Final Project Report

Spatial Modeling of Landscape Development Intensity And Water Quality in the St Marks River Watershed DEP Contract #GW138

to the
Department of Environmental Protection
Bureau of Water Resources Protection

By

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EXECUTIVE SUMMARY

Organization of the Report

The report is organized in chapters. Each chapter is self contained, having individually numbered pages, figures, and tables. In addition, each chapter contains a bibliography and appendices as required. The first chapter is an annotated bibliography of literature relevant to the St Marks Watershed and spatial and pollutant modeling. The second chapter reports on pollutant loading of the Lake Lafayette and Lake Munson sub-basins. In the third chapter a spatial model of nitrogen in surficial ground waters of Wakulla County is reported. The fourth chapter gives the results of the work conducted to relate Landscape Development Intensity (LDI) to spatial loading of Total Phosphorus within the St Marks. In the fifth chapter copies of slides used in a presentation given at the DEP Twin Tower offices in May, 1998 are provided. Finally. the sixth chapter is a proposal for continuation of work on spatial modeling and LDI's incorporating a statewide data base, expanding model parameters, and using statistical tests to validate model predictions.

Project Narrative

Research began in the late spring of 1997. Graduate students Neal Parker and Alan Foley spent several days within the St Marks Watershed traveling from top to bottom and throughout the basin learning first hand about its development patterns, drainage networks, and ecological systems. Additionally they spent several days retrieving literature from state and local agencies. On a separate occasion, Mark Brown traversed the basin for a two day period in June, 1997. In April and May, 1997 Mark Brown attended several meetings at DEP in Tallahassee discussing the scope and timing of the project and participating in kickoff presentations for the St. Marks Watershed Project.

The goal of the initial phase of our project was to support the publication of the document summarizing the watershed planning process for the St. Marks Basin. As a result, the first 4 months of the project were aimed at providing this support. First an annotated bibliography of relevant literature was complied. Second, simulations and analysis of water quality in the Lafayette and Munson sub-basins were completed, and third nitrogen loading resulting from septic tanks in areas near Wakulla Springs were evaluated. The reports that resulted from these investigations are included in this final report as Chapters 1, 2 and 3.

The goal of the overall project was to further develop Landscape Development Intensity (LDI) indices and demonstrate their link to stormwater quality using the St. Marks Watershed. Since water quality data within the St. Marks were few and far

(BMP's), and restoration of historic wetlands were tested with the models for the present and future conditions to evaluate their effectiveness in reducing pollutant loads.

Pollutant Loading and Percent Impervious Surface

The amount of impervious surface within watersheds is related to the intensity of human activity and as a result, is a good predictor of stormwater water quality. The graph in Figure 1 shows pollutant load in the sub-basins of Lake Lafayette and Lake Munson watersheds compared to impervious surface. As the graph shows, "imperviousness" is strongly correlated to pollutant load. Numerous other studies across the United States have suggested that impervious surface may be a very good predictor of stormwater quality and the health of downstream waterbodies.

Sub-basins of the Lafayette and Munson watersheds were ranked based on their imperviousness for the present land use conditions, and for future land uses based on maps provided by Leon County Planning (Figures 2 and 3). Based on previous studies by others which suggested that imperviousness was related to ecosystem health we concluded that at the present time 7 sub-basins out of the total number of 16 sub-basins within the Lafayette and Munson watersheds have sufficient impervious surface (greater than 30%) to warrant serious concern for the ecological health of surface water bodies. Further, using land use projections we suggested that 11 of the 16 sub-basins. Will have imperviousness greater than 30% raising concern for ecological health of water bodies within these basins.

Pollutant Loading

Using a GIS based spatial model, annual pollutant loads for each of the sub-basins in the Lafayette and Munson watersheds were modeled. The graphs in Figures 4 and 5 show comparisons of annual pollutant loading by sub-basin for the past, present and future conditions for the Lafayette and Munson Basins. The shortest bars in the graph are for the natural landscape, averaging about 0.4 lbs/acre*yr^{-1} (0.45 kg/ha*yr^{-1}). Urbanized areas have about 3 times these background loads (1.4 lbs/acre*yr^{-1} [1.48 kg/ha*yr^{-1}]). The biggest changes from present conditions to future conditions are found in the Lafayette Basin where sub-basins exhibit annual pollutant load increases of between 50 to 80%. The increases in the Munson basin between the present and future condition are much smaller, with only one basin exhibiting a 75% increase in annual load. The remaining basins all appear to exhibit increase of between 5 and 15%.

Pollutant Transfer

A second GIS based simulation model was developed that used an overland flow algorithm to converge and concentrate runoff. Annual pollutant load for the past, present, and future conditions in the Lake Lafayette and Lake Munson basins were generated using the model. Pollutant loads are summed along flow paths so that total load at any point in the drainage basin could be read from the resulting maps. Among the most significant simulation results in the Lake Lafayette basin the model simulated:

- a present day increase of 350% over historic pollutant loads in the lower reach of Alford Arm Branch.

between, developing links between LDI's and stormwater quality could not be accomplished using existing data. As a result it was necessary for Neal Parker to develop a GIS based spatial model for predicting stormwater pollutant loading in all sub-basins of the watershed. This model was an outgrowth of our initial work on the Lafayette and Munson sub-basins and is reported in Chapter 4 of this report.

Work progressed on the spatial model and the Wakulla County nitrogen loading model through the winter and spring of 1998 and we made a presentation In Tallahassee of our results to date in May, 1998. A copy of the PowerPoint presentation is included as Chapter 5. Since the May presentation, we have finalized the spatial simulation model and LDI analysis and the Wakulla County nitrogen loading model, taking several extra months since the end of the contract to finalize these reports.

As our worked progressed through the summer of 1998, it was apparent that a larger effort was needed to validate the pollutant loading model and LDI relationships that were explored with this initial investigation. Validation of the model was hampered by a lack of data within the St Marks basin and as a result water quality correlations to LDI's were somewhat tenuous. The final chapter (Chapter 6) in this report is a proposal for continuation of work on spatial modeling and LDI's incorporating a statewide data base, expanding model parameters, and using statistical tests to validate model predictions.

Project Summary

In this section the goals and results of each of the studies conducted during this project are summarized and major points are extracted and summarized from each of the chapter reports.

Chapter 1: Literature Review and Annotated Bibliography

Neal Parker and Alan Foley

The literature on land use based pollutant loading, pollutant loading models, and especially data sources for the St marks basin were reviewed and an annotated bibliography compiled. The bibliography contains over 130 entries.

Chapter 2: Stormwater, Pollutant Loads and Management in the Lafayette and Munson Basins. Mark Brown and Neal Parker

The effects of spatial distributions of land uses on pollutant loading received by surface water features were modeled using spatial models that incorporated land use and topography. Lake Lafayette and Lake Munson were divided into sub-basins to evaluate the various sub-basin contributions to each water body. In addition, pollutant loads were modeled for major water bodies and closed basins within each of the larger watersheds to provide perspective on areas of concern.

As a means of understanding loss of "basin function", pollutant loads were modeled for three time periods for each of the basins: past, present, and future, and then compared. Stormwater management options including Best Management Practices

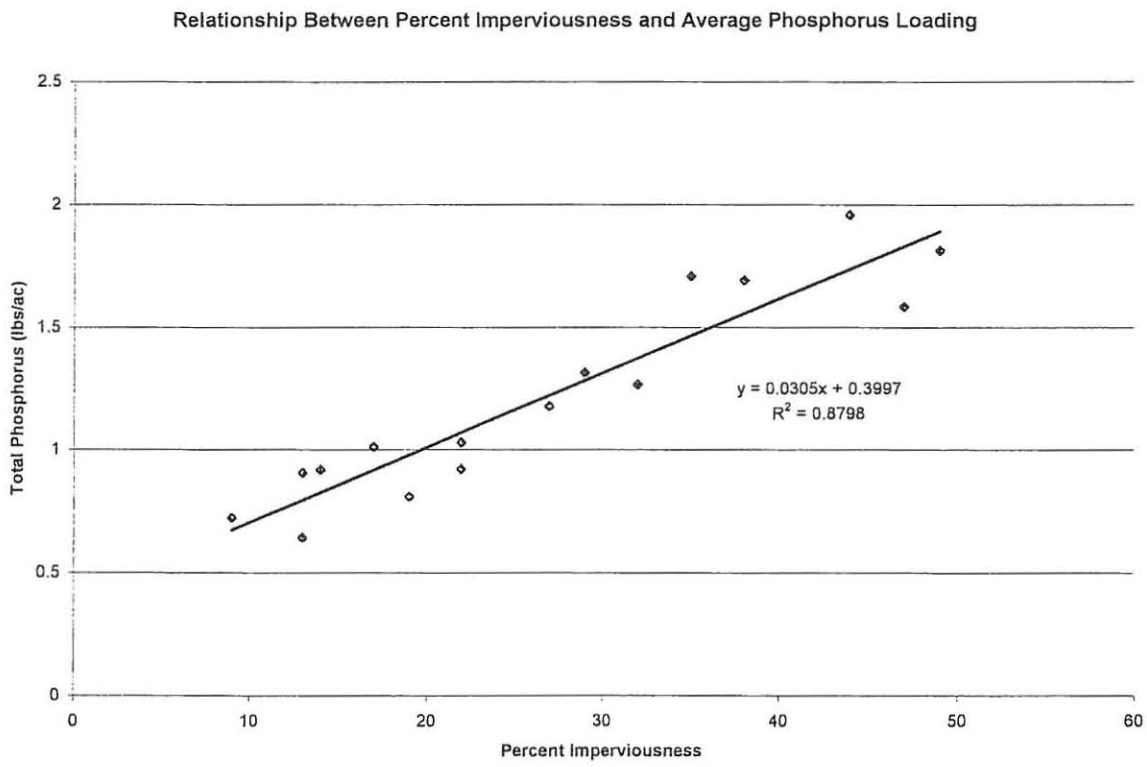
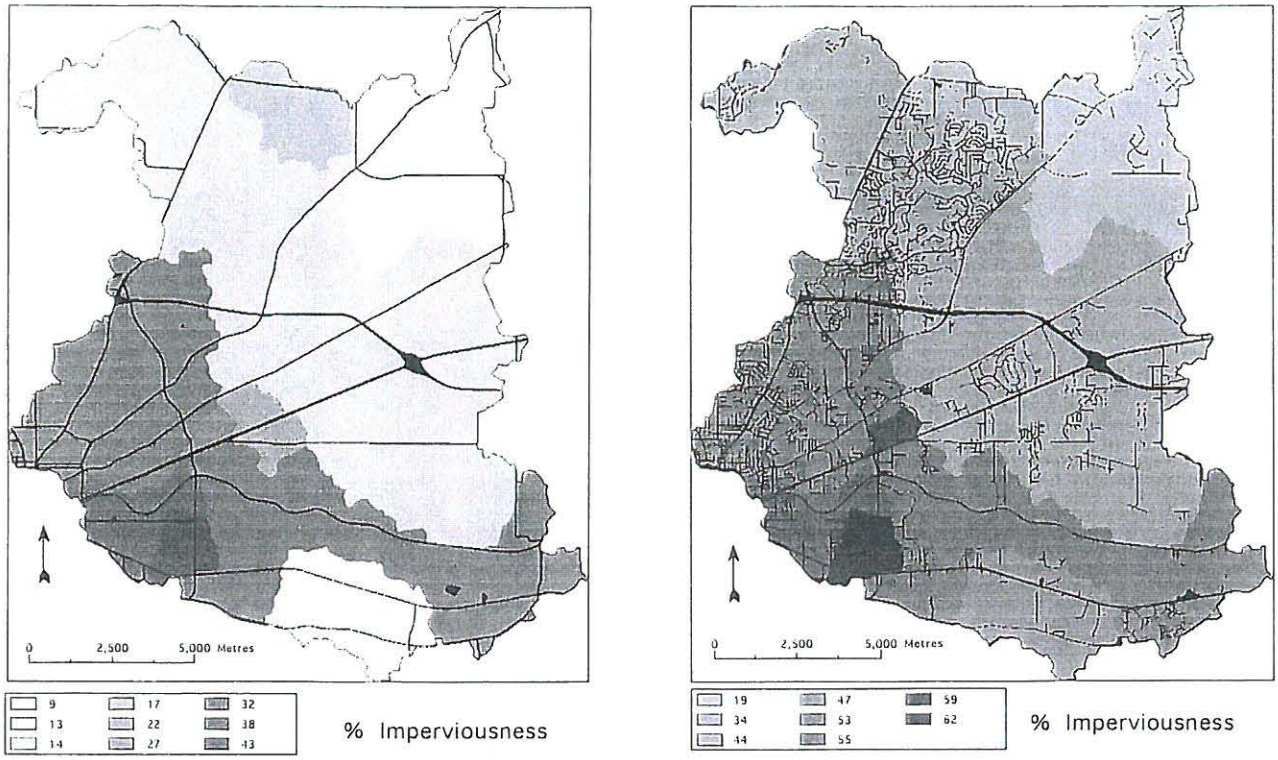
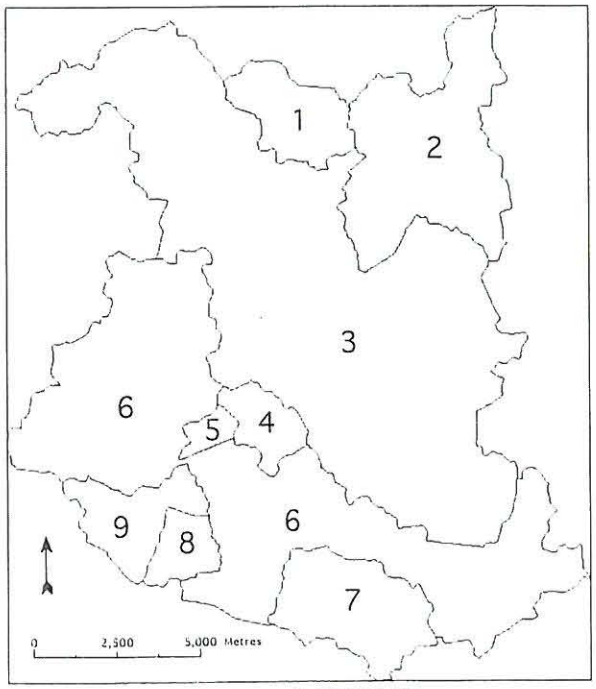


Figure E-1. Impervious surface vs. Phosphorus Load in lake Lafayette and Lake Munson sub-basins

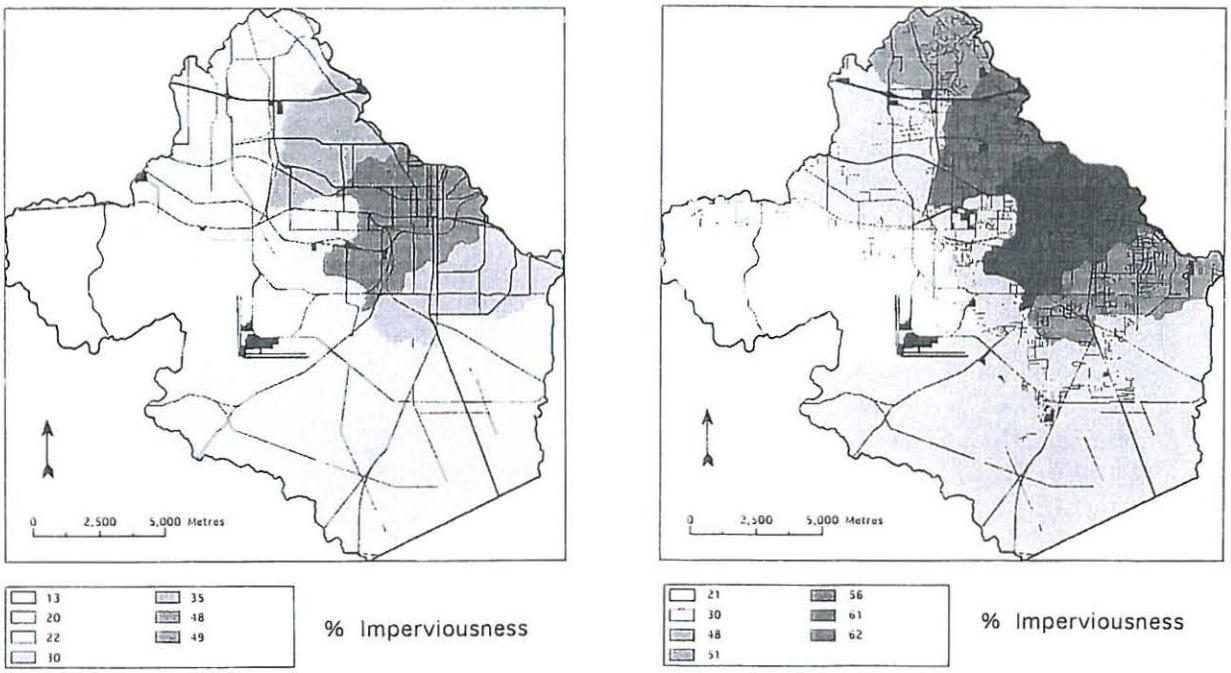


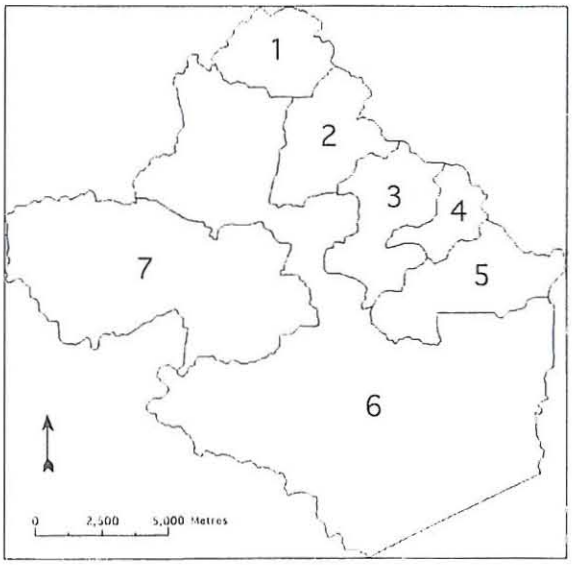


1. Gilberts Pond Outlet
2. Roberts Pond Outlet
3. Alford Arm
4. Buck Lake Outlet
5. Unnamed Run
6. Lake Lafayette Drain
7. Unnamed Slough
8. Unnamed Run
9. Mall Drainage Area

Fig. 9

Figure E-2. Maps of Impervious surface in Lake Lafayette sub-basins. Top left is present condition, top right is based on future land use.





1. Unnamed Run
2. Godby Ditch
3. Central Drainage Ditch
4. St. Augustine Branch
5. East Drainage Ditch
6. Munson Slough
7. Bradford Brook

Figure E-3. Maps of Impervious surface in Lake Munson sub-basins. Top left is present condition, top right is based on future land use.

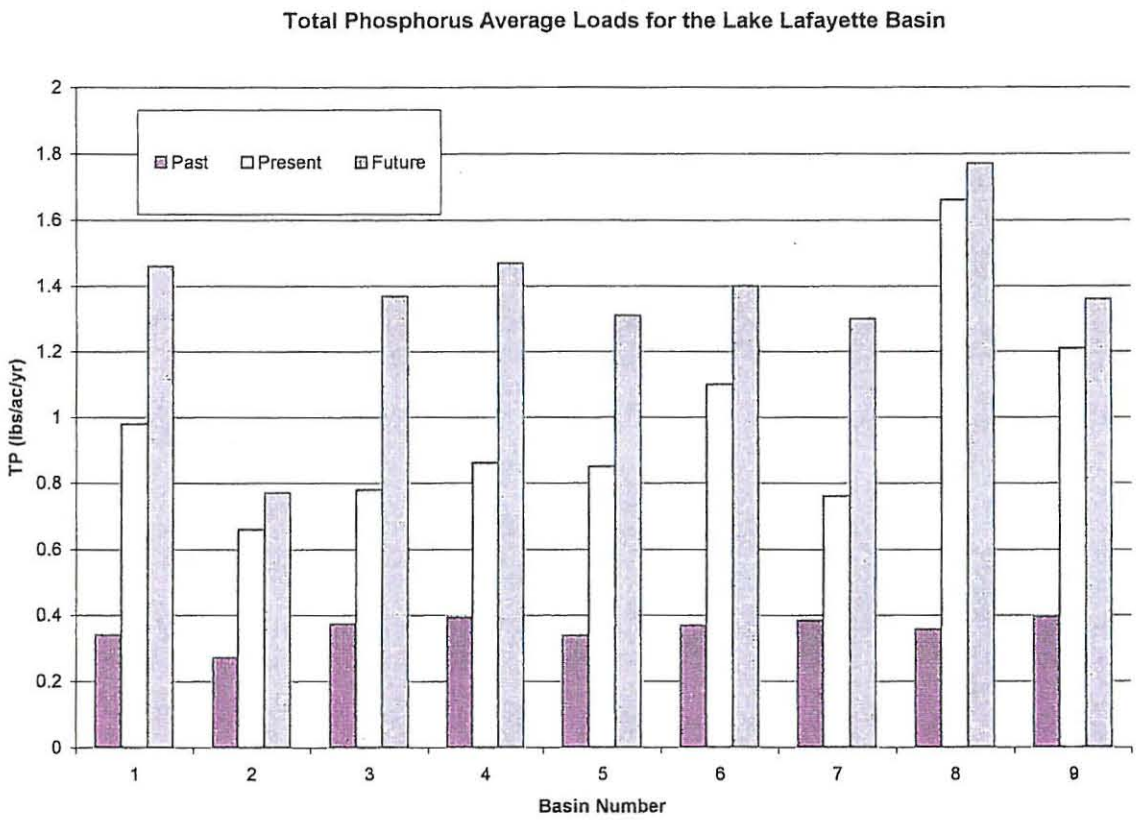


Figure E-4. Simulated phosphorus loads for the past, present, and future in Lake Lafayette sub-basins

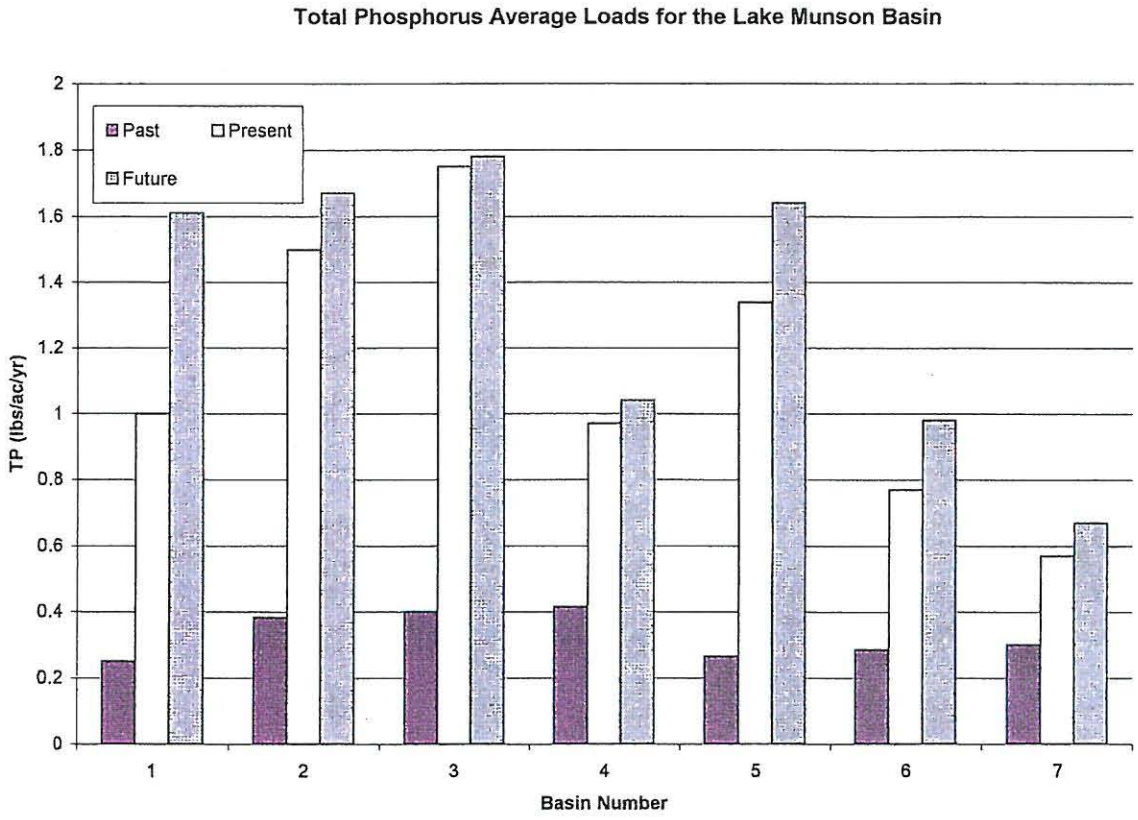


Figure E-5. Simulated phosphorus loads for the past, present, and future in Lake Munson sub-basins

- a 450% increase between the historic annual load and the present load in the Tallahassee Drainage inflow to Upper Lake Lafayette,
- an 80% increase in annual load to Lake McBride in the future. and
- a 70% increase for Killearn Lakes .

Significant simulation results in the Lake Munson basin included:

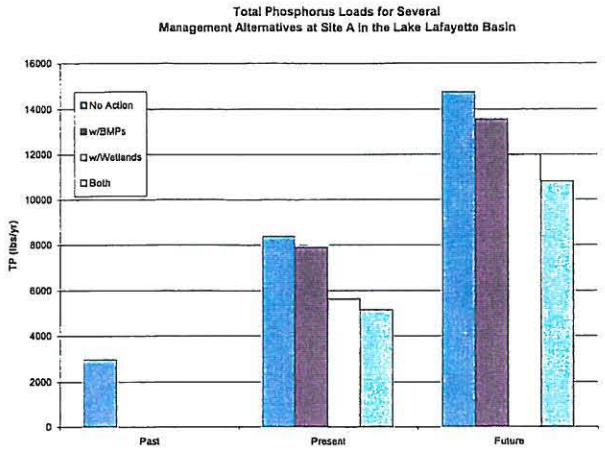
- little change in the pollutant load at the inflow to Bradford Lake between the past, present and future.
- an increase in pollutant load of 367% from past to present day at the inflow to Munson Slough with increases in the future of 30% over present day loads,
- an increase in the Northern Drainage Area of 233% from past to present increasing another 30% over current loads in the future.

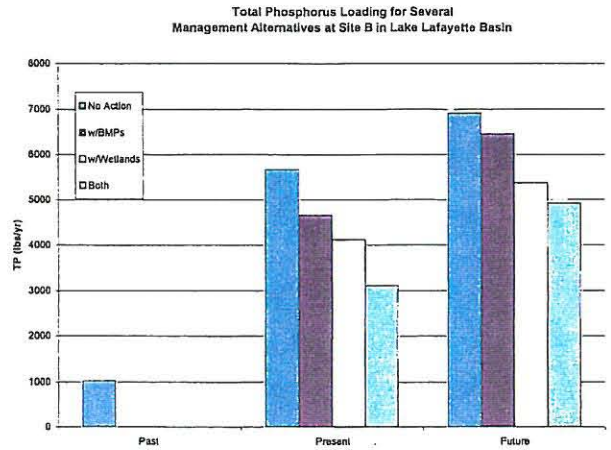
Stormwater Management Alternatives

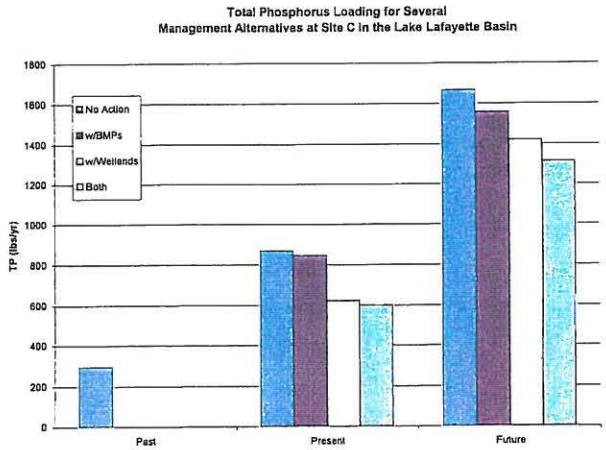
The GIS based simulation model was used to compared several stormwater management alternatives within the Lake Lafayette and Lake Munson basins. The graphs in Figures 6 and 7 show changes in pollutant loading at several locations in the basins and the effects of three management alternatives. In the first alternative, BMP's (street sweeping and swales) are used to reduce pollutant loads by 10% in residential areas and 20% in commercial areas. In the second alternative, wetlands are reconstructed throughout the watersheds to replace wetlands lost over the years to development. And the third alternative is to combine BMP's with the wetlands alternative. BMP's, an effective means of reducing some loads, appeared to reduce loads by about 10 - 15% in urbanized basins. The largest increase in water quality resulted from the wetlands reconstruction alternative, lowering pollutant loads in some cases by more than 50%. The combined approach , provided additional improvement in reducing total load.

Chapter 3. Spatial Modeling of Nitrogen Loading to a Surficial Aquifer from Residential On-site Sewage Disposal Systems in Wakulla County, Florida.
Alan Foley

This study investigated spatial modeling of nitrogen loading to the surficial aquifer resulting from septic tanks Wakulla County. The study focused on five hydrologic sub-basins surrounding Wakulla Springs State Park and the Wakulla River. The area is rapidly urbanizing and has a high potential for groundwater degradation.
A raster-based geographic information system was used to manipulate digital map layers to solve a one-dimensional analytical equation over two-dimensional space using a cell resolution of 30m X 30m. A coarse digital elevation model (DEM) was created by interpolation between 1:100,000 ft scale contours and the assumption was made that the phreatic surface within the surficial aquifer was generally a reflection of the land surface







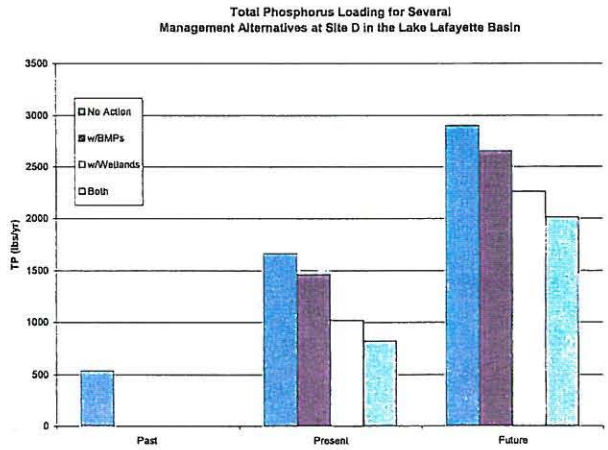
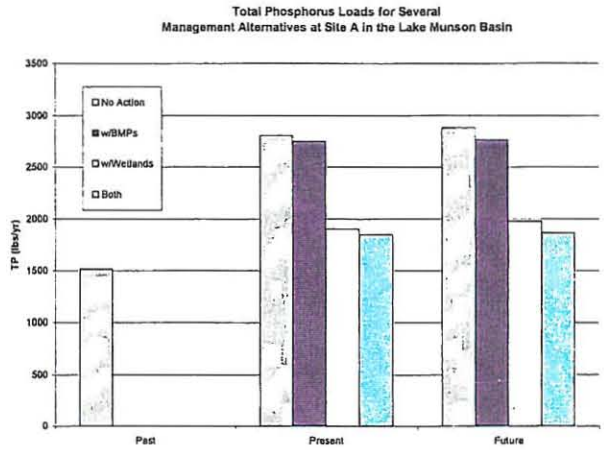
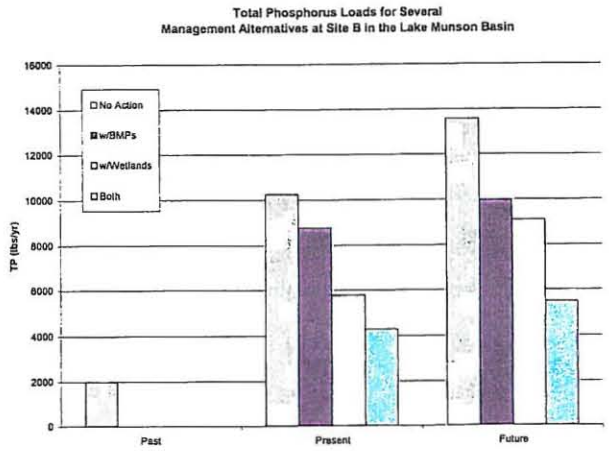


Figure E-6. Change in simulated phosphorus loads that result from BMP's and increases in wetland areas absorbing runoff in Lake Lafayette sub-basins





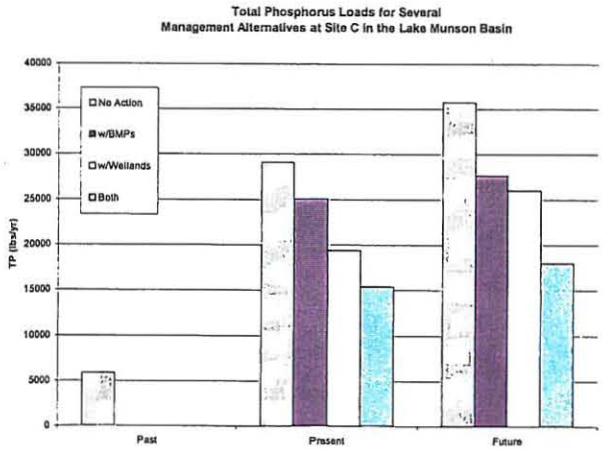


Figure E-7. Change in simulated phosphorus loads that result from BMP's and increases in wetland areas absorbing runoff in Lake Munson sub-basins

A general soils map of the area was used to generate soil hydraulic conductivity and soil carbon content coverages that were necessary for the simulation.

For the most part the simulation results show that the majority of NO_3^- attenuation in the soil occurs within the first 30 to 90 meters surrounding the land use that generates the nitrogen load. Concentrations generally drop to 2 mg NO_3^-/l within 30 to 90 m's of the septic tank source. Figure 8 shows a detail of the nitrogen model output, where the source of nitrogen is in the upper right hand corner. Evident is the drop in nitrogen concentrations within 1 to 3 cells from the source boundary.

The results confirmed that the high spatial and temporal variability of governing parameters – particularly hydrology – poses a significant challenge to modeling subsurface chemical dynamics. Isolating and modeling the effect of a single of nitrogen input source, like septic tanks, within a karst system requires either broad generalizations or a considerable amount of data. Model calibration was difficult due to the lack of and or questionable validity of ground water quality data.

Chapter 4. Spatial Models of total Phosphorus Loading and Landscape Development Intensity in the St Marks River Watershed. Neal Parker

A spatial pollutant load model was developed for total phosphorus (TP) and then model results were correlated with measures of landscape development intensity (LDI) in the St. Marks Watershed. A main feature of the model was to account for the mitigating effects of distance upon TP in stormwaters delivered to water bodies. in the spatial model, and (2) use model results to examine the relationship between development intensity and TP for sub-basins within the St. Marks Watershed.

The model used a geographic information system (GIS) with an overland flow algorithm to predict TP for every 100 meter by 100 meter cell within the watershed. The function of TP absorbed by the landscape with distance controlled the amount of TP that entered each stream. Each cell in the watershed was assigned a literature determined TP loading value dependent upon land use. Calibration against existing data from the watershed was used to specify TP uptake coefficients.

A best fit of modeled with observed TP concentrations ($r^2 = 0.48$) was found by using two distance decay factors, one for locations in the northern half of the watershed (-0.2) and another for the southern half (-0.3). These regions had distinct differences in surface geology and soil types. All combinations of distance decay factors were modeled until a best fit between predicted and observed TP concentrations at 16 locations within the basin were found. Both linear and exponential decay functions were modeled. Final results suggested that the exponential function produced best results, and that lands within 200 – 300 meters of surface water bodies were most influential in determining TP loads to the water bodies. Modeled TP concentrations ranged from background levels of 0.01 mg/l to 0.49 mg/l compared with a range of 0.01 mg/l to 0.87 mg/l for observed data. Model validation was hampered by the lack of available date in the St Marks Watershed.

Five LDI measures were developed from spatial coverages including two physical and three emergy indices. Emergy (spelled with an m) is the energy of one kind required directly and indirectly to make a product. The physical indices were percent impervious

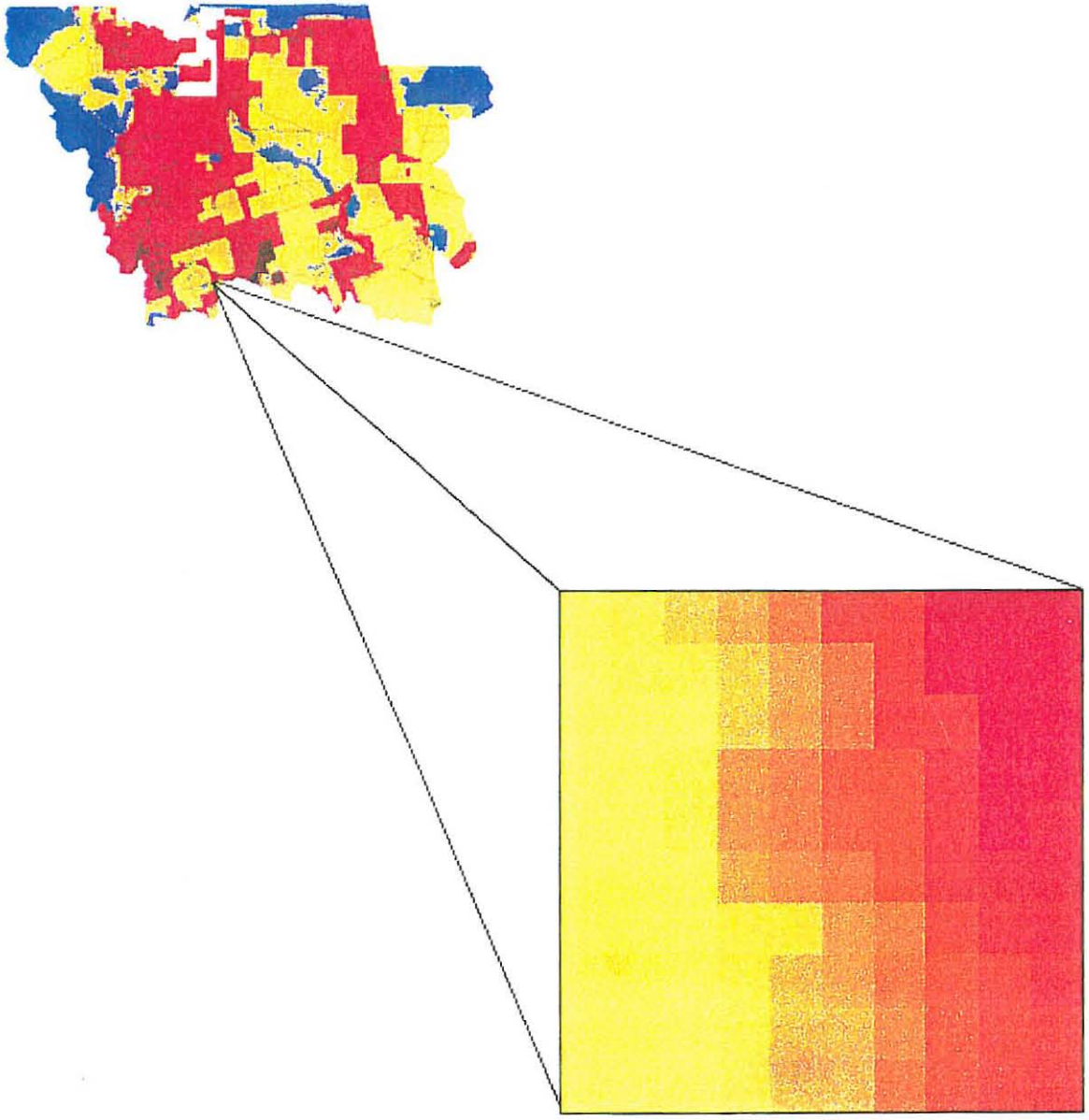


Figure E-8. Detail of the simulated nitrogen plume resulting from a septic tank source in the upper right hand corner. The map at the top shows the Wakulla County study area.

surface and a weighted land use intensity index. The emergy indices were total and developed emergy flow per area (empower density) and a ratio of nonrenewable to renewable inputs to the landscape (environmental loading ratio). These five LDIs were correlated with modeled TP loads for sub-basins of the St. Marks Watershed (Figure 9).

The imperviousness LDI exhibited the strongest correlation with TP ($r^2 = 0.74$ above 10% imperviousness. Weighted land use LDI had the second best fit ($r^2 = 0.67$) above an LDI of about 3.0. Graphs of LDI's vs. TP showed that developed land area (urban and/or agriculture) had to exceed about 25% to 30% of basin land area before TP in surface water bodies was detectable above natural background fluctuations .

In summary several important conclusions were made regarding pollutant load modeling and the relationship between LDIs and TP within the St. Marks Watershed .

1. Pollutant load models can be successfully developed that aggregate many parameters including soils, topography, and imperviousness into a decay coefficient, vastly simplifying the model algorithm and data requirements.
2. The pollutant load model suggested that the most important contributions of pollutant loads come from developed lands within between 200 and 300 meters of surface water bodies.
3. Management efforts may be best focused at locations where the spatial TP model predicted significantly higher TP concentrations than observed concentrations. These may be locations where development induced impacts are just beginning to occur.
4. It appears that there is a development threshold above which TP loads in surface waters are higher than natural background variability. Above this threshold, the LDI's have stronger correlations with TP loads. These thresholds occurred at 10% imperviousness for the imperviousness LDI, 3.5 for the weighted land use LDI, 50E15 sej/ha/yr for the total and developed empower LDIs, and 20 for the ELR LDI. For intensities above the LDI threshold, the imperviousness LDI had the highest correlation with TP loads ($r^2 = 0.60$).
5. Overall, the model suggested that TP loads become apparent above background levels when the area of development exceeds 30% of the total area contributing stormwater runoff to surface waters..

Chapter 5. DEP Presentations

A half-day presentation of project results was conducted in Tallahassee on May 1, 1998. The presentation included theory and principles of spatial analysis and spatial modeling and then we presented results of efforts to model pollutant loading in the St Marks Basin and application of the LDI to resulting model output.

Chapter 6. PROPOSAL: Development of a Spatial Model of Pollutant Loading and Water Quality for Florida Watersheds

The next step in developing a GIS based spatial model of pollutant loading is to statistically validate the model used in the St Marks analysis with water quality data from

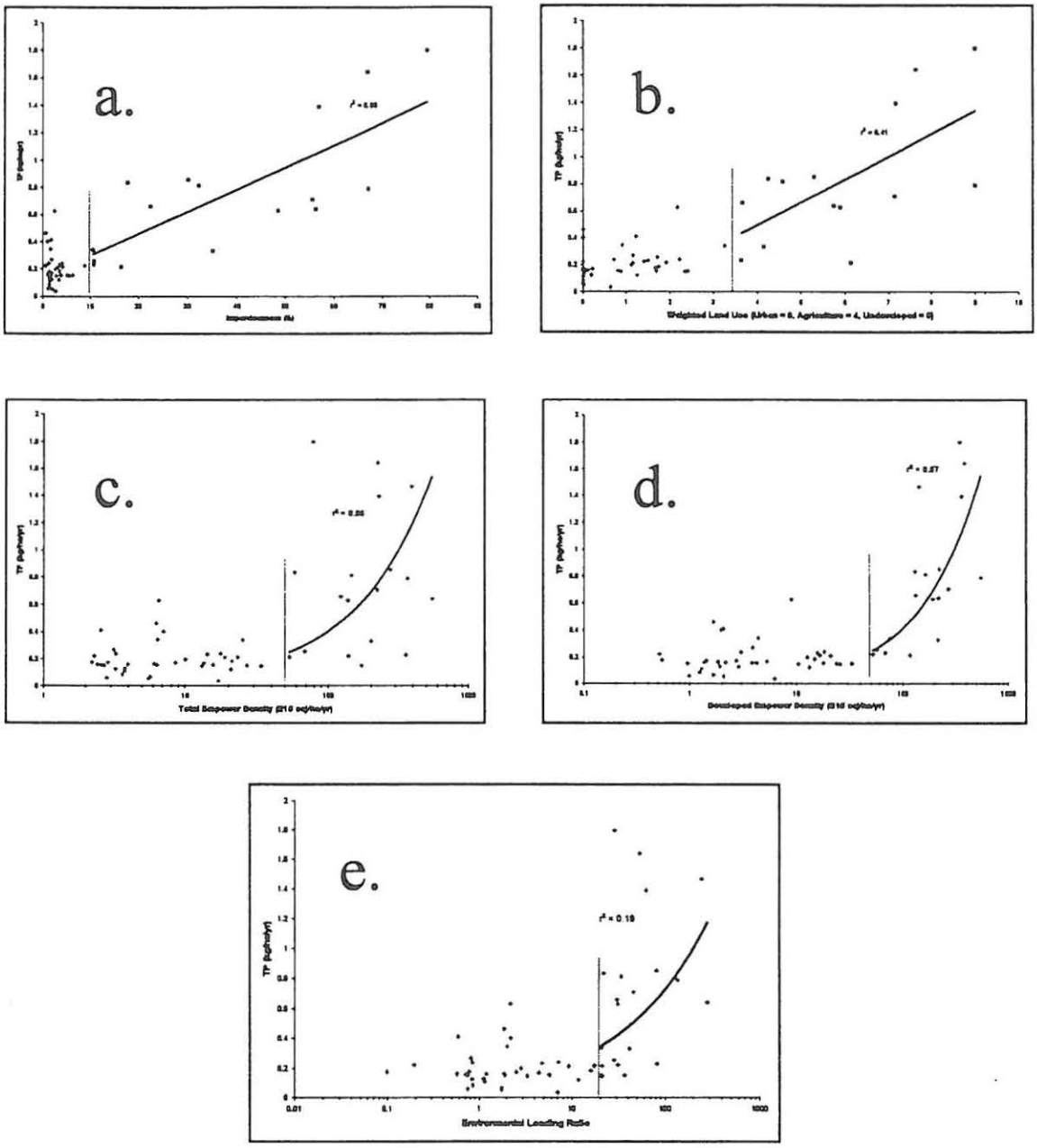


Figure E-9. Landscape Development Intensity indices graphed against simulated phosphorus loads in the St. Marks Watershed. a) Imperviousness, b) weighted land use, c)total empower density, d) developed empower density, e) environmental loading ratio.

around the state. We propose to select a large number of basins (stratified based on ecoregion.) that have good water quality data, and use statistical methods to validate the model, extending the number of parameters included as well as refining the spatial modeling algorithm.

It is our belief that efforts to screen Florida's surface waters for present and future quality problems can be significantly enhanced through the use of a relatively simple, yet sophisticated spatial model in a GIS environment. Rather than detailed computer simulation packages that require hundreds of coefficient s and months of validation before they can be applied to a single basin, the raster based spatial simulation requires few coefficients, and minor amounts of time. The complex simulation packages are good for detailed analysis and where legal challenges dictate scientific scrutiny. The proposed spatial model is needed to screen Florida's waters, and provide needed macro-scale information for the effective allocation of resources for management. The model will be beneficial in the following applications:

1. Estimate degree of impairment of water bodies
2. Spatially locate areas of impairment and identify sources of non point source pollutant generation
3. Direct field monitoring where: (a) the model predicts impairment and (b) where uncertainties are the greatest. In other words, suggest monitoring plan based on model output.
4. Predict future impairment based on changes in land use/land cover
5. Model could be used as the 305B model to estimate % of state water bodies that are within compliance

Chapter 1
Annotated Literature Review

Neal Parker and Alan Foley

St. Marks

Bartel, R., et. al. 1991. Lake Munson Basin Plan - City of Tallahassee and Leon County Stormwater Management Plan Vol. II. NWFWMD.

This report (Volume II) represents a description of the Lake Munson basin stormwater management plan developed by the Northwest Florida Management District in cooperation with Leon County and the City of Tallahassee. The flooding and water quality problems in four major basins in the study area, which included the Lake Munson, Lake Lafayette, Lake Jackson, and Fred George Sink basins. The technical details, which included the use of hydrologic models as a tool for plan development, have been described in a technical report (Volume VI) for this study. An overall summary of the problems identified and the recommended plan for the Lake Munson basin may be found in the Executive Summary (Volume I) of this study.

Bartel, R., et. al. 1991. Stormwater Management Plan for the City of Tallahassee and Leon County Vol. IV - Technical Report. NWFWMD.

This publication represents Volume VI of a series of six reports to document the Stormwater Management Plan for the City of Tallahassee and Leon County. It is provided as a guide for understanding the techniques, data, and principles used for the development of basin management plans in the project area. As such, it is mainly intended for the technical reader having some familiarity with hydrologic analyses and for City and county staff who may be involved in making recommendations to actually implement the basin management plans and updating the plan in the future.

Bartel, Ronald L. and A.E. Maristany, P.E. 1989. Wetlands and Stormwater Management: A Case Study of Lake Munson Part II: Impacts of Sediment and Water Quality. American Water Resources Association. pp. 231-246.

In early 1950, a 255-acre tract of cypress and gum swamp in southern Leon County, Florida was converted from a wetland into Lake Munson. The purpose for this shallow impoundment was to alleviated downstream flooding problems. Unfortunately, the area never reached its full potential as a recreational water body or aquatic habitat due to severe water quality impacts on the lake. The inadvertent discharge of wastewater effluents and stormwater have been the major cause for the degradation of this water body, to the point that in 1982, it was ranked on the trophic state index as the seventh most degraded lake in Florida. In 1984, the wastewater effluents were diverted to a land application system and lake water quality has improved. However the lake continues to suffer algal blooms, fish kills, depressed oxygen levels, and high nutrient and bacterial levels. Restoring ecologically sound conditions to the lake and developing its potential use as a recreation area will require upstream treatment of stormwater and possibly the removal of bottom sediment. Ironically, results show that upstream treatment may require a wetlands area at least the size of Lake Munson.

Bartodziej, W., A.J. Leslie. 1997. Water Hyacinth as a Biological Indicator of Water Quality. FDEP, TSS #97-100, Tall. FL.

Water Hyacinth production was measured against water quality in four north Florida mesotrophic lakes to see if plant growth related to nutrient enrichment. Being a large, conspicuous plant, water hyacinth may be an ideal biological indicator of nutrient enrichment if generalizations can be made about its response to water-column phosphorus and nitrogen concentrations in certain lake types.

Burton, T.M., Turner, R.R., and R.C. Harriss. 1977. Nutrient Export from Three North Florida Watersheds in Contrasting Land Use. Symposium Papers: Watershed Research in Eastern North America Volume II. Edgewater, MD. pp. 323-341.

Exports of nutrients from a forested-agricultural, a suburban, and an urban watershed in north Florida were measured from 1973 to 1975. The most significant impact of urbanization has been the change in temporal distribution and quantity of total and dissolved P exports. The urban watershed exports 16 times more total P than does the forested-agricultural watershed with the suburban watershed intermediate. Further, 98% of this total P is exported in quickflow (stormflow) on the urban watershed while only 53% is exported in quickflow on the forested-agricultural watershed. Dissolved P exports are relatively low from all 3 watershed with the primary effect of urbanization being the temporal changes in export, with 77% of the urban exports being in quickflow compared to 35% for the forested-agricultural watershed. Exports of inorganic N were highest from the suburban watershed, probably as a consequence of septic tank drainage; exports of NO3-N were 6 times higher for the suburban and 3 times higher for the urban, sewered watershed compared to the forested-agricultural watershed. Si and Cl were also monitored. Mechanisms controlling export of P, N, Cl, and Si are emphasized.

Burton, T.M., Turner, R.R., and R.C. Harriss. 1977. Descriptive Hydrology of Three North Florida Watersheds in Contrasting Land Use. Symposium Papers: Watershed Research in Eastern North America Volume II. Edgewater, MD. pp. 211-224.

Results of hydrologic studies on three adjacent watersheds in north Florida representing respectively urban (792 ha), suburban (430 ha) and forested-agricultural (611 ha) land uses support theory and the findings of others that urbanization (1) increases storm peak flows, (2) increases the ratio of quickflow volume to delayed flow volume and (3) increases annual runoff. Total runoff losses from the forested-agricultural, suburban and urban basins during the period July 1973 to June 1975 were respectively, 31.2, 38.1, and 48.3 cm of which 39%, 51%, and 82% were quickflow. Ratios of total quickflow volume to total precipitation over the study period were 0.05, 0.08, and 0.16 for the forested-agricultural, suburban and urban watersheds respectively. comparison of individual storm hydrographs also revealed striking contrasts in the relative magnitude and temporal distribution of streamflow from these watersheds. Maximum discharges during the study period were 0.19 m^3/sec/km^2 (forested-agricultural), 0.75 m^3/sec/km^2 (suburban) and 2.37 m^3/sec/km^2 (urban). When combined with results of contemporaneous water quality studies on the same watersheds these hydrologic findings offer an ideal example of some hydrochemical consequences of urbanization on a small watershed scale.

Clemens, Linda A. 1988. Ambient ground water quality in northwest Florida Part II: A Case Study in Regional Ground Water Monitoring – Wakulla Springs, Wakulla County, Florida. Havana, FL: NWFWMD. Water Resources Special Report 88-1.

Water samples were taken from within the caverns of Wakulla Springs. Sample water quality did not seem to vary considerably for different locations within the spring caverns.

Florida Department of Environmental Protection. 1996 Water-Quality Assessment for the State of Florida: Section 305(b) Main Report. Bureau of Water Resources Protection, Division of Water Facilities, FDEP, Tallahassee, FL.

This report covers water-quality information for the state with a section on basic facts, ecology, and human impacts within the St. Marks River Basin. The two main problem areas noted within the report include Munson Slough and Rattlesnake Branch. Munson Slough drains portions of the Tallahassee urban area and historically received wastewater from small package plants. A partial solution was to divert Tallahassee's treatment plant effluent to land-spreading operation. The second problem area is the St. Marks River downstream from Rattlesnake Branch (lower three to four miles) which received effluent from Seminole Refining corporation and Purdom Power Plant. Near the confluence with the St. Marks, the Wakulla receives nutrients from Boggy Branch. Lake Lafayette may also be an additional problem area which is still under investigation.

Florida. Bureau of Geology. 1972. Environmental geology and hydrology, Tallahassee area, Florida. Tallahassee, FL.

There is a distinct escarpment that separates the coastal lowlands from the Tallahassee Hills which ranges in elevation from sea level to approximately 260 feet. The lowland soils are sandy (immediate infiltration of rain) hence surface runoff is minimal. In the Tallahassee Hills, soils are clayey and several tens of feet thick, over permeable limestone promoting local surface drainage from the hills. Lake Lafayette is one of the headwater tributaries of the St. Marks River. The surface layer is ancient Miocene-Pliocene delta plain. Dissolution of the limestone around Tallahassee has created the three largest lake basins: Lake Jackson, Lake Iamonia, and Lake Lafayette. The St. Marks River has a poorly defined channel north of natural bridge. Additional water from the Floridan Aquifer is added to the river at natural bridge. Cores taken in the area reveal the following layers from shallowest to deepest: Miccosukee, Hawthorn, St. Marks, Suwannee Limestone, and Crystal River Formation. The deeper layers are limestones and dolomites.

Hendry, C.W. Jr., and C.R. Sproul. 1966. Geology and Ground-Water Resources of Leon County, Florida. Tallahassee, Published for The Florida Geological Survey.

The physiography of the gulf coast is generally divided into the highlands and lowlands which are further divided into the delta plain, tertiary highlands, terraced coastal and river valley lowlands. Leon County specifically contains the northern highlands, gulf coastal lowlands, and the river valley lowlands. The Tallahassee Hills are tertiary in age and deltaic in origin. The Miocene-Pliocene delta plain is continually. dissolved by streams and modified by subsurface solution. The Hills are composed of a heterogeneous mix of yellow-orange clays, silts, and sands that are weakly cemented. Soils are loamy and support much vegetation and sediments. The impermeable nature of these soils gives rise to small wet weather ponds and lakes. The gulf coastal lowlands are affected by Pleistocene erosion and deposition. Terraces have been created as a result of fluctuating sea level. Two major units exist in the Gulf Coastal Lowlands: the Appalachicola Coastal Lowlands and Woodville Karst Plain. The Appalachicola Coastal Lowlands, located mostly within the Appalachicola National Forest, are underlain by thick elastic deposits creating flat, sandy surfaces marked by shallow bays, few poorly defined creeks. The underlying sand and clay is less than eighty feet thick. During the rainy season, the area is swampy resulting from the water table approaching the surface. The Woodville Karst Plain ranges from zero to sixty feet in elevation. Characterized by loose quartz sands over a limestone substrata, the topography is one of sinkholes and sand dunes. Vegetation mostly consists of pines, black-jack, turkey oak, and cypress and bays in lower areas. Most streams disappear below ground. The St. Marks River Valley has a poorly perceptible flood plain valley due to the stream flowing upon slightly incised bedrock with a thin veneer of loose quartz sand. During high waters, Lake Lafayette contributes to St. Marks River. The limestone of this region becomes soluble with atmospheric CO2 and organic acids. The dense dolomite is least soluble while the porous calcitic limestones are most readily soluble.

Highley, Bradley A. et al. 1994. Recent sediments of the St. Marks River Coast, Northwest Florida, a low-energy, sediment-starved estuary. Abstracts with Programs - Geological Society of America. 26(4):20.

The St. Marks river of northwest Florida drains parts of the central panhandle of northwestern Florida, and a small area in southwestern Georgia. It traverses nearly 56.3 kilometers through a watershed of 1711 square kilometers. The slow-moving river carries little sediment and terminates in Apalachee Bay, a low-energy embayment in the northeastern most gulf of Mexico. The coastal region is characterized by mudflats, seagrass beds, and an absence of sandy beaches and barrier islands. Clastic sediments of the coast and shelf rest on a shallow-dipping carbonate platform. The upper surface of the platform is locally karstic. As a result, like other rivers in this region of northwest Florida, the St. Marks watershed is marked by sinkholes and disappearing streams. The fact that the river travels underground through part of its lower watershed serves to trap or sieve some of its clastic load. In the estuary, the undulating karst topography causes the estuarine sediments to vary in thickness from 0 to 4+ meters. In places, in both the estuary and lower river valley, the Tertiary carbonate units are exposed at the surface. The concave shape of the coastline and its orientations with respect to prevailing winds result in low average wave energy. Sedimentation is therefore controlled by riverine and tidal forces. The relatively low energy conditions result in good preservation of the sedimentary record in the St. Marks estuary. A suite of sediment cores has been collected in the lower river, estuary and adjacent Gulf of Mexico. Lead-210 dating results indicate a slow average sedimentation rate (~1mm/yr). Investigation of sedimentation rates and sediment characteristics (grain size trends and clay mineralogy) over time in the St. Marks estuary indicates that sedimentologic conditions in this low-energy environment have been relatively stable during the recent geologic history of the estuary.

Katz, B.G. and A.F. Choquette 1991. Aqueous geochemistry of the sand-and-gravel aquifer, Northwest Florida. Ground Water. 29(1):47-55.

The aqueous geochemistry of the sand-and-gravel aquifer in northwest Florida was characterized as part of the Florida Ground-Water Quality Monitoring Network Program, a multiagency cooperative study delineating baseline and/or background water quality for the major aquifer systems throughout the State. The aquifer is the principal source of water in northwest Florida and consists predominantly of quartz sand with smaller amounts of andesine, chlorite, calcite, kaolinite, and illite.

Livingston, Robert J., ed. 1991. The Rivers of Florida. New York: Springer-Verlag, Inc.

Discusses generalities of many Florida rivers. Discusses rivers of the northern gulf in some detail with mention to the St. Marks River and its underlying strata.

Macesich, M. and O.J. Kenneth. 1989. Uranium isotopic study of Wakulla Springs. Geological Society of America, Southeastern Section, 38th annual meeting. 21(3):49.

Uranium isotopes have been used as natural tracers in many hydrologic regimes. The identities of differing water sources are discriminated by their uranium (U238) concentrations and their isotopic activity ratios (U234/U238). This method was used to identify the multiple sources of water contributing to Wakulla Springs, a first magnitude spring located within the Woodville Karst Plain and flows through the Floridan Aquifer. Wakulla Springs has the greatest known range of discharge of all Florida springs but averages 390 cubic feet per second. Six samples were taken from different deep natural tributary conduits, from as much as 4600 feet from the spring mouth (up to 270 feet in depth). Surface and local water samples were tested for their uranium signatures. Preliminary results suggest that the conduits which feed the spring have both shallow and deep sources. There is a noticeable temporal isotopic variation in the spring water suggesting a varying degree of contribution among the conduits. Comparison of surface discharge and deep water samples suggest that a shallow seep component may also be involved.

Maristany, Agustin, et. al. 1988. Water Quality Evaluation of Lake Munson – Leon County Florida. Water Resource Assessment 88-1. NWFWMD.

This report details the history of Lake Munson's water quality since its formation from a cypress swamp in 1950. Many water quality parameters were looked at and how the lakes condition has deteriorated as a result of increased levels in some of these parameters. Finally, restoration techniques that would give greatest improvements to the lake are recommended.

Maristany, Agustin E. and R.L. Bartel. 1989. Wetlands and Stormwater Management: A Case Study of Lake Munson Part I: Long-Term Treatment Efficiencies. American Water Resources Association. pp. 215-229.

The use of wetlands or wet detention ponds for stormwater management represents a relatively new approach which has been successfully applied in recent years to address water quality problems in urban areas. Since most systems have been in operation for only a few years, questions have been raised concerning their long-term performance. It has been speculated that once these systems reach a state of dynamic equilibrium, nutrient removal may decline due to the reduced nutrient uptake of a mature ecosystem. This paper sheds some light on the subject based on a recent study by the Northwest Florida Water Management District of a 255-acre wetland/lake system which has received wastewater effluent and stormwater discharges for over 30 years. Nutrient and pollutant removal rates were estimated for a wide range of parameters based on concurrent sampling of stormwater inflows, outflows, and lake water quality. Long-term removal rates for Lake Munson compared favorably with rates reported for relatively new facilities.

Rupert, F. and Steve Spencer. 1988. Geology of Wakulla County, Florida. Bulletin No. 60. Tallahassee, Florida Geological Survey.

Wakulla County is located in the Gulf Coastal Lowlands amid poorly drained pine flatwoods, swamps, and river basins. The average slope of this region is about four feet per mile. There are about five relict marine beach ridges based on topographical elevation due to changing sea elevation (from shore inland): Silver Bluff Terrace, Pamlico Terrace, Talbot Terrace, Penholoway Terrace, and Wicomico Terrace. The Woodville Karst Plain is bounded on the west by Appalachicola Coastal Lowlands. Less than twenty feet of quartz sand lies on the karstic St. Marks and Suwannee Limestone. The St. Marks River headwaters is located in the Tallahassee Hills of eastern Leon County. Seven major springs exist in Wakulla County: Indian Springs, Kini, Newport, Panacea Mineral, River Sink, Wakulla, and Spring Creek.

Stephens, D.W., B. Waddell, and J.B. Miller. 1988. Reconnaissance Investigation of Water Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage in the Middle Green River Basin, Utah, 1986-1987. U.S. Geological Survey Water-Resource Investigations Report 88-4011. Salt Lake City, Utah.

Reconnaissance of wildlife areas in the middle Green River basin of Utah was conducted during 1986 and 1987 to determine whether irrigation drainage has caused, or has the potential to cause significant harmful effects on human health, fish, and wildlife, or may adversely affect the suitability of water for beneficial uses. Studies at Stewart Lake Waterfowl Management Area and Ouray National Wildlife Refuge indicated that concentrations of boron, selenium, and zinc in water, bottom sediment, and biological tissue were sufficiently large to be harmful to fish and wildlife, and to adversely affect pesticides in surface water generally were less than established standards with the exception of gross alpha radiation, which exceeded by factors of three to five times the standard of 15 picocuries per liter in water from two of the drains discharging into Stewart Lake.

Swanson, Sloan, and Chernets. 1996. Lake Lafayette Management: A Report Outlining Lake Shore, In-Lake, and Land Use Management Proposals. Dept. of Growth and Environmental Management, Tallahassee, FL.

This report addresses the degradation of Lake Lafayette by urban stormwater and point source discharge. Localities around the lake of greatest concern are highlighted. The report stresses the need for an immediate and comprehensive management program for Lake Lafayette. The offered management program consists of strategies for land use controls, in-lake management and clean-up, and subsequent research and monitoring.

Turner, R.R., et al. 1975. The Effect of Urban Land Use on Nutrient and Suspended-Solids Export from North Florida Watersheds. Florida State University, Tallahassee Dept. of Oceanography. in 'Mineral Cycling in Southeastern Ecosystems', (Conf-740513), p.868-888.

Two watersheds of similar size, geomorphology, and pedology representing forested-agricultural and residential-commercial (urban) land use were hydrologically instrumented to obtain comparative nutrient and suspended-solids export data. Constituents measured included suspended solids, total dissolved solids, dissolved silicon, and dissolved nutrients (nitrogen and phosphorus). Intensive hydrochemical analysis of runoff from 13 storms in the urban watershed and 8 storms in the forested watershed demonstrated a strong contrast in the magnitude and temporal distribution of nutrient and suspended-solids concentrations and exports. Suspended-solids concentration and export were directly dependent on stream discharge. Although concentrations of dissolved constituents were generally inversely dependent on stream discharge, export of dissolved constituents were directly dependent on stream discharge. Higher cumulative stream discharge in the urban watershed thus exported higher total storm loads of all constituents except dissolved silicon. Exports of suspended solids, total dissolved solids, and all the dissolved nitrogen species from the urban watershed were higher than the higher volume of stream discharge in this watershed might otherwise indicate, suggesting significant additional sources of these constituents in the urban watershed. Exports of dissolved phosphorus from the urban watershed were also higher but near what the higher volume of stream discharge might indicate. Exports of dissolved silicon from the urban watershed were lower than from the forested watershed despite the higher volume of stream discharge in the urban watershed. Observed differences in exports were related to the changes in hydrology associated with urban development, i.e., in streamflow rate, total volume of stream discharge, and the relative significance of various pathways of water movement, as well as to increased diffuse anthropogenic inputs in the urban watershed. Comparison of material loads exported by storm flow and low flow in each watershed suggested increased significance of storm events in materials export in the urban watershed.

Winchester, J.W., et al. 1995. Atmospheric deposition and hydrogeologic flow of nitrogen in northern Florida watersheds. Geochimica et Cosmochimica Acta. 59(11):2215-2222.

Atmospheric wet and dry deposition ("acid rain") appears to be the principal source of nitrogen in twelve northern Florida watersheds that range from Pensacola to Gainesville (Escambia to Alachua Counties). The study was based on statistical analysis of chemical concentrations measured for more than ten years in weekly rainfall samples of the National Atmospheric Deposition Program, NADP, and more than twenty years of river water samples of the US Geological Survey, USGS. River fluxes of total dissolved nitrogen average close to the atmospheric deposition fluxes of nitrate and ammonium ions. Factor analysis was applied to the datasets to resolve principal components: (1) in atmospheric data, that distinguish air pollution nitrate and sulfate from sea salt sodium and chloride, and (2) in surface water data, that distinguish ground water Ca, Mg, and silica from metioric water nitrate and sulfate. Relationships within the sets of measured concentration data suggest that , following atmospheric deposition, inorganic nitrogen undergoes biogeochemical transformation within the watersheds, which results in inorganic nitrogen being transformed to organic forms. River concentration ratios N/P in the watersheds are high, averaging twice the Redfield mole ration N/P=16 for aquatic plant nutrients. The results indicate that excess dissolved nitrogen could be temporarily recycled in the watersheds but not retained, so that it could eventually flow to the coastal zone where N may be a limiting nutrient for marine plants. Chemical interactions of meteoric water within the watersheds depend on geologic, hydrologic, and biogeochemical processes and are certainly complex. However, in one watershed that is geologically the simplest, separate statistical

analyses of river water composition during high and low flow conditions show nitrate and sulfate to be correlated during high flow, but not during low flow, providing further evidence for an atmospheric nitrogen source and watershed transformation after deposition.

Wooten, Nicholas, et. al. July 1991. Lake Lafayette Basin Plan – Stormwater Management Plan Vol. IV. NWFWMD.

This report (Volume IV) represents a description of the Lake Lafayette Basin Stormwater Management Plan developed by the Northwest Florida Management District in Cooperation with Leon County and the City of Tallahassee, Florida. The plan developed represents the culmination of a Five-Year Study to identify flooding and water quality problems in four major basins in the study area which included the Lake Munson, Lake Lafayette, Lake Jackson, and Fred George Sink Basins. The technical details, which included the use of hydrologic models as a tool for plan development have been described in a technical report (Volume VI) for this study. An overall summary of the problems identified and the recommended plan for the Lake Lafayette Basin may be found in the Executive Summary (Volume I) of this study.
Yon, J. Williams. 1966. *Geology of Jefferson County, Florida.* Tallahassee, Published for the Florida Geological Survey. This book provides a detailed description on the geomorphology of Jefferson County.

Nutrients

Alfoldi, Laszlo, 1983. Movement and Interaction of Nitrates and Pesticides in the Vegetation Cover – Soil Groundwater – Rock System. Environmental Geology vol. 5 no. 1, pp. 19-52.

Review of solute dynamics primarily in the vadose zone. Divides soil into three areas. (1) Near the root zone – constituent transformation system. (2) Between the root zone and water table – constituent transportation system. (3) In the water table – constituent accumulation system. Water provides horizontal transportation in area (3). In aquifers under natural conditions the lateral movement of water will not be significant, excluding the flow through karstic formations, and consequently the infiltrated pollution propagates only slowly and accumulates relatively rapidly. As a result of water intake (pumpage) operations, flow will turn toward a well with accelerating velocities, and pollution may rapidly reach the place of water intake.

Anderson, LJ and H Kristiansen, 1984. Nitrate in Groundwater and Surface Water Related to Land Use in the Karup Basin, Denmark. Environmental Geology, vol. 5, no. 5, pp. 207-212.

Investigation and analysis of the content and distribution of nitrate in the groundwater and surface water in the Karup Basin area. Soil profiles indicate (1) an upper oxidation zone where nitrate is present and iron is absent and (2) a lower reduction zone where iron is present but nitrate is absent. In cultivated areas with high nitrate concentrations, the oxidation zone is found at depths of 8-15m below the groundwater table. In forested areas, the oxidized nitrate zone is thinner, 5-7 m, and has a low nitrate content.

Asbury, C.E. and E.T. Oaksford. 1997. A comparison of drainage basin nutrient inputs with instream nutrient loads for seven rivers in Georgia and Florida, 1986-90. U.S. Geological Survey, Water-Resources Investigation Report 97-4006.

Instream nutrient loads of the Altamaha, Suwannee, St. Johns, Satilla, Ogeechee, Withlacoochee, and Ochlockonee River basins were computed and compared with nutrient inputs for each basin for the period 1986-90. Nutrient constituents that were considered included nitrate, ammonia, organic nitrogen, and total phosphorus. Sources of nutrients considered for this analysis included atmospheric deposition, fertilizer, animal waste, wastewater-treatment plant discharge, and septic discharge. Although instream nutrient

loads constitute only one of the various pathways nutrients may take in leaving a river basin, only a relatively small part of nutrient input to the basin leaves the basin in stream discharge for the major coastal rivers examined in this study. The actual amount of nutrient transported in a river basin depends on the ways in which nutrients are physically handled, geographically distributed, and chemically assimilated within a river basin.

Ayres and Associates, July, 1989. Onsite Sewage Disposal System Research in Florida– Performance Monitoring and Ground Water Quality Impacts of OSDS's in Subdivision Developments. Report to the State of Florida Department of Health and Rehabilitative Services. Tampa, Florida.

Progress report presenting the first results of field monitoring of ground water below unsewered subdivisions and monitoring of the performance of individual OSDS's. Ground water was monitored beneath four specific subdivisions in four different hydrogeologic regimes. The study sites were selected to represent subdivisions developed under the requirements of the 1983 revisions to Chapter 10D-6, Florida Administrative Code, Standards for Onsite Sewage Disposal Systems. The study areas were also intended to reflect the various hydrogeologic regimes found in Florida. Findings reavealed that septic tank effluent (STE) in Florida was similar in character to effluent generated in other areas of the U.S. Notable exceptions were the lower concentrations of most constituents in septage and the higher suspended solids in STE. These may be attributed to the comparitively higher temperatures of both waste streams. Ground water quality in the vicinity of relatively new subdivisions (Less than 20 years old) served by individual OSDS's had not suffered substantial widespread contamination. Localized areas of potential impact were observed. If subdivision impacts are to occur, they may take decades to manifest themselves due to the low ground water seepage velocities at the test sites. Report contains a synopsis of prior research as well as details of the methods and results of the investigation.

Ayres and Associates, March, 1993. Onsite Sewage Disposal System Research in Florida– The Capability of Fine Sandy Soil for Septic Tank Effluent Treatment: A Field Investigation at an In-Situ Lysimeter Facility in Florida. Report to the State of Florida Department of Health and Rehabilitative Services. Tampa, Florida.

Investigation into the treatment capabilities of fine sandy soils common to Florida under controlled experimental conditions in the field. A unique field lysimeter research facility was designed, constructed, and operated under controlled conditions. Key variables investigated were the thickness of unsaturated soil below the wastewater infiltration system and the hydraulic loading rate of effluent to the system. The report describes lysimeter site selection and characterization, investigative methodology, and the results of the lysimeter facility monitoring for the first six months of operation. Preliminary results showed substantial attenuation of key parameters related to septic tank effluent (STE) treatment in the fine sandy soils. Total phosphorous removal was high. There were indications that the phosphorous capacity of the soil was being approached. It appeared that phosphorous removal was effected negatively by less unsaturated zone travel and greater hydraulic loading. Total organic carbon reduction was on the order of 80 percent. Total Kjeldahl nitrogen reduction was in excess of 97 percent indicating almost complete nitrification of STE nitrogen. The nitrate produced by nitrification was transported through the system in relatively high concentrations.

Ayres Associates, 1987. The Impact of Florida's Growth on the Use of On-Site Sewage Disposal Systems. Report to the Florida Department of Health and Rehabilitative Services. Tampa, Florida.

Summaries of the major soil types in Florida and their distribution; critical characteristics of the soils affecting their ability to accept and treat on-site disposal system (OSDS) effluent; soil types supporting most of the current and future OSDS installations in Florida; and the density and geographic distribution of OSDS designs most commonly used in Florida.Results indicate a continuing trend of high OSDS use in

urban fringes and the increasing use of soils with severe limitations for proper operation of conventional OSDS designs. Much of the suitable soil in many counties has been developed and there is increasing pressure to utilize alternative OSDS designs on inadequate soils. The characteristics that are expected to limit the ability of these soils are saturated conditions under and around drainfields due to a high ground water table and excessive permeability of sandy soils.

Ayres Associates, February, 1993. An Investigation of the Surface Water ContaminationPotential From On-Site Sewage Disposal Systems (OSDS) in the Turkey Creek Sub-basin of the Indian River Lagoon. Report to St. Johns River Water Management District and the Florida Department of Health and Rehabilitative Services. Tampa, Florida.

Study conducted as part of the State of Florida's Surface Water Improvement and Management Program. The study assessed the impact of several existing OSDS on water quality in adjacent canals that eventually drain to the Indian River Lagoon. Ground water and surface water quality investigations were conducted around two existing OSDS and one control area. Pollutant loading to the ground water and adjacent drainage canals was estimated. Ground water flow characteristics and seepage rates into the canals were determined. Recommended estimation of a preliminary nutrient budget to ascertain whether nutrient reduction techniques for OSDS should be initiated.

Ayres Associates, March, 1993. Onsite Sewage Disposal System Research in Florida:An Evaluation of Current Onsite Sewage Disposal System (OSDS) Practices in Florida. Report to the Florida Department of Health and Rehabilitative Services. Tampa, Florida.

Chapter 1 Overview of the onsite wastewater system regulatory program in Florida. Evaluation of onsite wastewater treatment system performance. Summary of OSDS performance monitoring in Florida. Technical guidelines for site evaluation, system design, construction, operation, and maintenance. Review of regulations and enforcement. Recommendations for overall improvement of systems and regulations.

Bartholomew, WV and FE Clark (eds.), 1965. Soil Nitrogen. American Society of Agronomy. Madison Wisconsin.

Text reviewing nitrogen geochemistry.

Battoe, LE and EF Lowe, 1992. Acidification of Lake Annie, Highlands Co., FL. Water, Air, and Soil Pollution 65:69-80.

Lake Annie is a clear-water seepage lake in south-central Florida, remote from significant pollutant sources. It is suggested that the lake's acidification was a threshold phenomenon wherein, following depletion of the watershed's buffering capacity, acidification of the lake was rapid.

Berndt, MP, 1990. Sources and Distribution of Nitrate in Groundwater at a Farmed Field Irrigated with Sewage Treatment-Plant Effluent, Tallahassee, Florida. US Geological Survey Water Resources Investigations Report 90-4006. Tallahassee, Florida.

Effluent from a secondary sewage-treatment plant and fertilizers containing inorganic nitrogen were applied in conjunction with the operation of a commercial farm. Water samples indicated that conversion of nitrogen species to nitrate was complete before the nitrogen-enriched water reached the water table. Water samples from monitoring wells inside the sprayfield had nitrate concentrations in excess of the drinking water standard of 10 mg/L. Samples from wells outside of the sprayfield had background levels of nitrate. Isotopic analysis indicated that nitrogen contributions from fertilizers was significant in some test areas.

Bicki, Thomas J. et al, 1984. Impact of On-Site Sewage Disposal Systems on Surface and Ground Water Quality. Report to Florida Department of Health and Rehabilitative Services, Institute of Food and Agricultural Sciences. Gainesville, Florida.

Brief description of on-site disposal systems. Characterization of septic tank effluent constituents of concern - nitrogen, phosphorous, chlorides, sulfates, sodium, detergent MBAS, toxic organics, bacteria, and viruses. Review of constituent fate and transport, water quality surveys, ground water monitoring studies, lysimeter, sand filter, and column studies. Summary of on-site sewage disposal system density, ground water resources and research needs in Florida. Extensive list of references.

Boggess, C.F. 1994. The Biogeoeconomics of Phosphorus in the Kissimmee Valley. PhD dissertation, Environmental Engineering Sciences. Univ. of Florida, Gainesville. 234 pp.

This study tested ways of relating a chemical cycle to a regional economy, using phosphorus in the northern drainage basin of Lake Okeechobee, Florida as an example. Among the methods used were a mass balance approach to phosphorus budgeting, dynamic modeling for runoff simulation, cost effectiveness for economic analysis, and emergy evaluation techniques. Spatial data on land use and management practices were organized using a geographic information system. Phosphorus management scenarios were evaluated and compared in terms of their ability to meet alternative goals for the region: physical (i.e., target phosphorus load reduction to Lake Okeechobee); economic (i.e., minimize cost of phosphorus reduction); and energetic (i.e., maximize regional empower). Five principles for the new field of biogeoeconomics were proposed for managing at the interface between an elemental cycle, its role in the environment, and its economic use to enhance the self-organizing properties of the landscape.

Bradley, PM, CM Aelion, and DA Vroblesky, 1992. Influence of environmental factors on denitrification in sediment contaminated with JP-4 jet fuel. Ground Water, vol. 30, no. 6: 843-848.

Attempted to identify factors likely to influence microbial activity under denitrifying conditions in a shallow aquifer contaminated by an 83,000 gallon jet fuel spill. Examined the fate of amended NO_3, the effect of pH, NO_3 and PO_4 on denitrification, and the variability of denitrification in sediments collected at the site. Denitrification rates were at least 38% lower at pH=4 than observed at pH=7.

Brooks, R.P. et. al. 1989. A Methodology for Biological Monitoring of Cumulative Impacts on Wetland, Stream, and Riparian Components of Watersheds. Paper presented and submitted for inclusion in the proceedings of the International Symposium: Wetlands and River Corridor Management, Charleton, SC.

Biotic communities were compared between two watersheds in the Ridge and Valley Province of central Pennsylvania. An undisturbed watershed served as a reference for comparisons against a similar watershed that was disturbed by agricultural and developmental activities.

Byron, E.R. 1989. Land-use and water quality in tributary streams of Lake Tahoe, California-Nevada. Journal of Environmental Quality. 18:84.

Concluded that land use usually affects water quality to a greater degree than geomorphology or soil types of the drainages. Long-term average nutrient flux originating from non-point sources closely reflects intensity and location of development in a watershed. Disturbances that affects the most errosive areas of watershed has the greatest affect on water quality.

Canter, Larry W, 1996. Nitrates in groundwater. Lewis Publishers. Boca Raton, Florida.

Excellent text. Reviews nitrogen dynamics in groundwater, sources, fate, pollution control, modeling, and remediation.

Correll, D.L., T.E. Jordan, and D.E. Weller. 1992. Nutrient flux in a landscape: effects of coastal land use and terrestrial community mosaic on nutrient transport to coastal waters. Estuaries. 15:431-442.

Long-term interdisciplinary studies of the Rhode River estuary and its watershed in the mid-Atlantic coastal plain of North America have measured fluxes of nitrogen and phosphorus fractions through the hydrologically-linked ecosystems of this landscape. These ecosystems are upland forest, cropland, and pasture; streamside riparian forests; floodplain swamps; tidal brackish marshes and mudflats; and an estuarine embayment. Croplands discharged far more nitrogen per hectare in runoff than did forest and pastures. However, riparian deciduous hardwood forest bordering the cropland removed over 80 percent of the nitrate and total phosphorus in overland flows and about 85 percent of the nitrate in shallow groundwater drainage from cropland. Nevertheless, nutrient discharges from riparian forests downslope from croplands still exceeded discharges from pastures and other forests. The atomic ratio of nitrogen to phosphorus discharged from the watersheds into the estuary was about 9 for total nutrients and 6 for inorganic nutrient fractions. Such a low N:P ratio would promote nitrogen rather than phosphorus limitation of phytoplankton growth in the estuary. Estuarine tidal marshes trapped particulate nutrients and released dissolved nutrients. Subtidal mudflats in the upper estuary trapped particulate P, released dissolved phosphate, and consumed nitrate. This resulted in a decrease in the ratio of dissolved inorganic N:P in the estuary. However, the upper estuary was a major sink for total phosphorus due to sediment accretion in the subtidal area. bulk precipitation accounted for 31 percent of the total nongaseous nitrogen influx to the landscape, while farming accounted for 69 percent. Forty-six percent of the total non-gaseous nitrogen influx was removed as farm products, 53 percent either accumulated in the watershed or was lost in gaseous forms, and 1 percent entered the Rhode River. Of the total phosphorus influx to the landscape, 7 percent was from bulk precipitation and 93 percent was from farming. Forty-five percent of the total phosphorus influx was removed as farm products, 48 percent accumulated in the watershed, and 7 percent entered the Rhode River. These nitrogen and phosphorus discharges into the Rhode River, although a small fraction of total loading in the watershed, were large enough to cause seriously overenriched conditions in the upper estuary.

Degen, MB, RB Reneau, Jr., C Hagedorn, and DC Martens, 1991. Denitrification in Onsite Wastewater Treatment and Disposal Systems. Virginia Water Resources Research Center Bulletin 171. Blacksburg, Virginia.

Study evaluating the effects of effluent type, effluent loading rate, dosing interval, and temperature on denitrification in onsite wastewater treatment and disposal systems. From the study, a model was developed that predicted the mean nitrous oxide production for each combination of the experimental treatments. The results of the study and the model indicate that denitrification can be enhanced in OSWTDS's by the application of anaerobic effluent at the Virginia Department of Health's recommended effluent loading rate to surface soil horizons using a 48-hour dosing interval.

Dillon, P.J., and W.B. Kirschner. 1975. The effects of geology and land use on the export of phosphorus from watersheds. Water Research. 9:135-148.

The export of total phosphorus from 34 watersheds in Southern Ontario was measured over a 20-month period. The annual average export for igneous watersheds (i.e., those of the Canadian Shield) that were forested was 4.8 mg/m^2/yr, significantly different from the average (11.0 mg/m^2/yr) for watersheds that included pasture as well as forest. Similarly, on sedimentary rock, the mean export from forested

watersheds (10.7 mg/m^2/yr) differed significantly from those with forest and pasture (28.8 mg/m^2/yr). The difference between watersheds of different geology but similar land use were also highly significant. Additional data from the literature supported our conclusions. Other forested igneous watersheds of plutonic origin averaged 4.2 mg/m^2/yr of total phosphorus exported; forested igneous watersheds of volcanic origin, however, averaged 72 mg/m^2/yr. The overall average export from each type of watershed as classified by geology and land use was very similar to that for the same classification found in our study. The effects of agriculture and urbanization were to greatly increase the total phosphorus exported. Wide ranges of values probably reflet the intensity of land use.

Eisenreich, SJ (ed.), 1981. Atmospheric Pollutants in Natural Waters. Ann Arbor Science Publishers Inc. Ann Arbor, Michigan.

Text reviewing all aspects of atmospheric pollution. Discussions of nitrogen and phosphorous deposition in Florida, modeling of atmospheric removal processes, metals in the atmosphere, and anthropogenic inputs.

Farnworth, E. G. et al. 1979. Impacts of Sediment and Nutrients on Biota in Surface Waters of the United States. Environmental Research Laboratory, Office of Research and Development, U.S. EPA. Athens, Georgia. 134 pp.

A review of research on the impacts of sediment, nitrogen, and phosphorus on aquatic biota was performed to determine the influences of sediment and nutrients on biota, to suggest directions for future research, and to provide suggestions for management of freshwater systems across the United States. This report is divided into two sections. The first section provides an organization and background information to enable incorporation of large amounts of available information and allow assessment of impacts at several hierarchical levels. Included are a hierarchical scheme which is the foundation of the analytical study; a regional analysis of the concentrations of sediment, nitrogen, and phosphorus in surface waters; a review of biotic impact assessment approaches; and a review of modelling of sediment and nutrient impacts. The second section reviews the impacts of sediment, nitrogen, and phosphorus on biota, integrates this information into the hierarchical scheme developed in the first section, and shows how the hierarchical scheme can be used for impact analysis.

Florida Geological Survey, 1988. Geology of Wakulla County, Florida. Florida Geological Survey Bulletin No. 60. Tallahassee, Florida.

General overview of the geology and mineral resources of Wakulla County based on existing literature and well data on file at the Florida Geological Survey.

Frissel, MJ, and JA van Veen (eds.), 1981. Simulation of Nitrogen Behavior of Soil-Plant Systems: papers of a workshop, models for the behavior of nitrogen in soil and uptake by plants, comparison between different approaches. Center for Agricultural Publishing and Documentation. Wageningen, Netherlands.

Compilation of papers from a conference.

Groffman PM, Jaworski, NA, 1990. Watershed Nitrogen Management "Upper Potomac River Basin Case Study." Report to USEPA. Narragansett, RI.

The authors developed a watershed nitrogen mass balance. Results indicated that the highest nitrogen export occurred during high surface water flows. Monthly loading can vary by a factor of three, therefore annual mass balances are required to evaluate best management practices (BMP). BMP's must maintain their integrity throughout the year since most nitrogen transport is during high-flow periods. Current BMP's do not strongly affect nitrogen storage mechanisms within agricultural fields.

Groffman, Peter M. Ecology of Nitrification and Denitrification in Soil Evaluated at Scales Relevant to Atmospheric Chemistry. In Rogers, JE and WB Whitman (eds.) Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides, and Halomethanes. American Society for Microbiology. Washington DC.

Organismal scale processes are well understood – translation of knowledge to useful information applicable at a larger scale is difficult. Plant community patterns are often strongly related to nitrogen availability, and different plant communities should thus exhibit distinct patterns of nitrification. At the landscape scale, soil type and plant community type are useful conceptual regulators of denitrification. Soil texture and drainage are strong controllers of oxygen availability and indirectly regulate nitrate supply through their influence on nitrification. Similarly, plant community type affects nitrate supply by controlling the nitrification rate. Both soil type and plant community type have strong effects on the decomposition of plant material and thus influence carbon supply to denitrifiers. Since plant community composition differences are the integrative product of the same ecological factors that influence microbial trace gas fluxes (water and nutrient availability), remote sensing of plant variables should be useful for large-scale soil-atmosphere gas exchange studies.

Groffman, Peter M. and James M Tiedje, 1989. Denitrification in north temperate forest soils: relationships between denitrification and environmental factors at the landscape scale. Soil Biology and Biochemistry Vol. 21 No. 5, pp. 621-626.

Relationships between annual denitrification nitrogen loss and soil physical and biological factors were investigated in nine north temperate forest soils of different texture and drainage classes. Soil texture was analyzed numerically by using percentage of sand as a variable. Soil wetness was quantified by a continuous drainage index function. Found that most of the variability (86%) in annual nitrogen loss to denitrification was explained with a multiple regression model using soil texture (%sand) and soil drainage index as predictor variables. Analysis at the landscape scale was necessary to derive this relationship. Denitrification enzyme activity (DEA) and DEA-to-biomass C ratio (Strongest predictor observed) accounted for up to 96% of the variation in annual denitrification nitrogen loss (ratio was higher in poorly-drained soils than well-drained soils. Percentage sand and soil wetness could also account for a large proportion of the annual variation in denitrification rates among sites ($r^2 = 0.86$).

Groffman, Peter M. and James M Tiedje, 1989. Denitrification in north temperate forest soils: relationships between denitrification and environmental factors at the landscape scale. Soil Biology and Biochemistry Vol. 21 No. 5, pp. 621-626.

A report on different aspects of the above mentioned study. Found that denitrification activity was highest in the spring and fall, and lowest in the summer – in Michigan. Over 80% of the annual nitrogen loss to denitrification occurred during brief (3-6 week) periods of high activity in the spring and fall. Rates of denitrification during these periods exceeded 0.5 kg N/ha/day in some soils. Estimates of annual nitrogen loss to denitrification ranged from <1 kg N/ha/yr. in a well-drained sand soil to over 40 kg N/ha/yr. in a poorly drained clay loam soil. Lack of available nitrate was the primary factor limiting denitrification in summer, but available carbon was probably occasionally limiting, especially in the well-drained soils.

Heede, B.H. 1985. Interactions Between Streamside Vegetation and Stream Dynamics. Symposium paper presented at Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Tucson, Arizona. pp. 54-57.

Interrelationships between vegetation and hydrologic processes in riparian ecosystems must be considered by managers before they attempt to alter these natural systems. A 5-year experiment demonstrated that logs that fall across the channel from streamside forests dissipate flow energy, maintain channel stability, decrease bedload movement, and increase water quality.

Hendry, MJ, RW Gillham, and JA Cherry, 1983. An integrated approach to hydrogeologic investigations – a case history. Journal of Hydrology, vol. 63: 211-232.

Study addressing the occurrence and migration of nitrate in a thin (3-7 m) phreatic sand aquifer beneath agricultural land. High nitrate concentrations were found at shallow depths, while concentrations at greater depths were close to zero. Through the application of physical hydrogeologic methods of investigation, geochemical studies, environmental isotope studies, numerical simulations of groundwater flow and solute transport, it was shown that denitrification was the principal cause of the nitrate distribution. The study showed that denitrification can be an important process in groundwaters at a regional scale.

Hirsch, R.M., W.M. Alley, and W.G. Wilber. 1988. Concepts for a National Water-Quality Assessment Program. U.S. Geological Survey Circular 1021.

Outlines three major goals of the National Water-Quality Assessment Program. Goals include providing a nationally consistent description of current water-quality conditions for a large part of the Nation's water resources, define long-term trends (or lack of trends) in water quality, and identify, describe, and explain, as possible, the major factors that affect observed water-quality conditions and trends.

Hubbard, R.K. 1990. Dissolved and Suspended Solids Transport from Coastal Plain Watersheds. Journal of Environmental Quality. 19:413-420.

Excessive amounts of dissolved or suspended solids in surface runoff or base flow may degrade the quality of streams, lakes, or other water bodies. Loads of dissolved and suspended solids in streamflow reflect the quality of water entering via surface runoff or base flow. This study was conducted to determine the concentrations and loads of dissolved and suspended solids in Coastal Plain streamflow; to examine relationships between concentrations, loads, and flow rate; and to determine overall streamflow water quality for these parameters. Dissolved solids and suspended sediment concentrations were determined on a weekly or high-flow storm event streamflow samples collected at gaging stations on three subwatersheds of the Little River Watersheds located near Tifton, GA. Dissolved solids concentrations ranged from 19 to 159 mg/L, and generally decreased as per unit area instantaneous discharge rate increased. Suspended sediment concentrations ranged from 1 to 137 mg/L, and generally increased as per unit area instantaneous discharge rate increased. Regression analyses showed good relations between log transforms of both dissolved solids load and suspended sediment load, vs. total monthly runoff. Mean suspended sediment concentrations during high-flow events were greater than means from the overall data set, while mean concentrations of dissolved solids from these events were reduced relative to the overall data set. The study showed that dissolved solids are the major component of total solids in coastal Plain streamflow. The mean dissolved and suspended sediment concentrations during the study were 67, 60, and 51 mg/L and 14, 17, and 14 mg/L for Watersheds B, F, and K, respectively. Overall, the study showed that, as measured on these watersheds, Coastal Plain streamflow is of good quality in terms of both dissolved and suspended solids. This good quality may reflect land-use practices designed to prevent soil erosion, but primarily reflects the Coastal Plain landform shape, which causes sediments eroded from the uplands to be deposited in the riparian zone before they can enter streamflow.

Izuno, F.T., et al., 1991. Phosphorus concentrations in drainage water in the Everglades Agricultural Area. Journal of Environmental Quality. 20:608-619.

Phosphorus in drainage water leaving the Everglades Agricultural Area (EAA) in southern Florida is alleged to be contributing to the accelerated eutrophication of Lake Okeechobee and the degradation of the Water Conservation Areas and the Everglades National Park ecosystems. Agricultural "best management practices" (BMPs) offer a means for achieving reductions in P in drainage water. Prior to developing and implementing BMPs, it is necessary to establish baseline EAA P concentrations. Baseline total P (TP) and total dissolved P (TDP) concentrations for various crop and field conditions in the EAA were determined. Thirty-six 0.7-ha plots were installed at four locations. Average TP and TDP concentrations were derived

from 6 to 30 drainage events for each of five conditions between November 1988 and December 1989: sugurcane, radish, and cabbage production fields, flooded fallow fields, and drained fallow fields. Baseline TP and TDP concentrations for main farm canals and rainfall were also determined. Average TP concentrations ranged rom 0.25 mg/L for radishes to 1.03 mg/L during the drain-down of flooded fallow plots. Total dissolved P concentrations ranged from 48 to 80% of TP. Main farm canal TP concentrations averaged 0.16 mg/L. Total P concentrations in rainfall averaged 0.07 mg/L. Total P in drainage water during 1989 for sugurcane, cabbage, and drained fallow fields were 0.72, 1.38, and 0.59 kg/ha, respectively. During the radish season, drainage water TP loading was 0.8 kg/ha. Flooded fallow fields after radishes yielded a TP loading rate of 3.82 kg/ha. Total P loading to the fields from rainfall averaged 0.7 kg/ha. Total dissolved P loading rates ranged from 25 to 60% of TP. Potential areas for BMP development and implementation for P concentration and loading reduction in the EAA include drainage rate, volume, and timing management, fertilizer use reduction, and enhanced crop rotation strategies.

Jaworski, N.A., et al. 1992. A watershed nitrogen and phosphorus balance: The upper Potomac River Basin. Estuaries. 15(1):83-95.

Nitrogen and phosphorus mass balances were estimated for the portion of the Potomac River basin watershed located above Washington, D.C. The total nitrogen (N) balance included seven input source terms, six sinks, and one "change-in-storage" term, but was simplified to five input terms and three output terms. The phosphorus (P) balance had four input and three output terms. The estimated balances are based on watershed data from seven information sources. Major sources of nitrogen are animal waste and atmospheric deposition. The major sources of phosphorus are are animal waste fertilizer. The major sink for nitrogen is combined denitrification, volatilization, and change-in-storage. The major sink for phosphorus is change-in-storage. River exports of N and P were 17% and 8%, respectively, of the total N and P inputs. Over 60% of the N and P were volatilized or stored. The major input and output terms on the budget are estimated from direct measurements, but the change-in-storage term is calculated by difference. The factors regulating retention and storage processes are discussed and research needs are identified.

Johengen, T.H., A.M. Beeton, and D.W. Rice. 1989. Evaluating the effectiveness of best management practices to reduce agricultural nonpoint source pollution. Lake and Reservoir Management. 5:63-70.

The Saline Valley project is one of 20 national projects sponsored by the U.S. Department of Agriculture (USDA) under the Rural Clean Water Program (RCWP) to evaluate methods of controlling agricultural nonpoint source pollution. The goals of this project were (1) to evaluate whether a voluntary approach using cost-share incentives would produce adequate participation by local farmers and (2) to reduce phosphorus loads from the area by 40 percent. Water quality has been monitored since 1981 using weekly grab samples and flow measurements. Trends in empirical relationships between concentration and discharge at three sampling stations were used to examine the effectiveness of best management practices (BMP). These relationships were highly variable among the sub-basins and years, and did not appear to correlate with areal estimates of BMP implementation. Overall, low participation within the project area hindered the ability to quantify changes in water quality resulting from BMP implementation and prevented the project from meeting its phoshorus reduction goals.

Jones, GW, Dr. SB Upchurch, KM Champion, 1996. Origin of nitrate in Ground Water Discharging from Rainbow Springs, Marion County, Florida. Southwest Florida Water Management District. Brooksville, Florida.

Report from an investigation to determine the sources of increasing nitrate levels in Rainbow Springs. Historic data indicated a possible 20 fold increase in nitrate concentrations during the past 40 years. The investigation involved delineating the areas where nitrate is entering the aquifer system, identifying the land uses that are contributing the nitrate, and determining what can be done to slow or reverse the nitrification of ground water in the area. The three greatest sources of nitrate were determined to be pasture fertilization, horses, and cattle. Development-related nitrogen sources such as septic tanks, sewage, and

residential turf and golf courses were determined to be minor contributers of nitrogen to the study area. It was concluded that nothing could be done to reduce current nitrate loading from the springs to the river due to the lag-time between nitrogen application and flow from the springs. The effects of reduced nitrate loading to the land may not be evident in water quality for a decade or more. Development of best management practices was recommended.

Jones, R.C. and B.H. Holmes. 1985. Effects of Land Use Practices on Water Resources in Virginia. Bulletin 144, Virginia Water Resources Reserach Center, Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

This study reviews the relationship between land use and water resources in Virginia. It examines three major land uses in the state--agriculture, urban, and forestry activities. For each land use, the relevant literature and state management programs are reviewed. In addition, the report outlines research needs in each area.

Kirkner and Associates, Inc, 1987. Risk Assessment of On-Site Sewage Disposal Systems for Selected Florida Hydrologic Regions. Report to the Florida Department of Health and Rehabilitative Services. Lake Wales, Florida.

Preliminary report complimenting the work of Ayres and Associates, 1987. Describes techniques used to select and monitor high-density subdivisions utilizing on-site disposal systems (OSDS). Uncertainty and sensitivity analysis techniques were applied to eight hydrologic regions distributed throughout Florida.

Korom, SF, 1992. Natural denitrification in the saturated zone: a review. Water Resources Research, vol. 28, no. 6: 1657-1668.

Review synthesizing published literature on natural aquifer denitrification. Discusses microbial processes and environmental requirements. Suggests guidelines for future research.

Lawrence, SJ, 1996. Nitrate and Ammonia in Shallow Ground Water, Carson City Urban Area, Nevada, 1989. US Geological Survey Water Resources Investigations Report 96-4224. Carson City, Nevada.

A network of 26 wells at 20 sites was established to investigate groundwater quality beneath the oldest and most developed part of the Carson City urban area. Nitrate and ammonia concentrations were positively correlated with several other solute concentrations. Contamination sources might be nitrogen-based fertilizers, septic systems, or leaky municipal sewer lines. Nitrification and denitrification control nitrate and ammonia concentrations beneath the study area.

Lowrance, R., et al., 1984. Riparian forests as nutrient filters in agricultural watersheds. BioScience. 34:374-377.

Riparian (streamside) vegetation may help control transport of sediments and chemicals to stream channels. Studies of a coastal plain agricultural watershed showed that riparian forest ecosystems are excellent nutrient sinks and buffer the nutrient discharge from surrounding agroecosystems. Nutrient uptake and removal by soil and vegetation in the riparian forest ecosystem prevented outputs from agricultural uplands from reaching the stream channel. The riparian ecosystem can apparently serve as both a short- and long-term nutrient filter and sink if trees are harvested periodically to ensure a net uptake of nutrients.

Lowrance, RR and HB Pionke, 1989. Transformation and movement of nitrate in aquifer systems. In Follett RF (ed.). Nitrogen Management and Ground Water Protection. Elsevier, New York.

Reviews processes leading to changes in NO3 concentration in groundwater and discusses case studies.In general, NO3 reaching shallow aquifer systems, especially those with rapidly fluctuating water tables, has a good chance for removal by denitrification or uptake by deeply rooted vegetation. Drawdown of a deeper unconfined aquifer can cause inflow of water with higher NO3 concentrations from a shallower aquifer.

Magette, W.L. et al. 1983. Wastewater Treatment in Soil as a Function of Residence Time in the Root Zone. Bulletin 137, Virginia Water Resources Research Center, Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

A laboratory study was conducted to determine nitjrogen removal rates from a land-applied wastewater as a function of the length of time the wastewater remained in the root zone. A digital simulation model was used as an aid in describing soil-water (and wastewater) movement through the root zone under wet conditions (i.e., root zone 50-75 percent saturated). A procedure was developed to predict the rate and volume of drainage as a function of initial soil moisture content, amount of liquid applied, and time after liquid application. An exact relationship between nitrogen removals and wastewater residence time in the root zone could not be developed. However, removals of up to 95 percent of applied NH4+-N were observed in an 18-cm-deep root zone when residence times were as short as 2 hr.

Meals, D.W. 1993. Assessing nonpoint phosphorus control in the LaPlatte River Watershed. Lakes and Reservoir Management. 7:197-207.

Phosphorus loading from agricultural activities such as manure and fertilizer applications often contributes to eutrophication of surface waters. The primary goal of the LaPlatte River Watershed Project in northwestern Vermont was to reduce phosphorus loading from farmland through implementation of best management practices (BMPs). Eleven years of monitoring did not show a dramatic decrease in phosphorus concentration or load from the watershed. However, analysis controlling for hydrologic variability suggested significant decreases in phosphorus load from some subwatersheds following completion of the land treatment program. Post-BMP phosphorus load reductions of 26-44% (0.01-0.14 kg/ha/yr) were estimated using a paired regression technique that accounts for discharge differences between years. Phosphorus export was reduced under most circumstances, except under the highest runoff conditions, suggesting that the capacity of the land treatment system to control phosphorus may have been exceeded occasionally. Observed phosphorus reductions in treated watersheds appeared to be related to the degree of treatment afte a minimum threshold level of land treatment had been achieved.

Miller, WL, 1992. Hydrogeology and Migration of Septic Tank Effluent in the Surficial Aquifer System in the Northern Midlands Area, Palm Beach County, Florida. US Geological Survey Water Resources Investigations Report 91-4175. Tallahassee, Florida.

The northern Midlands area in Palm Beach County is an area of expected residential growth, but its flat topography, poor drainage, and near-surface marl layers retard rainfall infiltration and cause frequent flooding. Tests at three septic tank sites showed traces of effluent in groundwater (38-92 feet from the septic tank outlets) and that near-surface marl layers greatly impede the downward migration of the effluent in the surficial aquifer system throughout the northern midlands.

Northwest Florida Water Management District, February, 1994. Non-point Source Assessment: Deer Point Lake Watershed. Water Resources Special Report 93-6. Havana, Florida.

Evaluation of existing and potential pollution contributions to Deer Point Lake from various non-point sources. Estimated pollutant-loading rates from non-point sources. Ranked water quality parameter loading and calculated cumulative rank indices to provide an overall indication of potential water quality impacts. Methodology integrated GIS, satellite imagery, and land use/cover maps to model existing

development patterns. The project methodology was intended to provide a framework for the development and implementation of pollution load reduction goals, total maximum daily loads, best management practices, land development regulations, land preservation and acquisition water quality protection, and watershed management goals.

Pederson, JK, PL Bjerg, and TH Christensen, 1991. Correlation of nitrate profiles with groundwater and sediment characteristics in a shallow sandy aquifer. Journal of hydrology, vol. 124: 263-277.

Develops a characterization of sediment profiles according to total reduction capacity (TRC), that quantifies the total amount of reduced compounds in aquifer material. A distinct increase of TRC was observed at and below the oxidation-reduction front. The TRC profiles showed that some reduction capacity is still present above the oxidation-reduction front and is apparently able to support denitrification.

Peterson, GH and AD McLaren (eds.), 1967. Soil Biochemistry. Marcel Dekker Inc. New York.

Text covering soil chemical dynamics.

Pitt, R. and M. Bozeman. 1982. Sources of Urban Runoff Pollution and Its Effects on an Urban Creek. United States Environmental Protection Agency. EPA-600/S2-82-090.

Sources and impacts of urban runoff were studied for the Coyote Creek near San Jose, California. The 3-year monitoring study included three tasks: (1) identifying and describing important sources of urban runoff pollutants; (2) describing the effects of those pollutants on water and sediment quality, aquatic organisms, and associated beneficial uses of the creek; and (3) assessing potential measures for controlling the problem pollutants in urban runoff. Results indicated that various urban runoff constituents (especially organics and heavy metals) may be responsible for many of the adverse biological conditions observed in Coyote Creek. But adequate control of pollutants would require extremely high removals that would be difficult as well as costly to achieve.

Postma, FB, AJ Gold, and GW Loomis, 1992. Nutrient and microbial movement from seasonally-used septic systems. Journal of Environmental Health, vol. 55, no. 2.

Seasonal occupancy may promote the transmission of contaminants to groundwater due to incomplete formation of a biological clogging mat in soil adsorbtion systems. Groundwater surrounding three seasonally-used septic systems was monitored to determine the movement and attenuation of nitrogen, phosphorous, fecal coliforms, and *Clostridium perfringens*. The septic systems showed inadequate attenuation of these parameters. Biological clogging mats were not found when the systems were evaluated at the end of summer occupancy. Siting seasonally-used shoreline septic systems may require improved effluent distribution to achieve wastewater renovation.

Puckett, L.J. 1995. Identifying the major sources of nutrient water pollution. Environmental Science and Technology. 29:408A-14A.

Atmospheric nitrogen may be a very important source of nutrient contamination. It is generally not enough to know the land use alone, the type and intensity of that use are equally important.

Reddy, Konda Rameshwer, 1976. Nitrification denitrification reactions in flooded soil. Ph.D. dissertation, Louisiana State University and Agricultural and Mechanical College.

Research results and review of nitrogen dynamics in soil and groundwater under fluctuating hydrologic regimes.

Reneau et al, 1989. Fate and Transport of Biological and Inorganic Contaminants. Journal of Environmental Quality, vol. 18, Apr-Jun 1989. Review of contaminant dynamics in the subsurface environment – especially as related to on-site disposal systems.

Denitrification may be significant in soils with restricted drainage if conditions are also adequate for nitrification of NH4+ before it reaches the anaerobic zone and if an adequate C source is available. Residual soil organic matter is probably not a satisfactory long- term energy source for denitrification. Data indicates that in well-drained soils where nitrification occurs immediately below the OSDS, denitrification does not adequately remove NO3- and the most probable mechanism for reducing N concentrations is dilution.

Reynoldson, T.B. Jr., and H.R. Hamilton. 1982. Spatial heterogeneity in whole lake sediments - towards a loading estimate. Hydrobiologia 91:235-240.

Studies of nutrient loadings, to shallow culturally eutrophied Alberta lakes, suggest internal inputs are significant. In this regard, estimation of bottom sediment P loads to Lake Wabamun were examined.

Riekerk, H and Korhnak, LV, 1992. Rainfall and Runoff Chemistry of Florida PineFlatwoods. Water, Air, and Soil Pollution 65: 59-68.

Rainfall chemistry was monitored for ten years. and correlated with runoff chemistry of an undisturbed 140 ha pine-cypress flatwoods watershed in Florida. The seasonal variation in pH showed a minimum during the summer months. Levels of nitrate-N and phosphate-P showed a significant decrease while Ca showed a significant increase in runoff over time. The pine-cypress flatwoods ecosystem appeared to absorb most of the acidity N, and P while losing only a little of the bases. Florida's sandhill lakes are poorly buffered and sensitive to acidification by rainwater of pH 4.7 or less, but forest lands are less vulnerable as the tree canopy and soil offers better buffering capacity, and the low productivity becomes improved by atmospheric nutrient inputs.

Robertson, WD, JA Cherry, and EA Sudicky, 1991. Ground-water contamination from two small septic systems on sand aquifers. Ground Water, vol. 29, no. 1.

Study conducted at two single-family homes located on shallow unconfined sand aquifers in Ontario. As a result of low transverse dispersion in the aquifer, mobile plume solutes such as nitrate and sodium occurred at more than 50 percent of the source concentrations 130 m downgradient from the septic system. Almost complete nitrate attenuation was observed within the last 2 m of the plume flowpath before discharge to an adjacent river. This was attributed to denitrification occurring within organic matter-enriched riverbed sediments. Concluded that, for many unconfined sand aquifers, the minimum distance-to-well reegulations for permitting septic systems in most parts of North America should not be expected to be adequately protective of well-water quality in situations where contaminants are not attenuated by chemical or microbiological processes.

Rosswall, T, 1982. Microbiological regulation of the biogeochemical nitrogen cycle. Plant and Soil, vol. 67: 15-34.

Discusses some aspects of the individual microbiological processes in the nitrogen cycle and their importance for an efficient management of agroecosystems. The influence of abiotic factors such as

oxygen concentration, inorganic nitrogen concentration, and pH is discussed in relation to the different processes.

Sawhney, B.L. 1977. Predicting phosphate movement through soil columns. Journal of Environmental Quality. 6:86-89.

To assess the potential pollution of ground water with P from septic tank drainfields, sorption capacities of various soils were determined over an extended period of time and related to P movement through soil columns using solutions having P concentrations similar to waste waters. The amounts of P sorbed by fine sandy loam (fsl) and silt loam (sil) soil columns before breakthrough occurred were approximately equal to the sorption capacities determined from isotherms obtained over a sufficiently long reaction time of about 200 hours. In Merrimac fsl, breakthrough occurred after about 50 pore volumes of waste water had passed through the column while about 100 pore volumes passed through Buxton sil before the breakthrough occurred. Following breakthrough, concentration of P in the effluent continued to increase and approached the influent concentration after several hundred pore volumes of effluent had passed through the columns. The results, thus, suggest that while most deep soils should effectively remove P from waste water, ground water under drainfields installed in soils of low P sorption capacity after prolonged use may contain undesirably large concentrations of P.

Starr, JL and BL Sawhney, 1980. Movement of nitrogen and carbon from a septic system drainfield. Water, Air, and Soil Pollution, vol. 13: 113-123.

A two year study of vertical and horizontal movement of nitrogen and carbon from a septic system drainfield in Connecticut. Concluded that 20 to 25% of total effluent nitrogen will be mineralized, even in unusually wet years. The nitrogen will be nitrified and will move with infiltrating water to the groundwater. Calculated that under continuous use and with 5-households / ha, the nitrate transported to groundwater aquifers below the sandy, well drained septic systems is about 35 kg / ha.

Stephens, D.W., B. Waddell, and J.B. Miller. 1988. Reconnaissance Investigation of Water Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage in the Middle Green River Basin, Utah, 1986-1987.

U.S. Geological Survey Water-Resource Investigations Report 88-4011. Salt Lake City, Utah. Reconnaissance of wildlife areas in the middle Green River basin of Utah was conducted during 1986 and 1987 to determine whether irrigation drainage has caused, or has the potential to cause significant harmful effects on human health, fish, and wildlife, or may adversely affect the suitability of water for beneficial uses. Studies at Stewart Lake Waterfowl Management Area and Ouray National Wildlife Refuge indicated that concentrations of boron, selenium, and zinc in water, bottom sediment, and biological tissue were sufficiently large to be harmful to fish and wildlife, and to adversely affect pesticides in surface water generally were less than established standards with the exception of gross alpha radiation, which exceeded by factors of three to five times the standard of 15 picocuries per liter in water from two of the drains discharging into Stewart Lake.

Sweets, P Roger, 1992. Diatom Paleolimnological Evidence for Lake Acidification in the Trail Ridge Region of Florida. Water, Air, and Soil Pollution 65: 43-57.

Paper reviews the basic data on north Florida lakes from PIRLA I (Paleoecological Investigation of Recent Lake Acidification) and adds data inferred from Florida Panhandle lakes analyzed during PIRLA II. Of the lakes determined to have acidified, there is evidence for the existence of natural acidification processes. Hydrological processes may play a large role in determining which lakes and areas are susceptible to anthropogenic acidification. Climatic differences are a likely exacerbating factor. Individual lakes are being affected according to amounts of acidic deposition, the number of years deposition has been received, soil types, bedrock type, flushing rates, and assorted limnological variables such as productivity, DOC, and competing anthropogenic influences. Past lakewater chemistries were inferred from dated cores. Such

studies have provided strong evidence for acidification resulting from acidic deposition in a small subset of lakes from various regions of the US. This subset is defined geographically by different acidic deposition rates and bedrock types, and by the many limnological factors that contribute to lake pH and buffering capacity. Lakes in the Panhandle did not show evidence of recent anthropogenic acidification – though only a few lakes were studied.

Tippett, John. 1993. Linking land use to water quality. Water Environment and Technology. 5:17+.

A GIS was used to determine land use patterns of five riparian buffer zones at increasing distances from a stream. Multiple linear regression was used to relate soluble reactive phosphorus and nitrate-nitrogen at each sampling station to watershed area and land use patterns in each buffer zone. Four seasonal equations were developed for each of the five buffer zones.

Trudell, MR, RW Gillham, and JA Cherry, 1986. An in-situ study of the occurrence and rate of denitrification in a shallow unconfined sand aquifer. Journal of Hydrology, vol. 83:251-268.

An in-situ injection experiment was conducted using a specially designed injection-withdrawal-sampling drive point. Nitrate and a conservative tracer (bromide) were added to natural groundwater and injected at 3m depth into a shallow, unconfined aquifer. The relative changes in concentration over time were observed. The organic carbon source required for denitrification is either dissolved organic carbon or soil organic carbon. Soil organic carbon, at 0.08-0.16% by weight, is adequate to denitrify large amounts of nitrate. The measured rate of denitrification ranged from 0.0078 to 0.13 g $NO_3^-N/m^3/hr$.

Tsai, Yuong-How, 1989. Factors Affecting Denitrification Kinetics in Selected Florida Soils. Master of Science Thesis. Gainesville, Florida.

Study of denitrification rates under anaerobic conditions for 24 soil horizons. The soil horizons differed widely in pH and organic carbon content. Soil horizons were selected from six soil profiles representing several poorly-drained soil series in Florida. Denitrification rates were determined by measuring the rate of nitrate disappearance or the rate of nitrous oxide production. Linear correlation analysis showed that zero-order denitrification values were significantly related to total or soluble organic carbon content and with nitrate-nitrogen content. High denitrification rates were obtained in surface horizons. In subsoil horizons, denitrification rates were relatively low due to low organic matter content, low microbial population, and pH induced toxicities (Possibly high aluminum concentrations).

US Environmental Protection Agency, 1977. Alternatives for Small WastewaterTreatment Systems: On-Site Disposal / Septage Treatment and Disposal. EPA Technology Transfer Seminar Publication.

Identification of community needs and suitability for non-central wastewater facilities. Description of wastewater characteristics and soil treatment capabilities. Estimation of soil infiltrative and percolative capacities. Summary of on-site treatment and disposal system alternatives and alternative selection. Overview of septage generation, treatment, and disposal.

USGS, 1994. Nitrate in Ground Water and Spring Water Near Four Dairy Farms in North Florida, 1990-93. Water Resources Investigations Report 94-4162. Tallahassee, Florida.

Investigation of nitrate in groundwater. Analysis of groundwater parameters as indication of type and extent of microbiological activity in aquifer – particularly denitrification.

USGS, 1996. Hydrogeologic Investigation and Simulation of Ground-Water Flow in the Upper Floridan Aquifer of North-Central Florida and Southwestern Georgia and Delineation of Contributing Areas for Selected City of Tallahassee, Florida, Water Supply Wells. USGS Water-Resources Investigations Report 95-4296. Tallahassee, Florida.

Results of an investigation of the part of the Upper Floridan aquifer that underlies Tallahassee and Leon County, Florida, and the surrounding counties in North-Central Florida and southwestern Georgia. Previously collected hydrogeologic data was used in conjunction with a computer model to characterize ground-water flow in the study area. Computer simulation was performed using two USGS software packages: MODFLOW - a modular three-dimensional finite-difference ground-water flow model - and MODPATH - a particle-tracking program. Contributing areas for five City of Tallahassee water-supply wells were delineated. Computer simulation results were compared to analytical methods.

USGS, 1996. National Water Quality Assessment of the Georgia-Florida Coastal Plain Study Unit – Water Withdrawals and Treated Wastewater Discharges, 1990. Water-Resources Investigations Report 95-4084. Tallahassee, Florida.

Compilation of data – by county – on water use in the Georgia-Florida Coastal Plain Study Unit.

USGS. Relation of Nitrate Concentrations in Ground Water to Well Depth, Well Use, and Land Use in Franklin Township, Gloucester County, New Jersey, 1970-85. Water Resources Investigations Report 94-4174. USGS. West Trenton, New Jersey.

Research report on nitrate dynamics in an unconfined aquifer system in New Jersey. Primary sources of nitrogen were leachate from on-site disposal systems, runoff from animal feedlots, and leachate from nitrogen fertilizers. Nitrate concentration increased with the percentage of developed land in a well's buffer zone. Nitrate concentrations tend to decrease with well depth. Deep wells contain older water – nitrate concentrations are more likely to represent past rather than present use.

Vighi, M. 1991. Phosphorus loads from selected watersheds in the drainage area of the Northern Adriatic Sea. Journal of Environmental Quality. 20:439-44.

The Po Valley is one of the most productive agricultural areas in Europe and P losses from fertilizers are often accused of being among the main factors responsible for eutrophication of the Northern Adriatic Sea. to quantify nonpoint phosphorus loads in this area, 15 small watersheds were studied. Thirteen watersheds were in the intensive agricultural area near the coast and two watersheds were in the forested mountains. Land use in the watersheds was carefully examined and P loads from various sources were theoretically evaluated and experimentally measured. The results indicate fertilization does not increase the losses of P through leaching from the coastal soils, where the measured release were in the range 0.03 to 0.21 kg-P/ha/yr with a mean value of about 0.1 kg-P/ha/yr. There is, however, a greater loss of P through soil erosion from the mountain watersheds (0.6 kg/ha/yr). It can be concluded that the control of point sources

must take priority over nonpoint sources in efforts to reduce accelerated eutrophication of the Northern Adriatic Sea.

Walker, WG, J Bouma, DR Keeney, and PG Olcott, 1973. Nitrogen tansformations during subsurface disposal of septic tank effluent in sands: II Ground water quality. Journal of Environmental Quality, vol. 2, no.4: 521-525.

Groundwater samples were analyzed to establish patterns of nitrogen enrichment in the groundwater around seepage beds and to evaluate system performance in sands in terms of nitrogen removal. The data obtained suggested that, in sands, the only active mechanism of lowering the nitrate content is by dilution with uncontaminated groundwater – possibly requiring relatively large areas or low septic tank densities.

Waller, BG, B Howie, and CR Causaras, 1987. Effluent Migration from Septic Tank Systems in Two Different Lithologies, Broward County, Florida. US Geological Survey Water Resources Investigations report 87-4075. Tallahassee, Florida.

Two septic tank test sites, one in sand and one in limestone, were analyzed for effluent migration. Monitoring wells were sampled for a 16 month period. Results were graphically presented. Variances in hydrologic regimes posed difficulties to generalizations of results. Dilution appeared to be the main factor in reduction of nitrate concentrations.

Weier, KL and JW Gillham, 1986. Effect of acidity on denitrification and nitrous oxide evolution from Atlantic Coastal Plain soils. Soil Science Society of America Journal, vol. 50:1202-1205.

The effect of acidity on denitrification and nitrous oxide production in six soils from the Atlantic Coastal Plain was estimated using laboratory incubations of flooded soil for periods up to 21 days. Increased denitrification effects were associated with a decrease in cidity in all soils – most of the effect occurred above pH = 6.5. Nitrous oxide evolved increased with increasing acidity with a maximum at pH <= 5.8. Atlantic coastal Plain soils generally have a pH < 5.8.

Whigham, D.F. et. al. 1988. Impacts of Freshwater Wetlands on Water Quality: A Landscape Perspective. Environmental Management 12(5):663-671.

Suggest that a landscape approach might be useful in evaluating the effects of cumulative impacts on freshwater wetlands. The reason for using this approach is that most watersheds contain more than one wetland, and effects on water quality depend on the types of wetlands and their position in the landscape. Riparian areas that border uplands appear to be important sites for nitrogen processing and retention of large sediment particles. Fine particles associated with high concentrations of phosphorus are retained in downstream wetlands, where flow rates are slowed and where the surface water passes through plant litter. Riverine systems also may play an important role in processing nutrients, primarily during flooding events. Lacustrine wetlands appear to have the least impact on water quality, due to the small ratio of vegetated surface to open water. Examples are given of changes that occurred when the hydrology of a Maryland floodplain was altered.

Yates, MV, 1985. Septic Tank Density and Ground-Water Contamination. Ground Water, vol. 23, no. 5.

Reviews literature regarding septic tank density and waterborne disease in several states. Concludes that the single most important means of limiting groundwater contamination by septic tanks is to restrict the density of these systems in an area.

Models

Adamus, C.L. and M.J. Bergman. 1995. Estimating Nonpoint Source Pollution Loads with a GIS Screening Model. Water Resource Bulletin, AWWA 31(4):647-655.

The St. Johns River Water Management District (SJRWMD) is using a Geographic Information System (GIS) screening model to estimate annual nonpoint source pollution loads to surface waters and determine nonpoint source pollution problem areas within the SJRWMD. The model is a significant improvement over current practice because it is contained entirely within the district's GIS software, resulting in greater flexibility and efficiency, and useful visualization capabilities. Model inputs consist of five spatial data layers, runoff coefficients, mean runoff concentrations, and stormwater treatment efficiencies. The spatial data layers are: existing land use, future land use, soils, rainfall, and hydrologic boundaries. These data layers are processed using the analytical capabilities of a cell-based GIS. Model output consists of seven spatial data layers: runoff, total nitrogen, total phosphorous, suspended solids, biochemical oxygen demand, lead, and zinc. Model output can be examined visually or summarized numerically by drainage basin. Results are reported for only one of the SJRWMD's ten major drainage basins, the lower St. Johns River basin. The model was created to serve a major planning effort at the SJRWMD; results are being actively used to address nonpoint source pollution problems.

Bacon, PE, 1995. Nitrogen Fertilization in the Environment. Marcel-Dekker, Inc. New York.

Agriculturally-oriented reference text that reviews the interactions between nitrogen and the ecosystem and presents simulation models.

Caussade, B. and M. Pratt, 1990. Transport modelling in watersheds. Ecological Modelling, 52: 135-179.

Reviews modeling concepts and methodologies as related to watershed-scale modeling. Points out model requirements for various temporal and spatial scales. Includes a model review.

CDM, 199??. Master Stormwater Management Plan, City of Tallahassee, Florida. Camp, Dresser, and McKee, Tallahassee, FL.

Cushing, C.E. et. al. 1983. Relationships among chemical, physical, and biological indices along river continua based on multivariate analyses. Arch. Hydrobiol. 98(3):317-326.

A variety of multivariate analyses were applied to chemical, physical, and biological data from 16 stream sites to explore the usefulness of these factors in possible stream classification systems and to test hypotheses of the River Continuum Concept.

DeVantier, B.A., and A.D. Feldman. 1993. Review of GIS applications in hydrologic modeling. Journal of Water Resources Planning and Management. 119:246-261.

Geographic information systems (GIS) provide a digital representation of watershed characteristics used in hydrologic modeling. This paper summarizes past efforts and current trends in using digital terrain models and GIS to perform hydrologic analyses. There methods of geographic information storage are discussed: raster or grid, triangulated irregular network, and contour-based line networks. The computational, geographic, and hydrologic aspects of each data-storage method are analyzed. The us of remotely sensed

data in GIS and hydrologic modeling is reviewed. Lumped parameter, physic-based, and hybrid approaches to hydrologic modeling are discussed with respect to their geographic data inputs. Finally, several applications areas (e.g., floodplain hydrology, and erosion prediction) for GIS hydrology are described.

Domenico, PA and FW Schwartz, 1998. Physical and Chemical Hydrogeology. John Wiley and Sons, Inc. New York.

Hydrological reference text. Discussions of fundamentals of groundwater flow, contaminant transport, remediation, risk assessment, and modeling.

Donigian, A.S. and W.C. Huber. 1991. Modeling of Nonpoint Source Water Quality in Urban and Non-Urban Areas. EPA/600/3-91/039.

Nonpoint source assessment procedures and modeling techniques are reviewed and discussed for both urban and non-urban land areas. Detailed reviews of specific methodologies and models are presented, along with overview discussions focusing on urban methods and models, and on non-urban (primarily agricultural) methods and models. Simple procedures, such as constant concentration, regression, statistical, and loading function approaches are described, along with complex models such as SWMM, HSPF, STORM, CREAMS, SWRRB, and others. Brief case studies of ongoing and recently completed modeling efforts are described. Recommendations for nonpoint runoff quality modeling are presented to elucidate expected directions of future modeling effort.

Eckersten, H –Gardenas – Jansson, 1992. Modelling seasonal nitrogen, carbon, water and heat dynamics of the Solling spruce stand. Ecological Modelling 83: 119 -129.

The authors coupled two mechanistic models: SOIL and SOILN to generate a more comprehensive ecological model. The driving variables in the SOIL model were air temperature, precipitation, wind speed, vapor pressure, and global radiation – all of which were obtained from the nearest meteorological station. SOIL also considered soil water tension. The driving variables in SOILN were soil water flows/contents, soil temperature, and the ratio between actual and potential transpiration. Problems were encountered in model calibration.

ESRI, 1992. Cell-based modeling with GRID. Environmental Systems Research Institute, Inc., Redlands, California.

User manual for GRID appliction.

Follett, RF (ed.), 1989. Nitrogen Management and Ground Water Protection. Elsevier, New York.

Text that address nitrogen and groundwater concerns from an agricultural perspective. Contains chapters reviewing nitrogen transport mechanisms, groundwater concerns, fertilizer management, and modeling methodologies and concerns.

Frimpter, MH, JJ Donohue, IV, and MV Rapacz, 1990. A mass-balance model for predicting nitrate in ground water. New England Water Works Association, vol. 54, no. 4.

Development of a model to be used for prediction of nitrate concentrations in public-supply wells under steady-state conditions. Predicted concentrations are derived by calculating the concentration that results

from the total weight of nitrogen and total volume of water entering the zone of contribution to the well. Not a spatial model.

Grayson, R.B., I.D. Moore, and T.A. McMahon. 1992. Physically based hydrologic modeling 2. Is the concept realistic? Water Resources Research. 26:2659-2666.

Future directions for physically based, distributed-parameter models intended for use as hydrologic components of sediment and nutrient transport models are discussed. The attraction of these models is their potential to provide information about the flow characteristics at oints within catchments, but current representations in process-based models are often too crude to enable accurate, a priori application to predictive problems. The difficulties relate to both the perception of model capabilities and the fundamental assumptions and algorithms used in the models. In addition, the scale of measurement for many parameters is often not compatible with their use in hydrological models. The most appropriate uses of process-based, distributed-parameter models are to assist in the analysis of data, to test hypotheses in conjunction with field studies, to improve our understanding of processes and their interactions and to identify areas of poor understanding in our process descriptions. The misperception that model complexity is positively correlated with confidence in the results is exacerbated by the lack of full and frank discussion of a model's capability/limitations and reticence to publish poor results. This may ultimately diminish the opportunity to advance understanding of natural processes because the managers of research resources are given the impression that the answers are already known and are being provided by models. Model development is often not carried out in conjunction with field programs designed to test complex models, so the link with reality is lost.

Hantzsche, NN, and EJ Finnemore, 1992. Predicting Ground-Water Nitrate-Nitrogen Impacts. Ground Water vol. 30, no. 4.

Review of literature concerning the contribution and fate of nitrogen beneath septic tank disposal fields. Simplified methods are developed for estimating long-term groundwater nitrate increases on an area-wide basis. Predicted values are compared with actual monitoring data for three California communities to verify the reasonableness of the suggested methods. The simple model does not account for spatial variability.

Harper, Harvey H., 1992. Estimation of Stormwater Loading Rate Parameters for Central and South Florida. Environmental Research and Design, Inc., Orlando, Florida.

Provides loading and concentration data for various landuses in Florida.

Hart, R.L. ed. 1993. Management Guidelines and Goals for the Myakka River Basin. Florida Department of Environmental Regulation, Office of Coastal Management.

The objective of the Myakka River Basin Project was to provide a technical basis for management goals and recommendations that would protect the natural resources of the Myakka River and its estuary, Charlotte Harbor. The major task for the third year was to conduct a geographic information system (GIS) analysis and use the results to develop management goals and recommendations. The Myakka River drains a watershed of approximately 1,559 km^2. Much of the watershed consists of rural uses and publicly owned lands. Water quality of the river is generally good. However, population growth projections for the region and concern over the potential environmental impacts associated with growth require planning to protect river resources from future degradation. A GIS-based computer model was developed to illustrate how models can be used for projecting the runoff and chemical loadings to the Myakka River as a result of changes in land use. Three scenarios were modeled for a subbasin which is generally undeveloped at this time. The scenarios included urban development as projected by the County Planning Department with no preservation areas; urban development with preservation of wetland and hammock areas; and urban development with preservation of wetland, hammocks, and a 220-foot shoreline buffer.

He, C., J.F. Riggs, and Y. Kang. 1993. Integration of Geographic Information Systems and a Computer Model to Evaluate Impacts of Agricultural Runoff on Water Quality. Water Resources bulletin, AWRA. 29(6):891-900.

This study integrates an Agricultural Non-Point source Pollution Model (AGNPS), the Geographic Resource Analysis Support System (GRASS) and GRASS WATERWORKS (a hydrologic modeling tool box being developed at the Michigan State University Center for Remote Sensing) to evaluate the impact of agricultural runoff on water quality in the Cass River, a subwatershed of Saginaw Bay. AGNPS is used to estimate the amounts, origin, and distribution of sediment, nitrogen (N), and phosphorus (P) in the watershed. GRASS and GRASS WATERWORKS are used to generate parameters needed for AGNPS from digital maps, which include soil association, land use, watershed boundaries, water features, and digital elevation. Outputs of the model include spatially distributed estimates of volume and peak runoff, overland and channel erosion, sediment yields, and concentrations of nitrogen and phosphorus. Management scenarios are explored in the AGNPS model to minimize sedimentation and nutrient loading. Scenarios evaluated include variations in crop cover, tillage methods, and other agricultural management practices. In addition, areas vulnerable to erosion are identified for best management practices.

Heidtke, T.M. and M.T. Auer. 1993. Application of a GIS-based Nonpoint Source Nutrient Loading Model for Assessment of Land Development Scenarios and Water Quality in Owasco Lake, New York. Water Science and Technology. 28(3-5):595-604.

The magnitude and water quality implications of nonpoint source phosphorus loadings to Owasco Lake (New York) are evaluated through the application of a methodology which links geographic characteristics, long-term average runoff loads and a set of critical lakewide water quality response parameters. The approach utilizes the Universal Soil Loss Equation together with empirical loading functions to derive representative phosphorus export coefficients for the local drainage system. Cumulative loadings from individual subbasins within the watershed serve as input to a simple water quality model of Owasco Lake, showing the expected lake response in terms of average total phosphorus concentration, trophic state, water transparency, and minimum hypolimnetic dissolved oxygen concentration. The methodology facilitates easy andrapid assessment of general watershed management and development scenarios of interest. A unique aspect of the approach is its dependence upon descriptive data supplied by a Geographic Information System (GIS) to establish the coincidence of specific land use, soil texture and surface slope attributes within each of the hydrologic sub-basins comprising the overall watershed. The GIS-generated attribute matrices provide a much more accurate depiction of critical geographic characteristics known to impact nonpoint source runoff loadings, thereby improving the reliability of current and projected phosphorus loads to Owasco Lake.

James, C.R., et al. 1995. Vulnerability Zone Identification: A Watershed Management Tool for Protecting Reservoir Water Quality. Proc. 22 Annual Confr. Integr. Water Res. Plan 21 Century. ASCE. p.293-296.

As part of an extensive watershed management project, an approach was developed for integrating the potential impacts of watershed physical characteristics on raw water quality. This approach uses the capabilities of a geographic information system (GIS) database to produce water quality vulnerability zone maps which, in turn, can be used to develop watershed management strategies. Therefore, these vulnerability zone maps can be used by utilities to proactively manage the watersheds so that source protection becomes an effective first barrier to water quality degradation and to ultimately improve the water quality. An overview of the conceived approach used to produce the water quality vulnerability zone maps is documented here.

Jeton, A.E., and J.L. Smith. 1993. Development of Watershed Models for Two Sierra Nevada Basins Using a Geographic Information System. Water Resources Bulletin. 29(6):923-932.

Techniques were developed using vector and raster data in a geographic information system (GIS) to define the spatial variability of watershed characteristics in the north-central Sierra Nevada of California and Nevada and to assist in computing model input parameters. The U.S. Geological Survey's Precipitation-Runoff Modeling System, a physically based, distributed-parameter watershed model, simulates runoff for a basin by partitioning a watershed into areas that each have a homogeneous hydrologic response to precipitation or snowmelt. These land units, known as hydrologic-response units (HRU's), are characterized according to physical properties, such as altitude, slope, aspect, a land cover, soils, and geology, and climate patterns. Digital data were used to develop a GIS data base and HRU classification for the American River and Carson River basins. The following criteria are used in delineating HRU's: (1) Data layers are hydrologically significant and have a resolution appropriate to the watershed's natural spatial variability, (2) the technique for delineating HRU's accommodates different classification criteria and is reproducible, and (3) HRU's are not limited by hydrographic-subbasin boundaries. HRU's so defined are spatially noncontiguous. The result is an objective, efficient methodology for characterizing a watershed and for delineating HRU's. Also, digital data can be analyzed and transformed to assist in defining parameters and in calibrating the model.

Kalkhoff, S.J. 1993. Using a Geographic Information System to Determine the Relation Between stream Quality and Geology in the Roberts Creek Watershed, Clayton County, Iowa. Water Resources Bulletin. 29(6):989-996.

A geographic information system (GIS) was used to determine the relation between the stream-water quality and underlying geology in Roberts Creek watershed, Clayton County, Iowa, for base-flow conditions during the spring and summer of 1988-90. Geologic, stream, basin and subbasin boundaries, and water quality sampling-site coverages were created by digitizing available maps. A contour coverage was created from digital linegraph data. The areal extent of geologic units subcropping in each subbasin was quantified with GIS, and the results then were output and joined with the discharge and water-quality data for statistical analyses. Illustrations showing the geology of the study area and the results of the study were prepared using GIS. By using GIS and a statistical software package, a weak but statistically significant relation was found between the water temperature, pH, and nitrogen concentrations in Roberts Creek and the underlying geology during base-flow conditions.

Lahlou, M. et. al. May 1996. Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) Version 1.0 Users Manual. Office of Water (4305) EPA-823-R-96-001.

The EPA BASINS model is a watershed modeling package designed to integrate real data, and simulations with a spatial GIS framework. The model can be broken into two portions. The first is the Assessment/Planning Module which allows quick evaluation of selected areas, organize information and display results. Three scales of analysis can be performed including a Target or regional analysis, Assess at the watershed level, and Data Mining located at the station level. The second portion is the Modeling Module. This examines impacts of pollutant loadings from point and non-point sources. Three different models are contained within the Modeling Module. The NPSM (non-point source model) simulates nonpoint source runoff, pollution loadings and dissolved oxygen levels in runoff for selected runoff. QUAL2E is a 1-D steady-state water quality and eutrophication model that allows fate and transport modeling for point and nonpoint source loadings. TOXIROUTE is a screening level stream routing model that performs simple dilution/decay calculations under mean or low-flow conditions for a stream system within a given watershed.

Levine, D.A., and W.W. Jones. 1990. Modeling phosphorus loading to three Indiana reservoirs: a geographic information system approach. Lake and Reservoir Management. 6:81-91.

This paper describes a geographic information system (GIS) approach to modeling the effects of distance to water and slope angle on external phosphorus loading to lakes. A raster GIS database was created that included land use, topography, soils, and watershed boundaries for three Indiana reservoirs (Lakes Kickapoo, Lenape, and Shakamak). Three sets of phosphorus export coefficients were selected and assigned to each cell according to land use. Linear filters were designed and applied to the export coefficients such that areas nearest the water and with the steepest slopes would contribute the greatest amount of phosphorus relative to the intitial phosphorus export coefficient. These filtered coefficients were used to calculate aerial phosphorus loading and in-lake phosphorus concentrations. The predicted concentrations were within 5 ug/L of the observed phosphorus concentration in Lake Shakamak for three modeling scenarios, and within 22 ug/L of the observed concentration in Lake Lenape for three scenarios. Predictions of phosphorus concentrations in Lake Kickapoo were consistently low (35 percent to 95 percent). This may have resulted from the complex hydrology of Lake Kickapoo or the inability to accurately model the physical processes as intended. The GIS system was useful for modeling the effects of distance and slope on phosphorus loading and for providing data highlighting critical management areas.

McElroy, A.D. et. al., 1976. Loading Functions for Assessment of Water Pollution from Nonpoint Sources. EPA-600/2-76-151, U.S. Environmental Protection Agency, Washington, D.C., USA.

This is a user's handbook that provides two basic functions. First, it presents loading functions together with the methodologies for their use. Second, it presents some of the needed data, provides references to other sources of data, and suggests approaches for generation of data when available data ane inadequate. A corollary function consists of assessments of the adequacies of functions and their supporting inventories of data, and an assessment as well of the extent to which pollutants and nonpoint sources are adequately covered.

Mehran, M, J Noorishad, and KK Tanji, 1984. A numerical technique for simulation of the effect of soil nitrogen transport and transformations on groundwater contamination. Environmental Geology, vol. 5, no. 4, 213-218.

Development of a model intended as a tool for long-term prediction of the impact of agricultural activities on aquifer systems and evaluation of management alternatives. Nitrate pollution potential in groundwater is predicted using two numerical models. One model is for the vadose zone and includes transport by dispersion and convection of mobile species of nitrogen, ammonium ion exchange, first order nitrogen transformations, and plant uptake of nitrogen. The other model is for the aquifer system where transport of nitrate is assumed to be affected only by dispersion-convection phenomena. A simple hypothetical example problem is solved

Moosburner, GJ and EF Wood, 1980. Management model for controlling nitrate contamination in the New Jersey Pine Barrens aquifer. Water Resources Bulletin, vol. 16, no. 6: 971-978.

Application of a land use management model to Jackson Township of the New Jersey Pine Barrens. The model consisted of a simulation model for the transport of nitrates from septic tank systems through the aquifer and a multiobjective, goal programming optimization model to determine population density restrictions using planning population projections. Results showed that growth may have to be curtailed in areas of Jackson Township in order to maintain acceptable nitrate concentrations in groundwater.

Pastor, J and WM Post, 1986. Influence of climate, soil moisture, and succession on forest carbon and nitrogen cycles. Biogeochemistry, vol. 2: 3-27.

Report on a computer simulation developed to assemble a model ecosystem that links abiotic and biotic processes through equations that predict decomposition processes, actual evapotranspiration, soil water

balance, nutrient uptake, growth of trees, and light penetration through the canopy. The model can make accurate quantitative predictions of biomass accumulation, nitrogen availability, soil humus development and net primary production for forests in eastern North America.

Rifai, H.S. et al., 1993. Getting to the nonpoint source with GIS. Civil Engineering. 63(6):44-46.

As part of the Galveston Bay National Estuary Program in Texas, engineers have characterized the nonpoint pollution sources that are poisoning the bay. A geographic information system has helped them with extensive mapping-based calculations.

Smith, Richard A, Gregory E Schwarz, and Richard B Alexander. Regional interpretation of water-quality monitoring data. Water Resources Research, vol. 33, no. 12, pp. 2781-2798.

Paper describes a model SPARROW – Spatially referenced regressions on watershed attributes. The method was designed to overcome problems in data interpretation caused by factors that complicate regional water quality assessments: sparseness of sampling locations due to cost constraints, spatial biases in the sampling network, and drainage basin heterogeneity. The method is used to estimate the proportion of watershedss in the conterminous United States with outflow total phosphorous concentrations less than 0.1 mg/L and to classify cataloging units according to total nitrogen yield.

Soranno, P.A., S.L. Hubler, and R. Carpenter. 1996. Phosphorus Loads to Surface Waters: A Simple Model to Account for Spatial Pattern of Land Use. Ecological Applications 6(3):865-878.

This research models nonpoint-source phosphorus (P) loading from land to surface waters using a simple model that accounts for spatial pattern in topography and land use using geographic information system (GIS) databases. They estimated areas of the watershed that strongly contributed to P loading by approximating overland flow, and modeled annual P loading by fitting three parameters to data obtained by stream monitoring. The model was calibrated using P loading data from two years of contrasting annual precipitation for Lake Mendota, a Wisconsin eutrophic lake in a watershed dominated by agriculture and urban lands. Land-use scenarios were developed to estimate annual P loading from pre-settlement and future land uses. As much as half of the Lake Mendota watershed did not contribute significantly to annual P loading. The greatest contribution to loading came from a heterogeneous riparian corridor that varied in width from 0.1 km to ~6 km depending on topography and runoff conditions. They estimated that loading from pre-settlement land use was one-sixth of the loading from present land use. A future scenario, representing an 80% increase in existing urban land (from 9 to 16% of total watershed area, which would be reached in 30 yr with current landuse trends), showed only modest increases in annual P loading but possible significant effects on water quality. If the watershed were to become entirely urbanized, P loading to the lake would double and potential effects on water quality would be severe. Changes in P loading were strongest with conversion of undisturbed vegetated lands, especially riparian areas, to either urban or agricultural uses. Variability in total annual rainfall led to variability in the riparian area that affects P loading, with implications for policies intended to control nonpoint nutrient inputs.

Tim, U.S. and R. Jolly. 1994. Evaluating Agricultural Nonpoint-Source Pollution Using Integrated Geographic Information Systems and Hydrologic/Water Quality Model. Journal of Environmental Quality. 23(1):25-35.

Considerable progress has been made in developing physically based, distributed parameter, hydrologic/water quality (H/WQ) models for planning and control of nonpoint-source pollution. The widespread use of these models is often constrained by the excessive and time-consuming input data demands and the lack of computing efficiencies necessary for iterative simulation of alternative

management strategies. Recent developments in geographic information systems (GIS) provide techniques for handling large amounts of spatial data for modeling nonpoint-source pollution problems. Because a GIS can be used to combine information from several sources to form an array of model input data and to examine any combinations of spatial input/output data, it represents a highly effective tool for H/WQ modeling. This paper describes the integration of a distributed-parameter model (AGNPS) with a GIS (ARC/INFO) to examine nonpoint sources of pollution in an agricultural watershed. The ARC/INFO GIS provided the tools to generate and spatially organize the disparate data to support modeling, while the AGNPS model was used to predict several water quality variables including soil erosion and sedimentation within a watershed. The integrated system was used to evaluate the effectiveness of several alternative management strategies in reducing sediment pollution in a 417-ha watershed located in southern Iowa. The implementation of vegetative filter strips and contour buffer (grass) strips resulted in a 41 and 47% reduction in sediment yield at the watershed outlet, respectively. In addition, when the integrated system was used, the combination of the above management strategies resulted in a 71% reduction in sediment yield. In general, the study demonstrated the utility of integrating a simulation model with GIS for nonpoint-source pollution control and planning. Such techniques can help characterize the diffuse sources of pollution at the landscape level.

Vieux, B.E. and Scott Needham. 1993. Nonpoint-Pollution Model Sensitivity to Grid-Cell Size. Journal of Water Resources Planning and Management. 119(2).

Nonpoint-pollution models estimate loadings of chemicals, sediment, and nutrients that degrade water quality. Before controls can be implemented, location and severity of pollution must be identified in the watershed basin. Geographic information systems (GISs) are computer-automated, data management systems simplifying the input, organization, analysis, and mapping of spatial information. Because nonpoint-pollution models simulate distributed watershed basin processes, a heterogeneous and complex land surface must be divided into computational elements such as grid cells. Model parameters can be derived from each grid cell directly from maps using GIS. Cell size selection, if arbitrarily determined though, yields ambiguous if not erroneous results. This paper investigates the effects of cell size selection through a sensitivity analysis of input parameters for the nonpoint-pollution model, Agricultural Nonpoint source Pollution Model (AGNPS), using a GIS for a small research watershed. Model grid-cell sizes were found to be the most important factor affecting sediment yield. As the grid-cell sizes increase, stream meanders are short-circuited. The shortened stream lengths cause sediment yield to increase by as much as 32%.

Winchester, John W, and Ji-Meng Fu, 1992. Atmospheric Deposition of Nitrate and Its Transport to the Apalachicola Bay Estuary in Florida. Water, Air, and Soil Pollution 65: 23-42.

Estimation of Nitrate deposition based on statistical analysis. Used weekly data from five National Acid Deposition Program sites within the watershed and river water chemical data from the USGS. Other surface sources of nitrate and chemical transformations were not fully quantified. Atmospheric deposition appeared to be sufficient to account for essentially all the dissolved nitrate and ammonium and total organic nitrogen flow in the river.

Planning/Policy

Odum, H.T., C. Diamond, M.T. Brown. 1987c. Energy Analysis Overview of the Mississippi River Basin. Report to The Cousteau Society. Center for Wetlands (Publication 87-1), Univ. of Florida, Gainesville. 107pp.

Energy flow, quality, and embodied energy enables one to quantify and compare resource uses and determining the development strategies that maximize energetics of both human and natural systems in an increasing low energy world. Water cycles of the Earth are so important in the organization of the landscape, river basins form a natural unit for understanding, predicting, and planning for the future. Soil reserves within the basin were considered the most valuable long-range resource. Diking and channeling caused much of these reserves to be lost to the sea.

Romitelli, S. 1997. Energy Analysis of Watersheds. PhD Dissertation, Environmental Engineering Sciences. Univ. of Florida, Gainesville. 292 pp.

This research uses a new approach to study the organization of watersheds and to provide insight for their management. It evaluates work done by water energies on the landscape and explores an hypothesis that "self-organizing watersheds couple the geopotential and chemical potential energy use to maximize biological and geological production". Work of the mountains was measured by the geopotential energy use and related to work on terrestrial productivity of valleys measured by the chemical potential energy evapotranspired. Using data on rainfall and river flow data and topographic geographic information, spatial and temporal energy analysis and EMERGY evaluations were performed for six Brazilian watersheds of the Ribeira de Iguape River basin, and for the Coweeta River basin in North Carolina. EMERGY is the energy of one kind used directly and indirectly to make a product or service. Maps and graphs included the water energies used, empower, and river transformities. Transformity is EMERGY per unit of energy. Water Resources Council. 1979. *A Unified National Program for Flood Plain Management.* United States Water Resources Council. This report (1) sets forth a conceptual framework for floodplain management, (2) identifies available management strategies and tools for reducing the risk of flood loss, minimizing the impact of floods on human safety, health, and welfare, and restoring and preserving natural and beneficial floodplain values; (3) assesses the implementation capability of existing Federal and State agencies and programs; and (4) makes recommendations for achieving "A Unified National Program for Floodplain Managment." Although drafted as the result of a Federal initiative, the concepts and strategies of this report are presented from a national perspective and offer guidance applicable to all governmental and nongovernmental interests.

Whitfield, Douglas F. 1993. Emergy Basis for Urban Land Use Patterns In Jacksonville, Florida. Thesis, University of Florida, Gainesville, FL. 212 pp.

Over the last fifty years, American cities have experienced a large proportion of their growth in the suburbs. In many cities, residential and commercial activities are more dominant in the suburbs than in downtown. This suburban land use pattern has changed the level of resources used to connect and support urban functions. This study used Jacksonville, Florida to identify the major resources shaping the organization of urban systems. The relationship between land use patterns and urban infrastructure, resource consumption in transportation, and environmental contributions was investigated. Resources were evaluated with emergy, an energy based unit of value, to place all resource contributions, both natural and human, on a common basis. Emergy analysis represents a donor based measure of value in contrast to a user based approach used in economics.

Chapter 2
Stormwater, Pollutant Loads and Management in the Lafayette and Munson Basins

Mark T. Brown and Neal Parker

INTRODUCTION

Non-point sources of pollution have increasingly become the focus of attention as point sources such as the outfalls from sewage treatment and industrial plants have been eliminated from surface water bodies. Over the past decade, while great improvement of surface water quality was achieved with elimination of direct discharge of wastes, non-point sources became more obvious as an important contributor to surface water quality degradation.

In this study, two watersheds within the St. Marks watershed, the Lake Lafayette and Lake Munson watersheds are looked at in detail (Figure 2-1). The two watersheds are similar, yet exhibit differences that make the analysis of their stormwater, pollutant loads and the changes expected in the future instructive. Lake Lafayette basin has urban development along its western edge, with rural development and agricultural uses in the eastern and northern portions of the watershed. Lake Munson, on the other hand, is more developed. With the city of Tallahassee occupying most of its northern sub-basins, Lake Munson receives almost 2/3's of it total inflows from heavily urbanized areas. In all, the Lake Lafayette basin is less developed, but significant expansion of urban uses is occurring and anticipated in the future, while Lake Munson is more developed. The contrasts between an urbanized watershed and one that is beginning to urbanize, provide an interesting test of management alternatives. In the Lake Munson basin, reductions in pollutant loading will require retrofitting an already urbanized watershed, while in the northern portions of the Lake Lafayette basin, innovative development options that include wetlands and special development buffers for waterbodies can be tested.

Description of the Study Areas

The Lake Lafayette watershed (Figure 2-2) encompasses approximately 80 square miles of land in the northwest quadrant of the St. Marks Basin. The Lake is situated in the lower reaches of the watershed and has been impounded for years in three places. These impoundments basically divide the lake into 4 sections: Upper Lafayette, Piney Z, Lower Lafayette, and Alford Arm . Alford Arm receives surface water from the northern portions of the drainage basin which is predominately rural and contains a large amount of storage in natural depressions and man-made ponds, as well as numerous closed basins. Lake Lafayette receives drainage from areas immediately surrounding the lake and a major tributary called Northeast Drainage Ditch that drains the heavily urbanized areas to the west.

Adjacent to the Lake Lafayette drainage basin is the Lake Munson basin, encompassing approximately 69 square miles (Figure 2-3). The heavily urbanized areas of Tallahassee in the northern portion of the basin contributes a significant amount of the total drainage through what are called the West, Central, and East Drainage Ditches. The western portions of the basin are dominated by the Apalachicola National Forest which drains into Bradford Brook to the Bradford Lake Chain, eventually to Munson Slough. There are numerous closed basins and Bartel et al. (1991) estimate that about 25% of the basin does not contribute stormwater to lake Munson. In the past, several sewage treatment plants discharged to surface water, but these discharges were eliminated in 1984.

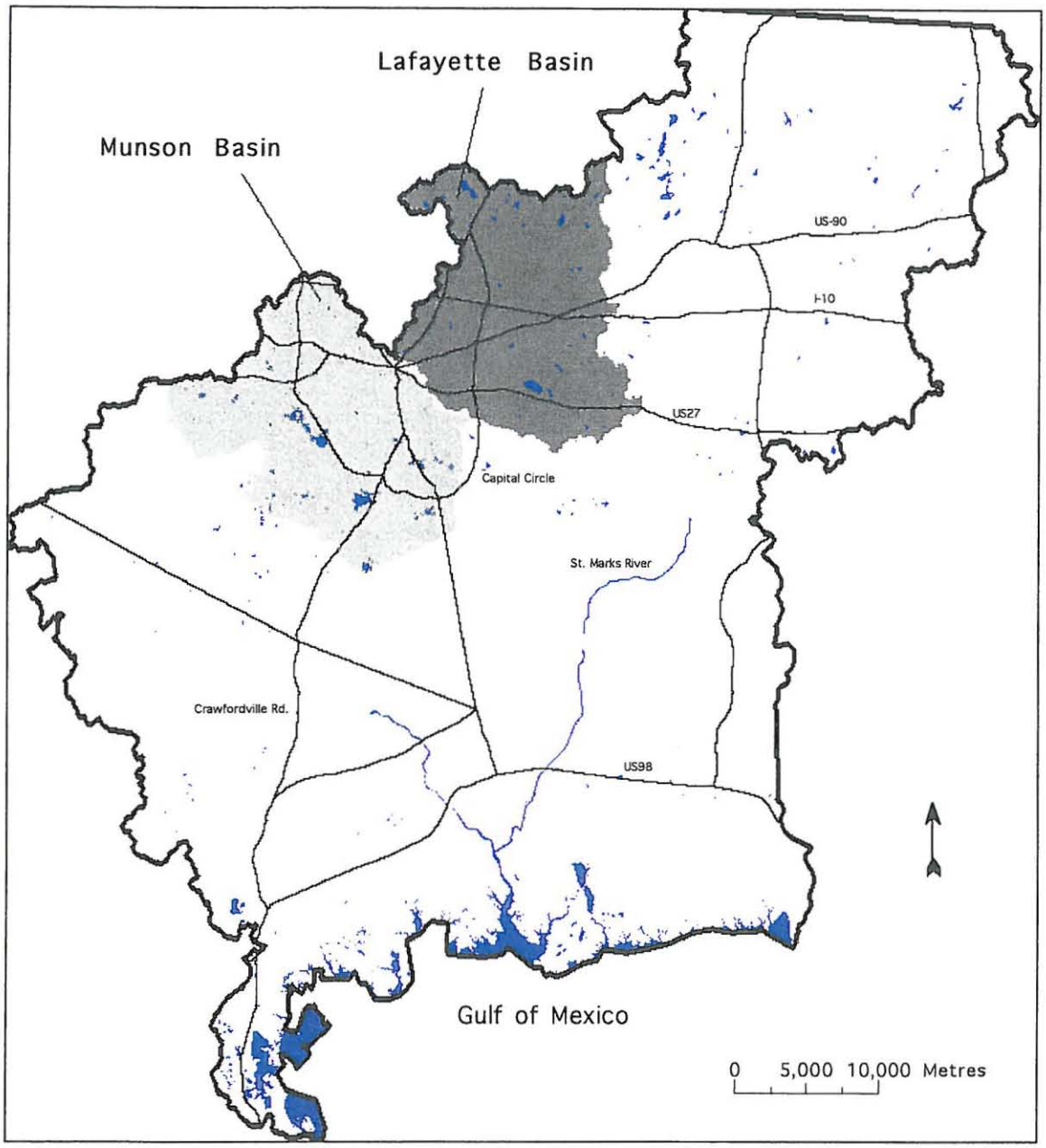


Figure 2-1. Map of the St. Marks Watershed showing Lake Lafayette and Lake Munson Basin

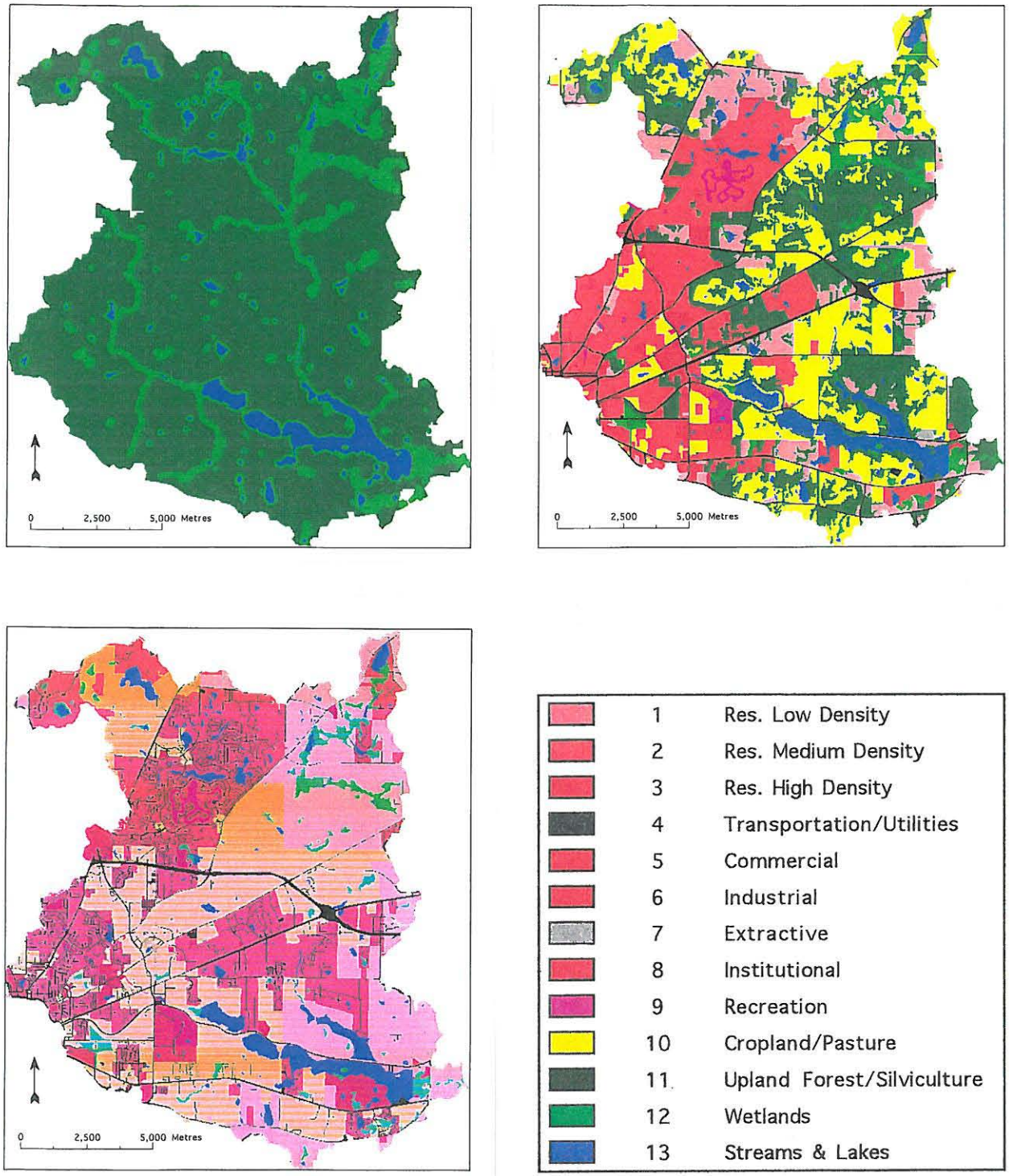


Figure 2-2. Maps of the land use and land cover in the Lake Lafayette Basin for the past (top left), present (top right) and future (bottom).

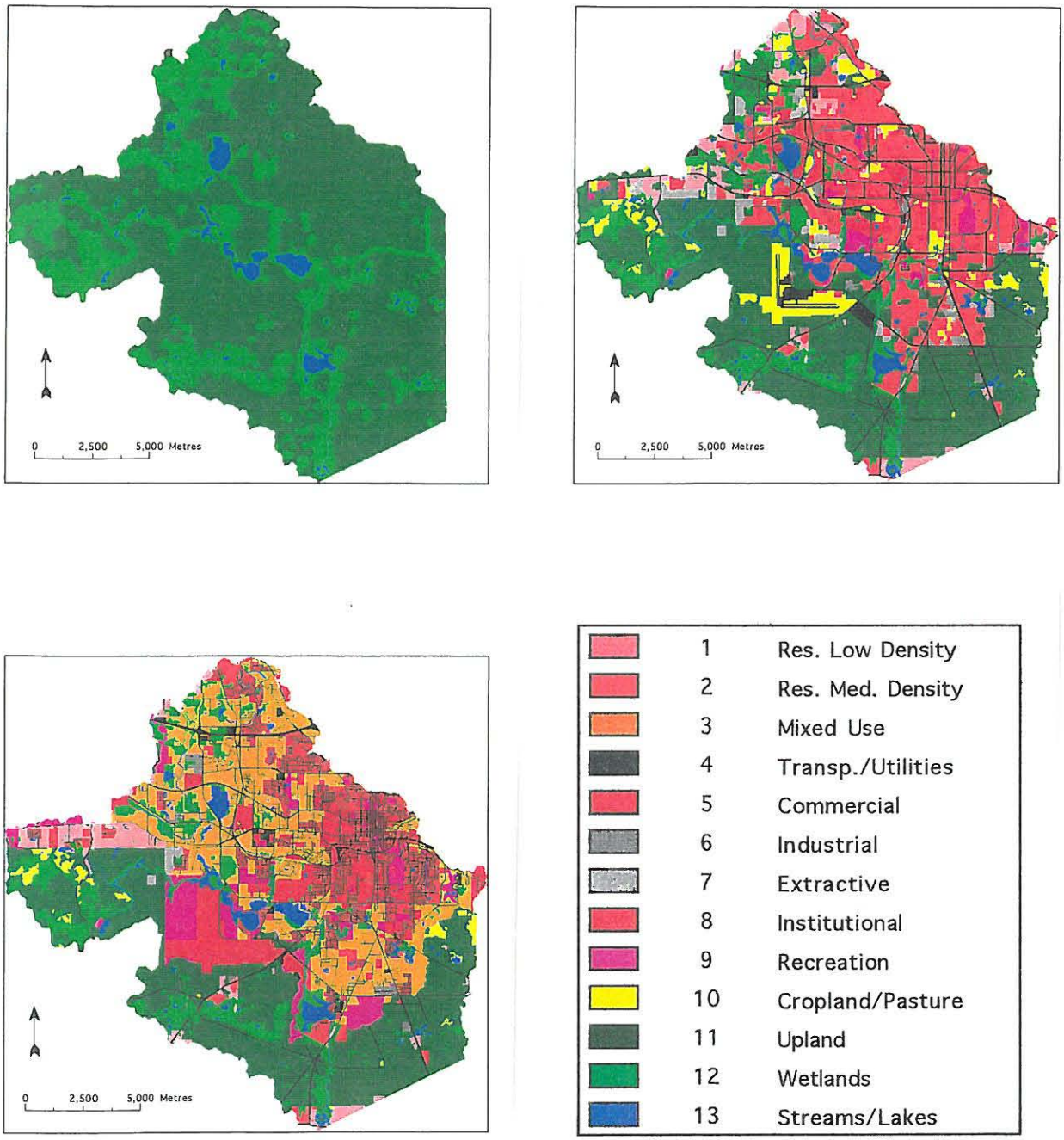


Figure 2-3. Maps of the land use and land cover in the Lake Munson Basin for the past (top left), present (top right) and future (bottom).

Plan of Study

To better understand the effects of spatial distributions of land uses on pollutant loading, spatial models that used land use and topography were developed that modeled pollutant load received by surface water features. Lake Lafayette and Lake Munson were divided into sub-basins to evaluate the various sub-basin contributions to each water body. In addition, pollutant loads were modeled for major water bodies and closed basins within each of the larger watersheds to provide perspective on areas of concern.

As a means of understanding loss of "basin function", pollutant loads were modeled for three time periods for each of the basins: past, present, and future, and then compared. Stormwater management options including Best Management Practices (BMP's), and restoration of historic wetlands were tested with the models for the present and future conditions to evaluate their effectiveness in reducing pollutant loads.

METHODS

In this study two different methods were used to model pollutant load using Geographic Information System (GIS) data layers. The first uses a fine resolution "Drain Model" for pollutant transfer that predicts yearly average load for all locations within a basin. The second uses a distance modified basin scale pollutant load model that sums potential load for sub-basins. Comparisons were made between past conditions assuming no development, present conditions based on 1989 land use / land cover, and for future conditions based on the Tallahassee-Leon County 2010 Comprehensive Plan.

In both models, total phosphorus (TP) is used as the constituent of concern. Other pollutants could have been used, by multiplying by their areal loading rates (Table 2-1). In previous studies (Brown and Tilley, 1995, Tilley and Brown, 1998) and in the Stormwater Water Management Plans produced by the NWFWMD (Bartel, et al, 1992) it has been determined that few significant differences in relative loading exist between constituents, thus in essence, TP can be thought of as an indicator for most other constituents of stormwater.

Drain Model

The Drain Model was used to model pollutant transfer, so that annual pollutant load could be estimated for any point in the watershed. Thus pollutant load delivered by several streams or ditches to a lake can be evaluated separately, and management actions taken accordingly. In like manner, the annual pollutant load from stormwater overland flow to sinks and smaller lakes can be determined as the sum of the flows entering through the lake or sink edge.

Figure 2-4 is a diagram of the steps in the GIS framework necessary to model pollutant transfer from land uses to surface water bodies. The model requires as inputs a coverage[1] of the surface water bodies, a topography coverage (DEM), and Land use / land cover. The surface water bodies and topography coverages were combined to produce a third coverage which depicts flow paths. Essentially, the flow path coverage shows the direction and volume of pollutant transfer from any point within the watershed "downhill" to the water bodies (or sink holes). A coverage of Land Use / Land Cover was used in combination with Areal Stormwater Pollutant Loading Rates (Table 2-1) to generate a Pollutant Loading coverage. The Pollutant Loading coverage was, in turn, combined, or drained through the Flow Path Coverage to generate the Pollutant Transfer coverage. Each cell in the Pollutant Transfer coverage has the total yearly pollutant load received from "uphill" which is passed on to cells "downhill." Thus by reading the value in any cell, one is reading the accumulated pollutant load in that cell.

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[1] A coverage is GIS terminology that refers to a thematic, GIS map. Usually maps are paper copies of GIS data, and several coverages can be combined to make one map.

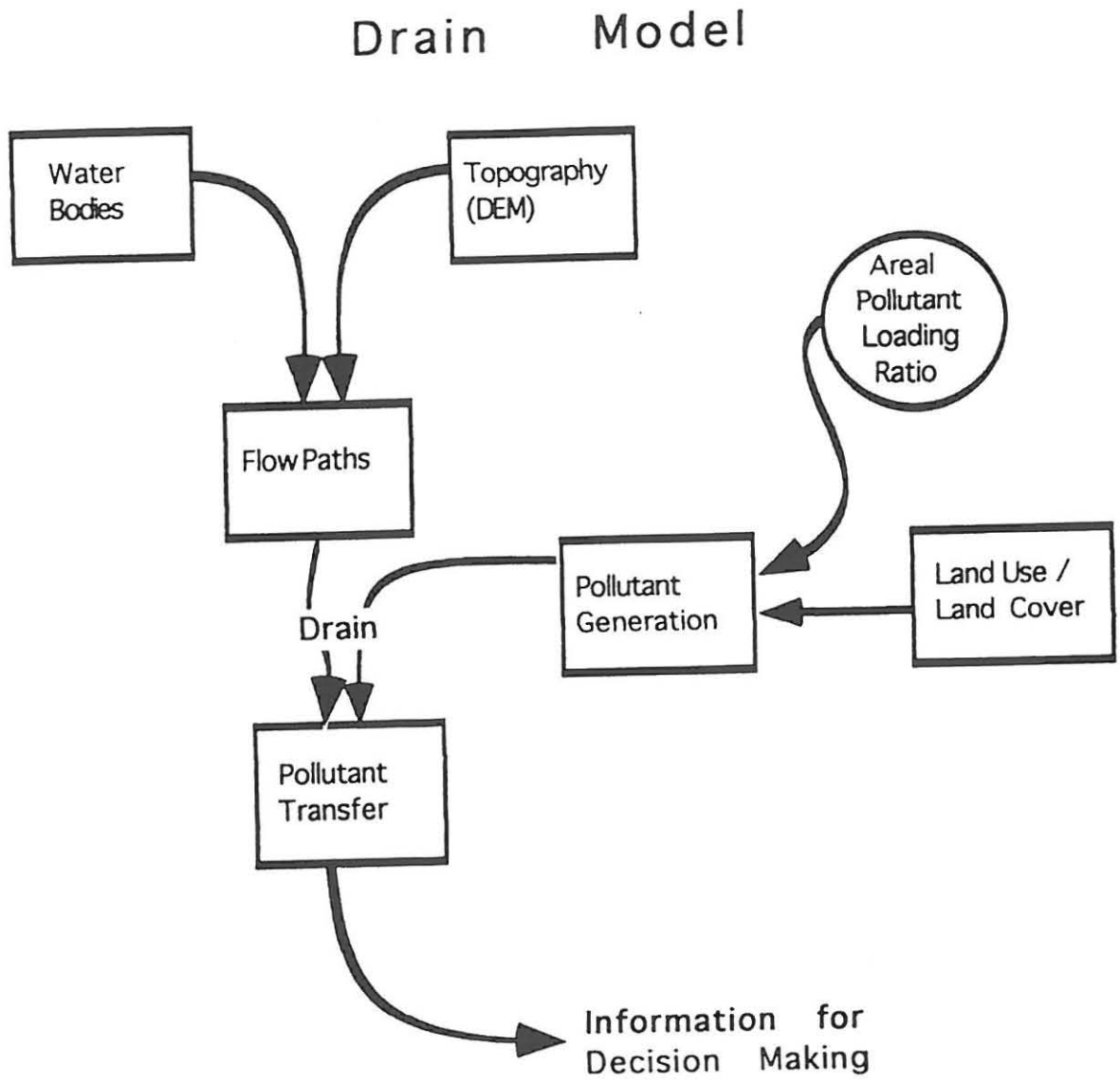


Figure 2-4. Diagram of the steps in the GIS framework for the DRAIN Model of pollutant transfer from land uses to surface water bodies

Table 2-1. Areal Stormwater Pollutant Loading Rates (after Harper, 1995)

Land Use Category	Areal Loading TP (kg/ha*yr)
Low density residential	1.53
Single family residential	2.84
Multi-family residential	8.24
Low intensity commercial	3.11
High intensity commercial	9.39
Industrial	5.94
Highways	6.32
Agriculture (pasture)	4.2
Agriculture (general)	2.64
Open Space	1.22
Mining	1.35
Wetland	0.00
Open water	0.00

Distance Modified Pollutant Load Model (DMPL Model)

This model was used to evaluate sub-basin annual pollutant loading to water bodies. Generally, the pollutant load flowing to water bodies is a function of the intensity of activity in a watershed and the physical characteristics of the watershed. Most pollutant load models deal with watersheds as single dimensional space, treating all pollutant loads the same regardless of their location within a watershed. With the increased availability of spatial data, it is possible to model pollutant loading spatially; based not only on slope, but also on location, or distance from the water body. The farther a source of pollutant generation is from a water body, the more natural "treatment" may occur in the intervening space and thus the lower the actual load received by the water body.

The Distance Modified Pollutant Load Model was developed to determine total pollutant load based on topography, "effective" distance from the water body, land use/land cover, and areal pollutant loading rates (Table 2-1). Figure 2-5 is a diagram of the steps in the GIS framework necessary to model distance modified pollutant load for sub-basins. Required inputs to the model are water bodies, topography (DEM's), and land use / land cover. The waterbody and topography coverages are combined to generate a coverage that measures effective distance from waterbodies. Effective distance is a slope modified straight line distance, so that steeper slopes have shorter effective distance for the same straight line distance. All locations within a watershed are assigned a unique distance to the nearest downhill water body, based on the relationship graphed in Figure 2-6. The coverage of land use / land cover is multiplied by aerial pollutant loading rates to create a pollutant generation coverage where each land use contains the unit area, annual pollutant generation. The pollutant generation coverage is multiplied by the effective distance coverage to produce the annual distance modified pollutant loading coverage. Summing by sub-basins provides total annual loads for each sub-basin.

Alternative Management Schemes

Three stormwater management schemes were tested using the Drain Model to determine how present and future pollutant loads might be reduced by management practices and ecological engineering. First, Best Management Practices (BMP's) were tested by assuming pollutant load reductions of 10% for residential land uses and 20% for commercial/ industrial land uses. BMP's might consist of street sweeping, the use of roadside swales instead of curb and gutter, and regular cleaning and maintenance of stormwater conveyance systems. The second management scheme consisted of a wetlands reconstruction alternative. For this alternative, all historic wetlands were reconstructed in the basins, regardless of their locations, or what present day structures or infrastructure might occupy the site. Finally the third alternative that was tested combined BMP's with wetlands reconstruction.

The objective of modeling these alternative management schemes was to evaluate the effects of management and ecological engineering on pollutant load. The BMP alternative represents a conservative approach and a conservative estimate of pollutant reduction, assuming current technologies with minor investments in stormwater management. The wetlands alternative represents, in some respects, a radical approach to stormwater management, by reconstructing and rehabilitating historic wetlands, regardless of their location, so they will function as stormwater treatment areas. In a sense, this alternative indicates how much basin function has been lost through years of development and engineering of stormwater systems that was concerned mainly with quickly and efficiently removing stormwater

Distance Modified Pollutant Load Model

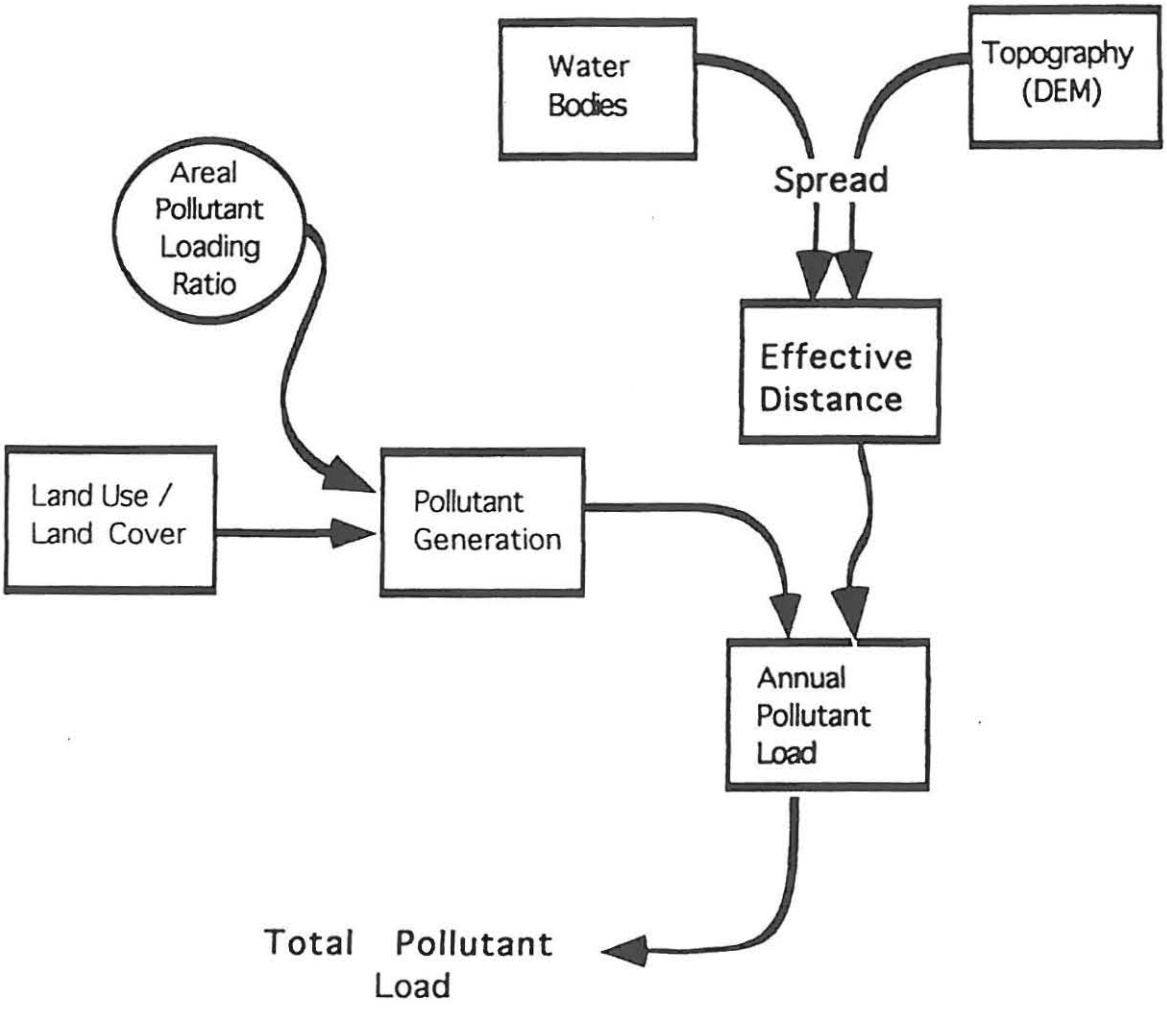


Figure 2-5. is a diagram of the steps in the GIS framework for the Distance Modified Pollutant Load Model (DMPL Model) of distance modified pollutant load for sub-basins.

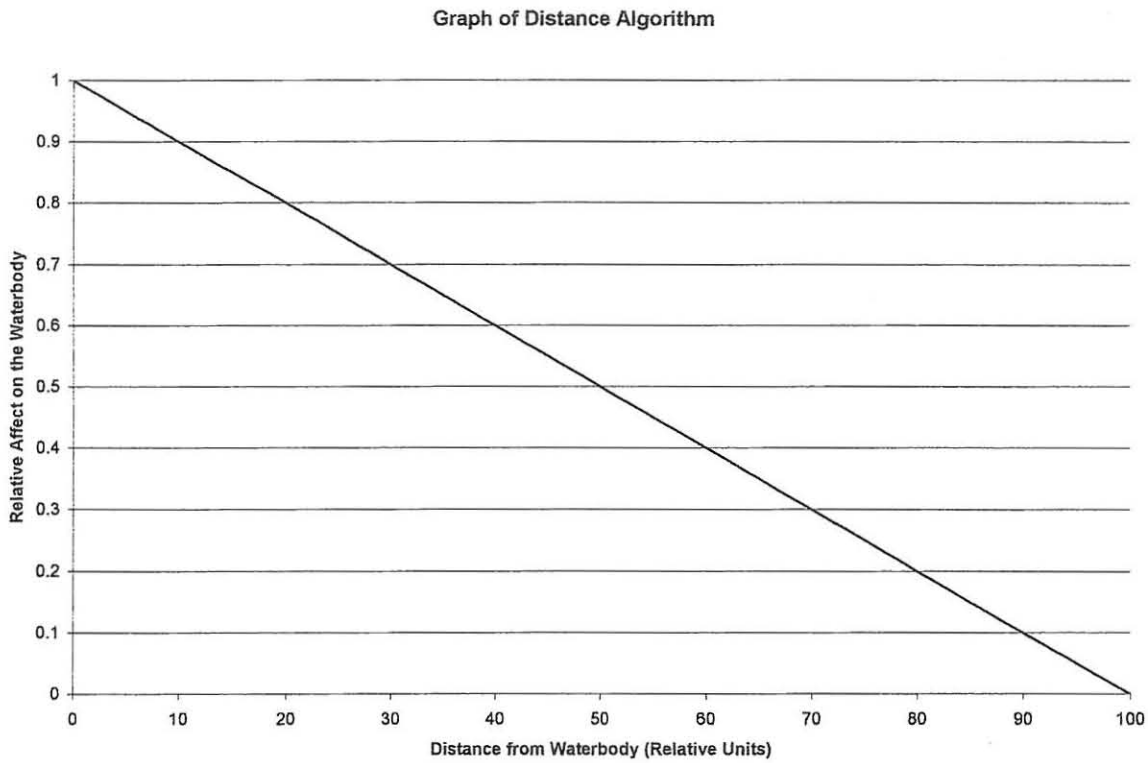


Figure 2-6. Graph of the distance algorithm used to assign all locations within a watershed a unique distance to the nearest downhill water body.

RESULTS AND DISCUSSION

Stormwater in the St. Marks Basin

Stormwater is the water that runs off lands during rainfall events. When graphed against time, the amount of stormwater runoff from a watershed exhibits a curve called a "runoff hydrograph." Two such hydrographs are depicted in Figure 2-7. The two graphs show how the amount and timing of runoff following a storm event changes when a watershed is developed. As a watershed becomes more urbanized, stormwater runoff increases in amount and the speed at which it flows off the land is increased as well.The lower graph is a hydrograph for an undeveloped watershed, showing the slow increase in runoff after a rainstorm, with volume peaking about 3 hours after it begins to rain. The total area under the curve is the total volume of runoff that results from the rainstorm. The higher graph is the hydrograph for the same watershed but with an increase in impervious surface that has resulted from urbanization. Two things are apparent. First, the timing of peak runoff has been changed. The peak comes 2 hours earlier than in the undeveloped watershed. Second the amount of runoff has increased (area under the second graph is larger than the area under the first graph) because the area of impervious surface forces more water to runoff instead of percolating into soils.

Pollutant Loading and Percent Impervious Surface

As water runs off the land it carries with it many kinds of materials of varying sizes. Larger things typically can be seen in drainage ditches, streams, and along the rivers and lakes of the St. Marks watershed as the "jetsam and flotsam" of wood, plastic, cloth, bottles, cans, etc. Stormwater carries sediments, or soil particles, that have been eroded from the watershed. It also carries many materials and chemicals that are invisible to the eye because their are dissolved, or are so small that they go unnoticed without the aid of a magnifying glass. All of these things might be considered "pollutants", that is, too much of something in the wrong place at the wrong time. However, while the big things carried by stormwater are the most noticeable, and often get the attention of the public to focus on clean up of a water body, it is the smallest of these...the dissolved solids, chemicals, and organic matter that cause the most trouble.

The things carried by stormwater are often called its pollutant load and depend on the type of land the water flows over. In watersheds that are covered by forests, water running off the land during storms might have a small amount of chemicals like soil nutrients, some organic compounds that are washed from the soil surface, and bits and pieces of decaying vegetation. Usually, without development, a watershed exports very little eroded soil, because the vegetative cover is good protection against erosion, and acts to filter sediments if some should begin to move.The volume of water flowing off undeveloped watersheds is slowed down by the vegetation and allowed to percolate into soils, reducing the erosive ability of runoff and minimizing velocity. When developed, with increasing amounts of impervious surface, the amount of water and the speed at which it runs off increase.These factors have profound influence on the pollutant load that stormwater carries.

Because stormwaters often end up in surface water bodies, the load of pollutants it carries is of interest to the public. Water bodies that receive stormwaters often exhibit unacceptable characteristics when polluted. Not only is the total load important, but the types of materials and chemicals that are carried by stormwater are important. Experience has shown that stormwater from urban areas contains, fertilizers, metals, and sediments, as well as a myriad of other chemicals in smaller amounts. Stormwaters from agricultural lands contain fertilizers, pesticides, sediments, and lessor amounts of metals. Which of these pollutant is of most concern, depends in large part on their concentration. Usually, in urban watersheds, the effects of phosphorus in stormwaters are more exaggerated than other chemicals. However, sediments can often be a major problem. In agricultural watersheds, often phosphorus is a problem, but pesticides can be extremely important because of the large effects that result form such small concentrations. Typical constituents of stormwater from urban watersheds are given in Table 2-2.

The amount of impervious surface within watersheds is related to the intensity of human activity and as a result, is a good predictor of stormwater water quality. The impervious surface in

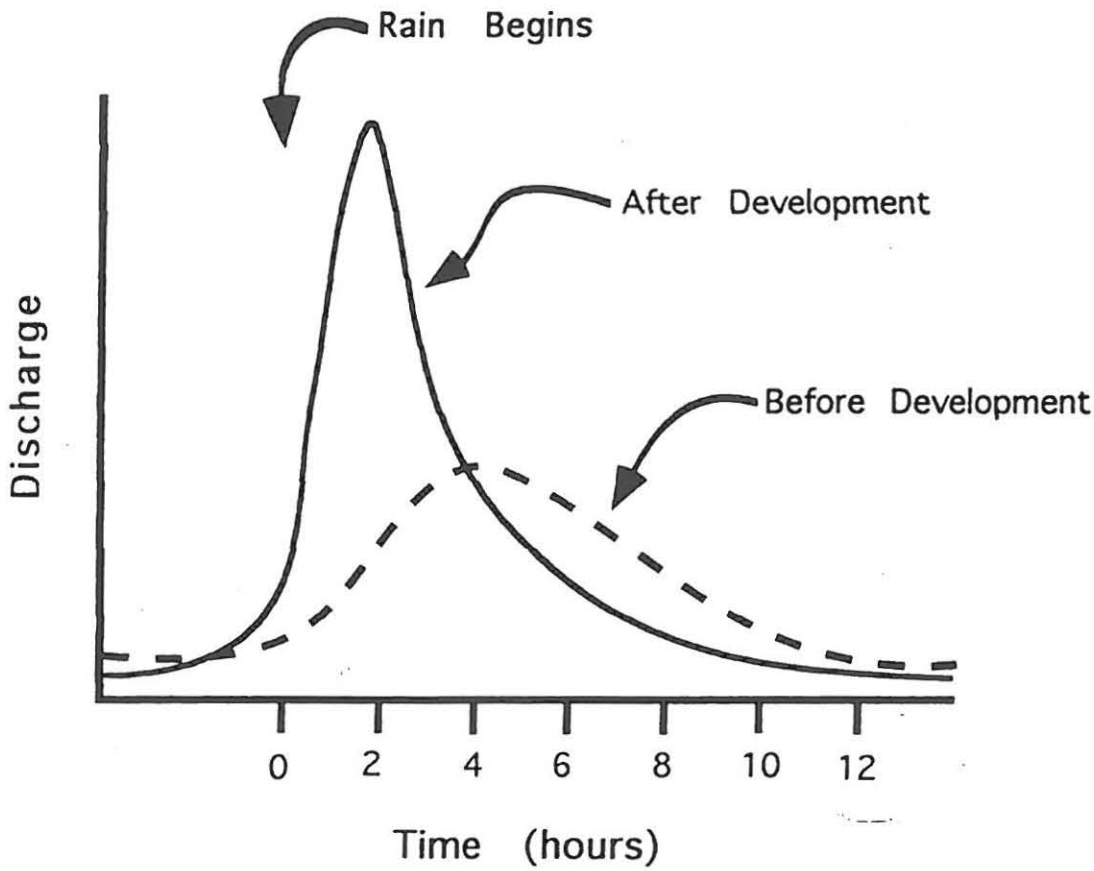


Figure 2-7. Stormwater hydrographs for a developed and undeveloped watershed showing how the amount and timing of runoff following a storm event changes when a watershed is developed.

Table 2-2. Characteristics of Urban Stormwaters (from Corbitt, 1990)

Parameter	Range
Biological Oxygen Demand	1 - 700 mg/l
Total Organic Carbon	1 - 150 mg/l
Chemical Oxygen Demand	5 - 3,100 mg/l
Suspended Solids	2 - 11,300 mg/l
Total Solids	200 - 14,600 mg/l
Volatile total solids	12 - 1,600 mg/l
Settleable solids	0.5 - 5,4000 mg/l
Organic N	0.01 - 16 mg/l
NH_3N	0.1 - 2.5 mg/l
NO_3N	0.01 - 1.5 mg/l
PO_4	0.1 - 10 mg/l
Total PO_4	0.1 - 125 mg/l
Chlorides	2 - 25,000 mg/l
Oils	0 - 110 mg/l
Phenols	0 - 0.2 mg/l
Lead	0 - 1.9 mg/l

itself is not the cause of poor water quality, but is a contributing factor, since increased impervious surface translates into increased runoff and decreased ability for pollutant processing in soils. But more importantly, as impervious surface area increases in watersheds, it indicates that intensity of human activity increases as well. Human presence and all the activities associated with modem life generate by-products, many of which lay around on the ground just waiting to be swept along with stormwater runoff as it is directed toward downstream locations. Oils and metals from automobiles, fertilizer runoff, and chemicals from combustion of fossil fuels are the most prevalent of these by-products. During rainstorms these materials and chemicals are washed from the ground surface, along with loose soil particles, organic debris, etc and deposited in streams and rivers.

When the amount of impervious surface within watershed areas is graphed against pollutant load as in Figure 2-8, very often there is a strong relationship between the two. In this graph, pollutant load in the sub-basins of Lake Lafayette and Lake Munson watersheds are compared to impervious surface. As the graph shows, "impervious-ness" is strongly correlated to pollutant load. Numerous other studies across the United States have shown consistently that there is a strong correlation between impervious-ness of drainage basins and the health of the receiving surface water body (Klein 1979; Griffin, 1980; Schueler, 1987; Todd, 1989; Schueler, 1992; Booth and Reinfelt, 1993; and Schueler, 1994). Thus impervious surface may be a very good predictor of stormwater quality and the health of downstream waterbodies.

A study of water quality in Wisconsin surface water bodies has shown that there is a shift in the health of waterbodies receiving stormwaters when impervious-ness exceeds about 10% - 20% (Schueler,1994). Water bodies receiving stormwaters from basins with between 10% and 20% imperviousness exhibited lower indices of ecosystem health than basins with less impervious surface. As the percent imperviousness increased the indices decreased. Schueler (1992) suggested that when imperviousness reached about 30% of basin surface area, water bodies receiving stormwaters exhibited degraded conditions. In another study of Wisconsin streams Wang et al. (1997) showed that indices of ecosystem health (they used Index of Biotic Integrity [IBI]) were strongly correlated with land use in watersheds. They found that when agricultural uses exceeded 50% of total basin area, there was a marked decline in the IBI. In addition their study reinforced the findings of Schueler (1994) that urban uses greater than 10% - 20% of basin area significantly lowered IBI's for basin streams.

In our study of the Lafayette and Munson watersheds, we ranked sub-basins based on their imperviousness for the present land use conditions, and for future land uses based on maps provided by Leon County Planning. Figures 2-9 and 2-10 show the sub-basins ranked by imperviousness for Lake Lafayette and Lake Munson Basins respectively. The lightest gray represents the lowest percent impervious surface, while the darkest gray has highest impervious surface area. In the Lafayette Basin (Figure 2-9a), 4 basins had percent imperviousness less than 20% (basins 1,2,3, and 9), and only basin No.1 had percent imperviousness less than 10%. In the Munson Basin (Figure 2-10a), two sub-basins had imperviousness equalt to or less than 20% (Sub-basins 6 and 7), while four sub-basins (2,3,4 and 5) had greater than 30% imperviousness surface.

Based on the previous studies by others which suggested that imperviousness was related to ecosystem health (Schueler, 1992,1994; Wang et al.,1997), we might conclude that at the present time 7 sub-basins out of the total number of 16 sub-basins within the Lafayette and Munson watersheds have sufficient impervious surface (greater than 30%) to warrant serious concern for the ecological health of surface water bodies within those sub-basins. Further, 6 sub-basins within these two watersheds have sufficient areas of impervious surface (10% - 20% imperviousness) to raise concern for the ecological health of their water bodies.

When imperviousness is mapped for the future land use condition as in Figures 2-9b and 2-10b, 14 basins have percent impervious surface greater than 30%, while no sub-basins have less than 10%. Clearly, there is cause for concern. As a predictor of ecological health, the increase in percent impervious surface within the Lafayette and Munson watersheds provides an indicator that out look for the health of surface water bodies within these watersheds is questionable.

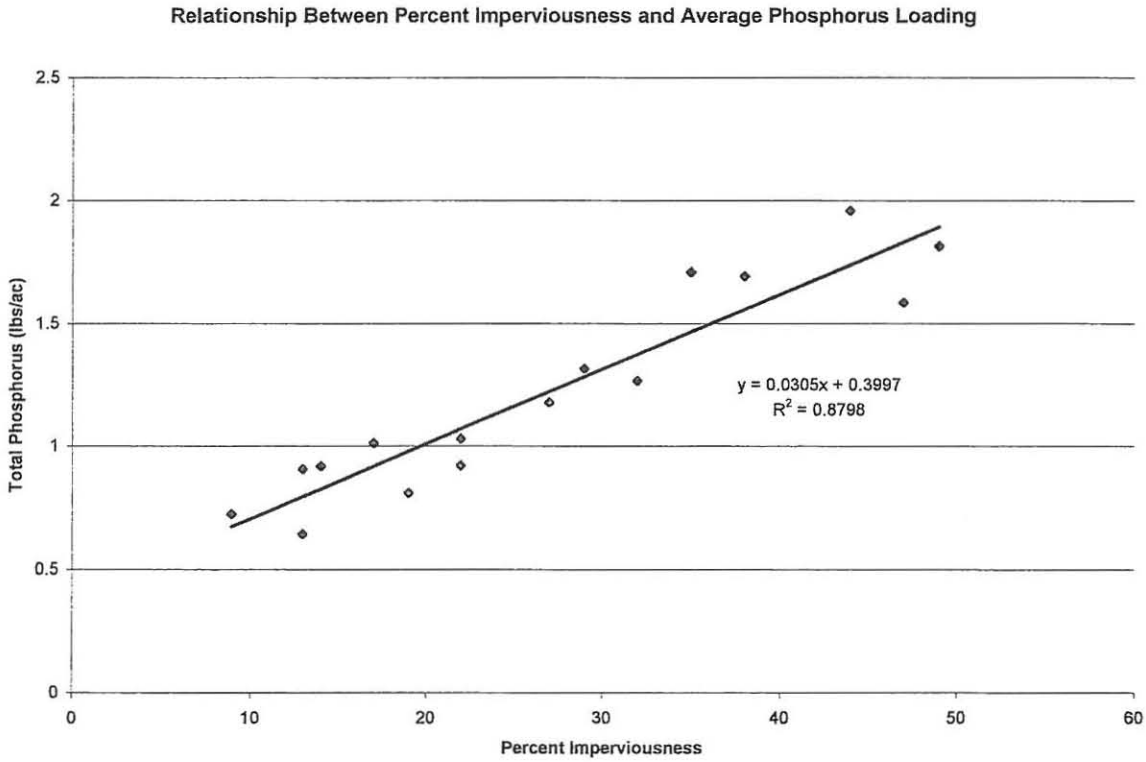
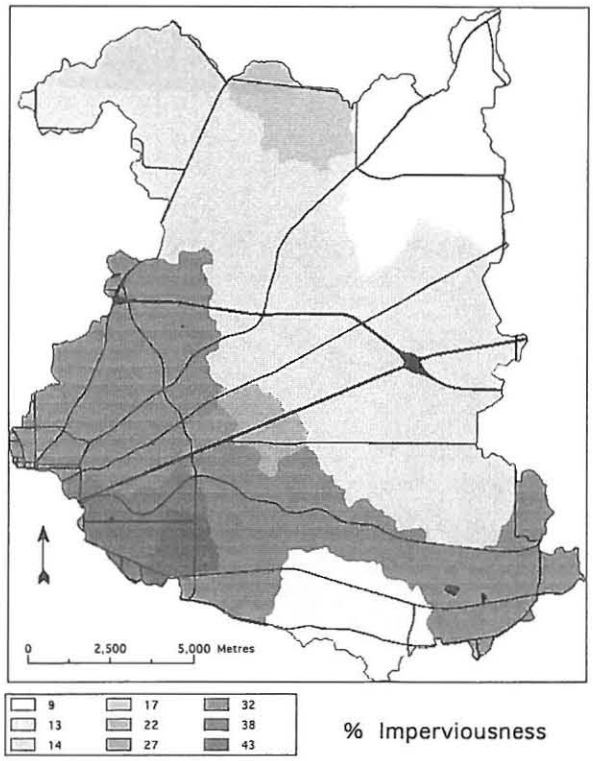
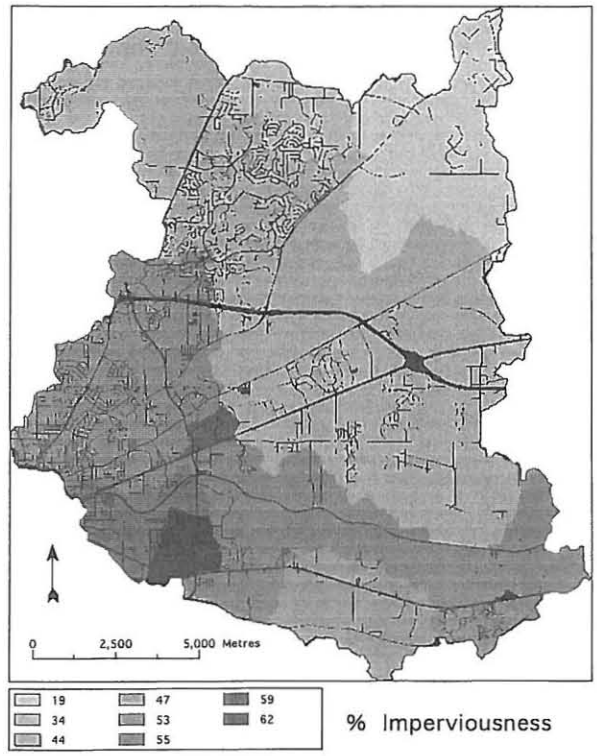
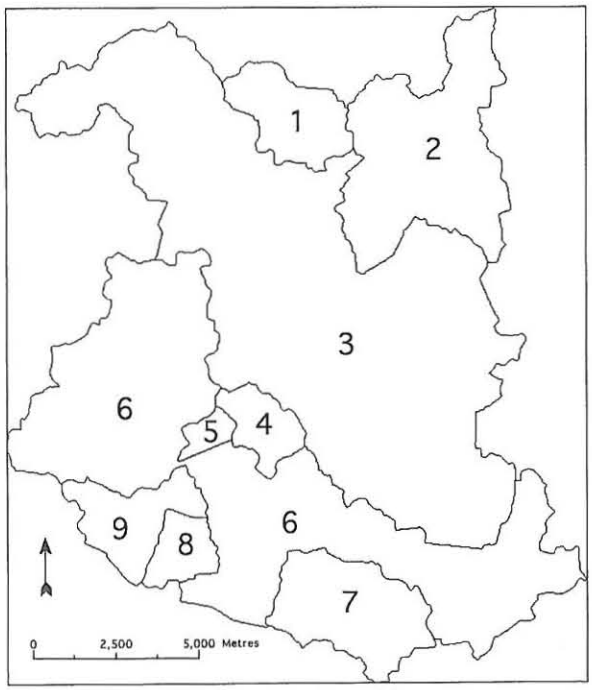


Figure 2-8. Pollutant load vs. impervious surface in the sub-basins of Lake Lafayette and Lake Munson watersheds showing a strong correlation between "impervious-ness" and pollutant load.



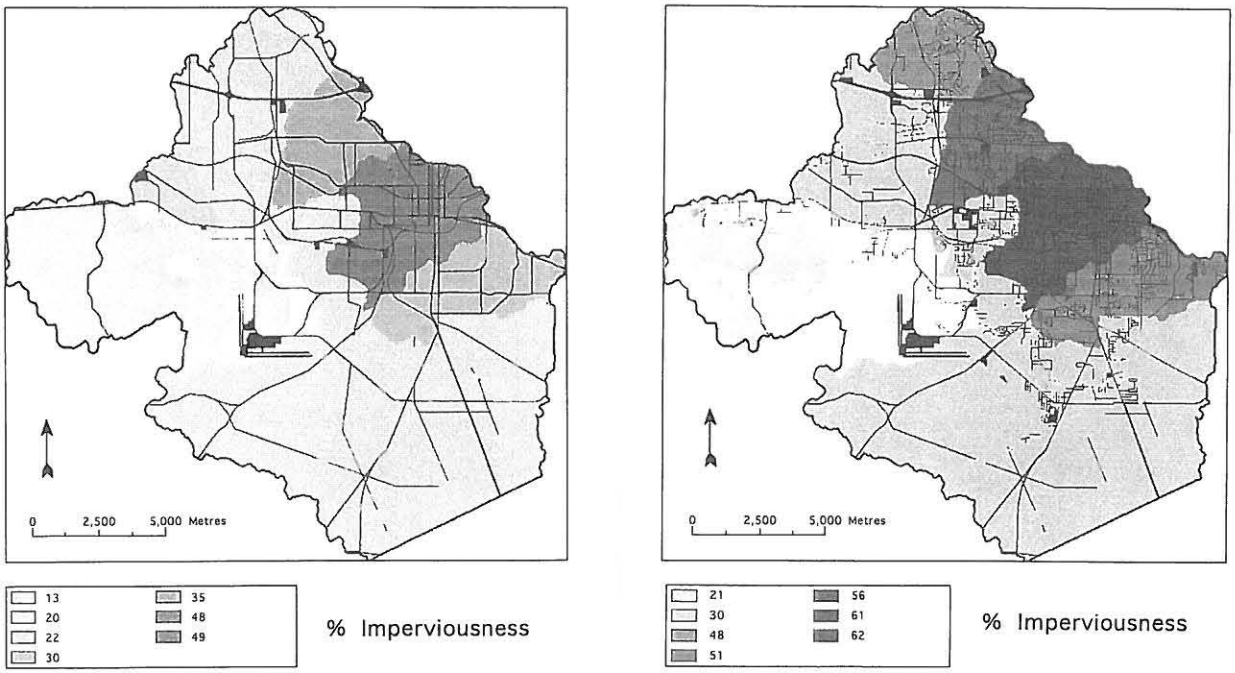


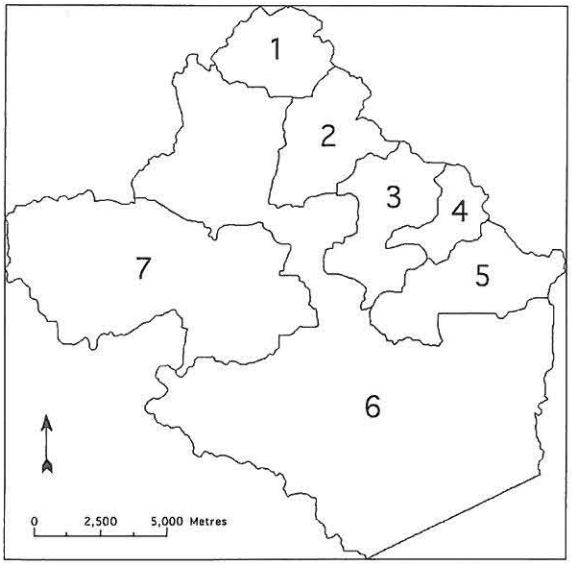


1. Gilberts Pond Outlet
2. Roberts Pond Outlet
3. Alford Arm
4. Buck Lake Outlet
5. Unnamed Run
6. Lake Lafayette Drain
7. Unnamed Slough
8. Unnamed Run
9. Mall Drainage Area

Fig. 9

Figure 2-9. Maps of the sub-basins of Lake Lafayette Basin ranked by imperviousness for the present condition (top left) and the future (top right). The lightest gray represents the lowest percent impervious surface, while the darkest gray has highest impervious surface area





1. Unnamed Run
2. Godby Ditch
3. Central Drainage Ditch
4. St. Augustine Branch
5. East Drainage Ditch
6. Munson Slough
7. Bradford Brook

Figure 2-10. Maps of the sub-basins of Lake Munson Basin ranked by imperviousness for the present condition (top left) and the future (top right). The lightest gray represents the lowest percent impervious surface, while the darkest gray has highest impervious surface area.

Pollutant Loading

Using the DMPL Model, annual pollutant loads for each of the sub-basins in the Lafayette and Munson watersheds were generated. The maps in Figures 2-11 and 2-12 show the spatial distributions of pollutant loading for the past, present and future. For comparative purposes the map values are given as load per area of sub-basin (kg/ha/yr). In this way the spatial generation of pollutant load can be compared. The Lake Lafayette and Lake Munson sub-basins are numbered as follows (Figures 2-11and 2-12):

Lake Lafayette Sub-basins

1) Gilberts Pond
2) Roberts Pond
3) Alford Arm
4) Buck Lake
5) Un-named Run
6) Lake Lafayette Drain
7) Un-named Slough
8) Un-named Run
9) Mall Drainage Area

Lake Munson Sub-basins

1) Un-named Run
2) Godby Ditch
3) Central Drainage
4) St. Augustine Branch
5) East Drainage Ditch
6) Munson Slough
7) Bradford Brook

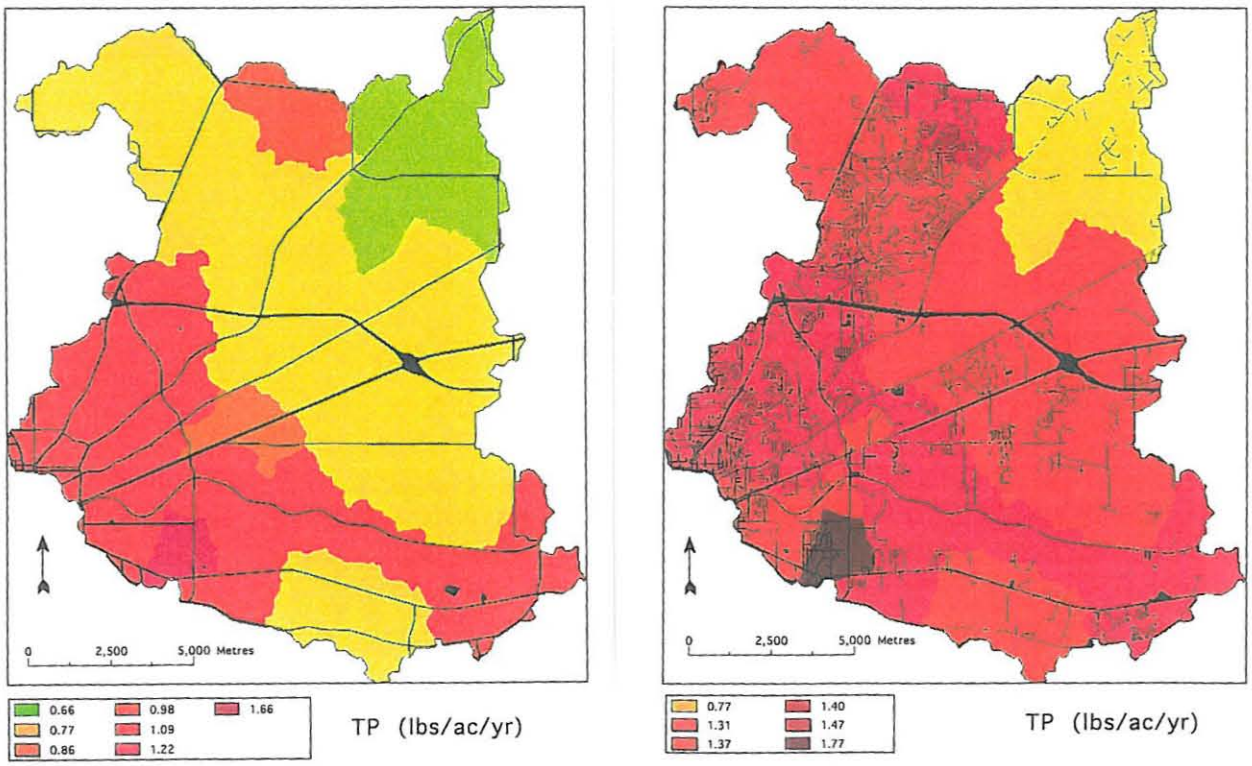
The maps show that future development in the Lafayette basin is moving northward, as the higher pollutant loads move from the southern sub-basins to the northern basins (Figure 2-11). Sub-basins within the presently urbanized area of Tallahassee have total annual pollutant loads of about 1.5 lbs/acre*yr^{-1} (1.68 kg/ha**yr^{-1}), while Gilberts Pond and Alford Arm sub-basins (sub-basins 2 and 3) have annual loads of about 0.7 lbs/acre*yr^{-1} (0.78 kg/ha*yr^{-1}). Future annual pollutant loads are significantly higher in the Alford Arm Basin (sub-basin 3) averaging about 1.37 lbs/acre*yr^{-1} (1.53 kg/ha*yr^{-1})

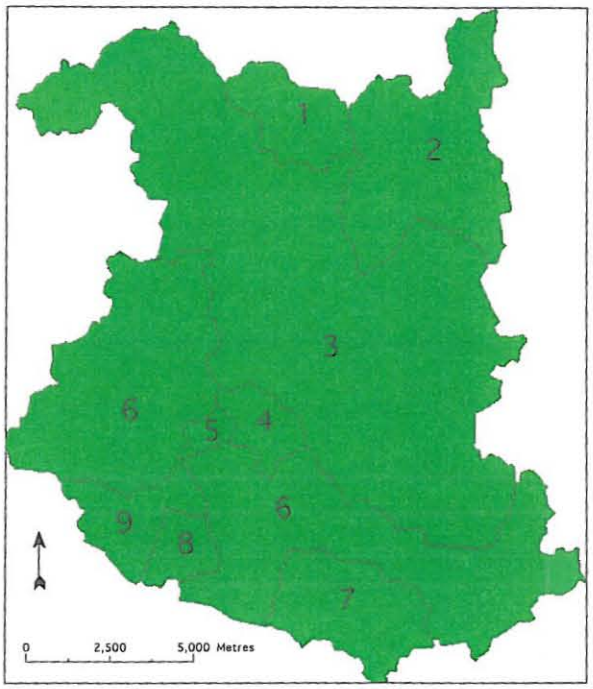
Development in the Munson basin is moving toward the southeast although the changes between the present annual pollutant loads and those of the future are not as significant as the changes in the Lafayette Basin. The Bradford Brook Sub-basin (sub-basin 7) has annual pollutant loads of about 0.7 lbs/acre*yr^{-1} (0.78 kg/ha*yr^{-1}) presently, increasing to about 0.8 lbs/acre*yr^{-1} (0.9 kg/ha**yr^{-1}) in the future. There is not much increase from the present to the future in total annual pollutant load in the Tallahassee sub-basins (sub-basins 2,3,4, and 5), since these areas are nearly fully developed.

The graphs in Figures 2-13 and 2-14 show comparisons of annual pollutant loading by sub-basin for the past, present and future conditions for the Lafayette and Munson Basins. The shortest bars in the graph are for the natural landscape, averaging about 0.4 lbs/acre*yr^{-1} (0.45 kg/ha*yr^{-1}). Urbanized areas have about 3 times these background loads (1.4 lbs/acre*yr^{-1} [1.48 kg/ha*yr^{-1}]). The biggest change from present conditions to future conditions are found in the Lafayette Basin where the moderately developed sub-basins exhibit annual pollutant load increases of between 50 to 80%. The increases in the Munson basin between the present and future condition are much smaller, with only one basin (Un-named Run #1) exhibiting a 75% increase in annual load. The remaining basins all appear to exhibit increase of between 5 and 15%.

Pollutant Transfer

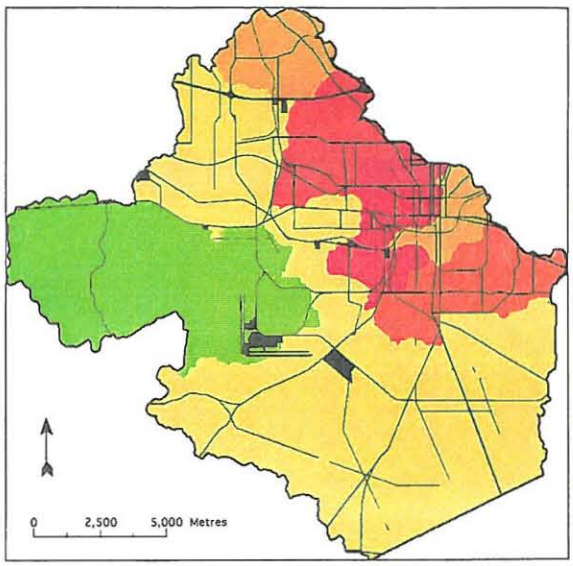
To better understand the spatial distribution of annual pollutant loads requires taking a closer look at the Lafayette and Munson Basins. The Drain Model uses an overland flow algorithm

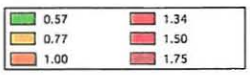




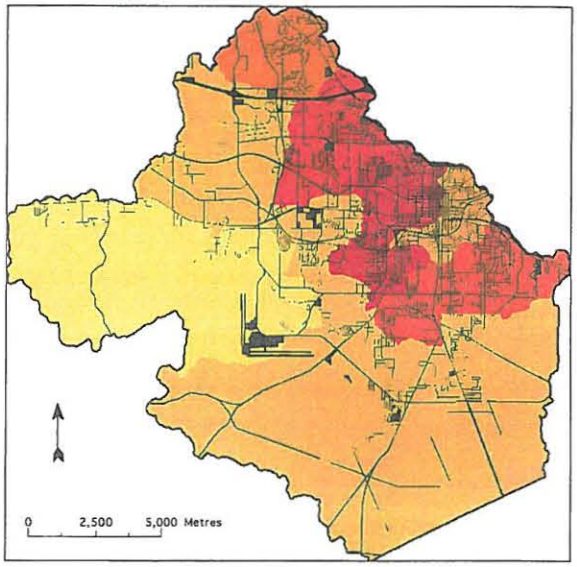
1. Gilberts Pond Outlet
2. Roberts Pond Outlet
3. Alford Arm
4. Buck Lake Outlet
5. Unnamed Run
6. Lake Lafayette Drain
7. Unnamed Slough
8. Unnamed Run
9. Mall Drainage Area

Figure 2-11.Maps of Lake Lafayette Basin showing the spatial distributions of pollutant loading for the present (top left) and future (top right). For comparative purposes the map values are given as load per area of sub-basin (lb/acre*yr^{-1}).

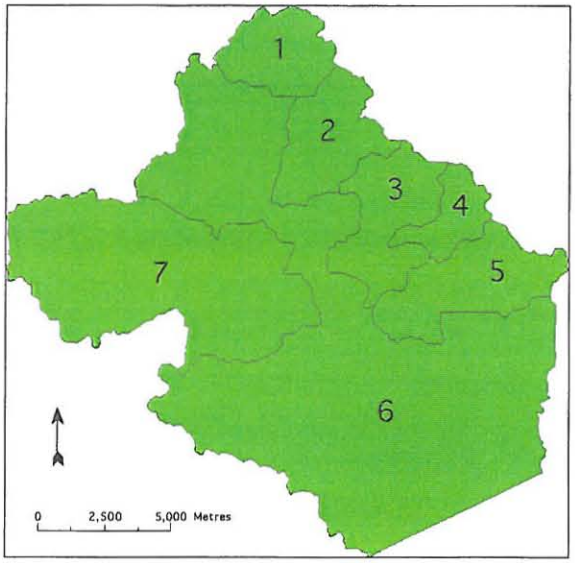




TP (lbs/ac/yr)



TP (lbs/ac/yr)



1. Unnamed Run
2. Godby Ditch
3. Central Drainage Diitch
4. St. Augustine Branch
5. East Drainage Ditch
6. Munson Slough
7. Bradford Brook

Figure 2-12.Maps of Lake Munson Basin showing the spatial distributions of pollutant loading for the present (top left) and future (top right). For comparative purposes the map values are given as load per area of sub-basin (lb/acre*yr^{-1}).

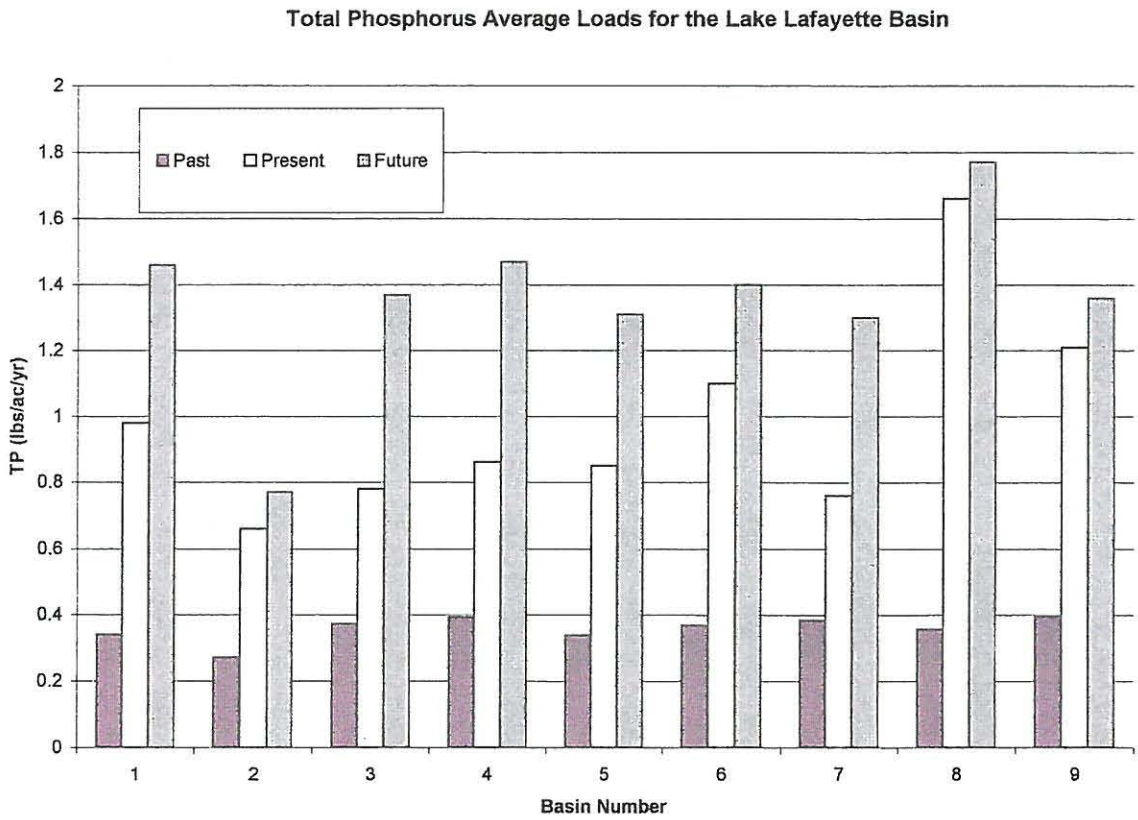


Figure 2-13. Comparison of annual pollutant loads by sub-basin for the past, present and future conditions for the Lake Lafayette Basin.

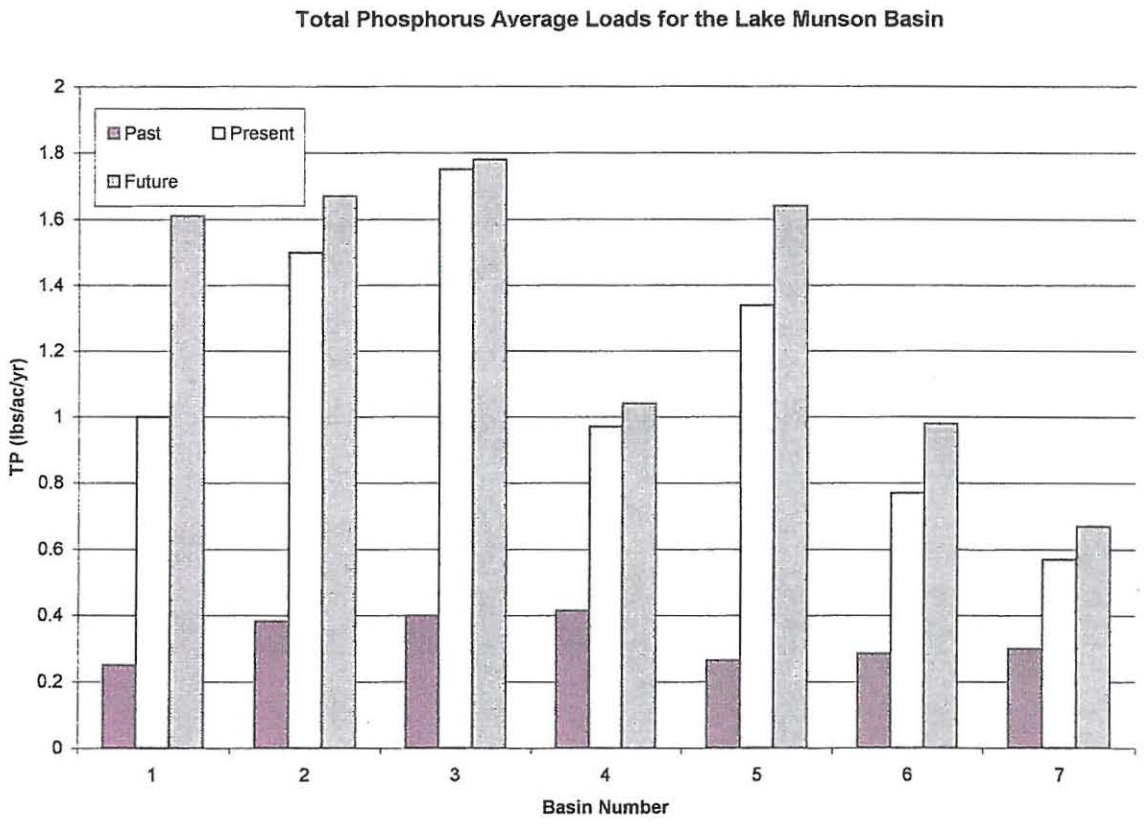


Figure 2-14. Comparison of annual pollutant loads by sub-basin for the past, present and future conditions for the Lake Munson Basin.

to converge and concentrate runoff much as water actually runs off land. The maps in Figures 2-15 and 2-16 depict annual pollutant load for past condition, present condition, and future condition in the Lake Lafayette and Lake Munson basins that were generated using the Drain Model. Darker areas indicate highest total pollutant load, while the lighter areas indicate lowest overall pollutant load. Evident in the maps is a dendritic pattern of pollutant convergence and concentration. Pollutant loads are summed along flow paths so that total load at any point in the drainage basin can be read from the maps. The graphs in Figures 2-17 and 2-18 show concentrations at several points within each of the basins for the three time periods. Figure 2-17 shows the annual pollutant loads in Lake Lafayette Basin at the following four locations : A) lower reach of Alfrod Arm Branch, B) Tallahassee Drainage inflow to Upper Lake Lafayette. C) Lake McBride, D) Killearn Lakes, . Figure 2-18 graphs the annual pollutant load at the following 3 locations in the Lake Munson Basin: A) Bradford Brook inflow to Bradford Lake, B) Northern Drainage Area (at the confluence of Gun Creek and West Drainage Ditch), and C) inflow to Lake Munson.

One of the most significant changes in the Lafayette Basin (Figure 2-17) is the 350% increase in annual pollutant load over historic loads in the lower reach of Alford Arm where because of the drainage basin size and increasing urbanization the total loads are very large. While there was a 450% increase between the historic annual load and the present load in the Tallahassee Drainage inflow to Upper Lake Lafayette the change from present to future is much smaller (15%) because the area contributing runoff is mostly built out. The model predicts an 80% increase in annual load to Lake McBride in the future and a 70% increase for Killearn Lakes.

Annual pollutant loads at the three locations within the Munson basin are shown in Figure 2-18. There is little change in the pollutant load at the inflow to Bradford Lake between the past, present and future. The most significant increase in the basin is at the inflow to Munson Slough where the increase from past to present was about 367%. The model predicts that the annual pollutant load will increase again in the future about 30% over current loads. The increase in the Northern Drainage Area was about 233% from past to present. The model predicts that annual pollutant loads at this location will increase another 30% over current loads

Ecological and Hydrological Function

Ecological function describes the normal properties and processes that an ecosystem exhibits over the course of time. Since the climate is not constant, but varies from day to day and season to season, ecosystem processes have to be thought of as average conditions. Sometimes ecosystem functions are called ecosystem services because often they are not "things" so much as they are processes that might be exploited. Probably the single most important ecosystem processes is primary production, since all other properties and processes occur as a result of the fact that an ecosystem is a living system that provides services as part of its everyday processing of energy and materials.

Watersheds like the St. Marks are a collection of ecosystems and human dominated systems. Ecosystem function of a watershed is the collection of properties and processes of the ecosystems of the watershed; for instance, a basin's ability to cycle nutrients, store water, provide erosion protection, and provide wildlife habitat. We might say that ecological function is 100% when fully natural (having no urbanization or agriculture uses) and potentially decreases as urbanization and agricultural uses increase.

Hydrologic function of a watershed is the sum total of properties and processes of water moving within a watershed. It is such things as storage and timing of water flows, watershed nutrients dynamics, and the erosion and sedimentation cycle driven by water. Again, we might say that hydrologic function is 100% when fully natural and potentially decreases as the watershed is urbanized, or converted to agricultural uses.

Measuring ecological or hydrologic function is no easy matter. The loss of ecological function might be measured by computing Indices of Ecological Health for all ecosystems within a basin, or by simply measuring the land areas of natural undisturbed lands verses urban and agricultural lands and equating decreases in indices or natural land area as loss of ecological function. Loss of hydrologic function might be measured by changes in stream hydrographs that

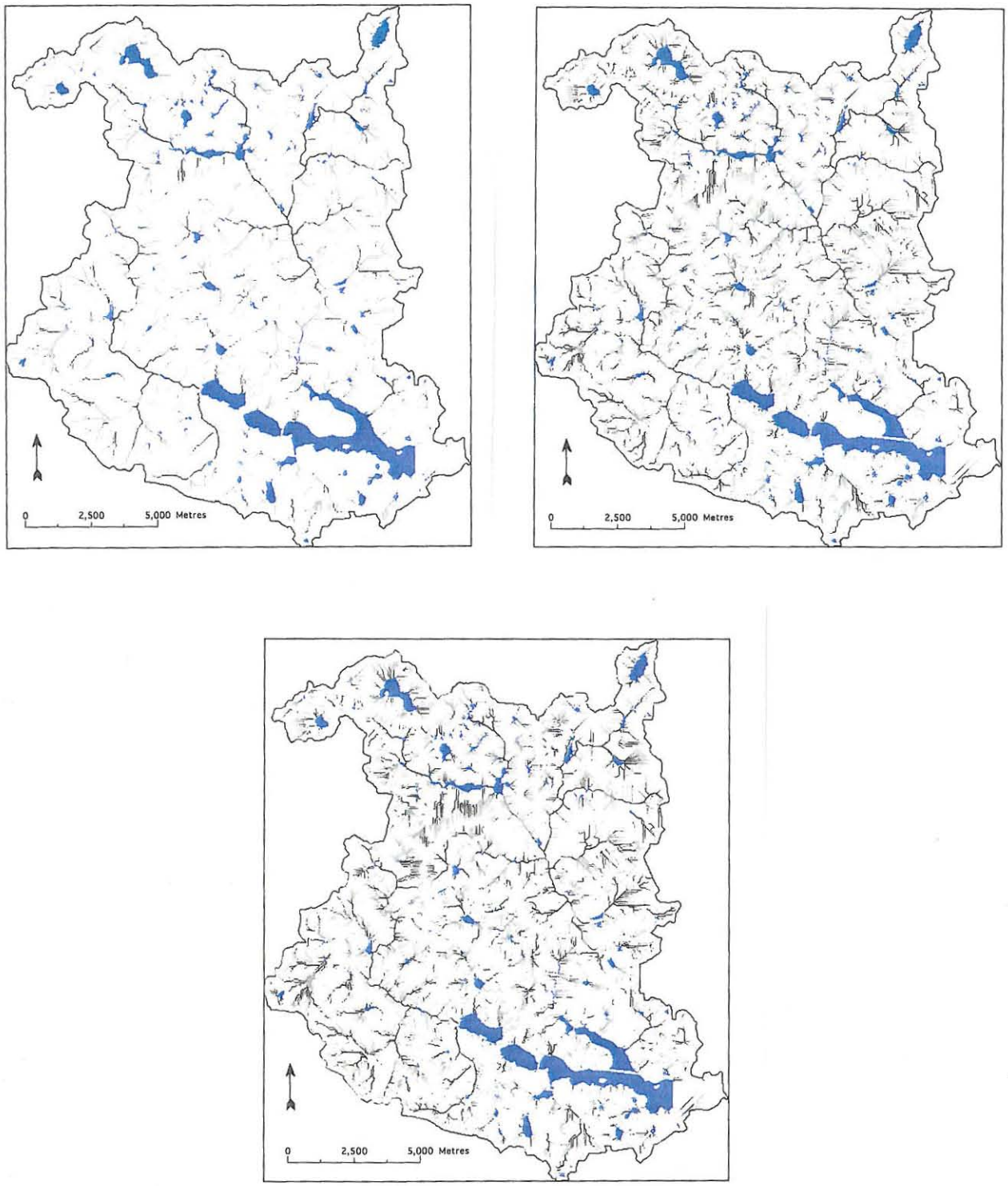


Figure 2-15. Maps depicting annual pollutant load for past condition (top left), present condition (top right), and future condition (bottom) in the Lake Lafayette Basin generated using the Drain Model. Darker areas indicate highest total pollutant load, while the lighter areas indicate lowest overall pollutant load.

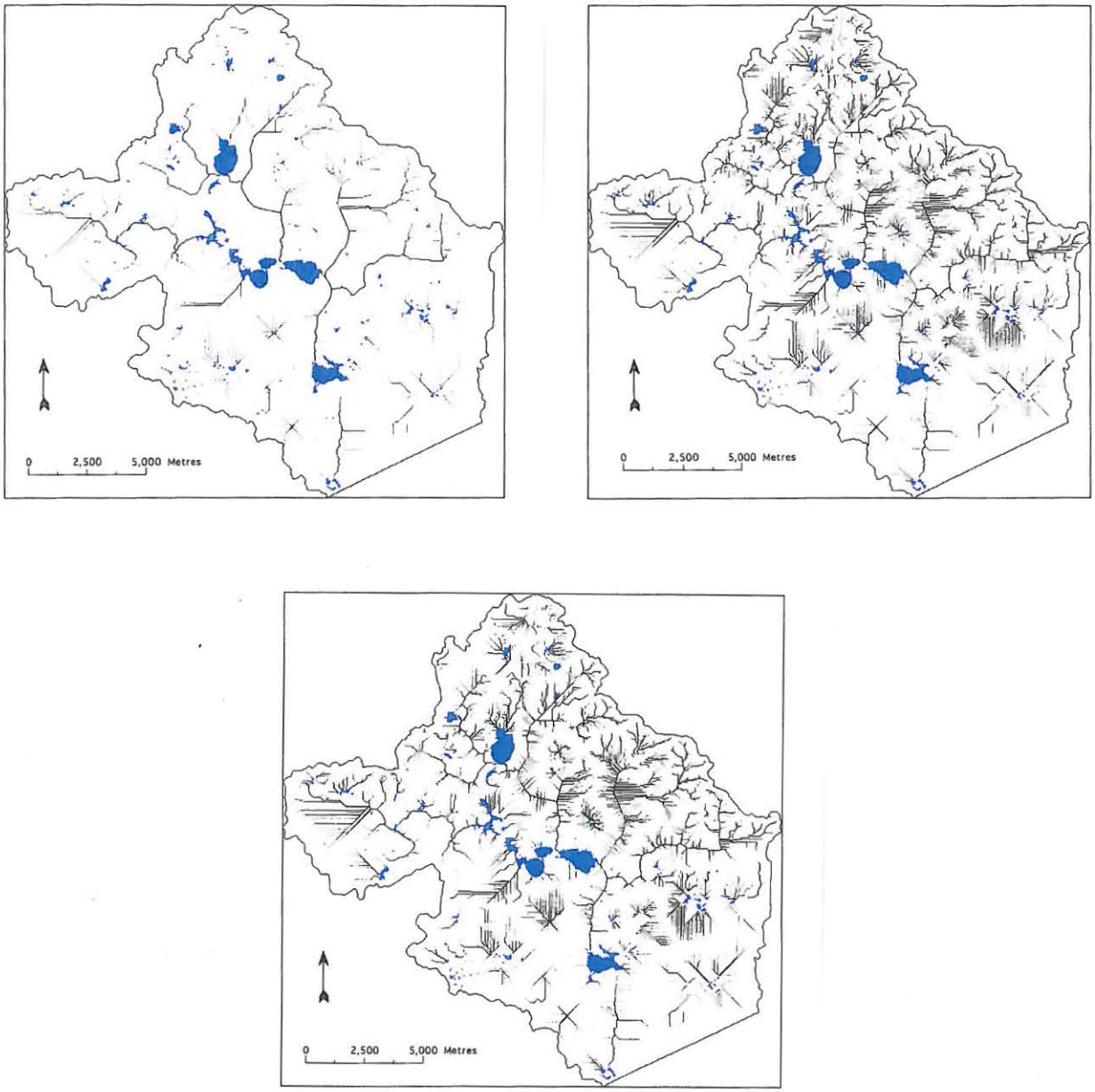


Figure 2-16. Maps depicting annual pollutant load for past condition (top left), present condition (top right), and future condition (bottom) in the Lake Munson Basin generated using the Drain Model. Darker areas indicate highest total pollutant load, while the lighter areas indicate lowest overall pollutant load.

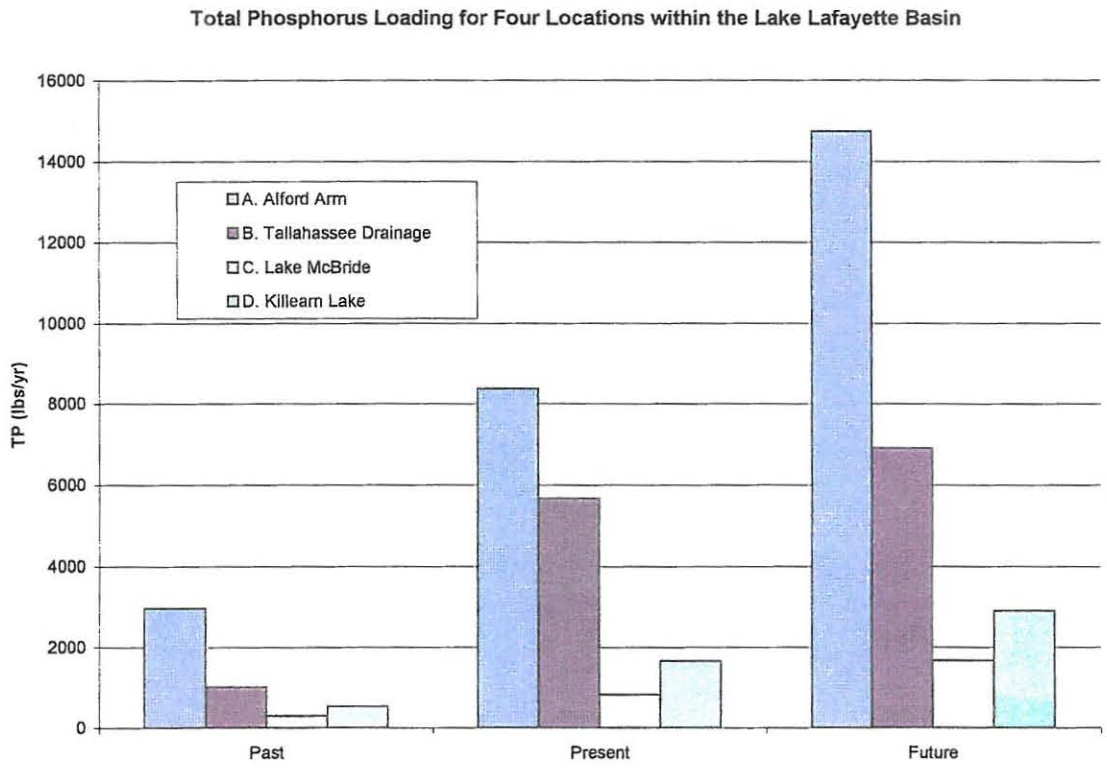


Figure 2-17. Graphs of the annual pollutant loads in Lake Lafayette Basin for the three time periods at four locations: A) lower reach of Alfrod Arm Branch, B) Tallahassee Drainage inflow to Upper Lake Lafayette, C) Lake McBride, D) Killearn Lakes.

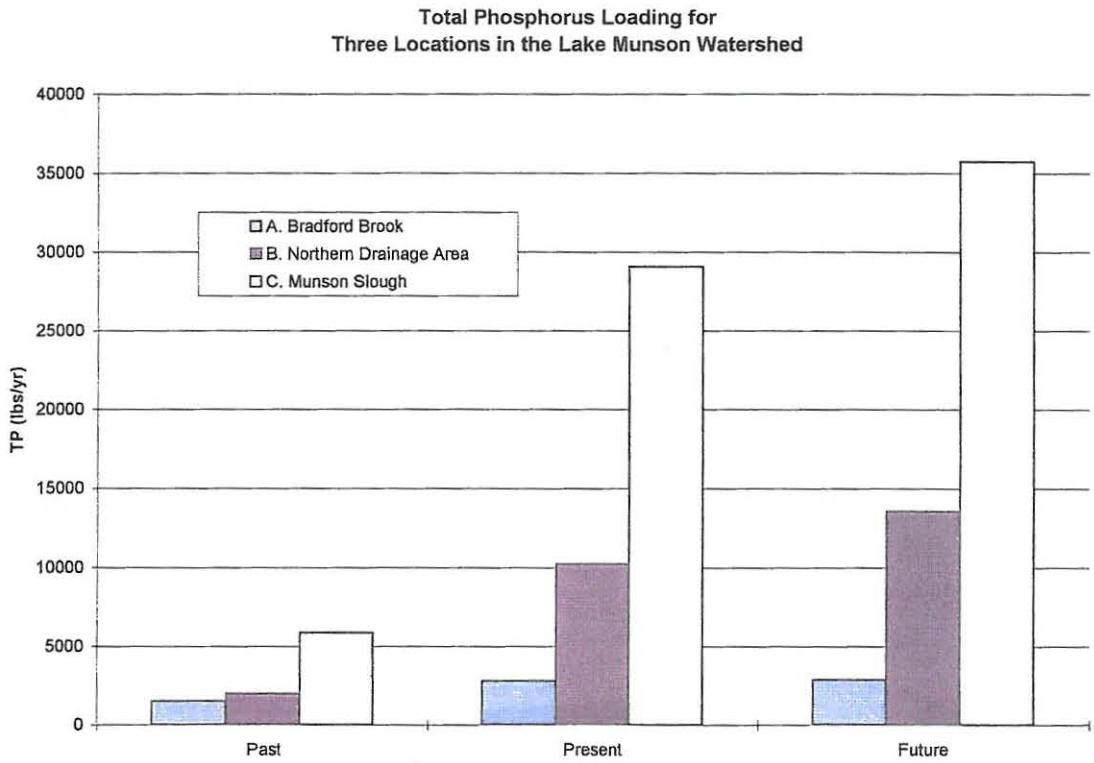


Figure 2-18. Graphs of the annual pollutant load for the three time periods at 3 locations in the Lake Munson Basin: A) Bradford Brook inflow to Bradford Lake, B) Northern Drainage Area (at the confluence of Gun Creek and West Drainage Ditch), and C) inflow to Lake Munson.

indicate loss of water storage capacity, changes in erosion and sedimentation rates, or changes in pollutant loads. Thus loss of function is relative; measured against some given condition.

Stream flow and the materials that it carries can be a good indicator of both hydrologic and ecological function. If a watershed is ecologically healthy, and hydrologically healthy, waters should flow without destructive quantities and timing and should contain normal amounts of sediments and nutrients. Changes in ecological function or hydrologic function should show up in the quantities and timing of water and in the constituents carried by water. So we might speak of the sum total of ecological and hydrological function as "Basin Function."

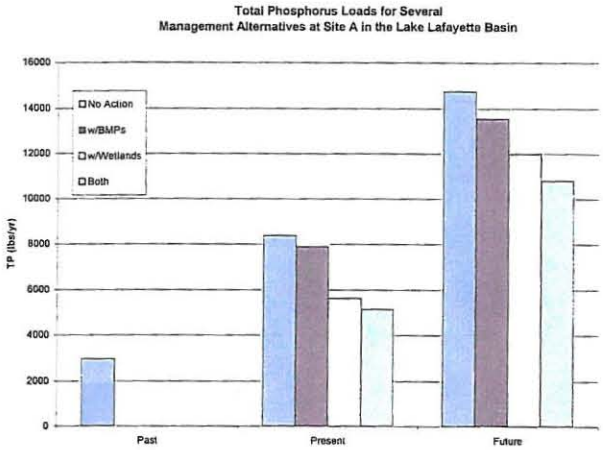
Comparing pollutant loading in the present condition with that of the past as in Figures 2-13 and 2-14 and Figures 2-17 and 2-18 provides a metric against which we might judge loss of basin function.. Potential losses can be measured by comparing past with future conditions assuming build out. The loss of basin function is shown as the increase in pollutant loading. Basins where present pollutant loadings are less than 50% greater than historic loads, while having lost some function are still functioning at reasonable levels (Roberts Pond Sub-basin in Lafayette Basin and Bradford Brook Sub-basin in Munson Basin). The remaining basins have greater than 100% change between historic loads and present loads; some as much as 450% increases. These basins have lost most hydrologic functions and will require serious efforts to ecologically engineer them to approach historic levels. In the following section, we suggest several alternatives for stormwater management that could increase basin hydrologic function. We use the Drain Model to test the effectiveness of these management alternatives.

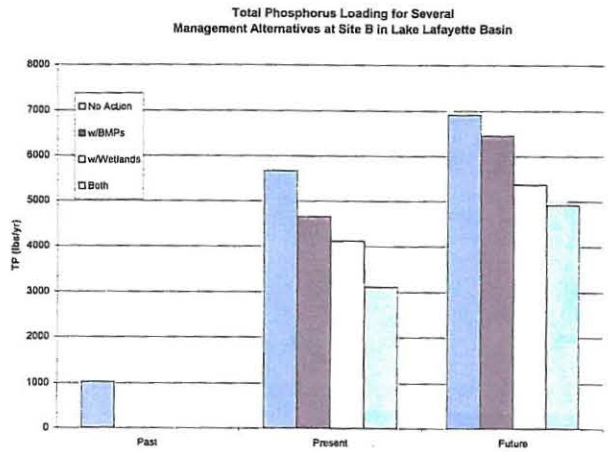
Stormwater Management Alternatives

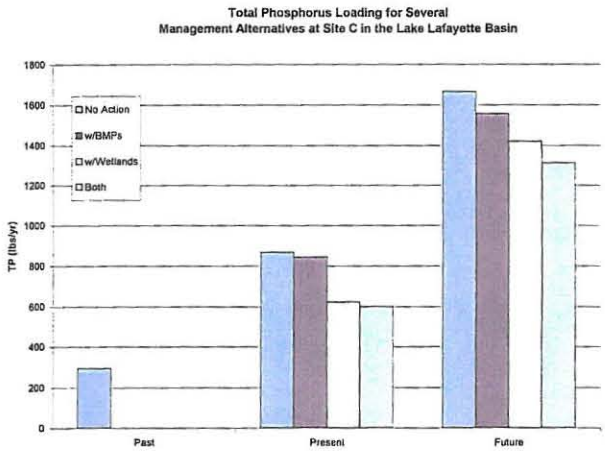
Water flows down hill. This simple truth is the basis for most stormwater management systems. Until very recently, stormwater management consisted of collecting and efficiently conducting stormwater runoff to the lowest point within a watershed. The lowest point often was a stream, river, or lake. The concern of stormwater managers was to get water off the land to minimize flooding. In the early days, when developed areas were still small and undeveloped areas large, this technique of stormwater management worked. Stormwaters were quickly and efficiently routed off developed lands through constructed ditches and deposited in downstream water bodies. However, as the amount of impervious surface within watersheds increased, the amount of runoff greatly increased and the pollutant load it carried increased as well. Receiving waterbodies began to show signs of stress; lakes became eutrophic, rivers exhibited fish kills and extreme sediment erosion and deposition, and ultimately coastal estuaries showed signs of toxic waste buildup and bacterial contamination.

In some respects, stormwater has become over managed. That is to say that through efficient removal of ever increasing volumes of stormwater carrying increasing pollutant loads, down stream waterbodies have suffered. The stormwater was managed, but the system was not. The time has come to revisit the concept of stormwater management, rethinking its goals, and redesigning stormwater management systems. Much work has already been done in developing and administering stormwater management regulations that have begun to reverse the trends of declining stormwater quality. New developments are required to retain and treat stormwaters, thus reducing quantity of runoff and improving quality. However, there is much stormwater runoff that does not benefit from these new regulations because it flows from areas that were developed prior to adoption of stormwater regulations. It is these older areas that often present the most difficult stormwater problems to solve.

The graphs in Figures 2-19 and 2-20 show changes in pollutant loading at several locations in the Lafayette and Munson basins and the effects of three management alternatives. In the first alternative, BMP's are used to reduce pollutant loads by 10% in residential areas and 20% in commercial areas. In the second alternative, wetlands are reconstructed throughout the watersheds to replace wetlands lost over the years to development. And the third alternative is to combine BMP's with the wetlands alternative. In almost every case, BMP's have minor effect on reducing pollutant load. On the other hand re-establishing wetlands throughout these basins can have a significant impact on water quality, reducing it, in some cases, to near historic levels. We recognize the radical nature of this proposal, and suggest it more in a comparative mode. It







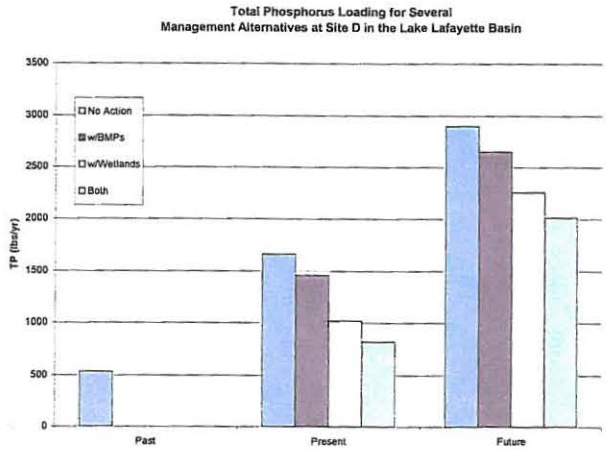
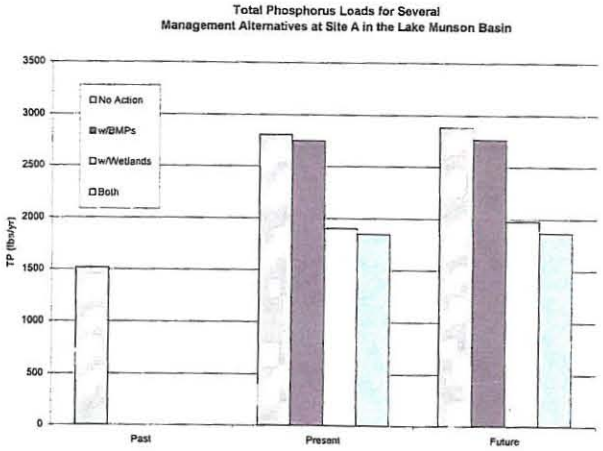
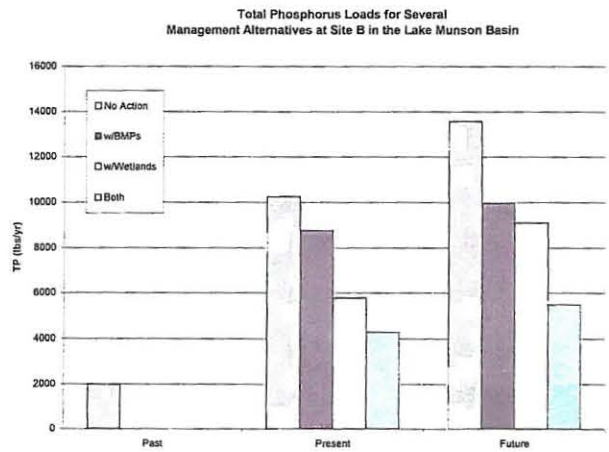


Figure 2-19. Graphs of the changes in pollutant loading at several locations in the Lake Lafayette Basin and the effects of three management alternatives: 1) the use of Best Management Practices (BMP's) that reduce pollutant loads by 10% in residential areas and 20% in commercial areas, 2). wetlands reconstruction throughout the watersheds to replace wetlands lost over the years to development, and 3) combination of BMP's with the wetlands alternative.





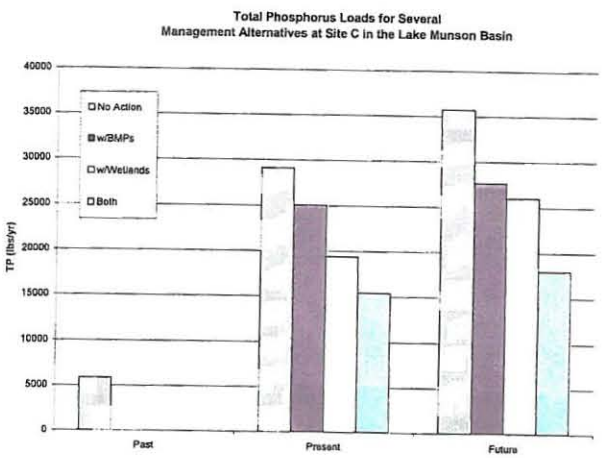


Figure 2-20. Graphs of the changes in pollutant loading at several locations in the Lake Munson Basin and the effects of three management alternatives: 1) the use of Best Management Practices (BMP's) that reduce pollutant loads by 10% in residential areas and 20% in commercial areas, 2). wetlands reconstruction throughout the watersheds to replace wetlands lost over the years to development, and 3) combination of BMP's with the wetlands alternative.

essentially says, that with more sensitive development, protecting wetlands and storing water within watersheds instead of diverting it downstream, significant losses of basin function can be avoided.

Figure 19 shows graphs of annual pollutant load at Sites A through D in the Lafayette basin (Figure 15). The historic pollutant load at each location is given first in each graph, and then loads for the present and future conditions are given. The first bar in each time period is the load predicted by the Drain Model with no management alternatives. The second and third bars result from instituting BMP's and the wetlands reconstruction alternatives respectively. The fourth bar represents total load when both BMP's and wetlands are combined as a management alternative. BMP's while an effective means of reducing some loads, appears to reduce loads by about 10 - 15% in urbanized basins. The largest increase in water quality resulted form the wetlands reconstruction alternative, lowering pollutant loads in some cases by more than 50%. The combined approach , provides additional improvement in reducing total load.

Pollutant loads at sites A through C in the Munson Basin (Figure 16) are given in Figure 20. The largest improvement in total load is achieved at Site B (the northern drainage area) where reconstruction of historic wetlands provided a 40% decrease in pollutant load. When the combined management alternative was tested, the improvement was almost 60%. This is a moderately developed basin that had relatively large historic areas of wetlands

Its important to note that these management alternatives do not lower pollutant loads to historic levels, but certainly should provide significant increases in water quality. We are quite aware that the reconstruction of historic wetlands is extremely difficult and probably highly unlikely, given the present amount of urban infra-structure throughout these basins. However, the alternative shows the impacts of their loss on basin hydrologic function, and the difference that wetland sensitive development could make today and in the future. The same level of treatment might be attained through the construction of wetland stormwater treatment areas within basins in locations where it is possible to do so, although finding suitable vacant property in basins that are nearly completely developed may be difficult at best. In these situations, plans now for the purchase of sites in future years that are suitable because of under utilization or pollutant status (Brown Fields) may be extremely desirable. In all, it is apparent that BMP's alone will not provide the level of treatment necessary to improve today's nonpoint source pollution problems in urbanizing areas, leaving the future in serious doubt. Even effective programs of ecological engineering, may find it difficult to provide the level of treatment necessary to improve the health of water bodies in urban landscapes.

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Chapter 3 :
Spatial Modeling of Nitrogen Loading to a Surficial Aquifer from Residential On-site Sewage Disposal Systems in Wakulla County, Florida.

Alan Foley

INTRODUCTION

This study developed a spatial modeling technique within a GIS environment to predict the fate of nitrogen loading from septic tanks to the surficial aquifer in Wakulla County (Figure 1).. The study focused on five hydrologic sub-basins surrounding Wakulla Springs State Park and the Wakulla River. This area is rapidly urbanizing and has a high potential for groundwater degradation. The model was used to evaslaute nitrogen loading that resulted from present development patterns and furture development patterns based on the Wakulla County Future Land Use Plan and assuming buildout to zoned densities (Figure 2)

Background

The estimated population of Wakulla County in 1985 was 13,159 people. The projected county population in 2005 is 20,000 people (Ayres and Associates, 1987). This population increase will proportionally increase potable water and wastewater demands.

The majority of residences in Wakulla County are un-sewered and therefore utilize on-site sewage disposal systems (OSDS's) for wastewater treatment. It is estimated that OSDS's provide about 70% of Wakulla County's wastewater treatment, (Ayres and Associates, 1987). OSDS's dispose of wastewater using a variety of components and configurations, the most common being the septic tank/soil absorption system (STSA), (Kirkner and Associates, 1987).

Physical, chemical, and biological processes in septic tanks and soil provide treatment of wastewater. Soil adsorbtion of chemicals in septic tank effluent is probably the major contaminant removal mechanism in the subsurface environment (Wilhelm et al., 1994, Canter and Knox, 1984). Adequate contaminant removal may not be possible in some areas of Wakulla County, (FDEP, 1979). STSA's have a direct influence on water quality in the surficial aquifer and may influence water quality in the deeper Floridan Aquifer – drinking water source for most of Wakulla County (Lowrance and Pionke, 1989, Kirkner and Assoc., 1987).

Increased STSA density has been related to groundwater degradation, (USGS, 1995, Yates, 1989). STSA induced groundwater degradation can be characterized by high nitrate (NO_3^-) and bacteria concentrations in addition to potentially significant amounts of organic contaminants. Septic tank failure is commonly due to exceedance of soil effluent absorption capacity. Soils with high permeability – such as those in the Woodville Karst Plain – can be rapidly overloaded with organic and inorganic chemicals and Canter and Knox, 1984). Current STSA design parameters and regulations may not provide for adequate protection of groundwater quality (Bicki, et al., 1984). Elevated NO_3^- concentrations have been detected at Wakulla Springs State Park and in some private wells (FDH, 1998, Hand, 1998).

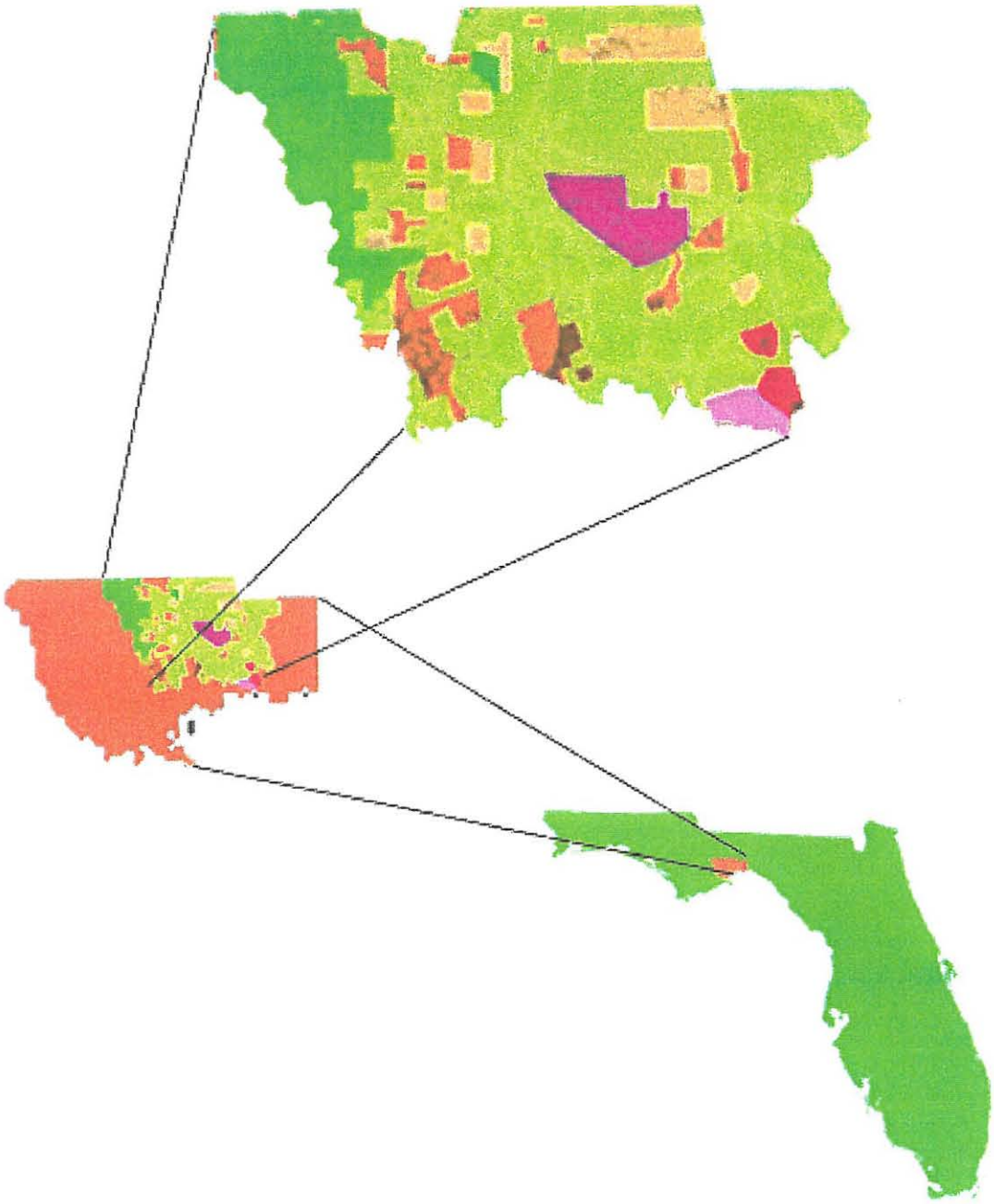
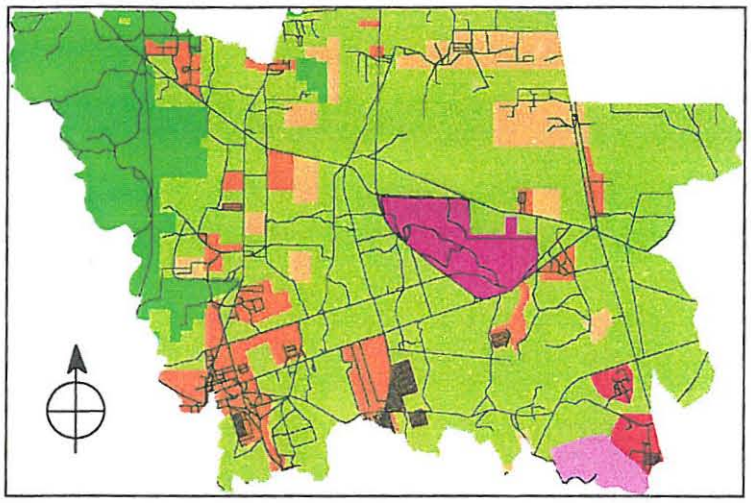


Figure 1. Location map of the Wakulla County Study Area. Shown are: (1) a portion of the Apalachcola National Forest (the dark green are in the western portion of the study area), (2) Wakulla Springs State Park (purple area in the center of the study area), (3) St Marks National Wildlife Area (lighter purple area in the southern tip of the study area), and (4) present zoneing classes (see Figure 2 for details).

Present Land Use



- Incorporated City
- Industrial
- Urban 1, 2 DU/ ac.
- Rural 2, 1 DU / 2 ac.
- Rural 1, 1 DU / 5 ac.
- Zoned Agriculture / Silviculture
- Wakulla Springs State Park
- Apalachicola National Forest
- St. Marks National Wildlife
- Roads

Future Land Use

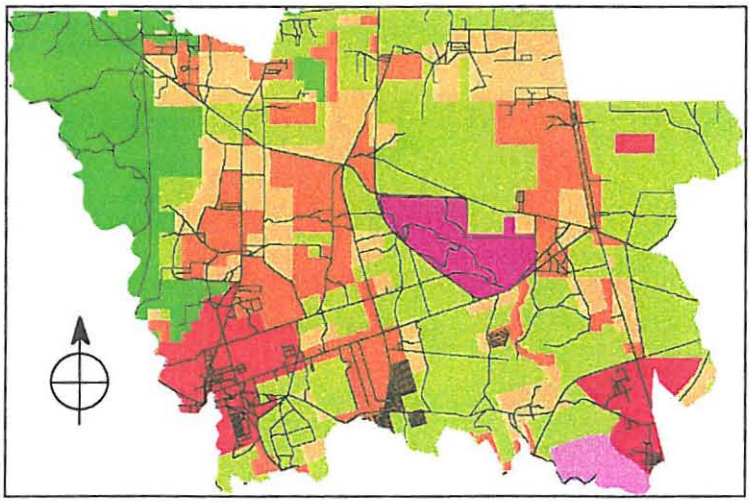


Figure 2. Present and future land use in the study area.

Throughout much of the populated areas of the county there is no confining unit overlying the Floridan aquifer – the primary source of drinking water in the county, and in much of northern Florida. The eastern half of Wakulla County is located on the Woodville Karst Plain – a groundwater discharge area with characteristic karst features such as springs and sinkholes. The limestone aquifer in this area is covered by a thin layer of fine, unconsolidated quartz sand and clay that generally is less than 20 feet thick (SCS, 1991).

Water in the sand and clay layer overlying the St. Marks formation is free to rise and fall and is referred to as the surficial aquifer. Ground water recharge in this area is derived mostly from precipitation. The Wakulla County Soil Survey describes nearly all of the soil types present as severely limited in regard to the placement of septic tank adsorption fields (FDEP, 1979, SCS, 1991). As the county continues to build out at rural densities, management agencies are challenged to accommodate growth while preserving the water resources and environmental features in Wakulla County.

The tight integration between hydrology, geology, and biology requires a multidisciplinary management approach where geographic information systems (GIS) may provide a means to work across disciplinary lines in problem analysis. The compilation and standardization of digital geographically referenced data sets enables spatial modeling which may aid in land use decisions and efficient allocation of management resources.

Review of the Literature

Regulation of nitrogen in drinking water

The US Environmental Protection Agency has set drinking water maximum contamination levels (MCL's) for two inorganic, nitrogen-containing compounds, nitrite (NO_2^-), and nitrate (NO_3^-). The current MCL for NO_2^- is 1 mg/L (as nitrogen). The current MCL for NO_3^- is 10 mg/L (as nitrogen). A third inorganic compound, ammonia (NH_4^+) , is on the Drinking Water Priority List (DWPL). The DWPL contains substances that are known or anticipated to occur in public water systems and that may require regulation under the Safe Water Drinking Act, (AWWA, 1992).

NO_2^- may combine with amines in the stomach to form potentially carcinogenic nitrosamines. High NO_3^- levels in drinking water can cause methemoglobinemia (Blue baby syndrome). Nitrate is converted to nitrite (NO_2^-) in the intestines. The NO_2^- combines with hemoglobin in the blood to form methemoglobin. Methemoglobin does not transport oxygen in the blood. High levels of methemoglobin can cause death due to asphyxiation. Babies and cattle are especially susceptible to this condition. The current drinking water limit for NO_3^- is a concentration of 10 mg/L.

Nitrogen in the environment

Nitrogen containing compounds can be introduced to the environment as solid, liquid, or gaseous forms. USGS chemical analyses of atmospheric precipitation yielded low nitrate concentrations relative to concentrations in groundwater, (USGS, 1996a). Atmospheric deposition of nitrogen was not considered in this analysis.

Agricultural practices such as crop fertilization and dairy farming can introduce large quantities of nitrogen into the environment. Approximately 2 percent of the land area in Wakulla

County is used for agriculture, predominantly for the production of row crops and as pasture, (SCS, 1991).

Treated wastewater from domestic and commercial OSDS's is discharged directly to the subsurface environment. OSDS effluent is the most frequently reported source of groundwater contamination, (Miller,1980).

Groundwater monitoring studies and laboratory column studies indicate that approximately 20 –40% of nitrogen in STSA effluent may be adsorbed or otherwise removed before the effluent reaches the groundwater, (Bicki, et al, 1984).

Nitrogen chemistry in soil - Several chemical processes govern nitrogen transportation and distribution in the subsurface environment. These processes convert nitrogen from organic to inorganic forms and include: ammonification, volatilization, nitrification, denitrification, dissimilatory nitrate reduction to ammonium, adsorption, and biological uptake, (Bicki et al, 1984). Denitrification is the only process that may serve as a major nitrogen sink, the other processes temporarily immobilize nitrogen, (Korom, 1992).

Ammonification - Ammonification (mineralization) is the conversion of organic nitrogen compounds such as proteins and amino acids to inorganic compounds such as ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-). Ammonification is performed by a wide variety of organisms (bacteria, actinomcetes, and fungi) and follows several reaction pathways. Ammonification typically refers to complete mineralization - when all organic nitrogen is converted to NH_4^+ (Bitton, 1994; Bartholomew, 1965).

Volatilization - NH_4^+ is the predominant reduced nitrogen form in acidic and neutral pH aquatic environments. As pH increases, NH_3 becomes more abundant and is released as a gas. Volatilization does not remove significant amounts of nitrogen from OSDS effluent due to the fairly neutral range of effluent pH (7.0 – 8.5). In addition, subsurface disposal systems do not provide adequate air-water contact for appreciable amounts of NH_3 to volatilize (Bicki et al, 1984). When conditions are favorable, NH_4^+ is oxidized to NO_2^- or NO_3^- before its assimilation by microbes (Bartholomew, 1965).

Nitrification - Nitrification is the oxidation of NH_4^+ to NO_3^-. Nitrification occurs in the temperature range of 15-35° C and is favored in wastewater effluents with a low biological oxygen demand (BOD) and a high NH_4^+ content, (Bitton, 1994). Nitrifying bacteria require an aerobic environment (oxygen concentration greater than 1.0 mg/L), and sufficient alkalinity to neutralize the hydronium ions produced during the oxidation of NH_4^+ (pH of 7.2-8.4 is optimal for nitrification). Acidic soil conditions will prevent nitrification, (Bartholomew, 1965). High nitrification rates may adversely affect denitrification by lowering pH, (Reneau et al., 1989).

The two-step nitrification reaction is performed by two groups of nitrifying bacteria. The Nitrosomonas group of bacteria converts NH_4^+ to NO_2^-. The Nitrobacter bacteria group then transforms the NO_2^- to NO_3^-, (Bicki et al., 1984; Bitton, 1994). NO_3^- is very soluble and chemically inactive under aerobic conditions. Once in the groundwater NO_3^- is highly mobile. Reduction of NO_3^- concentrations in an aquifer will occur due to dilution or denitrification (Bicki, et al., 1984).

Denitrification - The denitrification process reduces NO_3^- to gaseous N_2O or N_2. These gases will diffuse to the atmosphere, resulting in a loss of nitrogen from the subsurface system. Denitrification requires nitrogen oxides (NO_3^-, NO_2^-, NO, and N_2O), bacteria with the metabolic capacity to perform denitrification, anaerobic conditions, and suitable electron donors, (Korom, 1992). Chemical denitrification in shallow groundwater due to the oxidation of ferrous iron may occur, (Lowrance and Pionke, 1989).

Denitrification is a two step process; NO_3^- is converted to NO_2^-, which is then converted to N_2O or N_2, (Bitton, 1994). Nitrogen must be in an oxidized form for denitrification to take place. Nitrification provides the oxidized nitrogen compounds for denitrification. Aerobic nitrification must precede anaerobic denitrification in the soil.

The bacteria that carry out the denitrification process are facultative anaerobes – organisms capable of survival with or without O_2. Necessary anaerobic environments are present under saturated conditions. Organic matter (OM) is the most commonly used electron donor. Bacteria use O_2 to oxidize the OM until oxygen supplies are depleted. Facultative anaerobes then switch to using NO_3^- as an oxidizing agent (electron acceptor). An increase in O_2 levels will cause the bacteria to return to using O_2 instead of NO_3^- to oxidize OM, (Korom, 1992).

Denitrification rates depend on the types of bacteria present locally and whether they are actively denitrifying. Local denitrification rates need to be determined on an individual basis, (Korom, 1992). Consistent nitrification-denitrification processes are rare in a subsurface system due to the variability of the water table.

DNRA - Dissimilatory nitrate reduction to ammonium (DNRA), like denitrification, also occurs under anaerobic conditions. DNRA tends to conserve nitrogen within the soil system, while denitrification tends to export nitrogen from the soil system. DNRA is favored when NO_3^- supplies are limiting. Denitrification is favored when OM (electron donor) supplies are limiting.

Though the inorganic NH_4^+ ion is not water-soluble under aerobic conditions, it may later undergo nitrification to produce NO_3^-. Saturated soil conditions may create an anaerobic environment in which NH_4^+ can be leached to the ground water, (Korom, 1992). Anaerobic and aerobic conditions may alternate at a location due to water table fluctuations.

Adsorbtion - Soil adsorbtion of NH_4^+ can be significant under certain conditions. The factors influencing NH_4^+ adsorption are: the number of cation exchange sites in the soil, the affinity of those sites for NH_4^+, site saturation with NH_4^+, and the composition of the effluent. Other cations present in the effluent may outcompete NH_4^+ for exchange sites, (Bicki et al, 1984). Soils with a higher content of clay tend to adsorb more NH_4^+. Studies referenced in Bicki et al, 1984, found that NH_4^+ adsorbtion ranged from 2 mg/100 grams of sandy soil to 100 mg/100 grams of fine-textured soil with 30% clay content.

Saturation of cation exchange sites in the soil underlying an OSDS may occur. Cations in the OSDS effluent would no longer be adsorbed and would pass through the septic field and into the groundwater. Adsorbed NH_4^+ can be desorbed and nitrified if reaeration occurs due to water table fluctuation, (Bicki et al, 1984).

Biological uptake - Biological uptake of nitrogen by microbes and plants results in nitrogen immobilization for the life of the assimilating organism. The low carbon/nitrogen ratio of OSDS effluent limits microbial uptake. The amount of nitrogen generated by OSDS's typically exceeds

the capability of local plant uptake, (Bicki et al, 1984). Much of the nitrogen may not be available for plant use if hydraulic loadings cause the highly soluble NO_3^- to be leached below plant roots, (EPA, 1977).

Septic tanks operation

Septic tank effluent carries a variety of organic and inorganic chemicals into the subsurface environment. Chemical components are retained in the soil, assimilated biologically or transported by groundwater movement. Satisfactory effluent treatment depends on numerous factors including:

- characteristics of the wastewater
- design of system components
- construction techniques
- rate of hydraulic loading
- age of system
- periodic maintenance
- climate
- geology
- hydrology
- topography
- morphological, chemical, and physical properties of the soil

STSA's can provide varying levels of wastewater treatment. Proper STSA functioning is achieved only if a sufficient volume of aerobic, unsaturated soil is available to absorb the volume of effluent. It should be noted that a "properly" functioning septic system discharges significant quantities of NO_3^- into the subsurface environment, (Bicki et al., 1984, Wilhelm et al., 1994).

Most STSA constructions provide two zones where reduction/oxidation (redox) reactions occur. The first redox zone is the anaerobic septic tank. Most of the nitrogen (75-85%, Bicki et al., 1984) is released from the septic tank in the reduced NH_4^+ form. Septic tank retention times average about three days. Effluent is distributed to the drainfield with a typical areal loading of one to five cm/day.

The second redox zone is the aerobic drainfield. The typical O_2 demand exerted by wastewater ranges from 400 to 1500 mg/l. NH_4^+ oxidation in the drainfield results in NO_3^- concentrations roughly two to seven times the drinking water limit (10 mg/l as nitrogen equivalent to 45 mg/l as NO_3^-). Almost complete oxidation of NH_4^+ can occur within 1m of the distribution pipes and within a few hours exposure to oxygen (Bicki et al, 1984, Walker et al., 1973). The predicted nitrogen specie under drainfields are: nitrate in sandy soils, a mixture of nitrate and ammonium in silt loams, and ammonium in clays, (Bicki et al., 1984).

The largest changes in wastewater composition occur in the drainfield. As NH_4^+ is oxidized to NO_3^-, organic carbon in wastewater is oxidized to CO_2. CO_2 production can lower pH if the CO_2 remains in solution. A lowered pH may cause significant dissolution of calcium carbonate ($CaCO_3$) over an extended period, (Wilhelm et al., 1994).

If the native sediments in the drainfield are finer than the gravel surrounding the distribution pipes, a two to five cm thick mat will form beneath the gravel. This "biomat" is generally moist and has a high O_2 demand, resulting in local anaerobic conditions. Biomat

formation will alter the hydraulic conductivity in the drainfield. Drainfield specifications account for the permeability of the natural soil, but not for the permeability of the biomat.

Some STSA configurations provide a third redox zone in which denitrification occurs under anaerobic conditions. Most sites do not have the geology necessary for a third redox zone. Denitrification requires an approximate one to one ratio between organic carbon and NO_3^-. Organic carbon in the wastewater will have oxidized by this point. Long term denitrification requires significant amounts of available organic carbon, (Wilhelm et al., 1994).

STSA hydraulic loading and precipitation recharge will cause downward movement of any mobile constituents. NO_3^- contamination tends to remain as a distinct plume due to the typically laminar flow of groundwater, (Ayres and Associates, 1993). Denitrification in the saturated zone has not been found to be significant. It is thought that the organic carbon present is recalcitrant and therefore unavailable for microbial utilization.

Limited denitrification and low dispersivity in the saturated zone can result in locally elevated concentrations of NO_3^-. Harman et al. (1996) delineated 110 m of a septic tank plume in an unconfined sand aquifer. NO_3^- concentrations exceeded drinking water standards over the entire 110 m. Postma et al. (1992) detected average concentrations in the 20 – 50 mg NO_3^- /l range, with a peak concentration of 115 mg NO_3^- /l 2 m downgradient from a drainfield receiving effluent with 100 mg NO_3^- /l. In a laboratory study Willman et al. (1981) found that 77-92% of an initial 178 mg NH_4^+/l was converted to NO_3^- in 60 cm soil columns containing limestone/shale sand. Limestone columns with 0, 3, 6, and 12% clay reached a steady-state concentration near 150 mg NO_3^-/l (Willman et al., 1981). It is estimated that 20-40% of the nitrogen is removed from the effluent as it percolates through the soil (Bicki et al., 1984, Ayres and Associates, 1993).

Septic tank regulation

To prevent groundwater contamination, current regulations (FL Administrative Code Chap. 10D-6, 1985) specify a 24" separation distance between the bottom of the adsorbtion system and the seasonal high groundwater. A 24" separation distance may be inadequate. In-situ GW monitoring studies and lab column studies support a minimum depth of 36". STSA's may impact more severely in some locations than in others due to differences in land use, soil-water-landscape relations, STSA density, and recharge capability of local aquifers, (Bicki, et al, 1984).

Assimilative capacity of the environment

Environmental assimilation of STSA effluent is a function of hydrology, geology, and biology. The hydrologic and biologic components will exhibit some degree of seasonal variation. Water provides both NO_3^- transport and moisture to create saturated, anaerobic soil conditions. Water table fluctuations will cause alternating aerobic/anaerobic regimes. Denitrification may be enhanced or hindered depending on the frequency and amplitude of water table fluctuation. Greater groundwater velocities may enhance dilution of an NO_3^- plume.

Natural processes governed by the vegetative cover on the ground surface can affect contaminant concentrations in a phreatic aquifer system, such as the surficial aquifer in Wakulla County. The amount of rainfall recharge to groundwater will be affected by the plant evapotranspiration and by soil permeability. The quantity and quality of vegetative litter

deposited on the surface may affect soil characteristics such as permeability and chemical content.

Biologically, an overabundance of NO_3^- may cause eutrophication of water bodies and shifts in species composition. Microbes are typically carbon limited in the saturated zone. In some areas – such as organic riparian soils – microbes may be NO_3^- limited. Increases in NO_3^- loading may result in development of a microbial community with the ability to denitrify large amounts of NO_3^-. The presence of denitrifying bacteria does not necessarily indicate the occurrence of denitrification, (USGS, 1994).

The geologic composition of the subsurface will determine the assimilative capacity of the soils. Soils with higher clay content will have a larger cation exchange capacity (CEC) and will sequester a greater amount of NH_4^+ than more sandy soils. Willman et al. (1981) found that laboratory columns of limestone sand retained NH_4^+ for four to nine weeks. CEC can eventually be exceeded and the soil will no longer retard movement of NH_4^+, (Canter, 1996).

Wakulla County is in the Gulf Coastal Lowlands physiographic province. The county can be further subdivided into the Woodville Karst Plain and the Apalachicola Coastal Lowlands. The study area for this project is located in the Woodville Karst Plain. The sediments at the surface and near the surface are made up of quartz sand that is not more than 7 m thick. The sands overlie a karstic, early Miocene limestone, (SCS, 1991). Soils in the Woodville Karst Plain do not appear to contain the necessary components for significant denitrification.

Ayres and Associates (1987) determined that 65% of the soils in Wakulla County had severe to very severe soil limitations for OSDS use. Only 5% had slight limitations. The soils within the five sub-basin study area could generally be considered to be severely limited for OSDS use, (SCS, 1991).

Modeling

Groundwater models can be generally classified in two categories – flow and solute transport models. Flow models are concerned with groundwater movement while solute transport models describe the movement of constituents within the water. These two general categories can be further subdivided as either analytical or numerical models.

Analytical models describe the water behavior in an aquifer with differential equations derived from basic principles such as the laws of continuity and conservation of energy (Canter and Knox, 1984). The application of analytical models in a heterogeneous environment is limited by their inability to account for parameter variability.

Numerical models approximate analytical solutions as a system of algebraic equations, or alternatively by simulating transport through the spread of a large number of moving reference particles (Domenico and Schwartz, 1997). Numerical approaches can accommodate parameter variation and thus allow modeling of more complex geometries in multiple dimensions.

Contaminant transport models usually begin with some form of mass transport equation:

mass inflow rate – mass outflow rate = change in mass storage with time

The mass transport equation is then modified to account for processes such as chemical reactions, biological uptake, and molecular and mechanical dispersion.

METHODS

Analytical model

One-dimensional mass transport involving a first-order kinetic reaction can be described as:

$$C/C_0 = (1/2) \exp \{(x/2\alpha_x)[1-(1+(4\lambda\alpha_x)/v)^{1/2}]\} \operatorname{erfc} [(x-vt(4\lambda\alpha_x)/v)^{1/2}/2(\alpha_x vt)^{1/2}] \quad (1)$$

where:

C	=	constituent concentration (mg/L);
C_0	=	constituent concentration at time $t_0 = 0$ (mg/L);
x	=	distance (m);
α_x	=	longitudinal dispersion coefficient (m);
λ	=	decay constant (1/day);
v	=	groundwater velocity (m/day);
t	=	time (day);

erfc is the complimentary error function.

The above equation approaches a steady state as the argument of erfc approaches negative two. Assuming a continuous source, the steady-state solution to the above equation is:

$$C/C_0 = \exp \{(x/2\alpha_x)[1-(1+(4\lambda\alpha_x)/v)^{1/2}]\} \quad (2)$$

Using data compiled by Gelhar et al (1992), an α_x of 10m was selected to characterize longitudinal dispersion in the surficial aquifer.

Spatial implementation

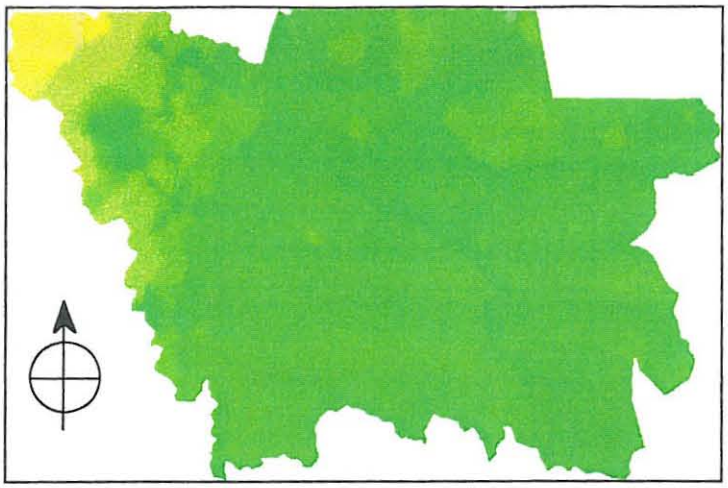
A raster-based geographic information system (GIS) was used to manipulate digital map layers to solve the above one-dimensional analytical equation over two-dimensional space. A cell resolution of 30m X 30m was used for all maps.

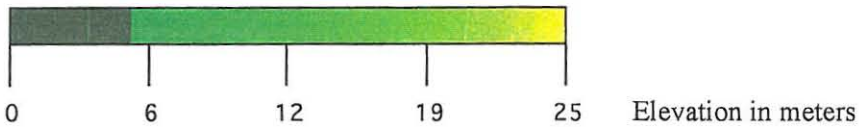
A coarse digital elevation model (DEM) was created by interpolation between 1:100,000 ft scale contours (See Figure 3). The interpolation results were modified to reflect an average north-south slope of 4 feet per mile (SCS, 1991).

The average slope within each cell of the DEM coverage was determined. It was assumed that the phreatic surface within the surficial aquifer is generally a subdued reflection of the land surface (USGS, 1991). Surface slope in this case can be interpreted as the change in hydraulic head with respect to distance – $\delta h/\delta x$.

A general soils map of the area was used to generate soil hydraulic conductivity and soil carbon content coverages (See Figure 3). Darcian velocities (q) within the surficial aquifer were determined using Darcy's Law:

Digital Elevation Map of the Study Area





Study Area Soil Characteristics

Soil Drainage and Type:

- Excessive to moderate - sandy soils
- Moderate to excessive - sandy soils
- Moderate - sandy soils
- Poor to moderate - sandy soils
- Poor - sandy soils
- Poor to very poor - sandy soils
- Very poor - organic soils
- Roads

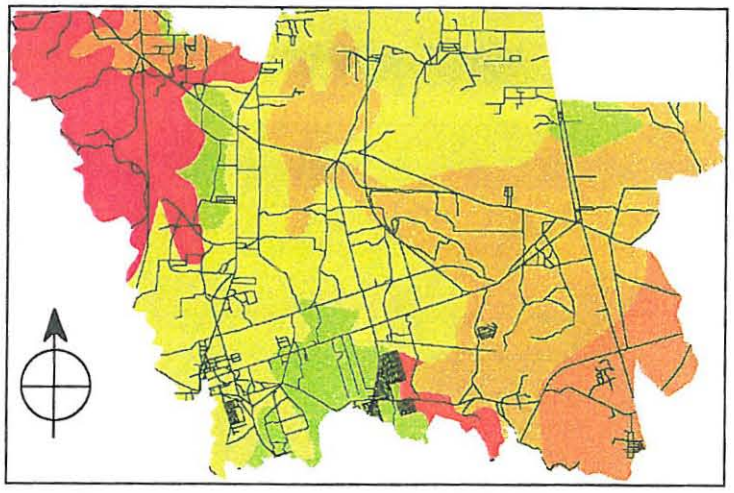


Figure 3. Digital Elevation Model (top map), and Soils coverage (bottom map) of the study area.

$$q = (-K/n)\ (\delta h/\delta x) \qquad (3)$$

where: K = hydraulic conductivity (m/day);
n = effective soil porosity (dimensionless ratio).

Assuming an average porosity of 0.3, Darcy's Law was solved spatially through algebraic manipulation of the polygon layers of hydraulic conductivities and surface slope map. The operation resulted in a map with each cell representing the Darcian velocity based on the pertinent parameters at the cell's location. This map was used as "v" in the analytical equation. Additional simulations were conducted with velocities double and half the initial "v" value.

Water surface slope (DEM) and hydraulic conductivity layers were used to generate flowpaths radiating away from the residential land use polygons. Flowpaths were generated from every cell on the perimeter of a residential land use polygon. The resulting flowpath coverage contained linearly distributed values increasing away from the land use polygons. Flowpath values were used as linear x coordinates along which the one-dimensional equation was applied. Flowpaths were immediately adjacent to each other, thus allowing a two-dimensional simulation with a one-dimensional equation.

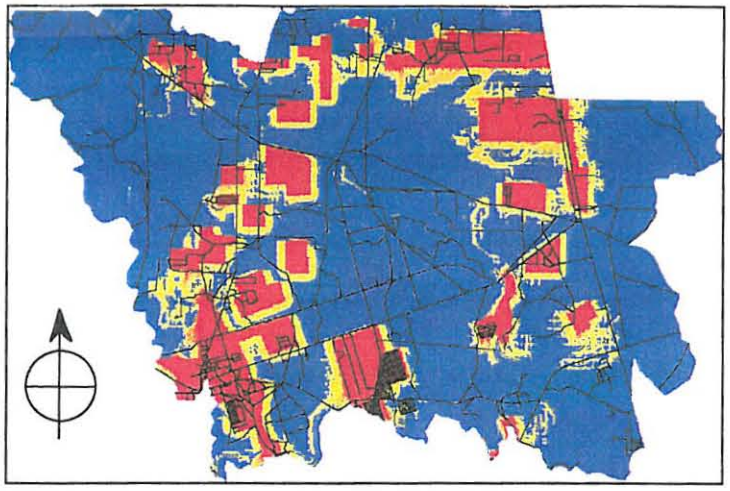
Research has shown that denitrification rates can be correlated with aoil organic carbon content, (Groffman and Tiedje, 1989, Tsai, 1989, Pratt et al., 1979, Yan, 1995). The soil carbon content layer was entered into a model of denitrification rates for sandy Florida soils (Tsai, 1989) to develop an initial estimate of potential denitrification rates for the soils in the study area. The potential denitrification rates were compared against an average initial NO_3^- concentration to develop a coverage representing λ in the analytical equation. Several simulations were conducted using a range of values for λ.

Accurate estimation of longitudinal dispersion is not possible without field studies. An average longitudinal dispersion coefficient of 10 m was selected based on data compiled by Gelhar et al. (1992). The land use polygons were simulated as an areal source (Mooseburner and Wood, 1980) with an initial NO_3^- concentration of 140 mg/l. Simulation under these conditions represents a steady-state scenario. It is important to note that the output maps depict <u>potential maximum</u> concentrations within each cell.

RESULTS

Figure 4 presents simulation results obtained using parameter values believed to be representative of conditions in the study area. The gradation of concentrations is not apparent at this map scale. Figure 5 shows details of NO_3^- attenuation in cells on the parameter of an area that is a source of N from septic tanks. The flow is right to left from the upper right hand corner. The majority of NO_3^- attenuation occurs within the first few cells surrounding the land use polygons.

Figure 6 is a bar graph showing the fraction of the study area having NO_3^- concentrations within given ranges. Six categories are presented in the graph legend – Present/Future – low/median/high. The low, median, and high terms refer to the flow rate used in obtaining the results. The median flow rate is the one believed to best characterize conditions in the study area. The low flow rate is half of the median and the high flow rate is double the median. Note



Potential Maximum NO3- Concentrations Under Present Zoning Plan

Unaffected by Residential Septic Tanks
2 mg NO3 / l
70 mg NO3 / l
140 mg NO3 / l
Roads

Potential Maximum NO3- Concentrations Under Future Zoning Plan

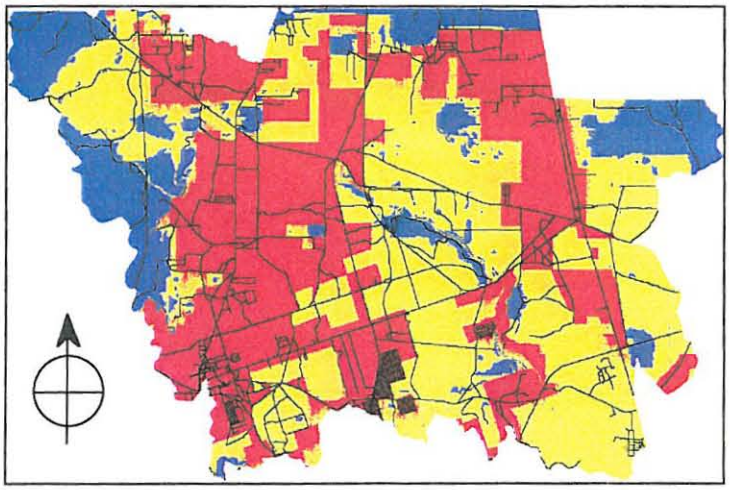


Figure 4. Spatial simulation results of NO_3^- loading the surficial aquifer from septic tanks. Top map shows present conditions and the bottom map shows conditions based on future land use.

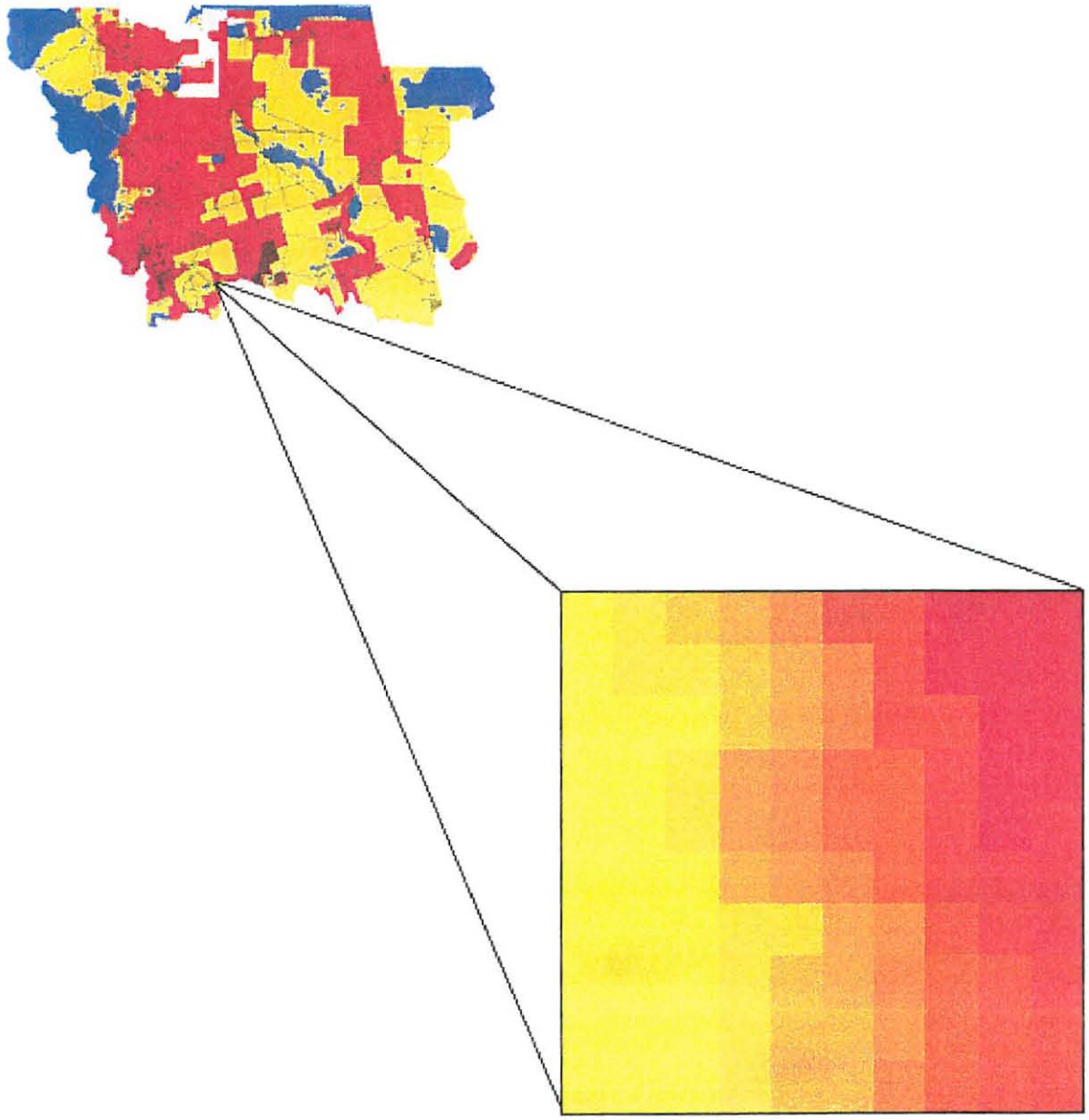


Figure 5. Details of the spatial simulation of nitrogen loading and attenuation of NO_3^- in an area adjacent to a septic tank. The upper right hand corner is the effluent source and flow is right to left..

Figure 7: Effect of Variations in Denitrification Rate

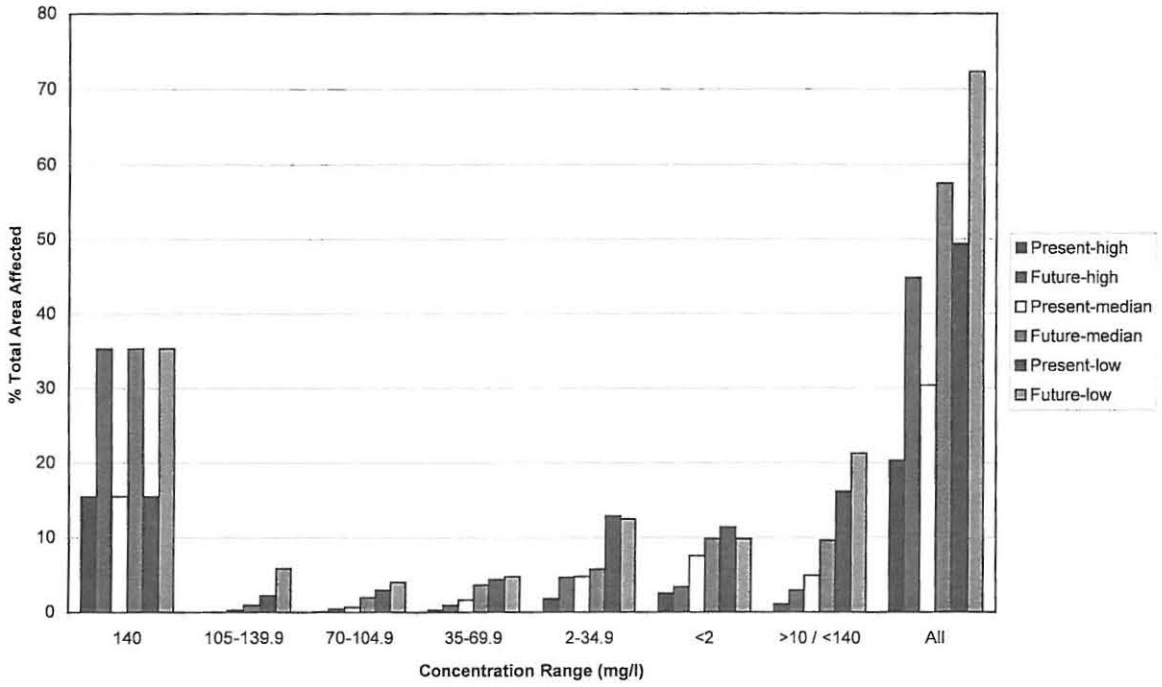


Figure 6. Graph of the percent of the study area influenced by various NO_3^- concentrations when uptake rate is held constant and ground water flow rate is varied one order of magnitude above and below median flow rate.

that percent area affected increases with increasing flow rate. Figure 7 presents results from simulations holding flow rate constant (median flow rate) and varying the NO_3^- uptake rates. The median uptake is believed to best characterize conditions in the study area. The low uptake rate is an order of magnitude lower than the median and the high uptake rate is an order of magnitude greater.

DISCUSSION

The simulation results in Figure 4 contain some anomalies resulting from the input data. As explained below, use of the DEM as a phreatic surface generated questionable groundwater velocities and isolated flowpaths in some small areas. The most striking feature of the maps is the extent of affected areas in the future scenario.

Ambient NO_3^- concentrations in this area are believed to be 2 mg NO_3^-/l or less (USGS, 1994, 1996a). The majority of the yellow areas in the maps represent ambient concentrations. Figure 5 provides a closer look at a plume segment to illustrate the gradation not visible on the maps. Initial concentrations of 140 mg NO_3^-/l are found under the land use polygons only. The percent area affected by 140 mg NO_3^-/l is equivalent to the percent area zoned for residential development (See Figures 6, and 7).

The majority of NO_3^- attenuation occurs within the first few cells surrounding the land use polygons. Concentrations generally drop to 2 mg NO_3^-/l within 30 to 90 m's. In Figure 6, the fraction area affected generally increases with an increase in flow velocity. Percent area affected by 2 mg NO_3^-/l decreases with increasing flow velocities under the future zoning plan. The decrease at this concentration range is attributable to increases in other concentration ranges. In the greater-than-10 / less-than-140 mg NO_3^-/l range the "Present-low velocity" shows a larger fraction affected area than the "Future-low velocity." The difference is due to a larger fraction of residential area and therefore a larger fraction being affected by 140 mg NO_3^-/l.

Variations in NO_3^- uptake rate show as similar trend as with variations in flow rate. No variations were made between the relative uptake rates of the soil types. Higher degrees of variation in uptake rates between soil types may exist. More variation was expected due to the spatial heterogeneity of the soils. Use of a more detailed soils coverage may result in greater variance.

The model results give an indication of potential STSA NO_3^- loading to the surficial aquifer system and the effects of variation in flow and NO_3^- uptake rates. Further modifications could allow the model to be used as a screening tool to highlight areas at potential risk of contamination. Modifications would require extensive data sets and a comprehensive hydrological analysis.

Use of a DEM to represent the phreatic surface is adequate for a first attempt at modeling. DEM derived values will under- and overestimate groundwater slope depending on local conditions. A karst aquifer system, such as that in the Woodville Karst Plain, has an intimate relationship between surface and groundwater. Both flow regimes need to be considered for a full analysis.

The springs and sinkholes present in the study area create highly variable flow patterns that shift in response to storm events. Karst aquifers often have discrete groundwater basins that receive recharge from the land surface through sinkholes and sinking streams. Basin divides are determined by highs in the water table. These boundaries may shift with the water table. Each

Figure 6: Impact of Flowrate Variations on % Total Area Affected

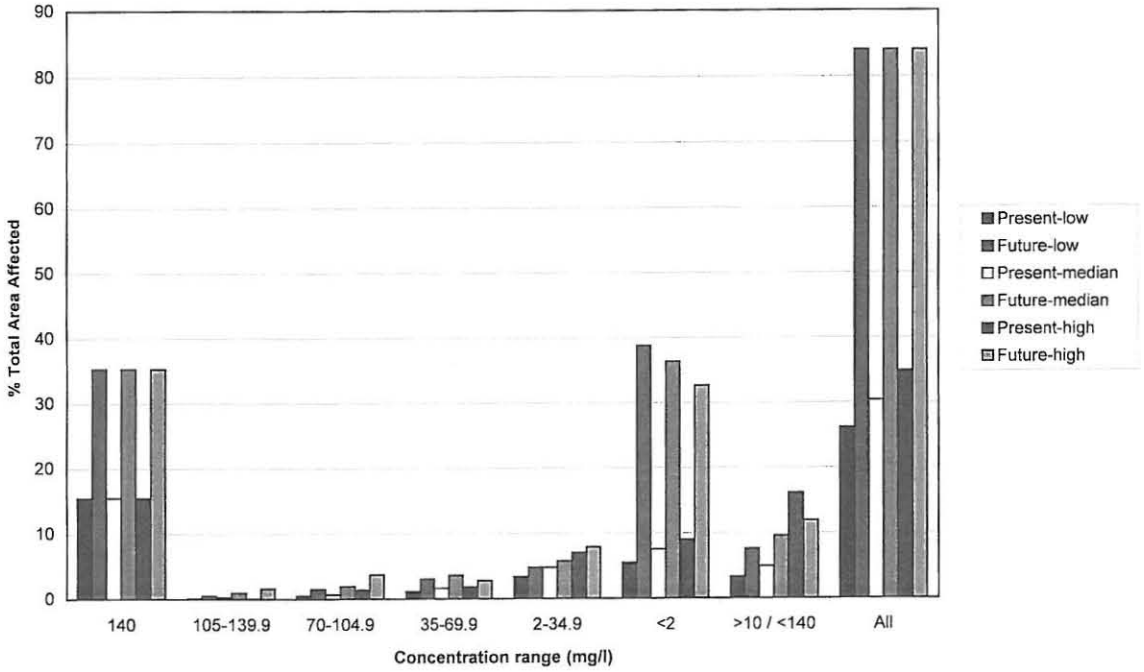


Figure 7. Graph of the percent of the study area influenced by various NO_3^- concentrations when groundwater flow rate is held constant and nitrogen uptake rate is varied one order of magnitude above and below median uptake.

groundwater basin typically drains to a spring or group of springs. Groundwater basins usually extend beyond surface basin boundaries and groundwater flow may not be in the same direction as surface water flow (White 1993).

The estimates of NO_3^- uptake used in the model are unverified. Wide fluctuations in the Wakulla County water table may result in significant changes in denitrification rates on a seasonal basis. NO_3^- reaching shallow aquifer systems, especially those with rapidly fluctuating water tables, has a good chance for removal by denitrification or uptake by deeply rooted vegetation (Lowrance and Pionke, 1989). Data are not available to verify the model. Local testing is necessary to characterize nitrogen dynamics within the soil system.

The high spatial and temporal variability of governing parameters – particularly hydrology – poses a significant challenge to modeling subsurface chemical dynamics in Wakulla County. Isolating and modeling the effect of one nitrogen input (e.g. STSA's) within a karst system requires either broad generalizations or a considerable amount of data. Canter and Knox (1984) described the application of groundwater models to septic tanks as "disappointing and frustrating." Model calibration is difficult due to the lack or questionable validity of input data. Traditional hydrologic modeling such as the approach used in this study may not be efficient. Intensive data gathering and complex modeling may yield no better than general results.

Hydrologic simulations are adapted to a larger time scale (annual) then physicochemical processes (days) (Caussade and Pratt, 1990). It does not seem possible to quantify NO_3^- contamination at a 30m X 30m cell size without a large amount of data gathering. Work has begun on a more general modeling approach that will allow the use of minimal data – sets derived from satellite images – to evaluate land use decisions.

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Chapter 4
Spatial Model of Total Phosphorus Loading and Landscape Development Intensity in the St Marks River Watershed

Neal M. Parker

INTRODUCTION

Statement of Problem

Development of landscapes leads to changes in land cover, surface water runoff, and regional energy flows, among other things. Each of these changes is related to the intensity of activity. Development intensity is related to load on the environment, where the higher the intensity of development, the greater the environmental load. With more integrated environmental policies such as ecosystem management, rapid assessment protocols, cumulative impact studies, and best management practices, there is increasing emphasis on the development of measures of environmental load to track environmental change.

One method of quantifying environmental load is total phosphorus (TP). Elevated TP levels often signify human effects upon ecosystems. Modeling TP concentrations accurately allows managers to first see where concern areas are located, and then focus monitoring efforts there. Predicting TP loads is a spatial phenomenon that requires knowledge of the local landscape geomorphology, land uses, and their spatial distribution. Since elevated TP concentrations are often linked to increased human development, metrics of development intensity would allow better estimates of TP concentrations to be made and help determine where limits are exceeded. Important to know is at what development intensity the first signs of increased TP concentrations are noticed and the general pattern between development and TP loads. The ability to predict TP levels within a watershed based upon development intensity measures would allow

environmental managers and policy makers to minimize water quality degradation by planning future developments in a more comprehensive, pro-active manner.

Plan of Study

In this study the spatial distribution of development intensity and its relationship to environmental load was investigated. Specifically, several LDIs were developed to examine their predictive potential of TP within the St. Marks Watershed located in the panhandle of Florida (Figure 1). A spatial model of TP was developed for the watershed using maps of land use/land cover, soils, topography, water bodies, and aereal pollutant loads. The calibrated model was used to investigate spatial patterns of TP loads and how it related to LDIs. The five measures of landscape development intensity included two physical indices and three energy-based indices. Indices were correlated with TP loads for subbasins of the St. Marks (Figure 2) to investigate the correspondence of landscape development metrics with environmental loading.

Review of Literature

Pollutant Load Models

Corbitt (1990) states that pollutant load models (PLMS) are commonly used to estimate loadings from different nonpoint sources and to evaluate different alternatives. The distribution of loads from various sources within a basin can be examined to identify areas of concern that are most responsible for changes in water quality. To better illustrate and define the numerous types of PLMS, they have been divided into three categories based upon complexity: simple, intermediate, and detailed models (Gilbert, 1997). Depending upon the use intended, one model may better meet the user's needs

Figure 1. Location of the St. Marks Watershed within the Panhandle of Florida.

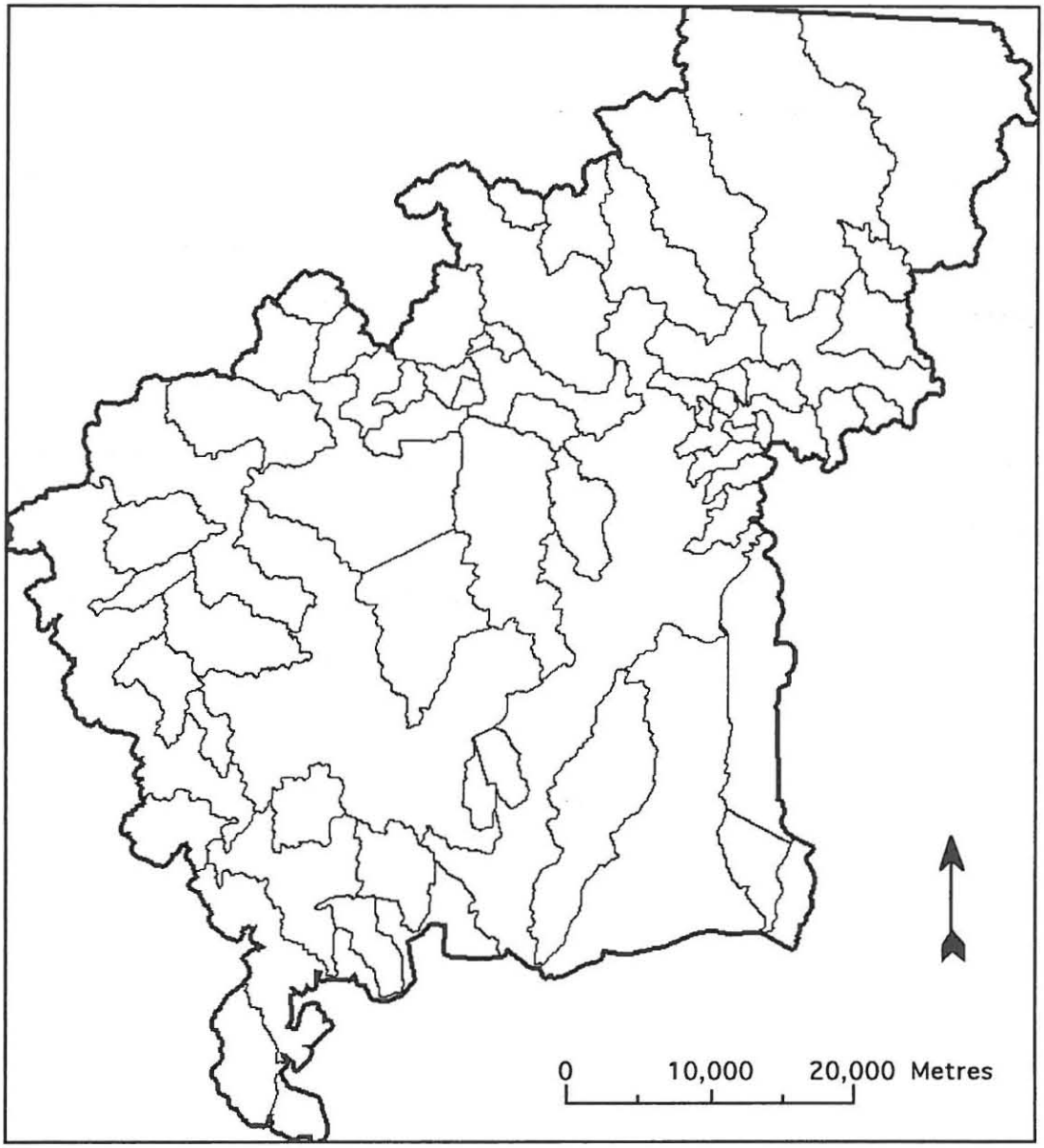


Figure 2. The sixty-four delineated subbasins within the St. Marks Watershed of North Florida. They range in size from 145 hectares to 26,785 hectares.

than another. In general, cost and level of effort to understand the model increases with model complexity.

Donigian and Huber (1991), in their meticulous review of nonpoint source models, suggested five common objectives for the utilization of PLMS: (1) characterize runoff quantity and quality for temporal and spatial detail, (2) provide input to a receiving water quality analysis, (3) determine effects, magnitudes, locations, and combinations of control options, (4) perform frequency analysis on quality parameters, and (5) provide input to cost-benefit analyses. In addition, Donigian and Huber (1991) as well as many other authors (Douglas and Burges, 1982; Kibler, 1982, Huber, 1986) have summarized some of the general modeling caveats as follows: (1) have a clear statement of project objectives, (2) use the simplest model that will satisfy the project objectives, (3) the quality prediction method should match the available data, (4) predict only the quality parameters of interest and only over a suitable time scale, (5) perform a sensitivity analysis as well as familiarize oneself with the selected model, and (6) if possible, calibrate and validate the model results.

Simple models

Gilbert (1997) classifies simple models as screening models that make use of existing data, involve little calibration or verification, and produce answers quickly. These models often take the form of a spreadsheet model that uses multipliers between land area and event mean concentrations (EMCs) for each land use type in order to achieve a total load. Simple models are useful in screening out areas of concern and prioritizing nonpoint source and point source problem areas. They are also useful in

allowing a "first cut" of where resources should be best allocated in order for more detailed modeling to be utilized (Gilbert, 1997).

Spreadsheets, statistical methods, and regression rating curves are some of the tools used by simple models. Spreadsheet models often determine runoff quantity by multiplying runoff coefficients by a rainfall depth. This in combination with an EMC can be used to calculate a total load. Spreadsheets are best suited for estimation of long-term loads because simple prediction methods usually perform better over a longer averaging time (Donigian and Huber, 1991). Statistical methods most commonly take advantage of the fact that EMCs are not constant, but exhibit a log-normal frequency distribution, which with similarly distributed runoff volumes can be used to derive loads (Donigian and Huber, 1991). Regression analysis has been used to relate loads and EMCs to hydrologic characteristics such as stream flow rates and/or volumes (Donigian and Huber, 1991) allowing the user to simply acquire information from one of the "rating curves" for their region. Unfortunately, as Huber points out, regression approaches are subject to large potential errors when used to extrapolate for different conditions. Regression equations, developed through the United States Geological Survey (USGS) (Driver and Tasker, 1988), were found to be most accurate for the arid Western US, and least accurate for areas with large mean annual rainfall.

Some common simple models include the EPA Screening Methods, EPA Simple Method, USGS Regression Method, Water Screen, WaterShed, and Watershed Management Model (WMM). Applications of simple models have been relatively widespread due to their ease of use and lack of need for extensive data.

Intermediate models

In many respects, intermediate models make use of simple model approaches integrated with a geographic information system (GIS). This brings a spatial component into modeling that facilitates both the input of data (Gilbert, 1997) and analysis. The use of GIS allows comparisons to be made between different watersheds or regions in a graphical manner and thus allocate resources accordingly. Intermediate models generally require some level of calibration and verification, requiring the input data to be of sufficient resolution (Gilbert, 1997). Oftentimes a GIS allows various development scenarios to be modeled in accordance with the local planning authority, providing a useful tool to planners and water quality experts.

There have been numerous applications of intermediate models. The Northwest Florida Water Management District's (NWFWMD) Deer Point Lake Study (1994) integrated GIS, satellite imagery, and land use/cover maps to model existing development patterns. This study evaluated existing and potential pollution contributions to Deer Point Lake from non-point sources. Tippett (1993) used GIS to vary five riparian buffer zones along a stream. A regression model was used to relate soluble reactive phosphorus and nitrate-nitrogen for each sampling station to watershed area and land use patterns in each buffer zone.

Adamus and Bergman (1995) developed a GIS nonpoint source screening model for the St. Johns River Water Management District (SJRWMD). Their model is contained entirely within the GIS framework, giving it greater flexibility, efficiency, and graphical capabilities. Model inputs include five spatial data layers, runoff coefficients, mean runoff concentrations, and stormwater treatment efficiencies. A similar study was

performed on the Myakka River Basin by Hart (1993) using a GIS based computer model. Numerous other GIS based watershed models have been developed around the country (He et al., 1993; Heidtke and Auer, 1993; Jeton and Smith, 1993; Levine and Jones, 1990). Soranno (1996) took a particularly interesting approach modeling nonpoint-source phosphorus loading by developing a GIS based overland flow algorithm that accounted for spatial patterns in topography as well as land use.

Detailed models

The most complex of nonpoint source (NPS) models, detailed models take into account many different chemical, physical, and biological processes within the watershed (Gilbert, 1997). Data inputs are extensive including stream flow, pollutant accumulation and decay, particle transport and settling, infiltration, runoff, and ground/surface water interactions (Gilbert, 1997). However, where data is limited, detailed models can often be run with many of these variables at default settings. Models of this type often require many man-hours, weeks, or months of training (Donigian and Huber, 1991). In addition, these models often require "experts" for consultation in their use, although if used properly, reasonable predictions can be made for any given point or points within a watershed (Gilbert, 1997).

Models of this type include the Storm Water Management Model (SWMM) and Hydrologic Simulation Program - FORTRAN (HSPF). The Linked Watershed Waterbody Model (LWWM) and Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) are two programs that interface many of these models in a GIS environment, providing the user with a graphical user interface to enter and alter parameters. BASINS is a recent development of the Environmental Protection Agency

(Lahlou, 1996) that contains two modules: an Assessment/Planning Module and a Modeling Module. The Assessment/Planning Module allows rapid evaluation and results for selected areas. Three scales of analysis allow the investigator to zoom in from a regional scale to the individual stations. The Modeling Module looks at pollutant loadings from point and non-point sources using three different models: a non-point source model (NPSM), fate and transport model (QUAL2E), and a stream routing model (TOXIROUTE).

Applications of detailed models are extensive and growing rapidly as technology continues to advance. In particular, SWMM has been used extensively for urban watersheds in Florida (Bartel et al., 1991b; Wooten et al., 1991), and elsewhere in the country.

Spatial Data Needs

Novotny and Chesters (1981) have summarized the general data needs for a non-point source model into three categories: watershed information, state variables, and input variables. Table 1 lists this summary.

Previous Studies within North Florida Watersheds

Nutrient and water quality studies within North Florida Watersheds have ranged from land use, nutrient budgets for several of its subbasins to detailed stormwater management studies within the Tallahassee urban area.

Early studies were conducted by Turner (1975) on the effects of urban land use on nutrient and suspended solids export from north Florida watersheds. Turner compared two watersheds of similar size, geomorphology, and pedology for nutrient and suspended solids export that represented forested-agricultural and residential-commercial land uses.

Table 1. General data needs for nonpoint source models.

Watershed Information	State Variables	Input Variables
Area of watershed	Ambient temperature	Precipitation
Areas of homogenous subareas (catchment)	Reaction rate coefficients	Atmospheric deposition
Imperviousness of each catchment	Adsorption/desorption coefficients	Evaporation and evapotranspiration
Slopes for each catchment	Growth stage for crops	Pollutant concentrations
Fraction of impervious areas directly connected to a channel	Growth stage for natural vegetative cover	Design flows for receiving waters
Total surface storage	Daily accumulation rates of litter	
Soil characteristics	Traffic density and speed	
Crop and vegetative cover	Potency factors for pollutant	
Curb densities or street gutter lengths	Solar radiation	
Sewer system characteristics		
Natural drainage characteristics		

Source: modified from Novotny and Chesters (1981).

Hydrochemical analysis of runoff from thirteen storms in the urban watershed and eight storms in the forested watershed showed a strong contrast in both the magnitude and temporal distribution of exports. Turner concluded that observed differences in exports were related to changes in hydrology associated with urban development. Comparisons of material loads exported by storm flow and low flow for the watersheds suggested increased significance of storm events in export from the urban watershed. Additional research conducted by Burton et al. (1977b) upon three north Florida watersheds; an urban, suburban, and forested-agriculture basin, supported findings that urbanization (1) increases storm peak flows, (2) increases the ratio of quickflow volume to delayed flow volume and (3) increases annual runoff.

Tallahassee's presence within the St. Marks Watershed has resulted in many stormwater studies on the city as well as the immediate surroundings. Lake Munson, a waterbody on the south side of Tallahassee that receives a significant quantity of the cities urban runoff, has been the focus of several studies. Maristany et al. (1988) looked at the historical changes in water quality parameters in Lake Munson since its formation from a dammed cypress swamp in 1950. The study concluded with recommendations for restoration measures that should be taken. Further studies by Maristany and Bartel (1989) focused on the use of wetlands or wet detention ponds for stormwater management at the Lake Munson site and their long-term treatment efficiencies. Adjacent to the Lake Munson subbasin is the Lake Lafayette watershed that also receives a significant portion of Tallahassee's runoff. Swanson et al. (1996) addressed the degradation of Lake Lafayette by urban stormwater and point source discharge.

Recommendations within the report included strategies for land use controls, in-lake management and clean-up, and subsequent research and monitoring.

An extensive stormwater management plan was developed for the City of Tallahassee and Leon County by the Northwest Florida Water Management District and Camp Dresser and McKee that included the Lake Munson and Lake Lafayette basins as well as other basins within the St. Marks Watershed (Bartel et al., 1991b; Wooten et. al., 1991; Camp Dresser and McKee, 1995). The most recent compilation of water quality parameter data for the St. Marks Watershed as well as the State of Florida is found in the Department of Environmental Protection's 305(b) Main Report. This report covers water-quality information for the state with a section on basic facts, ecology, and human impacts within the St. Marks Watershed.

Description of the Study Area

Location

The St. Marks Watershed is located in the panhandle portion of northwestern Florida at approximately 30° north latitude and 84° west longitude (see Figure 1). The watershed covers about 305,620 hectares and includes portions of three counties including Wakulla, Leon, and Jefferson. About 5% of the watershed extends over the state line into Georgia. Average rainfall measured at the Tallahassee Municipal Airport (1958-1987) for the region is about 165 cm per year. The wettest year on record was 1964 with 263 cm, while the driest year was 1961 with 118 cm (Wooten et al., 1991). Average monthly precipitation shows that July is the wettest month of the year and October is the driest (Bartel et al., 1991b). Average yearly temperature is 68 degrees

Fahrenheit with a record low in 1899 of -2 degrees and a record high in 1985 of 103 degrees Fahrenheit (Wooten et al., 1991).

A systems diagram is presented in Figure 3 showing the major energy flows within the St. Marks Watershed. The symbols and their definitions are given in Appendix A. The diagram shows the major inputs to the system including sunlight, wind, rain, groundwater, fuels, and goods and services. The production symbol on the left of the diagram represents the natural environment within the watershed including silviculture and agriculture operations. These systems rely largely upon sunlight and rainfall for support. On the far right is the urban sector, largely composed of Tallahassee but also several smaller towns throughout the watershed. The urban regions are dependent upon the environmental sectors as well as fuels and good and services to export their products to outside markets and maintain themselves.

In the middle of the diagram are three storage tanks that depict water storage within the system. Connecting these tanks and the natural and urban sectors are various pathways of water flow, emphasized by darker lines. The uppermost tank consists of surficial water or water within reach of the root zone that is fed directly by rainfall. As this water converges surficial groundwater and surface runoff, it enters water bodies portrayed by the second tank. Water from urban systems often bypasses the surficial aquifer altogether, entering directly into water bodies. This is portrayed with the interaction symbol and sensor between the rain, urban system, and middle tank. The amount of stormwater runoff into water bodies is proportional to the size of the urban system. These water bodies include streams, rivers, and lakes that recharge deeper aquifers and ultimately enter the gulf. Along the way this water is used by both natural

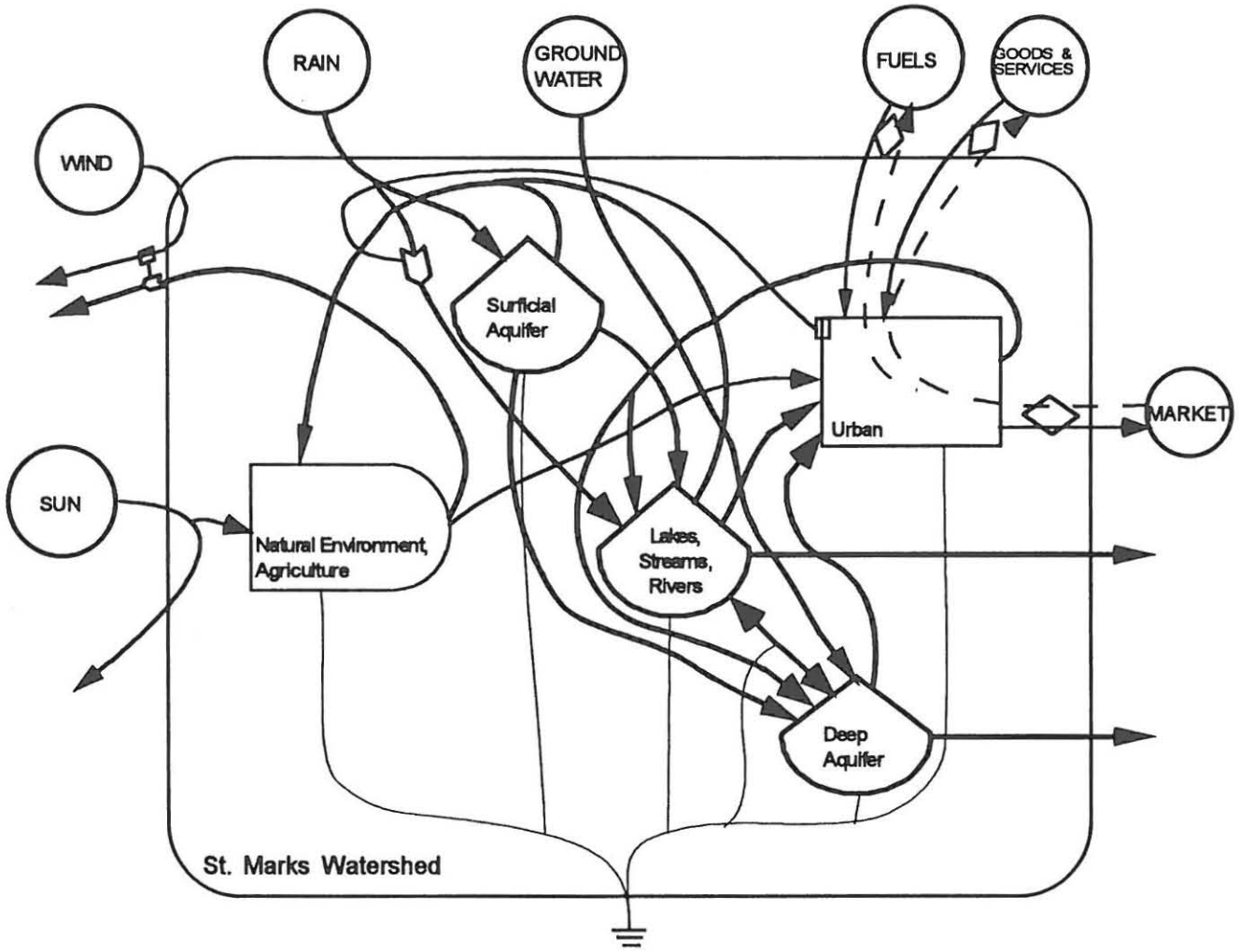


Figure 3. Aggregated systems diagram showing the major energy flows within the St. Marks Watershed of North Florida. The pathways of water flow are emphasized with larger lines.

and urban ecosystems. Carried along with the water as it flows through these systems are many constituents including phosphorus, the element of interest in this study.

Geomorphology

The St. Marks Watershed can generally be divided into the highlands and lowlands with further divisions into the delta plain, tertiary highlands, terraced coastal and river valley lowlands (Hendry and Sproul, 1966) (Figure 4). The highlands or northern half of the watershed is a region of undulating hills that range from fifteen to seventy meters above sea level. The St. Marks River begins here as many small swamps and bay heads. Soils of the highlands are composed of clays, silts, and sands that are weakly cemented together into a somewhat impermeable strata that forms many of the small ponds and lakes observed. Separating the highlands from the lowlands is the Cody Escarpment--an abrupt drop in elevation--that runs in an east-west direction south of Tallahassee.

The Gulf Coastal Lowlands can be divided into the Appalachicola Coastal Lowlands and the Woodville Karst Plain. Located mostly within the Appalachicola National Forest, the Appalachicola Coastal Lowlands are underlain with thick elastic clay deposits covered by sands and have many shallow bays and poorly defined streams (Hendry and Sproul, 1966). The relatively flat Woodville Karst Plain stretches from the Cody Escarpment to the Gulf and ranges from zero to about twenty meters in elevation. Loose sands overlie a limestone substrata creating a topography of sand dunes and sinkholes amid poorly drained pine flatwoods, swamps, and river basins (Hendry and Sproul, 1966; Rupert and Spencer, 1988).

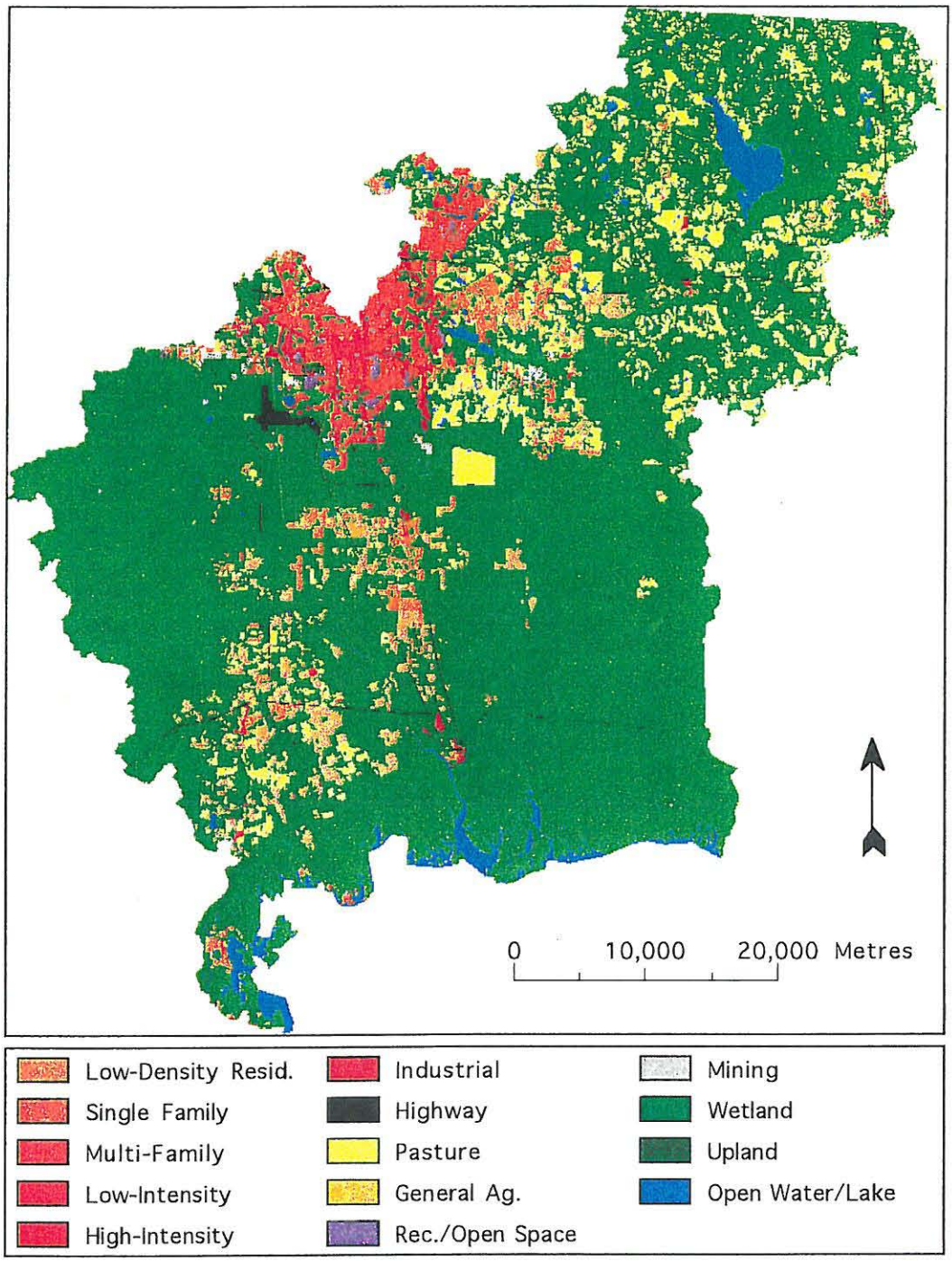


Figure 5. Land use map for the St. Marks Watershed of North Florida. Tallahassee is the large red assemblage. The northeastern region is dominated by agriculture as seen from the numerous yellow parcels.

At a point midway through the watershed known as Natural Bridge, the St. Marks flows underground for a short distance before reemerging. North of Natural Bridge, the St. Marks River has a poorly defined channel that becomes more discernible after groundwater flow is added to the river during its short subterranean traverse. The St. Mark's largest tributary, the Wakulla River, originates at Wakulla Springs and converges with the St. Marks about three miles north of the Gulf. Average yearly stream flow for the Wakulla and St. Marks Rivers at this point is 393 cubic feet per second (cfs) (Rupert and Spencer, 1988) and 700 cfs respectively, although both rivers' stream flow has significant variability. At the southern edge of the watershed, the St. Marks River enters the Gulf of Mexico through the salt marshes of the St. Marks National Wildlife Refuge.

Land Use

Current land use/land cover and its distribution is shown in Figures 5 and 6 for the St. Marks. Shades of red indicate urban areas while yellow is various types of agriculture or pasture. The lighter greens denote wetlands and the darker greens are upland systems including silviculture operations. Blue portrays open water, with the most obvious open water features being Lake Miccossukkee in the northern section of the watershed and the Gulf in the south. Tallahassee is the dominant urban feature in the watershed with various smaller communities located in the hinterlands. The northeastern portion of the watershed contains a significant amount of agriculture while the southern half of the watershed is relatively undeveloped with significant amounts of acreage devoted to pine plantations (see Figure 6). The Appalachicola National Forest encompasses the southwestern portion of the watershed. Vast salt marshes spread along the southern edge of the watershed marking the transition from fresh to saltwater and the Gulf.

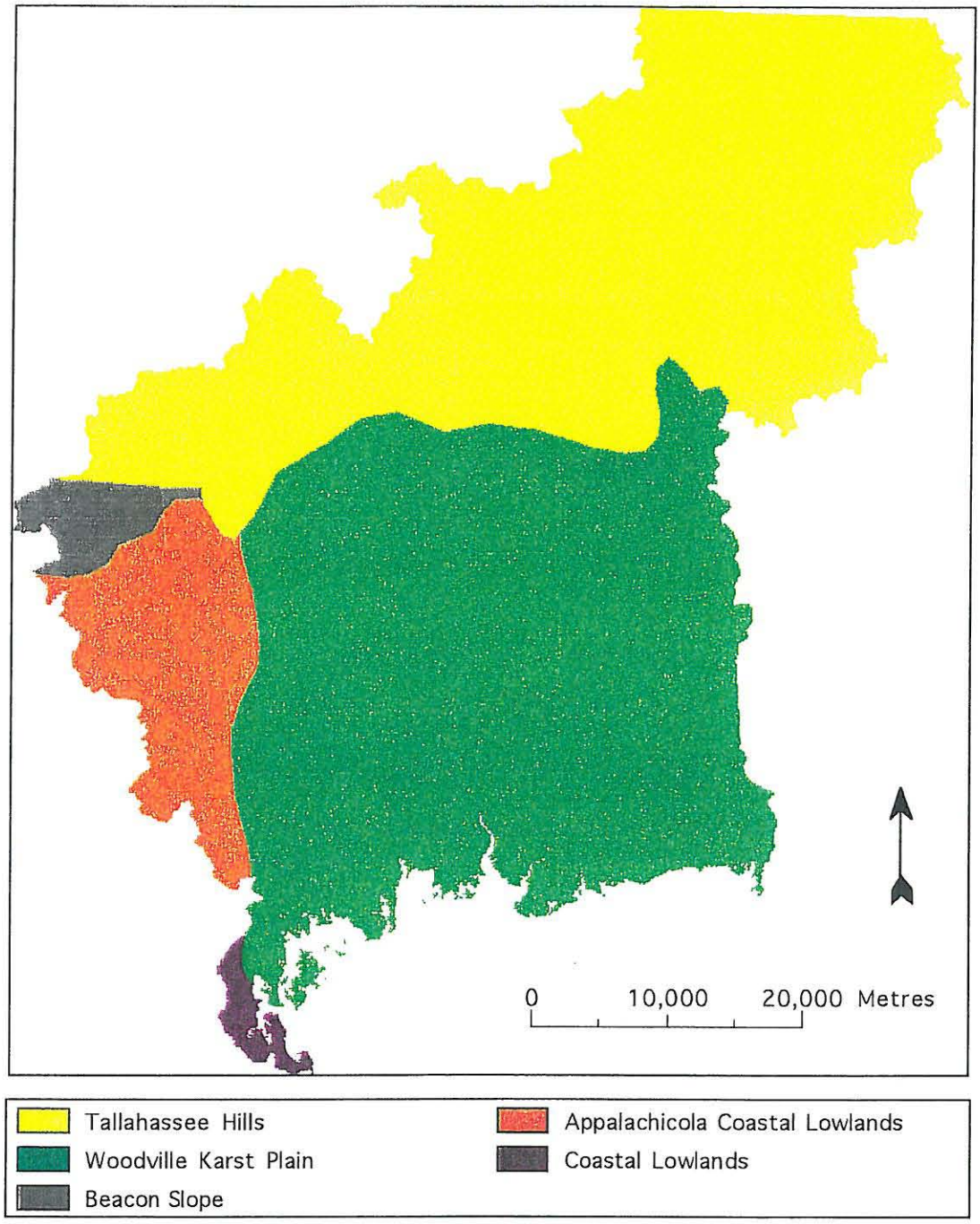


Figure 4. The five major physiographic regions within the St. Marks Watershed of North Florida.

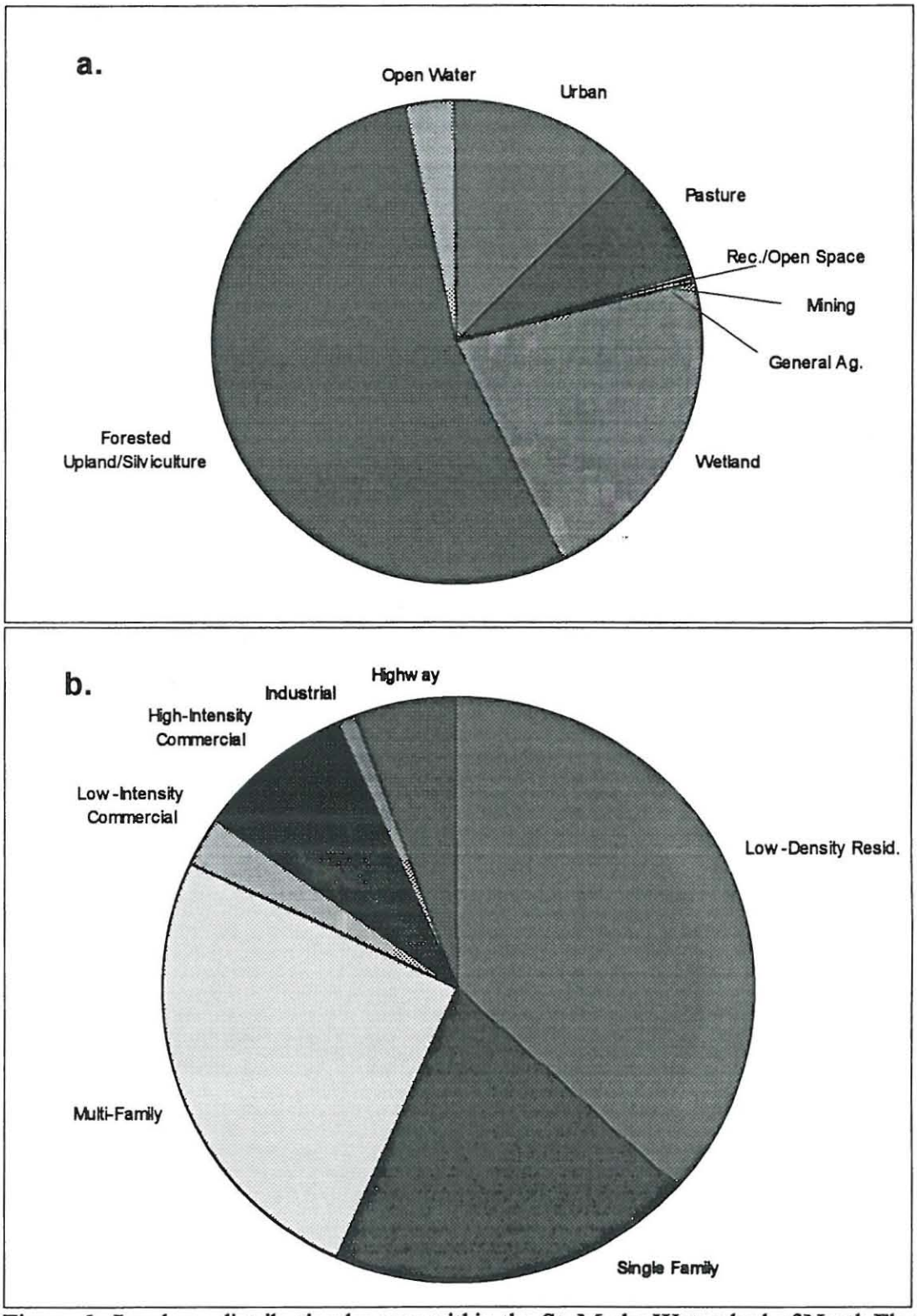


Figure 6. Land use distribution by area within the St. Marks Watershed of North Florida. a) All uses within the watershed (305,620 hectares); b) Urban land uses (11% of total; 34,347 hectares).

METHODS

Overview

The procedure for investigating the relationship between landscape development intensity and water quality was conducted as follows:

1. Spatial GIS coverages were assembled including land use/land cover, soils, topography, and hydrography. From these, other data layers were derived including aereal phosphorus loads, stream networks, and imperviousness.

2. A spatial phosphorus loading model for Total Phosphorus (TP), which includes particulate, dissolved and organic, was developed using these spatial data layers.

3. Output from this model was then compared with sampling points throughout the St. Marks Watershed. Model parameters were adjusted until a best fit between predicted and observed results was achieved.

4. Five LDIs were constructed for the watershed. These included two physical LDIs and three energy based LDIs.

5. Model results were then correlated with the five LDIs for subbasins of the St. Marks Watershed.

Techniques of Map Analysis

Methods of spatial analysis including those used in this study generally employ a technique known as map algebra. Just as two numbers can be added, subtracted, multiplied, or divided, a similar procedure can be used with maps. A map or coverage is

a spatial database where every point upon that map has a value that represents some characteristic of that location, whether it be elevation, land use, or soil type. For raster maps, these points are cells with known dimension of length and width. Each cell within the map has a value that corresponds to some known attribute. For example, on a map of land use, a cell value of 1 may represent low-density residential, a value of 4 wetlands, etc. For an analysis to be accomplished between two or more maps, the maps need to be georeferenced to each other. In other words, all cell locations upon one map should correspond to cell locations upon the other maps. If these maps were to be overlayed, cells from each map with the same spatial location would fall one on top of the other. Ian McHarg (1969) employed these techniques in the late 60s using clear plastic sheets for each coverage. Each sheet had a theme, such as a water bodies coverage or a vegetation coverage that when overlayed provided useful information such as where a future development could be located.

Shown in Figure 7 is an illustration of a simple map algebra process where two maps are added together. These maps are very simple, containing only four cells. Maps can have any number of cells and it is not uncommon for a map to have upwards of a million cells. Each cell within Map A overlays a corresponding cell in Map B. In this example, the lower right hand cell contains the value 1 in Map A and 2 in Map B. When the two maps are added, as in the statement "Map A + Map B", a third map is produced, Map C. In the previous example, the Maps A and B with cells of values 1 and 2 produce a new map with a cell of value 3. All corresponding cells within the two maps are added.

While this illustration is a relatively simple example of map algebra, the algebraic complexity possible with maps is limited only by the imagination. A map can be

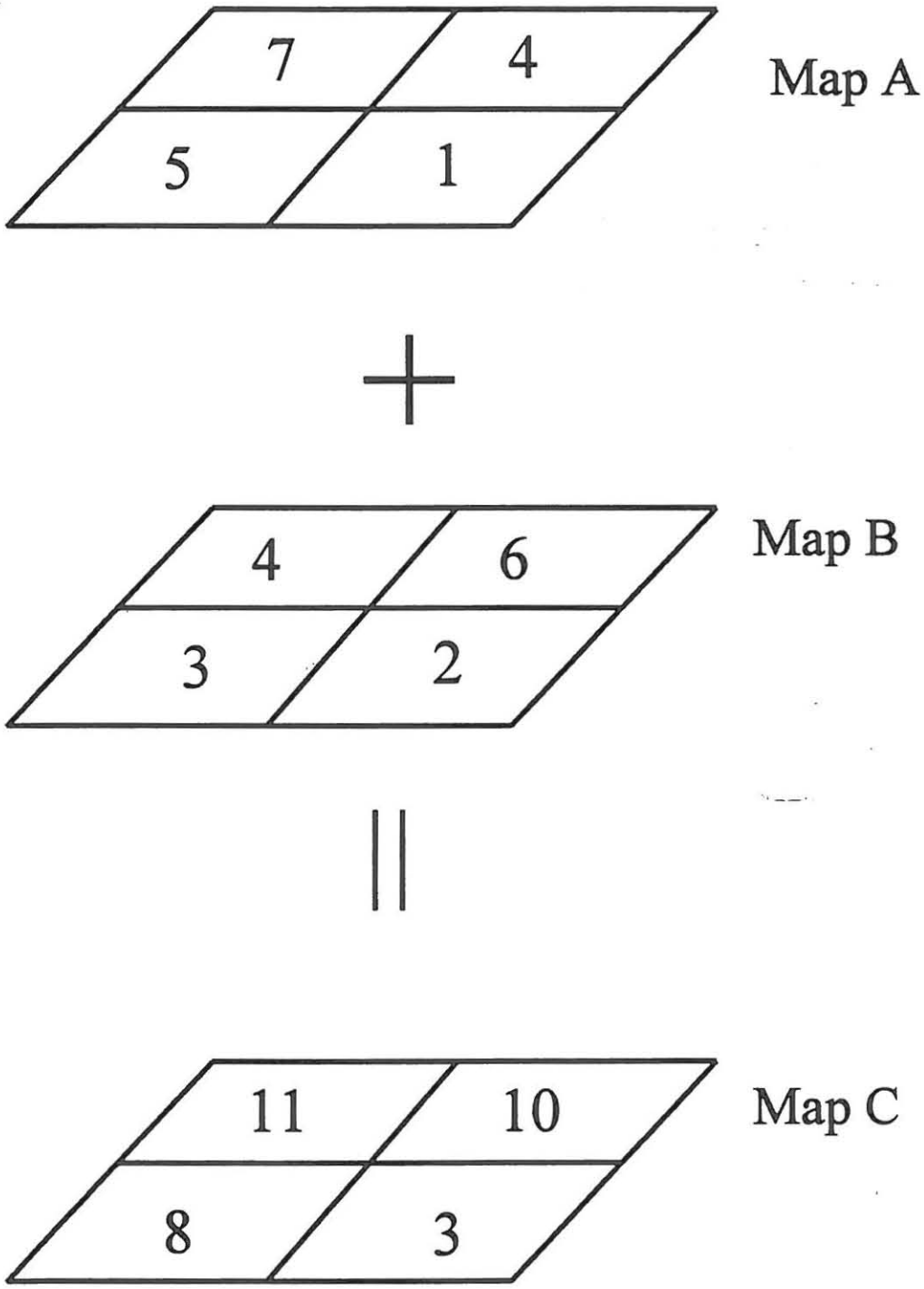


Figure 7. Technique of map algebra in which two maps (Map A and Map B) are added to produce a third map (Map C). All cells within the two maps are added.

substituted in place of a number for virtually any calculation, whether it be a simple addition or a complex formula. Besides basic arithmetic, new maps can also be created by extracting only the desired information from the old maps. For example, from a map of land uses, a new map of just wetlands can be created. If Map A in Figure 7 represented a map of land uses, where 1 = low density residential, 4 = wetlands, 5 = commercial, and 7 = pasture, a new map of just wetlands can be generated by selecting only those values in Map A equal to 4. In this case, only one of the four cells has a value coded for wetlands.

Much of the spatial analysis used within this study relied upon the techniques of map analysis. Of critical importance to the spatial TP model was an operation that adds cell values within one map in a downhill direction. Referred to from here on as a "drain" operation, it is useful for determining TP loads or the volume of surface water runoff that passes through a cell for a known time period. A drain operation drains a load map over an elevation map. The drain command and several other MapFactory operations used in this study are defined in Table 2. Figure 8 illustrates the drain methodology and results for a three celled map. The top map, referred to as a Load Map, could represent phosphorus or precipitation. When the drain command is applied to this Load Map upon an Elevation Map, the Load Map values will drain down slope, halting at the lowest elevation. This product map of the Load Map and slope is the Cumulative Load Map where cell values from the Load Map have been added in the downhill direction. The top value in the Cumulative Load Map (4), remains the same because there are no values above it in the Load Map. The middle cell value is the sum of 4 and 2, and the lowest cell value is the addition of all three Load Map values. If there was a lake below Load

Table 2. MapFactory GIS operations used in this study.

Operation Name	Description
Drain	Generates a map that represents the amount of precipitation that flows through the surface map along the steepest downhill path. Each cell value in the map indicates the amount of precipitation that has flowed through that cell.
Recode	Generates a map layer where each cell is assigned a specific value by the Assigning modifier based on the original value of that cell in the operand map layer.
Spread *uphill*	Generates a map layer of distance or costs from non-VOID cells. The direction of the spread can be controlled. Spread can take vertical distance into account. The uphill modifier restricts spreading to level and uphill paths only.

Source: MapFactory Module Reference (1996).

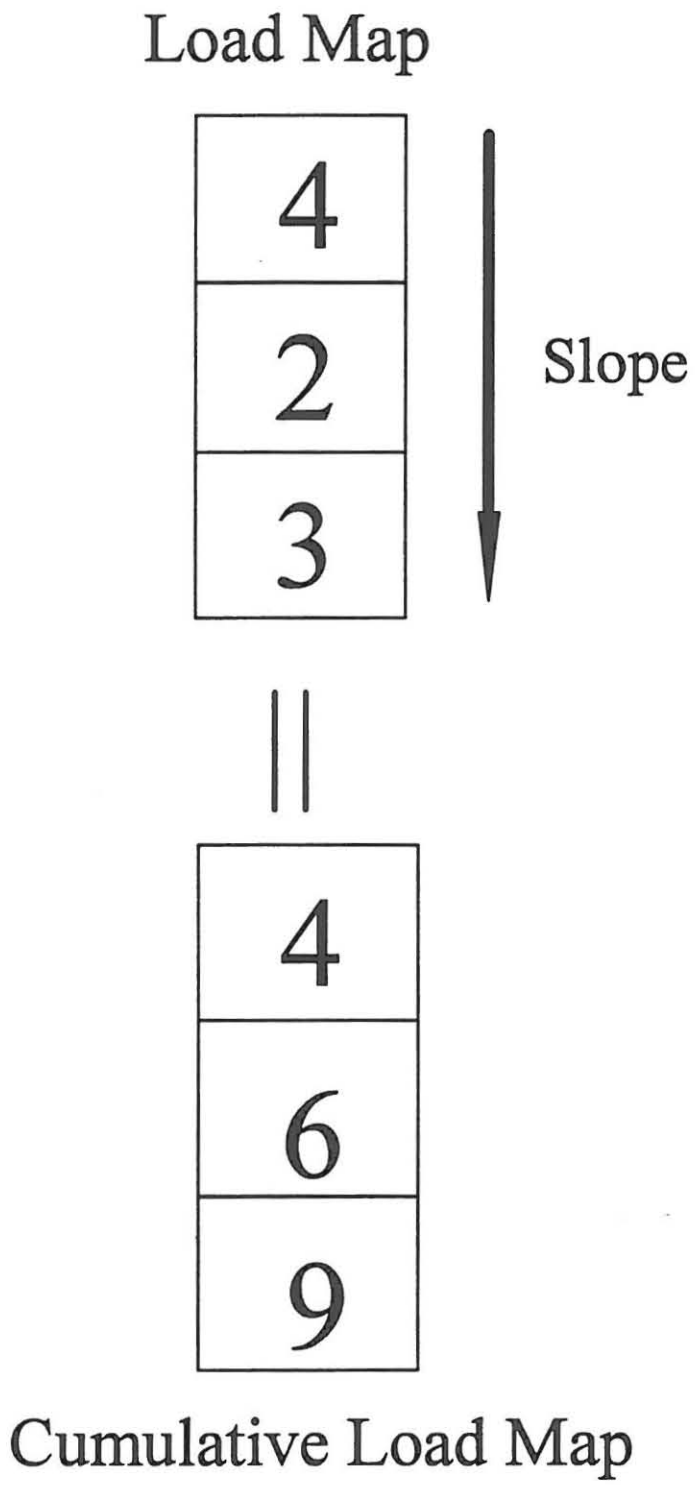


Figure 8. Overland flow technique of draining a map to produce a cumulative load map. All cells in the load map are added in the downslope direction.

Map value 3, it would receive a total load of 9. Figure 8 is a simplistic one-dimensional representation of what is actually a three-dimensional phenomenon. The St. Marks Watershed landscape would have many of these flow paths traveling in different directions, often times merging or splitting, depending upon the underlying elevation.

Many of the secondary and intermittent coverages used within this analysis were developed using techniques such as those previously mentioned. Map operations utilized in this study such as the drain command, are nothing more than several fundamental map algebra steps combined, making analysis more convenient for the user.

The GIS Model

The Spatial Phosphorus Model

Figure 9 illustrates the map algebra methodology for the GIS based spatial phosphorus load model. The model uses an overland flow algorithm that determines cumulative TP load to all receiving streams. TP loads accumulate along flow paths that are determined from the DEM. An underlying model assumption is that TP loads are transported with surface waters. The model utilized primary and secondary coverages to predict TP loads (Figure 9), and model results depict average yearly TP loads, concentrations, and runoff volumes for all raster cells within the watershed.

GIS coverages used in the model were obtained through the Florida Department of Environmental Protection. Analysis was accomplished using the GIS packages ARC/INFO and the Macintosh mapping analysis software MapFactory. All coverages were converted to raster or cell based grids with a 100 meter (1 hectare) cell resolution. The coverages used can be classified into primary and secondary depending upon whether they are source or source derived coverages.

Primary Coverages

Primary coverages consist of the spatial data layers that are the backbone of the analysis. These coverages were taken straight from the GIS source library with little or no manipulation. Primary source coverages include land use, general soils, topography, and hydrography.

Land use coverages were based upon 1994/1995 sources for northwestern Florida that were categorized using the Florida land use codes classification system (FLUCCs) (DOT, 1985). Land use data were aggregated into fourteen categories (see Figure 5).

General soils data, shown in Figure 10, were taken from general soils coverages for the state (USDA, 1991). The state soil geographic data base (STATSGO) provided information on hydrologic soil groups for each soil type necessary in determining runoff potential. The general soils coverage groups were aggregated into one of the four (A, B, C, D) hydrologic categories.

USGS topography data were assembled from 1:100,000 scale quadrangles for the region and then interpolated to derive a digital elevation map (DEM) (Figure 11.) Elevations within the watershed ranged from sea level at the Gulf to over seventy meters in the upper reaches of the watershed, a region known as the Tallahassee Hills. The lower half of the watershed, seen as the large dark area in Figure 11, is a relatively level region known as the Woodville Karst Plane which averages about five meters above sea level.

The hydrography coverage for the St. Marks Watershed is shown in Figure 12. This coverage includes the St. Marks River, all major stream systems, and delineated boundaries of all major lakes and coastal marshes.

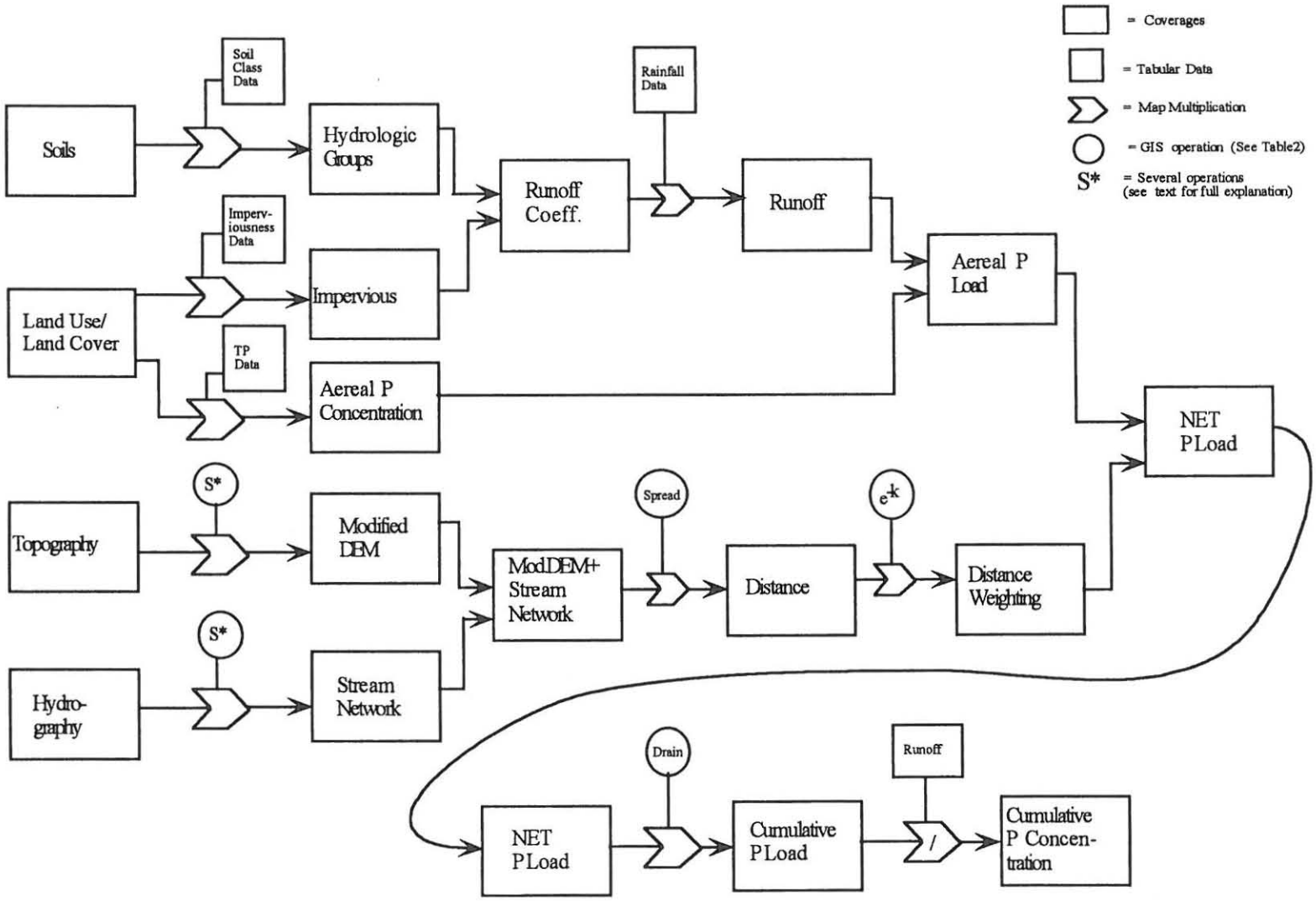


Figure 9. Flow chart of the spatial TP model algorithm.

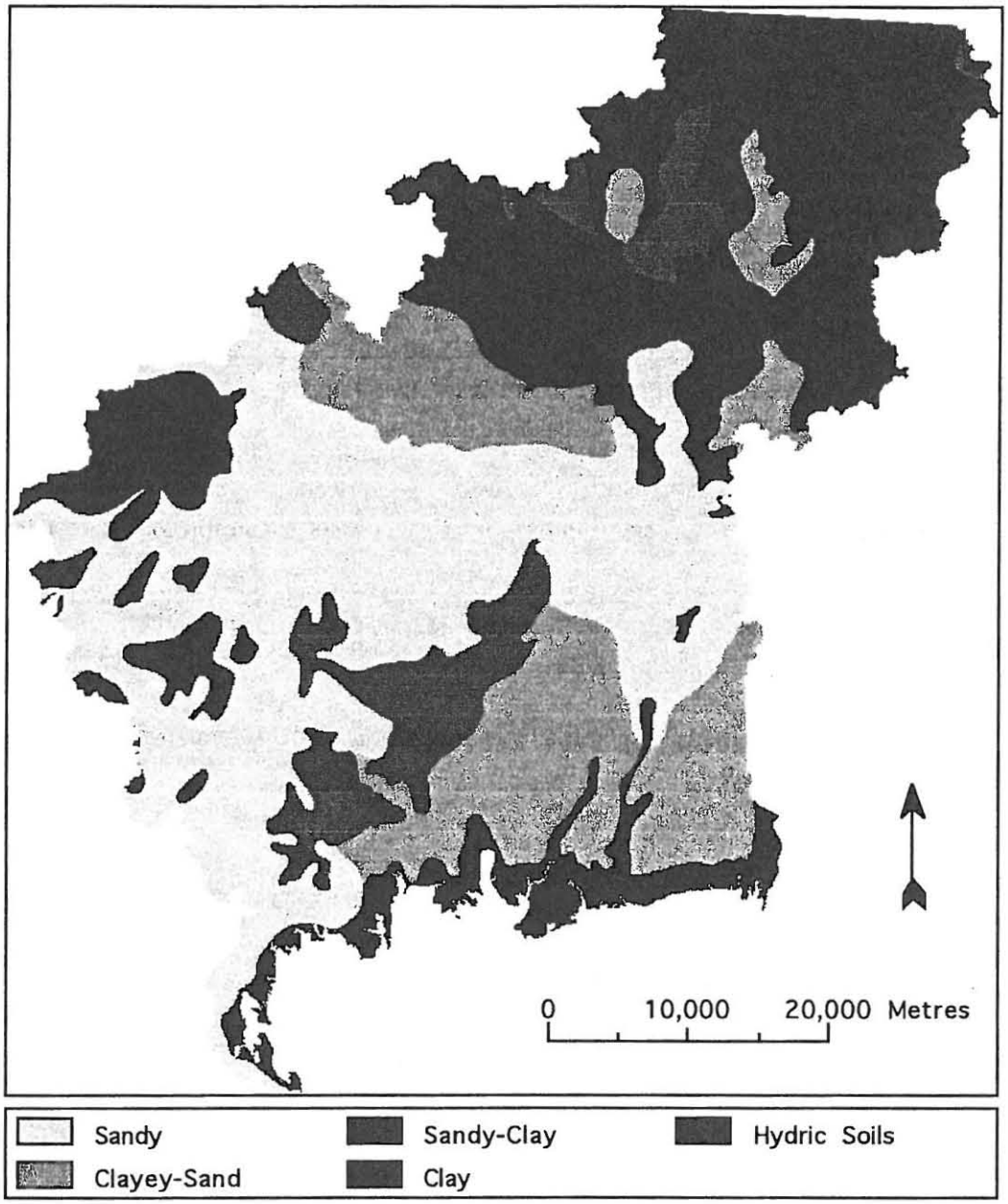


Figure 10. General soils for the St. Marks Watershed of North Florida. Different shades of gray denote different soil classes.

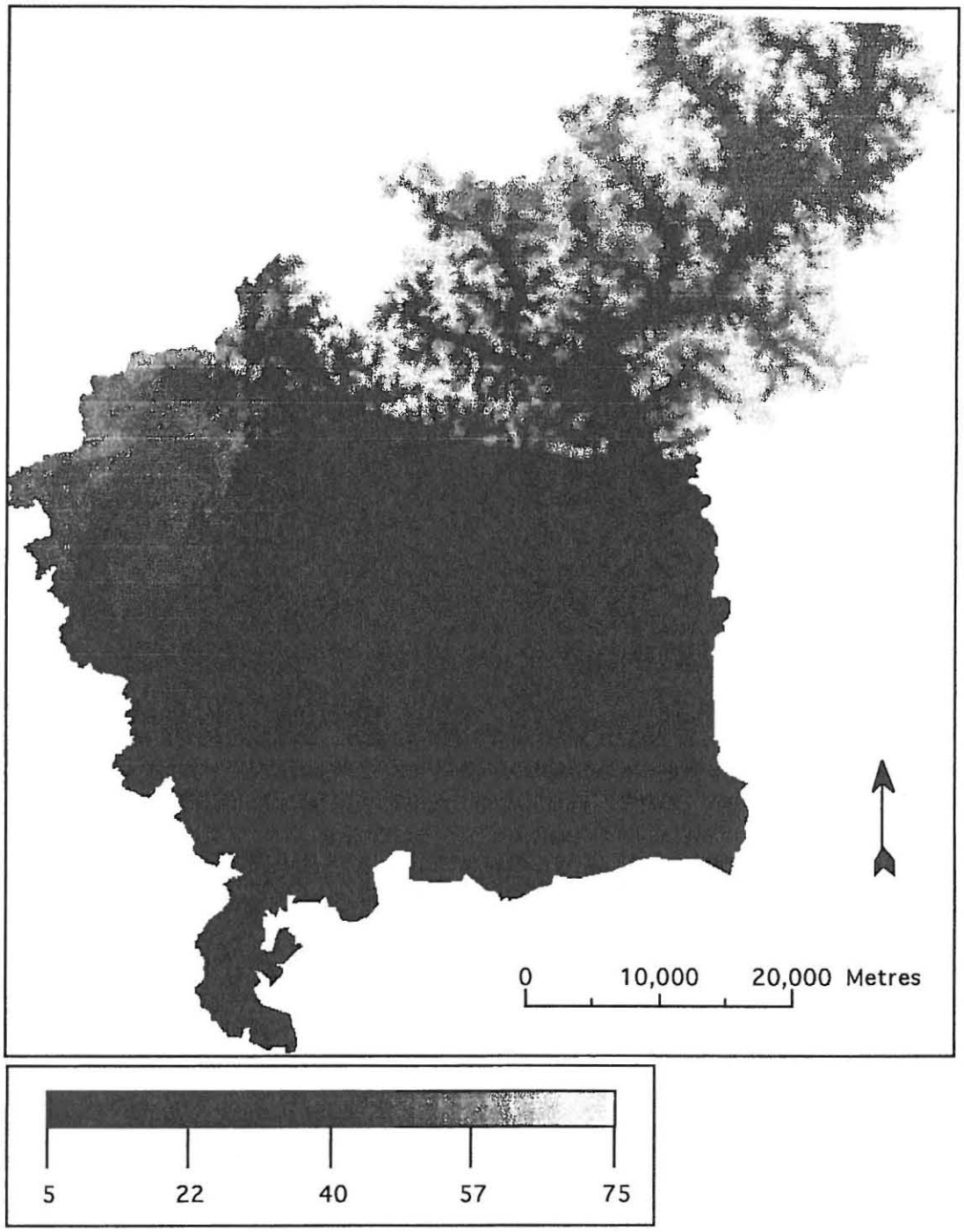


Figure 11. Digital elevation map (DEM) of the St. Marks Watershed of North Florida with values in meters above mean sea level. A distinct change in physiography can be seen from the more hilly northern half to the relatively flat southern coastal plains.

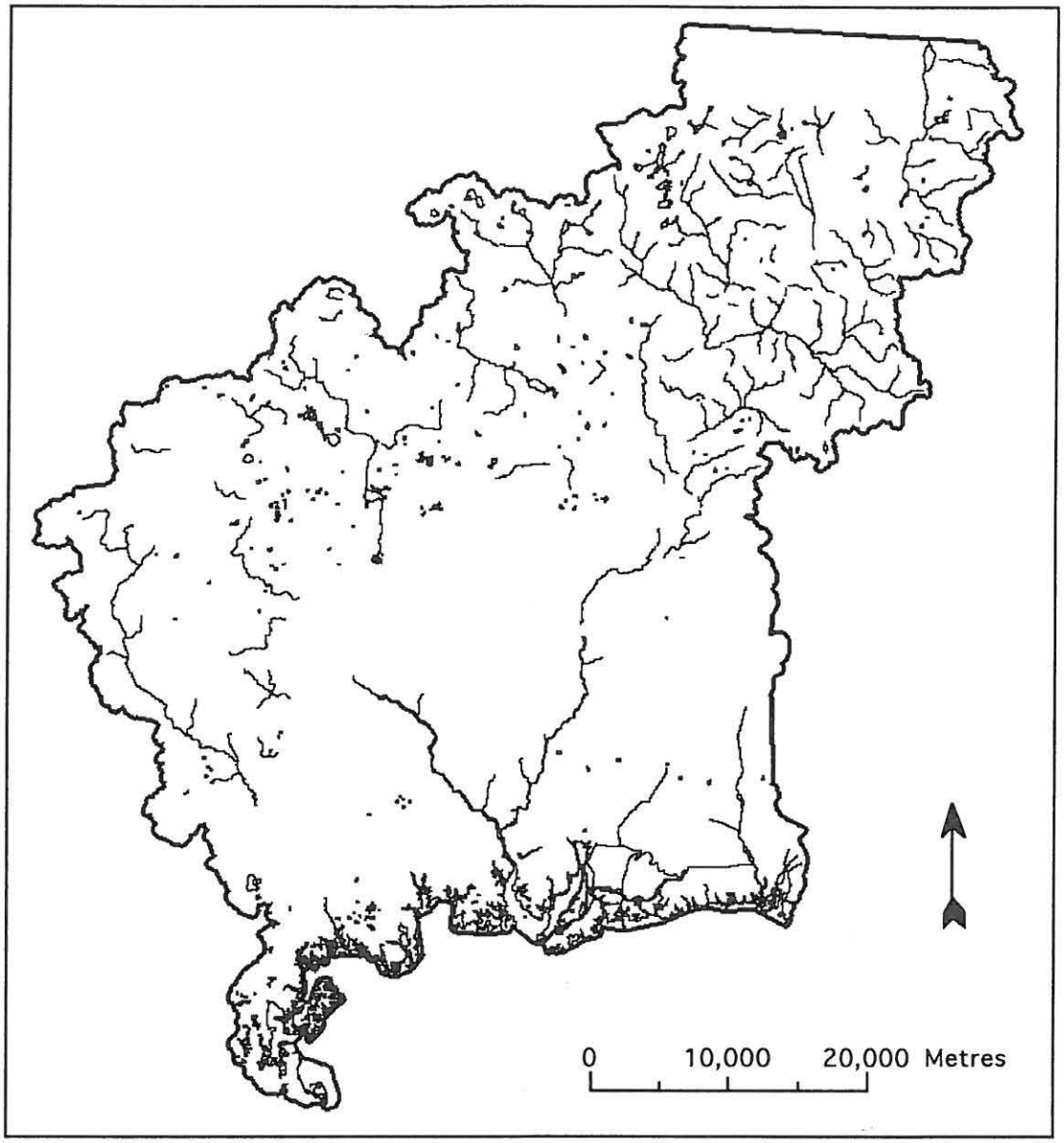


Figure 12. Hydrography showing all major stream networks for the St. Marks Watershed of North Florida.

Secondary coverages

Secondary coverages consisted of those spatial data layers that were derived from primary or source coverages. Secondary coverages included: (1) a modified DEM, (2) stream networks, (3) imperviousness, (4) runoff coefficients, (5) aereal TP loads, (6) distance weights, and (7) net TP loads. Construction of the secondary coverages are described in the following sections.

Modified DEM coverage--To operate successfully, the overland flow algorithm required sufficient topographical resolution for a small gradient to exist across all grid cells. Florida's low relief terrain only serves to exacerbate this problem. USGS 7.5 minute DEM quadrangles for the St. Marks Watershed only had an accuracy to 1 meter, consequently, many neighboring cells would have the same elevation when in actuality there was a difference between them, albeit small. If the original DEM was used without modification, modeled loads that flowed downhill would halt upon encountering two or more grid cells of the same elevation even though a slope may be present. To compensate for these "artificially flat" regions, the watershed was placed upon a slight tilt whereby all grid cells would have a downhill neighbor. Only a very slight tilt was necessary to overcome these hydrological "sticking points". This was accomplished by adding a very small amount to all DEM cell values. The amount added to the DEM cells was directly proportional to the DEM cells' distance from the watershed outlet. A small number was added to cell elevations farther from the Gulf--nearer the headwaters--and a very small number was added to cells closer to the Gulf.

Stream network coverage--A stream network coverage was created from a hydrography coverage. As with the DEM, the hydrography coverage did not have sufficient detail to

reveal many of the minor tributaries to the St. Marks River. To compensate for this inadequacy in resolution, stream networks were generated using the overland flow technique where a precipitation map was drained over the modified DEM coverage creating an output map that identifies all stream paths. These stream paths were added to the hydrography coverage to obtain the final stream network coverage. To allow for flow down all stream channels, a similar procedure of sloping was used as described above for the watershed to ensure stream channel slope from the headwaters to the north. Stream cells closer to the Gulf were given lower elevations than locations farther inland. Exact elevations were not necessary, only that a gradient was present, and that the entire stream network had a lower elevation than the ground surface.

Imperviousness coverage--An imperviousness coverage was constructed based upon imperviousness data for different land uses (Table 3). The imperviousness data along with the land use map were used in creating the imperviousness coverage.

Table 3. Percent imperviousness for land uses within the St. Marks Watershed of North Florida

Land Use	Imperviousness (%)
Low-Dens. Res.	14.7
Single-Family	27.8
Multi-Family	67
Low-Intensity Comm.	91
High-Intensity Comm.	97.5
Industrial	86.8
Highway	85
Pasture	0
General Ag.	0
Rec./Open Space	1.5
Mining	23
Wetland/Open Water	0
Upland	2

Source: modified from Harper (1994)

Runoff coefficients coverage--The amount of runoff from any given cell is dependent upon that cell's soil type and imperviousness combination. Table 4 shows the runoff coefficients used for the soil/land use combinations.

Table 4. Runoff coefficients for their respective land use/soil combinations used within the model.

Land Use	Soils			
	A	B	C	D
Low Density Residential	0.25	0.3	0.35	0.4
Single Family	0.3	0.37	0.43	0.5
Multi-Family	0.5	0.57	0.63	0.7
Low-Intensity Commercial	0.6	0.7	0.8	0.9
High-Intensity Commercial	0.65	0.75	0.85	0.95
Industrial	0.6	0.7	0.8	0.9
Highway	0.95	0.95	0.95	0.95
Pasture	0.15	0.23	0.3	0.37
General Agriculture	0.14	0.16	0.2	0.24
Mining	0.2	0.3	0.4	0.5
Wetland	0.23	0.23	0.23	0.23
Silviculture/Upland Forest	0.1	0.17	0.23	0.3
Open Water	—	—	—	—
Recreational/Open Space	0.1	0.17	0.23	0.3

Source: modified from Adamus (1995).

Runoff coefficients for the various land uses were derived based upon the combination of two coverages--imperviousness and soil type--as shown in Figure 13a. Using the four possible hydrologic soil groups and fourteen land uses, fifty-six combinations were possible (Table 4). Where dual soil classes occurred (A/D, B/D, or C/D), similar assumptions were made as by Adamus (1995). The first hydrologic

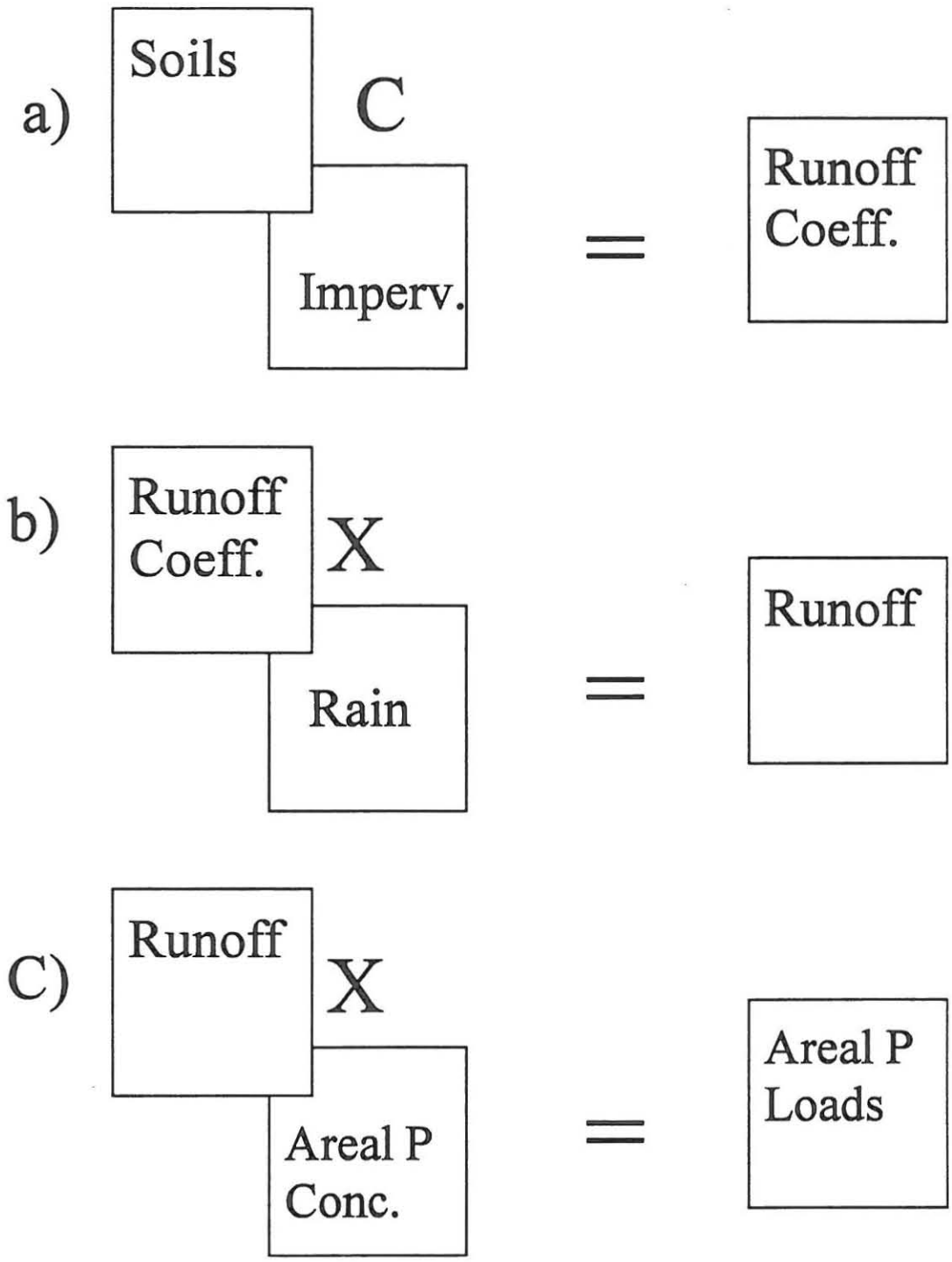


Figure 13. Map algebra procedure to generate several model output coverages. a) Runoff coefficients; b) Runoff; c) Total P loads. (C = combination of two maps, X = multiplication of two maps).

designation in each dual class represents drained conditions. For urban regions with dual soil classes, adequate drainage was assumed, while the D or poorly drained class was used for recreation/open space, pasture, or any undeveloped areas.

Runoff coverage--The runoff coverage was generated by multiplying the runoff coefficients map by a precipitation map (assumed to be constant over the entire basin) (shown in Figure 13b). Cell values within the runoff map contained the estimated volume of water that would leave that cell for total yearly precipitation.

Aereal TP load coverage--An aereal TP load coverage was derived using literature values (Harper, 1994) of TP concentrations assigned to their appropriate land uses (Table 5).

Table 5. TP concentrations for land uses used within the model.

Land Use	Total P (mg/l)
Low-Dens. Res.	0.177
Single Family	0.3
Multi-Family	0.49
Low-Intensity Comm.	0.15
High-Intensity Comm.	0.43
Industrial	0.31
Highway	0.34
Pasture	0.476
General Ag.	0.344
Rec./Open Space	0.053
Mining	0.15
Wetland	0.19
Upland	0.032
Open Water/Lake	0

Source: modified from Harper (1994)

The aereal phosphorus concentrations coverage (mg/l) was multiplied by the runoff coverage (liters) to produce a map of aereal phosphorus loads (mg) (Figure 13c).

Distance weighted coverage--Distance from water bodies was determined using a slope-modified distance algorithm in which steeper slopes had a shorter horizontal distance than lower grade terrains for any given cell. This "distance map" was created by accounting for all uphill cells above the receiving stream. Cell values within this distance map ranged from 0 at the stream edge to much higher numbers for cells that were a considerable distance from stream networks. This distance map was multiplied by an exponential decay coefficient to create a coverage of distance weighting. Shown below is the mathematical relationship used in developing this distance weighted coverage:

$$D_w = e^{(-k*d)} \qquad (1)$$

Where:

D_w = Weighted distance

k = Decay coefficient determined through model calibration (1/m)

d = The distance any cell is from the downhill receiving stream (m)

The decay coefficient (k) represents relative TP uptake by the landscape on it's path to receiving waters.

Net phosphorus load coverage--The distance weighted coverage, and the aereal TP load coverage were multiplied together to obtain a net TP load coverage that contained the TP load leaving any given cell. Depicted below is the equation for how this net phosphorus load coverage was calculated:

$$C = C_i * D_w \qquad (2)$$

Where:

$C =$ The net pollutant load that will leave a cell (kg/yr)

$C_i =$ The initial pollutant load in a cell before attenuation is taken into account (kg/yr)

$D_w =$ Weighted distance (see Equation 1)

The Spatial Phosphorus Model

To generate cumulative phosphorus loads for every cell in the watershed, the net phosphorus load coverage was drained over the modified DEM. Cell values in the cumulative phosphorus loads coverage contained the total amount of phosphorus passing through the cells in a year (kg/yr). A cumulative runoff coverage was also created by draining the runoff map over the modified DEM. All cells in the cumulative runoff coverage had runoff volumes that pass through those cells in a year (liters/yr). A spatial TP concentration map (mg/l) was generated by dividing the cumulative phosphorus loads coverage by the cumulative runoff coverage. The MapFactory algorithm for the spatial TP model is shown in Appendix B.

Model Comparison and Calibration

To see how well the overland flow algorithm predicted stream flow from runoff, model predicted stream flow was compared with measured stream flow at six locations within the watershed. The model was calibrated to TP data at twenty-four stations throughout the watershed. Model iterations were repeated for different decay factors until model predicted TP concentration results correlated closest with observed.

Comparing Model Results with Flow Data

Six sampling locations throughout the St. Marks Watershed were compared with modeled flow results (Table 6).

Table 6. Average stream flows for six monitoring stations used for comparison with modeled flows in the St. Marks Watershed of North Florida.

Station-ID	Station Name	Abb.	Time of Flow Record	Avg. Flow (cfs)
2326700	Lloyd Creek at Lloyd, FL	LC	1965-1971	14.31
2326838	NE DD at Miccosukee Rd, Tallahassee, FL	NEDD	1993-1995	10.1
2326900	St. Marks River nr Newport, FL	SMRN	1957-1994	718
2327000	Wakulla Sp. nr Crawfordville	WSC	1970-1976	470
301632084085201	St. Marks Sp. near Woodville	SMSW	1985	324
301910084175500	Munson Slough at eightmile sink nr Tall, FL	MS8	1976	13.1

cfs = cubic feet per second
Abb. = Abbreviation

These sites, referred to from now on by their abbreviations, are taken from the EPA's STORET database. Recorded flow data were relatively sparse within the watershed with only two sites (SMRN and WSC) having significant flow data on record. To examine variation in modeled flows, the model was run for a wet year (264 cm), dry year (117 cm), and average year's precipitation (165 cm). Model output was in the form of a cumulative runoff map from which flows at the six locations were extracted for comparison with observed flow data.

Calibrating Model Results with Water Quality Data

TP concentration data from twenty-four locations (shown in Figure 14) were compared with modeled TP concentration results. Data from these twenty-four sites are grab samples that were taken during the late 1980s and early 1990s and are published in the 1996 Florida Department of Environmental Protection 305(b) Main Report. TP levels at these sites along with the number of observations and time period over which the samples were taken are shown in Table 7. The summarized TP data are of varying quality, having often been taken as grab samples and where some locations had many observations, others are the result of only a few. Model output is in the form of a spatial TP concentration map that shows predicted TP concentrations for all streams within the St. Marks Watershed. The predicted TP concentrations for these twenty-four locations were extracted from the spatial concentration map and compared with the observed water quality data.

Numerous simulations of the model were repeated, varying the decay coefficient (k) and comparing model results with observed TP concentrations. The decay coefficients were selected that provided the highest correlation between model predicted and observed TP concentrations.

Landscape Development Intensity Indices

Five different measures of landscape development intensity (LDI) were developed to examine possible correlations with TP. They fell into two categories: physical and energy based LDIs. An average LDI was calculated for each index for the subbasins of the St. Marks Watershed.

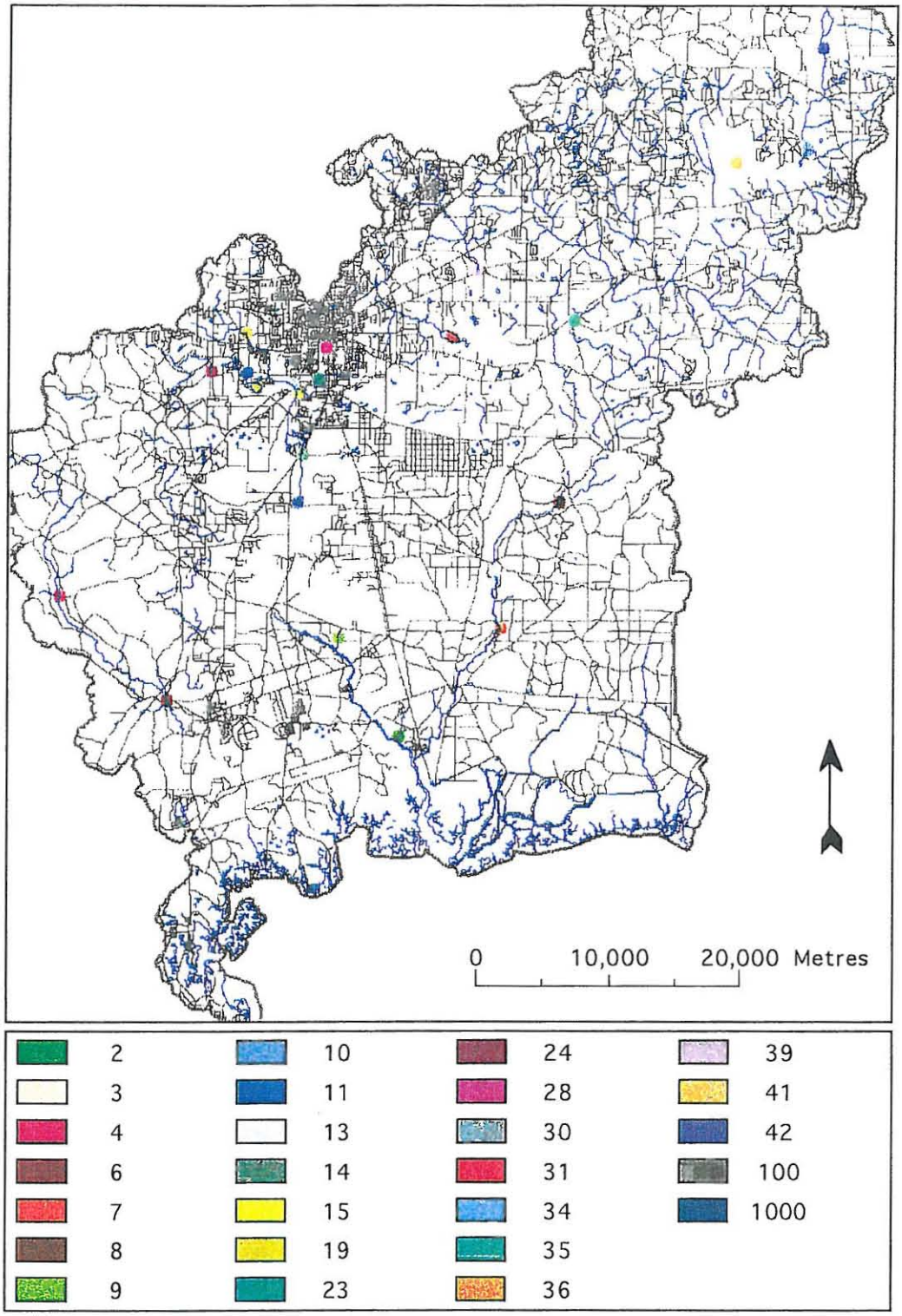


Figure 14. Water quality sampling locations used in model calibration. Numbers correspond to station ID numbers in Table 7.

Table 7. Average TP concentrations for twenty-four stations within the St. Marks Watershed of North Florida.

Station-ID	Station Name	Number of Observations	Time of Flow Record	Avg. TP (mg/l)
Water Body Type: Blackwater				
4	Unnamed Drain	3	1985	0.01
Water Body Type: Lake				
11	Clear Lake	6	1986	0.01
13	Lake Munson	891	1980-1987	0.35
19	Lake Bradford	8	1992	0.01
24	Bradford Brook	10	1985-1986	0.01
31	Lake Lafayette Drain	36	1992-1993	0.1
39	Alford Arm	16	1987	0.07
41	Lake Miccosukee	8	1992	0.05
Water Body Type: Spring				
8	St. Marks Spring	2	1985	0.05
Water Body Type: Stream				
2	Big Boggy Branch	8	1993	0.19
3	Wakulla River	24	1992-1994	0.02
6	Lost Creek	7	1993	0.01
7	St. Marks River	23	1993-1994	0.03
9	McBride Slough	13	1993	0.01
10	Munson Sink	15	1992-1993	0.13
14	Munson Slough (BL)	7	1993	0.09
15	Munson Slough (AL)	7	1993	0.12
23	East Drainage Ditch	20	1987	0.51
28	St. Augustine Branch	14	1988	0.87
30	Central Drainage Ditch	56	1987-1988	0.79
34	Lloyd Creek	6	1992	0.12
35	Copeland Sink Drain	5	1993	0.14
36	Godby Ditch	18	1987	0.33
42	Ward Creek	5	1992	0.07

BL = Below Lake, AL = Above Lake

Source: Florida Department of Environmental Protection 305(b) Main Report (1996)

Physical LDIs

Two physical LDIs were tested for the St. Marks Watershed; imperviousness and weighted land use. Past studies have shown that imperviousness has a strong correlation with decreasing watershed health (Schueler, 1994). Literature determined imperviousness values (Corbitt, 1990; Harper, 1994) were assigned to each land use type (see Table 3) to generate an imperviousness map (%).

The weighted land use coverage was developed by aggregating the land use map into three categories: urban, agriculture, and undeveloped. Weights for each of these categories were based upon average TP concentrations and runoff curve numbers. Calculations from Table 5 showed that urban uses have an average TP concentration of about 0.3 mg/l, agriculture about 0.4 mg/l, and undeveloped 0.11 mg/l. Average runoff curve numbers for urban, agriculture, and undeveloped were 0.60, 0.20, and 0.20 respectively (see Table 4). Multiplying each categories average TP concentration by its runoff curve number, a ratio of approximately 9 to 4 to 1 was found for urban to agriculture to undeveloped. Based upon these calculations, it was decided to give urban a weight of 9, and agriculture a weight of 4. Undeveloped was left at 0 to better distinguish background from human induced affects upon the landscape.

Energy Based LDIs

Indices of landscape development intensity based upon total energy flow expressed as emergy (spelled with an m) were also generated for subbasins of the St. Marks Watershed. These LDIs were total empower density, developed empower density,

and an environmental loading ratio (ELR). Similar to the physical LDIs, an average for each energy index was calculated for all subbasins on a per hectare basis.

Emergy is a measure of the work required to generate a product or service, and accounts for both environmental and economic inputs to a system. Emergy is expressed in units of the same form of energy, most often solar emergy (units are solar emjoules, abbreviated sej). Empower density is emergy per area per time (sej/ha/yr). Empower density can be interpreted as a measure of the aereal work per time. Transformity is a measure of energy quality and is a ratio of the emergy required to make something to its energy content (units are sej/J). The environmental loading ratio is the ratio of nonrenewable to free, renewable inputs upon a system, and therefore a possible indicator of environmental impact (Brown and Ulgiati, 1997; Ulgiati et al., 1996).

To compute empower density and the ELR, the dominant renewable and nonrenewable resources were determined for all subbasins. The renewables included sunlight inputs, together with the geopotential and chemical potentials of rain. Wind energies were found to be negligible and not considered. All renewable and nonrenewable energy sources were taken as flows on a per year basis. All of the following described emergy calculations are listed in detail in Appendix C.

Sunlight was assumed a constant across the watershed. The amount of sunlight that falls upon one hectare (one grid cell) of land was calculated and applied to the entire coverage. A coverage of sunlight energy was generated by assuming an albedo of 10% of solar inputs.

The rain geopotential is a measure of the land forming energies within rain that do the work of basin formation and sediment transport. The rain geopotential energy is a

measure of the elevational potential within a volume of water above a reference datum. Since the design of the LDI-TP spatial analyses was to compare LDIs with TP at the subbasin level, geopotential was calculated for subbasin cells relative to their subbasin's outlet, rather than for the entire St. Marks Watershed relative to the Gulf. The geopotential energies were measured as any subbasin cell's elevation above that subbasin's outlet elevation times the runoff volume for that cell. This coverage was then multiplied by the global rain geopotential transformity to generate a map of rain geopotential emergy.

The rain chemical potential is a measure of the purity of water relative to seawater. In energy terms, Gibbs free energy is the measure used to rate this purity. Generally, the majority of rainfall's purity is exploited through transpiration processes within plants. The rain chemical potential energy was computed as the percentage of water loss due to transpiration from any cell. To determine transpirational losses for the different land uses, land uses were grouped into low intensity development, high intensity development, and agriculture/undeveloped. Transpiration values as a percentage of rainfall of 50%, 5%, and 60% were assigned to each of these categories, respectively. The global rain chemical transformity was then multiplied by the map of transpiration volume to generate a map of rainfall chemical potential emergy.

The dominant nonrenewables within the St. Marks Watershed were topsoil loss, electricity, agriculture fertilizers, and fuel use. Topsoil loss was dependent upon the land use type. Values in grams per hectare per year and converted to emergy are listed in Appendix C. A map showing the emergy of topsoil loss for all land uses was then produced.

Direct energies consisted of electricity utilized by the land uses. Whitfield (1993) determined average electricity in kilowatt hours for various land uses. Multiplying the electricity energies by the transformity of electricity, a coverage of direct emergies was created.

Agriculture was divided into improved pasture and corn and empower densities were taken from calculations made by Brandt-Williams (1998). The dominant nonrenewable inputs to agriculture included electricity, potash, lime, phosphate, and nitrogen.

It was assumed that automotive gasoline consumption comprised virtually all non-electric fuel uses within the watershed. Fuel use calculations were divided into three components; Tallahassee, Interstate 10, and all rural roads within the county.

Tallahassee fuel uses were determined for each of the four major land uses within the city--low density residential, single family, multi-family, and commercial/industrial. Total fuel use per year for the city was calculated by multiplying fuel use per capita (477 gal/capita) (Florida Statistical Abstract (FSA), 1995) by the Tallahassee population (137,057 people). This total was then divided between each of the four land uses based upon their estimated percentage consumption of the total fuels.

The major traffic artery through the St. Marks Watershed is Interstate 10. Fuel use along the interstate portion that passed through the watershed was determined from Department of Transportation car counts (44,045 vehicles/day) and average miles per gallon per vehicle (22 mpg). To estimate fuel use on rural roads, the rural population was approximated by subtracting the Tallahassee population from the remainder of the watershed populace. Total rural fuel use was then evaluated from the average fuel use

per capita, rural population (83,737 people), and length of all rural roads (4340 km). To determine total fuel use for the watershed on an emergy basis, spatial maps of these three major fuel use sectors were summed and then multiplied by the transformity of gasoline.

Once all of the renewable and nonrenewable emergy maps were created, empower density and the ELR could be determined. The coverage of total empower density was calculated by adding the largest of the renewable emergy maps--to avoid double counting--and all of the nonrenewable emergy maps together as shown in equation 3:

$$\text{Empower Density} = (\text{RA,chem} + \text{RA, geo} + \text{EL} + \text{E} + \text{Ag.} + \text{FU}) \qquad (3)$$

Where:

Empower Dens. = Total emergy flow per area per time (sej/ha/yr)

RA,chem = Chemical potential emergy of rain

RA, geo = Geopotential emergy of rain

EL = Earth loss

E = Electricity use

Ag. = Agriculture energies

FU = Fuel use

Since the nonrenewables are based upon different sources over different time scales, they could be added. Two empower density LDIs were calculated, one of total empower density and one consisting of only the nonrenewable portions (developed empower density).

The empower density coverage portrayed all major energy flows within the watershed. Each cell within the empower density coverage would have a distinct emergy flow associated with it depending upon its location within the watershed.

The ELR coverage was computed by dividing the sum of the nonrenewable emergy maps by the largest of the renewable emergy maps as shown in equation 4:

$$ELR = \frac{(EL + E + Ag. + FU)}{(RA, chem + RA, geo)} \qquad (4)$$

Where:

ELR = Environmental Loading Ratio (unitless)

Refer to equation 3 for the definition of the symbols used. The value within each cell in the ELR coverage portrays environmental loading at that location. Greater values would signify a higher environmental loading.

Spatial Correlations

Once the five LDIs were generated, spatial correlations between the LDIs and TP could be explored. First, the appropriate decay coefficients were selected based upon comparison of modeled and observed data. These decay coefficients would establish an effective distance from the stream network where LDIs would have the greatest effect upon the stream. The effective distance where greater than 10% of the TP load was expected to reach streams was selected out for the LDI-TP analysis. Average LDIs were determined for the portions of all subbasins that fell within this zone by summing the LDI for each cell within the subbasin portion and dividing by the total number of cells within the portion. Similarly, model predicted net TP loads were summed and divided by the number of cells within the subbasin portion to determine its average TP load in kilograms per hectare per year. The LDI for each subbasin was then correlated with it's corresponding average TP loads.

RESULTS

Spatial Coverages

Seven secondary coverages were generated as part of the spatial model: modified DEM, stream networks, imperviousness, runoff coefficients, runoff, aereal TP loads, distance weighted, and net TP load (Appendix D).

Comparison of Modeled and Observed Stream Flow Data

To investigate how well the model predicted stream flow, the model was run for a wet year, a dry year, and an average year of precipitation and then compared with stream flow at six stations (see Table 6). Observed and model predicted stream flows are shown in Figure 15. The observed flows are averages for a variety of years. The data used to compute averages were not available so range bars are not shown. The upper and lower range bars on the modeled stream flows are for wet and dry years, respectively. Modeled average year stream flows when compared with observed stream flows were divided equally with predictions greater than observed flows for two stations (MS8, LC), approximately equal for two (NEDD, SMSW), and below average for two (WSC, SMRN). Note that for the station on the St. Marks River with the longest record of flow data (SMRN), the model predicted stream flows that were 55% of observed.

Calibration of Modeled and Observed Water Quality Data

Distance decay coefficients weighted for northern and southern locations were used to compare model output with observed water quality data. Table 8 presents model

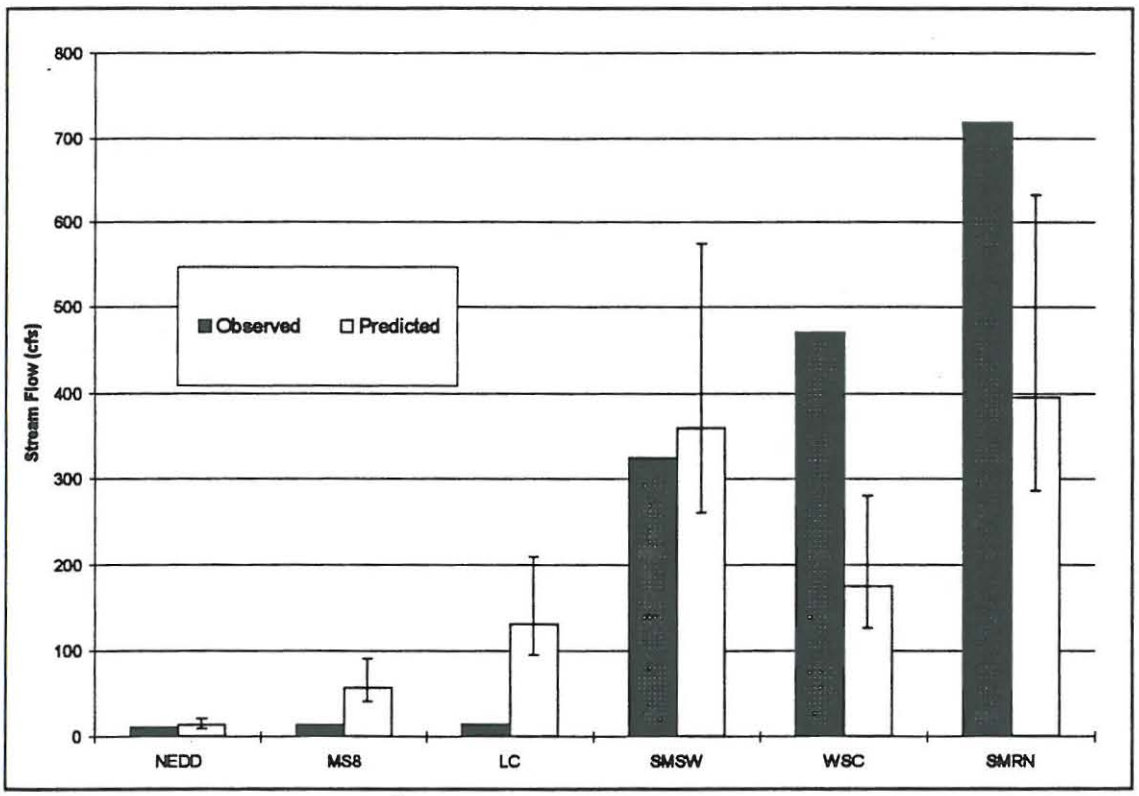


Figure 15. Observed and model predicted stream flows in cubic feet per second (cfs) for six stations in the St. Marks Watershed of North Florida. Range bars are for rainfall years of 117 cm, 165 cm, and 264 cm. Station abbreviations are identified in Table 6.

Table 8. Spatial TP model results compared with observed results for four decay coefficient weights. Model results are also shown for the case where no distance weight is given (0,0).

Location	Stations	Observed TP (mg/l)	PREDICTED (Decay Coefficients for North, South) (0,0) (mg/l)	(-0.03,-0.05) (mg/l)	(-0.2, -0.3) (mg/l)	(-0.2,-0.4) (mg/l)
North of Escarpment						
	Bradford Brook	0.01	0.19	0.163	0.069	0.069
	CLear Lake	0.01	0.336	0.307	0.185	0.185
	Lake Bradford	0.01	0.49	0.434	0.22	0.22
	Lake Miccosukee	0.05	0.209	0.166	0.07	0.07
	Ward Creek	0.07	0.336	0.136	0.021	0.021
	Alford Arm	0.07	0.475	0.461	0.389	0.389
	Munson Slough (BL)	0.09	0.49	0.381	0.222	0.201
	Lake Lafayette Drain	0.1	0.384	0.252	0.155	0.155
	Lloyd Creek	0.12	0.375	0.344	0.215	0.215
	Munson Slough (AL)	0.12	0.49	0.475	0.401	0.401
	Copeland Sink Drain	0.14	0.475	0.374	0.096	0.096
	Godby Ditch	0.33	0.43	0.417	0.352	0.352
	Lake Munson	0.35	0.426	0.26	0.14	0.127
	East Drainage Ditch	0.51	0.49	0.475	0.401	0.401
	Central Drainage Ditch	0.79	0.49	0.475	0.401	0.401
	St. Augustine Branch	0.87	0.49	0.49	0.49	0.49
South of Escarpment						
	Lost Creek	0.01	0.19	0.133	0.023	0.011
	McBride Slough	0.01	0.19	0.163	0.077	0.057
	Unnamed Drain	0.01	0.19	0.18	0.14	0.127
	Wakulla River	0.02	0.299	0.191	0.14	0.127
	St. Marks River	0.03	0.194	0.154	0.079	0.067
	St. Marks Spring	0.05	0.208	0.18	0.14	0.127
	Munson Sink	0.13	0.374	0.285	0.222	0.201
	Big Boggy Branch	0.19	0.215	0.145	0.073	0.064

BL = Below Lake, AL = Above Lake

Note: Coefficient weights are explained in text

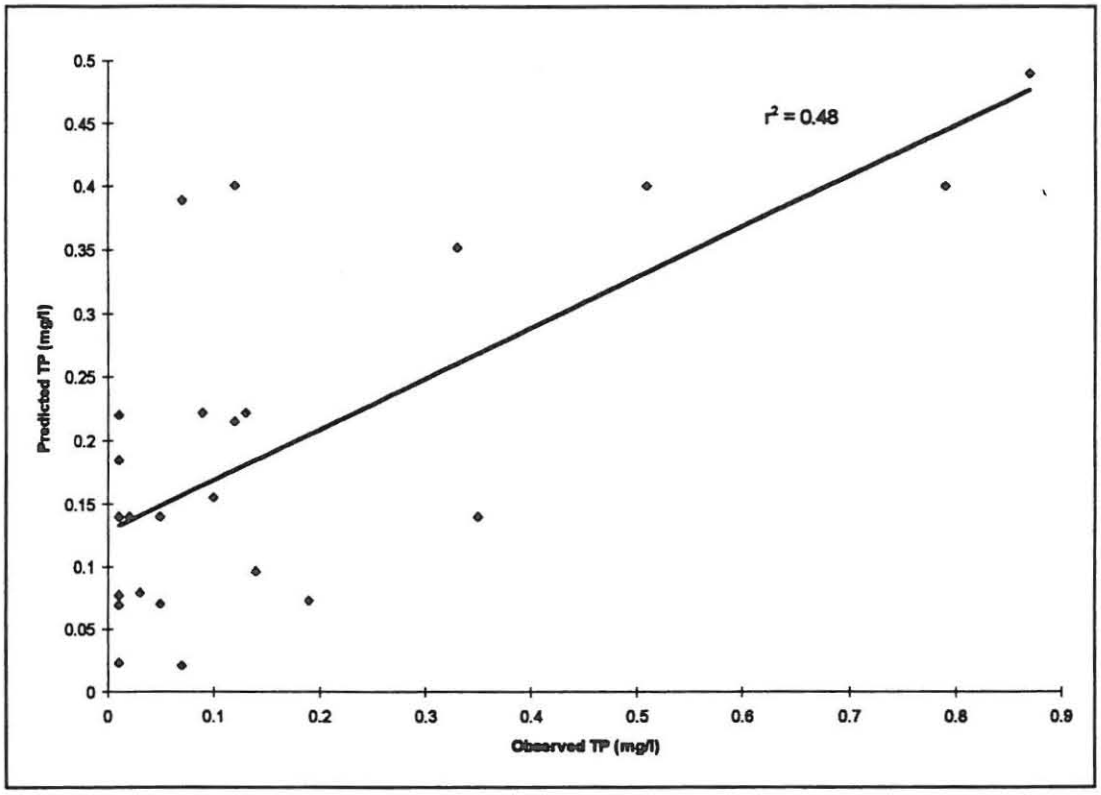


Figure 16. Predicted versus observed TP concentrations for the decay coefficients (-0.2, -0.3).

results for several of these decay coefficients. The first column of model predicted TP concentrations is for a case where no distance decay is taken into account (0,0). Generally, the model predicted TP concentrations decline as decay coefficients become more negative (Table 8). Decay coefficients of weight (-0.2,-0.3) were found to have the best fit to observed data ($r^2 = 0.48$) (Figure 16).

Model output with the decay coefficient weights (-0.2, -0.3) was compared with observed TP measurements for locations north and south of the escarpment (Figure 17). Range bars on the observed data represent the lower and upper range of TP concentrations measured at those sites. North of the escarpment (Figure 17a), the largest predicted and observed TP concentrations were St. Augustine Branch with concentrations of 0.49 mg/l and 0.87 mg/l respectively. The lowest predicted TP concentrations were for Ward Creek (0.02 mg/l) and Lake Miccosukee (0.07 mg/l)

South of the escarpment the model consistently predicted higher TP concentrations than observed TP concentrations for all stations except Big Boggy Branch (Figure 17b). The largest predicted TP concentration was at Munson Sink (0.22 mg/l). Lost Creek had the lowest predicted TP concentration (0.01 mg/l).

Spatial Simulation Results

The spatial coverage that weighted TP's influence upon the stream networks with the decay coefficients (-0.2, -0.3) is displayed in Figure 18. The gradation from red to yellow to green represents decreasing affect upon the streams, shown in blue. The green areas throughout the watershed have little if any direct effect upon the stream system.

Percentage of TP load delivered to streams decayed faster with distance from streams for locations south of the escarpment than for the north (Figure 19). South of the

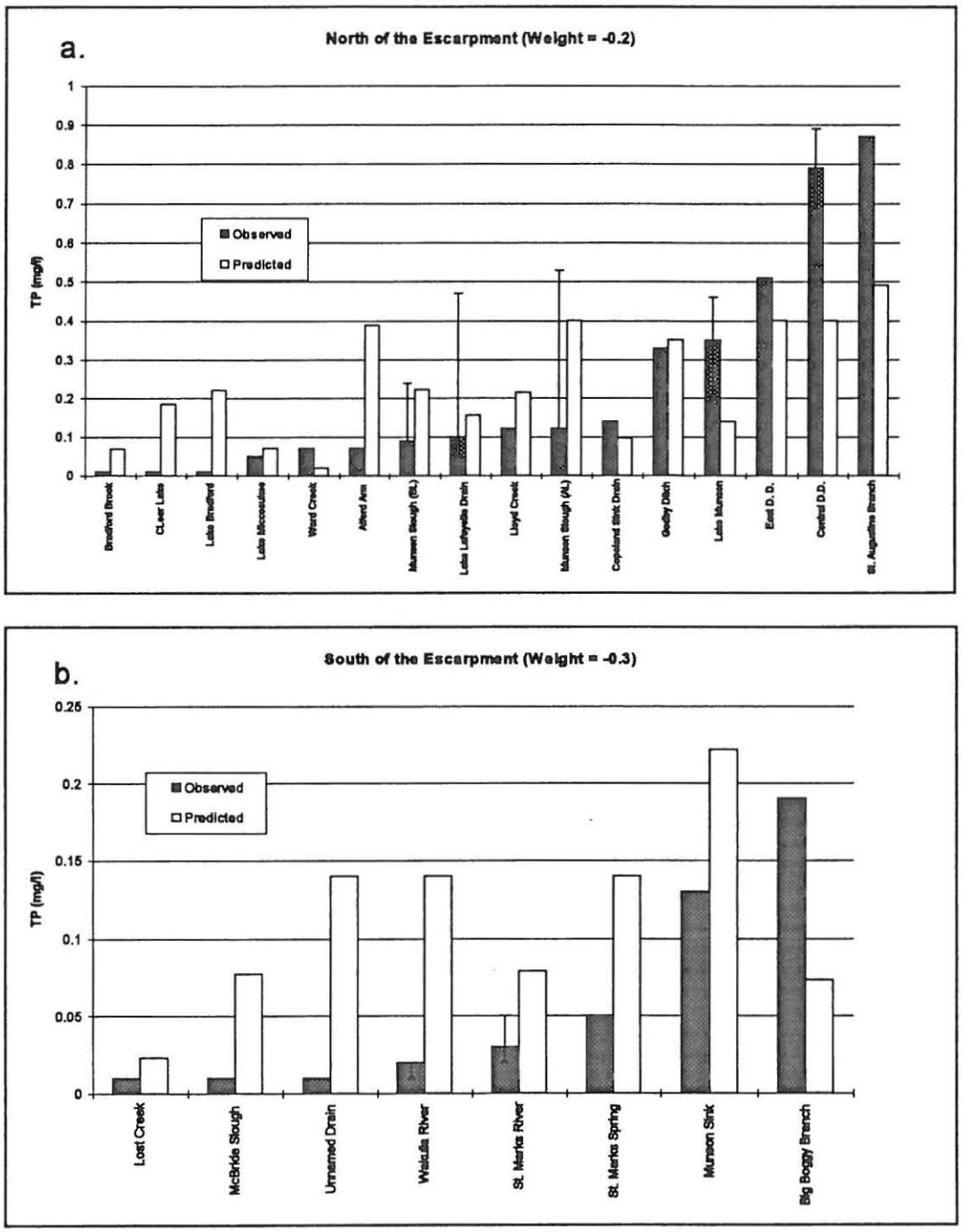


Figure 17. Graphs of the modeled and observed TP concentrations. a). Locations north of the Cody Escarpment (decay coefficient of -0.2); b). Locations south of the Escarpment (decay coefficient of -0.3). Range bars on observed are minimum and maximum TP concentrations recorded.

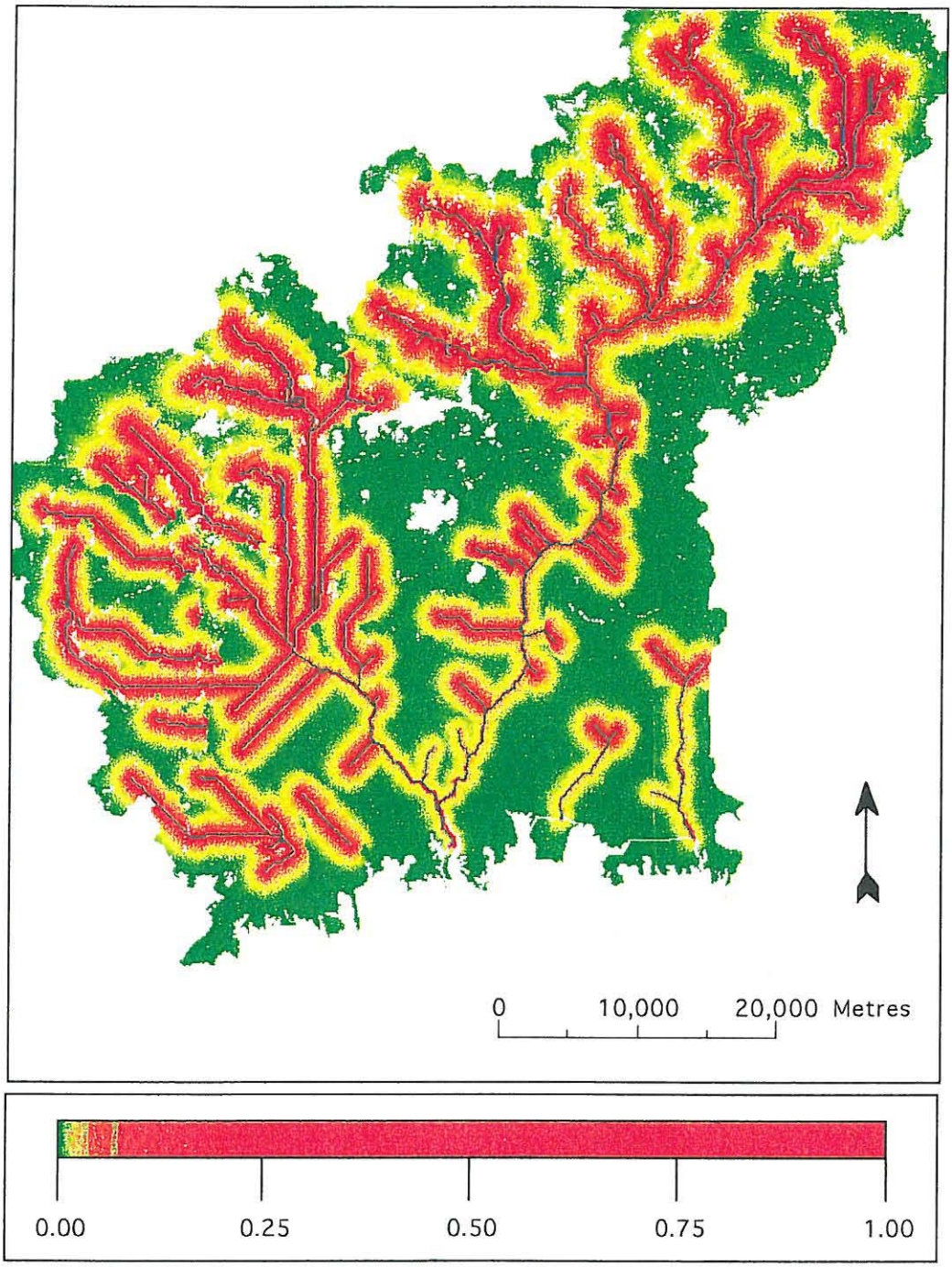


Figure 18. Spatial coverage showing the distance influence upon TP delivery to stream networks in the St. Marks Watershed of North Florida. Gradation from red to yellow to green indicates lessening percentage of TP delivered to the stream. The large areas of green have little affect upon the stream system. (1=100% influence, 0=0% influence).

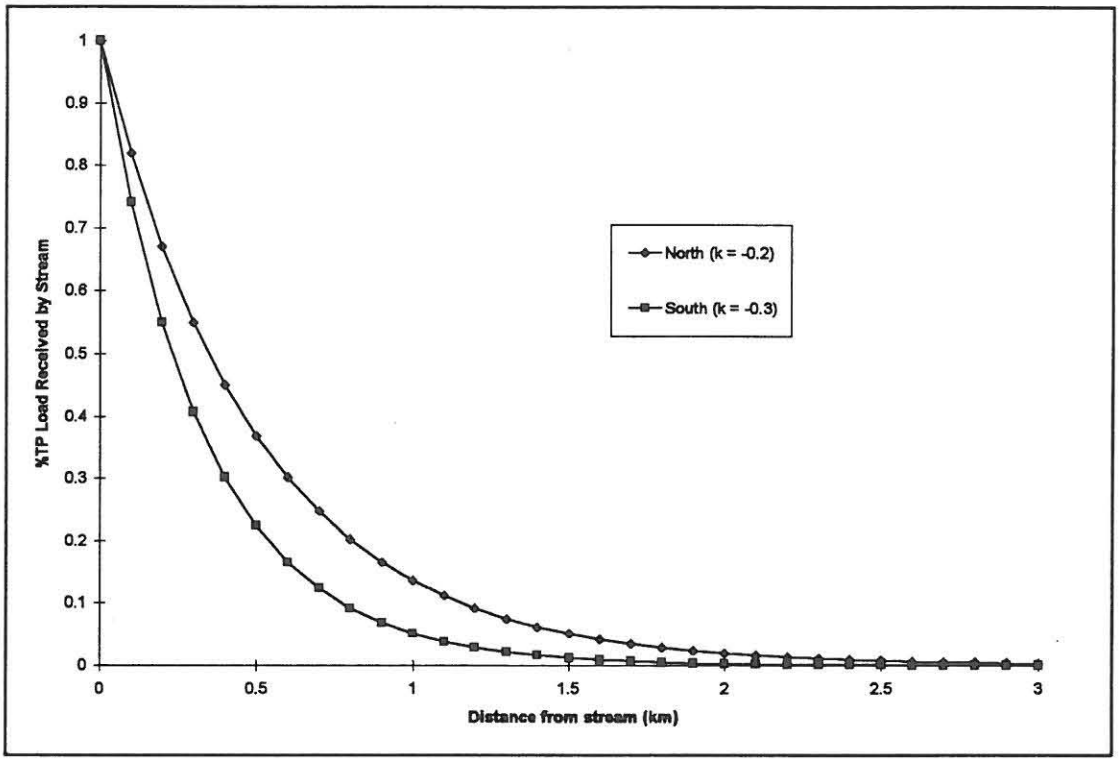


Figure 19. Graphical representation of the distance decay affect away from streams for the decay coefficients (-0.2, -0.3) at locations north and south of the Cody Escarpment. The stream receives a greater percentage of TP from locations closer to the stream.

escarpment, 50% of the TP load is received by the stream in the first 0.25 km, while north of the escarpment, 0.40 km distance from the stream is required to deliver the same load. The location where only 10% of the TP load reaches the stream occurs at about 0.80 km in the south, and 1.10 km in the north. The amount of TP load reaching the stream is approximately 0 at 2.50 km in the north and 1.70 km in the south.

The cumulative TP load map output for the decay coefficients (-0.2, -0.3) is shown in Figure 20. Each cell contains a TP flux (kg/yr). A dendritic pattern is observed in the upper half of the watershed where greater relief occurs. The southern half of the watershed tends to converge TP loads in a more linear fashion, reflecting the low relief of the landscape. TP loads of 11,570 kg/yr and 6,351 kg/yr are shown for selected locations on the St. Marks and Wakulla Rivers, respectively.

The cumulative TP load map (Figure 20) divided by a cumulative runoff map generated a TP concentration map (Figure 21). Values range from 0 mg/l for a large percentage of the watershed (light grey) to 0.49 mg/l for the highest TP concentrations (dark blue). Locations greater than background TP concentrations (0.01 mg/l) are generally found in streams and lakes.

Spatial Correlations of Water Quality with LDIs

Correlations between modeled TP loads and five LDIs within the St. Marks Watershed were carried out in stream zones predicted by the decay coefficients to contribute greater than 10% TP load to streams. This distance varied depending upon the location within the watershed. Although there are sixty-four subbasins within the St. Marks Watershed, not all subbasins fell within the stream zone of greatest effect, consequently less than sixty-four subbasins are plotted on the LDI-TP graphs.

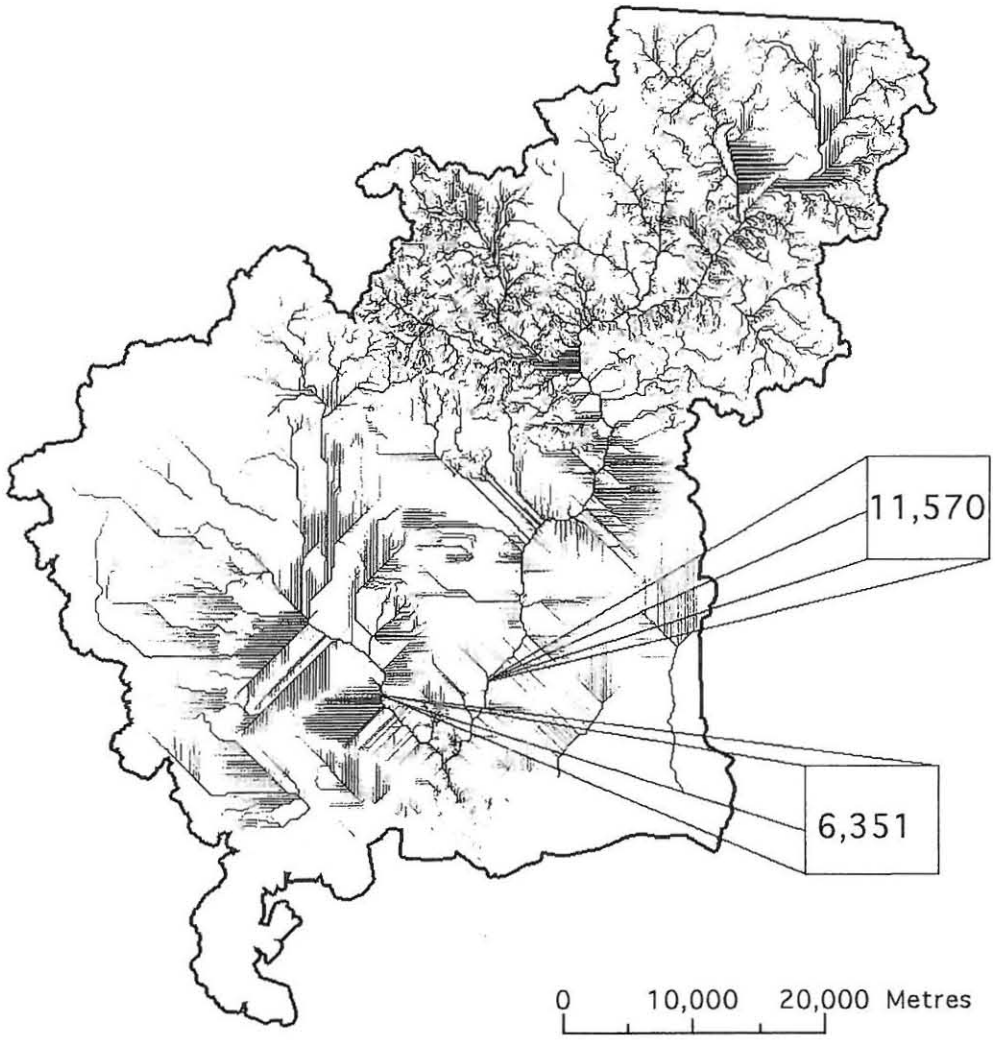
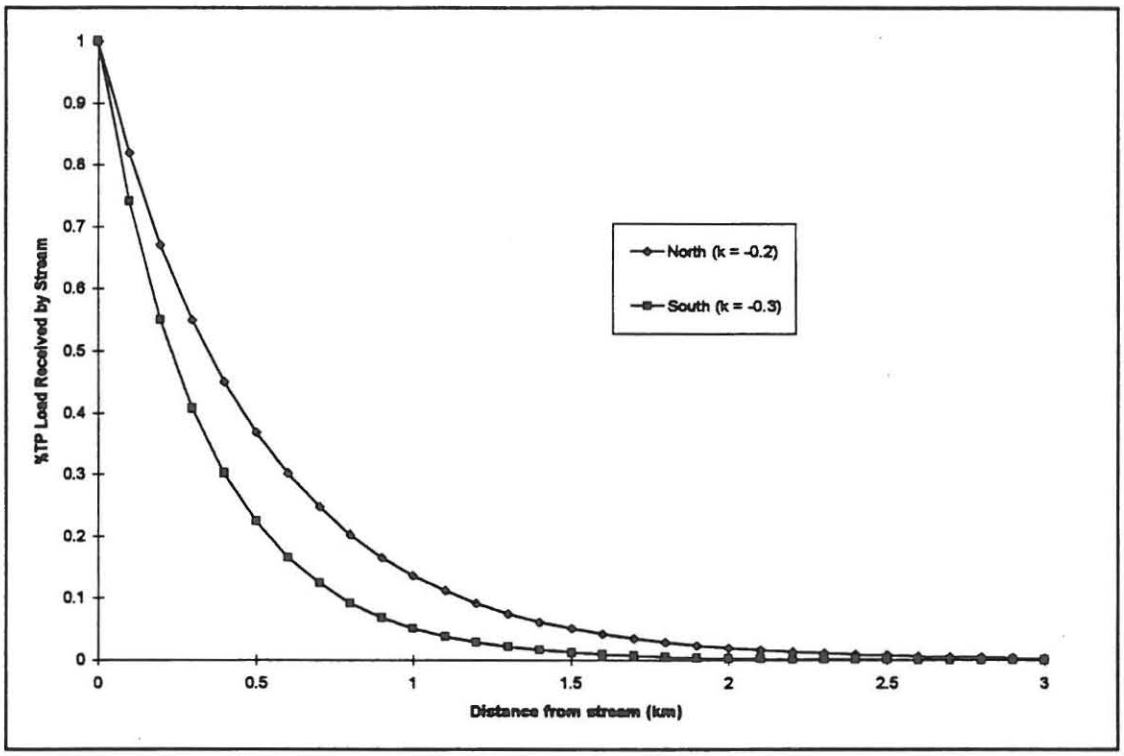


Figure 20. Model output showing cumulative TP loads (kg/yr) within each one hectare raster cell. TP loads at a location on the St. Marks River and the Wakulla River are 11,570 and 6,351 kg/yr respectively.



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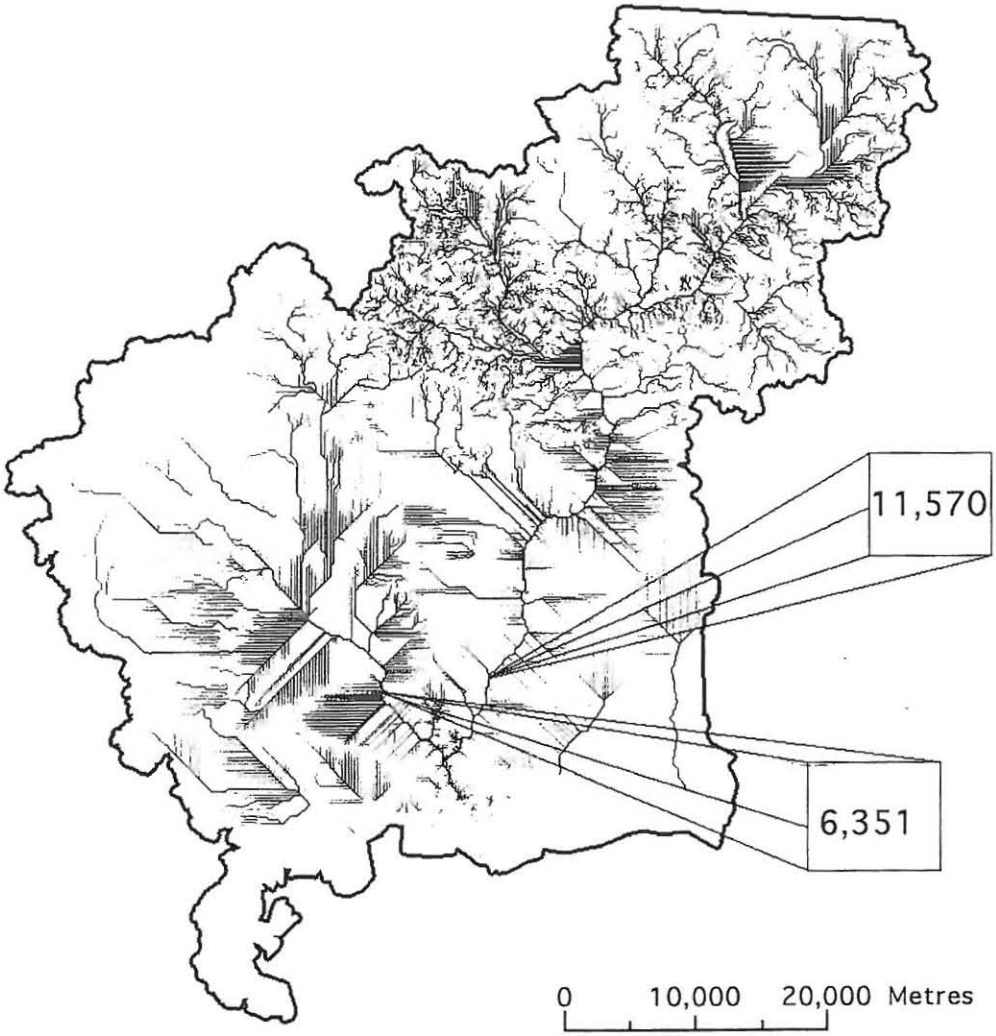

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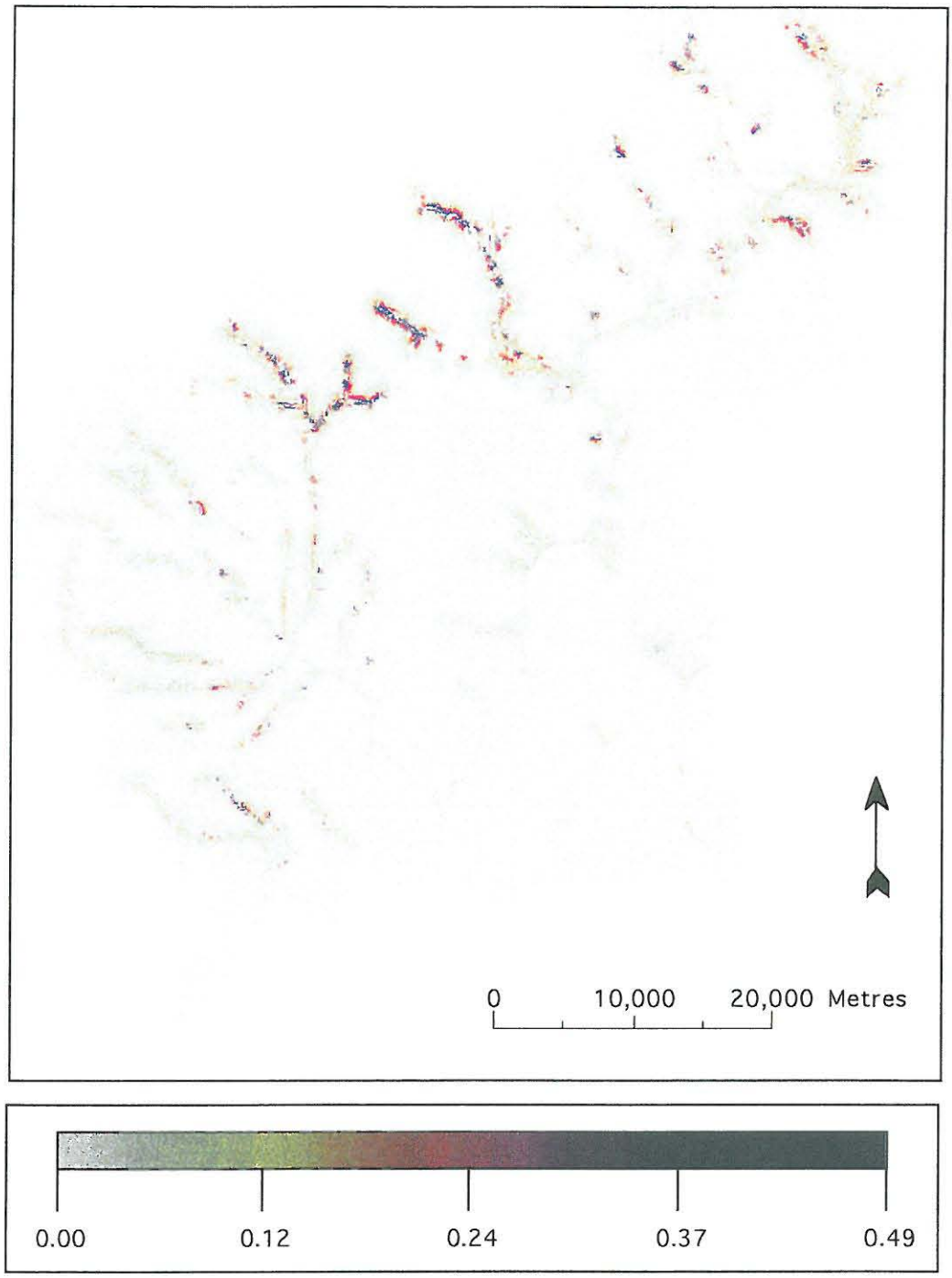


Figure 21. TP concentration map (mg/l) for the St. Marks Watershed of North Florida.

Physical LDIs

Imperviousness index

Imperviousness LDI has values ranging from 0% to 98% imperviousness (Figure 22). Darker cells are locations with greater imperviousness. The darkest regions is central Tallahassee. The majority of the watershed is a light gray averaging around 2% imperviousness.

TP loads increased for subbasins of greater than 10% imperviousness (Figure 23) Each point is a subbasin. Many rural subbasins of low imperviousness (0-10%) are clumped with TP loads ranging from 0 to about 0.4 kg/ha/yr. Above 10% imperviousness, TP values show a linear increase. Between 10% and 55% imperviousness TP loads are about 0.6 to 0.85 kg/ha/yr. Three subbasins have the greatest TP loads of 1.4 to 1.8 kg/ha/yr and have imperviousness of between 60 and 80%. A best fit linear regression of imperviousness and TP produced an $r^2 = 0.74$.

Weighted land use index

The weighted land use map with the three weights used; urban = 9, agriculture = 4, and undeveloped = 0 is shown in Figure 24. Urban is shown in black, agriculture in gray, and undeveloped in white. The majority of undeveloped land (including silviculture) is located in the southern half of the watershed. The largely agricultural northeast is seen by the numerous gray patches. Tallahassee and smaller communities south of it constituted the urban domain.

TP loads showed an increase for weighted land uses greater than 3.5 (Figure 25). Subbasin LDIs in the range of 0 to approximately 2 are clumped at TP values of from 0 to 0.4 kg/ha/yr. Approximately half of the subbasins on the graph fall within this 0 to 2

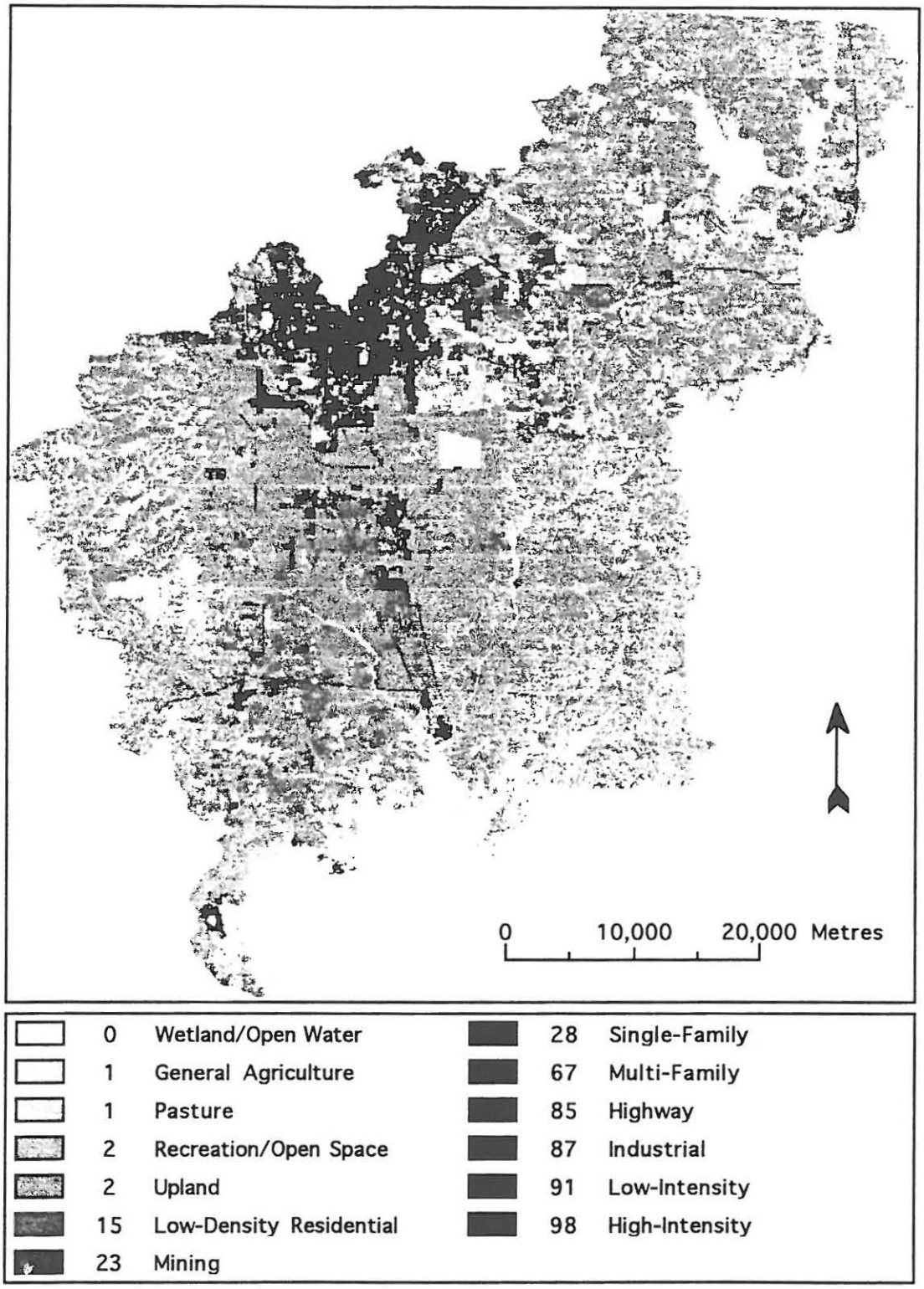


Figure 22. The imperviousness LDI. Darker cells portray greater imperviousness (%).

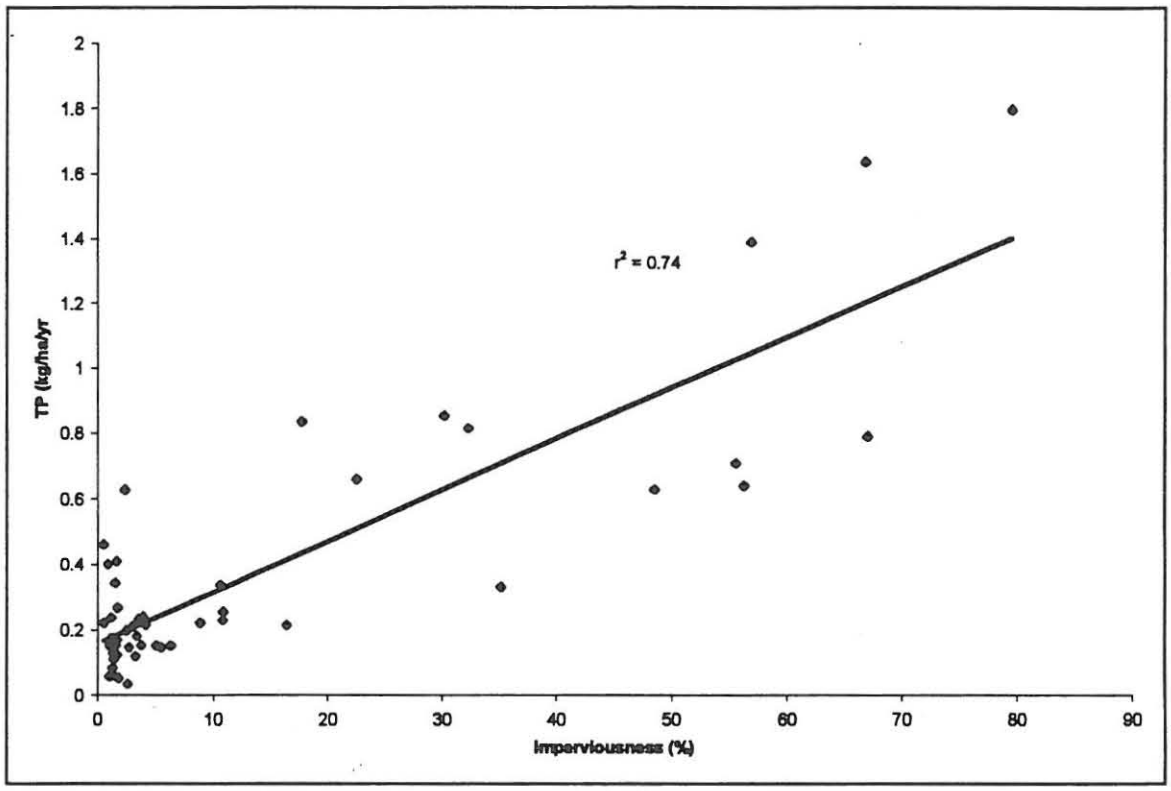


Figure 23. Percent imperviousness versus TP loads for subbasins of the St. Marks Watershed of North Florida.

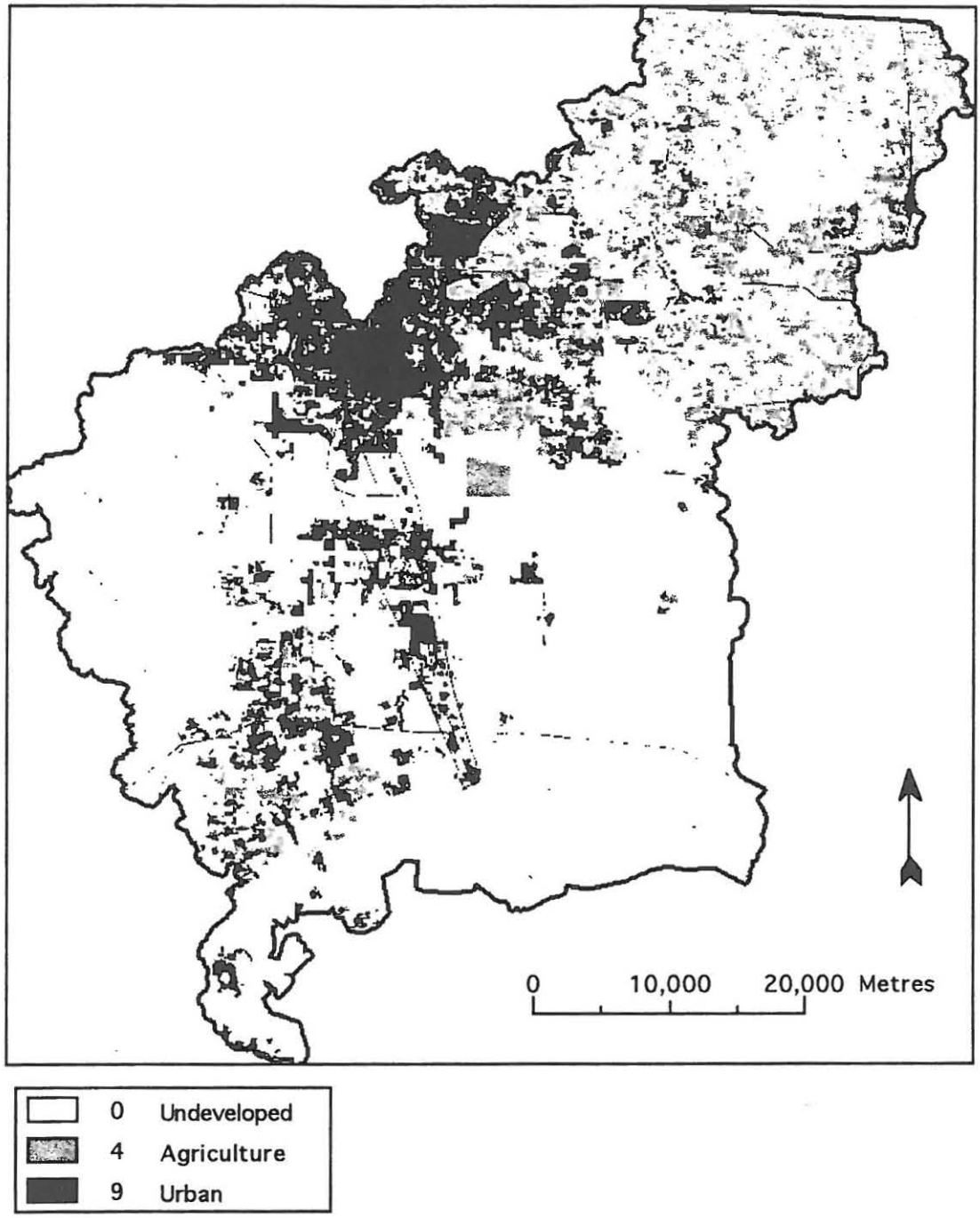


Figure 24. The weighted land use LDI. Weights used were urban = 9, agriculture = 4, and undeveloped = 0.

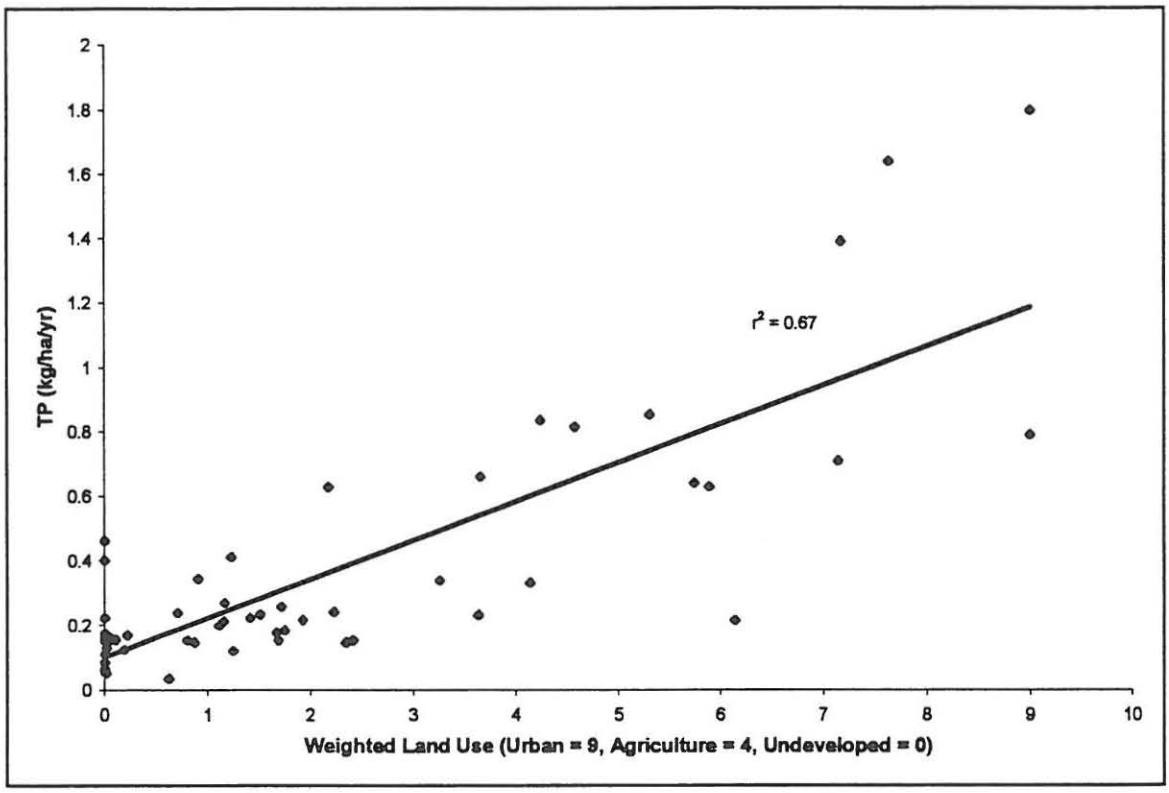


Figure 25. Weighted land use LDI versus TP loads for the subbasins of the St. Marks Watershed.

range. For LDIs ranging from 3.5 to 6, TP loads increased from 0.2 to 0.8 kg/ha/yr. The highest TP loads occurred at LDI values greater than 7. A linear correlation of $r^2 = 0.67$ was attained for this data set.

Energy Based LDIs

Empower density indices

Darker red cells in the total empower density coverage denote greater emergy flows (Figure 26). The largest empower density occurs in the urban regions in and around Tallahassee (250E15 sej/ha/yr to 1000E15 sej/ha/yr), where nonrenewable energy flows like electricity and fuel use dominate. South of Tallahassee are areas of rural development characterized by empower densities of 80E15 sej/ha/yr to 200E15 sej/ha/yr. The interstate east of Tallahassee and local roads also stand out in color from the lighter undeveloped landscape. Subtle color changes can be seen in the landscape where vertical relief affects the geopotential emergy of rainfall, becoming yellower at higher elevations. Values in these undeveloped regions range from about 15E15 sej/ha/yr at the summits of the Tallahassee Hills to approximately 1E15 sej/ha/yr near the gulf. These changes can also be observed by the yellow to green color transition from the north to south.

Total empower density was plotted on a log and a linear scale versus TP loads (Figure 27). TP loads do not begin to increase noticeably until empower density reaches 50E15 sej/ha/yr. Below 50E15 sej/ha/yr TP loads average about 0.2 kg/ha/yr. Above 50E15 sej/ha/yr a gradual trend is noticed between empower density and TP, becoming stronger as empower density increases. Above 100E15 sej/ha/yr the pattern is sharply vertical with several subbasins having TP loads of 1.4 to 1.8 kg/ha/yr. The total empower density LDI had a linear regression coefficient of $r^2 = 0.32$.

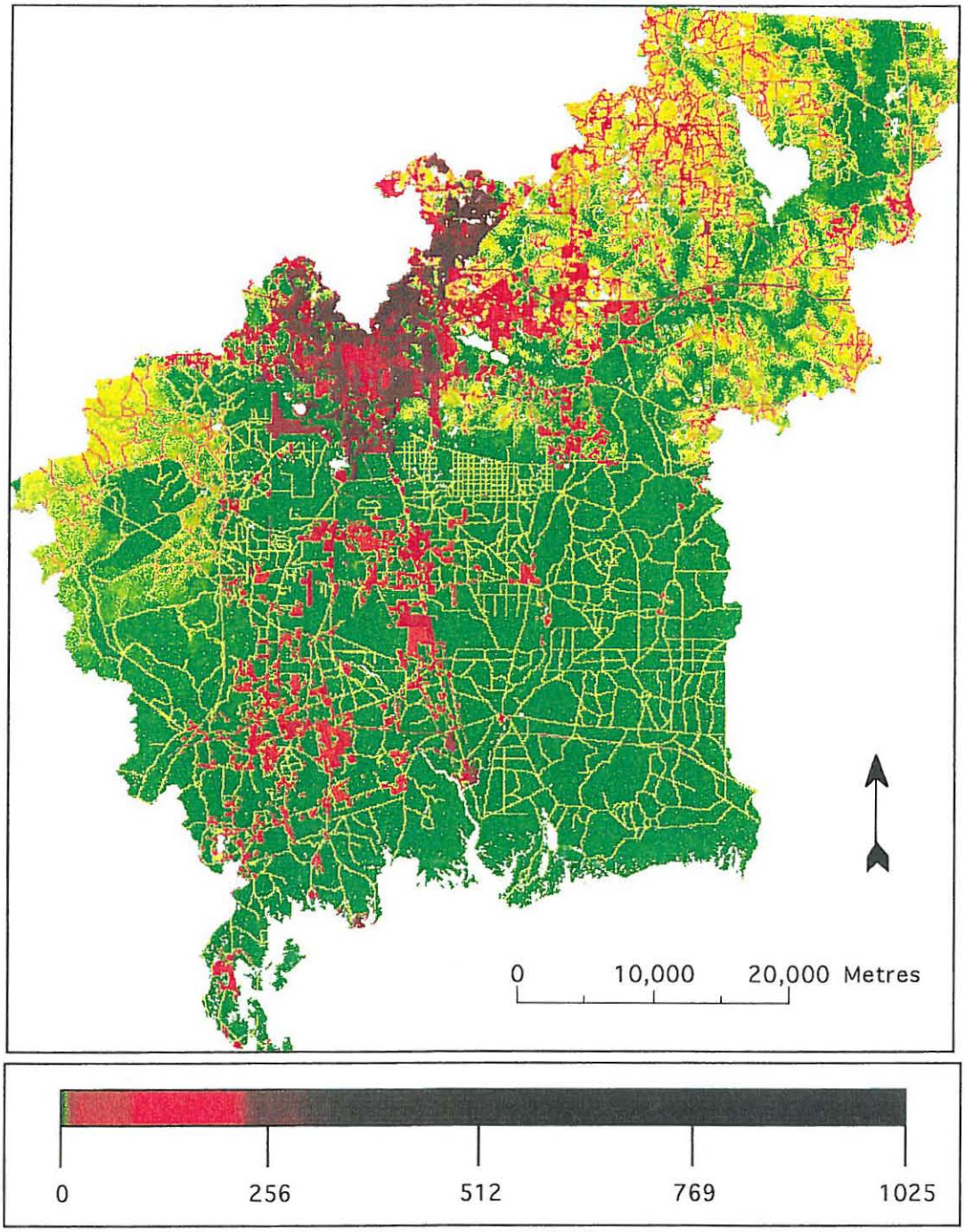


Figure 26. Total empower density LDI (E+15 sej/ha/yr) for the St. Marks Watershed of North Florida. Green to yellow to red denotes increasing empower density.

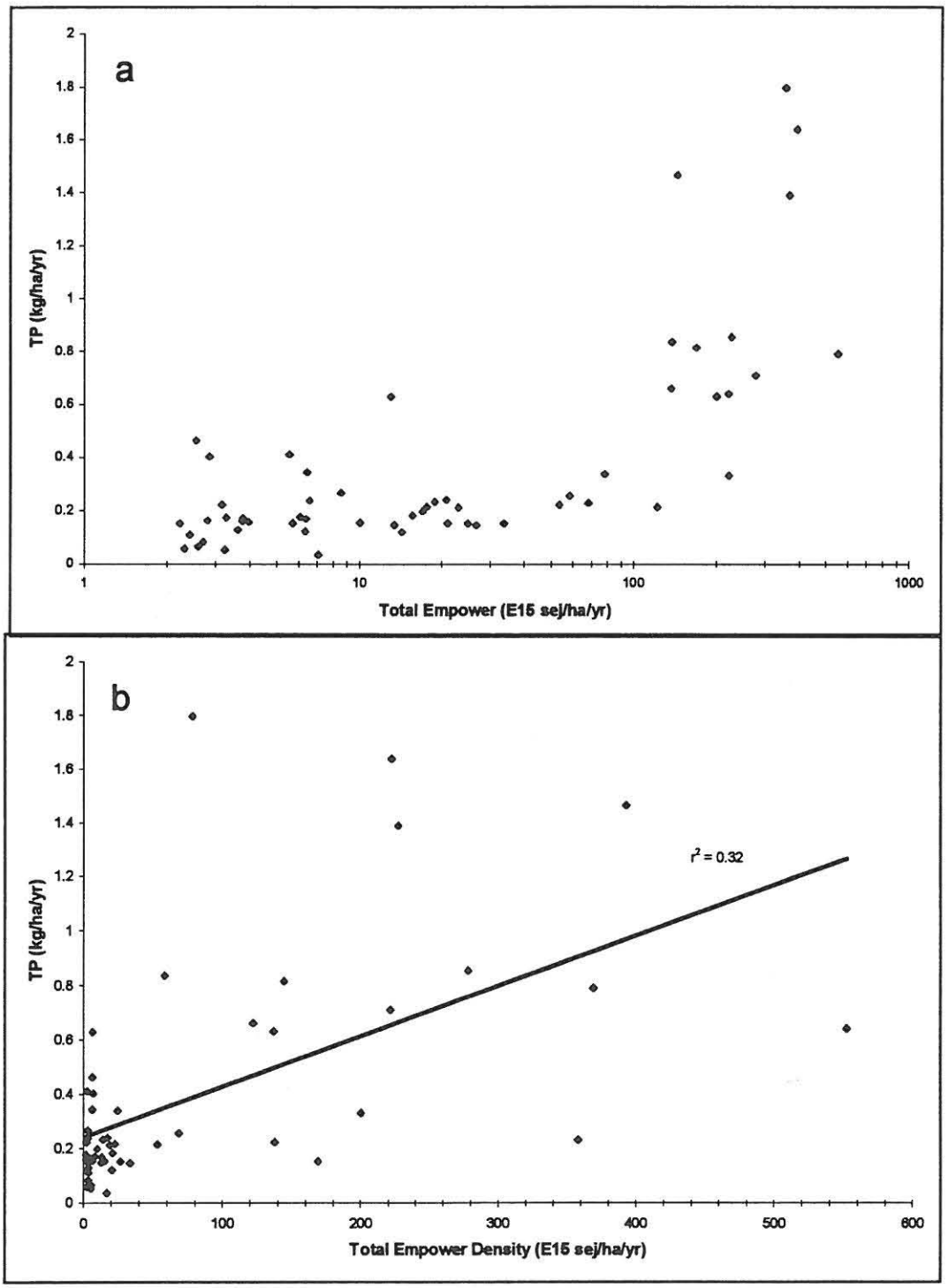


Figure 27. The total empower density LDI versus TP loads for subbasins of the St. Marks Watershed of North Florida. a) Log-scaled empower density; b) Linear scaled empower density.

The developed empower density LDI is shown in Figure 28. Results are similar as for total empower density with values range from 0.5E15 sej/ha/yr to 1000E15 sej/ha/yr. Developed empower density was also plotted on a log and linear scale versus TP loads (Figure 29).

ELR index

The environmental loading ratio LDI is shown in Figure 30. Darker areas are larger ELRs. The city of Tallahassee has the largest ELRs (25 to 6000). The highest values (1000 to 6000) are found at the Tallahassee Airport located just southwest of the city. Interstate values average about 100. The more rural towns' have ELRs from 10 to 100. County road ELR's average from 1 to 10. The undeveloped ELR indexes are less than one and often zero where locations receive only renewable energies.

The log and linear scaled ELR is plotted versus TP loads in Figure 31. The pattern is similar to Empower density (Figure 27), albeit with more scatter. For ELRs in the range of from 0.1 to about 20 little trend is noticed with TP loads ranging from about 0.1 to 0.5 kg/ha/yr. Above 20, TP loads generally increase. ELR's above 50 have loads ranging from 0.2 to 1.7 kg/ha/yr. The ELR had a linear regression coefficient of r^2 = 0.26.

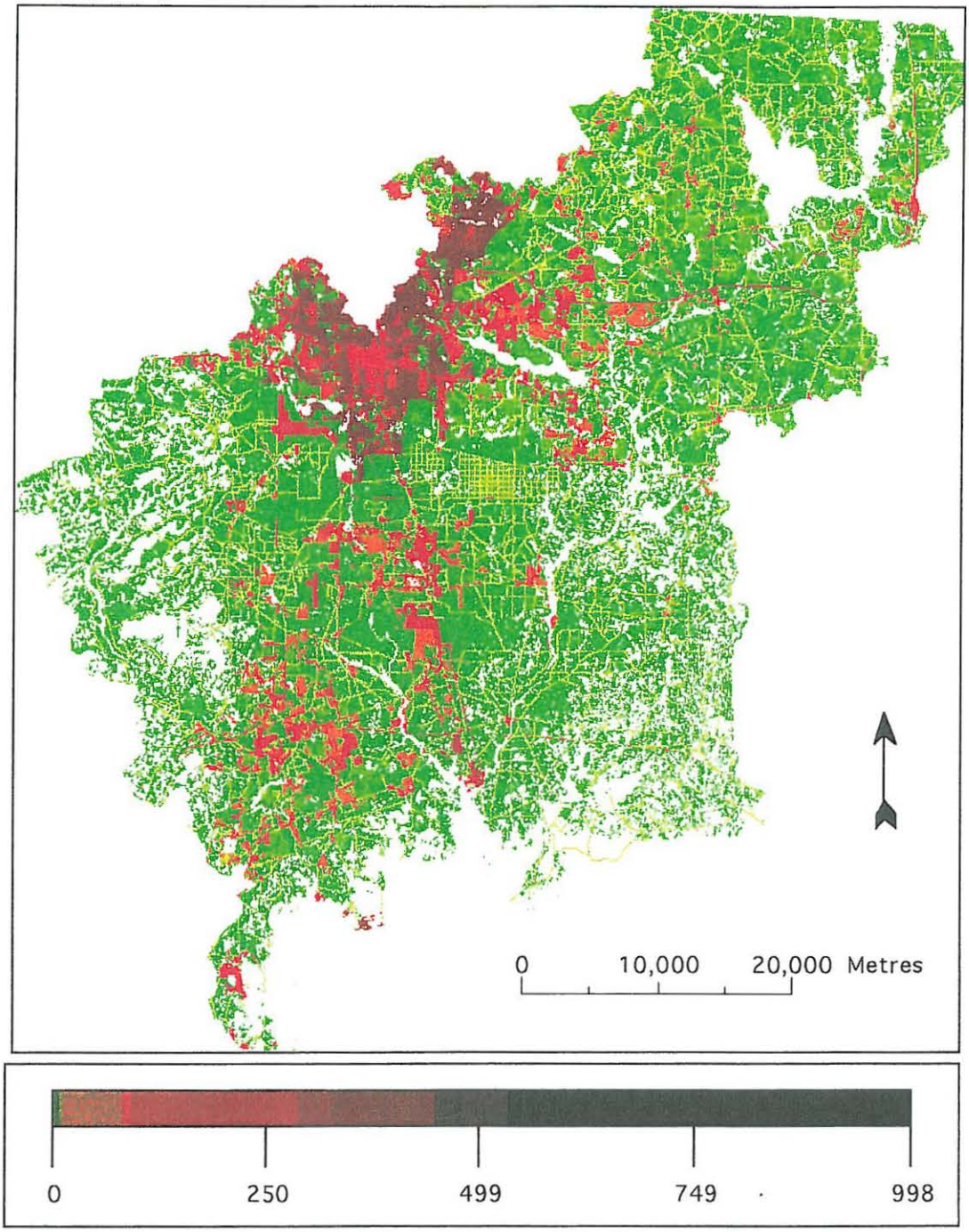


Figure 28. Developed empower density LDI (E+15 sej/ha/yr) for the St. Marks Watershed of North Florida. Green to yellow to red denotes increasing empower density

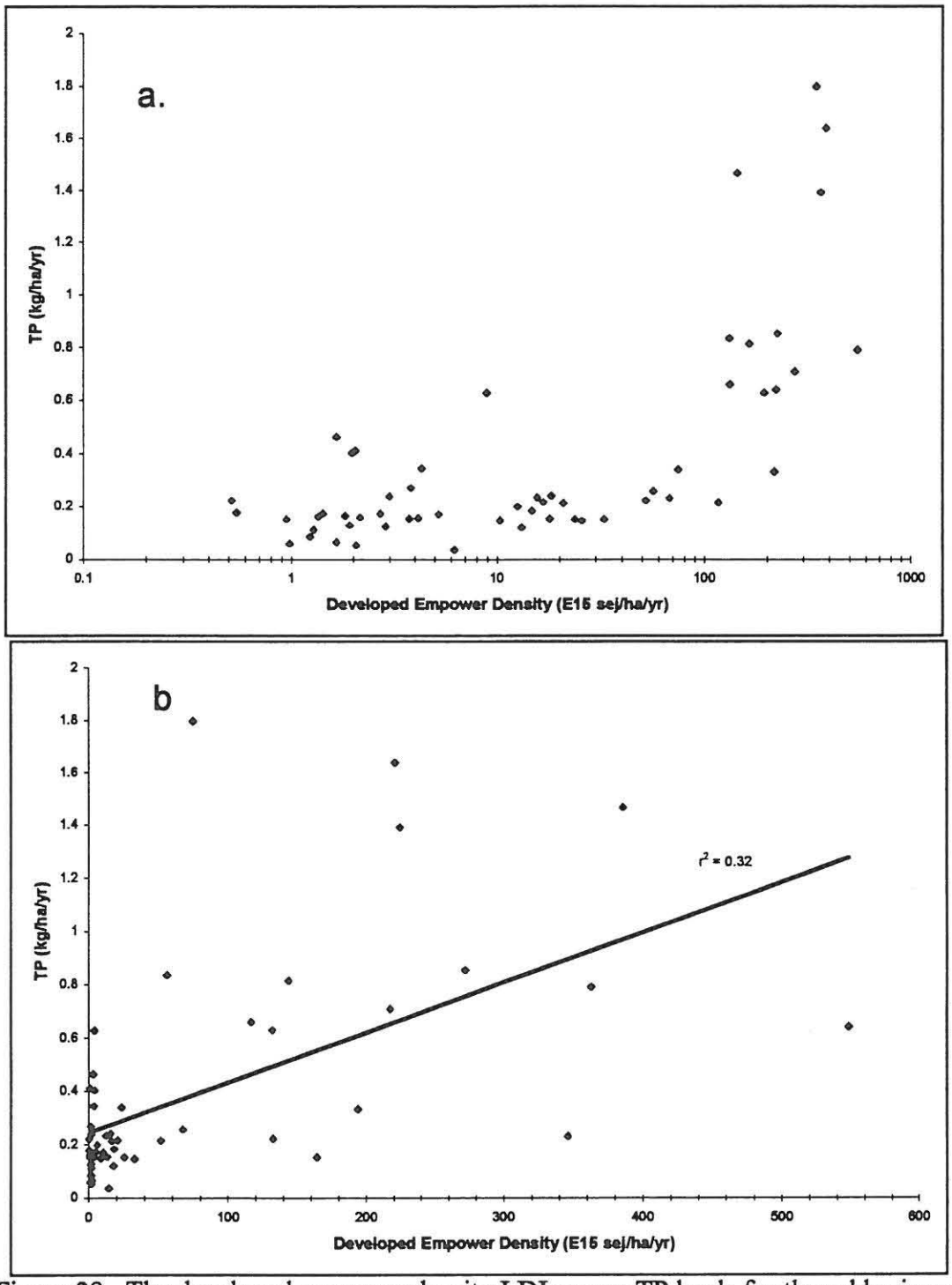


Figure 29. The developed empower density LDI versus TP loads for the subbasins of the St. Marks Watershed of North Florida. a) Log-scaled empower density; b) Linear scaled empower density.

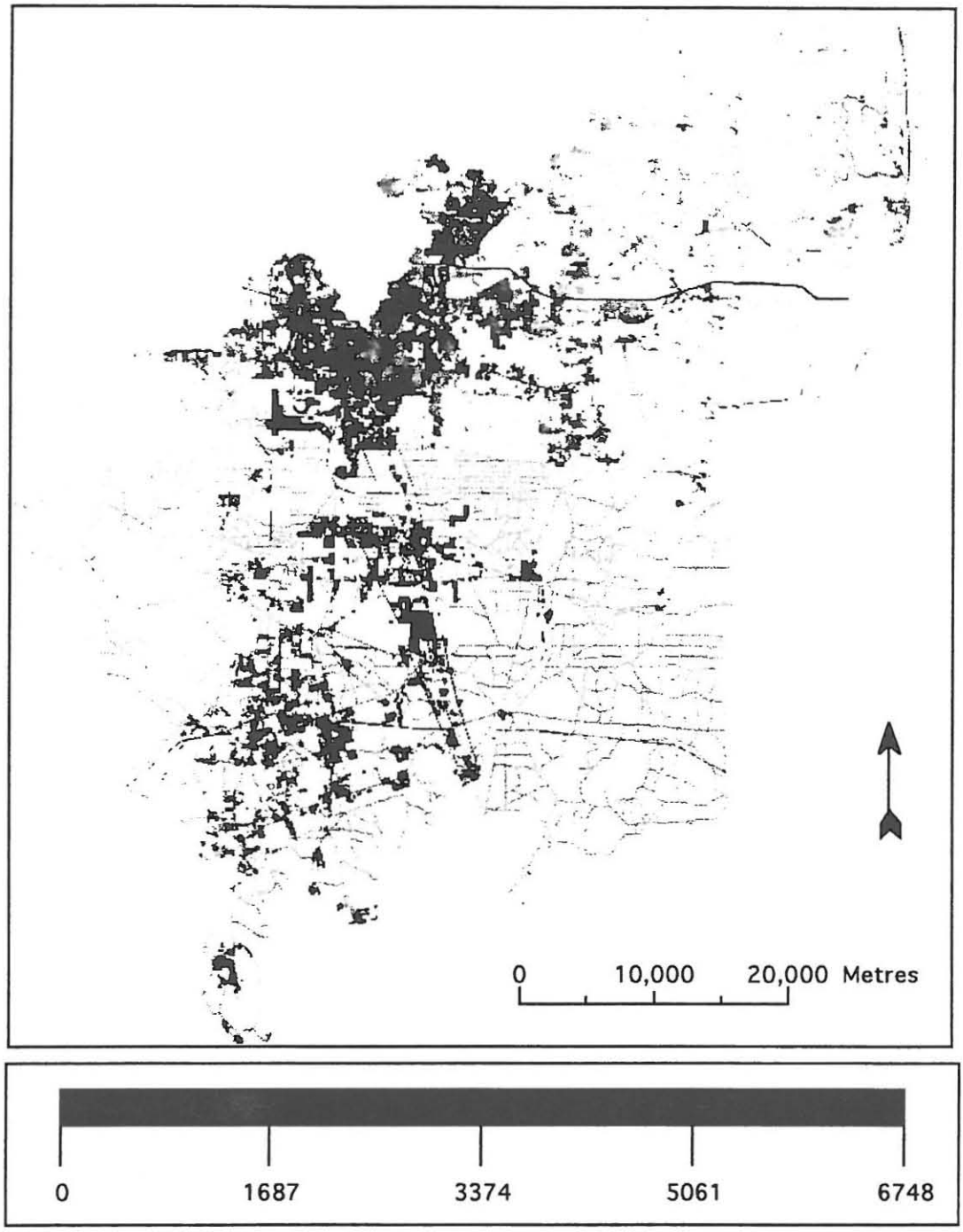


Figure 30. The Environmental Loading Ratio LDI for the St. Marks Watershed of North Florida. Darker regions have a greater Environmental Loading Ratio (range from 0 to 6700).

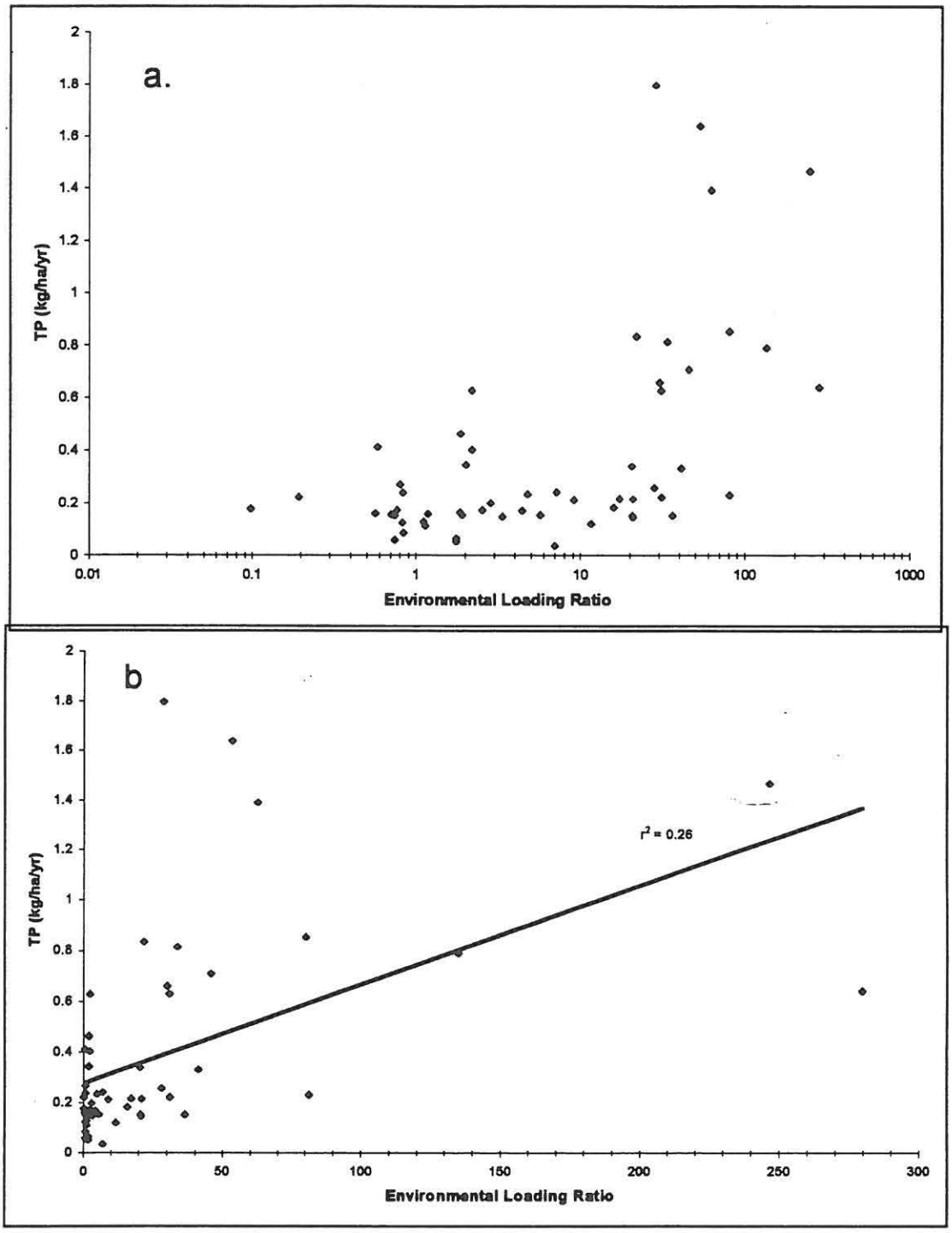


Figure 31. The Environmental Loading Ratio LDI versus TP loads for the subbasins of the St. Marks Watershed of North Florida. a) Log-scaled ELR; b) Linear scaled ELR.

DISCUSSION

In this study, a spatial TP model was developed that accounted for the mitigating factor of distance. Runoff generated by the model was compared with observed stream flow at several stations within the watershed. Modeled TP concentrations were calibrated to observed TP concentrations using two decay coefficients for the northern and southern halves of the watershed.

Five LDI measures were developed for the watershed. These included imperviousness, weighted land use, total empower density, developed empower density, and an environmental loading ratio. Modeled TP results were compared with the five measures of development intensity to investigate whether any LDI was a better predictor of TP loads within the watershed.

Comparison of Modeled and Observed Stream Flow Data

Since TP loads are intricately linked to surface water runoff, comparison of modeled and observed flow data may help in understanding the model's predictive behavior.

Although the observed and predicted flows for the SMSW station were close, modeled flows for the WSC and SMRN stations were about 50% of observed (see Figure 15). For locations south of the Cody Escarpment, ground water may add considerably to stream base flow. Thousands of springs of various sizes have been recorded throughout the Woodville Karst Plain. Flows from many of these springs are significant (most

notably Wakulla Springs). These groundwater influences within this karst domain make predicting stream flow based solely upon precipitation difficult. The model's behavior--being a surface overland flow model--is strongly influenced by direct precipitation and does not account for groundwater sources. At stations south of the Cody Escarpment (SMSW, WSC, and SMRN) the model would be expected to under estimate observed stream flow. The strong dependence of the WSC station's stream flow upon Wakulla Springs may explain the model predicting only 37% of the observed flow being predicted by the model, suggesting that the surface runoff contribution to WSC's stream flow is a relatively low percentage of total stream flow. Similarly for SMRN where model predicted stream flows indicate that approximately 45% of stream flow is attributed to groundwater. Although SMSW showed close agreement between observed and predicted stream flows, it is located closer to the Cody Escarpment than the WSC and SMRN stations and may not receive as large a groundwater influence.

Limitations in the observed stream flow data need consideration when making many comparisons. For several stations (MS8 and SMSW), stream flow data was available for only a short interval (refer to Table 6). Flow data for three stations (LC, MS8, and WSC) were only available for the late 1960s and early 1970s and may be affected by land use changes since then. Increased development near these stations over the past several decades would likely increase runoff and elevate stream flows. In fact the model predicted greater stream flows than observed for MS8 and LC (Figure 15). Both are located near urban centers (Tallahassee and Crawfordville) that have grown since the time of measurement (1970s).

The model predicted similar stream flow results as observed at the NEDD station (see Figure 15). The NEDD station, located in downtown Tallahassee, is a first order stream that receives the majority of it's flow from direct runoff.

Stream flow was modeled for an average, wet, and dry year's precipitation and not for the precipitation over the time of measurement at each individual station. Significant weather events during years of measurement such as a hurricane or drought year could affect model comparisons with observed stream flows.

Calibration of Modeled and Observed Water Quality Data

The spatial TP model predicts TP concentrations in surface runoff and not TP concentrations in water bodies after dilution has occurred. This may explain the model over predicting TP concentrations at Clear Lake and Lake Bradford, locations where observed TP concentrations were at or below detection limits (0.01 mg/l) (refer to Figure 17). Both Clear Lake and Lake Bradford are in close proximity to Tallahassee and likely receive urban runoff from the city. The observed TP concentrations from Clear Lake and Lake Bradford are grab samples from the lakes that have accounted for dilution processes. Alford Arm is also influenced by urban development, but still supports relatively low lake levels of TP (0.07 mg/l). The spatial TP model predicts higher TP concentrations in runoff (0.39 mg/l) entering the lake from surrounding development. With time many of these urban proximal lakes that contain low observed but high predicted TP concentrations (Clear Lake, Lake Bradford, Alford Arm), may begin to reflect the impacts of urban runoff (i.e., eutrophication). From a management perspective it would be useful to investigate locations where the model predicts significantly higher TP concentrations than observed. These water bodies may appear pristine, but in reality

be receiving large inputs of TP. Several of these locations included Unnamed Drain, Wakulla River, St. Marks Spring, Clear Lake, Lake Bradford, and Alford Arm.

The discrepancy between model predicted TP concentration of 0.49 mg/l and observed concentration of 0.87 mg/l for St. Augustine Branch is due to limitations in literature based aereal TP concentrations used by the model. The largest land use pollutant concentration used within the model was 0.49 mg/l, putting an upper bound on model predicted concentrations. This would also explain the model underestimating TP concentrations for Central Drainage Ditch, and East Drainage Ditch.

Limitations in the observed TP concentration data also need consideration. Number of observations and time of record varied from 2 for St. Marks Spring to 891 for Lake Munson (see Table 7). Dates of record extended from 1980 for the earliest Lake Munson readings to 1993 for a number of the sites. Changes in land use and discharge to streams is likely to vary considerably from early TP records. In one case a wastewater treatment plant located proximal to Munson Slough discharged a considerable amount of effluent into the Slough during the 1970s and early 1980s. The effects of this point source upon water quality are likely to be reflected in early observed TP concentrations for the Lake Munson site.

Observed TP concentrations for several locations (Munson Slough-below lake, Lake Lafayette Drain, Munson Slough-above lake, and Lake Munson) showed significant variation, as reflected in their error bars in Figure 17. All of these locations were found in urbanized areas where the time of sample--whether just after a storm event or during a dry spell--would greatly affect measured TP concentrations (Corbitt, 1990).

Spatial TP Model Operation

Distance was the one model parameter that most affected model results. Modeled TP output was also dependent upon the overland flow operation and found to be highly sensitive to stream density.

Distance

One goal of the spatial TP model was development of a model that could be calibrated spatially with as few parameters as possible. The decay coefficient aggregated many variables including soils, imperviousness, topography, land use, hydrology and distance. By varying the decay coefficient, the model could be adjusted for real water quality data. Since the northern and southern halves of the St. Marks Watershed differed in types of geology (clay versus limestone), a separate decay coefficient was chosen for each. The St. Marks Watershed's two distinct regions provided a unique opportunity to explore how the decay coefficient varied for different landscapes.

The spatial TP model results showed that there was a difference in the decay coefficients for the northern and southern halves of the watershed. Decay coefficients of (-0.2, -0.3) gave the best fit between observed data and predicted results (see Table 8). When no distance was accounted for at all (0,0), observed TP concentrations were overestimated for most sites suggesting that distance is an important factor and that TP load delivery is highly sensitive to this distance weight.

For decay coefficients (-0.2, -0.3), TP delivery to a stream would likely be greater north of the escarpment than south of the escarpment for the same given distance. The northern half of the watershed is more developed than the southern half, suggesting that the decay coefficient may also be land use specific. Greater development in the north

implies a greater affect of TP upon streams at greater distances from streams. If the southern half of the watershed had similar levels of development as the north, it may be plausible that the decay coefficients would be closer in value. This remains to be tested.

If the decay coefficient is closely related to topography, it should continue to increase with landscapes of greater relief. If this was the case, how development occurs in high relief watersheds would be more critical than for flatter landscapes, as a greater percentage of the watershed would contribute TP to streams.

Subbasin Contribution

The overland flow algorithm was essential to predicting runoff and TP concentrations. The model relied upon a modified DEM for the overland flow (drain) routine to perform. To explore the underlying workings of the overland flow algorithm, the drain operation was carried out for all sixty-four subbasins of the St. Marks Watershed to determine percentage contribution for each subbasin. Generally, due to the presence of sinks, internal drainage, and low spots, some fraction of the water falling upon a watershed does not leave through the surface outlet.

Percentage contribution for each subbasin was determined by draining a spatially homogenous rainfall event of one over the watershed--each cell received a value of one from the uniform rainfall. Assuming one-hundred percent of the watershed drained to the outlet, the value of the outlet cell in the cumulative runoff map would be equal to the number of cells within the subbasin. Dividing the outlet cell value by the total number of cells in the watershed gives percent contribution. This percentage contribution was then correlated with various subbasin characteristics to explore whether any one subbasin property was responsible for that subbasin's percentage contribution.

The average percentage contribution for the St. Marks Watershed was 47%, meaning 47% of the watershed contributed to the outlet through overland flow. Although a hydrological budget for the watershed has not been undertaken, 47% may be somewhat high when compared with other watersheds of similar relief, suggesting that the drain operation may over predict overland flow.

When percentage contribution for the sixty-four subbasins was compared with an assortment of subbasin characteristics including subbasin size, position within the watershed, stream density, topographical variation, and average grade, little correlation was found. This suggests that no one topographical factor overwhelmingly influences percentage contribution within a subbasin. Correlations between percentage contribution and topographical characteristics such as average grade may improve for watersheds with greater vertical relief.

Stream Resolution

The model's ability to predict TP concentrations was sensitive to stream resolution. Since the amount of phosphorus reaching a stream was dependent upon the pollutants distance from the stream, stream networks needed to be as detailed as possible. This sensitivity to stream networks was especially apparent in downtown Tallahassee. Here, where urban runoff is efficiently drained away from the city through numerous roadside ditches and stormwater drains, knowledge of the detailed routing is essential. For several of the Tallahassee subbasins, the model under predicted TP concentrations due to poor stream resolution. In these subbasins streams were sparse, while in reality there were probably a number of ditches within the subbasin. A coverage of ditches and

stormwater drains along with stream networks would greatly improve the spatial TP model's predictive power in urban regions.

Model Improvements

Comparison and calibration of the spatial TP model with real data helped to identify where model improvements were needed.

The output of the spatial TP model is only as good as the data inputs. Improved resolution of streams and topography coverages are needed. Data inputs from literature sources such as aereal TP loads should be further checked for regional validity. Aereal TP loads specific to land uses in north Florida need to be generated.

The spatial TP model could be modified to account for groundwater inputs. With improved knowledge of regional base flows, groundwater contribution could be determined through back calculation.

Inclusion of in-stream attenuation and biotic uptake of TP would further improve model realism. This would likely entail modifying the overland flow algorithm to recognize "environmental frictions" on a cell-by-cell basis. Uptake factors for streams and different biological communities could be translated into friction maps.

Accounting for the dilution processes that occur when runoff enters lakes would allow in-lake TP concentrations to be determined and not just TP concentrations in runoff entering the lakes. A coverage of waterbody volumes included within the algorithm when generating TP concentrations may solve this problem.

Although not a direct improvement upon the spatial TP model, observed data of better quality (both temporally and spatially) would increase confidence in calibrated

model results. This can only be achieved with improved sampling efforts, a task often limited by funding.

Spatial Correlations of Water Quality with LDIs

Five LDIs were chosen for this analysis, although numerous others could have been used. These five LDIs measure different aspects in development and vary in applicability. For example, although the imperviousness LDI only measures one item, it may provide planners with a "quick and dirty" method for assessing watershed development using the simplest of spatial data available (i.e., land use, imperviousness data, and subbasin coverages). The energy based indices are an assessment tool that encompasses many impacts.

All LDIs were determined for subbasins that fell within the effective distance from streams predicted to contribute 10% or greater TP load to the stream (based upon decay coefficients of (-0.2, -0.3)). Consequently, subbasins that fell outside of this effective distance were not included within the analysis, as their TP loads upon the streams were predicted to be negligible (refer to Figures 18 and 19).

When the five LDIs were plotted versus subbasin TP loads, a threshold LDI occurred above which TP loads corresponded with increased development intensity (Figure 32). Below this threshold it was difficult to distinguish developmental influences from background variation in TP loads. This suggests that there is an LDI above which development intensity has a noticeable effect upon in-stream TP concentrations. The approximate location of this threshold was determined for each of the LDIs as denoted by the vertical lines in Figure 32. This threshold value was chosen at the LDI where TP loads became consistently greater than background loads of 0.2 kg/ha/yr (approximately

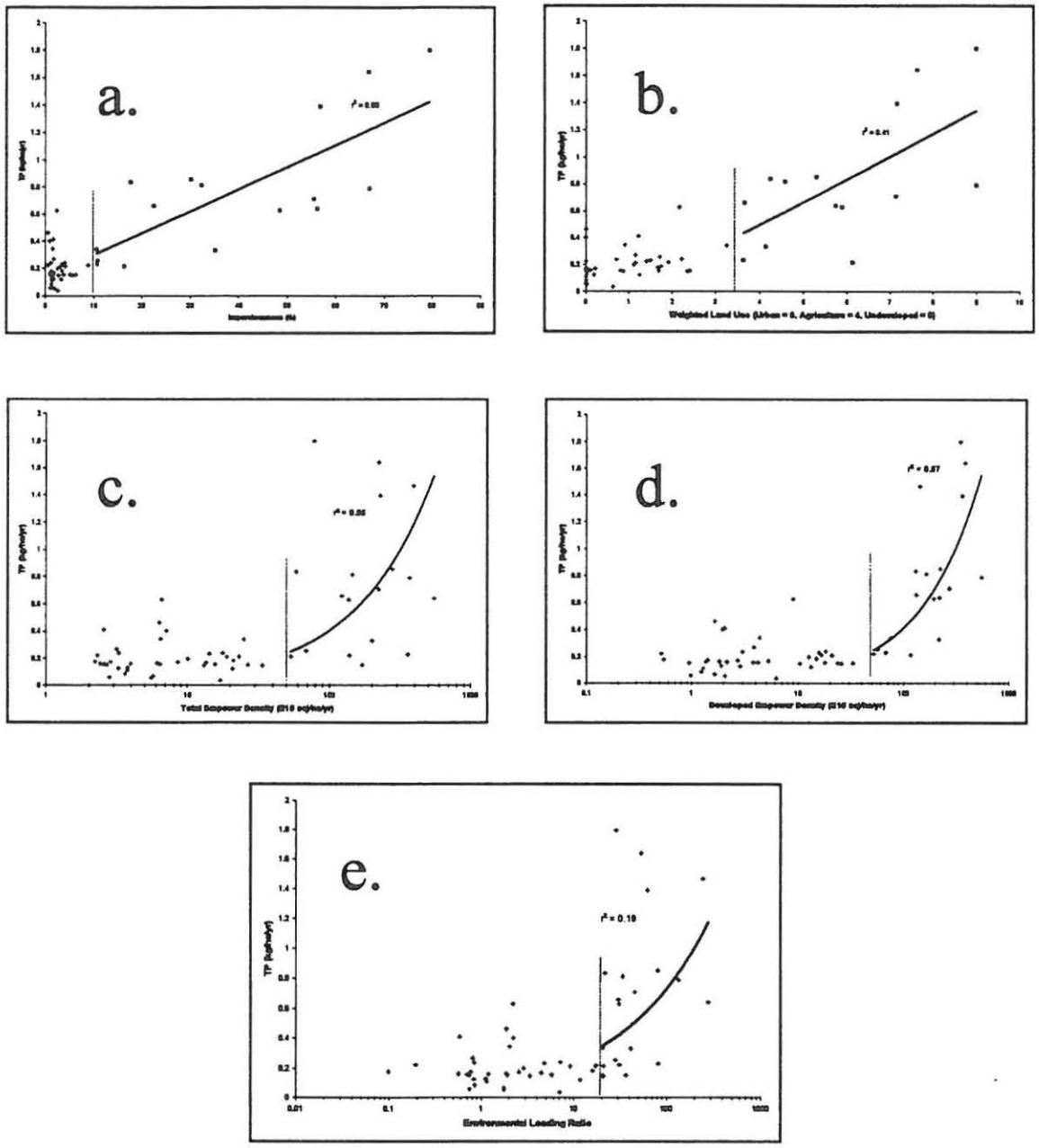


Figure 32. Best fit curve above the LDI threshold (denoted by the vertical line) for the five LDI-TP graphs. a) Imperviousness; b) Weighted land use; c) Total empower density; d) Developed empower density; e) Environmental loading ratio.

0.012 mg/l). Above this threshold improved regression coefficients were found between all LDIs and TP loads as seen by the best fit lines on the graphs. A linear regression best fit both physical indices, while a power regression best fit the emergy indices. This is probably due to the physical LDIs comprising only one to two orders of magnitude while the emergy LDIs encompass four to five orders of magnitude. Table 9 summarizes these threshold locations, regression coefficients, and landscape characteristics as percent developed (urban and agriculture) for subbasins above these threshold LDIs.

Table 9. Threshold LDIs, regression coefficients, and minimum percent development above the threshold for the five LDIs.

LDI	Threshold LDI	Units	r^2 Above Threshold	Minimum % Development above LDI Threshold
Imperviousness	10	%	0.60	30
Weighted Land Use	3.5	unitless	0.41	30
Total Empower	50	E+15 sej/ha/yr	0.56	25
Developed Empower	50	E+15 sej/ha/yr	0.57	25
ELR	20	unitless	0.19	20

Imperviousness LDI is a useful tool for predicting TP loads above 10% imperviousness as implied by the linear regression coefficient of $r^2 = 0.60$. The overall linear correlation for all imperviousness with TP loads was $r^2 = 0.74$. Subbasins with imperviousness greater than 10% were more than 30% developed. Below 10% imperviousness, urban development ranged from 0% to 25% and agriculture ranged from

0% to 30% of the subbasins. Above 10% imperviousness agriculture was less than 20% for all subbasins.

The weighted land use LDI had a linear regression coefficient of $r^2 = 0.41$ with TP loads above an LDI of 3.5. The overall linear correlation for all weighted land uses with TP loads was $r^2 = 0.67$. As with imperviousness, all subbasins were at least 30% developed for LDIs above 3.5. Agriculture was always less than 20% above 3.5 and ranged from 0% to 40% below 3.5. Since the weighted land use LDI weighted urban with 9 and agriculture with 4, a threshold of 3.5 may indicate that development intensities slightly below that of agriculture are when TP loads first become noticeable above background.

The total empower density LDI had a power regression coefficient of $r^2 = 0.56$ with TP loads above 50E15 sej/ha/yr. Above 50E15 sej/ha/yr the landscape was at least 25% developed. Results for developed empower density ($r^2 = 0.57$; threshold = 50E15 sej/ha/yr) were similar to total empower density indicating that renewable emergies have little affect upon predicting TP loads compared with nonrenewables.

Of the five LDIs, the ELR gave the lowest correlation with TP. The threshold LDI for the ELR occurred at about 20. A low power regression coefficient with TP was found ($r^2 = 0.19$). ELR had an overall correlation with TP loads of $r^2 = 0.26$. Development was generally greater than 20% for ELRs above 20 although significant land use variation was apparent above and below the threshold. Since the ELR is a ratio, it likely experiences greater sensitivity to the landscape than the empower densities, contributing to the greater scatter and thus lower correlation observed.

LDI Summary

The imperviousness LDI had the highest correlation with TP loads. The empower density LDIs had lower correlations and ELR the lowest (see Table 9). The higher correlations displayed by the imperviousness indice than the emergy indices with TP is related to it's relationship with the immediate landscape. Imperviousness has a direct relationship with the St. Marks Watershed. The emergy LDIs encompass both direct and indirect energies. The indirect component, while important in determining impact at a larger scale, probably does not have an affect on TP flows within the St. Marks Watershed. Perhaps the better use of the emergy LDIs is to illustrate the energy necessary to produce a given load on a landscape.

Exploring the relationship between development intensity and TP may ultimately provide planners with a tool for predicting ecosystem health. Although the interplay of many complicated factors is likely to influence the integrity of an ecosystem, it is well known that nutrients such as TP can change an ecosystem's composition as with eutrophication. Development intensities above a certain threshold may foreshadow long-term ecosystem change.

One of the most important generalities concluded from this study was that predicting TP loads remains difficult at low development intensities. The variation in background TP levels in these low ranges of intensity masks developmental influences. It is only when a threshold equivalent to 30% development is reached that TP loads become apparent above background levels.

Conclusions

The following conclusions can be made regarding pollutant load modeling and the effects of LDIs upon TP within the St. Marks Watershed of North Florida.

1. Pollutant load models can be successfully developed that aggregate many parameters including soils, topography, and imperviousness into a decay coefficient, vastly simplifying the model algorithm.
2. The spatial TP model results using decay coefficients of (-0.2, -0.3) for locations north and south of the escarpment were found to compare closest with observed TP data. The difference in decay coefficients for locations above and below the Cody Escarpment may be a reflection of the differences in geology, topography, and land use.
3. The spatial TP model showed that with the selected decay coefficients (-0.2, -0.3), locations north of the escarpment will deliver a greater TP load to a nearby stream than locations south of the escarpment for the same given distance.
4. Model predicted TP concentrations ranged from 0.00 mg/l to 0.49 mg/l. Observed TP concentrations ranged from 0.01 mg/l to 0.87 mg/l. Model predicted TP loads for subbasins ranged from 0.2 to 1.8 kg/ha/yr. Background loads of TP averaged about 0.2 kg/ha/yr.
5. Management efforts may be best focused at locations where the spatial TP model predicts significantly higher TP concentrations than observed concentrations. These may be locations where development induced impacts are just beginning to be felt.
6. All LDIs exhibited a threshold above which the LDI corresponded closer to TP

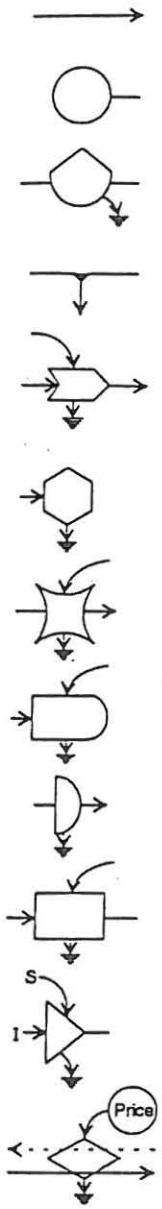
loads. These thresholds occurred at 10% imperviousness for the imperviousness LDI, 3.5 for the weighted land use LDI, 50E15 sej/ha/yr for the total and developed empower LDIs, and 20 for the ELR LDI. Below these thresholds the subbasins are more clumped, making it difficult to distinguish background TP levels from developmental influences.

7. For intensities above the LDI threshold, the imperviousness LDI had the highest correlation with TP loads ($r^2 = 0.60$).

8. The threshold LDI may indicate locations where impacts of development upon water quality are just beginning to be observed.

9. The model suggested that TP loads become apparent above background levels when development exceeds 30%.

APPENDIX A
ENERGY LANGUAGE SYMBOLS



Energy circuit: A pathway whose flow is proportional to the quantity in the storage or source upstream.

Source: Outside source of energy delivering forces according to a program controlled from outside; a forcing function.

Tank: A compartment of energy storage within the system storing a quantity as the balance of inflows and outflows; a state variable.

Heat sink: Dispersion of potential energy into heat that accompanies all real transformation processes and storages; loss of potential energy from further use by the system.

Interaction: Interactive intersection of two pathways coupled to produce an outflow in proportion to a function of both; control action of one flow on another; limiting factor action; work gate.

Consumer: Unit that transforms energy quality, stores it, and feeds it back autocatalytically to improve inflow.

Switching action: A symbol that indicates one or more switching actions.

Producer: Unit that collects and transforms low-quality energy under control interactions of high-quality flows.

Self-limiting energy receiver: A unit that has a self-limiting output when input drives are high because there is a limiting constant quality of material reacting on a circular pathway within.

Box: Miscellaneous symbol to use for whatever unit or function is labeled.

Constant-gain amplifier: A unit that delivers an output in proportion to the input *I* but is changed by a constant factor as long as the energy source *S* is sufficient.

Transaction: A unit that indicates a sale of goods or services (solid line) in exchange for payment of money (dashed line). Price is shown as an external source.

Figure A-1. Symbols of the Energy Systems Language (Odum, 1996).

APPENDIX B
TP MODEL ALGORITHM

```
/**ALGORITHM TO DETERMINE AMOUNT OF POLLUTANT THAT REACHES THE STREAM**/

/**DETERMINING STREAM REACH**/

drain_result = drain strmks_basin1 over "StMarks:StMarks:StMrk - Coverages:pseudo_DEM";

main_stream = recode drain_result assigning 1 to 500...137000;
save main_stream;

/**ACCOUNTING FOR OVERLAND ATTENUATION**/

strms_sprd = spread main_stream to 100 over "StMarks:StMarks:StMrk - Coverages:pseudo_DEM" uphill;

strms_sprd1 = float(strms_sprd);
save strms_sprd1;

zz_distance_effect = EXP(strms_sprd1*coefficient_map);
save zz_distance_effect;

/**DETERMINE VOLUME AND LOAD FOR A 65 INCH AVERAGE RAIN SEASON**/

zz_runoff_liters = 65*runoff_coeff*254000;
save zz_runoff_liters;

zz_load_kg = areal_poll_conc * zz_runoff_liters*zz_distance_effect/1000000;
save zz_load_kg;

zz_total_load = drain zz_load_kg over "StMarks:StMarks:StMrk - Coverages:pseudo_DEM";
save zz_total_load;

/**CREATION OF CONCENTRATION MAP BY DIVIDING LOAD MAP BY RUNOFF TOTAL MAP**/

zz_total_load_mg = 1000000*zz_total_load;

zz_total_runoff_liters = drain zz_runoff_liters over "StMarks:StMarks:StMrk - Coverages:pseudo_DEM";

zz_total_runoff_liters1 = recode zz_total_runoff_liters assigning void to 0, carryover;

zz_concentration_map = (zz_total_load_mg / zz_total_runoff_liters1);

concentration_map1 = prec(zz_concentration_map, 3);
save concentration_map1;

zz_station_score = score WQ_samp by zz_concentration_map1 maximum;

zz_WQ_compare = combine WQ_samp with WQ_observed with zz_station_score ;
save zz_WQ_compare;
```

APPENDIX C
EMERGY CALCULATIONS

Table C-1. Emergy calculations used to generate spatial emergy maps.

RENEWABLE ENERGY

SUNLIGHT

area = 1 ha
avg. insol. = (1.58E6 kcal/m^2/yr)(1-0.1)(1E4 m^2/ha)(4186 J/kcal)
= 5.95E13 J/ha/yr (Odum, 1996; p114)

Transformity = 1 sej/J

Sun (sej) = 59.5E12 sej/ha/yr

RAIN, GEOPOTENTIAL

Elevation difference was determined for all cells within a subbasin relative to that subbasins outlet

(runoff_mapm^3/yr)*(DEM_map-Minelev_mapm)(1E3kg/m^3)(9.8m/s^2)=_____J/ha/yr

Transformity = 10489 sej/J (Odum, 1996)

RAIN, CHEMICAL

Work calculated as the percentage of rainfall transpired from different land uses
Following percentages were used:

Transpired	*Land Use*	*Transpired*	*Land Use*
50%	low-dens. res.	60%	pasture
	single family		gen. ag.
	multi-family		mining
	rec/open space		wetland
			upland

Transpired	*Land Use*
5%	low-int. comm.
	high-int. comm.
	industrial
	highway

Table C-1--continued.

Average Rainfall = 65 in/yr = 1.651 m/yr

Transformity = 18,199 sej/J (Odum, 1996)

Rain (sej)=(1E4m^2/ha)(1.651m/yr)(Transpiration_map)(4.94J/g)(1E6g/m^3)(18199sej/J
= ____sej/ha/yr

NONRENEWABLE ENERGY

1 ha = 2.47 acres
1 lbs/acre/day = 409,304 g/ha/yr

EARTH LOSS

Transformity (topsoil) = 6.3E4 sej/J (Odum, 1996; p194)

(5.4 Kcal/g)(4186 J/Kcal) = 22604 J/g

(Corbitt, 1990)

	Rate (lbs/acre/day)	(g/ha/yr)	(sej/ha/yr)
Resid.	1	409,304	5.83E+14
Comm.	1.5	613,956	8.74E+14
Industry	1	409,304	5.83E+14
Forest	0.04	16,372	2.33E+13
Agric.	0.5	204,652	2.91E+14

The Following Assumptions were Used:

low-intensity comm. = strip comm.
high-intensity comm. = shopping centers

highway = major arterial
rec./open space = low-dens. res.
mining = industrial

Table C-1--continued.

Following based upon Whitfield, 1993

DIRECT ENERGY

(Whitfield, 1993)

Land Use	Direct_eng (E12 sej/ha/yr)
Low-Dens. Res.	82004
Single Family	213902
Multi-Family	533520
Low-Int. Comm.	249470
High-Int. Comm.	288496
Industrial	462384
Highway	407550
Pasture	2.72
Gen. Ag.	8.65
Rec./Open Space	82004
Mining	0
Wtld.	0
Upland	11.73
Open Water/Lake	0

AGRICULTURE

Bahia Pasture (Brandt-Williams, 1998)

Electric	4.00E+13	sej/ha/yr
Potash	4.00E+13	sej/ha/yr
Lime	3.70E+14	sej/ha/yr
Phosphate	1.60E+14	sej/ha/yr
Nitrogen	3.70E+14	sej/ha/yr
Total	*9.80E+14*	*sej/ha/yr*

Table C-1--continued.

Corn (Grain)	(Brandt-Williams, 1998)
Electric	1.30E+14 sej/ha/yr
Potash	1.20E+14 sej/ha/yr
Lime	3.70E+14 sej/ha/yr
Pesticides	3.00E+13 sej/ha/yr
Phosphate	4.60E+14 sej/ha/yr
Nitrogen	1.38E+15 sej/ha/yr
Total	*2.49E+15 sej/ha/yr*

VEHICULAR FUEL CONSUMPTION

This was broken into three sectors: Tallahassee, I-10 interstate, and Rural.

Tallahassee

Population of Tallahassee = 137,057 (1995, FSA)
Pop. of Tallahassee (assuming 10% average annual increase w/students and legislators = 150,057

Fuel use per capita per year = 477 gal/capita (1994) (1995, FSA)
Gibbs of gasoline (J/g) = 6.28E9 J/bbl = 1.50E8 J/gal (Odum, 1996)
Table C-1. continued.

Total fuel use in Tallahassee = 71,913,951 gallons/yr = 1.08E16 J/yr

Transformity = 6.6E4 sej/J (Odum, 1996)

Emergy = 7.13E20 sej/yr
Total Area = 12,241 ha
Avg. Empower Density = 5.8247E16 sej/ha/yr

	Percent*	Area (ha)	Fuel Used (sej/ha/yr)
Low-Dens. Res.	3	914	1.75E+15 (1747.41E12)
Single Family	7	563	4.08E+15 (4077.29E12)
Multi-Family	25	7748	1.46E+16 (14561.75E12)
Comm/Indust.	65	3016	3.79E+16 (37860.55E12)

* Percentage of total fuel used within each land use:

Table C-1--continued.

I-4 Interstate

cars/yr = 44,045 vehicles/day = 16,076,425 veh/yr (based on DOT traffic count)
Average MPG = 22 mpg
I-10 length = 20.13 mi = 3.24E4 m (324 cells)

Fuel Use = 20.13 mi * 1gal/22 mi * 16,076,425 vehicles/yr = 1.47E7 gal/yr
= 2.21E15 J/yr

Emergy Use = 2.21E15 J/yr * 6.6E4 sej/J = 1.456E20 sej/yr

Emergy/100m length=1.456E20sej/yr/324 cell =4.4947E17sej/100m/yr (449382.72E12)

Rural

Pop.in Leon/WakullaCnty.minusTallahassee=(271,500+17,000)-150,763=83,737people
Fuel use/capita = 477 gal/capita
Fuel Use = 39,942,549 gallons
Fuel Use = 3.95E20 sej/yr
Length of all rural roads = 4.34E6 m (43,360 cells)

Average Fuel Use/100m = 9.11E15 sej/100m/yr (9109.77E12)

APPENDIX D
SECONDARY COVERAGES

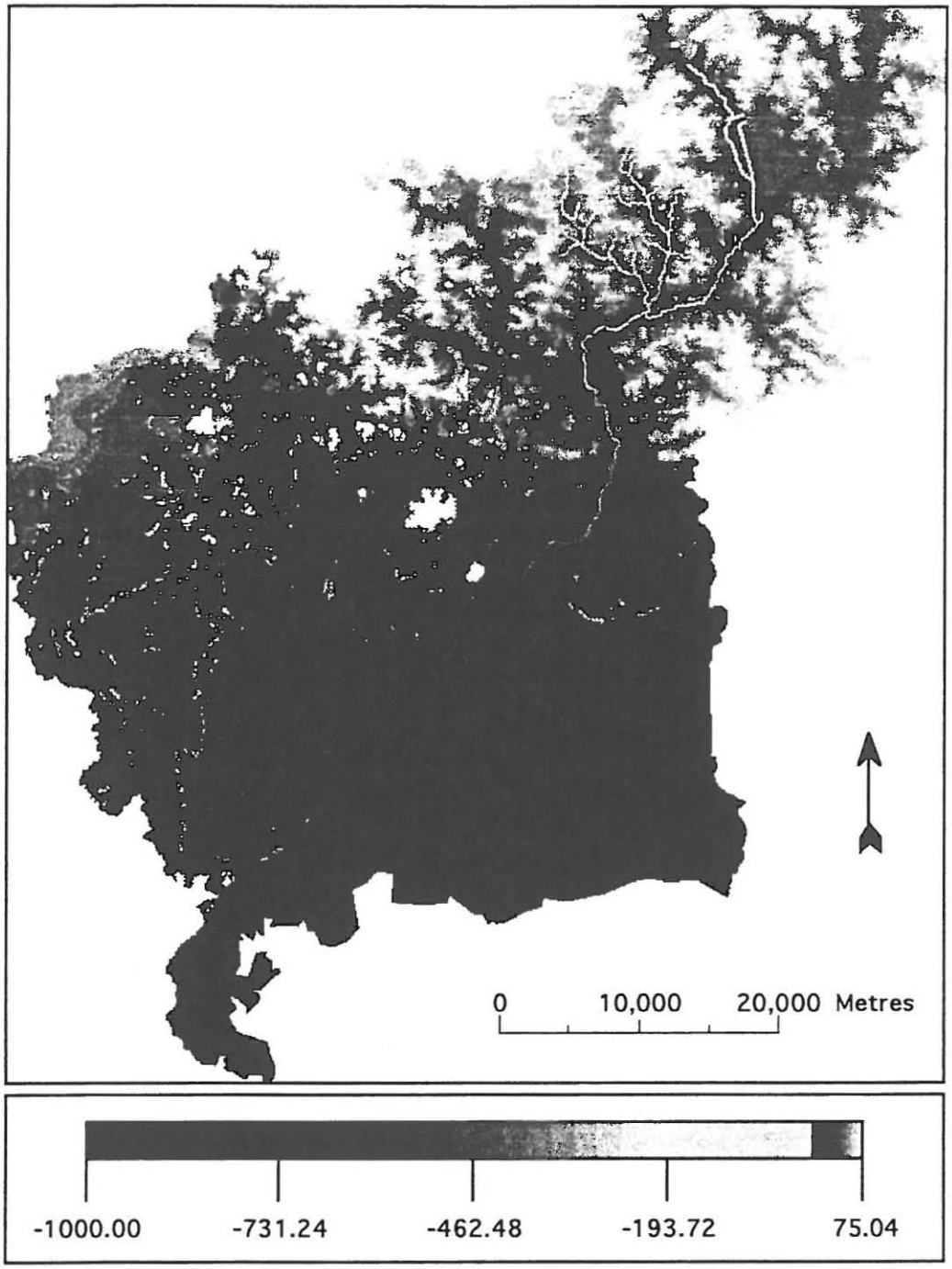


Figure D-1. The modified DEM for the St. Marks Watershed of North Florida. Stream bed elevations are negative. Positive values are meters above mean sea level.

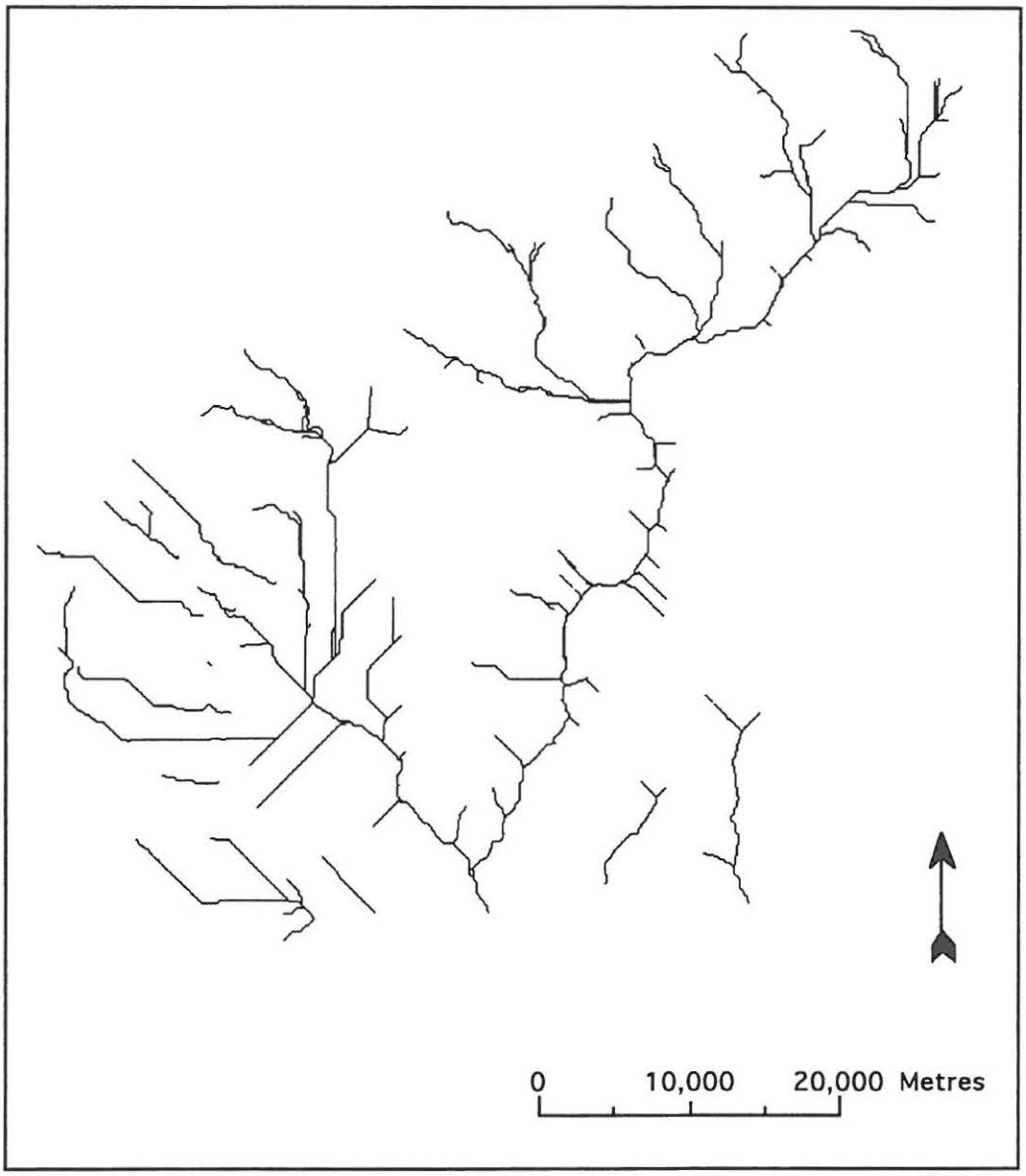
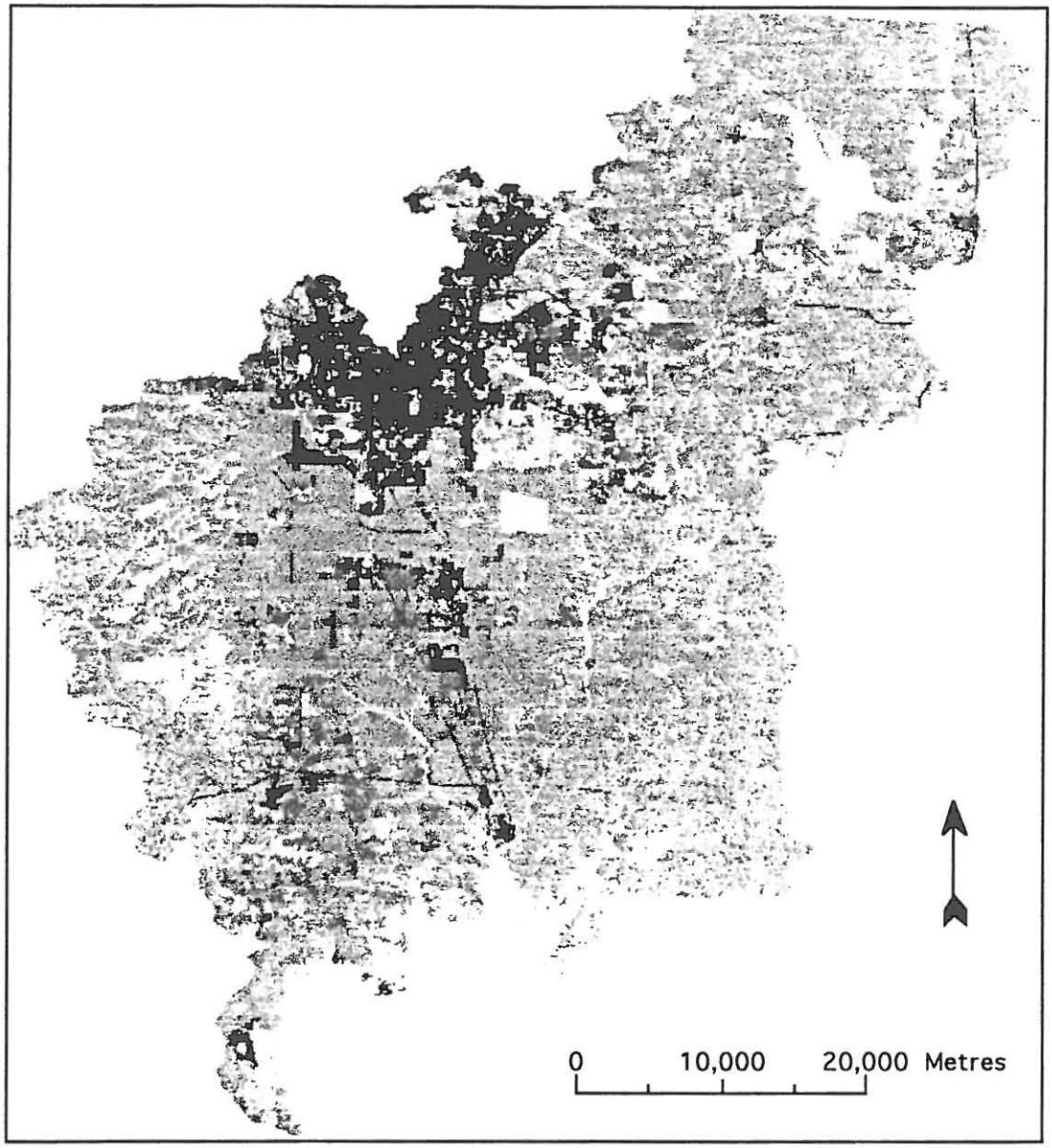


Figure D-2. Stream networks for the St. Marks Watershed of North Florida.



	0	Wetland/Open Water		28	Single-Family
	1	General Agriculture		67	Multi-Family
	1	Pasture		85	Highway
	2	Recreation/Open Space		87	Industrial
	2	Upland		91	Low-Intensity
	15	Low-Density Residential		98	High-Intensity
	23	Mining			

Figure D-3. Imperviousness (%) for land uses in the St. Marks Watershed of North Florida

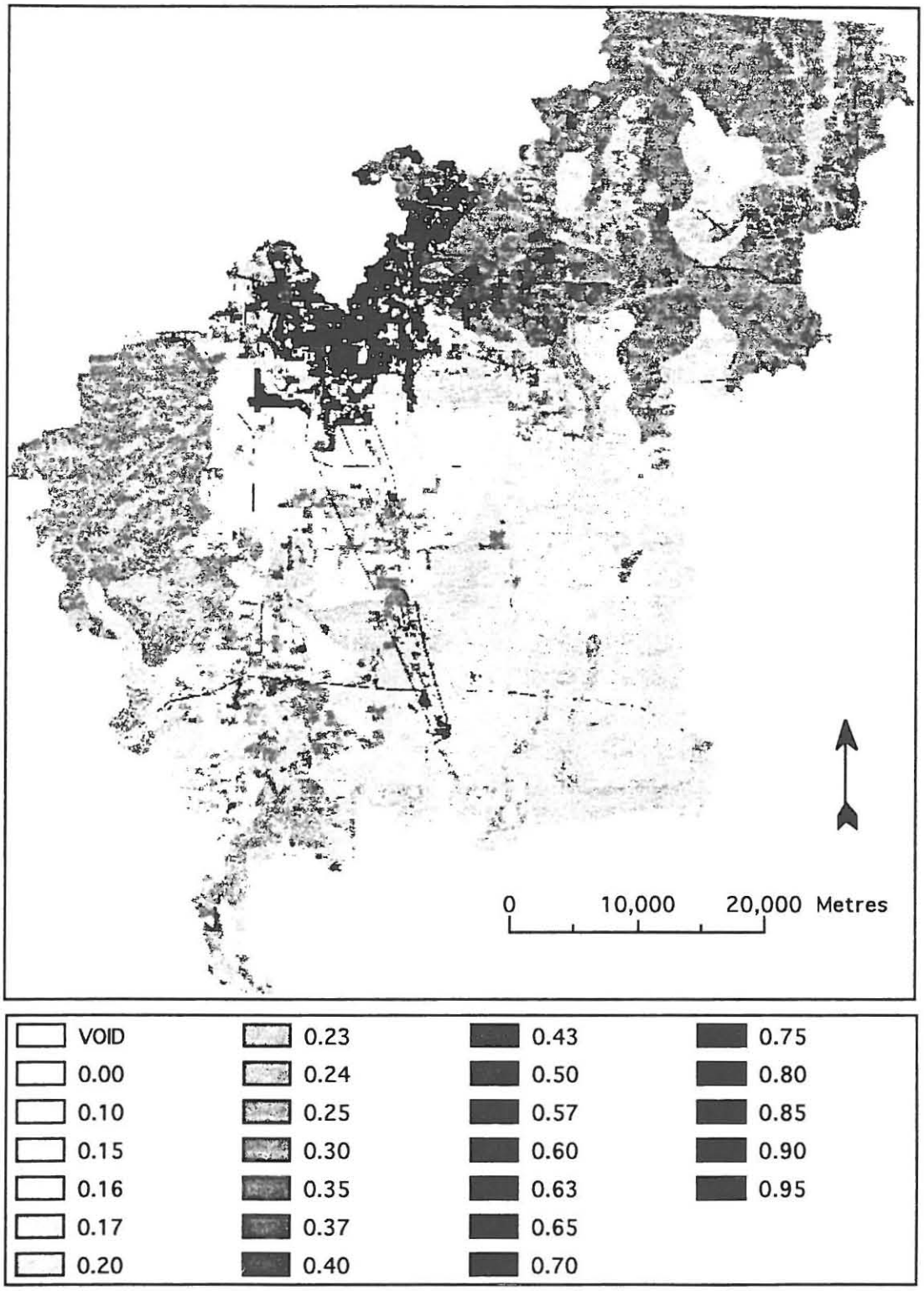


Figure D-4. Runoff coefficients for land use/soil combinations for the St. Marks Watershed of North Florida.

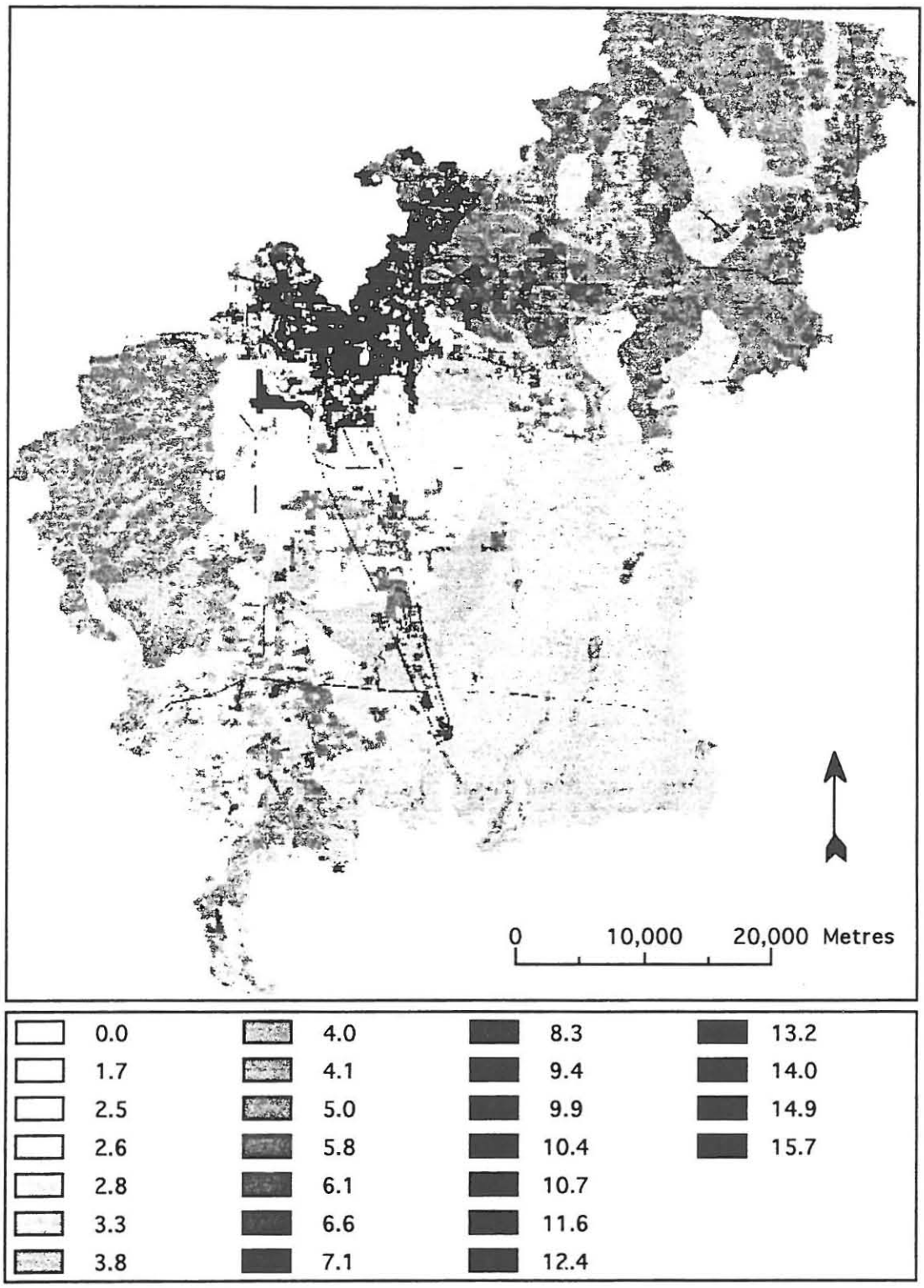


Figure D-5. Runoff (E6 liters/ha/yr) for the St. Marks Watershed of North Florida.

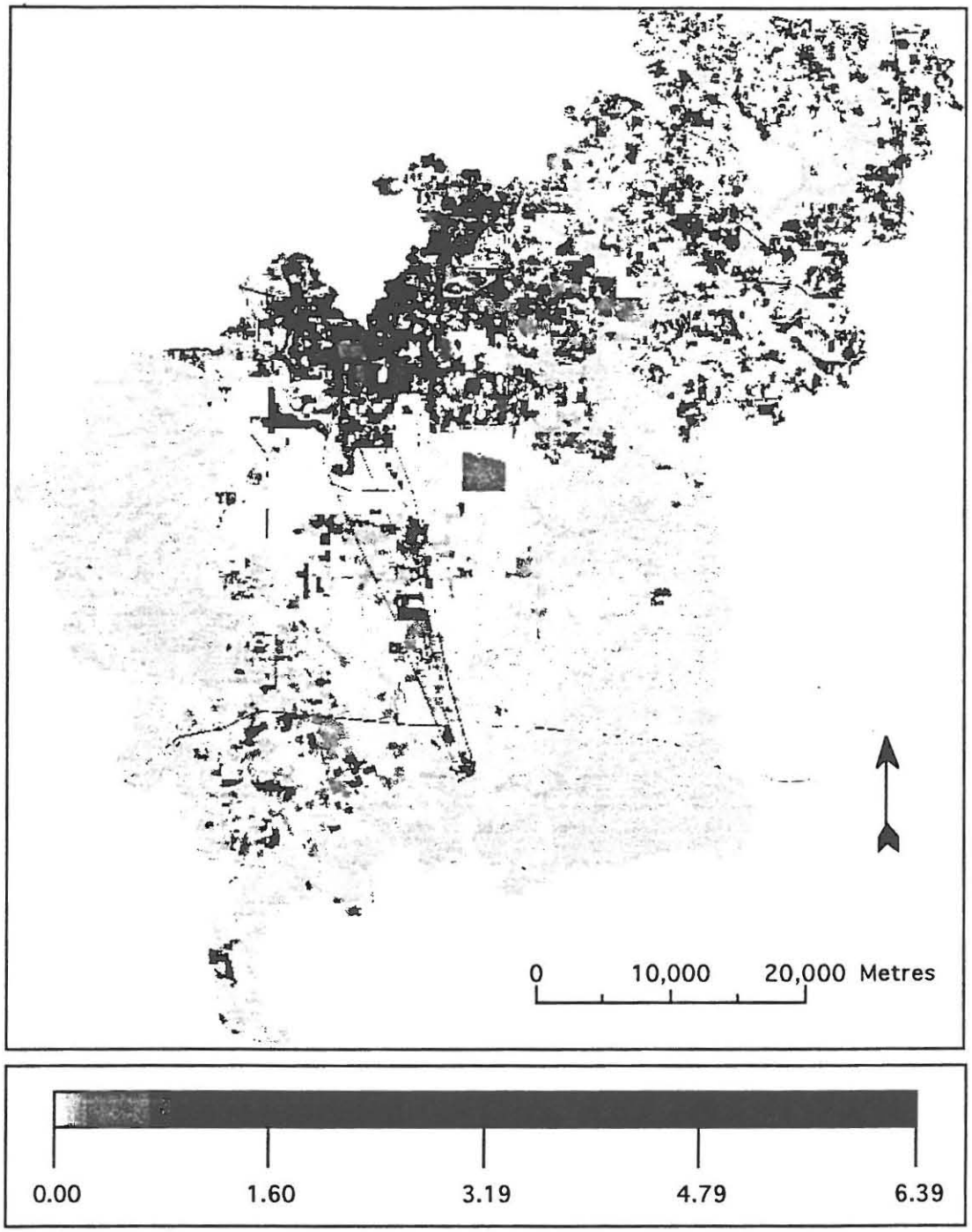


Figure D-6. Aereal TP loads (kg/ha/yr) for the St. Marks Watershed of North Florida.

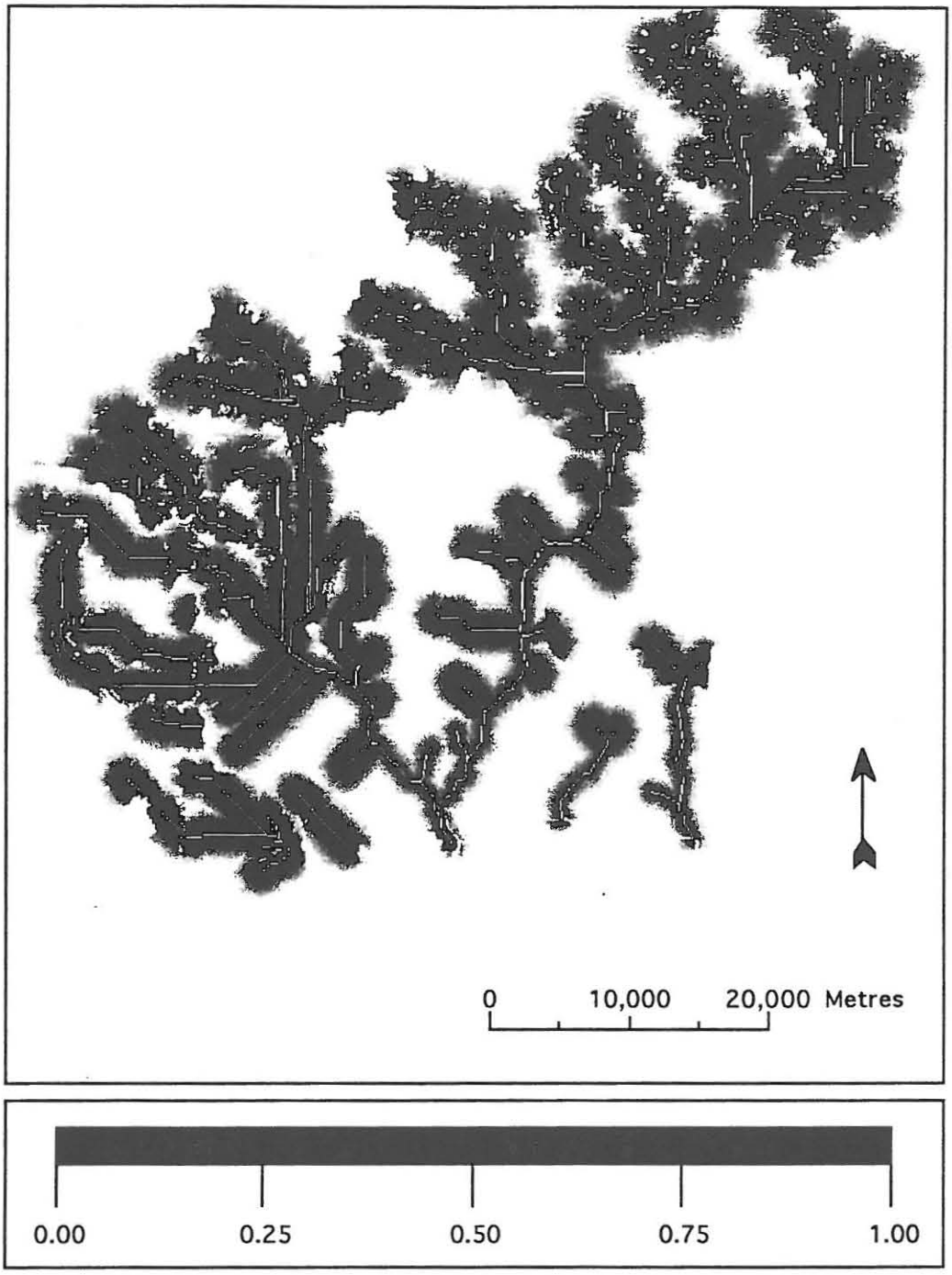


Figure D-7. Distance weights for the St. Marks Watershed of North Florida. Weights from 0 to 1 (white to dark) are increasing affects upon streams.

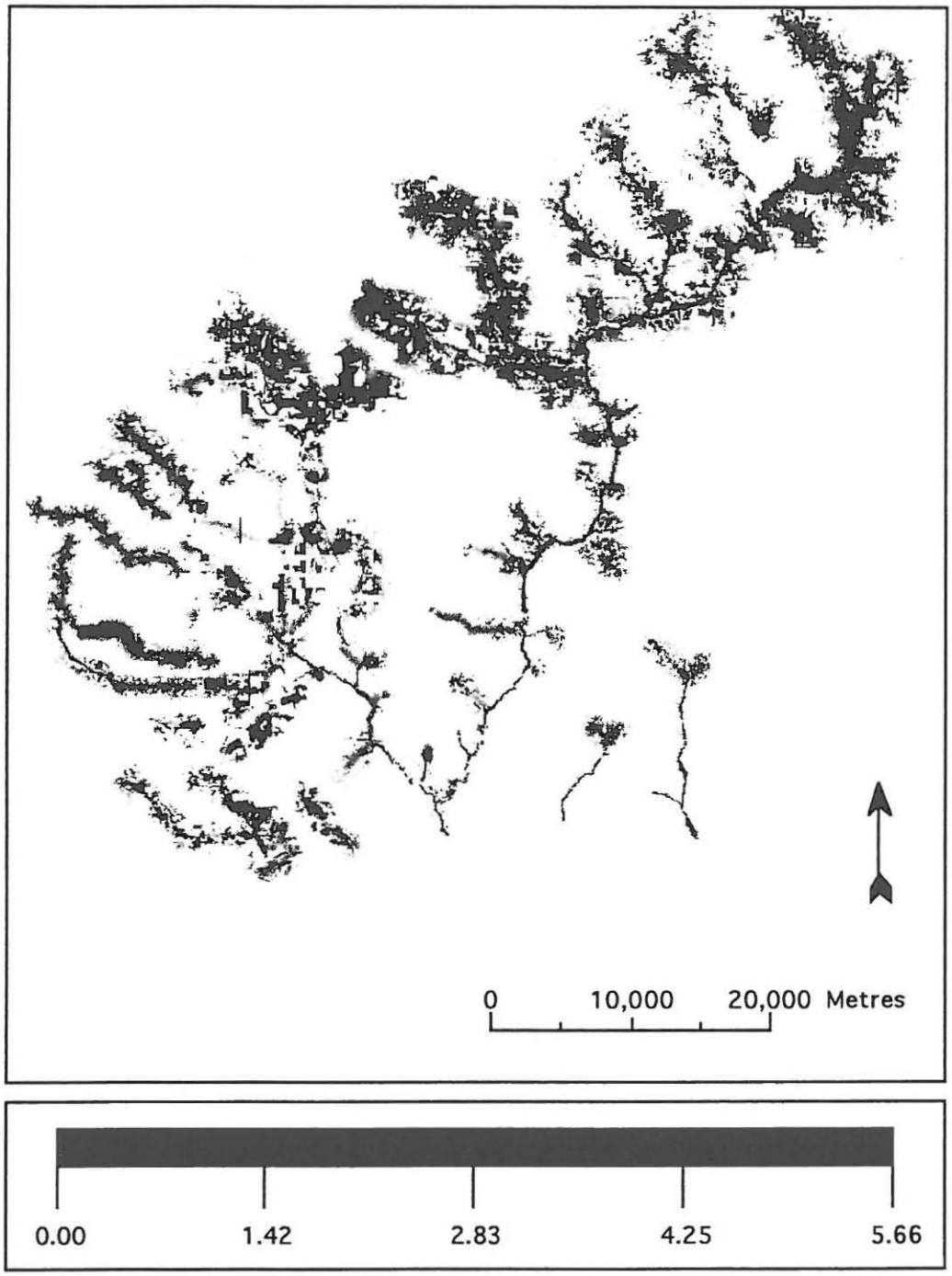


Figure D-8. Net aereal TP loads coverage (kg/ha/yr) for the St. Marks Watershed of North Florida.

APPENDIX E
SUBBASIN CHARACTERISTICS

Table E-1. Subbasin area percentages. Basins are ranked by increasing TP (kg/ha/yr)

bns #	Name	Total Area (ha)	Urban area (%)	Agric. area (%)	Natural area (%)
1	Unnamed Drain, drain*	928	0.43	0.00	99.57
2	Dickerson Bay, bay*	2860	13.36	0.38	82.87
3	Burnt Mill Creek, stream	2024	1.28	17.69	78.95
4	Chicken Branch, slough	2570	5.99	5.02	88.99
5	Unnamed Branch, stream	238	6.30	34.03	59.66
6	Fisher Creek, stream	9215	1.83	0.10	97.23
7	Limestone Creek, stream	1168	4.11	2.05	93.84
8	Unnamed Branch, stream	253	0.40	10.28	89.33
9	East River, stream	7964	0.28	0.00	99.72
10	Unnamed Run, stream*	1109	54.73	5.86	36.61
11	Moore Branch, stream	685	0.73	15.77	83.50
12	Mill Creek, drain	2296	2.61	0.44	93.86
13	Moriah Creek, stream	1184	5.66	0.00	94.34
14	Hall Branch, stream	664	2.26	13.40	84.34
15	Lost Creek, stream	12365	1.82	0.55	95.56
16	Pinhook River	1241	0.00	0.00	99.03
17	Unnamed Branch, stream	391	5.63	13.04	81.33
18	Direct Runoff to bay, runoff	7930	6.15	2.30	89.14
19	Direct Runoff to Bay, runoff	16345	0.24	0.46	99.21
20	Cuba Branch, stream	1024	1.95	25.59	70.90
21	Lutterloh Slough, slough	8771	7.49	19.36	73.15
22	Sweetwater Branch, stream	711	4.08	19.55	76.37
23	Polar Mill Branch, stream	1050	0.67	21.24	78.10
24	Dog Pond Drain, slough	2842	0.04	0.00	99.96
25	Hosford Branch Reach, stream	6484	0.08	0.00	99.91
26	Black Creek, stream	4132	5.64	2.78	91.58
27	Crow Pond Outlet, outlet	228	3.95	38.60	57.46
28	Lang Branch, stream	613	0.33	30.83	66.56
29	Walker Creek, bayou	1806	8.08	1.94	89.98
30	Cow Swamp Drain, drain	2134	0.00	0.00	100.00
31	Unnamed Slough, slough	2149	32.57	8.14	59.28
32	St. Marks river, stream	26785	6.28	4.89	88.78
33	Wakulla River, stream	24526	18.25	2.77	78.98
34	McBride Slough, stream	6817	32.15	3.37	64.47
35	Unnamed Drain, drain	1148	0.09	0.00	99.91
36	Direct Runoff to bay, runoff	940	3.40	0.00	94.36
37	West Goose Creek, bayou	2841	8.59	10.45	80.96
38	Unnamed Branch, stream	185	0.00	29.73	70.27
39	Pinhook River, stream	2181	0.00	0.00	99.86
40	Morris Branch, stream	2518	2.10	22.08	75.77

Table E-1.--continued.

bns #	Name	Total Area (ha)	Urban area (%)	Agric. area (%)	Natural area (%)
41	Robert's Pond Outlet, outlet	2404	13.48	22.30	62.56
42	Spring Creek, bayou	5475	14.54	12.09	73.37
43	Bradford Brook, stream	5063	22.34	1.78	75.57
44	Lake Drain, stream	18722	2.01	19.70	77.95
45	Black Creek, stream	5116	11.81	12.31	75.84
46	Ward Creek, stream	15518	3.63	21.81	74.34
47	Lloyd Creek, noncon	4603	2.76	24.70	72.28
48	Patty Sink Drain, stream	7697	5.05	24.98	69.20
49	Big Boggy Branch, stream	813	27.06	0.12	72.82
50	Godby Ditch, ditch*	1389	78.11	0.29	21.45
51	Munson Slough, stream	12371	29.98	0.93	68.61
52	Caney Branch, Stream	1405	8.26	38.15	53.59
53	Unnamed Slough, slough	1203	23.19	43.47	33.33
54	Copeland Sink Drain, drain	2493	21.70	17.05	61.25
55	St. Augustine Branch, stream*	629	99.68	0.00	0.32
56	Lake Lafayette Drain, drain	6419	52.33	7.38	39.23
57	Alford Arm, Slough	9401	42.95	16.41	39.70
58	East Drainage Ditch, ditch	1672	78.71	0.42	20.87
59	Buck Lake Outlet, outlet	416	62.26	10.10	27.64
60	Gilbert's Pond Outlet, outlet	763	64.88	8.26	26.08
61	Central Drainage Ditch, ditch	1517	92.88	0.00	7.12
62	Unnamed Run, stream	344	74.71	3.20	22.09
63	Unnamed Run, stream	145	55.86	1.38	42.76
64	Mall Drainage Ditch, stream	712	76.69	1.54	21.77

*** These basins were left out of the analysis because they lacked adequate drainage network resolution.**

Table E-2. Subbasin average TP and imperviousness.

bsn #	TP (kg/ha/yr)	TP (kg/yr)	Imperv. (ha)	Imperv. (%)
1	0.00	0.01	14.38	0.02
2	0.01	17.14	162.55	0.06
3	0.07	141.47	45.09	0.02
4	0.07	188.61	69.40	0.03
5	0.08	18.30	4.92	0.02
6	0.09	774.20	181.42	0.02
7	0.10	110.24	21.20	0.02
8	0.10	25.41	4.55	0.02
9	0.11	805.99	108.56	0.01
10	0.12	82.75	318.24	0.30
11	0.12	79.92	11.10	0.02
12	0.12	257.21	49.55	0.02
13	0.13	150.06	53.63	0.05
14	0.14	88.79	13.95	0.02
15	0.14	1498.60	212.82	0.02
16	0.14	104.95	2.38	0.00
17	0.15	56.61	8.96	0.02
18	0.15	924.46	199.40	0.03
19	0.16	1977.33	144.40	0.01
20	0.16	144.94	25.60	0.03
21	0.17	1410.24	411.71	0.05
22	0.17	115.37	14.83	0.02
23	0.17	164.87	18.08	0.02
24	0.17	480.13	34.61	0.01
25	0.17	1132.60	71.15	0.01
26	0.18	726.79	101.10	0.02
27	0.18	40.81	4.27	0.02
28	0.19	102.90	10.82	0.02
29	0.20	307.44	64.52	0.04
30	0.22	450.64	16.64	0.01
31	0.22	468.72	183.43	0.09
32	0.22	5845.11	704.07	0.03
33	0.23	5662.95	1384.00	0.06
34	0.25	1682.44	666.83	0.10
35	0.25	288.25	13.39	0.01
36	0.26	236.17	10.95	0.01
37	0.26	731.20	68.71	0.02
38	0.26	47.18	2.26	0.01
39	0.28	581.30	15.78	0.01
40	0.30	711.91	76.28	0.03

Table E-2.--continued.

bsn #	TP (kg/ha/yr)	TP (kg/yr)	Imperv. (ha)	Imperv. (%)
41	0.32	730.29	86.83	0.04
42	0.33	1790.40	225.50	0.04
43	0.33	1595.79	609.66	0.12
44	0.35	6049.20	367.61	0.02
45	0.36	1739.54	199.95	0.04
46	0.36	5309.20	395.01	0.03
47	0.37	1645.81	140.82	0.03
48	0.41	2945.20	184.64	0.02
49	0.43	346.68	144.50	0.18
50	0.43	584.51	746.81	0.54
51	0.43	5014.61	2502.55	0.20
52	0.44	587.30	41.12	0.03
53	0.49	532.11	97.25	0.08
54	0.55	1349.70	123.88	0.05
55	0.78	488.28	472.20	0.75
56	0.84	5175.64	2120.51	0.33
57	0.90	8060.03	1678.51	0.18
58	0.96	1587.48	849.77	0.51
59	0.97	391.56	86.35	0.21
60	1.03	742.20	265.79	0.35
61	1.15	1612.13	1046.96	0.69
62	1.34	456.28	195.58	0.57
63	1.46	211.35	53.87	0.37
64	1.49	1041.27	420.39	0.59

Table E-3. Subbasin sunlight and rain (geopotential) emergies.

bsn #	Sun (E12 sej/ha/yr)	Sun (E15 sej/yr)	Rain, geo (E15 sej/ha/yr)	Rain, geo (E15 sej/yr)
1	0.06	55.87	0.00	0.00
2	0.06	168.98	0.00	0.00
3	0.06	120.19	0.01	17.33
4	0.06	152.92	0.00	6.12
5	0.06	14.16	0.00	1.11
6	0.06	547.40	0.01	93.05
7	0.06	69.50	0.00	3.12
8	0.06	15.05	0.01	2.69
9	0.06	473.86	0.00	0.17
10	0.06	65.51	0.01	10.17
11	0.06	40.76	0.00	6.46
12	0.06	135.96	0.01	7.13
13	0.06	70.45	0.00	0.00
14	0.06	39.51	0.00	2.65
15	0.06	732.09	0.01	89.55
16	0.06	73.84	0.00	2.78
17	0.06	23.26	0.01	3.23
18	0.06	470.11	0.00	2.41
19	0.06	972.41	0.00	2.74
20	0.06	60.75	0.01	8.34
21	0.06	521.87	0.00	31.15
22	0.06	42.30	0.01	5.91
23	0.06	62.48	0.01	10.54
24	0.06	169.10	0.00	6.60
25	0.06	385.80	0.00	4.27
26	0.06	245.85	0.01	21.37
27	0.06	13.57	0.01	1.37
28	0.06	36.47	0.01	5.45
29	0.06	107.46	0.00	0.16
30	0.06	126.97	0.00	0.02
31	0.06	127.87	0.00	9.00
32	0.06	1593.71	0.00	66.58
33	0.06	1459.30	0.00	16.57
34	0.06	405.61	0.00	4.91
35	0.06	68.31	0.00	2.41
36	0.06	55.22	0.00	1.84
37	0.06	169.04	0.00	0.15
38	0.06	11.01	0.01	1.27
39	0.06	129.77	0.00	0.29
40	0.06	149.82	0.01	27.53

Table E-3.--continued.

bsn #	Sun (E12 sej/ha/yr)	Sun (E12 sej/yr)	Rain, geo (E12 sej/ha/yr)	Rain, geo (E12 sej/yr)
41	0.06	142.68	0.01	26.34
42	0.06	325.76	0.00	1.19
43	0.06	301.19	0.00	24.33
44	0.06	1112.95	0.01	221.58
45	0.06	304.40	0.01	60.00
46	0.06	923.14	0.01	101.05
47	0.06	273.88	0.01	56.53
48	0.06	457.08	0.01	101.27
49	0.06	48.37	0.00	0.00
50	0.06	82.65	0.02	22.97
51	0.06	735.36	0.01	79.96
52	0.06	83.60	0.01	15.77
53	0.06	71.58	0.01	8.60
54	0.06	148.33	0.01	16.52
55	0.06	37.43	0.03	16.15
56	0.06	380.98	0.02	116.97
57	0.06	558.65	0.02	163.72
58	0.06	99.48	0.02	25.12
59	0.06	24.75	0.02	8.63
60	0.06	45.40	0.01	7.99
61	0.06	90.26	0.02	27.96
62	0.06	20.47	0.02	8.13
63	0.06	8.63	0.02	2.45
64	0.06	42.36	0.02	17.38

Table E-4. Subbasin rain (chemical) and earth loss emergies.

bsn #	Rain, chem (E15 sej/ha/yr)	Rain, chem (E15 sej/yr)	Earth_loss (E15 sej/ha/yr)	Earth_loss (E15 sej/yr)
1	0.44	404.03	0.03	23.22
2	0.89	1779.11	0.10	266.18
3	0.88	1741.41	0.08	151.48
4	0.87	2232.12	0.07	167.32
5	0.88	208.84	0.14	34.44
6	0.88	8071.79	0.03	258.92
7	0.88	1033.07	0.04	51.14
8	0.89	225.17	0.05	12.95
9	0.86	6831.11	0.01	117.88
10	0.78	836.32	0.36	389.51
11	0.88	600.40	0.07	45.04
12	0.88	1956.87	0.03	63.29
13	0.86	1021.79	0.05	60.99
14	0.89	588.23	0.07	45.97
15	0.88	10706.58	0.03	322.40
16	0.77	951.13	0.00	2.77
17	0.88	344.95	0.09	33.68
18	0.77	5962.74	0.05	425.47
19	0.84	13718.55	0.01	194.73
20	0.88	882.93	0.10	101.93
21	0.85	7483.23	0.12	1054.31
22	0.87	618.44	0.09	67.07
23	0.89	933.17	0.08	85.98
24	0.85	2515.90	0.01	40.64
25	0.89	5763.92	0.01	80.85
26	0.88	3637.79	0.06	229.54
27	0.83	190.14	0.15	33.16
28	0.89	531.83	0.11	64.37
29	0.83	1491.44	0.06	111.38
30	0.89	1900.53	0.01	19.39
31	0.83	1785.93	0.23	490.79
32	0.85	22705.79	0.06	1717.03
33	0.85	20792.96	0.13	3232.93
34	0.83	5644.43	0.21	1462.65
35	0.89	1021.56	0.01	15.19
36	0.89	824.97	0.02	14.40
37	0.64	2424.47	0.09	262.55
38	0.88	162.98	0.10	17.94
39	0.87	1904.99	0.01	18.38
40	0.88	2205.04	0.09	230.13

Table E-4.--continued.

bsn #	Rain, chem (E12 sej/ha/yr)	Rain, chem (E12 sej/yr)	Earth_loss (E12 sej/ha/yr)	Earth_loss (E12 sej/yr)
41	0.84	1993.60	0.16	375.77
42	0.86	4702.80	0.14	741.55
43	0.80	4013.44	0.15	766.13
44	0.78	14464.96	0.08	1538.34
45	0.86	4420.58	0.12	613.66
46	0.87	13483.40	0.10	1516.61
47	0.88	4021.14	0.11	482.09
48	0.85	6487.69	0.12	898.03
49	0.73	595.36	0.17	139.34
50	0.66	919.66	0.50	698.74
51	0.76	9353.64	0.21	2613.37
52	0.86	1215.12	0.17	235.67
53	0.82	984.98	0.27	322.97
54	0.84	2106.03	0.19	469.13
55	0.42	264.28	0.72	451.97
56	0.69	4355.33	0.37	2335.32
57	0.79	7310.71	0.31	2876.04
58	0.65	1085.53	0.52	862.62
59	0.74	307.33	0.40	166.68
60	0.75	570.42	0.41	311.40
61	0.48	731.31	0.65	988.66
62	0.61	208.32	0.52	178.97
63	0.65	93.73	0.38	55.58
64	0.57	403.44	0.54	385.43

Table E-5. Subbasin direct (electric) and fuel use emergies.

bsn #	Direct_eng (E15 sej/ha/yr)	Direct_eng (E15 sej/yr)	Fuel (E15 sej/ha/yr)	Fuel (E15 sej/yr)
1	8.35	7669.17	0.41	300.62
2	29.28	80905.81	1.59	4536.67
3	3.26	6466.55	1.34	2705.60
4	9.23	23728.92	1.74	4463.79
5	5.17	1231.35	2.41	573.92
6	3.43	31338.58	1.85	17026.48
7	3.38	3944.40	1.29	1512.22
8	0.33	84.49	0.47	118.43
9	1.10	8741.46	1.20	9538.06
10	210.55	226973.73	5.13	5689.44
11	0.61	415.74	0.98	674.12
12	7.13	15858.51	1.09	2505.18
13	14.96	17708.38	1.29	1530.44
14	3.45	2291.20	0.60	400.83
15	2.69	32551.44	1.11	13674.00
16	0.00	1.40	0.59	728.78
17	4.62	1807.26	0.82	318.84
18	8.72	67478.85	1.11	8818.37
19	0.47	7635.35	1.13	18547.85
20	5.19	5228.82	0.69	710.56
21	12.91	113256.53	2.35	20595.29
22	3.81	2708.96	1.73	1229.82
23	0.56	583.20	1.08	1138.72
24	0.01	20.16	1.31	3716.78
25	0.32	2076.99	1.45	9383.19
26	7.39	30523.75	1.70	7005.48
27	3.24	739.44	1.12	255.07
28	1.37	819.74	0.73	446.38
29	21.38	38620.16	1.27	2286.55
30	0.00	9.76	0.64	1375.57
31	44.43	95479.94	2.87	6176.47
32	8.60	230160.29	1.92	51438.62
33	24.19	593205.82	1.72	42104.43
34	53.75	366447.46	1.70	11615.14
35	0.36	414.90	0.66	756.11
36	0.36	334.09	1.24	1147.83
37	6.88	19535.03	1.36	3853.43
38	0.01	1.12	0.79	145.76
39	0.00	9.26	0.85	1858.39
40	7.30	18365.07	9.18	23103.96

Table E-5.--continued.

bsn #	Direct_eng (E12 sej/ha/yr)	Direct_eng (E12 sej/yr)	Fuel (E12 sej/ha/yr)	Fuel (E12 sej/yr)
41	13.82	32670.61	1.88	4518.80
42	14.66	80242.55	1.37	7515.64
43	56.73	286334.53	2.26	11420.77
44	3.43	63908.08	2.18	40820.96
45	16.01	81864.04	5.45	27880.75
46	6.17	95553.34	1.51	23266.83
47	6.67	30629.52	5.51	25360.38
48	4.81	36719.53	2.16	16634.75
49	85.11	69190.38	1.52	1238.93
50	351.95	488155.58	13.74	19084.20
51	106.34	1309221.63	4.43	54746.09
52	9.42	13233.26	1.72	2414.09
53	47.51	57153.34	1.67	2013.26
54	20.46	50994.49	5.42	13515.19
55	373.00	234617.86	24.34	15312.63
56	200.20	1271481.22	10.08	64735.13
57	128.77	1199200.13	7.63	71775.84
58	322.57	539332.16	14.12	23604.22
59	151.68	63098.53	2.08	865.43
60	270.65	204879.02	6.51	4969.07
61	365.47	554414.78	21.51	32632.39
62	328.91	113145.06	16.26	5592.88
63	192.40	27897.38	6.60	956.46
64	320.42	228135.52	17.35	12353.10

Table E-6. Subbasin total empower and ELRs.

bsn #	Total Empower (E15 sej/ha/yr)	ELR
1	9.22	19.98
2	31.85	34.98
3	5.56	5.27
4	11.91	12.67
5	8.61	8.76
6	6.20	5.94
7	5.60	5.32
8	1.75	0.95
9	3.17	2.69
10	216.83	275.18
11	2.54	1.88
12	9.14	9.27
13	17.16	18.89
14	5.01	4.63
15	4.71	4.29
16	1.37	0.76
17	6.41	6.20
18	10.66	12.83
19	2.45	1.92
20	6.87	6.77
21	16.24	17.95
22	6.51	6.42
23	2.62	1.92
24	2.18	1.55
25	2.67	2.00
26	10.02	10.32
27	5.35	5.37
28	3.10	2.46
29	23.54	27.50
30	1.55	0.74
31	48.37	56.91
32	11.43	12.44
33	26.88	30.69
34	56.50	67.18
35	1.93	1.16
36	2.50	1.81
37	8.97	12.93
38	1.78	1.00
39	1.74	0.99
40	17.45	18.67

Table E-6.--continued.

bsn #	Total Empower (E12 sej/ha/yr)	ELR
41	16.71	18.56
42	17.02	18.81
43	59.94	73.92
44	6.47	7.23
45	22.45	24.63
46	8.66	8.87
47	13.17	13.83
48	7.95	8.21
49	87.53	118.53
50	366.87	538.84
51	111.74	144.84
52	12.18	12.90
53	50.28	59.87
54	26.92	30.61
55	398.51	892.86
56	211.36	299.17
57	137.51	170.35
58	337.86	507.63
59	154.92	202.97
60	278.33	363.31
61	388.13	774.47
62	346.32	549.40
63	200.04	300.57
64	338.90	572.40

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Chapter 5

Department of Environmental Protection Presentations

(May 1, 1998)

Mark T. Brown

Development of a Spatial Model of Pollutant Loading and Water Quality for Florida Watersheds

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Organization of Presentation

- Problem Statement
- Review of Other Pollutant Loading Models
- GIS Basics
- Project Overview
- Summary and Conclusions

Problem Statement

- Non-point sources of pollution have increasingly become the focus of attention
  - Elimination of point sources has increased attention
  - Increasing urbanization/development has heightened problem
- Non-point sources of pollution are spatial phenomena
- Management of Non-point sources requires a spatial perspective

Plan of Study

- Develop a spatial model of pollutant loading that:
  - » capitalizes on GIS capabilities
  - » accounts for temporal variation and landscape heterogeneity
  - » applicable to most watersheds
  - » easy to use, yet elegant

Resource Management

The model will be beneficial for use in resource evaluation and management

Estimate degree of impairment of water bodies

Spatially locate areas of impairment and identify sources of non-point source pollution

Direct field monitoring where a) the model predicts impairment and b) uncertainties are greatest

Model effects of changes in land use/land cover (anticipatory planning)

Watershed pollutant modeling approaches

- Dynamic simulation – uses equational structure for continuous simulation of pollutant transfer, decay, dilution
  - •Complex
  - •Non-spatial
  - •Data intensive
- Computational (ie spreadsheets) – calculates pollutant loads based on "pollutant accounting"
  - •Data intensive
  - •Non-spatial
  - •High resolution
- •Statistical – uses statistical measures (ANOVA, regression) to account for pollutant loading
  - •Non-spatial
- •GIS Spatial modeling – combination of any or all of the above in a spatial environment

GIS Based Pollutant Load Models

- SLOSS-PHOSPH: Sediment and Phosphorus Predication
- AGNPS: Agricultural Nonpoint Source Pollution Model
- LWWM: A Linked Watershed Waterbody Model
- BASINS: Version 1.0 (Version 2.0 Still in Beta release)
- Pollutant Load Screening Model (PLMS): GIS screening model

SLOSS-PHOSPH: Sediment and Phosphorus Predication

- GIS coupled model, rural only, annual only, no direct hydrology, and no pollutant routing.
- Output: Mean annual loads of sediment and phosphorus.

AGNPS: Agricultural Nonpoint Source Pollution Model

- Output: Storm runoff volume and peak flow. Sediment, nutrient, and COD concentrations ... viewed as 1) color GRASS maps displaying 4 ranges of simulated contaminant yields for the entire watershed or 2) a summary table for any cell selected.
- GIS (GRASS) used to compute values for most input variables.

LWWM: A Linked Watershed Waterbody Model

- Developed under contract to the Southwest Florida Water Management District by Dames & Moore, Inc. and ASci Corp.
- Uses GIS land use land cover and soils data layers for input
- Output: event driven runoff loads of nonpoint sources with SWMM, which feeds loadings and water volumes to hydrodynamic model (RIVMOD OR DYNHYD) to describe flow distribution in surface water and finally feed this information to WASP to simulate in stream impacts.

BASINS: Version 1.0 (Version 2.0 Still in Beta release) developed by Environmental Protection Agency

- Uses GIS data layers including: USGS land use/land cover, watershed boundaries, stream reaches, point sources, etc.
- Assessment and Planning Module organized under the GIS software
- Modeling Module uses:
  - NPSM-_HSPF: Version 10, a nonpoint source model for estimating loadings
  - QUAL2E: water quality and eutrophication model
  - TOXIROUTE: screening level stream routing model for dilution/decay of a stream system within a cataloging unit.
- Output: 50 water quality parameters at state or regional level

Pollutant Load Screening Model (PLMS)

St. Johns River Water Management District

- GIS Screening Model
- Uses GIS land use/land cover, soil hydrologic groups
- Output: total annual load, annual load per acre for: runoff, total nitrogen, total phosphorus, suspended solids, biochemical oxygen demand, lead, zinc

Other Pollutant Load Models

- Indian River Lagoon Pollutant Load Reduction Model (IRL-PLR Model)
- Watershed Management Model (WMM): Spreadsheet model
- SWMM: Storm Water Management Model: Dynamic spatial simulation

GIS Basics...

There is increasing evidence that not only is the earth not flat, but the occurrence of phenomena across the landscape is far from flat. We live in a lumpy world. Yet we treat it as one dimensional whenever we treat spatial data as averages, and then trip over lumps.

In other words, treating spatial data as we treat digital data (averages, normality, etc) is like flattening the earth.

GIS Basics...

The value of GIS lies in its ability to analyze spatial data as we once analyzed digital data.

GIS Basics...

Geographic Information System (GIS) - An organized collection of computer hardware, software, geographic data, and personnel designed to efficiently capture, store, update, manipulate, analyze, and display all forms of geographically referenced information. Certain complex spatial operations are possible with a GIS that would be very difficult, time-consuming, or impractical otherwise.

GIS Basics...

In the past, maps were primarily descriptive. They showed the precise locations of physical features, and other information needed for navigation.

Increasingly, maps have become prescriptive. Serving as data for determining appropriate management actions.

GIS Basics...

- Early uses of GIS involved **CARTOGRAPHIC MODELING**... mimicking manual map processing like:
  - map reclassification
  - overlay, and
  - simple buffering around features
- The new wave of applications concentrate on **SPATIAL MODELING**... Involving:
  - spatial statistics, and
  - advanced analytical operations

GIS Basics...

- Three categories of spatial modeling
  - **data mining**
  - **predictive modeling**
  - **dynamic modeling**

GIS Basics...

- **Data mining (data correlation)**...uses GIS to discover relationships among mapped variables.
  - For instance one map can be statistically compared with other "driving variable" maps, seeking correlation between the dependent map variable and the driving variables

GIS Basics...

Predictive modeling...most predictive modeling is non-spatial...involving regression equations. Average values for measured variables are used to solve a math model

A GIS solution spatially interpolates the field data into mapped variables, then solves the equation for all locations in space

GIS Basics...

Dynamic modeling.....usually interacts with a spatial model

1. Sensitivity analysis of driving variables determines relative weight of each mapped variable...
2. Address "what if" questions....

GIS Basics...

Maps as Data and Data Structure Implications

Maps actually map spatial variation.

- Manual cartographic manipulation is limited by its non-digital nature.
- Traditional statistics and mathematics are digital, but are limited by their generalization of spatial data
- Maps are now seen as geographically referenced digital data quantifying a system in prescriptive terms.

GIS Basics...

Implications of data structure.....

- The data structure used for storage has far reaching implications in how the data is encoded, analyzed, and displayed as digital maps
- All GIS are internally referenced...linking the data (thematic attributes) and the whereabouts (locational attributes) of those data.

TWO BASIC APPROACHES in describing locational attributes

VECTOR.....**RASTER**

GIS Basics...

While there are **significant practical differences** in these data structures, the primary theoretical difference is...

- GRID STRUCTURE stores information on the interior structure of areal features...implying boundaries
- LINE STRUCTURE stores information about boundaries, and implies interiors.

GIS Basics...

As a result of these differences thematic attributes are stored differently...

RASTER = set of numeric codes for each cell occupying interior space of an areal feature

VECTOR = a single numeric code assigned to the implied interior of a boundary

GIS Basics...

Vector vs. Raster Structure in Map Analysis

A grid-oriented system calculates the coincidence of variables at each cell location as if each were an individual polygon. Because these polygons are organized as a consistent uniform grid, the calculations are simply numeric evaluation, not geometric calculations for intersecting lines.

GIS Basics...

So what does this all mean...?

Whereas, in the past, data were collected from the landscape and may have been related to other variables that were characteristic of the landscape...(a correlation of 2 or more variables that change in space)

with GIS it is now possible to treat maps as variables and whatever mathematical or statistical operation one can use on a data set of variables, one can use on a set of maps

GIS Basics...

In this way, every map is a "surface" of data that can be added, subtracted, multiplied, divided, or correlated to, any other surface of data and variation, correlation, deviation can be evaluated as it changes in space.

GIS Basics...

And...

Probably most important, it is possible to analyze the influences of space, and spatially varying attributes on resource management issues

In other words...to use spatial analysis and spatial modeling to address resource management questions

GIS Basics...

Use of GIS to evaluate sensitivity to development

An example of cartographic manipulation of spatial data in the St. Marks Watershed

GIS Basics...

In this example, coverages of:

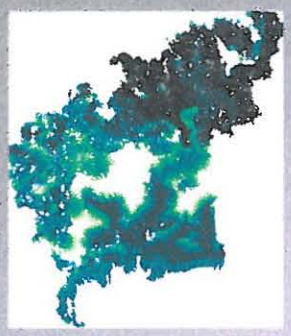
- Soils
- Water and wetlands
- Elevation, and
- Surficial geology

Are used to evaluate sensitivity to development based on potential impacts to surface and ground waters

Water Bodies and Wetlands

Water bodies and wetlands coverage is used to generate a coverage of effective distance by "spreading"

Effective Distance Coverage



Soil modified effective distance is generated by spreading from water bodies and wetlands over slope and modifying the resulting distance by soil permeability

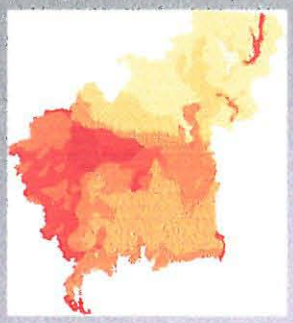
Surface Water Limitations

Surface water limitations are those areas that have the highest constraints on development because of potential impacts on surface waters. The coverage results from joining effective distance and water bodies and wetlands

Soils: Relative Permeability

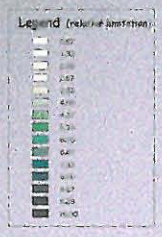
Relative Permeability is derived from hydrologic soils groups and used to determine groundwater limitations

Groundwater Limitations



Ground water limitations are those areas that have the highest constraints on development because of potential impacts on ground waters. The coverage results from joining Soil Permeability and geology of upper 30 meters.

Combined Surface and Ground Water Limitations



Combined limitations results from the sum of groundwater and surface water limitations

GIS Basics...

End. Part 1

Development of a Spatial Model of Pollutant Loading and Water Quality for Florida Watersheds

(Part 2)

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Equational Structure (1)

$$SP_j = \sum_{i=1}^{n} P_{li} T_i e^{-\alpha d_i} \qquad (1)$$

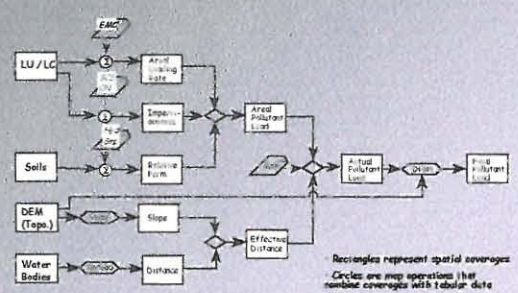
where
SP_j = non-point source pollutant load at stream cell j (kg/yr)
n = total cells in flow path to cell j
α = coefficient
d_i = overland flow distance of cell i from stream channel
P_{li} = pollutant load of land use l for cell i (kg P per cell per year)
$T_i = f(S_{li} Sl_i)$ = Transfer coefficient for cell i ($0 < T < 1$)
S_{li} = soil permeability of soil type l for cell i (% runoff)
Sl_i = Slope of cell i ($0 < Sl < 1$)

Equational Structure (2)

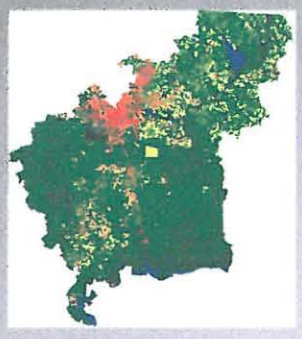
$$WP = \sum_{j=1}^{m} \left(PS_j + SP_j \right) A_j \qquad (2)$$

where
WP = Total pollutant load for watershed (kg/yr)
PS_j = point source load at cell j
SP_j = non-point source pollutant load at stream cell j (kg/yr) (from Eq 1)
m = number of stream cells along the water course
A_j = in-stream attenuation for stream cell j ($0 < A < 1$, based on stream slope)

GIS Spatial Model of Pollutant Loading



Land Use / Land Cover

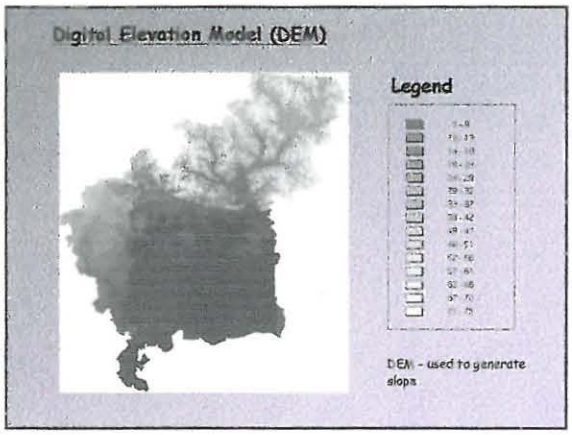


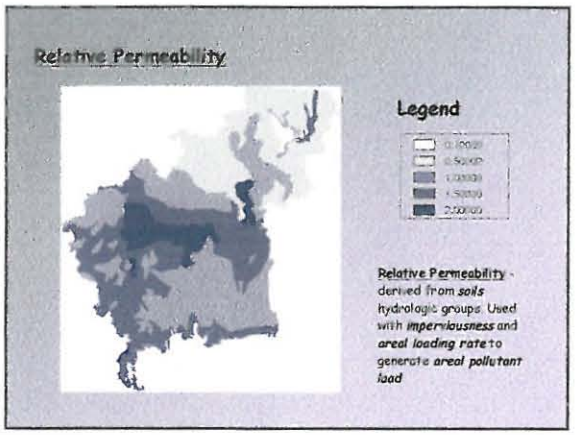
Legend

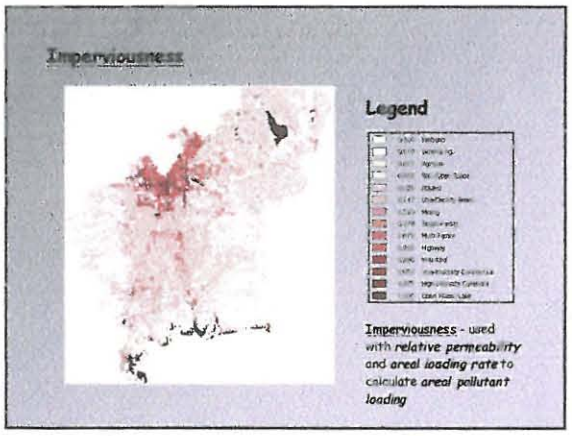
Land Use/ Land Cover - used with event mean concentrations (EMC's) to generate *areal loading rate* and to calculate *impervious ness*

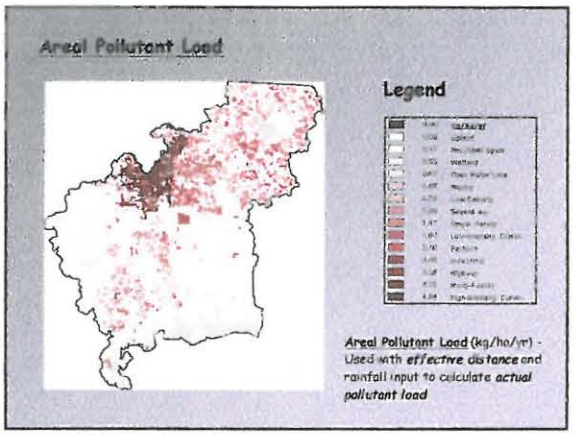
Water bodies

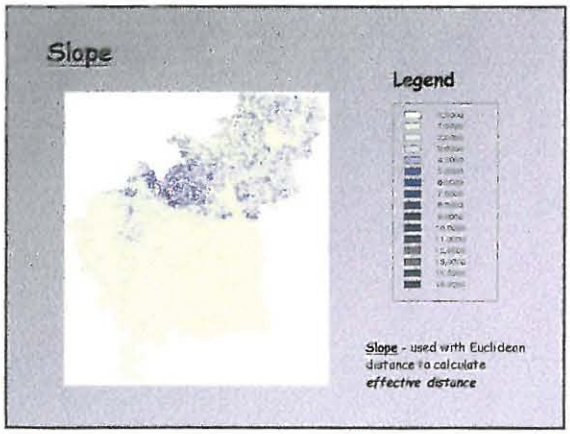
Water bodies - "spread" to generate euclidean *distance*

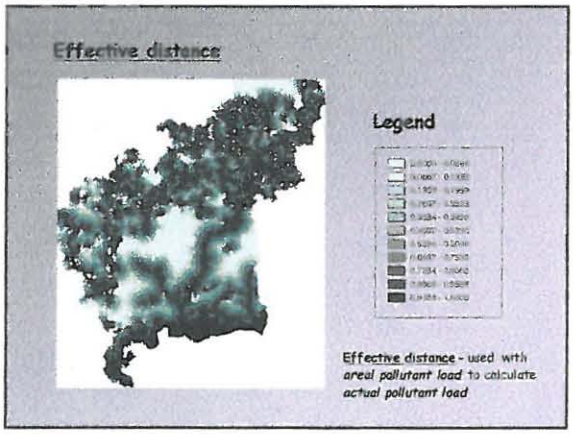

Digital Elevation Model (DEM)
Legend
DEM - used to generate slope


Relative Permeability
Legend
Relative Permeability - derived from soils hydrologic groups. Used with imperviousness and areal loading rate to generate areal pollutant load

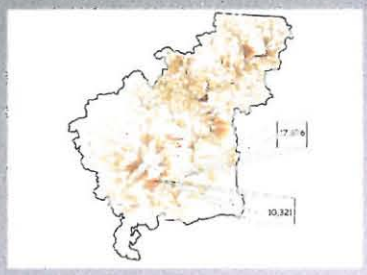

Imperviousness
Legend
Imperviousness - used with relative permeability and areal loading rate to calculate areal pollutant loading


Areal Pollutant Load
Legend
Areal Pollutant Load (kg/ha/yr) - Used with effective distance and rainfall input to calculate actual pollutant load

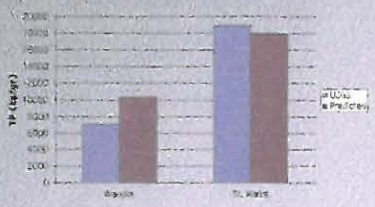

Slope
Legend
Slope - used with Euclidean distance to calculate effective distance


Effective distance
Legend
Effective distance - used with areal pollutant load to calculate actual pollutant load

Pollutant Load Coverage



Final Pollutant Load - ***actual pollutant load*** is "drained" over ***topography*** to calculate total load for any cell within the watershed based on any given rainfall event, or integrated over any time interval.



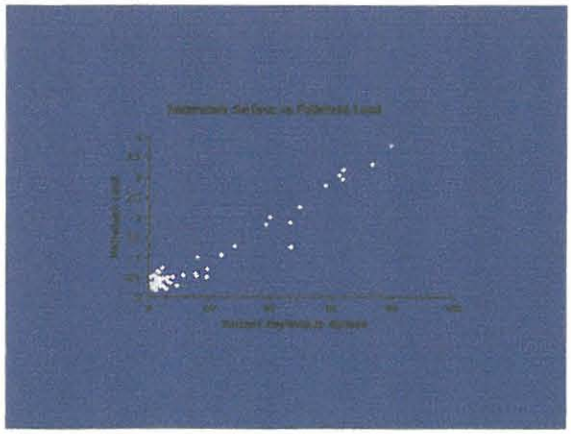
Landscape Development Intensity (LDI)

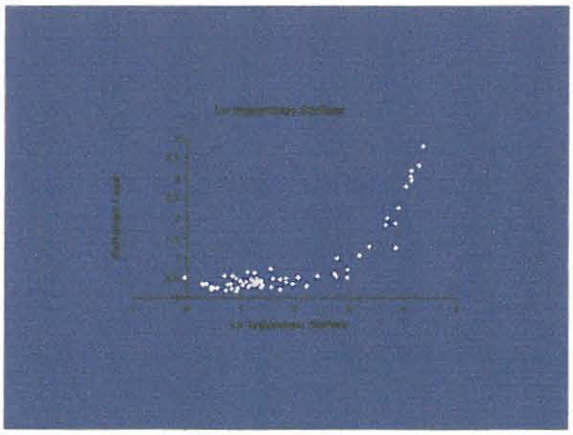
LDI is any index of development intensity that relates spatial data as a continuous variable. In this way spatial statistics can be used to evaluate correlations between development and pollutant generation.

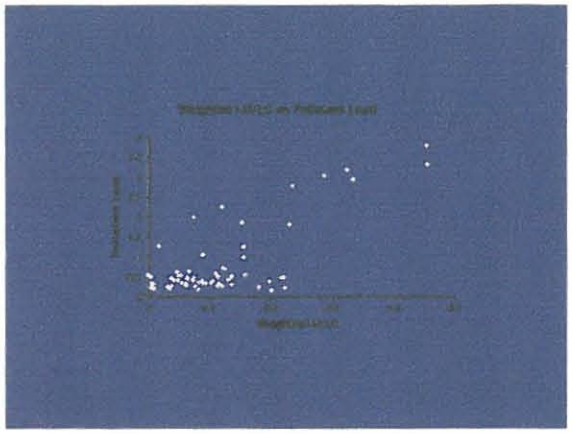
Landscape Development Intensity (LDI)

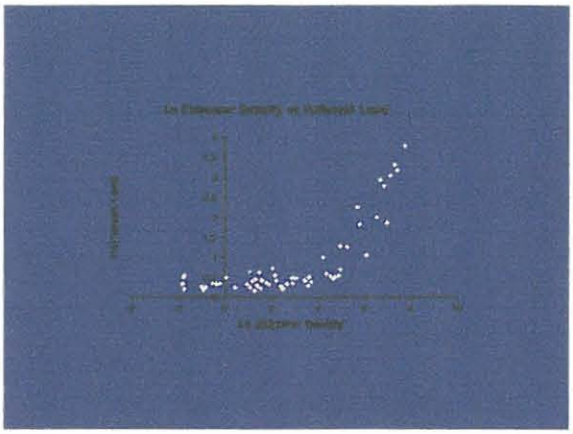
Examples of LDI's...

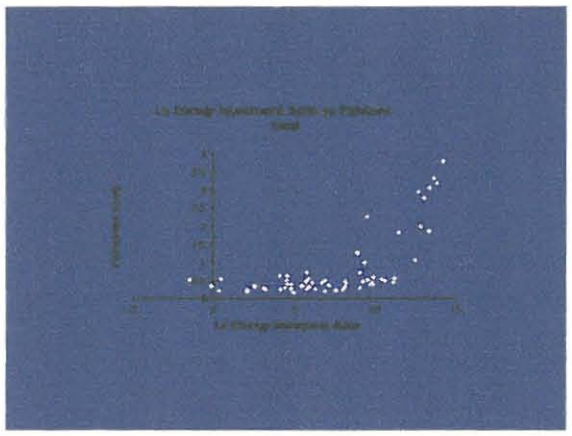
1. Impervious Surface
2. Weighted % Urban, Agriculture, Natural
3. Empower Density
4. Emergy Investment Ratio
5. Environmental Loading Ratio

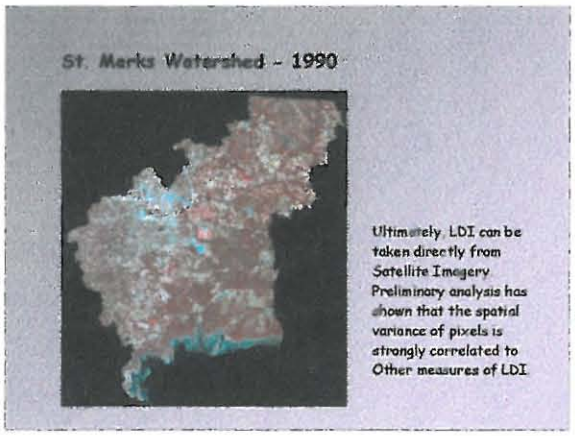











St. Marks Watershed - 1990
Ultimately, LDI can be taken directly from Satellite Imagery. Preliminary analysis has shown that the spatial variance of pixels is strongly correlated to Other measures of LDI.

Chapter 6
Development of a Spatial Model of Pollutant Loading and Water Quality for Florida Watersheds

a Proposal to
Florida Department of Environmental Protection
by

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Project Overview

This proposal is to continue to develop a spatially based pollutant loading model for surface waters using GIS data layers consisting of: land use/land cover, soils, and topography. The model modifies land use generated non-point source pollutant loading based on distance from surface water body, slope, and intervening soils and land cover and adds point sources from the NPDES data base. Non-point sources pollutant load received by the surface water body is a function of lateral distance decreased by slope, so that higher slopes between water body and pollutant generator decrease distance. Soils also modify pollutant load received by the surface water body through an inverse relationship to permeability. Pollutant load carried by the water course is decreased by in stream uptake that is a function of basin slope. The following equations describe the GIS model:

$$SP_j = \sum_{i=1}^{n} P_{li} T_i e^{-\alpha d_i} \qquad (1)$$

where :

SP_j = non-point source pollutant load at stream cell **j** (kg/yr)
n = total cells in flow path to cell j
α = coefficient
d = overland flow distance of cell **i** from stream channel
P_{li} = pollutant load of land use **l** for cell **i** (kg P per cell per year)
$Ti = f(S_{ti} SL_i)$ = Transfer coefficient for cell **i** (0<T<1)
S_I = soil permeability of soil type **t** for cell **i** (% runoff)
Sl_i = Slope of cell **i** (0<SL<1)

The total pollutant load at any point along a water course is the sum of non-point pollutant load from Equation 1 and point sources introduced directly into the water body, modified by in stream uptake as follows:

$$WP = \sum_{j=1}^{m} \left(PS_j + SP_j \right) A_j \qquad (2)$$

where:

$\boldsymbol{WP}$ = Total P load for watershed (kg/yr)
$\boldsymbol{PS_j}$ = point source load at cell **j**
$\boldsymbol{m}$ = number of stream cells along the water course
$\boldsymbol{A_j}$ = in-stream attenuation for stream cell **j** (0<A<1; based on stream slope)

The flow diagram in Figure 6-1 shows the GIS framework for the model.

At the present time the model generates annual load at any point in a watershed. We are generally impressed with the results, however the model has not been adequately validated or tested outside the St. Marks basin. In addition, the model does not include point source inputs to water courses. This proposal addresses validation of the model and the inclusion of point source data.

GIS software presently used for modeling spatial pollutant loads is a simple, but powerful raster based map analysis program called MAP•Factory, by Thinkspace, Inc. During the proposed project the model will be converted to ARC-Info *Spatial Annalist* and Macro's written that will reproduce the model in the ARC-View environment. A manual for use of the model will be written and included as part of the final report to DEP.

The proposed project will further validate the model by increasing the number and variety of watersheds for which predicted verses observed annual pollutant loads will be correlated using STORET data and point source data from NPDES permits . Annual loads for several hundred small watersheds throughout the state of Florida will be obtained from the STORET data base, along with watershed boundaries, and point source data. The actual number of watersheds depends on the number meeting selection criteria as follows: 1) Water quality and point source data are dependable, 2) Watershed boundaries can be accurately demarcated, 3) Stream pollutant load is dominated by point and non-point sources within the basin (i.e. entire watershed is incorporated in study), or pollutant load inflowing to the watershed is known, 4) Sampling station can be accurately located within the basin, and 5) Land use/ land cover, topography, and soils coverages exist for basin. Of lessor importance is the existence of a USGS gauging station near the water quality sampling station (while it is necessary that some of the watersheds have USGS flow data, not all of them need be gauged)

Calibration of the GIS model is an iterative process. First the model will be calibrated based on stream discharge. A subset of the total population of selected watersheds with discharge data will be used to calibrate overland flow . Then using the same subset of watersheds the model will be calibrated for annual P load. Calibration consists of setting parameters and using the GIS framework to spatially calculate discharge or load and then compare predicted verses observed discharge of load using linear regression. Model parameters will be adjusted, applied again to the sub-population of watersheds to give best fit to overall data. The iterative process will continue until there is no improvement in the regression equation's R^2.

Once calibration is completed. The model will be applied to calculate P loads for watersheds, correlating base flow conditions and wet season conditions with model output. The model to date has used only total phosphorus.

The model will be beneficial in the following applications:

1. Estimate degree of impairment of water bodies,
2. Spatially locate areas of impairment and identify sources of non point source pollutants,
3. Direct field monitoring where: (a) the model predicts impairment and (b) where uncertainties are the greatest (i.e. suggest monitoring plan based on model output),
4. Predict future impairment based on changes in land use/land cover, and
5. Model could be used as the 305B model to estimate % of state water bodies that are within compliance.

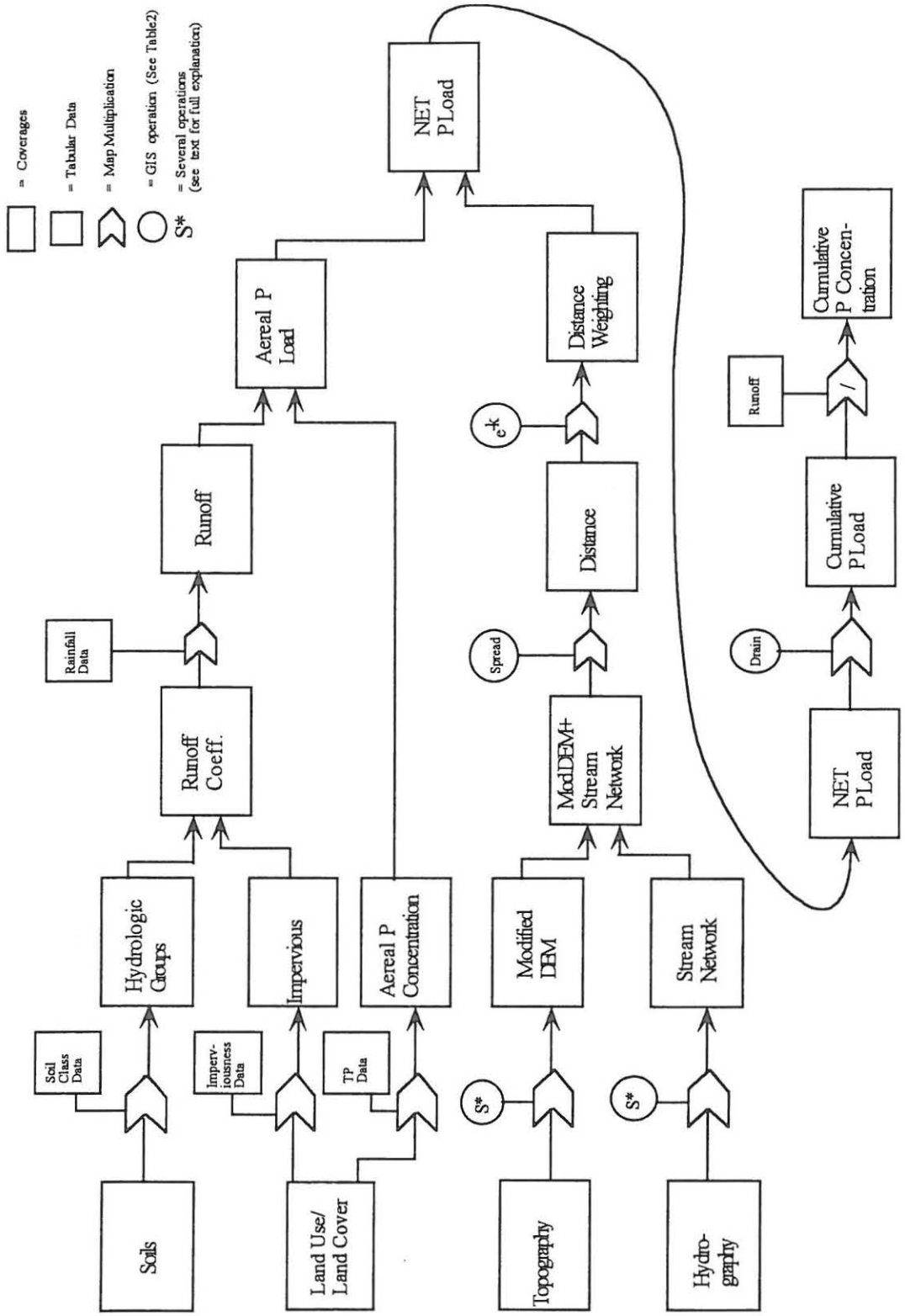

Figure 6-1. Flow diagram of the spatial model in the GIS framework

Future model refinements include correlating estimated pollutant loads with indices of Land Development Intensity (LDI) and incorporation of temporal simulation. Correlating model estimates with LDI's is the first step in eventually using satellite images directly to estimate pollutant loading. Preliminary research using satellite images covering Miami, Tampa, Orlando, and Jacksonville indicates that there are some relationships between reflectance in different bands of the images and intensity of use. Indices using satellite image bands in various combinations will be correlated with LDI's and then with model output.

Finally, work is progressing on a temporal simulation model that predicts water quality on a daily basis using real rainfall events over a year time frame. In the future the temporal model will be combined with the spatial GIS model to generate continuous water quality within a basin that results from continuous simulation of rainfall and runoff.

Scope of Work

The following scope provides details of project timing and deliverables to DEP over the two-year duration of the project.

Task 1. Select watersheds statewide and collect data
- 1.1 Evaluate STORET data to determine reliability of data, length, and frequency of measurements
- 1.2 Obtain point source NPDES data base for selected watersheds
- 1.3 Obtain soils coverages for each watershed
- 1.4 Obtain DEM's for each watershed
- 1.5 Obtain land use/land cover for each watershed

Task 2. Calculate annual phosphorus loads and P loads for wet season, and baseflow conditions
- 2.1 Use STORET data to calculate annual loads based on concentration and discharge
- 2.2 Using wet season months calculate wet season P loads
- 2.3 Using dry season months, calculated dry season P loads

Task 3. Calibrate model (Water discharge)
- 3.1 Using a subset of the selected watersheds (selection based on availability of good water discharge data), determine contributing area within each watershed based on DEMs
- 3.2 Using published runoff coefficients for land use types, calibrate model through best-fit correlations between predicted and observed discharge

Task 4. Calibrate model (P loads)
- 4.1 Calculate in-stream P attenuation coefficients using literature values and STORET data
- 4.2 Select a subset of the selected watersheds [selection based on: 1) watershed contains a 1st, 2nd, or 3rd order stream; 2) Minor NPDES activity within the basin; 3) All stream inflows from other watersheds have known annual loads]
- 4.1 Using published P loads per unit area of different land use types, calibrate model through best fit correlations between predicted and observed discharge.

Task 5. Test and refine model
- 5.1 Using the full data set of selected streams statewide, apply the model to predict discharge and P load
- 5.2 Compare predicted verses observed values
- 5.3 Refine model if necessary

Task 6. Final Report
- 6.1 Prepare final written report including documentation and statistical verification
- 6.2 Prepare model for presentation to DEP and train staff in use
- 6.3 Convert model to ARC-Info *Spatial Annalist* and prepare manual.

Project Timing and Deliverables

Task	Description	Due Date	Payment
Task 1 Select watersheds.	Working with DEP staff, select watersheds where data exist that represent a cross section of both size and intensity of development	3 months after project initiation	$10,000
Task 2 Calculate annual, wet, and dry season loads	Select a subset of streams where there are both discharge and water quality data	6 months after project initiation	$10,000
Task 3 Calibrate model for stream discharge	Using the subset of streams calibrate model by best fit between observed and predicted discharge	12 months after project initiation	$15,000
Task 4 Calibrate model for P load	Using the subset of streams calibrate model by best fit between observed and predicted P load	18 months after project initiation	$15,000
Task 5 Test and refine model	Using the full data set test and refine model	21 months from project initiation	$20,000
Task 6 Prepare final report	Working with DEP staff prepare final report and provide model and training	24 months from project initiation	$10,000