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John McCary
University of South Florida

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Incorporating Surficial Aquifer Ground-Water Fluxes
Into Surface-Water Resource Management Studies

by

John McCary

A thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science in Civil Engineering
Department of Civil and Environmental Engineering
College of Engineering
University of South Florida

Major Professor: Mark Ross, Ph.D.
Mark Stewart, Ph.D.
Xinjian Chen, Ph.D.

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DEDICATION

This is dedicated to all of those who have supported me in every venture throughout my life. I have been encouraged by many family and friends, and I would like to thank all of them collectively as I cannot thank all of them individually. In particular, to my wife and son: for being patient throughout this long, arduous, and sometimes painful journey, always encouraging me to finish; and to my father: for being a selfless friend, parent, and role model, always leading by example through hard work and helping others while expecting nothing in return. Thank you all as I could not have completed this without you!

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INCORPORATING SURFICIAL AQUIFER GROUND-WATER FLUXES INTO SURFACE-WATER RESOURCE MANAGEMENT STUDIES

John McCary

ABSTRACT

For surface-water resource management studies, it is important to quantify all of the mechanisms that contribute to water quantity and influence water quality. In this regard, various methods have been used to ground-water fluxes in lake systems. These have included physical measurements (e.g., seepage meters), flow-net analyses, water budgets, chemical tracers, ground-water flow models, and statistical analyses. The method developed for this study for calculating ground-water inflow uses a simplified, 1-layer (surficial aquifer) ground-water flow model. The test area was on a set of lakes known as the Winter Haven Chain of Lakes in Polk County, Florida. The technique combines the use of a numerical model (MODFLOW) with an inverse prediction technique (PEST) to determine net surficial recharge rates. Within the model, the lakes were represented as constant-head boundaries. A general, surficial ground water no-flow boundary was delineated around the entire lake system based on the topographic boundaries. The model used annual average lake elevations to create a constant-head boundary for each lake for each year. Annual average elevations of surficial well heads were used as target well data. Model results generally support previous studies in the region, concluding that the lake chain receives significant inflow from the surficial aquifer and leaks to the Floridan aquifer. As a consequence, ground-water quality constituency was found to be of critical importance. One of the most important observations from this study is the need for accurate ground-water concentrations for ridge lake water quality management. The initial measured values used in this study were highly variable, uncertain, and likely underestimated the effect that ground water has on nutrient loading to the Winter Haven Chain of Lakes. A more accurate spatial representation of ground-water concentrations is needed to better approximate the effect that ground-water loading has on the system. The technique used in this study whereby land-use based ground-water concentrations were developed from additional measured data appears to give a better representation of overall ground-water concentrations.

CHAPTER 1: INTRODUCTION

Characterizing the exchange of ground water with lakes is an important concern in water quality management of surface-water features. In Florida for example, interaction between surface water and ground water is an important part of the hydrologic cycle. A recent study of the Winter Haven Chain of Lakes (WHCL) in Florida shows the need to account for both volume and nutrient loading from the surficial aquifer to the lakes (McCary and Ross, 2005). This manuscript describes a process used to account for surficial ground-water loads to a surface-water body, considering the WHCL as a case study. Incorporated into this study is the importance of accounting for the spatial variability in ground-water nutrient concentrations.

The WHCL, shown in Figure 1, consists of 21 interconnected lakes within and around the City of Winter Haven, in north-central Polk County, Florida. The WHCL is split into two regions, the Southern Chain and the Northern Chain. The Southern Chain consists of 16 lakes: Winterset, Eloise, Summit, Lulu, Roy, Shipp, May, Howard, Cannon, Blue, Mirror, Spring, Idylwild, Jessie, Mariana, and Hartridge. The Northern Chain consists of 5 lakes: Conine, Rochelle, Haines, Smart, and Fannie. These lakes constitute a combined surface-water area of over 7,000 acres (refer to Table 1), or 11 square miles, and a watershed area of approximately 32 square miles (Polk County, 2002). The WHCL exist on a ridge in west-central Florida known as the Winter Haven Ridge, which is in the sand hills and ridges of Florida's Central Lakes District (Brooks, 1981). In these ridges, rainfall quickly percolates through the permeable sands to the water table, favoring ground-water flow over surface-water flow in pervious areas. Like many of Florida's lakes, the WHCL are situated in a layer of sand and clay that blankets an extensive and highly productive limestone aquifer, the Upper Floridan (Lee, 2002).

In the 1920s and 1930s, a series of canals were excavated in the WHCL in order to connect the lakes with navigable waterways. This substantially changed the natural hydrology and water quality of the lakes (SWFWMD, 1998). Prior to the construction of the canals, many of the lakes in the WHCL had small, isolated drainage basins. During this period, they were most likely one of the many seepage lakes in the state of Florida. Currently, about 70 percent of Florida's 7,800 lakes are seepage lakes, having no significant natural surface flow into or out of them (Palmer, 1984; Sacks et al., 1998). Heavy urbanization continued in the watersheds and the result was a deterioration of lake water quality. One of the areas of particular concern to the WHCL is the contribution of ground water, both in volume and nutrient loading. Citrus and phosphate industries are common in the area, making the quantification of ground-water inflow important in areas where nutrient-rich ground water can flow to the lakes. The need to account for the ground-water loadings to these lakes is the driving force behind the development of the process discussed in this manuscript and the consideration of the WHCL as a case study.

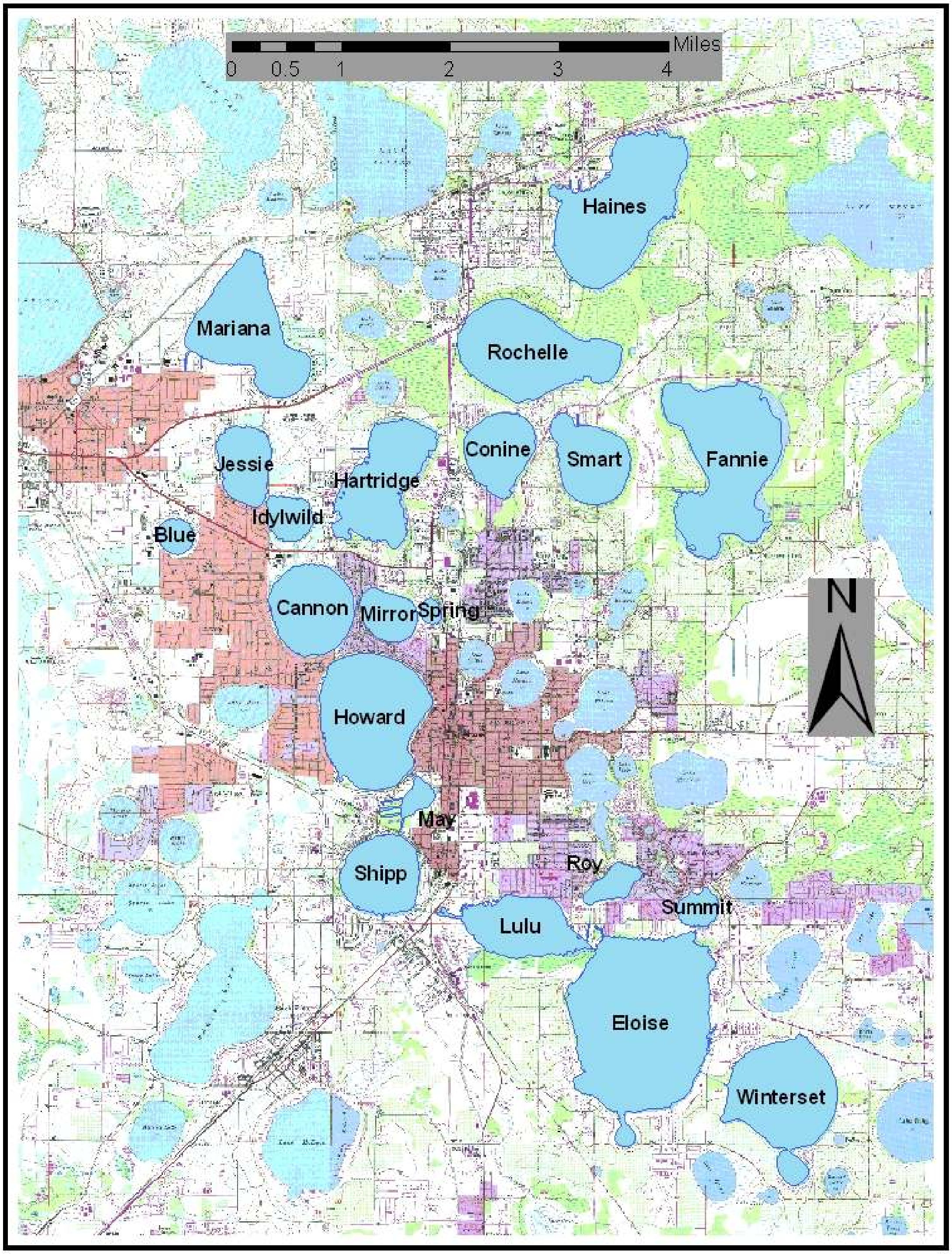

Figure 1: Winter Haven Chain of Lakes

Table 1: Winter Haven Chain Lake Areas

Lake Name	Area (ac)
Winterset	548
Eloise	1163
Summit	64
Lulu	307
Roy	74
Shipp	277
May	52
Howard	625
Cannon	328
Blue	54
Mirror	126
Spring	23
Idylwild	93
Jessie	186
Mariana	511
Hartridge	415
Conine	233
Rochelle	573
Haines	689
Smart	274
Fannie	738
Total	7351

1.1 Statement of the Problem

For lake/surface-water resource management studies, it is essential to quantify all of the mechanisms that generate water quantity and influence water quality. Traditionally, many hydrologic modeling applications to nutrient loading studies for surface-water bodies have attributed little influence to ground-water fluxes, which has been shown in several total maximum daily load (TMDL) studies (Vieux and Moreda, 2003; Vondracek et al., 2003; Dalzell et al., 2004) and may be an accurate assessment in some geologic and hydrologic conditions. In the WHCL area, there have been several studies with conflicting assumptions about the influence of ground water (Dames &

Moore, 1994; PBS&J, 2004). If ground water is assumed to be negligible when it is in fact significant, other factors that influence water quantity and quality will be calculated incorrectly in order to compensate for the lack of ground water. From a surface-water modeling standpoint, this usually involves increasing the impervious area and/or decreasing the soil infiltration values to unreasonable numbers to generate sufficient water quantity loading via runoff. Water quality is then accounted for by adjusting runoff concentration values within certain land-use types or regions to meet the loading requirements necessary to match the observed surface-water quality. While "calibrated" model results may appear to give values that are similar to the actual system, all of the mechanisms that influence the system are not accounted for, and those that are accounted for are incorrectly quantified. The model reliability as a predictive tool becomes poor. This makes it difficult or impossible to properly manage water quality from runoff mechanisms alone in regions heavily influenced by ground-water fluxes. In west-central Florida, it is also possible for evaporation losses to exceed precipitation. Because of this, net ground-water flow (inflow – outflow) can be very important for sustaining or depleting lake stage (Swancar et al., 2000; Lee, 2002). Properly accounting for ground-water contributions is an essential step in ridge lake management studies.

Even when ground water does not have a significant contribution to a lake's water budget, it can still heavily influence water quality. Solutes that originate in the shallow ground-water system can occur naturally from geologic or atmospheric sources (Stauffer, 1985; Baker et al., 1986; Pollman et al., 1991) or can be derived from anthropogenic sources such as septic tank leachate or fertilizers (Fellows and Brezonik, 1981; Stauffer, 1991; Tihansky and Sacks, 1997). In west-central Florida for example, many of the lake basins have been developed, and it is possible for ground water to become enriched in major ions and nutrients because of anthropogenic sources. If this enriched ground water flows into a lake, it can have a significant effect on the water quality of the lake. It is therefore necessary to quantify both the ground-water volume flux and nutrient flux in order to accurately determine lake water budgets and mass balances (Sacks et al., 1998).

Recent studies have focused on new techniques for estimating ground-water inflows to surface-water bodies. Many of these involve mass-balance approaches that use various environmental chemical tracers. While these studies have shown promise for the future, the historical data are not always available and the accuracy of the results is often highly variable. Traditional methods, including detailed, multi-layer flow models and detailed water budgets, can involve large amounts of time and money. In both the new and traditional techniques, unreasonable or unjustified assumptions can often still be made about the physical system in order to develop results. There is a need for a technique to approximate ground-water inflows to surface-water bodies using traditionally collected well data and aquifer parameters, as well as a method to account for the spatial variability in ground-water nutrient concentrations in order to be used in determining ground-water loading.

1.2 Background

Because of the need to account for ground-water fluxes in many lake systems, various methods have been used. These have included physical measurements (e.g., seepage meters), flow-net analyses, water-budget and mass-balance approximations, chemical tracers, ground water flow models, and statistical analyses. All of these methods must also incorporate some technique to determine accurate nutrient concentrations in order to approximate nutrient loadings.

In the past, seepage meters were used for localized data needs because they measure the amount of inflow, or seepage, at a given point over time. However, recent studies have shown that they to not provide accurate results. They also cannot be used for larger areas because they rely on point measurements, and these measurements must be extrapolated spatially and temporally in order to represent a larger area (Fellows and Brezonik, 1980; Belanger and Montgomery, 1992). In addition, they are only used to determine inflows. If a system does not receive any ground-water inflow, then an alternative technique would give more information, such as determining net ground-water flows through a water budget or a using a simplified ground-water model that could still give an approximation of ground-water outflow.

A flow-net is a simplified ground-water flow model that estimates or approximates steady-state conditions in order to determine ground-water flows to a surface-water body. In general, a flow-net is set up as a two-dimensional transect that represents a flow path from a ground-water flow divide to a constant-head boundary, such as a lake. In addition to a constant-head boundary and a ground-water flow divide, a flow-net analysis requires the aquifer parameters needed for the simulation of steady-state conditions, which are the depth of the flow field and the lateral hydraulic conductivity, but it does not require the aquifer parameters that are needed for volume storage/change used in transient models, which are the porosity and the specific yield.

This type of analysis can be beneficial if limited aquifer data are known, particularly if the limited data includes estimates of aquifer depth and hydraulic conductivity. It is also beneficial because it does not directly need to account for other hydrologic processes, such as surface-water flows, that could influence a transient model in cases where volume and elevation changes in the lake need to be considered. This technique yields good results when head data are representative of the entire study period, since it uses a steady-state approximation with no change in volume. For this reason, this is not a technique used to account for highly transient variables, such as the water table. In addition to the limits imposed by the steady-state approximation, another limit of using flow-nets involves the accurate delineation of the ground-water flow basin, which can be difficult for systems that have irregular shaped basin boundaries and/or possible flow-thru regions within the lake's ground-water flow basin.

Water budgets have been used extensively to account for net ground-water flows. They are typically used to determine net ground-water flows as the residual to the water budget as shown below:

$$\Delta G = E - P + S_o - S_i + \Delta V$$

where ΔG = net ground-water flow (inflow – outflow), P = precipitation, E = evaporation, S_i = surface-water inflow, S_o = surface-water outflow, and ΔV = change in volume. The problem with this technique is that it does not give a direct approximation of either ground-water inflow or outflow, only the difference between the two. Therefore, one of the terms must be approximated in order to come up with the other. These approximations usually involve assumed values for the hydrology and/or geology of the system. One example is the assumption that the most negative monthly net flow symbolizes only ground-water outflow (i.e. leakage) and does not vary substantially when compared with ground-water inflow. This is usually a period of dry weather, and it is assumed that there is little or no ground-water inflow. However, even if ground-water outflow does remain relatively constant, this assumption could still underestimate both ground-water inflow and outflow. It is still possible during dry periods to have surficial head gradients toward the lakes, resulting in ground-water inflow. Therefore, it is necessary to determine aquifer head gradients using observation wells or some other technique in order to make the assumption that there is no ground-water inflow. Incorrectly assuming that dry periods result in no ground-water inflow will perpetuate the errors encountered when estimating ground-water loading and nutrient concentrations. If the ground-water terms in the water budget are solved for incorrectly, then the ground-water nutrient concentrations will be “calibrated” to account for an incorrect nutrient loading value.

Walker and Havens (2002) used a simple numerical approximation to account for ground water in order to develop a method that could be used to determine a TMDL for phosphorus to Lake Istokpoga, Florida. They were able to measure surface-water inflows and outflows via gauged channel structures since a majority of the surface-water flows were channelized prior to interaction with the lake. The remaining terms of the water-budget equation could be accounted for, except for ground-water flow. Because they felt that ground-water outflow could be ignored based on the geology of the system, the only remaining term in the water budget equation was net ground-water inflow. An additional ungauged inflow value was included to reflect ground-water seepage from areas surrounding the lake. This ungauged inflow was approximated as a percentage of the gauged surface-water inflow in order to satisfy the water budget. The same approach was used in determining ground-water loading based on a mass-balance equation. Because all other loads and concentrations could be measured, ground-water loading/concentration could be calculated as the residual to the mass balance. For this system, this approximation may work well if the assumption about ground-water outflow is reasonable. However, in a system where ground-water outflow could be significant, or where surface-water flows could not be easily measured, these approximations could not be made and another technique must be used to account for these values.

PBS&J (2004) performed a detailed water-budget and mass-balance analysis on two of the lakes within the WHCL for one year. They solved the water budget equation for net ground-water flow and used Darcy's law to approximate the ground-water inflow term. Using Darcy's Law requires similar data as a flow-net analysis, including lateral hydraulic conductivity, head gradients, and contact surface area between the aquifer and the lakes. However, it could be difficult to approximate the variation in head gradients within the lake's ground-water flow basin using this method alone. Ground-water nutrient fluxes to the lakes were accounted for by multiplying the results of the volume flux by the average surficial-well nutrient concentrations for the basin.

Environmental chemical tracers have been used in techniques to quantify both surface-water and ground-water inflows to a system. Cimino (2003) has shown that stream-flow conductivity data can be used in non-urbanized environments to separate stream flows into surface-water and ground-water inflows. During periods of heavy runoff, stream conductivity decreases significantly. Alternatively, stream conductivity is very high during dry periods where streams are only receiving ground water. Using these two extremes as linear bounds, the segregation of stream water into surface water and ground water can be approximated based on its conductivity. This technique is much more difficult to use in urbanized environments due to elevated conductivity values of certain urban surface discharge sources including roadway runoff, fertilizer runoff, industrial discharges, sewage discharge, etc. This technique would also be difficult to use for a stagnant water body, such as a lake, because it would be difficult to measure the extreme bounds due to other important long-term effects, such as precipitation, evaporation, and leakage.

Sacks (2002) used the isotope mass-balance approach to estimate ground-water inflow to 81 lakes in the central highlands and coastal lowlands of central Florida. The isotope mass-balance approach uses the tracers Deuterium and Oxygen-18, which are naturally occurring stable isotopes of the water molecule. These isotopes are excellent conservative tracers because they are part of the water molecule itself, rather than dissolved constituents that may undergo reactions and dispersion. The difficulty with using the tracers, as with most tracers, is correctly measuring or quantifying all of the significant sources. In general, the approach combines the water-budget and mass-balance equations below:

$$\text{Water Budget:}\quad \Delta V = P - E + S_i - S_o + G_i - G_o$$

$$\text{Mass Balance:}\quad \Delta(V\delta_L) = P\delta_P - E\delta_E + S_i\delta_i - S_o\delta_L + G_i\delta_{Gi} - G_o\delta_L$$

where δ represents the isotopic composition for each water-budget term. Sacks' study focused on lakes with little or no surface-water drainage (seepage lakes) in order to eliminate the effect of uncertainty in surface-water flows and isotopic composition on calculated ground-water inflow. By eliminating the surface-water terms and rearranging the water-budget equation to solve for ground-water outflow, the water-budget equation can be substituted into the mass-balance equation to give the following formula for determining ground-water inflow:

$$G_i = \frac{P\delta_P - E\delta_E - (P-E)\delta_L}{\delta_L - \delta_{Gi}}$$

The study showed that in west-central Florida seepage lakes, computed ground-water inflow was most sensitive to uncertainty in variables used to calculate the isotopic composition of evaporating lake water. Sacks notes that the isotope mass-balance approach was most successful for lakes that have higher ground-water inflows, are deeper, and undergo less isotopic variability. The isotope mass-balance approach is better used to distinguish whether ground-water inflow quantities fall within a certain range of values, rather than for precise quantification. It is well suited for estimating ground-water inflows to lakes in geographic proximity, and by doing this, ground-water inflows can be extrapolated to lakes in the region that have similar geologic features but may also have significant surface-water flows. Sacks also notes that for multiple lakes, the technique is most successful when coupled with detailed data collection at a lake with a known water budget. However, it is important to note that this technique would not work directly with a lake that is influenced by surface-water flows unless these flows and their corresponding isotopic compositions were accounted for.

Detailed ground-water flow models or integrated surface-water/ground-water models can be the best methods to quantify ground-water flows in lake basins. However, these often lack sufficient data to properly parameterize the models. For instance, a detailed, multi-layered, transient ground-water flow model would require many aquifer parameters, including depths, vertical and lateral hydraulic conductivities, porosities, and specific yields for each layer. It would also require detailed transient water-budget terms, such as ground-water heads, lake stages, rainfall, and evapo-transpiration (ET). These parameters are not always readily available and can take large amounts of time and money to collect. If these data, along with high-resolution rainfall/recharge data, are available, then detailed modeling studies can provide good estimates for ground-water flows. High-resolution rainfall/recharge data are especially important in shallow water-table environments because long time steps can cause low ground-water volume predictions. These low predictions occur because recharge stresses and ground-water flow responses during rainy periods are averaged over too long a time period. When inflow is underestimated, leakage is also underestimated because inflow and leakage are correlated if lake stage is maintained over the long term. To simulate the total ground-water inflow to lakes, saturated-flow models of lake basins need to account for the potential effects of rapid and efficient recharge in the surficial aquifer system closest to the lake or other shallow water-table areas. In this part of the basin, the ability to accurately estimate recharge is crucial because the response time between rainfall and recharge is shortest (Swancar and Lee, 2003).

Statistical analyses can be coupled with ground-water applications in various ways. One way statistics can be used in ground-water applications is through the use of inverse modeling techniques, such as a nonlinear least-squares regression, to automatically adjust an unknown model parameter value in order to give the best prediction of a known result. Modeling calibration has traditionally been accomplished by the manual trial-and-error approach during which the modeler iteratively selects parameter values to improve the model results. However, a calibration obtained using this approach alone does not guarantee the statistically best solution. Inverse modeling techniques have been shown to improve the quality of ground-water models and yield results that are not readily available through trial-and-error calibration efforts alone (Poeter and Hill, 1996; Yobbi, 2000; Trout, 2002). This technique can be useful in steady-state predictions, because surficial ground-water heads are directly proportional to the ratio of recharge to lateral hydraulic conductivity. Therefore, if one of these values is known, an inverse modeling technique can be used to achieve the best approximation of the other value in order to match target data. Trout (2002) used the parameter estimation program, PEST (Doherty, 2001), to automatically adjust aquifer parameters within specified zonal boundaries to match target well data. Areas that had common residuals compared to target well data were initially grouped together in a common zone. After PEST was run on the model, zones that had overlapping confidence intervals could be combined to reduce the number of total zones within the model. This procedure was repeated until the best statistical model fit was achieved. This study showed the benefit of using an inverse modeling technique for simplifying ground-water flow models by reducing the number of zones with varying aquifer parameters and by finding a better statistical fit than that which could be achieved by trial-and-error techniques alone.

Another way statistics can be used with ground-water applications is through the use of physical characteristics and/or measurable values to create regressions that correlate these characteristics with ground-water flow values. These regressions can then be used to extrapolate values to other areas with similar characteristics. This technique obviously requires accurate ground-water flow values to create accurate regressions, so another technique would first be required to predict ground-water flows. It is therefore not a valid technique for calculating ground-water flows itself, but rather a way to extrapolate flows based on common characteristics. Sacks (2002) used multiple linear regression models to correlate basin characteristics with the calculated ground-water inflows determined from the study using the isotope mass-balance approach. The results from this study stated that geographically specific regression models were generally poorer than regression models for the entire study area, and that regression models should not be used for precise quantification of ground-water inflows to individual lakes, but rather in determining ground-water flows within a range of values as per the accuracy of the isotope mass-balance approach. Even though the regressions coupled with the isotope mass-balance approach should not be used to determine precise values, this technique does show the validity of using regressions to predict whether or not ground-water flows are significant within a given region.

CHAPTER 2: METHODS

The goal of this study was to develop a simplistic method to account for both ground-water volume and nutrient fluxes to surface-water bodies using traditionally collected data and/or data that can be easily acquired in the future. The methods described are designed to allow for ground water to be accounted for without having to quantify all other fluxes needed in a comprehensive analysis. However, the purpose behind the development of this technique is to incorporate ground water into surface-water management studies. For this reason, the WHCL was used as a case study to incorporate ground water into a pollutant load reduction goal (PLRG) for the WHCL (McCary and Ross, 2005). The PLRG study incorporated both surface-water and ground-water loads to the lakes in order to assess the water quality and develop load reduction scenarios for the lakes. Runoff loads and surficial ground-water loads were applied to the lakes through the use of modeling applications, and a simple transport model was used to generate flows between the lakes. A water quality model was developed to model lake water quality, incorporating all of the loads generated from runoff and ground water as well as the transport within the chain system. However, only the influence of ground water is discussed in this manuscript. For a detailed description of all the modeling applications, refer to McCary and Ross (2005). The time frame selected for that study was the ten-year period from 1990 – 1999, and this time frame was used as a guide for acquiring the data needed for method development. Because of the best overall availability for ground-water data, the time period from 1992 – 1999 was used for the modeling applications discussed in this chapter. This date range was expanded, including data collected on the spatial variability in ground-water concentrations from wells within different land uses, which is also discussed in this chapter.

2.1 Incorporating Ground-Water Volume Fluxes

Because of the potential for significant ground-water contribution to the WHCL, a method was developed to determine its magnitude. The method developed was the application of a steady-state, 1-layer, surficial aquifer flow model. Groundwater Vistas (Rumbaugh and Rumbaugh, 2001), a pre/post processor package for ground-water models, was used to create MODFLOW (McDonald and Harbaugh, 1988) data sets for each of the eight years of the model time period. In order to develop the models, specific data and procedures were required. The ground-water model domain was designed to represent 1-layer, the surficial aquifer system. For this modeling application, it was required to develop no-flow boundaries around the flow basin, constant-head boundaries for the lakes, target surficial well-head data for calibration purposes, and aquifer parameters needed for the flow system. A general, surficial ground water no-flow boundary was delineated around the entire lake system based on the topographic boundaries, which is a reasonable assumption for surficial aquifer systems in areas with relatively significant relief. The topographical information used was made available by the Southwest Florida Water Management District (SWFWMD). In the models, each lake is modeled as a constant-head boundary based on its annual average elevation. The constant-head boundary makes it an infinite sink or source in the model. Annual average surficial aquifer heads are used for target head values, requiring the collection of surficial aquifer well data within the ground-water flow basin. Also needed for the modeling analysis is the depth of the flow field and the hydraulic conductivity for the region. Data on aquifer depth and hydraulic conductivity were available through numerous reports published by both the SWFWMD and the United States Geological Survey (USGS). Surficial aquifer depths in the area varied from approximately 80 feet near the lakes to 120 feet in the higher upland areas. In the models, the bottom of the surficial aquifer was specified as 50 feet, which is approximately 80 feet below average lake elevations. A lateral hydraulic conductivity value of 8 ft/day is used to approximate the average hydraulic conductivity over the WHCL basin. Incorporated with the use of MODFLOW was a parameter estimation program, PEST. PEST uses an inverse prediction technique to find the best possible value for an unknown parameter. In this case, it was used to determine net surficial recharge rates in order to match model aquifer heads with target well data.

Lake stage data was collected from the SWFWMD for all lakes that had data within the WHCL. It appears that for some of the lakes, the values were only “checks” to compare their elevations to a lake with a more complete record. For this reason, a representative lake for both the Southern Chain and the Northern Chain are used to reflect all of the lakes within the system. Three of the lakes are only connected at high water elevations, so these lakes are considered separately. Lake Howard (Figure 2) had the most recorded data points in the Southern Chain, so it is shown as a representative of fourteen of the interconnected Southern Chain lakes. Lake Conine (Figure 3) had the most recorded data points in the Northern Chain, so it is shown as a representative of four of the interconnected Northern Chain lakes. Lake Blue, Lake Mariana, and Lake Fannie (Figures 4, 5, and 6, respectively) are only connected under elevated conditions, so they are shown separately.

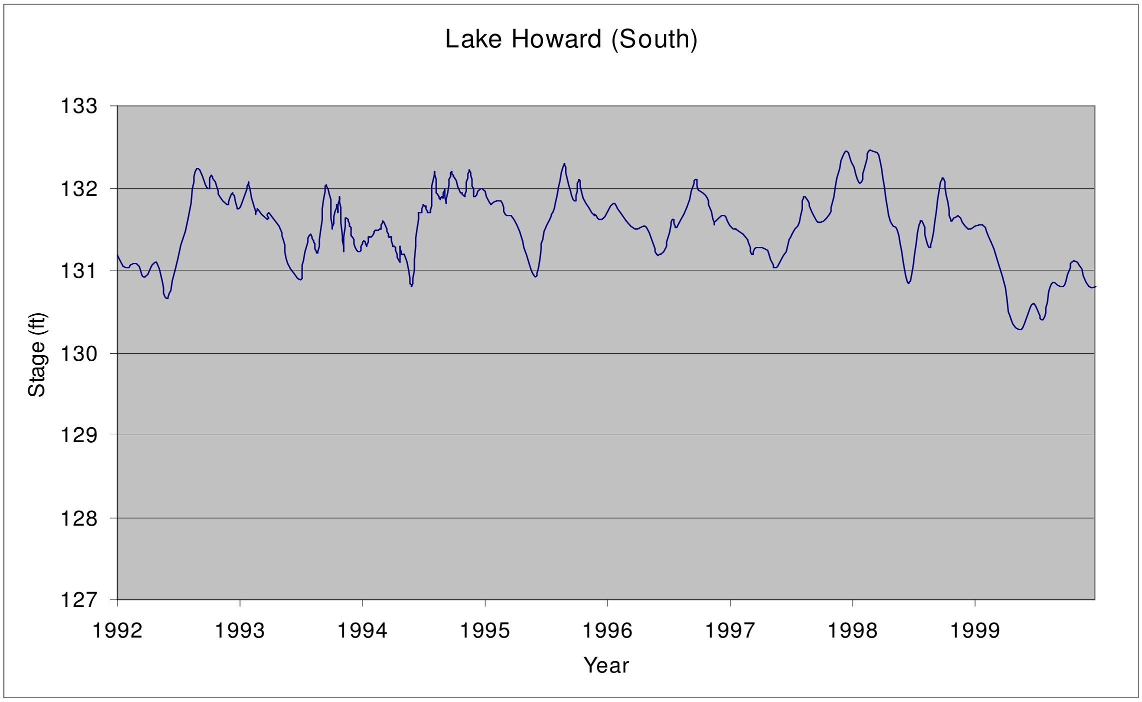

Figure 2: Lake Howard Stage vs. Time (Representing the Southern Chain Lakes)

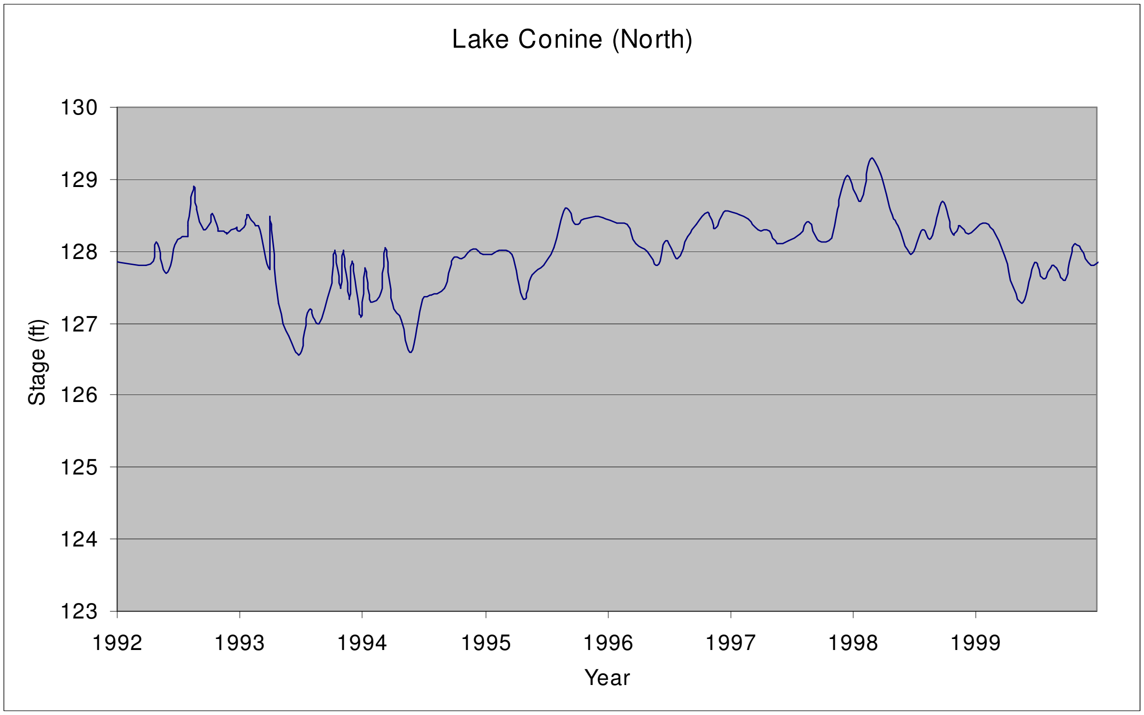

Figure 3: Lake Conine Stage vs. Time (Representing the Northern Chain Lakes)

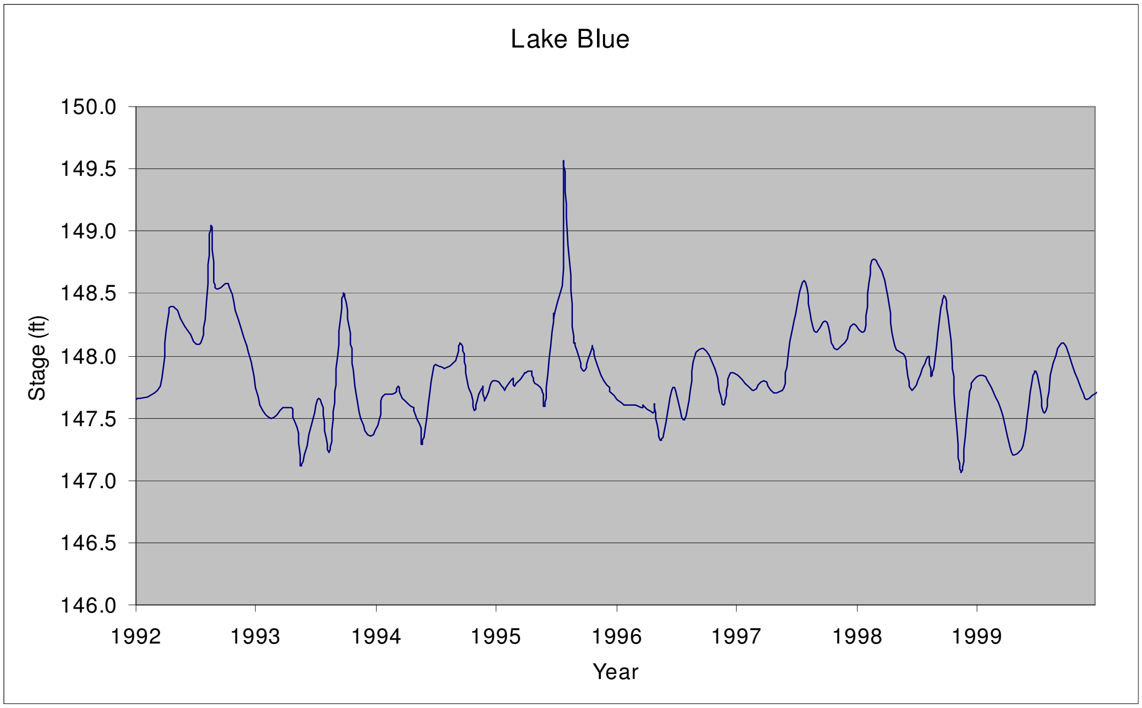

Figure 4: Lake Blue Stage vs. Time

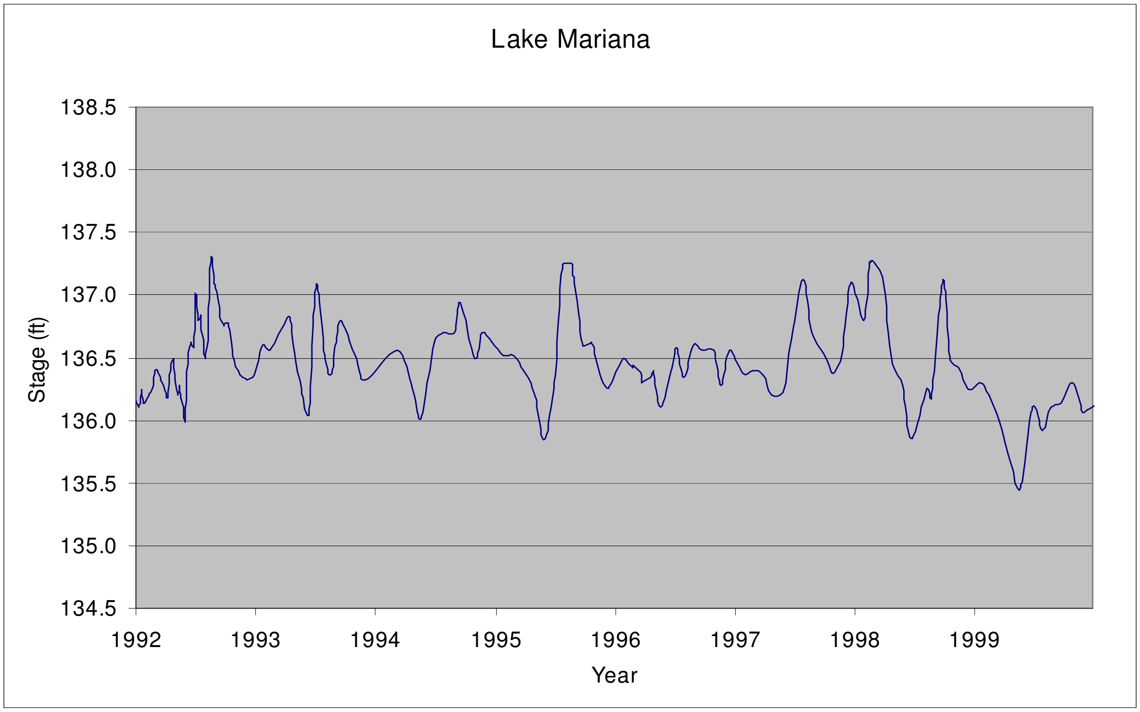

Figure 5: Lake Mariana Stage vs. Time

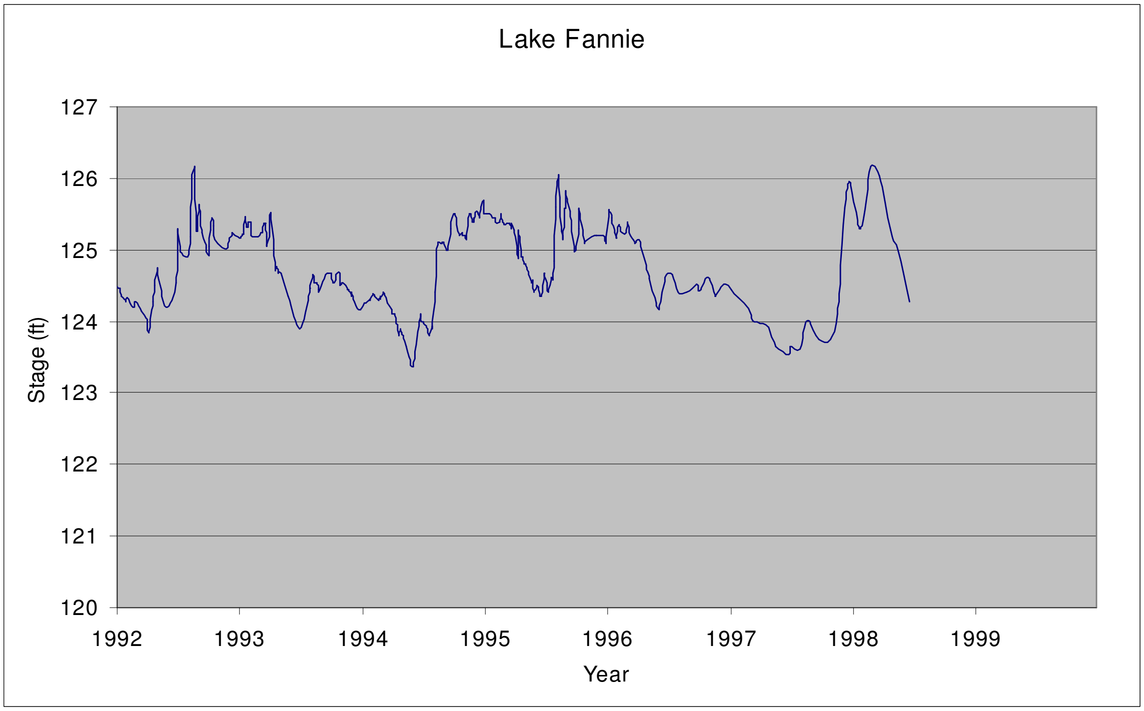

Figure 6: Lake Fannie Stage vs. Time

All available well-head data for Polk County were collected from the SWFWMD for the modeling period. There are a total of four wells located within the entire WHCL basin that have available aquifer head data: two surficial aquifer wells and two Floridan aquifer wells. This is also conveniently split between one well of each aquifer type located in the Northern Chain basin and the Southern Chain basin. These well-head values are shown in Figure 7. The locations of the wells are shown in Figure 8. Although there is limited well data within the basin, it clearly shows that the potentiometric heads for the Floridan aquifer are below both the surficial well heads and the lake stages shown in Figures 2 - 6. This means that the area is a Floridan aquifer recharge zone, which should be expected for the ridge system where the lakes are located. It also clearly shows that surficial well heads are above most of the lake stages shown in Figures 2 - 6, meaning that there is surficial inflow to those lakes.

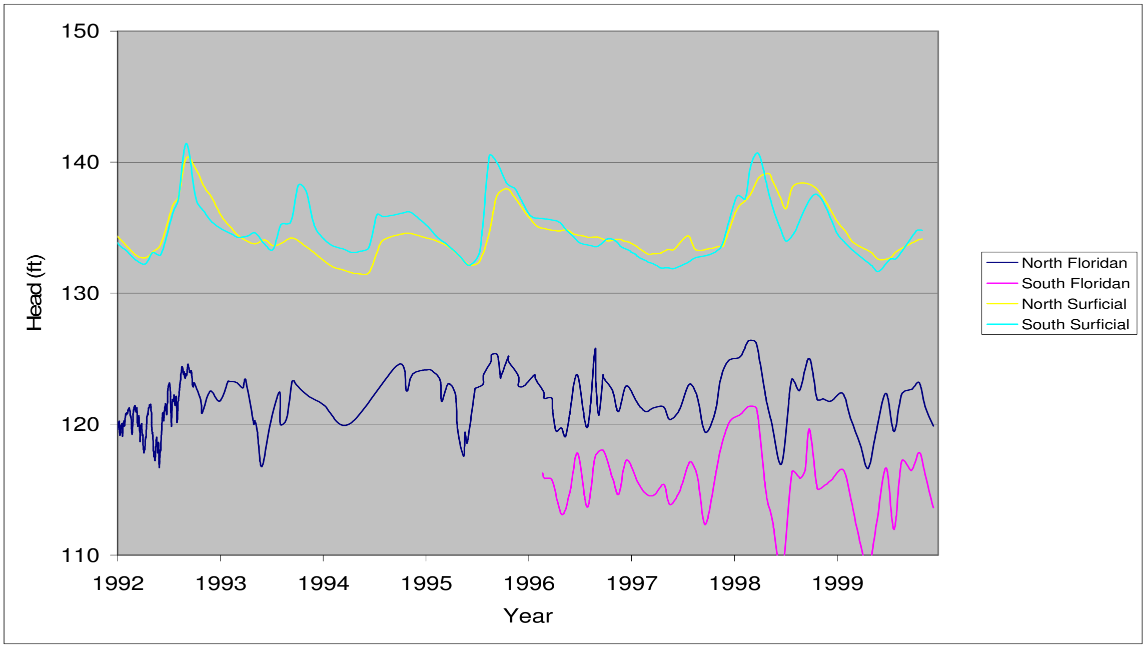

Figure 7: Well Heads vs. Time

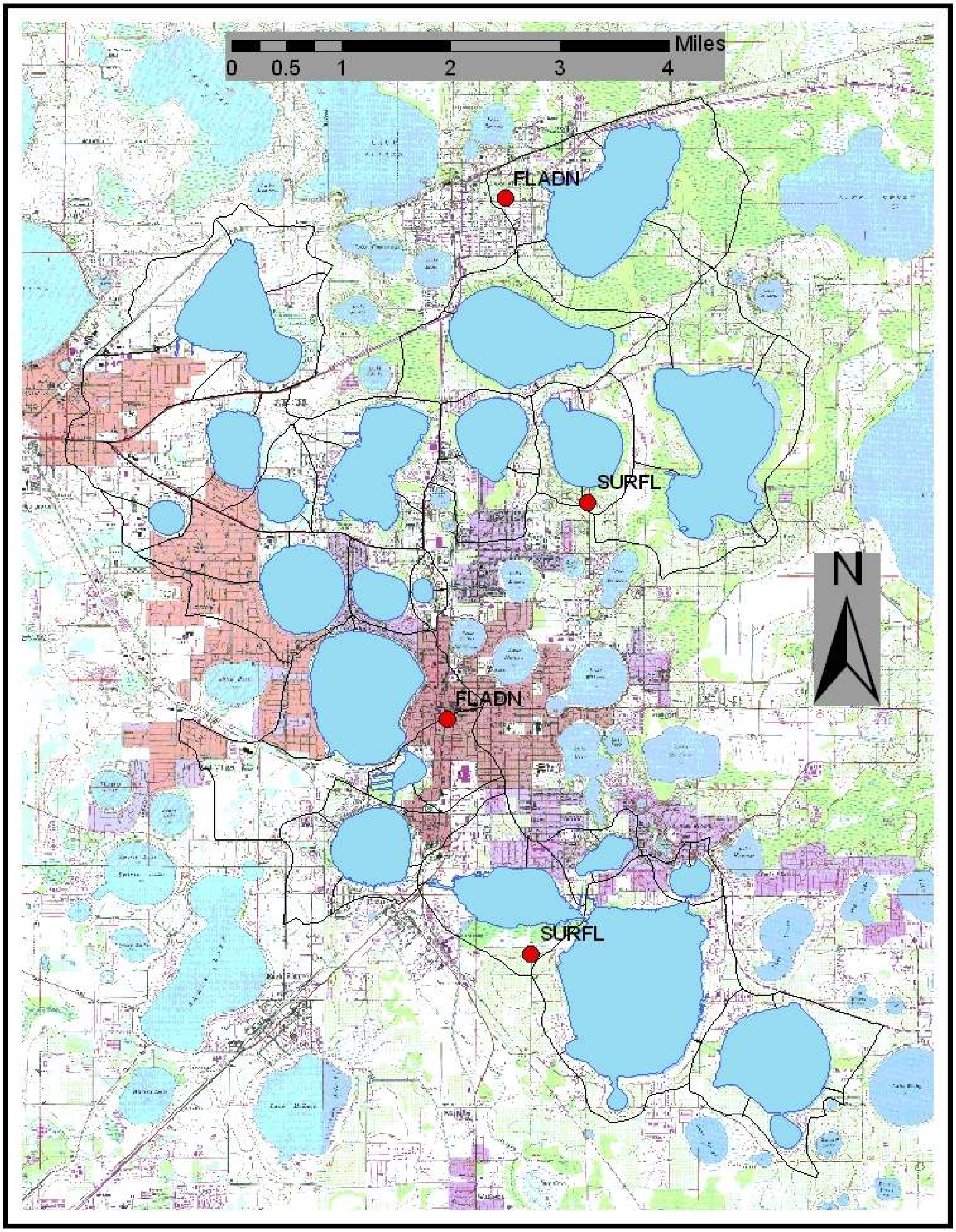

Figure 8: Well Locations within the WHCL Basin

As previously stated, the annual average of lake elevations were used to create a constant-head boundary for each lake, and annual averages of surficial well heads were used as target well data. Table 2 shows the annual average lake stages and surficial aquifer well heads that were used in the model. Recorded stage values for Lake Fannie for years 1998 and 1999 were incomplete, so the values used are the averages of the previous six years of data. The annual average values were used to approximate steady-state conditions for each year. Net surficial recharge rates are determined based on the application of PEST, using the given hydraulic conductivity to determine the values of annual average net recharge that yield the closest aquifer head values to target well data. Because there were two surficial wells within the model domain, one in the Southern Chain and one in the Northern Chain, two recharge zones were used to represent both sections of the chain.

By using the steady-state approach, target head values are simply a function of the ratio between recharge and hydraulic conductivity. Since the target well heads and hydraulic conductivity are known, PEST calculates the best statistical value for recharge rates. The determined recharge rates are considered net surficial recharge rates because they are actually recharge values minus ET and leakage to the underlying aquifer. By using a 1-layer surficial aquifer model and net surficial recharge rates, there was no need to account for the actual values of ET and leakage for the sake of determining ground-water inflows. Determining flow values was only a matter of determining head gradients and aquifer flow parameters since all model net recharge goes to the lakes. Once the values for net recharge were determined, they could be applied to determine the flows to each individual lake. The model is similar to a flow-net analysis, in that it uses a depth of flow field, recharge rate, and hydraulic conductivity. However, one major difference between this analysis and a flow-net analysis is that there is no approximation to an individual lake's contributing area or flow path. The entire WHCL basin is delineated as a whole, and the ground-water flow paths to individual lakes were determined by the model. This will allow for flow-thru areas within a lake's topographic basin that could not be accounted for by a flow-net analysis.

Table 2: Annual Average Lake Stages and Surficial Aquifer Well Heads

Site Name	1992	1993	1994	1995	1996	1997	1998	1999
Lake Howard Stage (ft)	131.55	131.55	131.63	131.68	131.62	131.50	131.72	130.84
Lake Conine Stage (ft)	128.27	127.71	127.53	128.10	128.23	128.37	128.49	127.85
Lake Blue Stage (ft)	148.31	147.63	147.74	147.99	147.67	148.02	148.04	147.69
Lake Mariana Stage (ft)	136.52	136.56	136.56	136.55	136.43	136.55	136.64	136.03
Lake Fannie Stage (ft)	124.93	124.78	124.57	125.12	124.82	123.99	124.70	124.70
South Surficial Well Head (ft)	135.10	135.12	134.49	135.73	134.37	132.60	136.91	133.35
North Surficial Well Head (ft)	136.16	134.09	132.70	134.71	134.46	133.40	137.77	133.74

2.2 Incorporating Ground-Water Nutrient Fluxes

The ground-water nutrient flux to the WHCL was to be accounted for by multiplying recorded surficial aquifer nutrient concentrations within the WHCL by the volume loading values determined in the modeling analysis. An initial investigation into ground-water quality data within the WHCL basin revealed nutrient data at only the one surficial aquifer well located in the Southern Chain basin. Because the PLRG study determined the lakes to be phosphorus limited (McCary and Ross, 2005), this analysis focused on incorporating phosphorus loads to the lakes and therefore used only the phosphorus concentrations recorded at that well. This well had three recorded data points, sampled on 3/17/1993, 3/4/1996, and 5/25/1999. Although there were not complete phosphorus data for all three sample dates, averages were taken from the data that were available. Average concentrations for total phosphorus, organic phosphorus, and ortho-phosphorus were 0.021 mg/L, 0.006 mg/L, and 0.015 mg/L respectively. The average concentrations of organic phosphorus and ortho-phosphorus were multiplied by the volume loading values determined in the modeling analysis to give nutrient loading results for both forms of phosphorus to each lake.

2.3 Analyzing Spatial Variability in Ground-Water Nutrient Concentrations

Because of the prospect of low representative nutrient concentrations used from the data at the Lake Eloise well, which would in turn generate low nutrient loading estimations to the WHCL, an attempt was made to account for the spatial variability in ground-water nutrient concentrations. In order to do this, a more detailed investigation into available well data needed to be considered. The method used was the development of ground-water nutrient concentrations by land use, which is similar to the event mean concentrations (EMCs) used for runoff values. The investigation included data for both nitrogen and phosphorus, and could be beneficial to use in areas with limited ground-water quality data. However, ground-water quality data were not abundantly available for each land use. There were five surficial wells in Polk County that had water quality data provided by the SWFWMD, and all of these wells were within an agricultural setting. In addition to these data, there were other limited ground-water quality data received from another study completed by PBS&J during the same time period as the PLRG study (PBS&J, 2004). That study incorporated four wells, two in the Northern Chain basin and two in the Southern Chain basin, and each well had two recorded data points. These wells were mainly in an urban setting. Adamski and Knowles (2001) reported on data collected in the Ocala National Forest region, with land being representative of natural, forested conditions. Metz and Sacks (2002) collected data for a study on three lakes in Hillsborough County, with well data being mixed between an urban setting and wetlands. In order to achieve the best spatial representation possible, only dates with complete data for either total nitrogen or total phosphorus were considered in this analysis. With the limited data that were available, results were developed for the following land uses: urban, agricultural, forested, rangeland, and wetlands.

CHAPTER 3: RESULTS

Using the methods described in Chapter 2, results for both the ground-water volume and nutrient flux analyses were generated and are shown in this Chapter. These results were also incorporated with runoff loading results for a comprehensive analysis on water quantity and quality in the WHCL (McCary and Ross, 2005), which was the purpose behind the development of the methods discussed in this manuscript. Results from the spatial variability analysis are also shown, confirming the suspicion of low nutrient concentrations at the Lake Eloise well.

3.1 Results of Ground-Water Volume Flux Analysis

As previously stated, there was one surficial well within the Southern Chain and one surficial well within the Northern Chain. Two recharge zones were used, one for both the Southern Chain and the Northern Chain, and these two wells were used as the target wells in each zone. For each year of the modeling study, recharge rates were estimated in each zone, resulting in an approximate annual net recharge rate for each year of the study for each zone. These net recharge rates were used to determine the approximate annual contribution from the surficial aquifer to each lake for all eight years of the study. The results are shown in Table 3.

Table 3: Surficial Ground-Water Volume Loading to the WHCL

Year	1992	1993	1994	1995	1996	1997	1998	1999	AVG
Lake Name	Flow (in/yr)								
Winterset	7.9	7.9	6.3	9.1	6.0	2.1	11.7	3.6	6.8
Eloise	4.9	4.9	3.9	5.6	3.7	1.3	7.2	2.2	4.2
Summit	16.2	16.3	13.0	18.6	12.2	4.4	24.0	7.4	14.0
Lulu	16.3	16.4	13.1	18.7	12.3	4.4	24.2	7.4	14.1
Roy	23.3	23.4	18.7	26.7	17.6	6.3	34.4	10.6	20.1
Shipp	26.1	26.2	20.9	29.9	19.7	7.1	38.6	11.9	22.6
May	49.0	49.3	39.3	56.1	37.0	13.3	72.5	22.3	42.3
Howard	12.8	12.9	10.3	14.7	9.7	3.5	19.0	5.8	11.1
Cannon	13.2	13.2	10.9	14.8	10.4	5.0	18.5	7.1	11.6
Blue	45.0	47.9	28.3	60.5	23.8	-24.1	92.5	-7.2	33.4
Mirror	10.0	10.0	8.0	11.4	7.5	2.7	14.8	4.5	8.6
Spring	33.3	33.5	26.7	38.2	25.1	9.1	49.3	15.2	28.8
Idylwild	22.3	22.3	18.5	25.0	17.6	8.6	31.3	12.1	19.7
Jessie	41.6	41.4	35.4	45.6	33.9	20.1	55.4	25.7	37.4
Mariana	18.8	18.9	14.9	21.7	14.0	4.3	28.4	7.9	16.1
Hartridge	12.9	11.4	9.0	13.3	9.6	4.5	18.3	6.3	10.7
Conine	36.3	28.0	24.9	31.0	29.5	24.3	42.7	24.9	30.2
Rochelle	22.4	16.6	14.6	18.7	17.9	14.6	26.4	15.1	18.3
Haines	34.6	25.6	22.4	28.8	27.7	22.8	40.8	23.4	28.3
Smart	29.6	21.7	18.8	24.5	23.3	18.4	34.9	19.6	23.8
Fannie	32.4	24.1	21.3	27.0	26.2	22.1	38.1	22.3	26.7
Southern Chain	14.3	14.2	11.3	16.2	10.7	4.0	21.0	6.5	12.3
Northern Chain	30.8	22.9	20.1	25.7	24.7	20.4	36.2	20.9	25.2
Total	19.9	17.2	14.3	19.4	15.5	9.6	26.2	11.4	16.7

In summary, the model results show that surficial ground water contributes an average of 12 in/yr in the Southern Chain, 25 in/yr in the Northern Chain, and 17 in/yr over the entire WHCL when averaged over the combined lake areas. It is important to note that these flux rates are normalized over the lake areas, not the basin areas.

3.2 Results of Ground-Water Nutrient Flux Analysis

The volume loading results from the modeling analysis were multiplied by the measured values of surficial aquifer phosphorus concentrations within the WHCL basin. These values were based on an assumed constant surficial ground-water concentration which was taken from the measured ground-water concentrations from the one surficial well at Lake Eloise. The results are shown in Table 4.

Table 4: Surficial Ground-Water Nutrient Loading to the WHCL

Year	1992	1993	1994	1995	1996	1997	1998	1999	AVG
Lake Name	Phosphorus Load (kg/yr)								
Winterset	9.4	9.4	7.5	10.7	7.1	2.5	13.8	4.3	8.1
Eloise	12.2	12.3	9.8	14.0	9.2	3.3	18.1	5.6	10.6
Summit	2.2	2.2	1.8	2.6	1.7	0.6	3.3	1.0	1.9
Lulu	10.8	10.9	8.7	12.4	8.2	2.9	16.0	4.9	9.4
Roy	3.7	3.7	3.0	4.3	2.8	1.0	5.5	1.7	3.2
Shipp	15.6	15.7	12.5	17.9	11.8	4.2	23.0	7.1	13.5
May	5.5	5.5	4.4	6.2	4.1	1.5	8.1	2.5	4.7
Howard	17.3	17.4	13.9	19.8	13.1	4.7	25.6	7.9	15.0
Cannon	9.4	9.3	7.7	10.5	7.3	3.5	13.1	5.0	8.2
Blue	5.2	5.5	3.3	7.0	2.8	0.0	10.7	0.0	4.3
Mirror	2.7	2.7	2.2	3.1	2.0	0.7	4.0	1.2	2.3
Spring	1.6	1.6	1.3	1.9	1.2	0.4	2.4	0.7	1.4
Idylwild	4.5	4.5	3.7	5.0	3.5	1.7	6.3	2.4	4.0
Jessie	16.7	16.7	14.3	18.4	13.6	8.1	22.3	10.3	15.1
Mariana	20.8	20.8	16.4	24.0	15.4	4.7	31.3	8.7	17.8
Hartridge	11.5	10.2	8.0	11.9	8.6	4.0	16.4	5.7	9.5
Conine	18.3	14.1	12.5	15.6	14.8	12.2	21.4	12.5	15.2
Rochelle	27.7	20.6	18.0	23.1	22.1	18.1	32.6	18.7	22.6
Haines	51.6	38.2	33.4	42.9	41.3	33.9	60.8	34.9	42.1
Smart	17.5	12.8	11.1	14.5	13.8	10.9	20.7	11.6	14.1
Fannie	51.7	38.5	34.0	43.1	41.8	35.3	60.8	35.5	42.6
Southern Chain	149	149	119	170	112	44	220	69	129
Northern Chain	167	124	109	139	134	110	196	113	137
Total	316	273	228	309	246	155	416	182	266

In summary, the model results show that surficial ground water contributes an average of 129 kg/yr of phosphorus to the Southern Chain, 137 kg/yr of phosphorus to the Northern Chain, and 266 kg/yr of phosphorus to the entire WHCL.

3.3 Results of Spatial Variability Analysis of Nutrient Concentrations

The data collected from wells within specified land uses were combined to create an approximation of ground-water concentrations by land use. While there was not an abundance of ground-water quality data, the data that were collected do show how ground-water quality can vary both between land use and within a given land use. The data also show that within a given land use, one nutrient can be present in much larger quantities when compared with the other. For instance, an agricultural land use can be a significant source of nitrogen while being relatively dilute in phosphorus. These values are shown in Tables 5 and 6.

Table 5: Surficial Aquifer Phosphorus Concentrations (mg/L) by Land Use

Land Use	Min	Mean	Max	Count
Urban	<0.01	0.11	1.20	31
Agricultural	0.02	0.08	0.42	30
Forested	0.01	0.22	0.83	8
Rangeland	0.09	0.18	0.27	2
Wetlands	<0.01	0.04	1.50	7

Table 6: Surficial Aquifer Nitrogen Concentrations (mg/L) by Land Use

Land Use	Min	Mean	Max	Count
Urban	<0.03	0.82	11.40	31
Agricultural	1.83	13.37	32.43	29
Forested	<0.22	0.36	0.83	10
Rangeland	4.60	5.15	5.70	2
Wetlands	<0.04	0.16	0.75	7

CHAPTER 4: DISCUSSION

The 1-layer, steady-state, surficial aquifer models developed in this study were designed to approximate surficial inflows to the WHCL. The model was calibrated using all available well data traditionally collected in the lake system. By using the steady-state approximation, the ratio of recharge to hydraulic conductivity is directly proportional to the target well heads. This makes it possible to understand the sensitivity of model results to changes in model parameters. For instance, a 50% increase in hydraulic conductivity would require a 50% increase in recharge values to sustain the target head value. This also allows for a convenient way to check recharge results to see if they are reasonable. If hydraulic conductivity is unknown and assumed to be too high, calibrated recharge values would be unreasonably high in order to sustain target well heads. This makes it important to have sufficient target well data that represents the system. If hydraulic conductivity values are reasonably understood, additional target well heads within a model domain will yield better model results. If hydraulic conductivity values are unknown, then additional target well heads will not yield better model results. Both target head values and hydraulic conductivity values must be known in order to solve for recharge, since it is the ratio of recharge to hydraulic conductivity, not the individual values, that is used to determine model head values.

Results from the models used in this study show that on average, surficial ground water contributes 12 in/yr in the Southern Chain, 25 in/yr in the Northern Chain, and 17 in/yr over the entire chain when averaged over all of the lake areas. Comparing these numbers to the runoff volumes determined in the PLRG study (McCary and Ross, 2005), runoff contributes an average of 18 in/yr in the Southern Chain, 6 in/yr in the Northern Chain, and 14 in/yr over the entire chain when averaged over all of the lake areas. These results clearly indicate that ground water is a major contributor to the volume loading to the WHCL.

Lake Eloise, which is the largest lake in the chain and has a relatively small basin, receives on average 4 in/yr of surficial inflow when averaged over the lake's area. This is the smallest contribution to any of the lakes. Lake May, which is one of the smallest lakes in the chain and has a relatively large basin, receives on average 42 in/yr of surficial inflow when averaged over the lake's area. This is the largest contribution to any of the lakes. Lake Blue, which is also one of the smallest lakes in the chain, has the highest variability in model results. It is also the only lake that model results show lateral flows from the lake to the surficial aquifer. This may be due to the high elevation of the lake and the lack of a surficial monitor well in the lake's basin resulting in large model uncertainty in that region.

Overall, the model results yield similar volume flux rates when compared to other studies done in the WHCL region. The Sacks (2002) study, which evaluated only seepage lakes and incorporated a detailed data collection effort for water isotopes, concluded that lakes in this region receive between 25% and 50% of inflows from the surficial aquifer. For a seepage lake that receives approximately 52 in/yr of rainfall, that would mean between 13 in/yr and 26 in/yr of surficial ground-water inflow, which is in agreement with the current modeling analysis. The PBS&J (2004) study resulted in a detailed water-budget and mass-balance analysis on two of the lakes for one year, Lake Shipp in the Southern Chain and Lake Haines in the Northern Chain. For the year, results show that Lake Shipp received 46 inches of surficial inflow and Lake Haines received 37 inches of surficial inflow. These values are higher than what was approximated in the current modeling analysis.

The models used in this study did not require the extensive detail that would have been needed for a detailed transient, multi-layered ground-water flow model. The main advantage to using a steady-state approximation over a detailed transient model is that aquifer storage parameters, such as porosity and specific yield, do not need to be estimated since there is no change in volume stored. This can be very beneficial since these parameters, in particular specific yield, can be very difficult to approximate. If specific yield is incorrectly accounted for, it will lead to a poor match between modeled well heads and measured observation well heads, and more importantly, increased flux uncertainty. It should be noted that the technique used in this study is not recommended to use over a time period in which lake stage and/or well heads change significantly as it can only yield average fluxes. When trying to account for a water body in which stage can fluctuate rapidly, or in an area with a shallow water table that can experience significant elevation fluctuations over short periods of time, the steady-state approximation is not valid. In these situations, a much shorter “steady-state” period must be approximated or a more detailed transient model with variable-head boundaries and aquifer storage parameters must be considered. The modeling technique used in this study was only designed to account for average flows over longer periods of time (i.e., years).

Results from the nutrient flux analysis show that on average, surficial ground water contributes 120 kg/yr of total phosphorus to the Southern Chain, 130 kg/yr of total phosphorus to the Northern Chain, and 250 kg/yr of total phosphorus to the entire WHCL. The runoff loading values determined in the PLRG study (McCary and Ross, 2005) show that runoff contributes approximately 3000 kg/yr of total phosphorus to the Southern Chain, 500 kg/yr of total phosphorus to the Northern Chain, and 3500 kg/yr of total phosphorus to the entire WHCL. The values predicted for ground-water loading were significantly less than those predicted for runoff loading. This is especially significant in the Northern Chain, where results show larger volume fluxes from ground water than from runoff. Loading reduction scenarios used in the PLRG study (McCary and Ross, 2005) for the Northern Chain show poor response in water quality to reductions in overall loading. This may be due to the underestimation of ground-water loading, since water quality results were calibrated with low loading values. Using low initial loading values results in reduction scenarios that remove little overall mass.

The low values of predicted surficial ground-water phosphorus loading to the WHCL result directly from the low ground-water phosphorus concentrations taken from the Lake Eloise well used to represent the study area. For comparative purposes, the PBS&J (2004) study had significantly different results for ground-water phosphorus loading to both Lake Shipp and Lake Haines. While that study only used one average ground-water concentration for each lake's ground-water flow basin, the values were taken from a better representation of sampled well data specific to the 1-year study period and the two lake basins. Their analysis concluded that for that one year, surficial ground water contributed 188 kg of phosphorus to Lake Shipp and 374 kg of phosphorus to Lake Haines. These results are significantly higher than the results obtained in this study.

The concentration variability of the limited number of wells analyzed strongly suggests the need for more ground-water constituency information. Perhaps additional sample data can support the development of ground-water concentrations by land use in order to account for the spatial variability in the WHCL and other areas. The limited results obtained for ground-water nutrient concentrations by land use, shown in Tables 5 and 6, indicate that there is significant potential for nutrient loading to be highly variable, and ground-water loading may be heavily dependent on the land use, age, and other factors within lake ground-water flow basins. The average value of total phosphorus concentration of 0.021 mg/L used in the nutrient flux analysis in this study is significantly less than the values reported in Table 5, which were taken from a larger set of sampled well data made after the PLRG study (McCary and Ross, 2005) was concluded. The values used in the PBS&J (2004) study, which were 0.069 mg/L in the Lake Shipp basin and 0.145 mg/L in the Lake Haines basin, are more representative of the average concentrations shown in Table 5.

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

Surface-water features are strongly coupled to ground water in west-central Florida, and in particular ridge lake systems, such as the WHCL, owe a significant portion of the volume flux to the surficial aquifer. In order to provide an accurate assessment of volume or nutrient loading to these surface-water systems, an analysis of ground-water fluxes must be included. As shown from this study, both the volume and nutrient loadings to a system must be determined to accurately quantify loadings.

The 1-layer, steady-state, surficial aquifer model used in this study is one method for approximating annual average flows to a surface-water system. When compared with limited but more detailed studies performed recently in the WHCL area, results are comparable, and show that the WHCL receives substantial volume and nutrient fluxes from the surficial ground-water system. The process that was developed works well, and because it uses data that is readily available in many areas, it can be applied readily elsewhere. Reasonable results can be obtained without the level of detail that would be involved in a detailed transient model. However, results could be improved and uncertainty could be diminished if additional target well data existed within the model domain.

One of the most important observations from this study is the need for more ground-water observations, especially water quality sampling data. The use of one data set from the one well at Lake Eloise probably resulted in the underestimation of the average concentration of phosphorus in the surficial ground-water system. This probably underestimated ground-water nutrient loading to the WHCL. The loading reduction scenarios performed in the PLRG study (McCary and Ross, 2005) show model calibration and prediction sensitivities when ground-water loads are underestimated. Basically, if loading predictions were higher, net benthic fluxes would be higher, and there would be more sensitivity to load reductions. When comparing the loading results to the PBS&J study (2004), the low values used to represent the surficial system were evident. This further supports that ground-water concentrations must be accurately quantified and that spatial variability should be accounted for in considering loading from the surficial system for PLRG studies and similar loading studies.

The method of accounting for ground-water nutrient concentrations by land use may be a good technique for showing the spatial variability in nutrient concentrations. This technique can be especially beneficial because it can be directly coupled with the EMC values developed for runoff concentrations commonly used for runoff loading analyses. Varying ground-water nutrient concentrations by land use are evident when comparing the data summarized in Tables 5 and 6. However, it should be cautioned that land-use correlations may be inappropriate to use in other areas unless similar hydrogeologic, climatologic, biological, and/or anthropogenic conditions exist. Since surficial wells are relatively abundant, it is recommended for future studies to select representative wells for sampling, considering land use, soil type, and time scale. Time scale is especially important, since it may be many years before a land use influences ground-water nutrient loading to a surface-water body. For this reason, it is recommended to select well sites as close to the study site as possible. Since these data are already available as a limited resource, another study combining existing data with further testing may be of great benefit for future loading studies.

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