estimation of magnitude. These sources include flows through ungauged tributaries, stormwater runoff from upland land uses, ground water seepage, and internal nutrient loading. Although there is likely to be significant error in the estimates of muck farm discharges, most of the farms have recently been acquired by SJRWMD and discharges have ceased (Table 2).

SJRWMD has contracted with USGS to measure flows through the Dead River, which connects Lakes Eustis and Harris-Little Harris. Although measurements are expected to be limited to 1–2 years, if discharges can be related to differences in water elevations between the two lakes, then flows may be estimated for other time periods. It is also recommended that flows in the Yale-Griffin Canal be measured, particularly if proposed changes in the regulation schedule for Lake Griffin are implemented. These changes would raise maximum water levels in Lake Griffin, probably increasing the frequency of flows from the lake into Lake Yale.

Although the estimates of stormwater runoff are indirect, based on measurements made in other areas of nutrient concentrations and runoff coefficients, it would be prohibitively expensive to attempt direct measurements of stormwater runoff in such a large drainage basin. However, there are two problems with the estimates of stormwater loading that could be addressed. First, the drainage subbasins used in the present loading estimates are very large (each lake is surrounded by a single subbasin, with zero to several peripheral subbasins). Thus, the locations of the loading sources have not been precisely determined. Second, no information was incorporated on any existing stormwater management facilities in the basin. Development of a finer-scale subbasin drainage map, which incorporates information on existing stormwater management facilities in the subbasins, would allow refinement of estimates of nutrient loading and determination of priority subbasins for stormwater management projects. Alternatively, priority areas for stormwater management could be determined by mapping nutrient runoff potentials for the basin using nutrient concentrations and runoff coefficients assigned to land use/soil combinations (Table 3). An initial prioritization by this method could focus more detailed drainage studies and the development of stormwater management plans on areas with high runoff potential.

Nutrient loading from ground water seepage can originate from two sources: (1) septic tank effluents in the zone immediately surrounding the lakes, which could contribute both nitrogen and phosphorus, and (2) more distant sources, which potentially include septic tanks and sprayfields for waste treatment and citrus processing plants; these sources are likely to contribute only nitrogen, due to the low mobility of phosphorus in soils. Septic tank effluents appear to make the most significant contributions to nutrient loading in Lake Weir, although even in this lake estimated loadings from septic tanks are less than 10% of total loadings. Substantial rates of failure of septic tanks would be required to significantly increase nutrient loading from septic tank effluents. The SWIM Plan for the UORB (Fulton 1995) includes a proposed project to contract with county health units to investigate failing septic systems and enforce existing septic tank regulations. Such an effort may be worth considering for Lake Weir, but seems unnecessary for the other lakes unless a study of Lake Weir indicates that failing septic tanks can significantly increase nutrient loading.

As discussed previously, ground water seepage is a potentially significant source of nitrogen for the UORB lakes. However, TN:TP ratios for the lakes and experimental studies of nutrient limitation in Lake Apopka (Schelske et al. 1992; Aldridge et al. 1993) suggest that phosphorus is the primary nutrient that contributes to eutrophication in the UORB lakes. Therefore, further investigation of nitrogen levels in ground water should be pursued if concerns are raised about ground water contamination. However, elevated nitrogen levels in ground water are unlikely to contribute significantly to eutrophication of surface waters in the UORB.

The internal phosphorus loading in the lakes will have major effects on responses of lake water quality to reduction in external loading, in both equilibrium in-lake nutrient concentrations and in the time to reach equilibrium. The equilibrium net sedimentation coefficients used in the present study are derived from empirical studies conducted in other areas. There are no measurements for this basin of temporal changes in sedimentation coefficients following a change in nutrient loading. Studies are needed of the nutrient content of lake sediments and nutrient sedimentation rates. Studies of past sedimentation rates can verify the estimates of sedimentation rates and expected equilibrium sedimentation rates under reduced nutrient loading used in the trophic modeling for the present study. Measurements of nutrient content of lake sediments may allow a rough estimate of the time period required for recovery of net sedimentation coefficients to a new equilibrium after a reduction in external nutrient loading (Sas 1989).

A study of nutrient and sediment deposition in Lake Griffin has been initiated with funding from Lake County Water Authority. One goal of the first phase of the study is to determine the feasibility and methodology to quantitatively estimate the basinwide net sedimentation rates of nutrients. Quantitative estimation of basinwide net sedimentation rates will be the focus of Phase II of the study. Phase I also will measure the nutrient content of sediments and use paleolimnological analyses of sedimentary diatoms to infer historic lake total phosphorus concentrations. The paleolimnological analysis of historic trophic state will provide a verification of the phosphorus concentration predicted under predevelopment conditions in the present study.

Finally, efforts should be made to expand and coordinate existing water quality monitoring programs in the basin to obtain reliable data for detailed modeling, assessment, and evaluation of effects of restoration and management actions. The SWIM Plan for the UORB (Fulton 1995) includes a project to coordinate current water quality sampling programs of all agencies to prevent future omissions in coverage, to eliminate redundant sampling, to maintain a high level of quality assurance, and to provide for future water quality sampling needs.

RECOMMENDATIONS FOR WATER QUALITY IMPROVEMENT—POLLUTANT LOAD REDUCTION GOALS

It is recommended that an interim PLRG for the UORB be a combination of reduction of muck farm discharges to levels expected from wetland areas by restoration or regulation with a reduction in Apopka-Beauclair Canal TP concentrations to the levels expected under Lake Apopka PLRGs. Estimates of nutrient loading and in-lake TP concentrations under current conditions and under the proposed PLRGs were calculated by the methods presented previously for the time period covered by the most recent land use map, 1986–90 (Table 26). The interim PLRGs are

Table 26. Estimated mean total phosphorus (TP) loadings to the upper Ocklawaha River Basin lakes and predicted equilibrium in-lake TP concentrations under existing conditions and under proposed interim pollutant load reduction goals (PLRGs), 1986–90

Lake	Current Conditions		Proposed Interim PLRG	
	TP Loading (kg/yr)	TP Concentration (mg/L)	TP Loading (kg/yr)	TP Concentration (mg/L)
Beauclair	21,296	0.250	4,434	0.065
Dora	16,288	0.108	6,835	0.047
Harris-Little Harris	20,255	0.025	17,798	0.023
Eustis	23,808	0.052	12,465	0.033
Griffin	38,887	0.102	13,276	0.046
Yale	6,617	0.032	6,617	0.032
Weir	3,259	0.007	3,259	0.007

predicted to have the greatest effects on the lakes with the worst water quality in the basin: Lakes Beauclair, Dora, Eustis, and Griffin. Implementation of the interim PLRGs in these lakes are predicted to reduce estimated total phosphorus loadings by 48–79%, and reduce estimated in-lake total phosphorus concentrations by 37–74% (Table 26). These actions would be expected to have minor effects on Lake Harris-Little Lake Harris and no effects on Lakes Yale and Weir, but all three of these lakes currently have better water quality (Table 11), and the lack of a single dominant nutrient source for these lakes makes water quality improvement more difficult.

Muck farm and Apopka-Beauclair Canal discharges are already being addressed through acquisition and restoration of muck farms in the UORB (Table 2), and through the Lake Apopka SWIM Program (Conrow et al. 1993). The primary nutrient source that is not addressed by these programs is stormwater runoff from upland land uses (primarily residential and agriculture). Stormwater loading is likely to increase in importance as urbanization increases in the basin and muck farm discharges are reduced or eliminated. The development of final PLRGs for the UORB should focus on reductions in nutrient loads associated with stormwater runoff.

As mentioned previously, development of finer-scale subbasin drainage maps or maps of stormwater runoff potential would assist in the determination of priority areas for stormwater management projects. Existing information can be used to judge which lakes would be expected to be most responsive to reductions in stormwater nutrient loading. Because the ability of a lake to assimilate nutrient loading is related to the size of the lake, a good indicator of the potential impact of nutrient loading from stormwater (or any other source) is the rate of loading per unit area of lake surface. By this measure ($mg/m^2/yr$), reductions in phosphorus loading from residential-urban runoff would have the greatest impacts on water quality in Lakes Eustis and Dora (Table 27). Reductions in phosphorus loading from upland agriculture runoff would have the greatest impacts on water quality in Lakes Yale and Beauclair. Reductions in

Table 27. Mean phosphorus loadings to the upper Ocklawaha River Basin lakes from stormwater runoff from residential-urban and upland agriculture land uses, 1986–90

Lake	Residential-Urban			Upland Agriculture		
	kg/yr	mg/m²/yr	% of Total	kg/yr	mg/m²/yr	% of Total
Beauclair	366	83.3	1.7	638	145.2	3.0
Dora	1,674	94.3	10.3	242	13.6	1.5
Harris-Little Harris	3,718	49.2	18.4	3,943	52.1	19.5
Eustis	3,323	105.9	14.0	1,909	60.8	8.0
Griffin	2,603	68.3	6.7	1,667	43.8	4.3
Yale	1,146	70.4	17.3	2,027	124.6	30.6
Weir	933	41.0	28.6	573	25.2	17.6

phosphorus loading from both residential-urban and upland agriculture runoff would have the greatest impacts on water quality in Lakes Eustis, Yale, and Beauclair. However, the high stormwater runoff loading per unit area to Lake Beauclair is primarily due to the very small size of the lake. As discussed previously, discharges from the Apopka-Beauclair Canal account for 85% of phosphorus loading, whereas stormwater runoff represents a very small portion of total phosphorus loading. The modeling of alternative restoration scenarios also shows the importance of stormwater runoff for Lakes Eustis and Yale. Reduction of TP loading to predevelopment levels, which in these basins is largely a reduction in stormwater runoff, produced substantial decreases in predicted equilibrium in-lake TP concentrations (Figures 30[B] and 32[B]).

Although phosphorus loading to Lake Harris-Little Lake Harris from stormwater runoff is high when expressed as either kilograms per year or percent of total loading, loading per unit area of lake surface is relatively low because of the large size of the lake (Table 27). As a result, reductions in phosphorus loading from stormwater runoff would be less effective in improving water quality in Lake Harris-Little Lake Harris. On the other hand, stormwater runoff is the primary controllable nutrient source for Lake Harris-Little Lake Harris, so PLRGs for this lake must address stormwater runoff. Similarly for Lake Weir, stormwater runoff is the primary controllable nutrient source, so even though present stormwater loading rates per unit area may be relatively low (Table 27), maintenance or improvement in water quality requires control of stormwater runoff.

The nutrient budgets identify a few other controllable nutrient sources for the lakes that could be addressed in the development of final PLRGs. Weak waste discharges from citrus processing plants represent a significant nutrient source for Lake Yale (Figure 22). Reduction in septic tank effluents may be considered, particularly for Lake Weir. Crisman et al. (1992) recommended installation of a central sewer system for heavily populated areas of the Lake Weir watershed. However, the nutrient budget for Lake Weir indicates that stormwater management would be a more effective control on nutrient loading to the lake (Figure 24). Implementation of the SWIM Plan project for the UORB (Fulton et al. 1995) to contract with county health units to investigate failing septic systems and enforce existing septic tank regulations may be a more cost-effective means of reducing nutrient loading from septic tank effluents than construction of central sewer systems.

Implementation of a full-scale marsh flow-way project may be an alternate means of achieving nutrient levels similar to predevelopment conditions in Lake Griffin, if significant reductions in stormwater loading are not feasible (Figures 31 and 34). High flow rates and effective retention of nutrients are essential for a flow-way to achieve significant reductions in Lake Griffin nutrient concentrations. Evaluations of nutrient retention efficiency by the Lake Apopka flow-way and the pilot-scale Lake Griffin flow-way projects will be important in determining the cost-effectiveness of a full-scale marsh flow-way for Lake Griffin.

Reductions in internal nutrient loading through sediment removal or treatment are expensive and of questionable utility for the

UORB lakes. Studies of sedimentary nutrient stores and internal nutrient cycling are necessary before decisions can be made regarding the necessity and efficacy of restoration actions to reduce internal nutrient loading.

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APPENDIXES

Appendix A—Land Cover Codes Included in Each Land Use Category

Table A1. Land cover codes included in each land use category

		1984 Land Use Maps		1987–89 Land Use Maps	
Land Use Group	Land Use	FLUCCS Code	Land Use	FLUCCS Code	Land Use
1	Residential low-medium density	110	Residential low density	110	Residential low density
		120	Residential medium density	120	Residential medium density
2	Residential high density	130	Residential high density	130	Residential high density
3	Commercial	140	Commercial and services	140	Commercial and services
		800	Utilities and communications	810	Transportation
				820	Communications
		810	Transportation	830	Utilities
4	Institutional	170	Institutional	170	Institutional
5	Industrial/mining	150	Industrial	150	Industrial
		160	Extractive	160	Extractive
		742	Borrow areas	742	Borrow areas
				743	Spoil areas
6	Recreation/open land	180	Recreation	180	Recreation
		190	Open land	190	Open land
				740	Rural disturbed land
7	Forest/rangeland	329	Grassy scrub	224	Abandoned tree crops
		411	Pine flatwood	310	Herbaceous rangeland
		412	Sandhill community	320	Shrub and brushland
		413	Sand pine scrub	330	Mixed rangeland
		421	Xeric hammock	410	Upland coniferous forest
		427	Mesic hammock	420	Upland hardwood forest
		441	Planted pine	430	Upland hardwood forest continued
		443	Clear-cut areas		
				440	Tree plantations

Table A1—*Continued*

Land Use Group	Land Use	1984 Land Use Maps: FLUCCS Code	1984 Land Use Maps: Land Use	1987–89 Land Use Maps: FLUCCS Code	1987–89 Land Use Maps: Land Use
8	Open water/wetlands	510	Rivers and streams	510	Rivers and streams
		520	Lakes and ponds	520	Lakes and ponds
		611	Bay swamps	530	Reservoirs
		615	Stream and lake swamps	610	Wetland hardwood forest
		617	Hydric hammock	611	Bay swamps
		621	Cypress forest	615	Stream and lake swamps
		641	Freshwater marsh	620	Wetland coniferous forest
		643	Wet prairies	621	Cypress forest
				630	Wetland forest mixed
				640	Vegetated nonforested wetlands
				641	Freshwater marsh
				643	Wet prairies
				644	Emergent aquatic vegetation
				645	Submerged aquatic vegetation
				646	Mixed scrub-shrub wetland
				710	Non-swimming beaches
9	Confined feedlots	230	Feeding operations	232	Poultry feeding operations
10	Pasture	211	Improved pasture	211	Improved pasture
				212	Unimproved pasture
				213	Woodland pasture
				251	Horse farms
11	Citrus groves	221	Citrus groves	221	Citrus groves
				223	Other groves
12	Other agriculture	210	Cropland	210	Cropland and pastureland
		214	Row crops	214	Row crops
		240	Nurseries and vineyards	215	Field crops
				216	Mixed crops
				240	Nurseries and vineyards
				241	Tree nurseries
				243	Ornamentals
				245	Floriculture
				260	Rural open lands-agriculture
				261	Fallow cropland
13	Muck farms	210	Cropland	211	Improved pasture
		211	Improved pasture	214	Row crops
		214	Row crops	224	Abandoned tree crops
				240	Nurseries and vineyards

APPENDIX B—LITERATURE SUMMARY OF NUTRIENT CONCENTRATIONS IN STORMWATER RUNOFF

Table B1. Literature summary of nutrient concentrations in stormwater runoff

Land Use	Parameter	Minimum Reported (mg/L)	Maximum Reported (mg/L)	Mean	Number of Studies Contributing to Mean	Source
Single-family residential (>1 dwelling unit/acre)						
	TP	0.073	1.69	0.499*	7	Harper 1992
	TN	0.605	4.62	2.29	7	Harper 1992
Multi-family residential						
	TP	0.2	0.73	0.47	7	Harper 1992
	TN	1.02	4.68	2.22	7	Harper 1992
Low-density residential (rural) (<1 dwelling unit/acre)						
	TP			0.177†		Harper 1992
	TN			1.77†		Harper 1992
Institutional/low-intensity commercial						
	TP	0.1	0.19	0.15	4	Harper 1992
	TN	0.89	1.53	1.18	4	Harper 1992
High-intensity commercial						
	TP	0.15	0.82	0.43	3	Harper 1992
	TN	2.15	3.53	2.83	3	Harper 1992
Industrial						
	TP	0.19	0.42	0.31	3	Harper 1992
	TN	1.42	2.53	1.79	3	Harper 1992
Pasture (primarily)						
	TP	0.27	0.697	0.476	3	Harper 1992
	TP			0.12	1	CH2M Hill 1978
	TN	2.37	2.58	2.477	3	Harper 1992
Citrus (primarily)						
	TP	0.09	0.24	0.14	3	Harper 1992
	TN	1.33	3.26	2.05	3	Harper 1992

Table B1—*Continued*

Land Use	Parameter	Miminum Reported (mg/L)	Maximum Reported (mg/L)	Mean	Number of Studies Contributing to Mean	Source
Row crops						
	TP			0.562	1	Harper 1992
	TP	0.126	0.34	0.233	2	CH2M Hill 1978
	TP	0.25	1.03	0.5675	4	Izuno et al. 1991
	TDP	0.2	0.51	0.325	4	Izuno et al. 1991
	TN			2.68	1	Harper 1992
Confined animal feedlots						
	TP	5.1	85	47.5	4	Reckhow et al. 1990
	TN	29.3	300	127.1	4	Reckhow et al. 1990
Recreational/open space/forest/rangeland						
	TP	0.02	0.07	0.053	3	Harper 1992
	TN	0.9	1.47	1.25	3	Harper 1992
Mining/extractive						
	TP			0.15	1	Harper 1992
	TN			1.18	1	Harper 1992
Wetlands						
	TP	0.09	0.33	0.19	4	Harper 1992
	TN	1.02	2.26	1.6	4	Harper 1992
Open water/lakes						
	TP	0.04	0.17	0.11	2	Harper 1992
	TN	0.73	2.22	1.25	3	Harper 1992

*A value of 0.3 mg TP/L was recommended for stormwater runoff from single-family residential land uses. The recommended value excludes data from one study which reported a mean value significantly greater than the other studies, due to several extreme values in the data set.

†No data; recommended values are the average of single-family residential and open space land uses.

APPENDIX C—LITERATURE SUMMARY OF DISSOLVED PHOSPHORUS, ORTHO-PHOSPHATE, AND TOTAL PHOSPHORUS CONCENTRATIONS IN RUNOFF

Table C1. Florida agriculture—Total dissolved phosphorus (TDP) and total phosphorus (TP) concentrations in runoff

Location/Land Use	TDP	TP	Proportion Dissolved	Source
	mg/L	mg/L		
EAA, FL sugarcane	0.170	0.280	0.607	Izuno et al. 1991
EAA, FL radish	0.200	0.250	0.800	Izuno et al. 1991
EAA, FL cabbage	0.310	0.560	0.554	Izuno et al. 1991
EAA, FL drained fallow	0.280	0.430	0.651	Izuno et al. 1991
EAA, FL flooded fallow	0.510	1.030	0.495	Izuno et al. 1991
SJWCD, FL citrus/pasture	0.060	0.090	0.667	Fall and Hendrickson 1988
Willowbrook Farms, FL row crop			0.550	Hendrickson 1987
USJ, FL pasture	0.140	0.230	0.609	Fall 1990
USJ, FL citrus/row crop	0.120	0.160	0.750	Fall 1990
	kg/ha/yr	kg/ha/yr		
Turkey Creek, FL TC9-pasture	0.110	0.200	0.550	Dierberg 1991
Mean Florida agriculture proportion TDP			0.623	

Table C2. Non-Florida agriculture—Total dissolved phosphorus (TDP) and total phosphorus (TP) concentrations in runoff

Location/Land Use	TDP	TP	Proportion Dissolved	Source
Row crops	kg/ha/yr	kg/ha/yr		
MN corn continuous	0.400	18.200	0.022	Reckhow et al. 1980
MN corn	0.300	13.700	0.022	Reckhow et al. 1980
MN corn surface spread manure	0.500	8.100	0.062	Reckhow et al. 1980
MN corn plowdown manure	0.400	9.800	0.041	Reckhow et al. 1980
MN corn rotation	0.170	3.140	0.054	Reckhow et al. 1980
MN corn continuous	0.330	5.550	0.059	Reckhow et al. 1980
IA corn continuous contour	0.190	0.496	0.383	Reckhow et al. 1980
IA corn continuous contour	0.085	1.033	0.082	Reckhow et al. 1980
IA corn continuous contour	0.237	2.118	0.112	Reckhow et al. 1980
IA corn continuous contour	0.040	0.594	0.067	Reckhow et al. 1980
IA corn continuous contour	0.175	0.279	0.627	Reckhow et al. 1980
IA corn continuous contour	0.019	0.092	0.207	Reckhow et al. 1980
IA corn continuous contour	0.043	0.287	0.150	Reckhow et al. 1980
IA corn continuous terraced	0.081	0.090	0.900	Reckhow et al. 1980
IA corn continuous terraced	0.009	0.024	0.375	Reckhow et al. 1980
IA corn continuous terraced	0.059	0.287	0.206	Reckhow et al. 1980
IA corn continuous terraced	0.119	0.613	0.194	Reckhow et al. 1980
IA corn continuous terraced	0.238	0.399	0.596	Reckhow et al. 1980
IA corn continuous terraced	0.018	0.050	0.360	Reckhow et al. 1980
IA corn continuous terraced	0.128	0.259	0.494	Reckhow et al. 1980
GA corn continuous	0.540	2.210	0.244	Reckhow et al. 1980
MS soybeans conventional	0.250	17.750	0.014	Reckhow et al. 1980
MS soybeans no till	1.800	2.900	0.621	Reckhow et al. 1980
OK cotton continuous	2.180	11.520	0.189	Reckhow et al. 1980
OK cotton continuous	0.680	2.380	0.286	Reckhow et al. 1980
OK cotton continuous	0.860	3.540	0.243	Reckhow et al. 1980
OK cotton continuous	1.060	5.070	0.209	Reckhow et al. 1980
OK cotton continuous	1.670	10.750	0.155	Reckhow et al. 1980
OK cotton continuous	0.510	2.070	0.246	Reckhow et al. 1980

Table C2—*Continued*

Location/Land Use	TDP	TP	Proportion Dissolved	Source
OK cotton continuous	0.700	3.500	0.200	Reckhow et al. 1980
OK cotton continuous	0.980	5.660	0.173	Reckhow et al. 1980
MS soybeans-corn no till	0.500	6.800	0.074	Reckhow et al. 1980
MS corn-soybeans no till	2.200	4.400	0.500	Reckhow et al. 1980
Non-row crops				
SD alfalfa-bromegrass	0.730	0.970	0.753	Reckhow et al. 1980
OK wheat continuous	0.610	3.340	0.183	Reckhow et al. 1980
OK wheat continuous	0.190	0.800	0.238	Reckhow et al. 1980
OK wheat continuous	0.360	0.960	0.375	Reckhow et al. 1980
OK wheat continuous	0.130	2.320	0.056	Reckhow et al. 1980
OK wheat continuous	0.520	4.290	0.121	Reckhow et al. 1980
OK wheat continuous	0.090	0.590	0.153	Reckhow et al. 1980
OK wheat continuous	0.260	0.790	0.329	Reckhow et al. 1980
OK wheat continuous	0.110	2.320	0.047	Reckhow et al. 1980
MN oats rotation	0.220	0.650	0.338	Reckhow et al. 1980
MN hay rotation	0.600	0.640	0.938	Reckhow et al. 1980
Pasture				
OH pasture	3.000	3.600	0.833	Reckhow et al. 1980
OH pasture	0.400	0.850	0.471	Reckhow et al. 1980
IA pasture	0.193	0.251	0.769	Reckhow et al. 1980
IA pasture	0.064	0.081	0.790	Reckhow et al. 1980
IA pasture	0.386	0.512	0.754	Reckhow et al. 1980
GA pasture	1.269	1.345	0.943	Reckhow et al. 1980
OK pasture	0.140	3.860	0.036	Reckhow et al. 1980
OK pasture	0.070	1.060	0.066	Reckhow et al. 1980
OK pasture	0.030	1.860	0.016	Reckhow et al. 1980
OK pasture	0.010	0.260	0.038	Reckhow et al. 1980
OK pasture	0.100	1.440	0.069	Reckhow et al. 1980
OK pasture	0.020	1.240	0.083	Reckhow et al. 1980

Table C2—*Continued*

Location/Land Use	TDP	TP	Proportion Dissolved	Source
OK pasture	0.020	0.270	0.074	Reckhow et al. 1980
OK pasture	0.000	0.020	0.000	Reckhow et al. 1980
OK pasture	3.270	4.900	0.667	Reckhow et al. 1980
OK pasture	2.430	3.090	0.786	Reckhow et al. 1980
OK pasture	0.010	0.760	0.013	Reckhow et al. 1980
OK pasture	0.020	0.200	0.100	Reckhow et al. 1980
Mixed agriculture				
IN mixed agriculture	0.240	5.400	0.044	Reckhow et al. 1980
IN mixed agriculture	0.090	1.100	0.082	Reckhow et al. 1980
IN mixed agriculture	0.460	5.000	0.092	Reckhow et al. 1980
IN mixed agriculture	0.220	1.000	0.220	Reckhow et al. 1980
IA mixed agriculture	0.319	0.648	0.492	Reckhow et al. 1980
IA mixed agriculture	0.190	0.270	0.704	Reckhow et al. 1980
Canada mixed agriculture	0.233	1.290	0.181	Reckhow et al. 1980
Canada mixed agriculture	0.210	1.280	0.164	Reckhow et al. 1980
Canada mixed agriculture	0.060	0.260	0.231	Reckhow et al. 1980
Canada mixed agriculture	0.500	0.910	0.549	Reckhow et al. 1980
Canada mixed agriculture	0.330	1.000	0.330	Reckhow et al. 1980
Canada mixed agriculture	0.430	1.530	0.281	Reckhow et al. 1980
Canada mixed agriculture	0.070	0.160	0.438	Reckhow et al. 1980
Canada mixed agriculture	0.030	0.080	0.375	Reckhow et al. 1980
Canada mixed agriculture	0.510	1.530	0.333	Reckhow et al. 1980
Canada mixed agriculture	0.200	0.490	0.408	Reckhow et al. 1980
Canada mixed agriculture	0.360	0.910	0.396	Reckhow et al. 1980
Canada mixed agriculture	0.360	0.810	0.444	Reckhow et al. 1980
Mean non-Florida agriculture proportion TDP			0.300	

Note: Abbreviations for location use the U.S. Postal Service 2-letter abbreviations for states.

Table C3. Mixed urban-agricultural—Total dissolved phosphorus (TDP) and total phosphorus (TP) concentrations in runoff

Location/Land Use	TDP	TP	Proportion Dissolved	Source
Residential	kg/ha/yr	kg/ha/yr		
FL, Turkey Creek TC0-residential	0.120	0.280	0.429	Dierberg 1991
FL, Turkey Creek TC8-residential	0.110	0.240	0.458	Dierberg 1991
Mixed urban-residential-agricultural				
FL, Turkey Creek TC7	0.100	0.190	0.526	Dierberg 1991
FL, Turkey Creek TPM	0.120	0.220	0.545	Dierberg 1991
FL, Tallahassee 67% residential	0.220	6.230	0.035	Reckhow et al. 1980
Canada >60% urban	0.107	1.630	0.066	Reckhow et al. 1980
S Africa residential	0.220	0.600	0.367	Reckhow et al. 1980
Mean Florida mixed urban-agricultural proportion TDP			0.399	
Mean other areas mixed urban-agricultural proportion TDP			0.216	

Table C4. Forested—Total dissolved phosphorus (TDP) and total phosphorus (TP) concentrations in runoff

Location/Land Use	TDP	TP	Proportion Dissolved	Source
	kg/ha/yr	kg/ha/yr		
Canada pine-spruce	0.032	0.060	0.533	Reckhow et al. 1980
Canada pine-spruce	0.024	0.036	0.667	Reckhow et al. 1980
NH maple, birch, beech	0.007	0.019	0.368	Reckhow et al. 1980
GA pine/hardwood	0.265	0.275	0.964	Reckhow et al. 1980
MS pine	0.094	0.281	0.335	Reckhow et al. 1980
MS pine	0.110	0.306	0.359	Reckhow et al. 1980
MS pine	0.097	0.357	0.272	Reckhow et al. 1980
MS pine	0.083	0.321	0.259	Reckhow et al. 1980
MS pine	0.055	0.226	0.243	Reckhow et al. 1980
Mean forested proportion TDP			0.444	

Note: Abbreviations for location use the U.S. Postal Service 2-letter abbreviations for states.

Table C5. Agriculture—Ortho-phosphate (PO_4) and total phosphorus (TP) concentrations in runoff

Location/Land Use	PO_4	TP	Proportion PO_4	Source
	mg/L	mg/L		
FL, Willowbrook Farms row crops	0.398	0.562	0.708	Hendrickson 1987
FL, Armstrong Slough citrus/pasture	0.035	0.090	0.389	Hendrickson 1987
FL, Ash Slough pasture	0.538	0.697	0.772	Hendrickson 1987
FL, SJWCD citrus/pasture	0.060	0.090	0.667	Fall and Hendrickson 1988
FL, USJ pasture	0.170	0.230	0.739	Fall 1990
FL, USJ citrus/row crop	0.090	0.160	0.563	Fall 1990
	kg/ha/yr	kg/ha/yr		
FL, Turkey Creek pasture	0.070	0.200	0.350	Dierberg 1991
FL, Alachua County	1.210	1.340	0.903	Baker et al. 1981
FL, Alachua County	0.630	0.860	0.733	Baker et al. 1981
FL, Near L. Jackson	0.140	0.510	0.275	Baker et al. 1981
Delaware	0.083	0.680	0.122	Baker et al. 1981
Delaware	0.078	0.480	0.163	Baker et al. 1981
Mean Florida agriculture proportion PO_4			0.610	
Mean non-Florida agriculture proportion PO_4			0.142	

Table C6. Florida residential/urban—Ortho-phosphate (PO_4) and total phosphorus (TP) concentrations in runoff

Location/Land Use	PO_4	TP	Proportion PO_4	Source
	kg/ha/yr	kg/ha/yr		
Near L. Jackson, FL urban	1.900	7.490	0.254	Baker et al. 1981
Orlando, FL urban	2.000	3.500	0.571	Baker et al. 1981
Ft. Lauderdale, FL urban	0.110	0.260	0.423	Baker et al. 1981
Near L. Jackson, FL residential	0.090	4.740	0.019	Baker et al. 1981
Orlando, FL residential	0.800	2.240	0.357	Baker et al. 1981
Turkey Creek, FL residential	0.100	0.280	0.357	Dierberg 1991
Turkey Creek, FL residential	0.120	0.240	0.500	Dierberg 1991
Mean Florida residential/urban proportion PO_4			0.354	

Table C7. Forested—Ortho-phosphate (PO_4) and total phosphorus (TP) concentrations in runoff

Location/Land Use	PO_4	TP	Proportion PO_4	Source
	kg/ha/yr	kg/ha/yr		
Alachua County, FL	0.300	0.330	0.909	Baker et al. 1981
Alachua County, FL	0.520	0.680	0.765	Baker et al. 1981
Bradford County, FL	0.200	0.400	0.500	Baker et al. 1981
Mississippi	0.029	0.300	0.097	Baker et al. 1981
Mean Florida forested proportion PO_4			0.725	
Mean non-Florida forested proportion PO_4			0.097	

APPENDIX D—ESTIMATED NUTRIENT LOADING AND PREDICTED TROPHIC STATE FOR LAKE BEAUCLAIR

Table D1. Lake Beauclair—Phosphorus loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading $mg/m^2/yr$	% of Total
Residential low-medium density	91.6	73.3	301.8	325.0	355.2	327.2	288.9	251.9	57.32	1.38
Residential high density	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Commercial	0.0	0.0	8.7	9.3	10.2	9.4	8.3	6.6	1.49	0.04
Institutional	0.0	0.0	1.4	1.5	1.7	1.5	1.4	1.1	0.24	0.01
Industrial/mining	0.0	0.0	27.9	30.0	32.8	30.2	26.7	21.1	4.80	0.12
Recreation/open	0.0	0.0	5.9	6.3	6.9	6.4	5.6	4.4	1.01	0.02
Forest/rangeland	53.6	42.9	38.0	40.9	44.7	41.2	36.3	42.5	9.68	0.23
Water/wetlands	322.2	257.8	514.7	554.3	605.7	557.9	492.6	472.2	107.45	2.58
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	90.1	72.1	133.0	143.2	156.5	144.2	127.3	123.8	28.17	0.68
Citrus groves	335.8	268.6	148.8	160.3	175.1	161.3	142.4	198.9	45.26	1.09
Other agricultural	170.8	136.7	320.8	345.4	377.5	347.7	307.0	286.6	65.21	1.57
Muck farms	758.3	899.4	1,368.6	1,343.8	1,526.6	986.0	950.5	1,119.0	254.66	6.11
Septic tanks	13.3	13.3	13.3	13.3	13.3	13.3	13.3	13.3	3.03	0.07
Point sources	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Precipitation	57.4	45.9	55.9	60.2	65.7	60.5	53.5	57.0	12.97	0.31
Dry deposition	151.7	151.7	151.7	151.7	151.7	151.7	151.7	151.7	34.52	0.83
Apopka-Beauclair Canal	9,416.7	8,226.4	17,578.4	31,982.9	22,457.6	10,934.6	8,283.6	15,554.3	3,539.76	84.98
Total	11,461.5	10,188.1	20,668.9	35,168.1	25,981.2	13,773.1	10,889.1	18,304.4	4,165.57	100.02

Table D2. Lake Beauclair—Nitrogen loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading $mg/m^2/yr$	% of Total
Residential low-medium density	699.1	559.3	2,303.9	2,481.1	2,711.4	2,497.4	2,205.0	1,922.5	437.50	0.72
Residential high density	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Commercial	0.0	0.0	61.1	65.8	71.9	66.2	58.5	46.2	10.52	0.02
Institutional	0.0	0.0	11.2	12.0	13.1	12.1	10.7	8.4	1.92	0.00
Industrial/mining	0.0	0.0	168.9	181.9	198.8	183.1	161.7	127.8	29.08	0.05
Recreation/open	0.0	0.0	138.8	149.5	163.4	150.5	132.9	105.0	23.90	0.04
Forest/rangeland	1,265.0	1,012.1	895.4	964.3	1,053.8	970.6	857.0	1,002.6	228.17	0.38
Water/wetlands	2,860.2	2,288.3	4,569.3	4,920.7	5,377.4	4,953.1	4,373.3	4,191.8	953.94	1.57
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	576.7	461.4	851.3	916.8	1,001.9	922.8	814.8	792.2	180.30	0.30
Citrus groves	4,916.7	3,933.6	2,179.0	2,346.6	2,564.4	2,362.0	2,085.5	2,912.5	662.82	1.09
Other agricultural	972.0	777.7	1,825.2	1,965.5	2,148.0	1,978.5	1,746.9	1,630.5	371.07	0.61
Muck farms	5,195.1	4,799.2	7,302.3	7,640.5	8,145.3	5,260.9	5,071.5	6,202.1	1,411.44	2.33
Septic tanks	235.6	235.6	235.6	235.6	235.6	235.6	235.6	235.6	53.62	0.09
Point sources	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Precipitation	2,768.4	2,214.8	2,695.2	2,902.5	3,171.8	2,921.5	2,579.5	2,750.5	625.95	1.03
Dry deposition	643.6	643.6	643.6	643.6	643.6	643.6	643.6	643.6	146.47	0.24
Apopka-Beauclair Canal	314,292.4	116,739.5	235,654.6	422,179.0	340,876.8	153,991.6	123,698.8	243,919.0	55,509.68	91.53
Total	334,424.8	133,665.1	259,535.4	447,605.4	368,377.2	177,149.5	144,675.4	266,490.3	60,646.37	100.00

Table D3. Lake Beauclair—Predicted lake trophic state variables

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90
Total P mean (mg/L)	0.149	0.199	0.257	0.339	0.240	0.210	0.205	0.228
Total P median (mg/L)	0.138	0.183	0.236	0.312	0.221	0.194	0.189	0.210
-1 standard error	0.089	0.119	0.153	0.202	0.143	0.125	0.122	0.136
+1 standard error	0.213	0.283	0.365	0.483	0.342	0.299	0.291	0.325
Total P (mg/L) (L&M)	0.119	0.187	0.239	0.304	0.220	0.204	0.181	0.208
Total P (mg/L) (C&B)	0.111	0.149	0.193	0.243	0.188	0.164	0.147	0.171
Total N mean (mg/L)	3.164	2.526	2.684	3.317	2.594	2.423	2.527	2.748
Total N median (mg/L)	3.029	2.418	2.570	3.175	2.484	2.320	2.419	2.631
-1 standard error	2.214	1.767	1.878	2.321	1.815	1.695	1.768	1.923
+1 standard error	4.145	3.309	3.516	4.345	3.399	3.174	3.310	3.600
Chlorophyll-*a* mean (μg/L)	50.609	44.429	50.812	67.657	48.234	43.406	44.829	49.997
Chlorophyll-*a* median (μg/L)	45.257	39.734	45.443	60.501	43.140	38.821	40.092	44.713
-1 standard error	27.301	23.970	27.414	36.497	26.024	23.419	24.186	26.973
+1 standard error	75.022	65.866	75.330	100.291	71.512	64.353	66.460	74.119
Secchi depth mean (m)	0.544	0.616	0.596	0.530	0.607	0.631	0.616	0.591
Secchi depth median (m)	0.506	0.574	0.555	0.493	0.565	0.587	0.574	0.550
-1 standard error	0.788	0.894	0.864	0.768	0.880	0.914	0.893	0.857
+1 standard error	0.349	0.396	0.383	0.340	0.390	0.405	0.396	0.380
TSI mean	76.767	77.201	80.165	84.857	79.221	77.300	77.469	78.997
TSI median	75.568	76.003	78.967	83.658	78.024	76.103	76.271	77.799
-1 standard error	68.416	68.851	71.815	76.506	70.872	68.951	69.119	70.647
+1 standard error	82.720	83.155	86.119	90.810	85.176	83.255	83.423	84.951

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

APPENDIX E—ESTIMATED NUTRIENT LOADING AND PREDICTED TROPHIC STATE FOR LAKE DORA

Table E1. Lake Dora—Phosphorus loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	222.9	178.4	738.9	795.7	869.5	800.9	707.2	616.2	34.73	4.25
Residential high density	1,700.6	1,360.5	394.9	425.2	464.7	428.0	377.9	736.0	41.48	5.07
Commercial	0.0	0.0	376.4	405.4	443.0	408.1	360.3	284.7	16.05	1.96
Institutional	1.8	1.4	18.3	19.7	21.5	19.8	17.5	14.3	0.81	0.10
Industrial/mining	49.5	39.6	43.2	46.5	50.8	46.8	41.3	45.4	2.56	0.31
Recreation/open	0.2	0.1	9.1	9.8	10.7	9.9	8.7	6.9	0.39	0.05
Forest/rangeland	28.4	22.7	40.5	43.6	47.7	43.9	38.8	37.9	2.14	0.26
Water/wetlands	535.4	428.3	448.0	482.4	527.2	485.6	428.7	476.5	26.86	3.28
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	68.7	55.0	23.6	25.4	27.7	25.5	22.5	35.5	2.00	0.24
Citrus groves	169.8	135.8	43.6	47.0	51.3	47.3	41.7	76.6	4.32	0.53
Other agricultural	0.0	0.0	161.3	173.7	189.8	174.8	154.3	122.0	6.87	0.84
Muck farms	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Septic tanks	111.7	111.7	111.7	111.7	111.7	111.7	111.7	111.7	6.30	0.77
Point sources	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Precipitation	231.7	185.3	225.5	242.9	265.4	244.5	215.9	230.2	12.97	1.59
Dry deposition	612.7	612.7	612.7	612.7	612.7	612.7	612.7	612.7	34.53	4.22
Lake Beauclair discharge	6,852.3	6,382.9	11,537.7	19,322.7	18,395.1	8,581.4	6,621.8	11,099.1	625.52	76.52
Total	10,585.7	9,514.4	14,785.4	22,764.2	22,088.8	12,040.9	9,761.0	14,505.7	817.53	99.99

Table E2. Lake Dora—Nitrogen loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	1,701.8	1,361.5	5,639.9	6,073.7	6,637.4	6,113.6	5,398.0	4,703.7	265.09	1.97
Residential high density	8,032.4	6,426.3	1,865.1	2,008.5	2,195.0	2,021.7	1,785.1	3,476.3	195.92	1.46
Commercial	0.0	0.0	2,649.1	2,852.8	3,117.6	2,871.6	2,535.5	2,003.8	112.93	0.84
Institutional	14.2	11.4	143.7	154.7	169.1	155.7	137.5	112.3	6.33	0.05
Industrial/mining	300.4	240.3	261.8	281.9	308.1	283.8	250.6	275.3	15.51	0.12
Recreation/open	4.4	3.5	215.0	231.5	253.0	233.0	205.8	163.7	9.23	0.07
Forest/rangeland	670.3	536.3	955.0	1,028.5	1,123.9	1,035.2	914.0	894.7	50.43	0.37
Water/wetlands	4,752.8	3,802.5	3,976.8	4,282.6	4,680.1	4,310.8	3,806.2	4,230.3	238.41	1.77
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	440.0	352.0	150.7	162.3	177.4	163.4	144.3	227.2	12.80	0.10
Citrus groves	2,486.0	1,988.9	638.7	687.8	751.7	692.4	611.3	1,122.4	63.26	0.47
Other agricultural	0.2	0.1	917.6	988.2	1,079.9	994.7	878.3	694.1	39.12	0.29
Muck farms	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Septic tanks	1,970.9	1,970.9	1,970.9	1,970.9	1,970.9	1,970.9	1,970.9	1,970.9	111.07	0.83
Point sources	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Precipitation	11,178.8	8,943.6	10,883.3	11,720.3	12,808.1	11,797.3	10,416.4	11,106.8	625.95	4.65
Dry deposition	2,599.0	2,599.0	2,599.0	2,599.0	2,599.0	2,599.0	2,599.0	2,599.0	146.47	1.09
Lake Beauclair discharge	232,502.7	113,311.9	203,870.7	276,705.7	324,158.9	165,885.9	120,068.8	205,214.9	11,565.38	85.94
Total	266,653.9	141,548.2	236,737.3	311,748.4	362,030.1	201,129.0	151,721.7	238,795.4	13,457.90	100.02

Table E3. Lake Dora—Predidcted lake trophic state variables

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90
Total P mean (mg/L)	0.055	0.056	0.071	0.092	0.087	0.062	0.058	0.069
Total P median (mg/L)	0.051	0.052	0.065	0.085	0.080	0.057	0.053	0.063
-1 standard error	0.033	0.034	0.042	0.055	0.052	0.037	0.034	0.041
+1 standard error	0.079	0.080	0.101	0.131	0.123	0.089	0.082	0.098
Total P (mg/L) (L&M)	0.066	0.103	0.106	0.122	0.118	0.102	0.090	0.101
Total P (mg/L) (C&B)	0.061	0.075	0.084	0.098	0.096	0.079	0.071	0.081
Total N mean (mg/L)	1.470	1.562	1.485	1.405	1.521	1.522	1.426	1.484
Total N median (mg/L)	1.407	1.495	1.421	1.345	1.456	1.457	1.365	1.421
-1 standard error	1.028	1.093	1.039	0.983	1.064	1.064	0.997	1.038
+1 standard error	1.925	2.046	1.945	1.841	1.993	1.993	1.867	1.944
Chlorophyll-*a* mean (µg/L)	18.047	19.220	19.622	20.105	21.315	19.331	17.733	19.339
Chlorophyll-*a* median (µg/L)	16.139	17.185	17.546	17.979	19.061	17.285	15.857	17.293
-1 standard error	9.736	10.367	10.585	10.846	11.498	10.427	9.566	10.432
+1 standard error	26.753	28.488	29.086	29.804	31.596	28.653	26.286	28.667
Secchi depth mean (m)	0.990	0.957	0.984	1.015	0.971	0.971	1.007	0.985
Secchi depth median (m)	0.922	0.891	0.916	0.945	0.904	0.904	0.937	0.917
-1 standard error	1.435	1.388	1.427	1.471	1.408	1.408	1.460	1.428
+1 standard error	0.636	0.615	0.632	0.652	0.624	0.624	0.647	0.633
TSI mean	59.951	60.698	62.352	64.199	64.442	61.397	60.012	61.864
TSI median	58.753	59.497	61.153	63.001	63.243	60.197	58.812	60.665
-1 standard error	51.601	52.345	54.001	55.849	56.091	53.045	51.660	53.513
+1 standard error	65.905	66.649	68.305	70.153	70.395	67.349	65.964	67.817

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

APPENDIX F—ESTIMATED NUTRIENT LOADING AND PREDICTED TROPHIC STATE FOR LAKE HARRIS-LITTLE LAKE HARRIS

Table F1. Lake Harris-Little Lake Harris—Phosphorus loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	908.0	726.4	1,739.4	1,873.1	2,047.0	1,885.4	1,664.7	1,549.1	20.48	7.92
Residential high density	429.0	343.2	551.3	593.7	648.9	597.6	527.7	527.3	6.97	2.69
Commercial	464.1	371.3	889.9	958.3	1,047.3	964.6	851.7	792.5	10.48	4.05
Institutional	20.9	16.8	72.1	77.7	84.9	78.2	69.0	59.9	0.79	0.31
Industrial/mining	162.2	129.8	240.2	258.7	282.7	260.4	229.9	223.4	2.95	1.14
Recreation/open	7.6	6.0	18.4	19.8	21.7	19.9	17.6	15.9	0.21	0.08
Forest/rangeland	107.3	85.9	255.3	274.9	300.5	276.7	244.4	220.7	2.92	1.13
Water/wetlands	3,237.1	2,589.8	3,330.3	3,586.4	3,919.3	3,610.0	3,187.4	3,351.5	44.31	17.13
Feedlots	883.2	706.6	0.0	0.0	0.0	0.0	0.0	227.1	3.00	1.16
Pasture	2,400.2	1,920.3	1,348.3	1,452.0	1,586.7	1,461.5	1,290.4	1,637.1	21.65	8.37
Citrus groves	1,665.7	1,332.6	391.7	421.8	460.9	424.6	374.9	724.6	9.58	3.70
Other agricultural	46.9	37.6	1,983.1	2,135.6	2,333.8	2,149.6	1,898.0	1,512.1	19.99	7.73
Muck farms	1,907.4	1,536.7	2,080.4	2,405.8	3,800.3	2,545.6	2,159.8	2,348.0	31.05	12.00
Septic tanks	522.6	522.6	522.6	522.6	522.6	522.6	522.6	522.6	6.91	2.67
Citrus plant	122.6	122.6	122.6	122.6	122.6	122.6	122.6	122.6	1.62	0.63
Precipitation	987.5	790.0	961.4	1,035.3	1,131.4	1,042.1	920.1	981.1	12.97	5.01
Dry deposition	2,611.3	2,611.3	2,611.3	2,611.3	2,611.3	2,611.3	2,611.3	2,611.3	34.53	13.34
Palatlakaha River inflow	2,956.9	77.5	539.4	1,251.3	1,485.3	461.4	45.1	973.8	12.88	4.98
Spring discharges	1,170.3	1,167.1	1,167.1	1,167.1	1,170.3	1,167.1	1,167.1	1,168.0	15.44	5.97
Total	20,610.8	15,094.1	18,824.8	20,768.0	23,577.5	20,201.2	17,904.3	19,568.6	258.73	100.01

Table F2. Lake Harris-Little Lake Harris—Nitrogen loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	6,930.9	5,545.0	13,277.1	14,298.2	15,625.3	14,392.1	12,707.5	11,825.2	156.36	5.22
Residential high density	2,026.4	1,621.2	2,604.2	2,804.5	3,064.8	2,822.9	2,492.5	2,490.9	32.94	1.10
Commercial	3,266.1	2,613.0	6,262.4	6,744.0	7,369.9	6,788.3	5,993.7	5,576.8	73.74	2.46
Institutional	164.7	131.8	567.5	611.2	667.9	615.2	543.2	471.6	6.24	0.21
Industrial/mining	983.7	787.0	1,457.0	1,569.0	1,714.7	1,579.4	1,394.5	1,355.0	17.92	0.60
Recreation/open	178.2	142.6	433.9	467.3	510.6	470.3	415.3	374.0	4.95	0.17
Forest/rangeland	2,531.3	2,025.1	6,021.4	6,484.5	7,086.3	6,527.1	5,763.0	5,205.5	68.83	2.30
Water/wetlands	28,737.7	22,991.4	29,564.9	31,838.6	34,793.7	32,047.7	28,296.5	29,752.9	393.40	13.14
Feedlots	2,363.3	1,890.8	0.0	0.0	0.0	0.0	0.0	607.7	8.04	0.27
Pasture	15,362.6	12,290.8	8,629.7	9,293.4	10,156.0	9,354.5	8,259.5	10,478.1	138.54	4.63
Citrus groves	24,390.7	19,513.7	5,735.2	6,176.2	6,749.5	6,216.8	5,489.1	10,610.2	140.29	4.69
Other agricultural	267.1	213.7	11,283.8	12,151.6	13,279.4	12,231.4	10,799.7	8,603.8	113.76	3.80
Muck farms	12,975.9	8,288.4	12,735.0	14,214.6	15,686.1	13,582.8	11,621.5	12,729.2	168.31	5.62
Septic tanks	9,224.5	9,224.5	9,224.5	9,224.5	9,224.5	9,224.5	9,224.5	9,224.5	121.97	4.07
Citrus plant	284.6	284.6	284.6	284.6	284.6	284.6	284.6	284.6	3.76	0.13
Precipitation	47,647.6	38,120.2	46,387.9	49,955.4	54,592.0	50,283.5	44,397.7	47,340.6	625.95	20.91
Dry deposition	11,077.6	11,077.6	11,077.6	11,077.6	11,077.6	11,077.6	11,077.6	11,077.6	146.47	4.89
Palatlakaha River inflow	52,573.3	941.6	6,947.5	20,995.7	22,325.6	5,928.2	583.1	15,756.4	208.34	6.96
Spring discharges	42,763.3	42,646.4	42,646.4	42,646.4	42,763.3	42,646.4	42,646.4	42,679.8	564.32	18.85
Total	263,749.5	180,349.4	215,140.6	240,837.3	256,971.8	226,073.3	201,989.9	226,444.4	2,994.13	100.02

Table F3. Lake Harris-Little Lake Harris—Predicted lake trophic state variables

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90
Total P mean (mg/L)	0.026	0.019	0.024	0.026	0.028	0.025	0.023	0.024
Total P median (mg/L)	0.024	0.018	0.022	0.024	0.026	0.023	0.021	0.023
-1 standard error	0.016	0.012	0.014	0.016	0.017	0.015	0.014	0.015
+1 standard error	0.038	0.028	0.034	0.037	0.040	0.035	0.033	0.035
Total P (mg/L) (L&M)	0.052	0.084	0.071	0.054	0.063	0.075	0.074	0.067
Total P (mg/L) (C&B)	0.045	0.047	0.049	0.046	0.050	0.051	0.049	0.048
Total N mean (mg/L)	0.687	1.197	0.877	0.645	0.680	0.887	0.961	0.848
Total N median (mg/L)	0.658	1.145	0.840	0.618	0.651	0.849	0.919	0.812
-1 standard error	0.481	0.837	0.614	0.452	0.476	0.621	0.672	0.593
+1 standard error	0.900	1.567	1.149	0.846	0.891	1.162	1.258	1.111
Chlorophyll-*a* mean (µg/L)	6.978	10.848	8.532	6.547	7.017	8.724	9.266	8.273
Chlorophyll-*a* median (µg/L)	6.242	9.697	7.630	5.857	6.277	7.802	8.285	7.399
-1 standard error	3.766	5.850	4.603	3.533	3.787	4.706	4.998	4.463
+1 standard error	10.348	16.074	12.648	9.709	10.405	12.933	13.734	12.264
Secchi depth mean (m)	1.653	1.214	1.443	1.712	1.662	1.434	1.372	1.499
Secchi depth median (m)	1.539	1.131	1.343	1.593	1.548	1.335	1.278	1.395
-1 standard error	2.396	1.761	2.092	2.481	2.410	2.079	1.990	2.173
+1 standard error	1.062	0.780	0.927	1.099	1.068	0.921	0.881	0.963
TSI mean	45.527	48.202	46.912	44.790	45.903	47.387	47.679	46.629
TSI median	44.331	46.999	45.713	43.595	44.708	46.189	46.480	45.431
-1 standard error	37.179	39.847	38.561	36.443	37.556	39.037	39.328	38.279
+1 standard error	51.483	54.151	52.865	50.747	51.860	53.341	53.632	52.583

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

Appendix G—Estimated Nutrient Loading and Predicted Trophic State for Lake Eustis

Table G1. Lake Eustis—Phosphorus loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading $mg/m^2/yr$	% of Total
Residential low-medium density	634.1	507.3	1,395.5	1,502.8	1,642.3	1,512.7	1,335.6	1,218.6	38.82	4.95
Residential high density	3,487.9	2,790.5	1,013.0	1,090.9	1,192.2	1,098.1	969.5	1,663.2	52.98	6.76
Commercial	46.2	37.0	570.6	614.5	671.5	618.5	546.1	443.5	14.13	1.80
Institutional	47.1	37.7	74.3	80.1	87.5	80.6	71.2	68.4	2.18	0.28
Industrial/mining	4.9	3.9	68.7	74.0	80.9	74.5	65.8	53.2	1.70	0.22
Recreation/open	8.9	7.1	15.7	16.9	18.5	17.1	15.1	14.2	0.45	0.06
Forest/rangeland	42.0	33.6	80.7	86.9	94.9	87.4	77.2	71.8	2.29	0.29
Water/wetlands	1,110.2	888.2	1,111.8	1,197.3	1,308.4	1,205.1	1,064.1	1,126.4	35.88	4.58
Feedlots	11,304.8	9,044.4	458.2	493.5	539.3	496.7	438.6	3,253.6	103.65	13.22
Pasture	1,026.6	821.3	495.9	534.1	583.6	537.6	474.6	639.1	20.36	2.60
Citrus groves	366.4	293.1	122.4	131.8	144.1	132.7	117.2	186.8	5.95	0.76
Other agricultural	56.4	45.1	725.9	781.7	854.2	786.8	694.7	563.5	17.95	2.29
Muck farms	3,184.7	2,347.7	3,049.5	3,472.8	3,812.0	2,372.3	2,899.1	3,019.7	96.20	12.27
Septic tanks	235.7	235.7	235.7	235.7	235.7	235.7	235.7	235.7	7.51	0.96
Precipitation	409.9	327.9	399.0	429.7	469.6	432.5	381.9	407.2	12.97	1.65
Dry deposition	1,083.9	1,083.9	1,083.9	1,083.9	1,083.9	1,083.9	1,083.9	1,083.9	34.53	4.40
Lake Dora discharge	4,697.7	3,223.7	6,979.1	21,707.0	13,524.5	4,506.7	3,271.7	8,272.9	263.54	33.62
Lake Harris discharge	3,439.8	309.8	1,316.4	4,053.6	4,173.6	1,611.6	1,106.1	2,287.3	72.86	9.29
Total	31,187.2	22,037.9	19,196.3	37,587.2	30,516.7	16,890.5	14,848.1	24,609.0	783.95	100.00

Table G2. Lake Eustis—Nitrogen loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading $mg/m^2/yr$	% of Total
Residential low-medium density	4,840.4	3,872.5	10,652.2	11,471.5	12,536.2	11,546.8	10,195.2	9,302.1	296.33	2.30
Residential high density	16,474.9	13,180.6	4,784.8	5,152.8	5,631.0	5,186.6	4,579.5	7,855.7	250.26	1.94
Commercial	325.4	260.4	4,015.6	4,324.4	4,725.8	4,352.8	3,843.3	3,121.1	99.43	0.77
Institutional	370.4	296.3	584.8	629.8	688.3	634.0	559.8	537.6	17.13	0.13
Industrial/mining	29.9	23.9	416.9	448.9	490.6	451.9	399.0	323.0	10.29	0.08
Recreation/open	209.1	167.3	371.2	399.7	436.8	402.3	355.2	334.5	10.66	0.08
Forest/rangeland	990.2	792.2	1,902.1	2,048.4	2,238.5	2,061.9	1,820.5	1,693.4	53.95	0.42
Water/wetlands	9,855.8	7,885.0	9,869.8	10,628.8	11,615.4	10,698.7	9,446.4	10,000.0	318.56	2.47
Feedlots	30,249.4	24,200.8	1,226.1	1,320.4	1,443.0	1,329.1	1,173.5	8,706.0	277.34	2.15
Pasture	6,570.7	5,256.9	3,174.1	3,418.2	3,735.5	3,440.7	3,037.9	4,090.6	130.31	1.01
Citrus groves	5,364.9	4,292.2	1,792.6	1,930.5	2,109.6	1,943.1	1,715.7	2,735.5	87.14	0.68
Other agricultural	320.7	256.6	4,130.1	4,447.8	4,860.6	4,477.0	3,952.9	3,206.5	102.15	0.79
Muck farms	19,992.8	12,526.7	16,271.5	18,530.0	20,339.9	14,089.7	15,468.9	16,317.1	519.80	4.03
Septic tanks	4,161.2	4,161.2	4,161.2	4,161.2	4,161.2	4,161.2	4,161.2	4,161.2	132.56	1.03
Precipitation	19,776.6	15,822.2	19,253.7	20,734.4	22,658.9	20,870.7	18,427.7	19,649.2	625.95	4.85
Dry deposition	4,597.8	4,597.8	4,597.8	4,597.8	4,597.8	4,597.8	4,597.8	4,597.8	146.47	1.14
Lake Dora discharge	223,402.9	95,614.2	186,482.7	321,481.6	326,904.6	149,661.7	114,119.1	202,523.8	6,451.67	50.03
Lake Harris discharge	173,732.2	13,055.5	85,817.5	172,002.6	162,373.2	75,936.3	56,475.3	105,627.5	3,364.91	26.09
Total	518,265.3	206,262.3	359,504.7	587,728.8	591,546.9	315,842.3	254,328.9	404,782.6	12,894.91	99.99

Table G3. Lake Eustis—Predicted lake trophic state variables

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90
Total P mean (mg/L)	0.063	0.052	0.047	0.069	0.058	0.043	0.042	0.053
Total P median (mg/L)	0.058	0.048	0.043	0.063	0.053	0.039	0.039	0.049
-1 standard error	0.038	0.031	0.028	0.041	0.034	0.026	0.025	0.032
+1 standard error	0.090	0.075	0.067	0.098	0.082	0.061	0.060	0.076
Total P (mg/L) (L&M)	0.084	0.170	0.073	0.088	0.074	0.068	0.071	0.090
Total P (mg/L) (C&B)	0.074	0.092	0.064	0.079	0.069	0.060	0.059	0.071
Total N mean (mg/L)	1.068	1.506	1.098	1.024	1.029	1.038	1.098	1.123
Total N median (mg/L)	1.023	1.441	1.051	0.980	0.985	0.994	1.051	1.075
-1 standard error	0.748	1.053	0.768	0.716	0.720	0.726	0.768	0.786
+1 standard error	1.400	1.972	1.439	1.341	1.348	1.360	1.439	1.471
Chlorophyll-*a* mean (µg/L)	13.818	18.169	12.990	13.611	12.967	11.977	12.563	13.728
Chlorophyll-*a* median (µg/L)	12.358	16.241	11.616	12.173	11.597	10.711	11.234	12.276
-1 standard error	7.455	9.798	7.008	7.343	6.996	6.462	6.777	7.405
+1 standard error	20.485	26.923	19.256	20.179	19.225	17.756	18.623	20.350
Secchi depth mean (m)	1.260	1.041	1.241	1.290	1.287	1.280	1.241	1.234
Secchi depth median (m)	1.173	0.970	1.155	1.201	1.198	1.192	1.155	1.149
-1 standard error	1.827	1.510	1.799	1.871	1.866	1.857	1.799	1.790
+1 standard error	0.809	0.669	0.797	0.829	0.827	0.822	0.797	0.793
TSI mean	57.991	59.679	55.520	58.368	56.769	54.123	54.478	56.704
TSI median	56.793	58.477	54.322	57.171	55.572	52.925	53.280	55.505
-1 standard error	46.641	51.325	47.170	50.019	48.420	45.773	46.128	48.353
+1 standard error	63.945	65.629	61.474	64.323	62.724	60.077	60.432	62.658

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

APPENDIX H—ESTIMATED NUTRIENT LOADING AND PREDICTED TROPHIC STATE FOR LAKE GRIFFIN

Table H1. Lake Griffin—Phosphorus loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	446.3	357.1	1,335.8	1,438.5	1,572.0	1,447.9	1,278.5	1,125.2	29.54	2.97
Residential high density	2,344.7	1,875.9	445.2	479.5	524.0	482.6	426.1	939.7	24.67	2.48
Commercial	127.7	102.2	435.9	469.5	513.0	472.5	417.2	362.6	9.52	0.96
Institutional	20.5	16.4	98.7	106.2	116.1	106.9	94.4	79.9	2.10	0.21
Industrial/mining	48.4	38.7	134.1	144.5	157.9	145.4	128.4	113.9	2.99	0.30
Recreation/open	1.3	1.0	8.4	9.0	9.8	9.1	8.0	6.7	0.17	0.02
Forest/rangeland	124.6	99.7	110.5	119.0	130.1	119.8	105.8	115.6	3.04	0.31
Water/wetlands	1,385.0	1,108.1	1,287.8	1,386.8	1,515.6	1,396.0	1,232.6	1,330.3	34.93	3.52
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	1,136.3	909.1	941.0	1,013.3	1,107.4	1,020.0	900.6	1,004.0	26.36	2.65
Citrus groves	255.5	204.4	98.4	106.0	115.8	106.7	94.2	140.1	3.68	0.37
Other agricultural	103.0	82.4	534.9	576.1	629.5	579.8	512.0	431.1	11.32	1.14
Muck farms	24,302.0	19,140.0	21,268.4	24,411.1	27,090.4	22,201.1	18,258.5	22,381.6	587.61	59.18
Septic tanks	380.1	380.1	380.1	380.1	380.1	380.1	380.1	380.1	9.98	1.01
Citrus plant	212.1	212.1	212.1	212.1	212.1	212.1	212.1	212.1	5.57	0.56
Precipitation	497.3	397.9	484.2	521.4	569.8	524.8	463.4	494.1	12.97	1.31
Dry deposition	1,315.1	1,315.1	1,315.1	1,315.1	1,315.1	1,315.1	1,315.1	1,315.1	34.53	3.48
Haines Creek discharge	8,214.5	1,464.8	5,421.3	12,340.2	14,676.4	4,500.0	3,202.6	7,117.1	186.85	18.82
Lake Yale discharge	1,692.9	0.0	51.4	85.2	30.7	11.9	29.3	271.6	7.13	0.72
Total	42,607.3	27,705.0	34,563.3	45,113.6	50,665.8	35,031.8	29,058.9	37,820.8	992.96	100.01

Table H2. Lake Griffin—Nitrogen loading (kg/yr)

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	3,406.8	2,725.6	10,196.4	10,980.5	11,999.7	11,052.6	9,758.9	8,588.6	225.49	1.78
Residential high density	11,075.1	8,860.5	2,103.0	2,264.8	2,475.0	2,279.6	2,012.8	4,438.7	116.53	0.92
Commercial	898.9	719.2	3,067.8	3,303.7	3,610.3	3,325.4	2,936.2	2,551.6	66.99	0.53
Institutional	161.5	129.2	776.1	835.8	913.3	841.2	742.8	628.6	16.50	0.13
Industrial/mining	293.7	235.0	813.5	876.1	957.4	881.9	778.6	690.9	18.14	0.14
Recreation/open	29.9	23.9	197.2	212.4	232.1	213.8	188.7	156.9	4.12	0.03
Forest/rangeland	2,937.7	2,350.3	2,606.4	2,806.8	3,067.3	2,825.3	2,494.6	2,726.9	71.59	0.57
Water/wetlands	12,295.4	9,836.9	11,432.6	12,311.8	13,454.5	12,392.7	10,942.1	11,809.4	310.05	2.45
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	7,273.0	5,818.7	6,022.7	6,485.9	7,087.9	6,528.5	5,764.3	6,425.9	168.71	1.34
Citrus groves	3,741.7	2,993.5	1,441.4	1,552.3	1,696.4	1,562.5	1,379.6	2,052.5	53.89	0.43
Other agricultural	586.1	468.9	3,043.7	3,277.8	3,582.1	3,299.4	2,913.2	2,453.0	64.40	0.51
Muck farms	91,225.6	83,249.7	89,703.3	100,539.3	117,612.1	85,981.8	80,812.1	92,732.0	2,434.61	19.27
Septic tanks	6,709.4	6,709.4	6,709.4	6,709.4	6,709.4	6,709.4	6,709.4	6,709.4	176.15	1.39
Citrus plant	3,915.2	3,915.2	3,915.2	3,915.2	3,915.2	3,915.2	3,915.2	3,915.2	102.79	0.81
Precipitation	23,996.5	19,198.2	23,362.0	25,158.7	27,493.8	25,324.0	22,359.7	23,841.8	625.95	4.95
Dry deposition	5,578.9	5,578.9	5,578.9	5,578.9	5,578.9	5,578.9	5,578.9	5,578.9	146.47	1.16
Haines Creek discharge	381,576.4	61,816.2	211,064.1	469,530.9	627,868.6	193,781.8	140,609.4	298,035.3	7,824.71	61.93
Lake Yale discharge	42,214.2	0.0	2,519.4	3,667.9	2,149.4	2,175.0	2,872.5	7,942.6	208.53	1.65
Total	597,916.0	214,629.3	384,553.1	660,008.2	840,403.4	368,669.0	302,769.0	481,278.2	12,635.62	99.99

Table H3. Lake Griffin—Predicted lake trophic state variables

Land Use	1984	1985	1986	1987	1988	1989	1990	Mean 1984–90
Total P mean (mg/L)	0.083	0.097	0.103	0.105	0.118	0.110	0.105	0.103
Total P median (mg/L)	0.076	0.089	0.095	0.097	0.108	0.101	0.097	0.095
-1 standard error	0.049	0.058	0.061	0.063	0.070	0.066	0.063	0.061
+1 standard error	0.118	0.138	0.147	0.150	0.168	0.156	0.150	0.147
Total P (mg/L) (L&M)	0.063	0.215	0.142	0.112	0.130	0.156	0.128	0.135
Total P (mg/L) (C&B)	0.064	0.113	0.104	0.096	0.107	0.109	0.095	0.098
Total N mean (mg/L)	0.981	2.147	1.686	1.574	2.001	1.846	1.727	1.709
Total N median (mg/L)	0.939	2.055	1.614	1.507	1.916	1.768	1.654	1.636
-1 standard error	0.686	1.502	1.180	1.102	1.400	1.292	1.209	1.196
+1 standard error	1.285	2.812	2.209	2.063	2.622	2.419	2.263	2.239
Chlorophyll-*a* mean (µg/L)	13.810	30.655	24.789	23.350	30.385	27.560	25.526	25.154
Chlorophyll-*a* median (µg/L)	12.357	27.410	22.171	20.886	27.173	24.647	22.830	22.496
-1 standard error	7.454	16.535	13.375	12.599	16.392	14.868	13.772	13.571
+1 standard error	20.484	45.437	36.753	34.622	45.045	40.857	37.845	37.292
Secchi depth mean (m)	1.111	0.719	0.822	0.854	0.747	0.781	0.811	0.835
Secchi depth median (m)	1.034	0.669	0.765	0.795	0.696	0.728	0.755	0.777
-1 standard error	1.611	1.042	1.192	1.238	1.084	1.133	1.176	1.211
+1 standard error	0.714	0.462	0.528	0.549	0.480	0.502	0.521	0.536
TSI mean	59.682	68.891	67.092	66.598	70.013	68.579	67.522	66.911
TSI median	58.489	67.691	65.895	65.402	68.815	67.381	66.324	65.714
-1 standard error	51.337	60.539	58.743	58.250	61.663	60.229	59.172	58.562
+1 standard error	65.641	74.843	73.047	72.554	75.967	74.533	73.476	72.866

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of arsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

APPENDIX I—ESTIMATED NUTRIENT LOADING AND PREDICTED TROPHIC STATE FOR LAKE YALE

Table I1. Lake Yale—Phosphorus loading (kg/yr)

Land Use	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	Mean 1980–90	Mean Areal Loading $mg/m^2/yr$	% of Total
Residential low-medium density	139.9	120.1	205.9	174.8	147.9	118.3	310.0	333.9	364.9	336.1	296.7	231.7	14.24	3.86
Residential high density	327.3	281.0	481.7	409.0	346.0	276.8	261.4	281.5	307.7	283.4	250.2	318.7	19.59	5.31
Commercial	0.0	0.0	0.0	0.0	0.0	0.0	203.5	219.2	239.5	220.6	194.8	98.0	6.02	1.63
Institutional	0.0	0.0	0.0	0.0	0.0	0.0	132.2	142.4	155.6	143.3	126.5	63.6	3.91	1.06
Industrial/mining	35.2	30.2	51.8	44.0	37.2	29.8	172.2	185.4	202.6	186.6	164.8	103.6	6.37	1.73
Recreation/open	7.7	6.6	11.3	9.6	8.1	6.5	2.6	2.8	3.1	2.8	2.5	5.8	0.36	0.10
Forest/rangeland	128.9	110.7	189.7	161.1	136.3	109.0	115.4	124.3	135.9	125.1	110.5	131.5	8.09	2.19
Water/wetlands	1,373.4	1,179.2	2,021.4	1,716.2	1,451.8	1,161.5	1,359.6	1,464.1	1,600.0	1,473.7	1,301.2	1,463.8	89.99	24.39
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	924.6	793.9	1,360.9	1,155.5	977.4	782.0	617.6	665.1	726.9	669.5	591.1	842.2	51.78	14.03
Citrus groves	375.1	322.1	552.1	468.7	396.5	317.2	99.2	106.8	116.7	107.5	94.9	268.8	16.52	4.48
Other agricultural	28.1	24.1	41.4	35.1	29.7	23.8	1,197.0	1,289.0	1,408.6	1,297.5	1,145.6	592.7	36.44	9.87
Septic tanks	94.7	94.7	94.7	94.7	94.7	94.7	94.7	94.7	94.7	94.7	94.7	94.7	5.82	1.58
Citrus plants	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	1,007.6	61.94	16.79
Precipitation	200.9	172.5	295.7	251.1	212.4	169.9	206.8	222.7	243.3	224.1	197.9	217.9	13.40	3.63
Dry deposition	561.7	561.7	561.7	561.7	561.7	561.7	561.7	561.7	561.7	561.7	561.7	561.7	34.53	9.36
Total	5,205.1	4,704.4	6,875.9	6,089.1	5,407.3	4,658.8	6,341.5	6,701.2	7,168.8	6,734.2	6,140.7	6,002.3	369.00	100.01

Table 12. Lake Yale—Nitrogen loading (kg/yr)

Land Use	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	Mean 1980–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	1,067.9	916.9	1,571.8	1,334.5	1,128.9	903.1	2,366.5	2,548.5	2,785.0	2,565.2	2,265.0	1,768.5	108.72	3.26
Residential high density	1,546.0	1,327.4	2,275.4	1,931.9	1,634.2	1,307.4	1,234.8	1,329.8	1,453.2	1,338.5	1,181.8	1,505.5	92.55	2.78
Commercial	0.0	0.0	0.0	0.0	0.0	0.0	1,432.4	1,542.5	1,685.7	1,552.6	1,370.9	689.5	42.38	1.27
Institutional	0.0	0.0	0.0	0.0	0.0	0.0	1,040.0	1,120.0	1,224.0	1,127.4	995.4	500.6	30.78	0.92
Industrial/mining	213.6	183.4	314.3	266.9	225.8	180.6	1,044.2	1,124.5	1,228.8	1,131.8	999.4	628.5	38.64	1.16
Recreation/open	181.8	156.1	267.6	227.2	192.2	153.7	62.0	66.7	72.9	67.2	59.3	137.0	8.42	0.25
Forest/rangeland	3,040.4	2,610.6	4,475.0	3,799.5	3,214.0	2,571.3	2,722.8	2,932.2	3,204.3	2,951.4	2,605.9	3,102.5	190.73	5.72
Water/wetlands	12,192.2	10,468.5	17,944.7	15,236.0	12,888.0	10,311.0	12,069.5	12,997.7	14,204.1	13,083.1	11,551.7	12,995.1	798.87	23.98
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	5,918.1	5,081.4	8,710.3	7,395.5	6,255.8	5,004.9	3,953.1	4,257.2	4,652.3	4,285.1	3,783.5	5,390.7	331.39	9.95
Citrus groves	5,492.6	4,716.1	8,084.1	6,863.8	5,806.1	4,645.1	1,452.0	1,563.7	1,708.8	1,574.0	1,389.7	3,936.0	241.97	7.26
Other agricultural	159.9	137.3	235.4	199.8	169.0	135.2	6,810.7	7,334.5	8,015.2	7,382.6	6,518.5	3,372.6	207.33	6.22
Septic tanks	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	1,671.0	102.72	3.08
Citrus plants	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	5,602.9	344.44	10.34
Precipitation	9,695.0	8,324.3	14,269.2	12,115.3	10,248.2	8,199.1	9,977.3	10,744.6	1,741.9	10,815.2	9,549.2	10,516.3	646.49	19.40
Dry deposition	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	2,382.6	146.47	4.40
Total	49,164.0	43,578.5	67,804.3	59,026.9	51,418.7	43,067.9	53,821.8	57,218.4	61,632.7	57,530.6	51,926.8	54,199.3	3,331.91	99.99

Table I3. Lake Yale—Predicted lake trophic state variables

Land Use	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	Mean 1980–90
Total P mean (mg/L)	0.026	0.026	0.032	0.029	0.028	0.025	0.031	0.033	0.033	0.032	0.030	0.029
Total P median (mg/L)	0.024	0.024	0.029	0.026	0.025	0.023	0.028	0.030	0.030	0.029	0.028	0.027
-1 standard error	0.015	0.016	0.019	0.017	0.016	0.015	0.018	0.020	0.019	0.019	0.018	0.018
+1 standard error	0.037	0.038	0.046	0.041	0.039	0.035	0.044	0.047	0.046	0.045	0.043	0.042
Total P (mg/L) (L&M)	0.087	0.089	0.069	0.076	0.082	0.094	0.092	0.085	0.088	0.092	0.099	0.087
Total P (mg/L) (C&B)	0.056	0.055	0.056	0.057	0.056	0.055	0.061	0.061	0.063	0.062	0.062	0.059
Total N mean (mg/L)	0.812	0.964	0.611	0.665	0.787	0.968	0.779	0.724	0.696	0.761	0.864	0.785
Total N median (mg/L)	0.778	0.923	0.585	0.637	0.754	0.927	0.746	0.694	0.666	0.729	0.827	0.751
-1 standard error	0.568	0.675	0.428	0.465	0.551	0.677	0.545	0.507	0.487	0.533	0.604	0.549
+1 standard error	1.064	1.263	0.801	0.871	1.031	1.268	1.021	0.949	0.912	0.997	1.131	1.028
Chlorophyll-*a* mean (µg/L)	8.139	9.661	6.606	6.919	8.054	9.508	8.250	7.832	7.529	8.142	9.031	8.152
Chlorophyll-*a* median (µg/L)	7.279	8.638	5.910	6.189	7.203	8.501	7.379	7.006	6.735	7.282	8.076	7.291
-1 standard error	4.391	5.211	3.565	3.734	4.345	5.129	4.451	4.226	4.063	4.393	4.872	4.398
+1 standard error	12.066	14.319	9.796	10.260	11.941	14.093	12.231	11.613	11.164	12.072	13.388	12.086
Secchi depth mean (m)	1.519	1.381	1.780	1.699	1.547	1.378	1.555	1.619	1.656	1.575	1.469	1.562
Secchi depth median (m)	1.415	1.286	1.657	1.581	1.440	1.283	1.448	1.508	1.542	1.467	1.367	1.454
-1 standard error	2.203	2.003	2.581	2.463	2.242	1.999	2.255	2.348	2.401	2.284	2.130	2.264
+1 standard error	0.976	0.887	1.143	1.091	0.993	0.885	0.999	1.040	1.064	1.012	0.943	1.003
TSI mean	46.978	48.899	46.155	45.915	47.284	48.322	48.228	48.063	47.634	48.283	48.996	47.705
TSI median	45.780	47.699	44.961	44.719	46.086	47.122	47.030	46.866	46.438	47.086	47.798	46.508
-1 standard error	38.628	40.547	37.809	37.567	38.934	39.970	39.878	39.714	39.286	39.934	40.646	39.356
+1 standard error	52.932	54.851	52.113	51.871	53.238	54.274	54.182	54.018	53.590	54.238	54.950	53.660

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

Appendix J—Estimated Nutrient Loading and Predicted Trophic State for Lake Weir

Table J1. Lake Weir—Phosphorus loading (kg/yr)

Land Use	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	Mean 1980–90	Mean Areal Loading mg/m²/yr	% of Total
Residential *low-medium* density	1,072.1	909.1	1,579.8	1,544.9	817.1	890.3	811.7	893.7	975.9	894.6	714.0	1,009.4	44.36	28.90
Residential high density	222.8	188.9	328.4	321.1	169.8	185.0	13.5	14.9	16.3	14.9	11.9	135.2	5.94	3.87
Commercial	0.0	0.0	0.0	0.0	0.0	0.0	6.7	7.4	8.0	7.4	5.9	3.2	0.14	0.09
Institutional	12.3	10.5	18.2	17.8	9.4	10.2	9.3	10.3	11.2	10.3	8.2	11.6	0.51	0.33
Industrial/mining	0.0	0.0	0.0	0.0	0.0	0.0	11.8	13.0	14.1	13.0	10.3	5.7	0.25	0.16
Recreation/open	22.6	19.1	33.2	32.5	17.2	18.7	29.5	32.4	35.4	32.5	25.9	27.2	1.19	0.78
Forest/rangeland	24.6	20.9	36.3	35.5	18.8	20.4	28.8	31.7	34.6	31.7	25.3	28.0	1.23	0.80
Water/wetlands	295.4	250.5	435.4	425.7	225.2	245.3	372.5	410.1	447.8	410.5	327.6	349.6	15.36	10.01
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	326.8	277.1	481.6	470.9	249.1	271.4	137.8	151.8	165.7	151.9	121.3	255.0	11.21	7.30
Citrus groves	210.2	178.2	309.7	302.9	160.2	174.5	105.6	116.3	127.0	116.4	92.9	172.2	7.57	4.93
Other agricultural	8.2	7.0	12.1	11.8	6.3	6.8	298.8	328.9	359.2	329.3	262.8	148.3	6.52	4.25
Septic tanks	222.3	222.3	222.3	222.3	222.3	222.3	222.3	222.3	222.3	222.3	222.3	222.3	9.77	6.37
Point sources	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Precipitation	334.7	283.8	493.2	482.3	255.1	277.9	303.3	333.9	364.6	334.2	266.8	339.1	14.90	9.71
Dry deposition	785.7	785.7	785.7	785.7	785.7	785.7	785.7	785.7	785.7	785.7	785.7	785.7	34.53	22.50
Total	3,537.7	3,153.1	4,735.9	4,653.4	2,936.2	3,108.5	3,137.3	3,352.4	3,567.8	3,354.7	2,880.9	3,492.6	153.48	100.00

Table J2. Lake Weir—Nitrogen loading (kg/yr)

Land Use	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	Mean 1980–90	Mean Areal Loading mg/m²/yr	% of Total
Residential low-medium density	8,183.8	6,939.3	12,059.4	11,793.0	6,237.1	6,795.6	6,196.1	6,822.0	7,449.1	6,828.7	5,450.3	7,704.9	338.58	18.56
Residential high density	1,052.5	892.5	1,550.9	1,516.7	802.1	874.0	64.0	70.4	76.9	70.5	56.3	638.8	28.07	1.54
Commercial	0.0	0.0	0.0	0.0	0.0	0.0	47.0	51.8	56.5	51.8	41.4	22.6	0.99	0.05
Institutional	97.1	82.3	143.1	139.9	74.0	80.6	73.3	80.7	88.1	80.8	64.5	91.3	4.01	0.22
Industrial/mining	0.0	0.0	0.0	0.0	0.0	0.0	71.4	78.6	85.8	78.6	62.8	34.3	1.51	0.08
Recreation/open	531.9	451.0	783.8	766.5	405.4	441.7	694.9	765.1	835.5	765.9	611.3	641.2	28.18	1.54
Forest/rangeland	580.8	492.5	855.8	836.9	442.6	482.3	678.6	747.1	815.8	747.8	596.9	661.5	29.07	1.59
Water/wetlands	2,622.8	2,224.0	3,864.9	3,779.6	1,998.9	2,177.9	3,306.6	3,640.6	3,975.2	3,644.2	2,908.6	3,103.9	136.40	7.48
Feedlots	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Pasture	2,091.7	1,773.6	3,082.3	3,014.2	1,594.2	1,736.9	882.3	971.4	1,060.7	972.4	776.1	1,632.3	71.73	3.93
Citrus groves	3,077.7	2,609.7	4,535.2	4,435.1	2,345.6	2,555.7	1,546.5	1,702.7	1,859.3	1,704.4	1,360.4	2,521.1	110.79	6.07
Other agricultural	46.7	39.6	68.8	67.3	35.6	38.8	1,699.9	1,871.6	2,043.7	1,873.5	1,495.3	843.7	37.08	2.03
Septic tanks	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	3,924.0	172.43	9.45
Point sources	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00
Precipitation	16,149.1	13,693.3	23,796.8	23,271.2	12,307.7	13,409.8	14,632.9	16,110.9	17,592.0	16,126.8	12,871.5	16,360.2	718.93	39.41
Dry deposition	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	3,333.1	146.47	8.03
Total	41,691.2	36,454.9	57,998.1	56,877.5	33,500.3	35,850.4	37,150.6	40,170.0	43,195.7	40,202.5	33,552.5	41,513.0	1,824.26	99.98

Table J3. Lake Weir—Predicted lake trophic state variables

Land Use	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	Mean 1980–90
Total P mean (mg/L)	*0.007*	*0.006*	*0.010*	*0.010*	*0.004*	*0.006*	*0.007*	*0.008*	*0.008*	*0.007*	*0.005*	*0.007*
Total P median (mg/L)	0.007	0.006	0.009	0.009	0.004	0.006	0.006	0.007	0.007	0.007	0.004	0.007
-1 standard error	0.004	0.004	0.006	0.006	0.003	0.004	0.004	0.004	0.005	0.004	0.003	0.004
+1 standard error	0.010	0.009	0.014	0.014	0.006	0.009	0.010	0.011	0.011	0.010	0.007	0.010
Total P (mg/L) (L&M)	0.057	0.074	0.040	0.042	0.155	0.078	0.059	0.049	0.046	0.055	0.156	0.074
Total P (mg/L) (C&B)	0.033	0.033	0.033	0.034	0.033	0.033	0.032	0.031	0.032	0.032	0.032	0.033
Total N mean (mg/L)	0.540	0.792	0.348	0.356	1.739	0.828	0.627	0.494	0.459	0.573	1.929	0.790
Total N median (mg/L)	0.517	0.757	0.333	0.341	1.661	0.792	0.600	0.473	0.439	0.548	*1.842*	*0.755*
-1 standard error	0.378	0.554	0.243	0.249	1.214	0.579	0.438	0.346	0.321	0.401	1.346	0.552
+1 standard error	0.707	1.036	0.456	0.467	2.273	1.084	0.821	0.647	0.601	0.751	2.521	1.033
Chlorophyll-*a* mean (µg/L)	3.759	5.236	2.725	2.749	9.960	5.387	4.250	3.495	3.309	3.990	11.147	5.092
Chlorophyll-*a* median (µg/L)	3.360	4.678	2.437	2.459	8.888	4.812	3.799	3.125	2.959	3.567	9.945	4.548
-1 standard error	2.027	2.822	1.470	1.483	5.362	2.903	2.291	1.885	1.785	2.152	5.999	2.744
+1 standard error	5.570	7.755	4.040	4.076	14.733	7.977	6.297	5.180	4.905	5.913	16.485	7.539
Secchi depth mean (m)	2.324	1.878	2.968	2.928	1.213	1.832	2.139	2.441	2.544	2.248	1.145	2.151
Secchi depth median (m)	2.164	1.750	2.763	2.726	1.131	1.707	1.992	2.273	2.368	2.094	1.067	2.003
-1 standard error	3.370	2.725	4.303	4.245	1.761	2.659	3.102	3.541	3.689	3.261	1.663	3.120
+1 standard error	1.493	1.207	1.906	1.880	0.780	1.178	1.374	1.569	1.634	1.445	0.736	1.382
TSI mean	31.068	33.767	29.817	29.607	38.118	33.762	31.921	30.580	30.346	31.747	39.552	32.753
TSI median	29.866	32.559	28.622	28.411	36.897	32.553	30.717	29.379	29.146	30.544	38.329	31.547
-1 standard error	22.714	25.407	21.470	21.259	29.745	25.401	23.565	22.227	21.994	23.392	31.177	24.395
+1 standard error	37.018	39.711	35.774	35.563	44.049	39.705	37.869	36.531	36.298	37.696	45.481	38.699

All trophic state variables are predictions of EUTROMOD (Reckhow 1991), except Total P (mg/L) (L&M)-trophic state model of Larsen and Mercier (1976), with coefficients recalculated using Canfield and Bachmann's (1981) data set, and Total P (mg/L) (C&B)-trophic state model of Canfield and Bachmann (1981).

Appendix K—Land Uses in Contributing Subbasins of the Upper Ocklawaha River Basin, From 1984 and 1987–89 Land Use Maps

Table K1. Land uses in the upper Ocklawaha River Basin contributing subbasins (in ha), from 1984 land use maps

Land Use Code	Land Use	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin 1980-82	Lake Griffin Subbasin 1983-84	Lake Yale Subbasin	Lake Weir Subbasin
110	Residential low density	0.36	118.85	17.78	68.30	21.34	21.34	0.00	352.13
120	Residential medium density	79.99	124.21	857.61	556.63	419.78	419.78	122.71	517.31
130	Residential high density	0.00	633.53	164.63	1,535.11	1,053.21	1,053.21	104.13	73.63
140	Commercial and services	0.00	0.00	8.92	25.29	51.91	51.91	0.00	0.00
150	Industrial	0.00	22.66	36.95	0.00	27.59	27.59	13.72	0.00
160	Extractive	0.00	0.00	52.95	0.00	17.81	17.81	6.73	0.00
170	Institutional	0.00	1.62	28.48	42.24	19.91	19.91	0.00	9.83
180	Recreational	0.00	3.08	101.94	51.91	30.08	30.08	55.85	0.00
190	Open land	0.00	10.00	17.67	51.15	0.00	0.00	0.00	320.96
210	Cropland and pastureland	302.67	0.02	365.10	382.41	8.98	8.98	36.75	9.02
211	Improved pasture	96.99	101.91	3,225.65	1,211.74	1,872.07	1,803.25	939.24	373.50
214	Row crops	0.00	0.00	0.00	0.00	1,640.54	1,460.49	0.00	0.00
221	Citrus groves	1,423.58	681.35	8,422.55	1,687.62	962.64	962.64	1,617.81	678.77
230	Feeding operations	0.00	0.00	6.19	72.69	0.00	0.00	0.00	0.00
240	Nurseries and vineyards	0.00	0.00	0.00	1.70	0.00	0.00	0.00	0.00
329	Grassy scrub	309.26	120.74	310.91	11.64	167.67	167.67	202.65	15.55
411	Pine flatwood	88.39	53.35	249.78	296.58	616.12	616.12	745.53	86.85
412	Sandhill community	0.00	0.00	10.70	51.15	19.55	19.55	4.40	107.74
413	Sand pine scrub	0.00	0.00	13.41	0.00	0.00	0.00	30.15	0.00
421	Xeric hammock	0.00	0.00	0.00	26.94	79.16	79.16	0.00	0.00

Table K1—*Continued*

Land Use Code	Land Use	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin 1980-82	Lake Griffin Subbasin 1983-84	Lake Yale Subbasin	Lake Weir Subbasin
427	Mesic hammock	378.05	159.29	869.56	140.59	425.41	425.41	70.57	35.32
441	Planted pine	0.00	0.00	31.55	5.89	0.00	0.00	67.95	10.11
443	Clear-cut areas	0.00	12.58	85.74	20.07	22.60	22.60	0.00	0.00
510	Streams and waterways	0.00	0.00	0.00	0.00	4.05	4.05	0.00	0.00
520	Lakes and ponds	883.06	1,873.40	8,209.05	3,796.21	3,877.35	3,877.35	2,230.72	2,404.38
611	Bay swamps	73.83	642.74	833.22	814.22	592.14	592.14	148.38	92.68
615	Stream and lake swamps	46.14	0.00	1,763.44	196.30	374.16	374.16	14.29	13.22
617	Hydric hammock	79.20	0.00	65.50	0.00	25.62	25.62	273.77	0.00
621	Cypress forest	0.00	0.00	343.75	78.86	0.00	0.00	3.28	0.00
641	Freshwater marsh	3.27	50.95	2,702.34	538.56	1,104.84	1,353.72	1,443.57	123.45
643	Wet prairies	0.00	0.00	177.83	0.00	0.00	0.00	25.17	0.00
742	Borrow areas	0.00	14.01	54.31	4.11	0.00	0.00	0.00	0.00
800	Utilities and communications	0.00	0.00	214.05	0.00	0.00	0.00	0.00	0.00
810	Transportation	0.00	0.00	120.65	0.00	10.69	10.69	0.00	0.00
	Total hectares	3,764.79	4,624.29	29,362.21	11,667.91	13,445.22	13,445.23	8,157.37	5,224.45

Table K2. Land uses in the upper Ocklawaha River Basin contributing subbasins (in ha), from 1987–89 land use maps

Land Use Code	Land Use	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin	Lake Yale Subbasin	Lake Weir Subbasin
110	Residential low density	250.53	269.16	678.03	285.87	482.88	108.71	583.56
120	Residential medium density	94.67	443.46	1,156.98	1,210.44	1,027.36	161.91	151.71
130	Residential high density	0.00	153.66	234.53	420.97	181.11	84.21	4.94
140	Commercial and services	0.00	184.86	337.16	251.19	244.29	50.66	3.56
150	Industrial	8.87	34.65	93.77	38.22	93.45	51.91	0.00
160	Extractive	0.00	0.00	125.20	10.33	24.87	108.29	9.34
170	Institutional	1.31	17.01	77.12	76.27	100.38	85.17	8.03
180	Recreational	42.26	17.02	119.14	97.08	53.09	0.00	11.68
190	Open land	25.12	101.64	96.57	108.76	77.48	1.22	317.57
210	Cropland and pastureland	0.00	0.00	45.83	0.00	0.00	0.00	0.00
211	Improved pasture	168.22	19.55	1,585.59	341.34	929.18	408.92	161.10
212	Unimproved pasture	0.00	1.51	165.84	199.08	136.30	177.45	8.79
213	Woodland pasture	6.86	0.00	156.89	49.97	124.21	62.02	32.21
214	Row crops	204.84	1.21	332.67	376.41	1,623.65	5.89	6.94
215	Field crops	40.61	8.34	237.04	111.02	63.27	107.14	98.22
216	Mixed crops	30.52	12.20	25.32	3.61	0.00	13.68	50.23
221	Citrus groves	627.81	181.06	1,919.37	562.45	377.67	376.57	378.84
223	Other groves	0.00	0.00	0.00	0.00	0.00	0.00	13.10
224	Abandoned tree crops	249.08	129.54	2,954.60	310.13	161.95	189.25	256.37
232	Poultry feeding operations	0.00	0.00	0.00	3.22	0.00	0.00	0.00
240	Nurseries and vineyards	5.44	0.00	31.87	3.52	41.38	0.79	0.00
241	Tree nurseries	0.00	0.00	1.57	0.00	0.00	0.00	0.00
243	Ornamentals	0.01	0.00	74.20	6.62	0.00	20.60	1.40

Table K2—*Continued*

Land Use Code	Land Use	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin	Lake Yale Subbasin	Lake Weir Subbasin
245	Floriculture	3.35	0.00	11.38	2.23	0.00	1.93	0.00
251	Horse farms	26.89	6.75	36.14	0.00	0.00	7.44	0.00
260	Rural open lands-agriculture	35.42	0.00	34.26	0.00	0.00	0.00	2.40
261	Fallow cropland	236.09	199.04	2,420.43	836.58	439.86	1,122.59	163.91
310	Herbaceous rangeland	4.55	2.92	85.46	0.54	0.00	0.34	0.00
320	Shrub and brushland	55.39	73.66	324.65	50.88	46.97	89.31	8.40
330	Mixed rangeland	47.38	65.48	235.19	154.82	230.91	68.68	0.00
410	Upland coniferous forest	61.74	101.25	276.32	252.75	196.17	231.95	1.25
420	Upland hardwood forest	0.00	0.00	10.76	5.51	0.00	74.91	42.38
430	Upland hardwood forest continued	182.77	61.25	818.83	355.87	565.95	408.90	108.39
440	Tree plantations	4.07	0.00	0.00	8.55	16.98	0.00	3.72
510	Streams and waterways	0.00	0.70	40.44	12.37	91.20	2.19	5.97
520	Lakes and ponds	861.86	1,909.62	7,877.89	3,708.63	3,889.56	2,226.13	2,359.10
530	Reservoirs	47.94	15.42	195.12	49.15	46.92	19.00	3.30
610	Wetland hardwood forest	0.00	0.00	0.00	0.00	0.00	0.00	7.87
611	Bay swamps	51.36	360.55	479.17	265.78	225.65	187.17	20.51
615	Stream and lake swamps	95.48	18.98	1,051.28	104.17	147.63	0.00	0.00
620	Wetland coniferous forest	10.32	2.26	26.85	50.21	20.07	11.39	0.00
621	Cypress forest	0.00	0.00	43.98	73.62	14.01	39.40	0.00
630	Wetland forest mixed	90.45	50.49	1,490.34	315.57	365.11	118.82	144.68

Table K2—*Continued*

Land Use Code	Land Use	Lake Beauclair Subbasin	Lake Dora Subbasin	Lake Harris Subbasin	Lake Eustis Subbasin	Lake Griffin Subbasin	Lake Yale Subbasin	Lake Weir Subbasin
640	Vegetated nonforested wetland	0.00	0.00	0.00	0.00	0.15	0.00	0.00
641	Freshwater marsh	40.99	70.22	1,705.32	557.04	405.65	1,061.89	70.79
643	Wet prairies	18.54	12.05	424.75	51.70	60.17	96.64	1.85
644	Emergent aquatic vegetation	11.02	29.57	269.61	42.04	32.41	58.00	49.49
645	Submerged aquatic vegetation	0.00	0.00	0.00	0.00	0.00	5.42	0.00
646	Mixed scrub-shrub wetland	94.83	32.76	681.47	201.44	845.21	265.58	71.79
710	Non-swimming beaches	0.00	0.00	0.00	0.00	0.00	0.00	60.32
740	Rural disturbed land	3.07	0.00	42.44	4.36	2.32	15.76	0.77
742	Borrow areas	0.00	0.00	0.00	3.41	6.55	0.00	0.00
743	Spoil areas	18.06	0.00	4.18	9.36	0.00	0.00	0.00
810	Transportation	6.04	31.77	315.39	76.85	39.72	17.47	0.00
820	Communications	0.00	0.00	3.62	0.00	3.38	0.00	0.00
830	Utilities	1.12	4.63	7.73	7.93	10.29	12.10	0.00
	Total hectares	3,764.88	4,624.24	29,362.29	11,667.86	13,445.36	8,157.40	5,224.48