AN INVESTIGATION OF THE SOURCES AND TRANSPORT OF NONPOINT SOURCE NUTRIENTS IN THE ILLINOIS RIVER BASIN IN OKLAHOMA AND ARKANSAS

By

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CHAPTER I

INTRODUCTION

Nitrogen and phosphorus are considered by most limnologists to be the nutrients most commonly limiting algal growth in surface waters. Quantities of these plant nutrients that are available for transport to surface waters are influenced by anthropogenic activities. When these nutrients are present in surface waters in excessive quantities, algal populations tend to experience seasonal growth blooms that may detract from aesthetic characteristics of streams and lakes, and contribute to accelerated eutrophication of water bodies.

Watersheds are ecosystems composed of a mosaic of different land uses connected by a network of streams. In-stream conditions are determined by processes occurring within the watershed and cannot be isolated from or manipulated independent of the watershed. In this study, the stream network of interest drains into a single large reservoir, and the water quality in the reservoir is then directly related to conditions and processes occurring within the watershed. An evaluation of the movement of certain pollutants, such as the nutrients nitrogen and phosphorus, from the watershed to the reservoir should include information about land uses, soil types, geology, topography, precipitation, and surface drainage patterns. Geographic information systems

(GIS) allow for the inclusion of all these aspects in an overall model of nutrient transport.

The Illinois River Basin in Oklahoma and Arkansas has been the subject of numerous water quality studies, primarily concerned with factors affecting water quality in the scenic river portions of the Illinois River in Oklahoma, and nutrient loading to Lake Tenkiller. The state (Oklahoma) scenic river status of the Illinois River from Watts to Tahlequah, and the tributaries Flint Creek and Baron Fork Creek, legislates that these designated stream sections receive special attention regarding increases in levels of pollution.

The area including the Illinois River Basin has experienced significant economic development in the past two decades. Changes that likely have affected the water quality of the Illinois River and Lake Tenkiller include expanding urban development and an increase in confined animal production. While assessing blame on any particular industry or urban center within the basin without a definitive study is difficult, and perhaps politically unwise, there is evidence that an expanding population and a booming poultry industry are significant contributors to the increasing levels of nutrients transported via the Illinois River and its tributaries to Lake Tenkiller. A 1989 Soil Conservation Service (SCS, 1989) inventory estimated more than 93,400,000 birds are raised in the basin per year producing approximately 366,000,000 kilograms of manure per year. Estimated nitrogen (N) and phosphorus (P) quantities in this manure are 4,680,000 kilograms N per year and 1,400,000 kilograms P per year. These nutrients are used to fertilize pasture and crop lands in the basin to promote the

growth of forage for cattle. Significant fractions of this fertilizer are likely not incorporated into herbaceous or animal biomass and are susceptible to transport into surface streams during runoff events, and leaching, especially for nitrogen, to ground water.

Lake Tenkiller, the ultimate receptacle of nutrients flowing into the Illinois River and its tributaries, has displayed evidence of accelerated eutrophication over the past twenty years. The influx of the nitrogen and phosphorus provide essential nutrients for the growth of algal populations. Dense spring algal blooms have been observed in the reservoir in recent years and, during periods of summer stratification, the hypolimnion of the reservoir has become anoxic. Continued input of these nutrients will likely exacerbate problems already observed and contribute to decreased aesthetic and water quality in the reservoir.

The objectives of the study were to:

1) identify the areas and land uses which are most highly correlated with the transport of the nutrients nitrogen and phosphorus from the watershed to tributaries, the Illinois River, and Lake Tenkiller;
2) quantify the amounts of nitrogen and phosphorus coming from point and nonpoint sources using a comprehensive nonpoint source hydrologic water quality model; and,
3) determine the potential beneficial effects, in terms of

reduced nutrient loading, of specific reductions of nonpoint nutrient quantities within the basin.

A continuous simulation phosphorus transport model called SIMPLE (Sabbagh et al., 1995) developed by the Biosystems and Agricultural Engineering Department at Oklahoma State University, interfaced with the GIS GRASS 4.0 (CERL, 1988), was used to model nonpoint phosphorus transport from subwatersheds of the Illinois River Basin to the Illinois River and its tributaries. The model incorporates the SCS Curve Number Method for determination of runoff, the Revised Universal Soil Loss Equation for the calculation of sediment loss, a linear isotherm to simulate the desorption of phosphorus in the soil matrix, and a mass balance accounting of the soil phosphorus. GIS data layers include topographical data from USGS 1:24,000 scale quadrangle maps, soil types and characteristics from county soil surveys, land uses derived from aerial photography, and animal densities and the locations of poultry houses.

This model was developed for the determination of long-term phosphorus loading. SIMPLE was used to estimate nonpoint source phosphorus export from the watershed to the receiving stream. This model was applied to subbasins defined by a Digital Terrain Model (DTM), a component of the SIMPLE Model, comprising the total Illinois River Basin including Lake Tenkiller.

Outputs from runs of the phosphorus transport model were routed through a riverine water quality model, QUAL2EU (Brown and Barnwell, 1987), incorporating uncertainty analysis, which simulates the fate of nutrients as they

move downstream to Lake Tenkiller.

The SIMPLE model does not simulate the transport of nitrogen, but nitrogen is a nutrient of interest to limnologists concerned with the effect of nutrient loads on streams and lakes (and reservoirs). Thus, unit area loadings for nitrogen, based on land uses, were developed based on literature values to estimate total nitrogen export from the defined subbasins. Unit area loads are simple estimates expressing pollution generation per unit area and time unit for each typical land use (Novotny, 1986). This method was used to estimate nitrogen export from subbasins in the watershed for use as QUAL2EU input. Results of the modeling effort were compared with nutrient loading quantities developed from historical discharge and concentration data measured at water quality monitoring stations in the Illinois River Basin.

The SIMPLE model, with its GIS interface, was used to identify those areas in the basin that are most highly correlated with phosphorus transport based on soil characteristics, land uses, slope, and fertilizer application. The SIMPLE model and unit area loadings were used to quantify nutrient loads available for transport to Lake Tenkiller. The QUAL2EU model was used to route and quantify point and nonpoint nutrient loads to Lake Tenkiller. The QUAL2EU model offers uncertainty analysis in the form of Monte Carlo simulations. These were employed to estimate the frequency distributions of nutrient concentrations and loads at specific locations on the Illinois River and its tributaries, and the upper portion of Lake Tenkiller, based on variable nutrient concentrations and stream discharge.

I analyzed historical water quality data to determine annual average nutrient concentrations and loads, temporal trends for nutrient concentrations and loads, and estimates of the sources of nutrients reaching the headwaters of Lake Tenkiller. These results, based on historical data, were compared with the modeled transport of nutrients down to the headwaters of Lake Tenkiller. Estimated reductions of nutrient transport to Lake Tenkiller due to reductions in nonpoint source inputs of nutrients were quantified. Trophic state indices were used to estimate the trophic state of the headwaters of Lake Tenkiller under simulated average annual conditions and with 25 and 50 percent reductions in nonpoint source inputs.

Chapter II is a brief description of the study area, the Illinois River Basin in Oklahoma and Arkansas. Chapter III is a review of literature applicable to this study touching on water quality in the Illinois River Basin, phosphorus and nitrogen properties and characteristics, nutrient unit area loading methods, characteristics of animal wastes, nonpoint source pollution, and nonpoint source pollution modeling.

Chapter IV outlines the methodology followed concerning data gathering, and analysis for automatic sampling selected stream sites, historical data, SIMPLE input data sets, and QUAL2EU data sets. Chapter V gives results of automatic sampling, analysis of historical water quality data, SIMPLE runs on the Illinois River Basin, and results of QUAL2EU model runs.

Chapter VI is a discussion of the results with comments about management practices, and, Chapter VII offers some concluding remarks.

CHAPTER II

DESCRIPTION OF THE STUDY AREA

The Illinois River above 650 feet mean sea level, was designated as a state "scenic river" in 1969 by an act of the Oklahoma Legislature in an attempt to preserve and protect the qualities of the river that make it unique and attractive. The Oklahoma Scenic Rivers Act of 1969 states " . . . some of the free-flowing streams and rivers of Oklahoma possess such unique natural scenic beauty, water conservation, fish, wildlife and outdoor recreational values of present and future benefit to the people of the state that it is the policy of the Legislature to preserve these areas for the benefit of the people of Oklahoma" (OK Statutes, Title 82 O.S. Supp. 1981, Sec. 1451).

A supplement to the Scenic Rivers Act in 1981 designated portions of two major tributaries of the Illinois River, Flint Creek and Baron Fork Creek, as state scenic rivers as well. The act provides that designated scenic river areas be preserved in their free-flowing forms, and directs and authorizes the Director of the Oklahoma Water Resources Board and other state water pollution control agencies to assist in preventing and eliminating pollution of waters within a scenic river area (OK Statutes, Title 82 O. S. Supp. 1981, Sec. 1451).

An Act of the Oklahoma Legislature in 1977 provided for the formation of theOklahoma Scenic Rivers Commission with responsibility to carry out the intended purpose of the Oklahoma Scenic Rivers Act. The Illinois River has been, and continues to be, a premier tourist attraction of the state, drawing thousands of people annually to northeastern Oklahoma. The region offers abundant camping, hunting and fishing, and canoeing opportunities in a setting that is unique to the Ozark region.

The Oklahoma Water Quality Standards of 1988 (OWRB, 1989) list beneficial uses of the scenic river portions of the Illinois River, Flint Creek, and Baron Fork Creek as: 1) public and private water supply, 2) smallmouth bass fishery, 3) primary recreation, 4) agriculture (non-irrigation), and 5) aesthetics. These standards list no numerical standards for nutrient concentrations. However, an Anti-Degradation Policy in section 200.4, which applies to designated scenic river segments, states "no degradation shall be allowed in waters which constitute an outstanding resource or have exceptional recreational value and/or ecological significance."

The listed beneficial use as aesthetics does have a stipulation regarding nutrients. It states, "nutrients from point source discharges or other sources shall not cause excessive growth of periphyton, phytoplankton, or aquatic macrophyte communities which impairs any existing or designated beneficial use." These same scenic river segments are also classified as "Outstanding Resource Waters" and as such, non-point source discharges are to be controlled using best management practices in the watersheds.

Illinois River Basin

The headwaters of the Illinois River are in the Boston Mountains of northwestern Arkansas in Washington County, about 24 kilometers southwest of Fayetteville. The stream flows in a northerly and westerly direction through this Ozark region, crossing the Oklahoma/Arkansas state line near Siloam Springs, Arkansas. The river continues in a westerly direction until it is joined by Flint Creek. It then flows in a southerly direction to its confluence with the Arkansas River in Sequoyah County near Gore, Oklahoma. The river flows approximately 260 kilometers from its headwaters to its confluence with the Arkansas River (Figure 1).

Two of the major tributaries of the Illinois River also have their origins in the Ozark region of Arkansas. Flint Creek, originating in Benton County, Arkansas, flows in a westerly direction out of Arkansas through Delaware County, Oklahoma and joins the Illinois River from the north just south of Kansas, Oklahoma. Baron Fork Creek originates in Washington County, Arkansas and flows in a southwesterly direction to its confluence with the Illinois River just south of Tahlequah, Oklahoma. Both of these tributary basins are largely forested. Osage Creek, a smaller tributary of the Illinois River, flows southwesterly from Rogers, Arkansas to its confluence with the Illinois River about 16 kilometers east of Siloam Springs, Arkansas. Sager Creek is a tributary of Flint Creek that originates just east of Siloam Springs, Arkansas and flows northwesterly joining Flint Creek in Delaware County, Oklahoma about five kilometers west of the Oklahoma/Arkansas state line.

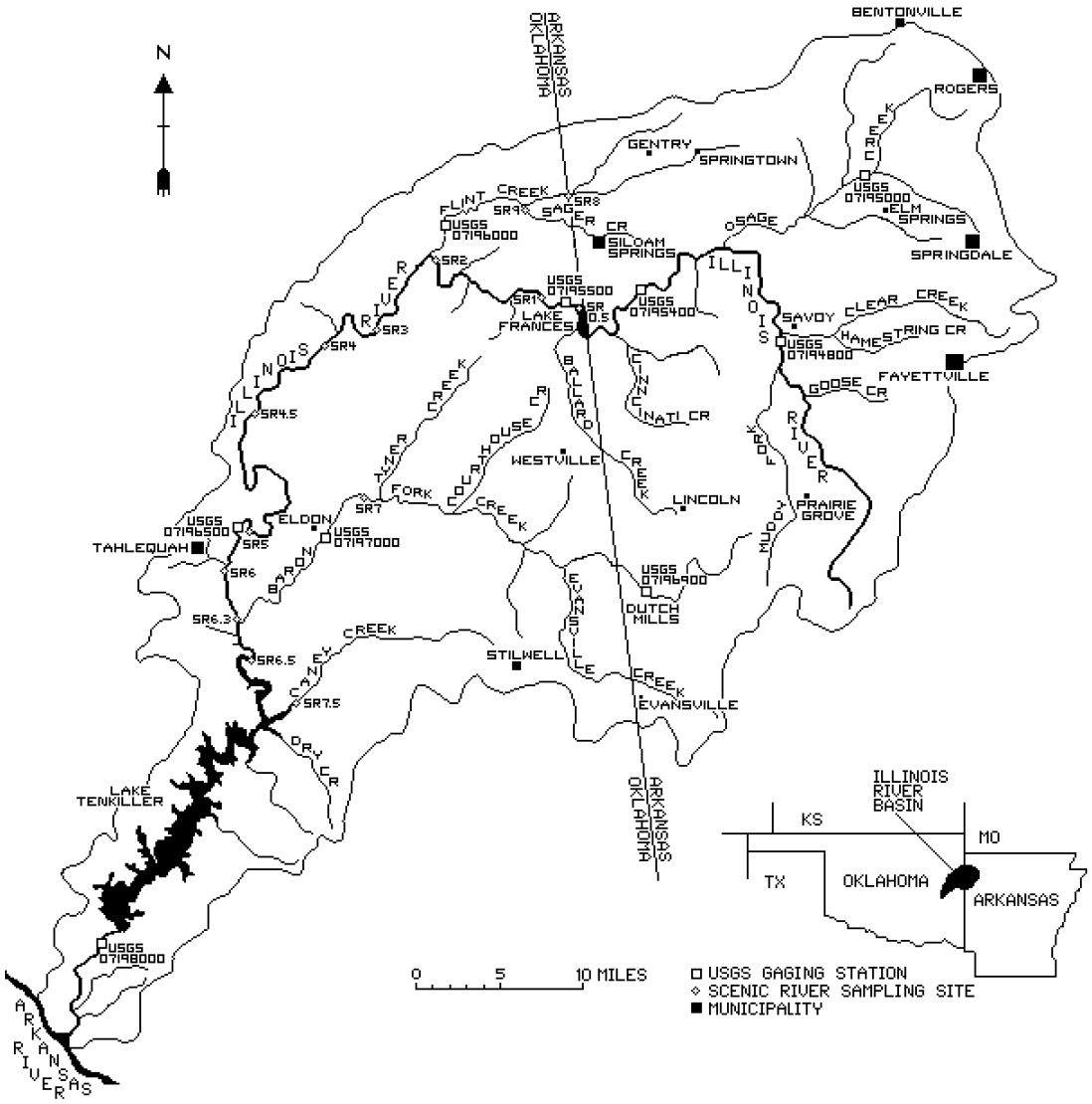

Figure 1. A Location Map of the Illinois River Basin in Oklahoma and Arkansas Indicating Locations of Water Quality Monitoring Stations, Municipalities, the Illinois River and its Major Tributaries.

The Illinois River Basin, including about 4,300 square kilometers, lies within the southwestern portions of the physiographic province called the Ozark Uplift that covers nearly 103,600 square kilometers in Missouri, Arkansas, and Oklahoma. Approximately 53% of the Illinois River Basin is in Oklahoma while the remaining 47% is in Arkansas (Lyhane, 1987). The Illinois River and its

tributaries are included in a part of the Ozark Uplift called the Springfield Plateau. This plateau is generally deeply dissected with rolling upland areas separated by V-shaped stream valleys that range from 60 to 90 meters in depth. Geologic processes have created cliffs of erosion resistant rock along much of the Illinois River and to a lesser extent on the Flint Creek and Baron Fork Creek basins (U.S. Dept. of Interior, 1979).

Soil types in the basin range from soils derived from sandstones, shale, clay, and some limestones. These soils support vegetation ranging from tall grasses to oak, hickory and pine forests (Lyhane, 1987). Estimated land use percentages calculated by the U.S. Dept. of Agriculture - Soil Conservation Service of Arkansas and Oklahoma (USDA-SCS AR/OK) in a 1989 report show that about 42% of the basin is forested, 48% is grassland, 3% is urban, 2% is cropland, and the remaining 5% is a mixture of water, feedlots, and other minor land use types.

Georeferenced land uses in the Illinois River Basin were obtained from the U.S. Environmental Protection Agency (U.S. EPA). Maps were derived from photo interpretation of 1:24,000 scale color infrared aerial film positives flown from August 30 - September 1, 1985. This information was transferred to clear mylar overlays based on USGS 7.5 minute (1:24,000 scale) quadrangles. Digitization and polygon attribution was performed based on these mylar overlays. Land uses derived from this information were reclassed into seven broad categories for use in the SIMPLE model. Table I shows the land use/land cover statistics for the Illinois River Basin above the Lake Tenkiller Dam based

on these 1985 data. The results are very similar to Lyhane (1987) with 49% of the basin classified as Pasture/Range and nearly 44% Forest.

TABLE I

LAND USE TYPES AND AREAS FOR THE ILLINOIS RIVER BASIN IN OKLAHOMA AND ARKANSAS BASED ON 1985 DATA

Land Use Description	Oklahoma Portion Area (ha)	Oklahoma Portion Area (%)	Arkansas Portion Area (ha)	Arkansas Portion Area (%)	Illinois River Basin Area (ha)	Illinois River Basin Area (%)
crop	880	0.4	3,890	2.0	4,770	1.2
confined animals	224	0.1	1,400	0.7	1,620	0.4
forest	121,000	56.2	56,900	29.5	178,000	43.6
pasture /range	84,100	38.9	118,000	61.2	202,000	49.4
roads & ROW	478	0.2	654	0.3	1,130	0.3
urban	2,850	1.3	11,500	6.0	14,400	3.5
water	6,090	2.8	649	0.3	6,740	1.7
totals	216,000	100	193,000	100	409,000	100

The significant economic benefits of recreation on the Illinois River can be explained in part by the extensive use of the Illinois River by canoeists. The Oklahoma Scenic Rivers Commission has tallied the $1.00 per canoe user fee paid at the numerous canoe rental operations on the river. Results show 52,000 to 67,000 canoes were rented each year from 1984 to 1988 (USDA-SCS AR/OK, 1989). Lake Tenkiller is also a popular recreation area with more than 1.5 million visitors each year from 1992 to 1994 (OWRB, 1996).

Lake Frances

Lake Frances, one of two impoundments on the mainstem of the Illinois River, a 570 surface acre lake located in Adair County in Oklahoma and Benton County, Arkansas, was first impounded in 1931. In 1954, the City of Siloam Springs, Arkansas purchased the dam and most of the adjacent land with the intentions to rebuild the dam and use the reservoir as a water supply source. The dam is considered the upper limit of the scenic river portion of the Illinois River.

Water supply is the major commercial use of Lake Frances. It serves as a water supply source for Siloam Springs and other small communities in the area both in Arkansas and Oklahoma. The lake provides some recreational uses as a fishery but the lake is generally too shallow to be used for recreational boating other than fishing.

Concerns have been raised about the dam impounding Lake Frances. The U. S. Army Corps of Engineers and the Oklahoma Water Resources Board have declared the dam a safety hazard. The Oklahoma Water Resources Board has ordered the City of Siloam Springs to repair the aging dam. The City of Siloam Springs has since offered to sell the dam and the lake for a nominal fee. There are currently alternatives being discussed which include removing the dam and draining the lake, or repairing the dam and dredging the lake. Part of the top of the dam broke off during flooding in May 1990.

Lake Frances is relatively shallow with a mean depth of 1.2 meters. The lake has a short hydraulic retention time of about two days (Threlkeld, 1983).

The lake suffers substantial seasonal algal blooms that deter from its attractiveness as a recreational area. These blooms have been cited as a possible cause for decreased water quality in the Illinois River several kilometers below the dam (Burks et al., 1991).

Lake Tenkiller

The Lake Tenkiller dam is on the Illinois River about 11 kilometers northeast of Gore, Oklahoma. The lake extends more than 40 kilometers up the Illinois River in Cherokee and Sequoyah counties and at normal power pool of 632 feet mean sea level has a surface area of approximately 12,900 acres, 209 kilometers of shoreline, and a volume of 654,100 acre-feet (U. S. Army Corps of Engineers, 1988). The lake was completed by the U.S. Army Corps of Engineers in 1952 with authorized project purposes being flood control and hydro-power generation. The lake also serves as a water supply source for numerous municipalities in the immediate vicinity and is valued as a prime recreational facility (Nolen et al., 1988). The lake is primarily fed by the Illinois River with the main tributaries being Flint Creek, Baron Fork Creek, and Caney Creek which enters the lake directly.

Several studies have been performed on the lake to estimate its current and future trophic status because concerns have been raised about water quality deterioration in the Illinois River Basin. Because flow velocities along the mainstem of the Illinois River are relatively high even during low flow periods, and because Lake Frances has a low mean hydraulic retention time, nutrient

discharges from the upper Illinois River watershed are likely to end up in Lake Tenkiller (Walker, 1987a).

CHAPTER III

LITERATURE REVIEW

Illinois River Basin Water Quality Studies

Several water quality studies have been performed on various segments of the Illinois River Basin over the past 15 years. The Oklahoma State Department of Health (1977) conducted studies in the Illinois River Basin from June 1975 to October 1977 that included assessments of Lake Frances and Lake Tenkiller, the Illinois River from Lake Frances to Lake Tenkiller, and portions of Flint Creek and Baron Fork Creek. Lake Frances was described as being in the late stages of eutrophication. The impact of the outflow from Lake Frances was determined to extend downstream to the Illinois River's confluence with Flint Creek. Flint Creek was shown to be carrying elevated loads of nutrients. Baron Fork Creek was judged to have superior water quality. Water quality in the Illinois River generally improved going downstream from Lake Frances to Lake Tenkiller. Lake Tenkiller was described as having high water quality and was classified as mesotrophic.

Threlkeld (1983) conducted a diagnostic feasibility study for the potential restoration of Lake Frances from October 1981 to October 1982. The study

included regular sampling of sites in Lake Frances, inflows from the Illinois River and Ballard Creek, and the outflow from Lake Frances. The lake was described as very eutrophic and the primary cause was attributed to phosphorus entering the system from discharges from Springdale and Rogers' wastewater treatment plants (WWTPs). It was concluded that Lake Frances was heavily loaded with both nitrogen and phosphorus but that the lake retained negligible amounts of these nutrients partially due to the short hydraulic retention time of about 2.4 days.

Threlkeld (1983) concluded that dredging of the upper end of Lake Frances was necessary to increase the residence time of waters in the lake to allow for greater retention of nutrients by the lake. Also, the treatment of phosphorus in the WWTPs at Springdale and Rogers would greatly reduce phosphorus entering the lake. Nutrient loading from the Lake Frances watershed was determined to contribute to water quality degradation in the Illinois River downstream of Lake Frances.

The U.S. Geological Survey (Terry et al., 1984) conducted an extensive water quality study on the Illinois River Basin above Lake Frances from September 1978 to September 1981. The purposes of that study were to determine existing water quality conditions and to calibrate and verify an in-stream water quality model that would be used to simulate changes in water quality caused by changes in nutrient loadings. The study concluded that existing water quality in the Illinois River, and several major tributaries, did not meet the Arkansas State Guideline of 100 µg/l total phosphorus (as P) in

streams.

Roberts/Schornick and Associates (1984) reviewed studies of the Illinois River Basin for the Office of the Attorney General of Oklahoma in response to the City of Fayetteville's plan to upgrade their existing wastewater treatment plant and divert part of the effluent into a sub-tributary of the Illinois River. They concluded that the quality of water in the Illinois River apparently improved going downstream from Lake Frances, but indicated that the river was probably assimilating as much waste as possible and that increased loads of nutrients would generate increasing water quality problems.

Oklahoma's 305(b) Report (Oklahoma Department of Pollution Control, 1984) included an assessment of trends for certain water quality parameters at USGS gaging stations 07195500, 07196000, 07196500, and 07197000 for the period from 1975 to 1983 done by the Oklahoma Department of Pollution Control (ODPC). It was concluded there was an apparent increasing trend in concentrations of total phosphorus at all four stations. Nitrite + nitrate (NO_2+NO_3) trend tests showed no apparent trend at USGS stations 07195500 and 07197000. USGS 07196000 showed an apparent decreasing trend and 07196500 showed a possible decreasing trend. The ODPC used a U.S. EPA software package that applied Spearman's Rho and Sen test statistics to analyze the data. These are nonparametric tests for trend based on rank correlated with time. In the report, an ‘apparent’ trend was defined as being statistically significant at the 90% level, and a ‘possible’ trend was defined as being statistically significant at the 80 - 90% level.

Gakstatter and Katko (1986) performed an intensive study of the Illinois River Basin in both Arkansas and Oklahoma in August 1985. This study was done in response to concerns that water clarity had decreased in the reach of the Illinois River between Lake Frances and Lake Tenkiller, the designated scenic river portion. The survey included water sample collection and analysis of 24 mainstem and tributary sites throughout the basin. The study concluded that background phosphorus concentrations in the basin were generally very low. However, Osage Creek, which receives wastewater effluent from the cities of Rogers and Springdale, Arkansas, typically had much higher phosphorus concentration levels that substantially affected concentration levels in the Illinois River above Lake Frances and in Lake Frances. It was also concluded that the effects of water flowing through Lake Frances, sustaining substantial algal growth, adversely affected water clarity for some 32 kilometers below the Lake Frances dam.

Walker (1987a) prepared a report for the Office of the Attorney General of Oklahoma in response to the proposed discharge of a portion of Fayetteville's effluent into the Illinois River Basin. Reviewing data in the EPA's STORET data base and Gakstatter and Katko's data, Walker concluded that phosphorus concentrations have increased by a factor of roughly two to three over the past decade. The increased levels of stream phosphorus have been accompanied by substantial increases in chlorophyll *a* concentrations in both Lake Frances and Lake Tenkiller. Chlorophyll *a* is a pigment produced by algae and is an indicator of algal density. Walker used flow-weighted annual mean total phosphorus

concentrations to develop conclusions about trends. He suggested that it would only be proper to compare years of comparable flow to determine if total phosphorus concentrations had indeed increased. Walker also concluded the most probable cause for accelerated eutrophication in Lake Tenkiller is increased point source nutrient loadings. Generally, nonpoint sources tend to be rich in nitrogen while point sources tend to be rich in phosphorus (Walker, 1987a).

A study done by the U. S. Army Corps of Engineers (1988) on Lake Tenkiller in 1985 and 1986 showed relatively high concentrations of nutrients in the upper portion of the lake which gradually decreased going downstream toward the dam. Using a trophic state index proposed by Carlson (1977), which provides a numeric measure of trophic status using total phosphorus data, the lake was classified as eutrophic throughout the lake. Carlson's index can also use chlorophyll *a* and secchi disk data to determine trophic status. Using these data the lake was shown to be borderline eutrophic at the upper end decreasing to mesotrophic near the dam. It was concluded "immediate and intense" efforts by federal and state agencies, municipalities, industries, and private landowners would be required to control point and non-point sources of nutrients to protect Lake Tenkiller from further deterioration.

Burks and Kimball (1988) performed a study evaluating existing concentrations of nutrients transported by the Illinois River to make an assessment of the potential effects on water quality in Lake Tenkiller. They found the highest levels of nutrient concentrations (nitrogen and phosphorus)

just below Lake Frances with a steady decline downstream to Tahlequah where that city's WWTP effluent caused an apparent increase. A steady state computer model, QUAL2E, of the lower reaches of the Illinois River above Lake Tenkiller and the upper segment of Lake Tenkiller was developed. They found that a projected decrease in phosphorus concentration input from Tahlequah's WWTP, after construction and implementation of a phosphorus removal system, would be adequate in reducing the rate of eutrophication of Lake Tenkiller. However, they concluded that other point and non-point sources within the basin would still contribute to the further deterioration of water quality in Lake Tenkiller. They recommended concerted efforts by public and private agencies to reduce phosphorus input into Lake Tenkiller to prevent further deterioration.

Harton (1989) performed a modeling study of the Illinois River in an attempt to analyze contributions of point and nonpoint source phosphorus loading on Lake Tenkiller. Included in the objectives of the study was an attempt to determine the effects of the discharge of half of Fayetteville, Arkansas' treated effluent into a tributary of the Illinois River and the subsequent effects on eutrophication in Lake Tenkiller. The Fayetteville wastewater treatment plant effluent was determined to have no observable effect on eutrophication in Lake Tenkiller. Harton concluded that the substantial distance from the point of entry of the effluent into the Illinois River to Lake Tenkiller was sufficient to allow for nearly total removal due to sedimentation and biological activity. Non-point source total phosphorus loadings from Oklahoma and Arkansas were found to be the main loading sources to the lake. Harton concluded that removal of 70 to

90% of the total phosphorus loading from point and nonpoint sources would be necessary to bring eutrophication under control at Lake Tenkiller.

A study by Burks et al. (1991) attempted to evaluate factors affecting water quality in the Illinois River. It was found that in-stream total phosphorus concentration exceeded the 0.1 mg/l level recommended by the EPA (U. S. EPA, 1986) to prevent enrichment of streams or tributaries to standing bodies of water. They suggested that there was "overwhelming evidence" that phosphorus loading to the upper end of Lake Tenkiller was excessive, and predicted decreases in water quality for the lake. Total nitrogen loading also was shown to be increasing over time. They suggested strict reduction of both point and nonpoint nutrient inputs into the system, and suggested that the focus be placed on phosphorus.

The Phase I Diagnostic and Feasibility Study on Tenkiller Lake (OWRB, 1996) found that mean annual concentrations of phosphorus, nitrogen, and chlorophyll *a* measured throughout Lake Tenkiller were indicative of eutrophic conditions. The entire lake was found to be affected by eutrophication as evidenced by the presence of anoxic conditions in the hypolimnion during summer stratification. Recommendations for control of eutrophication were focused on the reduction of phosphorus from both point and nonpoint sources.

Evidence suggests that there have been upward trends in the levels of nitrogen and phosphorus at water quality monitoring stations within these scenic river segments. Gade (1990) presented temporal trend tests (Kendall Tau) on flow adjusted concentrations of total phosphorus at USGS gaging stations

07195500 (Illinois River at Watts, OK), 07196000 (Flint Creek near Kansas, OK), 07196500 (Illinois River near Tahlequah, OK), and 07197000 (Baron Fork Creek near Eldon, OK) all indicated highly significant upward trends for the period from 1976 to 1986. Flow adjusted concentrations of nitrogen, measured as NO_2+NO_3, showed highly significant increasing trends at 07195500 near Watts and at 07194800 (Illinois River near Savoy, AR, not a scenic river segment) using the Kendall Tau test over the same time period.

Phosphorus

From a biological standpoint, phosphorus is necessary in small quantities for the proper nutrition of plants, animals, and man. The reproductive processes of life depend on the phosphorus-containing genetic code carriers DNA and RNA. Life cannot proceed without the energy exchanges mediated by vital phosphates such as ATP (Toy and Walsh, 1987). No other element in fresh waters has been studied as intensively as phosphorus. Ecological interest in phosphorus stems from its major role in biological metabolism, and the relatively small amounts of it available in the hydrosphere. It is the least abundant of the major structural and nutritional components of biota (carbon, hydrogen, oxygen, nitrogen, sulfur, and phosphorus) and most often limits biological productivity in aquatic environments (Wetzel, 1983).

In aquatic ecosystems, as in all ecosystems, specific amounts of nutrients and trace elements must be available to living organisms in order for them to survive and proliferate. In many freshwater aquatic ecosystems, phosphorus as

phosphate is often referred to as a "limiting nutrient" implying that it is normally not abundantly available in forms usable by organisms. If the amount of phosphate available to organisms is very small, then the growth of organisms is limited by that low availability. If the amount of phosphate is abundant or in excess of the amounts required for life maintenance of organisms, then the organisms will tend to proliferate until another of the essential nutrients or elements becomes limiting, or until the organisms begin to alter their environment in such a way that is detrimental to themselves or other life forms.

Concerning freshwater ecosystems, excessive input of phosphate tends to accelerate the natural aging process of lakes and streams (eutrophication). Eutrophication, as a natural geological process, involves a body of water where organic life develops and multiplies through time. Organisms flourish and die in normal life cycles and, with time, the bottom surface of the water body accumulates the remnants of organic life and inorganic sediments, and slowly builds up. As the water body becomes shallower, the aquatic organisms change character. Eventually the water body becomes so shallow that it becomes a marsh, bog, or swamp. Finally, the water body no longer exists as such, and becomes dry land. This process normally takes thousands of years for a relatively large body of water (Toy and Walsh, 1987).

Our present concern is that with the inflow of excessive quantities of nutrients into water bodies, caused by the activities of man, eutrophication is rapidly accelerated. An example is Lake Erie's shallow western basin. It has been estimated that in the past 50 years this area of the lake has aged an

equivalent of 15,000 years (Toy and Walsh, 1987).

Phosphorus is the eleventh most abundant element in igneous materials in the earth's crust. Average concentration of phosphorus in this environment is estimated as about 1% by weight and thus it is classified as a trace element. Only a fraction of this amount is concentrated in deposits consisting mainly of phosphate minerals (Holtan et al., 1988). Pure phosphorus is a white, transparent, crystalline, waxy solid that glows in the dark and ignites in air just above room temperature (30 ^{o}C) and melts at 44.2 ^{o}C (Toy and Walsh, 1987).

Phosphorus in the environment occurs almost exclusive as phosphate in the +5 oxidation state. Phosphine (PH_3) appears to be absent from biochemical and geochemical systems. In its general chemistry, phosphorus has a strong affinity for oxygen (Corbridge, 1985). Orthophosphate (H_3PO_4), the form of phosphorus found in natural environments, has a melting point of 42.35 ^{o}C, a boiling point of 213 ^{o}C, solubility of 1.0×10^{6} at 25 ^{o}C, specific gravity of 1.834, and vapor pressure of 0.0285 mm Hg at 20 ^{o}C. It is a strong acid and therefore in adequate concentration is corrosive (U.S. EPA, 1984). The acid dissociates in three distinct stages in water with dissociation constants of $k_1 = 7.1 \times 10^{-3}$, $k_2 = 7.99 \times 10^{-8}$, and $k_3 = 4.8 \times 10^{-13}$. It is soluble in water and alcohol, and has a molecular weight of 98.00 (Budavari, 1989).

The most significant form of inorganic phosphorus in natural waters is orthophosphate (PO_4^{-3}). A large proportion of phosphate in freshwater occurs as organic phosphate and cellular constituents in biota adsorbed to inorganic and dead particulate organic matter. Total inorganic and organic phosphorus are

frequently separated by chemical analyses but these fractions often poorly relate to the metabolism of phosphorus. Commonly, the most important measure is total phosphorus of unfiltered water which consists of phosphate in the particulate and dissolved phases. Particulate phosphate is phosphate in organisms such as nucleic acids and phosphoproteins (not involved in the rapid cycling of phosphate), low molecular weight esters of enzymes, vitamins, and nucleotide phosphates, mineral phases of rocks and soil in which phosphate is absorbed onto inorganic complexes such as clays, carbonates, and ferric hydroxides, and phosphate absorbed onto dead particulate matter. Dissolved phosphate is defined as orthophosphate, polyphosphate (as in synthetic detergents), organic colloids or phosphate combined with absorptive colloids, and low molecular weight phosphate esters (Brock, 1985).

Much emphasis on analytic evaluation results in the recognition of eight different forms of phosphorus with four operational categories. The four operational categories are soluble reactive phosphorus, soluble unreactive phosphorus, particulate reactive phosphorus, and particulate unreactive phosphorus. These operational categories do not necessarily correspond with chemical species of phosphorus or to their role in the biotic cycling of phosphorus.

Separation of total phosphorus into inorganic and organic fractions in a large number of lakes indicates that a large majority of the total phosphorus is in an organic phase (usually around 90%). About 70% of the organic phosphorus is found within particulate organic material and the remainder found as dissolved

or colloidal organic phosphorus. Inorganic phosphorus is consistently present in very low concentrations and is cycled very rapidly in the zone of utilization. The ratio of inorganic phosphorus to other forms is approximately 1:20 or about 5%. The percentage as orthophosphate is then less than 5% (Wetzel, 1983).

Phosphate is present in lakes at only one oxidation state (PO_3^-), so that complications regarding oxidation and reduction do not arise. Soluble phosphorus is that fraction assayable by orthophosphate assay after filtration of a water sample through a membrane filter (pore size generally 0.45 µm). Total phosphorus is that assayable after digestion of an unfiltered sample with a strong oxidizing agent (perchlorate or persulfate) followed by an orthophosphate assay. Insoluble phosphorus is the difference between total phosphorus and soluble phosphorus. The soluble form is almost certainly available to phytoplankton, whereas the particulate form may or may not be available (Brock, 1985).

Phosphate deposits may be classified into three broad apatite [$(Ca_5(PO_4,CO_3)_3(F,Cl,OH)$] groups which are apatite deposits of igneous and metamorphic origin, sedimentary phosphorites, and guano and related deposits. More than 200 natural phosphates have been described. Most (95%) occur as fluorapatite (Holtan et al., 1988).

Calcium phosphate from animal bones was used as a source of inorganic phosphorus in the past. Present sources of calcium phosphate are minerals commonly known as phosphorite or phosphate rocks mined in enormous open pits. The phosphate rocks have an approximate composition of the mineral

fluorapatite $Ca_5(PO_4)_3F$, a calcium phosphate containing 4% fluorine. Largest American deposits of phosphate rocks lie in Florida, Tennessee, North and South Carolina, Kentucky, Virginia, Utah, Idaho, and Montana. Florida is largest producer of phosphate rock which is a five billion dollar a year business. Large deposits are also found in Kola Peninsula in Scandinavia, Algeria, Tunisia, Egypt, and Morocco (Holtan et al., 1988). In 1980, approximately 37 million tons of phosphate rock was consumed in the U.S.. About 7% of this was used to produce elemental phosphorus and the remainder was converted to phosphoric acid and fertilizer products. In 1984 the U.S. produced more than 750 million pounds of the element (Toy and Walsh, 1987).

Origin of these rocks is believed to be the small quantities of phosphate normally present in granitic rocks. Over eons of time, through weathering and leaching, phosphates found their way into the sea where marine animals absorbed and concentrated them into their shells, bones, and tissues. The remains of these animals accumulated on the bottom of the sea and over the years, through geological changes, some phosphatic sediments appeared as deposits on dry land (Altschuler, 1973).

The release and export of phosphate from uncultivated soil is a function of the geology and soil composition, air temperature, precipitation, and hydrological conditions. Generally, waters in contact with unfertilized soils give a range of phosphorus concentrations less than 0.01 mg/l. Areas dominated by sedimentary rock systems tend to contribute more phosphorus to runoff than areas dominated by igneous rock formations (Holtan et al., 1988).

A major fraction of the total phosphate carried out of a watershed by streams is associated with the particulate matter suspended in the water, much of which is delivered during a relatively short period of time when discharge rate is high and the velocity of the flowing waters is sufficient to transport particles of varying sizes and densities. A smaller fraction is dissolved in the water.

An adult human normally excretes 1.3 - 1.5 grams of phosphorus per day and disposes of an additional quantity through the use of synthetic detergents. Phosphate detergents have become one target in the attempt to reduce phosphorus release into the environment. Phosphorus removal from sewage by municipal waste water treatment plants has become a standard process in some regions (Holtan et al., 1988).

Another major source of phosphorus is agricultural activity, either derived from crop production (fertilizers) or from animal wastes. Application of fertilizers to agricultural land to increase productivity has resulted in the off-site transport of some of the nutrients through leaching and runoff. The recent trend to produce livestock in larger farm units has, in some areas, led to high concentrations of livestock wastes washing into streams and lakes (Holtan et al., 1988).

After use by man, a significant fraction of phosphorus is transported to watercourses in particulate and dissolved forms by means of agricultural runoff or as point sources from municipal waste water treatment plants. Ground water discharge and atmospheric deposition also contribute to phosphate loading of streams and lakes (Holtan et al., 1988). There is great regional variation in phosphate concentration in precipitation. This can be a major source in some

lakes. Phosphorus sources which accelerate eutrophication are often anthropogenic and include point sources of treated wastewater effluent and nonpoint sources associated with agricultural activities.

The movement of phosphate through the inorganic and organic forms can be resolved into three main cycles. This is, in essence, two biological cycles superimposed on an inorganic cycle. The inorganic cycle turns very slowly with its rate of revolution measured in 10^9 years. The two biocycles turn rapidly in comparison. The land-based biocycle may have an annual turnover of some of the phosphate, although there are reservoirs of phosphate in the soil that can store it for centuries. In the water-based biocycle phosphate turnover is often measured in days and months.

The primary inorganic phosphate cycle is probably better described in terms of a spiral rather than a cycle. Phosphate originally comes from igneous rocks and, via weathering and biological processes, passes through the soil to the rivers and on to the sea, where it eventually ends up in sediment. If the sediment is part of the deep ocean there is little chance of that phosphate going through the cycle again. However, some ocean phosphate does return to land via ocean currents, fish, and the birds that feed on them. If the sediment is part of an inland sea or continental shelf there is a good chance it will once again be part of dry land as a result of geological uplift. The overall natural movements tend to concentrate phosphates in large deposits in a few areas of the globe, chiefly along the western coasts of the continents. The spiral would tend to end there except for the intervention of man who can release this locked-up

phosphate to the inorganic cycle again (Emsley, 1980).

The atmosphere does not serve as a significant reservoir of phosphate, but its contribution to the phosphates found in water bodies through wet and dry deposition is significant. The development of phosphorus budgets for water bodies are necessarily required to account for atmospheric deposition of phosphorus on lake and stream surfaces. The importance of atmospheric sources of phosphorus increases when the land area/lake area ratio increases. In some very dilute lakes of Western Ontario, precipitation may account for as much as 80% of the phosphorus input (Ahl, 1988).

Phosphorus exists in the atmosphere in dust particles and microbial debris. That means that all phosphorus collected as wet or dry deposition has earth as its origin (Ahl, 1988). Atmospheric dust originates from three major sources which are terrestrial airborne matter, volcanic, and cosmic. Of these, the terrestrial dust makes up the main bulk. Deserts and other dry regions act as major sources of atmospheric phosphorus and the soil composition of a wind eroded area can be reflected 200 - 250 km away (Ahl, 1988). Anthropogenic contamination of the atmosphere includes phosphorus coming from fertilized soils.

Differentiation of terrestrial and marine atmospheric phosphorus contributions can be made on the basis of the relative quantities of reference elements (Si, Na and Cl for marine, and Al and Fe for terrestrial) in the atmospheric dust particles. Generally, air masses with oceanic origin have much lower content of phosphorus than air masses with continental origin. Ahl (1988)

described a study of coastal aerosols in a low industrial area in Brittany, France at the Atlantic coast and found that air masses of continental origin contained phosphorus of crust origin up to 40%. The anthropogenic phosphorus content, independent of origin, varied from about 60 to 90% of the total amount.

Most of the terrestrial dust carries phosphorus in the inorganic form. A U.S. average estimate of background atmospheric deposition is in the range of 10-46 kg phosphorus $km^{-2}yr^{-1}$. Lowest deposition figures indicate that lowest phosphorus deposition in the Northern Hemisphere is found in remote areas (Ahl, 1988).

Soil formation (pedogenesis) is an ongoing process that is partly mechanical (weathering) and partly a consequence of supporting living things. Pedogenesis slowly releases inorganic phosphate from apatite fractions of the original rocks to the water in the soil, where it becomes available for plant nutrition. This source of phosphate is released too slowly to represent a significant contribution to the phosphate required for a particular growing season. The water in the soil generally has a concentration of 10^{-5} to 10^{-6} M phosphate but may be as low as 10^{-8} M in some tropical soils (Emsley, 1980). It has been observed that the concentration of plant available phosphorus in the top soil remains almost constant when crops are being grown which indicates that there are insoluble sources of phosphate in the soil which supply the soil water with phosphate to maintain an equilibrium concentration.

In addition to the supply of phosphate from the original rocks and rainwater, it is possible to identify three pools of phosphate that can supply it to

the soil water. These can be described as either non-equilibrium pools, such as the original apatite and the organic phosphate, from which the phosphate is slowly released, or equilibrium pools such as the precipitated and adsorbed phosphate pools. These labile pools can release phosphate to soil water or remove phosphate from soil water as conditions permit (Emsley, 1980).

Phosphate has a special relationship with three metals in soils (calcium, iron and aluminum). Calcium phosphate is essentially insoluble but does not remove phosphate permanently and remains in equilibrium with a certain concentration of phosphate in the soil solution. Aluminum hydroxide, on the soil particle surface, is the principle holding agent for inorganic phosphate and acts as a ligand to it in the $H_2PO_4^-$ form. It holds the HPO_4^{-2} ion less securely and for this reason an increase in soil pH will release phosphate bound to aluminum (Altschuler, 1973). Thus, liming soil has the effect of releasing more aluminum bound phosphate than it precipitates as calcium phosphate. Phosphate attached to aluminum is labile and crystals of definite composition may form (Emsley, 1980).

Phosphate attached to iron is immobile and is described as occluded. Ferric hydroxide sites on the soil surface can hold phosphate permanently, and the gradual buildup of occluded phosphate is due to its migration to and retention to these sites. Phosphate attached to iron is not labile nor does it crystallize (Emsley, 1980). During periods of lake stratification, where the hypolimnion can become anoxic, decreased redox potentials can result in the release of phosphate and iron from lake sediments into solution (Wetzel, 1983).

Biological interactions with phosphates in the soil and soil solution can be used to describe the land-based biocycle. Roots are good at scavenging phosphate from the soil water and then it begins through a biocycle. The ratio of plants to animals is the determining factor in the rate of phosphate flow through the system. In systems with little animal activity, the uptake of inorganic phosphate from the soil may be as little as half that from systems supporting dense animal activity. Roots can transfer phosphate against a concentration gradient from soil water to plant. By doing so they create in turn a concentration gradient for the precipitated and adsorbed phosphate in the soil, causing them to release phosphate to the soil solution.

The role of soil fauna and microbes is a key part of the phosphate cycle. These decomposers require phosphate for their own metabolic needs and studies have shown that they take up four to five times more soil phosphate than do plants living in the same soil. They also add organic phosphates to the soil. It has been estimated that 75% of organic phosphate in the soil comes from microbial sources (Emsley, 1980).

In general, the loss of phosphate from soils via leaching is relatively small since it is tightly bound to particulate matter. The amount lost by leaching is probably compensated for by phosphate arriving in rainwater.

Phosphate cycling in water bodies via biological means can be described as the water-based biocycle. The routes for conveying phosphate from the land cycle to the water-based cycles are for the most part rivers and streams. It has been estimated that rivers carry 17 million metric tons of phosphorus to the seas

each year, half due to natural processes and half due to the activities of man (Emsley, 1980).

Phosphate moves rapidly through the water-based system. The two factors which determine the extent of biomass in most lakes and seas are the availability of phosphate and the amount of solar energy. This cycle again represents a downward spiral as the processes of precipitation and sedimentation are responsible for the rapid loss of phosphate from the aqueous phase to the solid phase at the bottom.

Phosphate uptake in the aquatic system into biomass is very rapid. Algae, like land plants, can take in more phosphate than they actually need and can store enough surplus to support them through about three cell divisions without additional input (Emsley, 1980). Algae are able to directly utilize the inorganic phosphate and given the combination of a plentiful supply and enough sunlight they can multiply to such an extent as to color the water green or red in lakes. Often two blooms of algae appear each year. In Spring the sunlight and mixing waters rich in phosphate will often promote a bloom. In Fall, phosphate mixed upward from the deeper layers of the lake becomes available to the algae once again and a second bloom may appear. Phosphate in algae can be taken up and released rapidly. Within hours of death, the cellular phosphate is released into the water again as either inorganic phosphate or soluble organic phosphate which can be readily taken up by living algal cells (Emsley, 1980).

It has been observed that relatively low concentrations of phosphate in waters could support algal blooms, which seems at odds with the idea of

phosphate as a limiting nutrient. But, it is not the concentration of inorganic phosphate which is most important, but rather the total phosphorus concentration, since rapid recycling occurs.

The lake sediment is often the final resting place for phosphate. It may remain here for millions of years. Phosphate in the top layer of sediment can, however, return to the waters of a lake by various means. Decomposer organisms, enzymes, worms and other benthic organisms may assist this process. Certain forms of organic phosphate can form stable complexes with metals and this phosphate is not released.

When the waters of a lake become stratified the lowest layer may become anoxic due to oxygen depletion as a result of the degradation of organic debris settling down from the upper layers. The hypolimnion may also become slightly acidic. Both of these conditions, a reducing and an acidic solution, will facilitate the return of phosphate from the upper layer of the sediment back into solution. Phosphate held in the sediment as insoluble $FePO_4$ will be reduced to soluble $Fe_3(PO_4)_2$ and insoluble $CaHPO_4$ will be acidified to soluble $Ca(H_2PO_4)_2$. Under these conditions the release of phosphate from the sediment can sometimes exceed the downward flow of phosphate to it. The top layer of sediment in a eutrophic lake may hold enough phosphate to keep the lake in a eutrophic state despite effective steps taken to curb the inflow of phosphate into the lake (Emsley, 1980).

Sediments play an important role in the overall phosphorus metabolism of lakes acting as both a source and a sink. In most lakes there is a net deposit of

phosphorus in the sediments. However, in lakes of higher trophic status, phosphorus release may exceed phosphorus sedimentation for periods ranging from days to a year. Oxygenated sediments tend to retain phosphorus by fixation to iron (III) while reduced sediments release phosphorus by reduction of iron and subsequent dissolution of iron-phosphorus complexes (Bostrom et al., 1988). Lake sediments generally contain much higher concentrations of phosphorus than the overlying water. Under aerobic conditions the exchange equilibria is directed largely toward the sediment. Under anaerobic conditions, inorganic exchange at the sediment-water interface is strongly influenced by redox conditions.

Phosphorus mobilizing bacteria of the genera *Pseudomonas*, *Bacterium*, and *Chromobacterium* are important in mobilizing phosphorus into the interstitial waters of the sediments, but bacterial metabolism at the interface has relatively little effect on the on biogenic fixation and removal of phosphorus from overlying waters. Under aerobic conditions bacteria at the interface increase transport of phosphorus to the sediments due to sedimentation (Wetzel, 1983).

In very productive lakes, with high rates of sedimentation of organic matter to the sediments, the decomposition of the sedimenting organic matter produces anoxic conditions, hydrogen sulfide, and ferrous sulfide (FeS) is precipitated. FeS is exceedingly insoluble and forms at a redox potential of about +100 mv. If large quantities of FeS precipitate, sufficient iron can be removed to permit some of the phosphate accumulated in the hypolimnion to remain in solution which can then become available to algal organisms when the

water body mixes (Wetzel, 1983).

Only part of the total phosphorus load to lakes will be available to the lake biota for production of organic matter. Basically, bioavailable phosphate can be defined as the sum of immediately available phosphate, and, phosphate that can be transformed into an available form by naturally occurring physical (desorption), chemical (dissolution), and biological processes (enzymatic degradation) (Bostrom et al., 1988). Orthophosphate seems to be the only directly available phosphorus source for planktonic algae and bacteria although the direct uptake of some organic phosphates cannot be excluded.

Schaffner and Oglesby (1978) defined biologically available phosphate as soluble reactive phosphate (SRP), soluble unreactive phosphate (SUP), and labile phosphate (associated with soil particles). SRP is considered to be entirely biologically available, SUP is considered to be available by enzymatic hydrolysis, and labile phosphate may be available as it dissolves in an aqueous solution.

Bacteria and algae are the only two components of aquatic ecosystems which depend, for their nutrition, on the transport of dissolved substances from the water, through the cell wall/membrane barrier, to the interior of the cell. Utilization of complex organic phosphates has to be preceded by external liberation of orthophosphate by means of enzymes such as phosphatases.

Studies of *E. coli* show that bacterial uptake of phosphate is maintained by two kinetically distinct systems called high affinity and low affinity components. The high affinity system is activated when internal phosphate is

depleted. The low affinity system supplies phosphate at a rate equal to phosphate metabolism within the cell. The high affinity system allows the bacteria to accumulate phosphate against a concentration gradient at a rate faster than the phosphate utilized in the cell (Schaffner and Oglesby, 1978). A membrane localized protein binds one molecule of phosphate per protein molecule. During phosphate starvation the concentration of this protein is capable of increasing by at least one hundred times. The energy source for these systems has been proposed to be ATP, although a close connection between potassium and phosphate transport has been reported with phosphate transport systems appearing to be dependent on the presence of potassium with phosphate and potassium being taken up simultaneously.

Algae also depend on active transport systems for the accumulation of internal phosphate against a concentration gradient. The physiological details of the algal phosphate uptake system are less well known than those of bacteria. It is commonly observed that phosphate-starved algae take up phosphate faster than non-starved algae and in fact rates of uptake may increase 10-100 times. It appears that this response is controlled by internal inorganic phosphate, particularly polyphosphates. The polyphosphate pool is filled during periods of high phosphate supply and can sustain growth for several generations in the absence of external phosphate sources. Light induced energy as ATP from photophosphorylation and oxidative phosphorylation supplies the energy for the uptake (Jansson, 1988).

The sources of phosphate that can be used by algae are varied. The

primary source is inorganic phosphate but other sources can be of importance. It has been shown that polyphosphate with chain lengths of up to 53 units will support growths of *Chorella*. The algae have an adaptive enzyme which hydrolyzes pyrophosphate bonds releasing phosphate which can be used directly. *Chorella* has also been shown to be capable of deacylating phospholipids on their cell surface and absorbing the released phosphate (O'Kelley, 1973).

The level of phosphate required for optimal growth differs among species of algae. Investigations of minimal and maximal phosphorus concentration, by Chu (1943) and Rodhe (1948) in Wetzel (1983), grouped freshwater algae into categories according to their tolerance ranges at or around 20 µg phosphate as P per liter. Species whose optimum growth and upper tolerance limit is below 20 µg PO_4-P l^{-1} included *Uroglena* and some of the species of *Chara*. Species whose optimum growth was below 20 µg PO_4-P l^{-1} but had an upper limit above that concentration included *Asterionella* and other diatoms. Species whose optimal growth and upper tolerance limit is above 20 µg PO_4-P l^{-1} included green algae such as *Scenedesmus*, *Ankistrodesmus*, and many others. In these studies the phosphorus concentrations of the culture media required for optimal growth were nearly always higher than those in natural habitats where the algae were growing. The best explanation for this difference is that the chemical mass of inorganic phosphate in the water has only a tenuous relationship to growth kinetics. The factor of greatest importance is the rapidity with which phosphate is cycled and exchanged between the particulate phosphate and soluble

inorganic and organic phases (Wetzel, 1983).

The proper functioning of living cells depends on the action of many phosphorus containing compounds. The most important of these are DNA and RNA. These compounds are located in the cells of all living organisms and carry the genetic code. Another important biochemical is ATP, the universal energy carrier in living things. ATP serves as a major link between energy yielding and energy requiring chemical reactions. When one phosphate group in ATP is hydrolyzed at neutral pH at 25 ^{o}C to ADP and inorganic orthophosphate, 7,300 cal/mol of energy is liberated (Wetzel, 1983). This free energy of hydrolysis is used to perform biological work. ATP is not the only phosphorus energy carrier in the biological system. Some compounds have a higher free energy of hydrolysis than ATP. These compounds are used for a vast number of biochemical reactions.

The fundamental event in photosynthesis by green plants is the absorption of light by chlorophyll, which raises some of the electrons in the molecule to higher energy levels. The excited electrons are then transferred to electron-carrying enzymes which use the energy in a variety of ways, including the production of ATP (through photophosphorylation) and NADPH. These two energy carriers provide the energy necessary to convert carbon dioxide to glucose in a complex cycle of enzyme-catalyzed reactions. Biophosphates are important as energy carriers and as products of many of these reactions (Corbridge, 1985).

Phosphates are excreted as orthophosphate and some organic

phosphates in soluble form. Laboratory cultures of various species of algae have shown that more than 20% of the total phosphorus in a system may be excreted in organic soluble forms. This dissolved phosphorus can then be readily assimilated by either the species that produced it or another species (O'Kelley, 1973).

A great amount of attention has, and continues to be, directed to the role of phosphorus in eutrophication and the development of algal blooms. It has become widely accepted that the growth of phytoplankton in many bodies of freshwater tends to be limited by the supply of available phosphate. Thus, the addition of quantities of available phosphate to a system, with adequate quantities of all other essential elements and nutrients present, allows the phytoplankton to multiply.

As discussed above, phosphates play a role in a vast number of biochemical reactions. Phytoplankton, as well as all other living organisms, have a requirement for phosphate which, if met, allows normal life processes to proceed. It seems clear that a relationship exists between phytoplankton biomass and phosphorus. A number of studies have shown relationships between phosphorus-loading, summer phosphorus concentration, spring phosphorus concentration, etc., and chlorophyll. When phosphorus concentrations increase the algal biomass also tends to increase (Wetzel, 1983).

Phosphorus limited phytoplankton can respond to changes in phosphorus concentrations by a change in biomass and/or a change in rate of production. If the phosphorus is used to produce biomass that is quickly eliminated, by high

grazing pressure for example, the relationship between phosphorus and biomass would weaken unless grazing pressure is also a function of phosphorus. Phytoplankton in different lakes exhibit different strategies for the allocation of phosphorus for either a fast turnover or a high biomass with little turnover. The available nutrients other than phosphorus, as well as environmental conditions such as light, pH, and temperature, may also change the dynamics of phytoplankton in a way that alters its response to phosphorus (Heyman and Lundgren, 1988).

One of the best indicators of advanced eutrophication is the presence of large blooms of algae. Under proper conditions of adequate nutrients, ideal temperature, and correct pH, the rapid growth of microscopic phytoplankton can easily result in highly turbid water. Depending on the species of algae present, the water may turn "pea-green" or red. These algal cells eventually die and decomposition of this organic matter begins. Significant quantities of the organic matter settle to the lake bottom where decomposition continues. This bacterial decay consumes large quantities of dissolved oxygen, sometimes to such an extent that the normal diffusion of oxygen to water is not enough to replenish the supply. Deeper lakes, in regions where conditions exist such that a lake stratifies, eventually have nearly all dissolved oxygen removed from the hypolimnion where it is not readily replenished due to the stability of the stratification and the lack of diffusion from oxygenated layers. The resulting oxygen depletion in the lake effectively excludes oxygen requiring organisms from areas of the lake (Toy and Walsh, 1987).

Numerous definitions of lake eutrophication exist based on a variety of applicable conditions associated with increased productivity. Among limnologists, eutrophication is synonymous with increased growth of biota of lakes, and the rate of increasing productivity is accelerated over that rate which would have occurred in the absence of perturbations to the system. The most conspicuous measurable criterion of accelerated productivity is an increased quantity of carbon assimilated by algae and larger plant biomass per unit area.

Under most lake conditions, the most important nutrient factors causing a shift from a lesser to a more productive system are nitrogen and phosphorus. Typical plant organic matter of aquatic algae and macrophytes contains phosphorus, nitrogen, and carbon in approximately the ratios 1:7:40 per 100 dry weight (Wetzel, 1983). If one of these three elements is limiting and the others are present in excess of physical needs, phosphorus can theoretically generate 500 times its weight in living algae (nitrogen - 71 times, and carbon - 12 times). A comparison of the relative amounts of elements required for algal growth with supplies available in fresh water illustrates the general importance of phosphorus and nitrogen. Average plant tissue requirements for carbon, nitrogen, and phosphorus are 6.5, 0.7 and 0.08% respectively. The average supply of these elements in fresh waters is 0.0012, 0.000023, and 0.000001% respectively. The ratios of plant content to supply available then is 5000, 30,000, and 80,000, respectively (Wetzel, 1983). Variations in conditions of solubility or availability may at times make very abundant elements nearly unobtainable, but generally phosphorus and nitrogen are the first to impose limitations on the system

(Wetzel, 1983).

When phosphorus is added as a pulse to an unproductive system, the usual response is a very rapid increase in algal productivity. The increased productivity is not sustained but decreases rather rapidly to levels prior to the addition. Losses to the colloidal fraction and from sedimentation result in continuous losses from the trophogenic zone. Thus, sustained productivity requires the constant input of nutrients. In order to reduce productivity, algal growth is usually most effectively decreased by the reduction of phosphorus inputs into the system (Wetzel, 1983).

The science of ecology teaches us that all living and non-living things in an ecosystem are interrelated and interdependent. Thus, when a system such as a lake rapidly changes, we would expect to find changes in the distribution and presence of species in that community. In response to eutrophication there are documented changes in the distribution and numbers of individuals and species living in the system.

The response of lakes, reservoirs, and slow moving rivers to the over enrichment by nutrients is an increase in fertility, the consequences of which (reduced transparency, increased organic substances, depletion of dissolved oxygen, possible fish kills) adversely affect the different uses of water such as water supply, fisheries and recreation, and impair aesthetic qualities (Vighi and Chiaudani, 1987). Fish species change from salmonid and coregonid species of stringent low thermal and high oxygen requirements to warm water species that are increasingly tolerant of eutrophic conditions.

Benthic organisms are also affected. If lakes become extremely enriched to the point that the population densities of the phytoplankton and epiphytes become so great that they shade out the submersed macrovegetation, and then the habitat diversity of the littoral zone diminishes. With increases in eutrophication, and lengthening of the period of hypolimnetic oxygen reduction and associated chemical changes, the rates of respiratory activity of the adapted benthic animals are reduced. Rates of growth and survival also change. Under severe conditions essentially all aquatic insects are eliminated. Oligochaete annelids may be the only group of benthic fauna able to survive. More eutrophy shifts the percentage composition of the two dominant groups of benthic animals in the profundal zone of lakes, the Chironomidae and the oligochaetes. The former are dominant in oligotrophic conditions and the latter in eutrophic conditions (Wetzel, 1983).

In addition to increased populations of all species of phytoplankton with the influx of limiting nutrients, prolonged eutrophic conditions can lead to a change in the types and quantities of species present. Oligotrophic water bodies tend to be dominated by *Chrysophyceae, Cryptophyceae, Dinophyceae,* and *Bacillariophyceae*. Eutrophic water bodies tend to be dominated by *Bacillariophyceae, Cyanophyceae, Chlorophyceae, and Euglenophyceae* (Wetzel, 1983).

A national water quality standard for total phosphorus in freshwater has not been established. While a relatively firm relationship between high phosphorus concentrations and accelerated eutrophication has been established

for many lakes, reservoirs, and streams, certain conditions exist in some areas which complicate the establishment of firm criteria. Some water bodies with phosphorus concentrations higher than suggested limits show no signs of eutrophy while others with concentrations of phosphorus lower than suggested limits exhibit eutrophic qualities.

Naturally occurring phenomena may limit plants as nuisances. Turbidity due to suspended inorganic particles may reduce light penetration and thereby decrease the euphotic zone of optimal light for algal and aquatic macrophyte growth. Water body morphology (steep banks, great depth, and high discharge rates) may be effective in reducing nuisance plant growth. Some water bodies are managed for waterfowl or other wildlife and thus more productive communities are desirable. Some studies have shown nutrients other than phosphorus as the limiting nutrient.

EPA *Quality Criteria for Water 1986* (U. S. EPA, 1986), offers a discussion of phosphate phosphorus and discusses some of these problems. Suggested goals for total phosphorus concentrations in freshwater are recommended based on a great volume of work relating phosphorus to eutrophication of lakes, streams, and other water bodies. For streams, where they enter a lake or reservoir, 50 µg/l total phosphorus is suggested as the limit (U. S. EPA, 1986). Within lakes or reservoirs, 25 µg/l is the recommended limit (U. S. EPA, 1986). The goal for prevention of nuisance plant growth in streams an other flowing waters not directly discharging into lakes or impoundments is 100 µg/l total phosphorus (U. S. EPA, 1986). Part of the above goals were developed based

on the fact that most uncontaminated lakes have surface water concentrations of total phosphorus in the range of 10 - 30 µg/l (U. S. EPA, 1986).

Vollenweider (1976, in Wetzel, 1983) suggests total phosphorus loading in terms of $g/m^2/yr$ critical for eutrophic conditions within the receiving waterway based on water volume and mean depth/hydraulic detention time. Table II summarizes phosphorus loads determined to be "permissible", allowing a water body to remain oligotrophic, and "critical", loads at or above which a water body may begin to show signs of accelerated eutrophication.

TABLE II

VOLLENWEIDER'S SUGGESTED PHOSPHORUS LOADINGS FOR LAKES

(Mean Depth)/ (Hydraulic Detention Time) (m/yr)	Oligotrophic or Permissible Loading ($g/m^2/yr$)	Eutrophic or Critical Loading ($g/m^2/yr$)
0.5	0.07	0.14
1.0	0.10	0.20
2.5	0.16	0.32
5.0	0.22	0.45
7.5	0.27	0.55
10	0.32	0.63
25	0.50	1.00
50	0.71	1.41
75	0.87	1.73
100	1.00	2.00

Source: from Wetzel (1983)

Since phosphate, which is normally available in limited supplies, acts as a

fertilizer for aquatic algae and plants, methods for reducing harmful blooms of algae have been attempted. In general, algae are best controlled by limiting the inflow of phosphate for the following reasons (Thomas, 1973):

1. Phosphate is present in only trace amounts in oligotrophic lakes.
2. Natural tributaries running into these lakes contain very little phosphate unless they are subjected to pollution by the influence of man.
3. Fewer phosphate than nitrogenous compounds are washed out of agricultural lands.
4. Rainwater often contains large quantities of nitrogenous compounds that can be utilized by plants.
5. Bacteria and blue-green algae living in lake water are able to fix nitrogen or to produce the growth factors for algae.
6. The addition of phosphate only to lake water is sufficient to increase the growth of bacteria and blue-green algae.
7. Some blue-green algae produce toxins that are very toxic to warm-blooded animals.
8. Nitrogenous compounds from putrefied parts of organisms and sludge return to the biochemical cycle in larger quantities than phosphate compounds.
9. In eutrophic lakes, nitrates are eliminated from time to time by the process of denitrification.

10. It is relatively less expensive and easier to eliminate phosphates from sewage water (by alum or $FeCl_3$) in the activated sludge process than removing nitrogenous compounds.

Precise levels of phosphate in the environment which are "safe" for humans related to the accelerated eutrophication of lakes have not been determined. A variety of different responses by lakes and streams in different regions to levels of phosphate input compound the difficulty of determining such a number. A number of eminent researchers have developed measures of lake trophic status based in part on the input of phosphates.

Nitrogen

Problems associated with excessive nitrogen in surface waters are primarily associated with eutrophication while high levels of nitrate in drinking water may lead to human health or animal health problems. Control of nonpoint sources of nitrogen is difficult due to the dynamic nature of nitrogen transformations and transport, and the many natural and anthropogenic sources of this element. Sustained high crop production usually requires that nitrogen fertilizers be added to the soil on a short-term basis due to the many avenues of loss and the low crop recovery of nitrogen.

Nitrate poisoning in humans is rare since acute poisoning in an adult requires a single oral ingestion of one to two grams of nitrate as nitrogen, far above the normal exposure episodes. Rather the health effects of nitrate are usually due to nitrite which is formed by reduction of nitrate by intestinal flora in

some animals and the human infant during the first three to four months of life. Nitrite is rapidly absorbed into the blood from the stomach. The toxic effects are due to the fact that nitrite acts as an oxidant, converting the Fe(2) of hemoglobin to Fe(3). The oxidized hemoglobin (methemoglobin) cannot transport oxygen. The high susceptibility of newborns and infants relative to older children and adults is due to the fact that: 1) nitrate reduction is rapid because of the higher gastric pH of infants (favors bacterial growth); 2) fetal hemoglobin is more readily oxidized; and, 3) the activity of enzymes that restore methemoglobin to hemoglobin is lower in infants. Cyanosis is observed when about 5% of the hemoglobin is converted to methemoglobin. The observed toxic effects of nitrate in livestock are qualitatively similar to those in humans, but the dose required to produce toxicity is often much higher and is dependent on the species and diet. Nitrate poisoning in livestock is not widespread (Keeney, 1983).

The over-enrichment of surface waters with nutrients, primarily nitrogen and phosphorus, results in a number of water quality changes which are usually regarded as undesirable. These effects include increased algal blooms, greater aquatic macrophyte growth, and dissolved oxygen depletion and are much more evident in lakes and impoundments than surface waters. Considerable research has been directed at this problem over the past three decades and efforts to classify lakes according to their trophic status have been developed by evaluating nutrient levels, transparency, and algal chlorophyll *a* concentrations.

Most models have been oriented primarily toward phosphorus due to the finding that phosphorus rather than nitrogen is usually the nutrient limiting lake

productivity. This is due in large to the relatively greater mobility of nitrogen compared to phosphorus, and to the uncontrollable sources of nitrogen. However, it is erroneous to assume that all lakes are phosphorus rather than nitrogen limited.

The major combined nitrogen species (organic nitrogen, ammonium ions, and nitrate) in the biosphere are interrelated by a complicated series of chemical and biological reactions referred to as the nitrogen cycle. Key biological transformations are: 1) immobilization or the assimilation of inorganic forms of nitrogen by plants and microorganisms to form organic nitrogen compounds; 2) ammonification or mineralization which is the decomposition of organic nitrogen to ammonium; 3) nitrification, the microbial oxidation of ammonium to nitrite; 4) denitrification, the reduction of nitrate or nitrite to dinitrogen oxide and nitrogen gas; and, 5) nitrogen fixation, the reduction of nitrogen gas to ammonia (Keeney, 1983).

Important chemical reactions in the soil nitrogen cycle are: 1) ammonia volatilization or sorption, the release or uptake of atmospheric ammonia by soils or plants; 2) ammonium exchange, the rapid and reversible exchange of ammonium from soil cation exchange sites to soil solution; 3) ammonium fixation, the entrapment of ammonium in the interlayers of 2:1 clay minerals such that it is only slowly released; and, 4) chemical denitrification, the reaction of nitrite with soil constituents at acid pH or under elevated temperatures to yield nitrogen gas or nitrogen oxides (Keeney, 1983).

Mineralization and immobilization operate as a dynamic equilibrium in

unfertilized soils and the amount of inorganic nitrogen in these soils is a function of the relative magnitude of these two opposing processes. As long as conditions are favorable for biological activity, these processes are constantly occurring. Net mineralization (inorganic nitrogen release in excess of that immobilized) often is 2-4% of the total soil N/yr in temperate climates. The net amount of nitrogen mineralized or immobilized in a given time is a function of many factors including type of substrate, temperature, and water and aeration status (Keeney, 1983).

Nitrification transforms the relatively immobile ammonium into nitrate which then can be readily leached, taken up by plants, or denitrified. The two-step nitrification process is almost exclusively mediated by obligate aerobic bacteria. *Nitrosomonas* is considered the most common bacteria for the first step: ammonium - oxygen yields nitrite, water, and hydrogen. *Nitrobacter* is the most common and best known nitrite oxidizer: nitrite plus oxygen yields nitrate. Nitrite is normally oxidized as rapidly as it is formed and does not accumulate except under high pH, high ammonia conditions which are unfavorable for *Nitrobacter* activity (Keeney, 1983).

Denitrification can be a major source of nitrogen loss. It is carried out by bacteria that are adapted to use nitrate or nitrite as a terminal electron acceptor when oxygen has been depleted. In addition to an anaerobic environment, organic matter is essential as a source of electrons and energy. The commonly accepted pathway is nitrate to nitrite to nitrogen oxide to dinitrogen oxide to nitrogen gas (Keeney, 1983).

Nitrogen is transported from agricultural lands by erosion, runoff, and leaching. It is difficult to distinguish between these routes on a watershed basis. A host of climatic, soil, topographic, land use, and management factors affect the forms and amounts of nitrogen reaching the receiving waters. The complexity of these factors along with the dynamic nature of the nitrogen cycle greatly complicate modeling of nonpoint source nitrogen pollution from agricultural lands.

In most cases the nitrogen leaving agricultural fields by surface runoff is in the organic form associated with eroded soil. The eroded soil is usually of finer particle size and has more associated organic matter than the bulk soil resulting in an enrichment of the eroded sediment nitrogen content. The ratio of sediment nitrogen to bulk soil nitrogen is the enrichment ratio. Menzel (1980) concluded that this ratio decreases as erosion increases and arrived at values ranging from 2.5 to 7.5 for cropland.

Surface runoff will also contain soluble nitrogen, primarily ammonium and nitrate. The concentrations and export of these inorganic forms are the net result of many factors and thus average values have little meaning. Precipitation adds significant inorganic nitrogen (5-10 kg/ha/yr) and usually more nitrogen is added in precipitation than is lost in surface runoff and baseflow. When precipitation or irrigation begins these two forms are leached into the soil. Ammonium will be retained at soil cation exchange sites, may nitrify, and erode with soil particles or reenter the soluble nitrogen fraction. Nitrate can move through the soil and enter into the soil nitrogen cycle, reappear in surface flow

downslope, or be leached to ground water. When precipitation exceeds infiltration the runoff water can obtain additional inorganic nitrogen from vegetation (plant residues). The ratio of inorganic nitrogen to total nitrogen tends to increase as more land in a watershed is in agricultural production (Keeney, 1983).

Nitrate leached below the root zone may end up in interflow, ground water, or reappear as surface or base flow. In many cropland situations this is the major path of nitrogen loss. The main processes in the movement of nitrate in soil are: 1) bulk flow (convection) of dissolved substances in soil solution due to mass flow of soil water; 2) molecular diffusion due to concentration gradients; and, 3) hydrodynamic dispersion due to mixing resulting from local differences in water flow velocity and direction. Soil water flow is caused by a driving force resulting from a potential gradient, and flow occurs in the direction of decreasing potential with the rate of flow proportional to the potential gradient and affected by the geometry of the pore channels. If the soil is saturated with water, gravitational force dominates and the soil water has a positive pressure potential as compared to a free water surface. The energy of water in soils is commonly expressed in bars, which is equivalent to the pressure exerted on a unit area by a 1020 mm column of water. Most of the water in soils is under matric potential but some is also acted on by gravitational and osmotic forces. These three forces combine to form the total potential energy of soil water (Ψ_{tot}). The rate of flow of water in soils is a result of: 1) the change in water potential per unit of distance; and 2) the hydraulic conductivity (K) which is the ability of the soil to

transmit water. These factors are related by a form of Darcy's Law: $q=K*\Psi_{tot}/L$, where q is the rate of flow and L is the distance between the two locations. When the soil is saturated the matric and solute forces are zero, and gravitational potential energy difference is the only driving force. The dominant factor affecting the rate of water movement to a lower position in the soil is the size and continuity of the pores in the soil (Keeney, 1983).

The nitrogen and hydrologic cycles are intimately linked and cannot be considered separately when evaluating control of nonpoint source nitrogen pollution. Suggested methods for the control of nitrogen loss from croplands include: 1) agricultural BMPs which reduce erosion and runoff; 2) efficient use of irrigation to minimize leaching of nitrate from the root zone before plant uptake can occur; 3) consideration of cropping sequences or cover crops that might utilize residual nitrate; 4) improved fertilizer management (timing, rates, evaluation of soil nitrogen availability); 5) use of slow release nitrogen fertilizers for long-season crops; and, 6) use of nitrification and urease inhibitors where appropriate (Keeney, 1983).

Nutrient parameters are of special interest. Historically, attention has been given to phosphorus and nitrogen because they are often limiting nutrients which are necessary for algal growth. Carbon is often found to be particularly abundant and thus attempts to limit excessive algal growth have focused on phosphorus and nitrogen.

Eutrophication is a natural process of lake aging whereby a lake matures from a relatively unproductive oligotrophic status to a highly productive eutrophic

state. Unfortunately, anthropogenic wastes have greatly accelerated this process in many lakes in developed areas. The nutrients nitrogen and phosphorus, abundant in anthropogenic wastes, are not often found in abundance in natural conditions. If anthropogenic activities produce significant quantities of these nutrients, and if they are allowed to enter stream and lake systems, they provide ample nutrients for accelerated growth. As a lake becomes increasingly productive certain species are no longer able to compete and diversity decreases. Accelerated rates of eutrophication of lakes have been attributed to increased amounts of nutrients discharged into waters flowing into the lakes (Warren, 1971). If nutrient levels are left unchecked, their abundance may lead to undesirable water quality conditions. The water quality problems can include reduced diversity of organisms and conditions which are aesthetically undesirable such as extensive algal blooms, reduced water clarity, and offensive odors. If the water body is used as a water supply source, increased cost for water treatment may be required.

Streams can also be adversely affected by increased levels of nutrients. It is certain that enrichment causes changes in the flora and encourages the growth of periphyton and macrophytes. In some areas of the United States there has been a documented decrease in clear-water fish species as a result of nutrient enrichment. These species are often replaced by warm-water species. Additionally, turbidity levels may increase due to increased suspended algae (Hynes, 1970).

Precipitation as a Source of Nutrients

The concentration and accumulation of phosphorus by precipitation is small relative to that of nitrogen. No phosphorus gases exist in the atmosphere. The only sources are airborne particulate matters, mostly dust, and a significant portion of that is organic in nature. A significant amount of phosphorus is removed from the atmosphere as dry fallout. Most of the phosphorus in rain is associated with dust. The amount of phosphorus added by precipitation is insignificant compared with that present in soils and fertilizers. The phosphorus loading by precipitation can be significant when compared with concentrations of phosphorus in surface runoff, especially from land not in agricultural production (Tabatabai, 1983 in Schaller and Bailey, 1983). Table III is a summary of estimated concentrations and loadings of phosphorus forms present in precipitation at various locations in the U.S.

Precipitation adds significant inorganic nitrogen (5-10 kg/ha/yr) and usually more nitrogen is added in precipitation than is lost in surface runoff and baseflow (Keeney, 1983). Tabatabai (1983) also estimated the areal loadings of various nitrogen forms from precipitation based on numerous studies. Table IV gives these estimates for various locations in the U.S.

TABLE III

AMOUNTS OF PHOSPHORUS DEPOSITED IN PRECIPITATION AT VARIOUS LOCATIONS

State	PO_4 as Phosphorus (mg/l)	Total Phosphorus (mg/l)	PO_4 as Phosphorus (kg/ha/yr)
Arizona	0.002	...	5.8
California	0.019 - 0.670 (0.260)	...	...
Florida	0.019 - 0.024	0.034	0.41 - 1.02*
Indiana	0.020 - 0.040	...	...
Iowa	0.089 - 0.259 (00.138)	...	0.71 - 2.07 (1.10)
Minnesota	...	0.028 - 0.060 (0.036)	0.22 - 0.49*
Nebraska	0.030 - 0.200 (0.100)	...	0.10 - 1.20 (0.40)
Ohio	0.036	0.027 - 0.190 (0.080)	...
Wisconsin	...	0.021	0.16*

* indicates total phosphorus. Numbers in parentheses are median values.
Source: from Tabatabai (1983) in Schaller and Bailey (1983)

TABLE IV

AMOUNTS OF INORGANIC NITROGEN DEPOSITED
BY PRECIPITATION AT VARIOUS LOCATIONS

State	NH_4 as Nitrogen (kg/ha/yr)	NO_3 as Nitrogen (kg/ha/yr)	Total Inorganic Nitrogen (kg/ha/yr)
Arizona	...	3.5	3.5
Florida	1.2 - 1.5	2.3 - 2.8	3.5 - 4.3
Iowa	3.7-10.9 (7.2)	4.4 - 9.2 (6.1)	9.8 - 17.6 (13.5)
Michigan	2.1 - 3.1	3.2 - 4.0	5.3 - 7.1
Minnesota	...	...	5.8 - 13.6 (8.7)
Nebraska	3.9 - 13.7 (4.9)	1.8 - 5.8 (3.6)	5.7 - 19.5 (8.5)

Numbers in parentheses are median values.
Source: Tabatabai (1983) in Schaller and Bailey (1983)

Unit Area Loading

Beaulac and Reckhow (1982) conducted an extensive literature review of nutrient export studies. They reviewed and evaluated the studies based on sampling design criteria and compiled the results according to land use for both total phosphorus and total nitrogen. Land uses identified in their review included row and non-row crops, pasture, mixed agriculture, urban, forest, and feedlots. Ranges of export coefficients for total phosphorus and total nitrogen were determined (Table V).

Vaithiyanathan and Correll (1992) examined the effects of land use on phosphorus transport in forested and agricultural soils of the Rhode River watershed, MD. Total phosphorus export coefficients for forest and agriculture were 0.31 ± 0.07 and 2.41 ± 0.85 kg/ha/yr, respectively. Sharpley et al. (1992) determined phosphorus transport from cropped and uncropped (grass) watersheds in the Southern Plains of the U.S. over a five year period. Export coefficients for grass watersheds ranged from 0.02 to 0.31 kg P/ha/yr, and from 0.29 to 14.9 kg P/ha/yr for cropped watersheds. Dillon and Kirchner (1975) measured export of total phosphorus from 34 watersheds in Southern Ontario over a 20 month period. Export coefficients for total phosphorus ranged from 0.025 to 0.15 kg/ha/yr for forested areas and from 0.080 to 0.37 kg/ha/yr for areas of mixed forest and pasture. Clesceri et al. (1986) derived total phosphorus and total nitrogen export coefficients for land uses of forest, mixed forest and agriculture, and agriculture from data for 17 watersheds in Wisconsin.

Forest export coefficients ranged from 0.09 to 0.13 kg P/ha/yr and 3.50 to 3.93 kg N/ha/yr. Mixed land use coefficients ranged from 0.14 to 0.24 kg P/ha/yr and 2.50 to 5.61 kg N/ha/yr. Coefficients for agricultural land uses ranged from 0.11 to 0.30 kg P/ha/yr and 3.24 to 16.4 kg N/ha/yr.

TABLE V

QUARTILE RANGES OF LAND USE/NUTRIENT EXPORT COEFFICIENTS

Land Use	Total Phosphorus (kg/ha/yr)	Total Nitrogen (kg/ha/yr)
Row Crops	0.9 - 5.3	4.0 - 21.8
Non Row Crops	0.6 - 1.5	4.1 - 6.5
Pasture	0.2 - 2.6	2.4 - 10.9
Mixed Agriculture	0.5 - 1.4	9.4 - 25.5
Urban	0.6 - 2.7	4.0 - 11.2
Forest	0.1 - 0.3	2.2 - 3.3
Feedlot	170 - 425	1,580 - 3,425

Source: Beaulac and Reckhow (1982)

Omernick (1977), summarizing data from the National Eutrophication Survey (1972 - 1975), provides some information specific to the Lake Tenkiller area for that period. Over the broad range of land uses in the Illinois River Basin above Lake Tenkiller, mean export coefficients of 0.10 kg/ha/yr for total phosphorus, and 3.63 kg/ha/yr for total nitrogen were calculated. More general values for the Central Region of the U.S. were determined by combining data from watersheds. Regional export coefficients for forest and agricultural land uses were calculated as 0.124 and 0.19 kg P/ha/yr, respectively, and 3.11 and

4.40 kg N/ha/yr, respectively.

In the Final Report for Beaver Lake - Phase I Clean Lakes Feasibility Study (FTN Associates, Ltd., 1992) phosphorus and nitrogen export coefficients were developed for the Beaver Lake Reservoir watershed. Literature including Reckhow et al. (1980) and Omernick (1977) was reviewed in obtaining the estimates export coefficients for land uses including crop, pasture, urban, forest, and other. Coefficients in this study were calibrated using discharge monitoring data from various sites around the lake. The export coefficient for the land use category of pasture was adjusted to equilibrate the results of the unit area loading method with the loadings calculated from discharge data. A summary of the export coefficients used in that report follows in Table VI.

TABLE VI

LAND USE/NUTRIENT EXPORT COEFFICIENTS USED FOR BEAVER LAKE WATERSHED (AR)

Land Use	Total Phosphorus (kg/ha/yr)	Total Nitrogen (kg/ha/yr)
Crop	0.6	4.22
Pasture	0.2 - 0.65	4.0 - 9.5
Urban	0.6 - 2.0	3.0 - 8.0
Forest	0.025	2
Other	0.045 - 0.05	1.2

Source: FTN Associates, Ltd., 1992

Confined Animal Waste Characteristics

Large confined animal production facilities and the large amounts of wastes produced have become a disposal problem. Water pollutants resulting from pastured animal production or application of animal wastes to the land include nitrogen, phosphorus, oxygen-demanding biodegradable substances, and pathogenic organisms. Of these, phosphorus and nitrogen are probably the runoff constituents of greatest significance for water quality since the others are attenuated markedly during transport in lotic systems.

Wastes from confined animal operations are generally concentrated. The significant impact of confined animal wastes can be attributed to the fact that the animals are managed in a relatively small area and their wastes concentrated and distributed over a limited area resulting in a high loading potential. Some wastes are collected in lagoons for biological processing while those from poultry tend to be stored for composting or field spreading. Characteristics of animal wastes vary based on the type of animal. Reddy (1981) in Overcash and Davidson (1981) compiled a set of animal waste characteristics based on research results. Table VII describes some characteristics that can be used in estimating the percent of nutrients in dry organic wastes.

TABLE VII

SELECTED CHARACTERISTICS OF ORGANIC WASTES

Waste Type	Nitrogen	Phosphorus
	Percent Dry Basis	
Animal Waste		
Beef	0.60 - 3.7	0.11 - 1.4
Dairy	1.50 - 3.7	0.41 - 0.7
Poultry	1.10 - 6.7	0.80 - 1.8
Swine	1.10 - 6.0	1.0 - 2.5
Sheep	4.0	---
Sewage Sludge	1.0 - 6.0	0.8 - 4.0
Digested Sewage Sludge	1.07	0.34
Digested Municipal Sludge	5.50	---
Plant Residues		
Corn	1.15	0.30
Rice	0.43	0.25
Alfalfa	4.0	0.50
Orchard Grass	2.9	0.50
	Parts Per Million	
Sewage Effluent	15 - 40	0.5 - 40
Swine Lagoon Effluent	224	50

Source: Reddy (1981) in Overcash and Davidson (1981)

Nonpoint Source Pollution

Progress in the control of point sources of pollution to water bodies has resulted in a relative increase in the contribution made by nonpoint sources. Nonpoint source (NPS) pollutants in rainfall induced runoff have become the primary focus in water quality improvements since they are the principal cause of

remaining water quality problems. NPS can be best described as those pollutant sources not identified, regulated, or managed as point sources. Generally, NPS pollution is generated from diffuse sources including: runoff from agricultural, silvicultural, and urban land uses; construction and resource extraction; leaks and spills; land disposal of wastes; and atmospheric deposition (U.S. EPA, 1991a).

Novotny and Chesters (1981) identified general characteristics of NPS:

* NPS discharge into surface waters in a diffuse and intermittent manner, usually associated with some meteorological event;
* NPSs are impacted from extensive land areas and the pollutant is transported overland to surface waters;
* NPSs are not readily monitored from the point of origin, nor is the overland path easily identified;
* Control of NPS pollution should be directed at specific sites with the most feasible operations including land management and conservation practices;
* NPS cannot be measured in terms of effluent limitations and compliance is generally based on land practices rather than water based technology; and,
* NPS are related to meteorological events, geologic and geographic conditions which are not readily controlled for pollution abatement.

NPS pollution can be derived from either man-made or natural processes. Natural processes include weathering of geologic materials. Agricultural runoff,

a primary contributor to lake and stream eutrophication, is an example of man-made source of NPS. Nonpoint source discharges are diffuse in nature and primarily occur during rainfall events when storm runoff from the land surface carries sediment, sediment-adsorbed chemicals, and dissolved chemicals into receiving water systems. Dissolved chemicals may also percolate through the soil to interflow regions and ground water and may eventually reappear as surface waters in baseflow.

Nonpoint source pollution is stochastic and dynamic and has multimedia dimensions. It is dynamic in the sense that land uses and configurations change over time making the pollutant mix vary both spatially and temporally. Table VIII describes some nation-wide estimates of the contributions of point and nonpoint sources of specific water pollutants. Over 80% of the discharge of total P and total N to U.S. waterways is attributed to agricultural nonpoint sources.

TABLE VIII

U.S. ANNUAL DISCHARGE OF POLLUTANTS TO WATERWAYS IN MILLIONS OF TONS (% OF TOTAL)

Source	BOD_5	Total Suspended Solids	Total Dissolved Solids	Total Phosphorus	Total Nitrogen
Industrial	3.8 (29.9)	22.9 (2.1)	132 (21.1)	0.16 (11.7)	0.25 (4.6)
Municipal	2.6 (20.5)	2.7 (0.2)	14.5 (2.3)	0.05 (3.6)	0.50 (9.3)
Total Point Source	6.4 (50.4)	25.6 (2.3)	146 (23.5)	0.21 (15.3)	0.75 (13.9)
Cropland	3.2 (25.2)	479 (43.4)	216 (34.7)	0.43 (31.4)	2.37 (43.9)
Woodland	0.8 (6.3)	97.3 (8.8)	55.4 (8.9)	0.12 (8.8)	0.49 (9.1)
Pasture	0.5 (3.9)	78.8 (7.1)	32.8 (5.3)	0.07 (5.1)	0.34 (6.3)
Range	1.8 (14.2)	424 (38.4)	173 (27.7)	0.54 (39.4)	1.45 (26.9)
Total Agricultural Nonpoint Source	6.3 (49.6)	1080 (97.7)	477 (76.5)	1.16 (84.7)	4.65 (86.1)
Total Point & Agricultural Nonpoint Source	12.7	1100	624	1.37	5.40

Source: After Duttweiler and Nicholson (1983)

Nonpoint Source Modeling

Novotny (in Girgini and Zingales , 1986) describes nonpoint source models and their characteristics. Nonpoint source simulation programs are part of a category of loading models which describe formation of runoff and generation of pollutants from a source area. They can be divided into continuous simulation or event oriented models. They also can be based on the distributed parameter or lumped parameter concept. They range from small field

sized application models to large deterministic, process-oriented large watershed models. Most available models range from the simple application of the Universal Soil Loss Equation (USLE) to multiple parameter-multiple component models.

Nonpoint source models fall generally into two categories, screening (unit loads) planning models, and hydrological assessment models. Screening models are usually simple tools which identify problem areas in a large basin. They usually rely on assessment of unit loads of pollution to the various lands within the basin. Unit load is a simple value or function expressing pollution generation per unit area and time unit for each typical land use. The concept provides a simple mechanism and quick answers to pollution problems of large areas where more complicated efforts would fail due to the enormous amounts of information required to model many complicated processes. The land/use pollutant loading is compatible with overview modeling where unit loads are combined with information on land use, soil distribution, and other characteristics to yield watershed loadings, or to identify areas producing or causing the highest amount of nonpoint pollution. Sediment loading functions for agricultural areas are usually based on the USLE. Use of the unit load presumes that an adequate inventory of land data is available from maps, aerial surveys, remote surveys, and local information.

Hydrologic models simulate the behavior of a hydrologic phenomenon in detail but they do not reproduce the phenomenon itself. Abstract models attempt to represent the system theoretically in mathematical form. These models

replace the relevant features of the system with a set of mathematical relationships.

There are basically two approaches to modeling nonpoint pollution. The most common are the lumped parameter models, while some are based on the distributed parameter concept. Lumped models treat the watershed, or a large portion of it, as one unit. The various characteristics of the unit are then lumped together, often with the use of an empirical equation, and the final form and magnitude of the parameters are simplified to represent the model unit as a uniform homogeneous system. Distributed parameter models divide the system into elements. Each element has uniform system parameters, soils, imperviousness, crop, slope, etc. The mathematical foundation of these models commonly uses the finite difference, or finite element representation of the basic differential equation governing flow and mass continuity and motion in one, two or three dimensions.

Models can be designed to run on an event or continuous basis. Discrete event modeling simulates the response to a watershed to a major rainfall or snowfall event. The principal advantage of event modeling over continuous simulation is that it requires relatively little meteorological data and can be operated with shorter computer run time. The principal disadvantage of event modeling is that it requires specification of design storm and antecedent moisture conditions, or detailed rainfall input, thereby assuming equivalence between the recurrence interval of the storm and the recurrence interval of the runoff. The principal advantage of continuous modeling is that it provides a long-term series

of water and pollutant loadings that can be analyzed statistically as to their frequency. A principal disadvantage of continuous modeling is that it requires long simulation runs, thus it may restrict the number of alternatives that can be efficiently investigated.

Typical components of nonpoint pollution models are (Novotny, 1986, in Girgini and Zingales, 1986):

1) A surface water generation component which describes the transformation of precipitation into runoff and overland flow components;
2) Soil and ground water components which describe movement of water through the unsaturated soil zone and into the saturated ground water zone;
3) Runoff and sediment routing component to obtain runoff flow or pollution histograms from the excess rain and eroded soil routing to the watershed outlet;
4) Erosion component which estimates soil loss from pervious areas;
5) Pollutant accumulation and wash-off from impervious areas component estimating pollutant mass; and,
6) The soil adsorption/desorption component which determines the distribution of adsorbed and dissolved fractions of pollutants in soil solution and runoff.

CHAPTER IV

METHODOLOGY

Autosampler Placement and Sample Collection

To obtain data concerning in-stream phosphorus and nitrogen concentrations during runoff events in the Illinois River Basin, automatic water samplers were placed in the field at four locations within the Oklahoma portion of the basin. Three American Sigma (Model 702), and one Isco (Model 2900) samplers were used. These models are discrete samplers with a capacity to collect a total of 24 individual samples. Using a 12-volt DC power system, the samplers can be programmed to collect up to 500 ml per sample at selected time intervals. Each sampler was equipped with a liquid level actuator to activate the samplers when the stream level reached a predetermined level. The sampling interval was set so that the samplers would collect samples throughout the duration of a runoff event (2 - 3 hours between samples). The samplers were programmed to record the time and date of each sample collected. The last available sample bottle in each sampler was filled with deionized distilled water to serve as an analysis control.

The samplers were located at USGS gaging stations so that reasonably

accurate stream discharge measurements could be obtained to correspond with sample concentrations. The four sites chosen were: the Illinois River near Watts, OK (USGS 07195500), Flint Creek near Kansas, OK (USGS 07196000), Baron Fork Creek near Eldon, OK (USGS 07197000), and the Illinois River near Tahlequah (USGS 0716500).

The USGS ADAPS database, which allows a user to obtain 'real-time' stream gage height and discharge at USGS gaging stations, was used to determine when the samplers were activated and when a sampling session was complete, as well as the stream discharge at the times when samples were taken by the automatic samplers. Samples were retrieved within 24 hours after the last sample was drawn and were preserved under ice for transport and storage. Analyses were performed within 48 hours of collection.

Total phosphorus was analyzed using the persulfate digestion method of Menzel and Corwin (1965) as described by Lind (1985). Total nitrogen was analyzed using a persulfate digestion method described by Bachman and Canfield (1991).

Analysis of Historical Water Quality Data

Historical water quality data were obtained from U. S. EPA's STORET database for eight long-term USGS monitoring stations within the Illinois River Basin. The request was formulated to include discharge, nitrogen, and phosphorus data collected by federal and state agencies from water years 1980 through 1993. A list of the USGS gaging stations and their locations is given in

Table IX. Mean daily discharge at each of the eight gaging stations were obtained from the USGS ADAPS database for water years 1980 through 1993.

TABLE IX

ILLINOIS RIVER BASIN USGS WATER QUALITY MONITORING GAGING STATIONS

Station Identification	Verbal Description	Latitude/Longitude	River Kilometer
USGS 07194800	Illinois River W of Savoy, AR at Hwy 16 bridge	36 06 11/94 20 39	214
USGS 07195400	Illinois River at Hwy 16 S of Siloam Springs, AR	36 08 41/94 29 41	186
USGS 07195500	Illinois River at Hwy 54 N of Watts, OK	36 07 48/94 34 12	171
USGS 07196500	Illinois River at Hwy 62 NE of Tahlequah, OK	35 55 17/94 55 15	90
USGS 07195000	Osage Creek near Elm Springs, AR	36 13 19/94 17 18	16
USGS 07196000	Flint Creek at Hwy 412 near Kansas, OK	36 11 54/94 42 30	4.5
USGS 07196900	Baron Fork Creek at Hwy 59 near Dutch Mills, AR	35 52 48/94 29 11	59
USGS 07197000	Baron Fork Creek at Hwy 51 at Eldon, OK	35 55 16/94 50 18	14

Simple descriptive statistical analyses were computed for each of the gaging stations for nitrogen and phosphorus concentrations, discharge, and total nitrogen and total phosphorus loads. Additionally, a statistical software package called FLUX (Walker, 1987b) was used to estimate nutrient loads at the gaging stations using a flow-weighting technique. Concentration and discharge data were matched by date and divided into three strata based on stream discharge for the entire period of record (WTRs 80 - 93) at each gaging station. The first

stratum (low flow) included those observations with stream discharge from zero to one-half of the mean discharge (low flow $\leq$ ½Q_μ). The second stratum (medium flow) included all those observations with discharge from one-half mean discharge to twice mean discharge (½Q_μ < medium flow $\leq$ 2Q_μ). The third stratum (high flow) included all those observations with discharge greater than twice mean discharge (2Q_μ < high flow).

A flow-weighted mean concentration method was used to calculate average annual loading. Flow-weighted concentration was calculated as:

$$c_{fw} = c_i \ (Q_\mu / q_i)$$

where

c_{fw} = flow-weighted concentration,

c_i = i^{th} observed concentration,

Q_μ = mean discharge, and

q_i = i^{th} observed discharge.

Flow-weighted concentrations were calculated for all observations within a stratum using the mean discharge (Q_μ) for that stratum. The mean discharge for a stratum was calculated as the sum of all long-term flows in that stratum divided by the number of observations. The mean flow-weighted concentration for each stratum was calculated as the sum of all flow-weighted concentrations in the stratum divided by the number of observations. The average annual load contributed by each stratum was then calculated as the mean flow-weighted

concentration for stratum *i* multiplied by the mean discharge for stratum *i* multiplied by the percent frequency of observed flows within stratum *i*. Total average annual load was calculated as the sum of the annual loads of each stratum.

First-Order Kinetic Assimilation Rates

When stream nutrient fluxes are calculated, to determine loading quantities at a particular point in a stream system, the water column nutrient concentration and stream discharge are used. Over a period of time and stream distance, average nutrient concentrations in the water column may appear to decrease from an upstream point to a downstream point, assuming no additional nutrient inputs between the points, due to biological incorporation of nutrients and particulate settling. The rate for this assimilation can be estimated if in-stream data are available, and an assumption is made that no significant additional sources of nutrient pollution enter the stream between the upstream and downstream points.

Keup (1968) reviewed sources of phosphorus in flowing water on the basis of origin and quantities, and determined that phosphorus within a stream is assimilated from the flowing mass by biota and sedimentation. He found this assimilation rate to be measurable, and estimated the rate of removal using first-order kinetics as 0.007 per kilometer for the South Platte River (CO), and 0.012 per kilometer for the Pigeon River (NC). Simmons and Cheng (1985) examined the rate and pathways of phosphorus loss from the Nepean River (AUS).

Phosphorus added to the system was found to follow first-order kinetics in its removal from the flowing system at a rate of 0.005 per kilometer. Reddy et al. (1996) conducted field and laboratory studies to determine phosphorus assimilation capacity of Otter Creek (FL) and calculated phosphorus uptake as a function of distance using first order kinetics. Reddy et al. (1996) obtained values for two different times to be 0.11 per kilometer and 0.023 per kilometer.

Allen (1995) summarized research attempting to describe nutrient cycling in streams. Nutrient cycling is the process where an atom in the water column, in a dissolved phase, is incorporated into living tissue, passes through several links in the food chain, is released by excretion, and, through decomposition processes, is made available again. Allen (1995) describes work by Newbold et al. (1981, 1982, 1983) developing a theoretical framework for a concept called nutrient spiraling. Nutrient spiraling describes the path and distance traveled of a nutrient as it is transported downstream. Newbold et al. (1981) describe two major compartments of the spiraling distance consisting of the water column and biota. The water compartment is a measure of the distance traveled by a nutrient atom in inorganic form until uptake by biota occurs. The biota incorporating the nutrients are usually attached microorganisms, periphyton, and benthic invertebrates. The biota compartment measures the distance biota retain the nutrient until release back into the water column. First-order assimilation rates can be used to estimate the distance a nutrient atom travels in the water compartment of Newbold et al.'s (1981) framework.

First-order kinetic rates for in-stream assimilation of nutrients due to

biological uptake and settling were calculated from differences in total phosphorus and total nitrogen concentrations at two sets of water quality monitoring sites on the Illinois River in Oklahoma. STORET water quality data from 1986 to 1993 for two sets of monitoring stations, SR1 and SR2, and SR3 and SR4, were paired by date, and USGS ADAPS data were used to determine flow regimes for the dates based on discharges at USGS 07195500 on corresponding dates. These scenic river monitoring sites are indicated in Figure 1. SR1 is 3.2 km downstream from USGS 07195500. SR2 is just above the confluence with Flint Creek 16.7 km below SR1. SR3 is 11.4 km below SR2. SR4 is 7.1 km below SR3. It was assumed that no significant nutrient inputs occurred between the upstream and downstream stations.

The kinetic assimilation rates were then calculated as:

$$k_s = \frac{\ln\left(\frac{c_s}{c_o}\right)}{s}$$

where

k_s = in-stream first-order kinetic assimilation rate,

c_o = average concentration at the upstream station,

c_s = average concentration at distance s downstream, and

s = the distance between the stations in km (Nemerow, 1985).

First-order kinetic assimilation rates were calculated for three flow regimes (low, medium, and high) discussed earlier for both total phosphorus and for total nitrogen concentration observations. Kinetic assimilation rates calculated for

each flow regime from the two pairs of water quality monitoring stations were averaged to obtain estimates of the rates for the Illinois River and its tributaries as a whole for each nutrient and each flow regime. The resulting first-order assimilation rates applied in this study were -0.022, -0.016, -0.001 per kilometer for phosphorus assimilation at low, medium, and high flows, respectively. First-order assimilation rates for nitrogen were calculated as -0.017, -0.009, -0.006 per kilometer at low, medium, and high flows, respectively.

The development of the discharge sensitive first-order kinetic assimilation rates then allows for estimation of the contribution of point source nutrients to total nutrient loads at some point downstream given varying flow using the equation:

$$L_s = L_o e^{k_s s}$$

where

L_s = load remaining at distance s downstream,

L_o = the initial load,

k_s = the first-order kinetic assimilation rate, and

s = the distance downstream.

Point Sources

Estimates of municipal point source nutrient loads were obtained from personal communications with William Keith of the Arkansas Department of Pollution Control and Ecology (ADPCE) and Mark Derichsweiler of the Oklahoma

Department of Environmental Quality (ODEQ), and Permit Compliance System (PCS) output for dischargers in the Illinois River Basin. The ADPCE data were estimates for calendar years 1990 to 1991 or later. ODEQ data were estimates based on data from 1992 to 1993. PCS data for 1992 to 1993 were used for Tahlequah since their newer system went on-line in 1991. PCS data estimates for Fayetteville's discharges were made from the same period. The 1993 Curtis Report (OK Department of Agriculture, and Plant Industry and Consumer Services, 1993) was used to obtain estimates for nutrient discharges from plant nurseries in the Oklahoma portion of the Illinois River Basin. Point source discharge, concentration, and annual loading estimates are given in Tables XXII, XXXII, and XXXIII.

Background Loading

An EPA screening method (Mills et al., 1985) was used to approximate background nutrient loads that represent the "chemical . . . composition of surface waters which would result from natural causes and factors." While this concept may be somewhat artificial in that few, if any, waterbodies in the U.S. remain unaffected by human activity, it can provide an estimate of a "baseline level of pollution which cannot be eliminated by local or area-wide water quality management" (Mills et al., 1985).

Background pollution levels could be determined by measuring water quality levels in areas which are free of human activity. In the absence of such data, approximations were made using information from the USGS Hydrological

Benchmark Network (McElroy et al., 1976). These data represent data from monitoring stations considered free of human disturbance. Additionally, nutrient concentration data from the U.S. National Eutrophication Survey (Omernick, 1977) for areas of "≥90% Forest" category were used to estimate background nutrient concentrations in the Illinois River Basin.

USGS Water Resources Data for Oklahoma (Blazs et al., 1991) was used to obtain the drainage areas of the Illinois River Basin watershed above USGS gaging stations. Average annual runoff was determined by dividing average annual streamflow by the drainage area. Estimates of background concentration were obtained from Omernick (1977) for the Central U.S. Region category of "≥90% forest", and background concentrations of total phosphorus and total nitrogen were estimated as 0.02 mg P/l and 0.5 mg as N/l.

GIS Modeling - SIMPLE

The Spatially Integrated Model for Phosphorus Loading and Erosion (SIMPLE) (Storm et al., 1997; Sabbagh et al., 1995) is a continuous simulation, distributed parameter modeling system designed to predict sediment transport and phosphorus loading to surface waters from nonpoint sources on a basin scale using a geographic information system (GIS), Geographic Resources Analysis Support System (GRASS), for data storage and display. SIMPLE was written in UNIX-C and operates in the SUN Openwindows environment (Sunsoft, 1991). The spatial component of SIMPLE is raster based, using either a single cell or a field consisting of multiple cells as the computational unit. Fields were

delineated in this study by land use. Input variables for SIMPLE are entered manually and/or as GRASS raster files. SIMPLE input data files consist of input variables for each data group for each computational unit. The modeling system consists of a phosphorus and sediment transport model, a detailed digital terrain model, a database manager, and a menu driven user interface. SIMPLE estimates daily sediment-bound phosphorus, soluble phosphorus, and sediment loading from each cell or field. Average phosphorus loading statistics are calculated on a daily, monthly, and annual basis.

Rainfall

Rainfall is the driving force for transporting phosphorus from a site, either solubilized in surface water or leachate, or adsorbed to sediment particles subject to erosion. Rainfall intensity, volume, interception, infiltration and storage are all factors defining surface runoff and soil erosion which control the phosphorus lost from fields in soluble and sediment-bound forms. SIMPLE accepts measured daily rainfall data which is representative of typical rainfall in the region of interest.

Surface Runoff

Runoff volume from a rainfall event is estimated by the Soil Conservation Service (SCS) curve number method (SCS, 1985). This method is widely used due to its simplicity and the availability of data. Because daily rainfall is used, the rainfall and subsequent runoff is treated as a 24 hour event. To estimate the

daily curve number, a weighted curve number, CN, is estimated by:

$$CN = W_1 \; CN_1 \quad W_2 \; CN_2 \quad W_3 \; CN_3$$

where W_1, W_2, and W_3 are weighting factors, and CN_1, CN_2, and CN_3 are curve numbers for antecedent soil moisture conditions 1, 2, and 3, respectively. The weighting factors are estimated using:

$$W_1 = 1 \quad if \; V_p \quad f_1; \quad W_1 = \frac{f_1}{V_p} \quad if \; V_p \quad f_1$$

$$W_2 = 0 \quad if \; V_p \quad f_1; \quad W_2 = \frac{V_p \quad f_1}{V_p} \quad if \; f_1 \quad V_p \quad f_2; \quad W_2 = \frac{f_2 \quad f_1}{V_p} \quad if \; V_p \quad f_2$$

$$W_3 = 0 \quad if \; V_p \quad f_2; \quad W_3 = \frac{V_p \quad f_2}{V_p} \quad if \; V_p \quad f_2$$

where V_p is rainfall volume (cm), f_1 and f_2 are 1.25 and 2.75 cm during the dormant season, and 3.5 and 5.25 cm during the growing season, respectively (Smedema and Rycroft, 1983).

<u>Erosion</u>

Soil erosion is an important transport mechanism for phosphorus. Sediment-bound phosphorus losses can be calculated from the amount of eroded soil and the phosphorus content of the sediment. Gross annual soil loss is estimated by applying the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) on a daily event basis given as:

$$A_e\ (t) = 2.24\ R\ K\ LS\ C\ P$$

where $A_e(t)$ is gross annual soil loss (Mg/ha/yr)on day *t*, R is a rainfall factor (English units), K is a soil erosivity factor (English units), LS is the length and slope factor, C is a cover factor, and P is a practice factor. For a 24 hour rainfall, the rainfall factor is calculated on a storm basis using (Cooley, 1980):

$$f(D) = D^{0.0086}$$

and

$$R = \alpha\ D^{-\beta} \left(\frac{P_r}{25.4} \right)^{2.119\ f(D)}$$

where D is storm duration (hr), P_r is total storm rainfall (mm), and and are constants for a given rainfall distribution. Using a Type II rainfall distribution with a 24 hour duration, the two above equations can be simplified to:

$$R = 4.811 \left(\frac{P_r}{25.4} \right)^{2.178}$$

The USLE length and slope factors are estimated based on McCool et al. (1989) and McCool et al. (1987), respectively.

<u>Phosphorus Transformations</u>

The transformations of plant available phosphorus to mineral phosphorus

is described by the function developed by Jones et al. (1984):

$$TR_q\ (t) = q_i\ (t) \quad q_m\ (t)\ \frac{PSP}{1 \quad PSP}$$

where $TR_q(t)$ is the amount of phosphorus transformed (g P/g soil), $q_i(t)$ is plant available phosphorus (g P/g soil) at day *t*, $q_m(t)$ is mineral phosphorus (g P/g soil) at day *t*, *t* represents the current simulation day, and PSP is the phosphorus sorption coefficient taken as 0.5. If $TR_q(t)$ is greater than zero, then:

$$q_m\ (t\ \ 1) = q_m\ (t) \quad TR_q\ (t)$$

$$q_i\ (t\ \ 1) = q_i\ (t) \quad TR_q\ (t)$$

where (t+1) represents the next simulation day. If TR_q is less than zero, then:

$$q_m\ (t\ \ 1) = q_m\ (t) \quad 0.1\ \ TR_q\ (t)$$

$$q_i\ (t\ \ 1) = q_i\ (t) \quad 0.1\ \ TR_q\ (t)$$

At the start of the computer simulation, the initial plant available phosphorus, q_i, is a user specified input, the initial mineral phosphorus, q_m, is assumed to be equal to q_i, and the initial organic phosphorus, q_o, is approximated by the percent organic carbon in the soil assuming a C:P ratio of 100:1.

<u>Dissolved Phosphorus Concentration</u>

Several methods have been used to estimate dissolved phosphorus concentration in surface runoff. In the SIMPLE model, the concentrations of dissolved phosphorus are estimated based on the adsorption-desorption processes. A linear isotherm of the form:

$$q_i\ (t) = K_d\ C\ (t)$$

where K_d is a constant taken as 175 cm^3/g (Williams et al., 1984) and $C(t)$ is the dissolved phosphorus concentration (mg/l). The application of the linear isotherm assumes that the rainfall fully reacts with the top 10 mm of the soil.

Phosphorus and Sediment Loading to Receiving Waters

Sediment and Sediment Bound Phosphorus. Sediment bound phosphorus and sediment loss can be calculated from the amount of eroded soil and the phosphorus content in the sediment. The amount of sediment bound phosphorus leaving the field may be reduced along its route to the receiving stream due to deposition. The flow path, surface roughness, and slope must be known to estimate the actual loading. SIMPLE uses the relationship developed by Heatwole and Shanholtz (1991) to calculate a delivery ratio as:

$$S_f = S_{fmin} \quad \exp[\ k_2\ (S \quad S_o)]$$

$$DR = \exp(\ k_1\ D_s\ S_f)$$

where DR is a delivery ratio, D_s is distance to the stream (m), S is slope (m/m),

k_1, k_2, S_o, and S_{fmin} are constants. Based on delivery estimates from Draper et al. (1979), Heatwole and Shanholtz (1991) defined $k_1 = 0.0161\ m^{-1}$, $k_2 = 16.1$, $S_o = 0.057$, and $S_{fmin} = 0.6$. The amount of sediment reaching the stream, $A_s(t)$ (Mg/ha) on day t, is estimated by:

$$A_s(t) = A_e(t)\ DR$$

Due to the selective deposition process, sediment contains finer soil particles than the original soil matrix. Thus, the phosphorus content is higher in the sediment than in the soil due to the higher adsorptive capacity of the finer soil particles. Phosphorus concentration in the sediment is estimated based on the soil phosphorus content in the soil matrix and an enrichment ratio. The phosphorus enrichment ratio is defined as the ratio between phosphorus contents in sediment and soil. The soil phosphorus content is obtained based on a daily mass balance. The phosphorus enrichment ratio is estimated by (Menzel, 1980):

$$\ln(PER(t)) = 2 \quad 0.2 \ln(1000\ A_e(t))$$

where PER(t) is the phosphorus enrichment ratio. For sediment bound phosphorus, the phosphorus reaching the stream is found using a delivery ratio and PER, given as:

$$P_i(t) = 0.001\ q_i(t)\ A_e(t)\ PER(t)\ DR$$

$$P_o(t) = 0.001\ q_o(t)\ A_e(t)\ PER(t)\ DR$$

$$P_m\ (t)\ =\ 0.001\ q_m\ (t)\ A_e\ (t)\ PER\ (t)\ DR$$

$$P_{sed}\ (t)\ =\ P_i\ (t)\quad P_o\ (t)\quad P_m\ (t)$$

where for the event $P_{sed}(t)$ is the sediment bound phosphorus loss (kg/ha) on day *t*, $P_i(t)$ is plant available phosphorus loss (kg/ha) on day *t*, $P_o(t)$ is organic phosphorus loss (kg/ha) on day *t*, and $P_m(t)$ is mineral phosphorus loss (kg/ha) on day *t*.

<u>Dissolved Phosphorus</u>. Dissolved phosphorus in surface runoff is assumed to be conservative, such that losses do not occur during transport to the receiving stream. Under this assumption, dissolved phosphorus loading to the receiving stream is equal to the dissolved phosphorus loss from the field. The dissolved phosphorus loading for an event can then be expressed as:

$$P_{sol}\ (t)\ =\ 0.01\ V_q\ (t)\ C\ (t)$$

where $P_{sol}(t)$ is the dissolved phosphorus loading in runoff (kg/ha) on day *t*, $V_q(t)$ is runoff volume (mm) on day *t*, and $C(t)$ is phosphorus concentration in runoff (mg/l) on day *t*.

<u>Total Phosphorus</u>. Total phosphorus loading for each day, $P_t(t)$ (kg/ha), is calculated from:

$$P_t\ (t)\ =\ P_{sol}\ (t)\quad P_{sed}\ (t)$$

where $P_{sol}(t)$ and $P_{sed}(t)$ are equal to zero if there is no rainfall for the day.

Monthly and annual phosphorus loading is obtained by summing the loading for each day.

<u>Phosphorus Mass Balance</u>. SIMPLE performs a mass balance on the top 10 mm of soil. The total phosphorus in the soil is defined as:

$$q_t\ (t) = q_i\ (t) \quad q_o\ (t) \quad q_m\ (t)$$

where $q_t(t)$ is total soil phosphorus (g P/g soil), and $q_o(t)$ is organic phosphorus (g P/g soil) all on day *t*. The initial phosphorus in the top 10 mm of soil is evaluated daily based on the gain of phosphorus due to application of fertilizer or manure, and loss in surface runoff. The initial dissolved phosphorus for each day is evaluated by the following equations:

$$q_i\ (t\ \ 1) = q_i\ (t) \quad q_f\ (t) \quad \frac{100\ \ P_i\ (t)}{\beta_b\ d} \quad \frac{100\ \ P_{sol}\ (t)}{\beta_b\ d}$$

$$q_m\ (t\ \ 1) = q_m\ (t) \quad \frac{100\ \ P_m\ (t)}{\beta_b\ d}$$

$$q_o\ (t\ \ 1) = q_o\ (t) \quad \frac{100\ \ P_o\ (t)}{\beta_b\ d}$$

where $q_f(t)$ is the increase of total available soil phosphorus content due to application of fertilizer (g P/g soil), $_b$ is the soil bulk density (g/cm^3), d is the depth of interaction (mm) assumed to be 10 mm. It should be noted that interactions between organic with mineral and available phosphorus are

neglected.

The application of commercial or manure fertilizer is assumed to increase the plant available soil phosphorus content immediately after application. When applied by surface broadcast, all phosphorus is assumed to be added to the top centimeter of the soil. When the fertilizer is incorporated by other methods, the phosphorus is assumed to be evenly distributed throughout the application depth. The available phosphorus content added to the top centimeter of soil due to fertilizer application, $q_f(t)$, is calculated as:

$$q_f\,(t) = 10\,\frac{P_f\,P_c}{\beta_b\,d_f}$$

where P_f is the amount of fertilizer applied (kg/ha), P_c is the available phosphorus content fraction in the fertilizer or manure, $_b$ is soil bulk density (g/cm^3), and d_f is the fertilizer application depth (mm) which must be greater than or equal to 10 mm.

Some phosphorus in animal manure is in organic forms, while some is in inorganic forms immediately available for plant uptake. Organic phosphorus can be mineralized and become available for plant uptake. The rate of mineralization depends on various factors such as soil microbial activity and temperature. SIMPLE assumes mineralization in animal manure is rapid such that all phosphorus becomes immediately available after application. This assumption will initially overestimate the available phosphorus content of the soil.

Plant uptake is another potential loss of phosphorus from the soil, and is a function of plant type and state of growth. The uptake process occurs in the

entire active root depth. Because the top centimeter of soil is used for mass balance calculation, and this depth is relatively small compared to the active root depth, phosphorus uptake by plants is assumed to be negligible. SIMPLE also neglects phosphorus losses due to percolation.

SIMPLE Data Inputs

There are essentially three data layers required for running the SIMPLE model: soils, elevation, and land use. Assistance in the creation of these data layers was provided by individuals in OSU's Agronomy Department and Biosystems & Agricultural Engineering Department.

Soils. Soils data were obtained by digitizing and attributing 1:20,000 scale Natural Resources Conservation Service (NRCS) county soil survey maps for all areas included in the basin and then "patching" them together to create a final attributed vector product. The University of Arkansas digitized the Arkansas portion of the basin. This vector file was then converted into a 30 meter resolution raster file within GRASS for use with the SIMPLE model. Figure 2 shows soils in the watershed differentiated by color.

Elevation. The elevation data layer was created by patching together all available USGS 1:24,000 (30 meter resolution) scale quadrangle maps covering the Illinois River Basin. Seven quadrangles in the Arkansas portion of the basin did not have 1:24,000 scale data available. Here, 80 meter resolution data, which were available, were patched in using a smoothing technique to minimize any dramatic variations between the data of different resolutions. The elevation

map for the basin is shown in Figure 3.

Land Use. The land use layer was digitized from 1985 aerial photography of the basin. The Oklahoma portion of the basin was digitized and attributed by OSU Department of Agronomy. The Arkansas portion of the basin was obtained from the University of Arkansas. These products also had to be patched together and cleaned up to create a continuous vector file encompassing the entire Illinois River Basin. This vector file was then converted to a 30 meter resolution raster format within GRASS for use with the SIMPLE model. The land uses were reclassed into seven categories to simplify the modeling process (Figure 4). The seven resulting land use classes were:

1) crop (all agricultural crop types),
2) feedlots (including all confined animal operations),
3) forest (all types),
4) pasture and range land,
5) roads and rights-of-way (including all utility lines and roads of all classes),
6) urban (including residential, commercial, and industrial areas), and
7) water (including streams, ponds, and reservoirs).

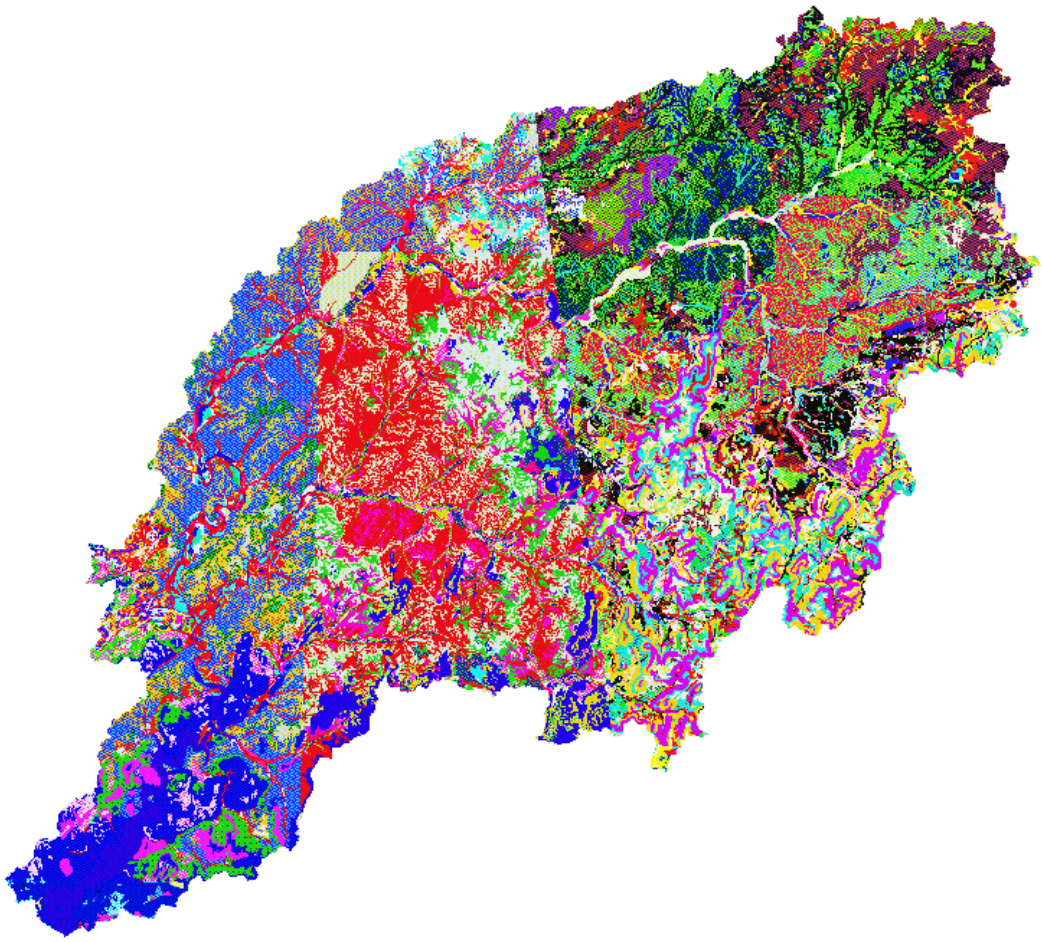

Figure 2. Soil Types in the Illinois River Basin Derived From County Soil Surveys.

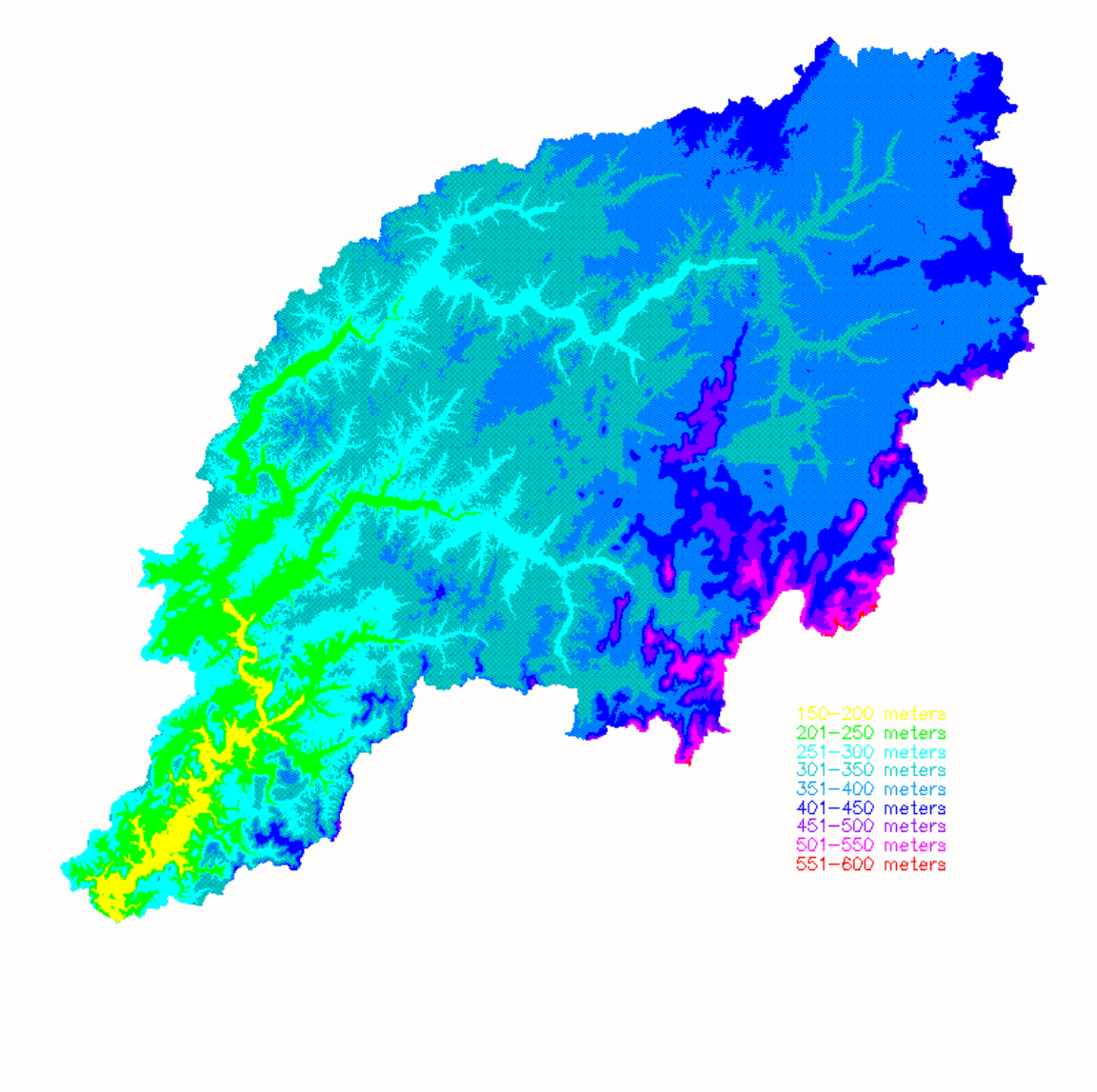

Figure 3. Elevation in Meters in the Illinois River Basin Derived From 7.5 Minute Quadrangle 30 Meter Resolution DEMs and 80 Meter Resolution DMA Data.

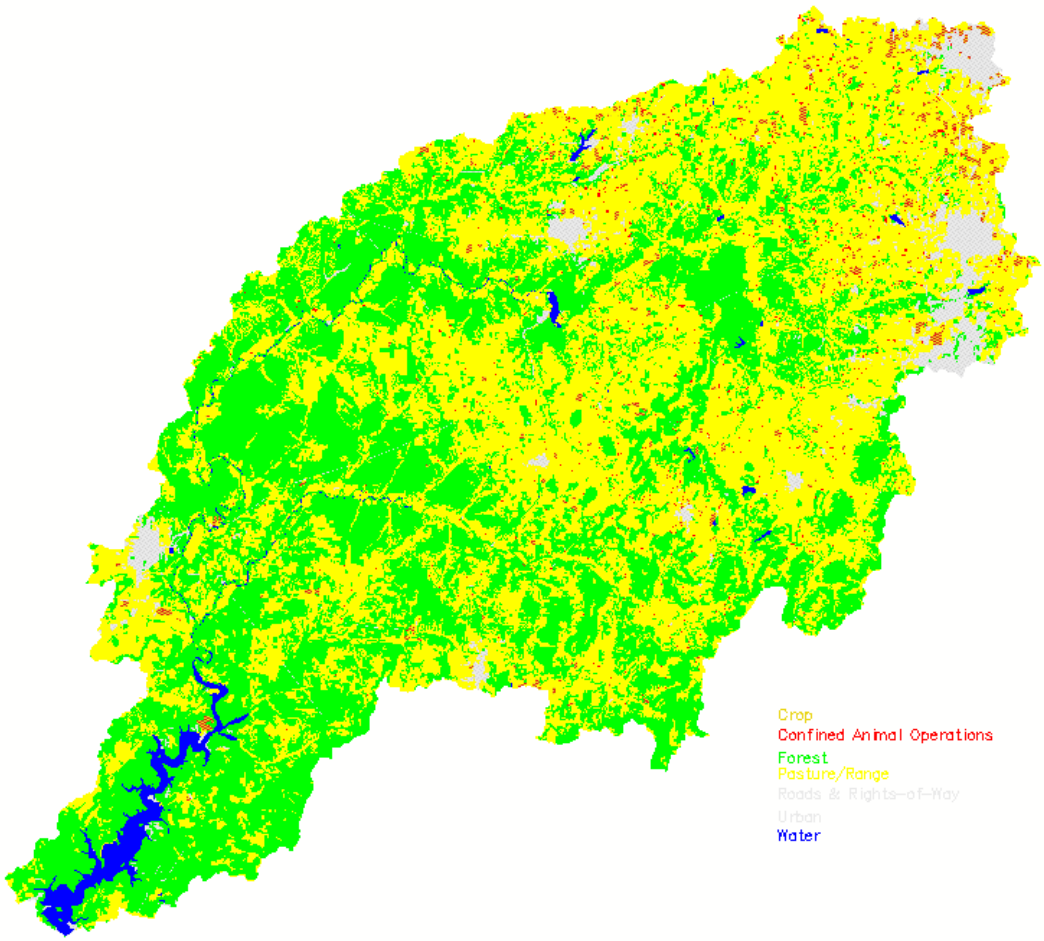

Figure 4. The Seven Major Land Uses, Delineated for the SIMPLE Modeling of the Illinois River Basin, Derived From 1985 U.S. EPA Data.

Subbasin Delineation

The basin was divided into smaller subbasins of manageable size using a digital terrain model (DTM) (Sabbagh et al., 1994) incorporated into the SIMPLE model. The DTM was applied to the digital elevation layer. The DTM delineates and enumerates flow networks, outlines subwatershed boundaries, and determines on a subwatershed basis the area, average slope, maximum travel distance and the slope to the receiving stream. The model also estimates for each cell or field, slope, path length to the stream, and the path slope (Sabbagh et al., 1994).

The DTM is divided into five components that contain procedures to (1) detect and fill depressions, (2) define flow direction and calculate flow accumulation values, (3) delineate channel networks, (4) define drainage boundaries, and (5) extract cell and drainage characteristics such as slope, path length, and slope of the path (Sabbagh et al., 1994).

Rainfall

Weather stations in and around the Illinois River Basin with at least 25 years of precipitation data (1960 - 1985) were used to provide rainfall input. The weather stations used in this study included Bentonville, Fayetteville, Siloam Springs, and Odell in Arkansas, and Kansas, Stilwell, Talhequah, and Webbers Falls in Oklahoma. Table XIX lists the weather stations used for watersheds within the Illinois River Basin.

SIMPLE Parameter Estimation

Methods used to estimate parameters required for input into the SIMPLE model of the Illinois River Basin are outlined in Storm et al. (1996) and are described below.

Topographic Parameters

SIMPLE requires cell/field slope, slope length, distance to stream and slope of distance to stream. The DTM used the 30 m DEM to estimate cell slope, and distance and slope to stream using procedures described by Sabbagh et al. (1994). However, the DEM was not detailed enough to estimate slope length. Therefore, slope length was estimated using a modified procedure developed by the Oklahoma NRCS. Slope length ( ), as used in the USE, was estimated based on county soil classification using two categories, upland soils and bottom land soils. All bottom land soils were assumed to have a slope length of 50 feet. The slope length for the upland soils was based on the soil mapping field slope as follows:

0 to 1 percent slope	=>	600 foot slope length,
1 to 3 percent slope	=>	500 foot slope length,
3 to 5 percent slope	=>	400 foot slope length,
5 to 8 percent slope	=>	300 foot slope length,
8 to 12 percent slope	=>	200 foot slope length, and
> 12 percent slope	=>	50 foot slope length.

Table X presents field slope and slope length statistics for each watershed, and Table XI gives the slope length for each soil type.

The next step was to define the stream network using the DTM. For each subbasin an arbitrary cut off value was selected to define the stream network. By trial and error the cut off value was changed until the stream network visually approximated the 1:24,000 USGS blue line streams (continuous and intermittent flow steams). Next, distance to stream was estimated based on the flow path predicted by the DTM. The slope of this distance to stream was calculated as the ratio of the elevation drop to the stream and the distance to the stream. Distance to stream and slope of distance to stream is given in Table X for each watershed.

Soil and Management Parameters

Based on the NRCS County soils surveys, Table XII describes the slope range and area for each soil type by county. Table XIII gives the USLE cover and management factors by land use based on USDA-SCS Handbook Number 537 (SCS, 1978). Hydrologic soil groups are given by land use in Table XIV based on NRCS County Soil Surveys.

Soil Phosphorus Parameters

Initial soil phosphorus is a very important input parameter for SIMPLE. Mehlich III soil test values were used as an estimate of the available soil phosphorus that was input into SIMPLE. Soil test phosphorus is typically estimated for a field using a composite of 0 to 6 inch soil samples. It should be noted that SIMPLE requires the amount of available soil phosphorus in the upper

one centimeter (cm) of the soil. However, based on validation and testing studies, we use the 0 to 6 inch composite Mehlich III soil test directly as the available soil phosphorus in the upper one cm of soil.

Several data sources of soil phosphorus for the Upper Illinois River Basin were available. However, detailed soil test phosphorus data were available only for a few small watersheds within the basin. Therefore, a method was developed to estimate soil phosphorus for the entire basin. First, all available soil test results were obtained from the Oklahoma State University Soil, Water, and Forage Analytical Laboratory. Data from Delaware County were from January 1993 through April 1995, Cherokee County data were from February 1993 through December 1994, and Adair County data were from January 1993 through May 1995. These were identified by land use and county, but their specific locations were unknown. Next, soil testing data were obtained from the Arkansas Soil and Water Conservation during the period December 1991 through April 1995. These data were only for pasture and were identified by watershed. A summary of the soil test phosphorus data for pasture is given in Table XV. It should be noted that the assumption was made that these data were representative of soil test phosphorus levels. This assumption is untested, but was the best available.

Soil phosphorus was assigned to fields based on land use for all land uses except pasture. A summary of the assigned soil phosphorus levels is given in Table XVI. The poultry, dairy, and hog houses were assumed to be land use of rooftop, and thus had a zero soil phosphorus status. For pasture a physically-

based method for assigning initial soil phosphorus was developed. This option assigned initial soil phosphorus to pasture as a function of distance from poultry house(s) and the average soil test phosphorus by county or watershed. The rational for using distance from poultry house is that owners of the poultry houses tend to apply litter on adjacent fields to minimize transportation costs. If the litter is applied to meet the nitrogen needs for forage production, then phosphorus will be over applied and will build up in the soil profile with time. High soil test phosphorus levels have been observed in the Battle Branch and Peacheater Creek watersheds under the recent USDA Hydrologic Unit Projects in Oklahoma (Sabbagh et al., 1995).

The first step in assigning initial soil phosphorus to pasture was to determine the number of poultry houses per county or watershed. The NRCS 1985 poultry house survey was utilized. It should be noted that there has been a significant expansion of poultry operations in the basin from 1980 through 1990. However, in the absence of more recent data, the 1985 survey was used. The Phase I Clean Lakes Report for Lake Tenkiller (OWRB, 1996) indicated that the number of chickens raised in the Illinois River Basin (Figure 5) peaked first in the late 1960s, declined, and has since exceeded the previous peak. The number of broilers in the basin (Figure 5) peaked first in the mid-1960s and has experienced phenomenal growth since the early-1980s.

The NRCS survey identified 1,246 sites that had from one to 11 poultry houses summing to a total of 2,692 houses. Figure 6 identifies the locations of poultry sites in the Illinois River Basin.

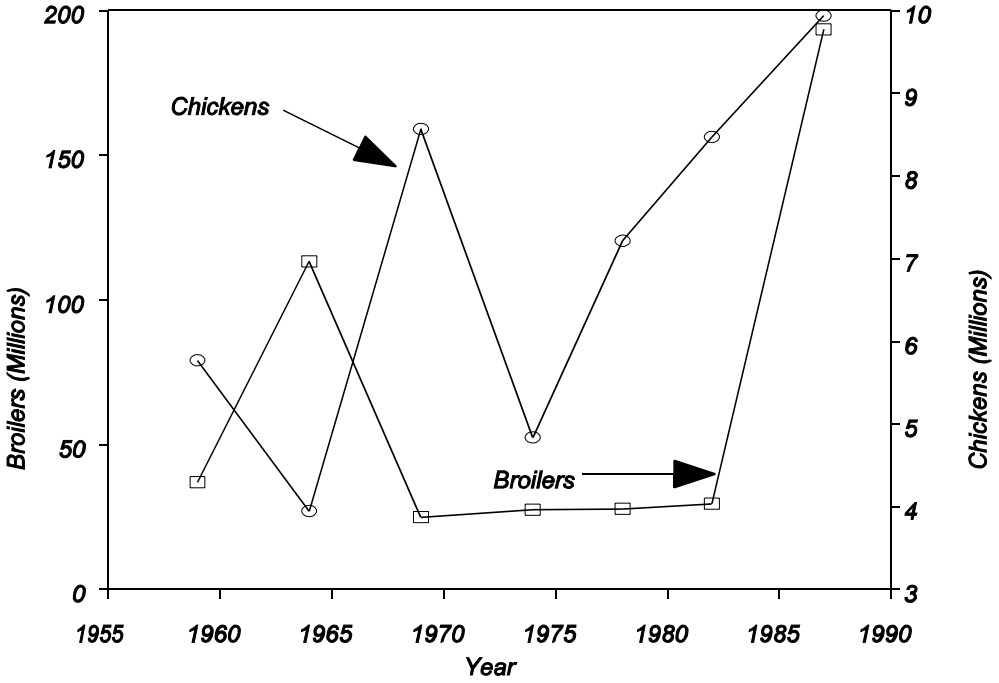

Figure 5. Number of Broilers and Three Month Old or Older Chickens in the Illinois River Basin.

The area of influence for each site was mapped using the GRASS 4.1 command *s.voronoi*, which mapped a relative area of influence for each site. Due to GRASS limitations from the large number of sites, *s.voronoi* was run for each county and watershed independently. Next, the distance from poultry house data layer was calculated for the entire basin simultaneously using the GRASS 4.1 command *r.cost*.

An average number of poultry houses per site was calculated for each county or watershed (Table XVII) and a weighing factor, W, was defined as:

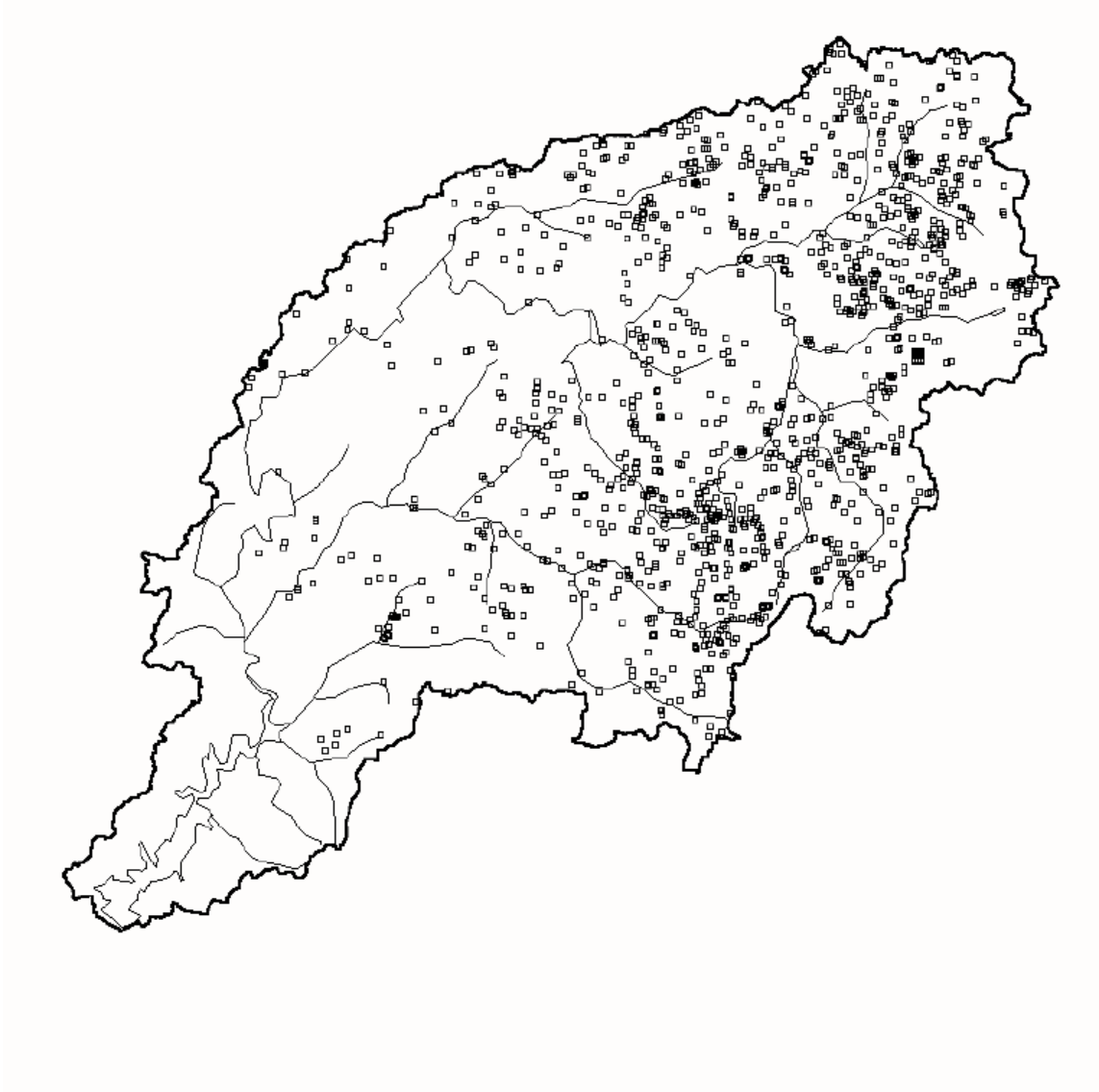

Figure 6. An Illinois River Basin Map Describing the Locations of Poultry Houses Based on 1985 Data.

$$W = \frac{\overline{P_{st}}\ H_n}{\overline{H_n}}$$

where $\overline{P_{st}}$ is the average soil test phosphorus for a county or watershed, H_n is the number of poultry houses per site, and $\overline{H_n}$ is the average number of poultry

houses per site for a county or watershed. It should be noted that there are multiple weighting factors, W, one being calculated for each H_n.

The first approximation of the initial soil phosphorus for each 30 m cell, P_{soil1}, in the county or watershed was calculated using:

$$P_{soil1} = W \frac{D_{max} \quad D_H}{D_{max}}$$

where D_{max} is the distance in meters at which the soil phosphorus level reaches the native background level, and D_H is the distance from poultry house estimated from the *r.cost* function in meters. Next, the estimated average initial soil phosphorus, $\overline{P_{soil1}}$, for the county or watershed was calculated and an adjusted initial soil test phosphorus for each 30 m cell, P_{soil2}, was calculated using:

$$P_{soil2} = \frac{P_{soil1} \overline{P_{st}}}{\overline{P_{soil1}}}$$

To keep realistic initial soil phosphorus values, P_{soil2} was bounded between 15 and 1,200 lbs/ac. After bounding the data by 15 and 1,200 lbs/ac, a new county or watershed average was calculated and the weighting function in equation was employed a second time to ensure the average observed and predicted county or watershed soil phosphorus levels agreed. This process was repeated until the predicted and observed average county or watershed soil phosphorus were within five percent.

This methodology assigns a relatively high soil test phosphorus at a poultry house location, with phosphorus levels decreasing with distance from the

poultry house. The rate at which the initial soil phosphorus decreased was governed by D_{max}. To estimate D_{max} the Peacheater Creek and Battle Branch (OK) watersheds were examined. These are smaller subbasins within the Illinois River Basin in Oklahoma which have been used for validation of the SIMPLE model. For these watersheds detailed soil testing was conducted by the Oklahoma Cooperative Extension Service as part of two USDA Hydrologic Unit Area Projects. Based on a linear regression and assuming a native soil phosphorus level of 15 lbs/ac, D_{max} is 2,500 and 1,500 meters for the Peacheater Creek and Battle Branch watersheds, respectively.

The above methodology was initially applied to the Upper Illinois basin using a D_{max} of 2,500 meters. However, there was a significant portion of the estimated soil phosphorus levels that were in excess of 1,200, and some levels exceeded 3,000 lbs/ac. By trial and error a D_{max} of 8000 meters was selected. The 8000 meter distance was selected based on visual comparison, and thus no statistical criteria were used. Using 8000 meters resulted in reasonable soil phosphorus levels compared to the observed soil test data.

Next, the methodology was applied to the entire basin. In general, the frequency distributions for the observed and predicted soil test values agreed. Figure 7 shows the initial soil phosphorus layer developed for the basin using the methods described, and used in SIMPLE.

The soil phosphorus data had units of lb P/ac. However, SIMPLE requires units of µg P/g soil. To convert lbs/ac to µg/g we assumed a dry soil bulk density of 1.5 g/cm^3 and a soil depth of 0.5 ft, thus yielding:

$$\frac{lb\,P}{ac} * \frac{kg}{2.2\,lbs} * \frac{10^9\,\mu g}{kg} * \frac{ac}{43560\,ft^2} * \frac{1}{0.5\,ft} * \frac{(0.0328)^3\,ft^3}{cm^3} * \frac{cm^3}{1.5\,g\,soil} \quad 0.49\,\frac{\mu g\,P}{g\,soil}$$

or

$$\frac{lb}{ac} = 0.49\,\frac{\mu g}{g}$$

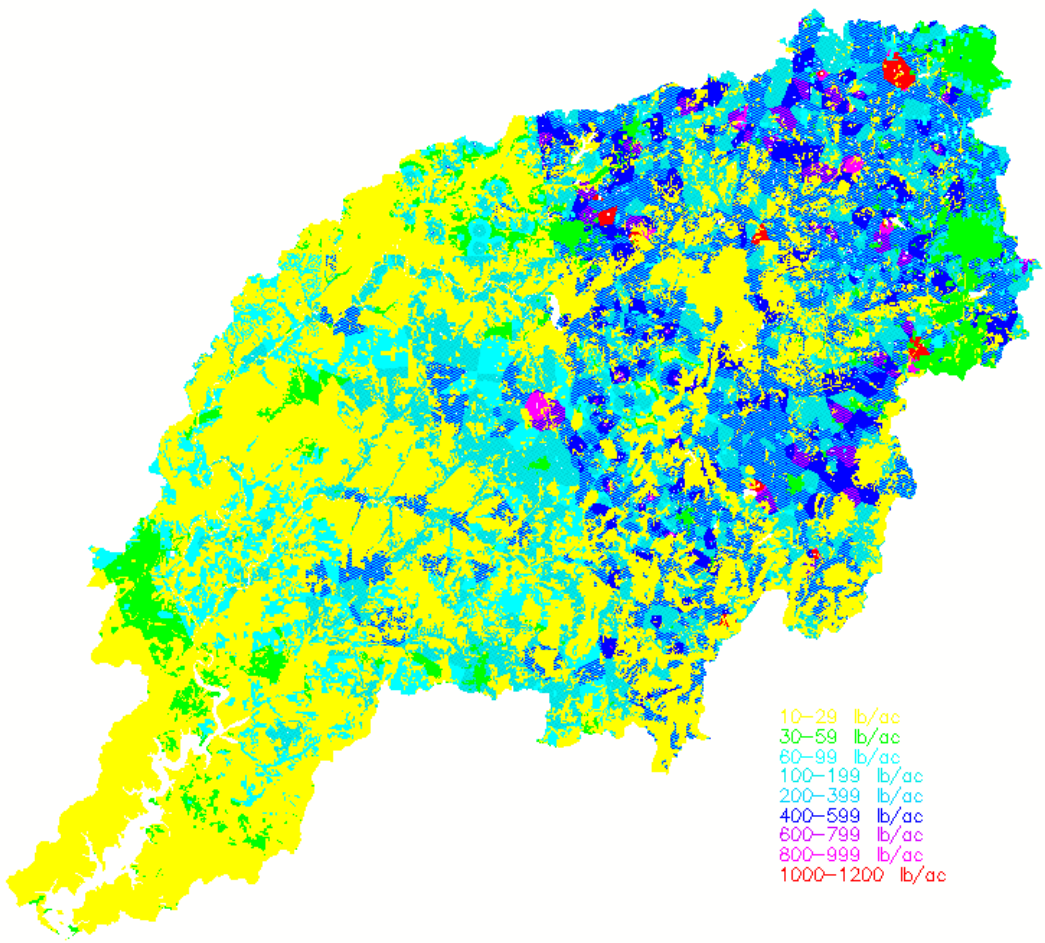

Figure 7. Estimated Initial Soil Phosphorus Levels, in lb/ac, Determined for the SIMPLE Modeling of the Illinois River Basin.

Fertilization

For the SIMPLE computer simulations, poultry litter was assumed to be applied to pasture/range land every April at a rate based on the number of poultry houses contained in the watershed. Each poultry house was assumed to hold 20,000 broilers and would produce 90,900 kilograms litter per year. This was based on 95.2 $kg/m^2/yr$ (Finley et al., 1994) and a 15.2 m by 61 m house. Next it was assumed the litter contained 1.5 percent phosphorus, and thus each house produced 1,400 kg phosphorus per year. The litter application rate to pasture for each of the watersheds is given in Table XVIII. It should be noted that commercial fertilizer, dairies, layers, pullets, and turkeys, and human water recreation impacts are neglected. However, relative to the broiler production these inputs are small.

For crop land an application of 20 kg P/ha/yr was assumed. For the remaining land uses a phosphorus application rate was selected that would keep the soil at approximately the same initial soil phosphorus level. Assumed application rates included: 0.3 kg P/ha/yr for urban areas; 0.06 kg P/ha/yr for transportation and utilities; 0.3 kg P/ha/yr for orchards, vineyards, and nurseries; and 0.03 kg P/ha/yr for forest land.

Full credit for performing the SIMPLE parameter estimations and model simulations must be given to Storm et al. (1996) since my involvement was primarily in the preparation of soils, elevation, and land use data layers.

TABLE X

TOPOGRAPHIC STATISTICS BY WATERSHED
FOR THE UPPER ILLINOIS RIVER BASIN

Watershed	Parameter	Slope (%)	Slope Length (meters)	Distance to Stream (meters)	Slope to Stream (%)
Osage	Mean	5.2	81	650	2.5
	SD	4.5	47	463	2.2
	Minimum	0	15	0	0
	Maximum	30.8	306	2,930	22.4
Clear	Mean	5.4	72	799	2.2
	SD	4.7	40	576	1.9
	Minimum	0	15	0	0
	Maximum	30	183	3,850	19.2
Fork	Mean	2.1	85	622	0.8
	SD	5.1	34	896	2.6
	Minimum	0	15	0	0
	Maximum	42	183	5,380	22
Flint	Mean	6.8	83	601	3.1
	SD	5.6	44	423	2.5
	Minimum	0	15	0	0
	Maximum	32.5	183	2,430	19.7
Baron	Mean	5.3	65	810	2.8
	SD	6.2	42	488	3.8
	Minimum	0	10	0	0
	Maximum	72	189	3,150	36
Caney	Mean	8.6	101	566	4.6
	SD	6	39	415	3.2
	Minimum	0	15	0	0
	Maximum	33	189	2,190	25.3
Benton	Mean	5.8	65	974	3
	SD	6	45	423	3.6
	Minimum	0	15	0	0
	Maximum	50	201	2,110	35.6
River	Mean	6.8	98	590	3.2
	SD	6.4	42	414	2.9
	Minimum	0	15	0	0
	Maximum	26.6	189	1,870	18.5

TABLE X

TOPOGRAPHIC STATISTICS BY WATERSHED
FOR THE UPPER ILLINOIS RIVER BASIN

Watershed	Parameter	Slope (%)	Slope Length (meters)	Distance to Stream (meters)	Slope to Stream (%)
BORD	Mean	11.3	68	546	4.1
	SD	7.6	39	413	3.7
	Minimum	0	15	0	0
	Maximum	34.7	183	1,940	52
TYNER	Mean	8.2	105	515	5.5
	SD	6.6	30	397	4.1
	Minimum	0	15	0	0
	Maximum	40.2	184	2,090	37.8
WEST	Mean	8.6	98	554	3.6
	SD	6.2	35	432	2.7
	Minimum	0	15	0	0
	Maximum	33	189	2,260	23.3
BBARON	Mean	6.9	81	590	3.9
	SD	6.1	45	496	3.6
	Minimum	0	15	0	0
	Maximum	29.2	183	3,220	30.4
BILIN	Mean	7.3	75	648	3
	SD	6.9	43	518	2.8
	Minimum	0	15	16	0
	Maximum	38.7	183	2,900	16.2
LAKEUP	Mean	6.3	97	629	1.9
	SD	5	43	523	2.1
	Minimum	0	15	15	0
	Maximum	23.6	183	2,040	10.8
LAKE	Mean	8.5	95	684	5
	SD	6	47	497	5.5
	Minimum	0	0	0	0
	Maximum	40.4	168	3,350	117

TABLE XI

SOIL CHARACTERISTICS FOR THE
UPPER ILLINOIS RIVER BASIN

Soil Number	USLE K (English)	Hydrologic Soil Group	pH	Organic Carbon (%)	Clay (%)	Bulk Density (g/cm3)	Slope Length (m)
1	0.28	B	6.10	0.44	14	1.45	122
2	0.28	B	5.25	0.44	14	1.45	61
3	0.28	B	5.25	0.44	14	1.45	61
4	0.37	C	5.25	0.44	25	1.45	152
5	0.43	B	5.00	0.74	25	1.43	152
6	0.43	B	5.00	0.74	25	1.43	152
7	0.37	B	5.00	1.18	25	1.39	189
8	0.37	B	5.00	1.18	25	1.39	152
9	0.37	B	5.00	1.18	25	1.39	152
10	0.37	B	5.40	1.18	25	1.39	122
11	0.01	B	7.00	0.01	0.01	1	15
12	0.1	C	5.80	0.74	13	1.51	152
13	0.19	C	5.00	0.85	17	1.5	152
14	0.28	B	6.70	2.65	25	1.28	15
15	0.28	B	6.70	2.65	24	1.34	15
16	0.43	C	5.80	0.01	18	1.51	189
17	0.43	C	5.50	1.47	18	1.39	152
18	0.28	B	4.55	1.03	10	1.52	152
19	0.28	B	4.55	1.03	10	1.52	122
20	0.28	B	4.55	1.03	19	1.48	122
21	0.28	B	4.55	1.03	19	1.48	122
22	0.28	D	6.20	1.47	37	1.29	15
23	0.49	D	5.80	0.44	25	1.45	183
24	0.32	D	7.25	0.01	33	1.54	152
25	0.37	C	6.45	1.18	33	1.34	183
26	0.37	C	6.45	0.1	33	1.34	152
27	0.37	C	6.45	1.18	33	1.34	122
28	0.37	C	6.45	1.18	33	1.34	122
29	0.43	D	5.00	2.06	18	1.34	15
30	0.49	C	5.55	0.44	25	1.45	183
82	0.01	D	7.00	0.01	0.01	1	152
87	0.01	D	7.00	0.01	0.01	1	152
88	0.01	D	7.00	0.01	0.01	1	152
98	0.01	B	7.00	0.01	0.01	1	152
102	0.28	B	5.25	1.74	18	1.37	152

TABLE XI

SOIL CHARACTERISTICS FOR THE
UPPER ILLINOIS RIVER BASIN

Soil Number	USLE K (English)	Hydrologic Soil Group	pH	Organic Carbon (%)	Clay (%)	Bulk Density (g/cm3)	Slope Length (m)
103	0.28	B	5.50	1.74	18	1.37	152
104	0.33	B	5.25	1.18	14	1.42	122
105	0.43	B	5.50	1.18	12	1.43	152
108	0.28	B	4.80	0.74	12	1.46	122
109	0.28	B	4.80	0.74	25	1.43	61
110	0.28	B	4.80	0.74	25	1.43	30
114	0.37	D	6.05	1.18	25	1.39	122
116	0.37	A	6.45	0.88	25	1.42	15
117	0.1	C	5.80	0.74	10	1.54	122
118	0.19	B	5.00	0.88	10	1.53	152
119	0.43	C	5.80	0.01	18	1.51	183
120	0.28	B	4.55	1.03	10	1.52	137
121	0.37	B	5.00	0.59	12	1.48	152
122	0.37	B	6.05	1.18	18	1.41	183
123	0.37	B	6.05	1.18	18	1.41	152
124	0.37	B	6.05	1.18	18	1.41	91
128	0.43	C	6.45	1.18	33	1.38	152
129	0.43	C	6.45	1.18	33	1.38	122
130	0.28	D	6.45	1.47	45	1.31	15
132	0.01	D	7.00	0.01	0.01	1	152
133	0.37	B	6.45	0.74	12	1.46	15
134	0.37	B	6.45	0.74	12	1.46	15
135	0.37	B	6.45	0.74	15	1.54	15
136	0.37	B	6.45	0.74	15	1.54	107
137	0.32	B	6.45	1.76	25	1.35	15
138	0.3	B	6.45	1.76	24	1.35	15
139	0.49	D	5.00	1.18	12	1.43	183
140	0.37	C	6.45	1.18	33	1.34	137
141	0.49	C	5.55	0.44	25	1.45	183
142	0.32	D	8.15	1.18	24	1.46	107
143	0.32	D	8.15	1.18	24	1.46	30
206	0.23	C	5.00	1	13	1.51	15
210	0.19	D	5.50	0.88	11	1.53	122
211	0.19	D	5.50	0.88	11	1.53	90
212	0.37	C	4.80	1.1	11	1.48	122

TABLE XI

SOIL CHARACTERISTICS FOR THE
UPPER ILLINOIS RIVER BASIN

Soil Number	USLE K (English)	Hydrologic Soil Group	pH	Organic Carbon (%)	Clay (%)	Bulk Density (g/cm3)	Slope Length (m)
221	0.43	B	5.00	1.03	15	1.43	152
222	0.43	B	5.00	1.03	18	1.43	122
223	0.43	B	5.00	1.03	18	1.43	122
229	0.37	B	5.00	0.1	22	1.43	15
234	0.32	D	7.25	0.01	33	1.54	15
236	0.49	D	4.75	1	15	1.44	183
238	0.49	C	5.00	1.18	12	1.43	152
241	0.37	C	6.45	1.18	33	1.34	152
320	0.28	B	5.50	1.76	18	1.38	122
321	0.28	B	5.50	1.76	18	1.38	61
322	0.28	B	5.50	1.76	18	1.38	30
323	0.28	B	5.50	1.76	18	1.38	15
335	0.37	C	5.25	0.88	15	1.43	107
336	0.37	C	5.25	0.88	15	1.43	61
345	0.43	B	5.50	1.18	8	1.45	152
346	0.43	C	5.50	1.18	12	1.43	400
348	0.43	B	5.50	1.18	8	1.45	122
349	0.43	B	5.50	1.18	8	1.45	122
352	0.43	B	6.20	0.74	18	1.47	152
356	0.28	B	4.80	0.74	25	1.44	30
357	0.28	B	4.80	0.74	25	1.44	15
374	0.37	A	6.45	0.88	8	1.51	15
381	0.28	B	6.70	1.76	25	1.36	15
401	0.37	B	6.05	1.76	14	1.38	15
402	0.2	B	7.00	0.01	8	1.27	122
404	0.37	B	6.05	1.76	14	1.38	15
409	0.43	B	5.80	0.01	6	1.53	152
410	0.43	B	5.80	XX	6	1.53	122
411	0.43	B	5.50	0.88	12	1.47	152
413	0.43	C	5.50	0.88	12	1.47	152
414	0.32	C	4.55	1.18	18	1.43	183
415	0.37	C	4.55	1.18	18	1.43	152
423	0.49	C	6.20	1.18	35	1.34	152
442	0.33	B	6.05	1.18	18	1.43	152
443	0.32	B	5.00	1.18	18	1.43	107

TABLE XI

SOIL CHARACTERISTICS FOR THE
UPPER ILLINOIS RIVER BASIN

Soil Number	USLE K (English)	Hydrologic Soil Group	pH	Organic Carbon (%)	Clay (%)	Bulk Density (g/cm3)	Slope Length (m)
444	0.32	B	5.00	1.18	18	1.43	61
445	0.43	C	5.00	1.18	18	1.43	107
453	0.28	B	5.50	1.18	18	1.43	61
454	0.28	B	5.50	1.18	18	1.43	30
455	0.28	B	5.50	1.16	18	1.43	15
464	0.32	B	5.90	1.76	25	1.36	152
465	0.32	B	5.25	1.76	25	1.36	122
466	0.32	B	5.25	1.74	18	1.41	107
467	0.37	B	5.25	1.16	12	1.45	152
469	0.37	B	5.25	1.18	12	1.45	107
471	0.43	B	5.25	1.03	18	1.44	152
472	0.43	B	5.00	1.03	18	1.44	107
473	0.43	B	5.00	1.03	18	1.44	107
474	1.43	B	5.00	1.03	18	1.44	61
489	0.37	B	6.70	1.18	18	1.49	15
493	0.37	B	6.70	1.18	18	1.43	15
494	0.37	B	6.70	1.18	18	1.43	15
497	0.01	D	1.00	0.01	0.01	2.65	152
501	0.32	B	5.80	1.18	17	1.41	15
506	0.37	B	6.95	2.65	25	1.29	15
507	0.32	C	7.25	0.01	25	1.51	152
515	0.37	D	6.45	1.18	42	1.31	183
516	0.37	D	6.45	1.18	42	1.31	152
517	0.37	D	6.45	1.18	42	1.31	107
518	0.37	D	6.45	1.18	42	1.31	76
519	0.37	D	6.45	1.18	42	1.31	30
520	0.37	D	6.45	1.76	33	1.29	107
521	0.37	D	6.45	1.76	33	1.29	61
522	0.37	D	6.45	1.47	37	1.3	152
523	0.49	C	5.55	0.44	12	1.47	183
524	0.49	D	5.55	0.44	12	1.47	152
525	0.49	C	5.55	0.44	12	1.47	152
526	0.37	B	5.00	0.01	13	1.53	107
533	0.28	A	5.80	0.74	8	1.48	107
534	0.28	A	5.80	0.74	8	1.48	61

TABLE XI

SOIL CHARACTERISTICS FOR THE
UPPER ILLINOIS RIVER BASIN

Soil Number	USLE K (English)	Hydrologic Soil Group	pH	Organic Carbon (%)	Clay (%)	Bulk Density (g/cm3)	Slope Length (m)
601	0.24	A	4.55	1.47	19	1.46	107
602	0.24	A	4.55	1.47	19	1.46	91
603	0.24	A	4.55	1.47	19	1.46	61
604	0.24	A	4.55	1.47	19	1.46	61
605	0.24	A	4.55	1.47	19	1.46	15
611	0.28	A	5.00	1.03	12	1.5	107
612	0.28	A	5.00	1.03	12	1.5	61
613	0.28	A	5.00	1.03	12	1.5	31
614	0.28	A	5.00	1.03	19	1.5	15
615	0.28	A	5.00	1.03	19	1.5	61
622	0.19	A	5.50	0.88	15	1.53	15
627	0.37	A	5.00	1.18	19	1.49	152
628	0.37	A	5.00	1.18	19	1.49	91
629	0.37	A	5.00	1.18	19	1.49	91
630	0.2	C	6.05	0.59	13	1.55	350
638	0.43	C	5.90	1.18	13	1.43	152
639	0.43	C	5.90	1.18	13	1.43	152
640	0.32	B	6.45	1.18	16	1.51	15
645	0.37	A	6.45	0.88	12	1.47	15
646	0.37	A	6.45	0.88	19	1.51	15
655	0.32	C	4.55	1.18	19	1.49	91
656	0.32	C	4.55	1.18	19	1.49	91
657	0.32	C	4.55	1.18	19	1.49	91
658	0.32	C	4.55	1.88	19	1.49	61
659	0.32	C	4.55	1.18	19	1.49	61
662	0.32	C	4.55	1.76	16	1.47	91
664	0.32	C	4.55	1.76	16	1.47	30
668	0.28	C	4.55	1.61	19	1.45	61
669	0.28	C	4.55	1.61	19	1.45	61
684	0.24	B	6.05	1.18	16	1.51	91
685	0.24	B	6.05	1.18	16	1.51	61
686	0.24	B	6.10	1.18	16	1.51	31
687	0.24	B	6.10	1.18	16	1.51	15
688	0.17	B	5.90	1	13	1.53	10
689	0.15	C	5.50	0.88	10	1.55	152

TABLE XI

SOIL CHARACTERISTICS FOR THE
UPPER ILLINOIS RIVER BASIN

Soil Number	USLE K (English)	Hydrologic Soil Group	pH	Organic Carbon (%)	Clay (%)	Bulk Density (g/cm3)	Slope Length (m)
690	0.15	C	5.50	0.88	10	1.55	61
691	0.15	C	5.50	0.88	11	1.55	61
708	0.28	B	4.55	1.03	4	1.52	107
712	0.33	B	4.55	1.03	19	1.5	152
714	0.28	B	4.55	1.03	19	1.5	91
716	0.28	B	4.55	1.03	13	1.51	91
717	0.28	B	4.55	1.03	13	1.51	61
724	0.28	C	5.25	0.74	18	1.47	91
725	0.2	B	5.25	1.18	6	1.54	30
726	0.2	B	5.25	1.18	6	1.54	91
727	0.2	B	5.25	1.18	6	1.54	15
791	0.49	C	6.05	1.76	25	1.36	152
794	0.49	C	6.05	1.76	25	1.36	152
795	0.37	B	4.55	1.03	10	1.54	152
796	0.37	B	4.55	1.03	10	1.54	91
834	0.32	C	4.55	1.76	16	1.47	15
852	0.25	C	4.90	1.1	13	1.53	30
882	0.28	B	4.55	1.03	4	1.52	15
917	0.17	D	5.25	1.18	10	1.53	90
931	0.2	B	5.00	1.03	16	1.53	120
938	0.26	B	4.75	1.1	16	1.52	30
939	0.26	B	4.75	1.1	16	1.52	15
999	0.01	D	7.00	0.01	0	1	152

TABLE XII

ILLINOIS RIVER BASIN SOILS DATABASE

County	Soil Number	Soil Name/Classification	Slope Range (%)	Area (ha)	Watershed Coverage (%)
Adair, OK	1	Bodine very cherty silt loam	1-8	21,753	5.17
	2	Bodine stony silt loam	5-15	5,279	1.25
	3	Bodine stony silt loam	steep	30,284	7.2
	4	Craig cherty silt loam	1-5	417	0.1
	5	Dickson silt loam	1-3	5,339	1.27
	6	Dickson cherty silt loam	0-3	8,370	1.99
	7	Etowah silt loam	0-1	601	0.14
	8	Etowah silt loam	1-3	2,215	0.53
	9	Etowah gravelly silt loam	1-3	4,038	0.96
	10	Etowah and Greendale soils	3-8	6,376	1.52
	11	Gravelly alluvial land	---	3,245	0.77
	12	Hector complex	---	6,397	1.52
	13	Hector-Linker fine sandy loams	1-5	1,815	0.43
	14	Huntington silt loam	---	400	0.1
	15	Huntington gravelly loam	---	993	0.24
	16	Jay silt loam	0-2	1,258	0.3
	17	Lawrence silt loam	---	231	0.05
	18	Linker fine sandy loam	1-5	556	0.13
	19	Linker fine sandy loam	3-5	109	0.03
	20	Linker loam	3-5	473	0.11
	21	Linker loam	3-5	117	0.03
	22	sage clay loam	---	178	0.04
	23	Parsons silt loam	0-1	203	0.05
	24	Sogn soils	---	562	0.13
	25	Summit silty clay loam	0-1	254	0.06
	26	Summit silty clay loam	1-3	379	0.09
	27	Summit silty clay loam	3-5	163	0.04
	28	Summit silty clay loam	3-5	63	0.02
	29	Taft silt loam	---	600	0.14
	30	Taloka silt loam	0-1	81	0.02
	82	Borrow Pits	---	30	0.01
	83	Gravel Pits	---	34	0.01
	87	Pits Quarries	---	6	0
	88	Quarries	---	36	0.01
	98	water	---	5,730	1.36

TABLE XII

ILLINOIS RIVER BASIN SOILS DATABASE

County	Soil Number	Soil Name/Classification	Slope Range (%)	Area (ha)	Watershed Coverage (%)
	102	Baxter silt loam	1-3	1,069	0.25
Cherokee & Delaware, OK	103	Baxter cherty silt loam	1-3	1,070	0.25
	104	Baxter-Locust complex	3-5	1,317	0.31
	105	Captina silt loam	1-3	2,504	0.60
	108	Clarksville very cherty silt loam	1-8	10,941	2.60
	109	Clarksville stony silt loam	5-20	6,575	1.56
	110	Clarksville stony silt loam	20-50	30,516	7.25
	111	Collinsville fine sandy loam	2-5	14	0.00
	114	Eldorado silt loam	3-5	625	0.15
	115	Eldorado soils	3-12	267	0.06
	116	Elsah soils	---	4,451	1.06
	117	Hector fine sandy loam	2-5	2,072	0.49
	118	Hector-Linker association hilly	---	12,681	3.01
	119	Jay silt loam	0-2	611	0.15
	120	Linker fine sandy loam	2-5	664	0.16
	121	Locust cherty silt loam	1-3	3,539	0.84
	122	Newtonia silt loam	0-1	58	0.01
	123	Newtonia silt loam	1-3	827	0.20
	124	Newtonia silt loam	3-5	338	0.08
	125	Newtonia silt loam	2-5	100	0.02
	127	Okemah silty clay loam	0-1	366	0.09
	128	Okemah silty clay loam	1-3	708	0.17
	129	Okemah silty clay loam	3-5	162	0.04
	130	Osage clay	---	377	0.09
	132	Rough stony land	---	2,698	0.64
	133	Sallisaw silt loam	0-1	383	0.09
	134	Sallisaw silt loam	1-3	1,549	0.37
	135	Sallisaw gravelly silt loam	1-3	2,149	0.51
	136	Sallisaw gravelly silt loam	3-8	5,125	1.22
	137	Staser silt loam	---	1,106	0.26
	138	Staser gravelly loam	---	2,748	0.65
	139	Stigler silt loam	0-1	925	0.22
	140	Summit silty clay loam	2-5	317	0.08
	141	Taloka silt loam	0-1	323	0.08
	142	Talpa-Rock outcrop complex	2-8	1,294	0.31
	143	Talpa-Rock outcrop complex	15-50	4,771	1.13
Sequoyah, OK	203	Cleora fine sandy loam	---	21	0.01

TABLE XII

ILLINOIS RIVER BASIN SOILS DATABASE

County	Soil Number	Soil Name/Classification	Slope Range (%)	Area (ha)	Watershed Coverage (%)
	206	Hector-Linker-Enders complex	5-40	7,110	1.69
	210	Linker-Hector complex	2-5	1,118	0.27
	211	Linker-Hector complex	5-8	64	0.02
Sequoyah, OK	212	Linker and Stigler soils	2-8	50	0.01
	216	Mason silt loam	---	269	0.06
	221	Pickwick loam	1-3	307	0.07
	222	Pickwick loam	3-5	414	0.10
	223	Pickwick loam	2-5	56	0.01
	224	Razort fine sandy loam	---	62	0.01
	227	Rosebloom silt loam	---	21	0.01
	229	Rosebloom and Ennis soils broken	---	325	0.08
	230	Sallisaw complex	8-30	14	0.00
	231	Sallisaw loam	1-3	24	0.01
	232	Sallisaw loam	3-5	59	0.01
	233	Sallisaw loam	2-5	34	0.01
	234	Sogn complex	10-25	483	0.11
	236	Stigler-Wrightsville silt loams	0-1	104	0.02
	238	Stigler silt loam	1-3	414	0.10
	239	Stigler silt loam	2-5	7	0.00
	241	Summit silty clay loam	1-3	56	0.01
	242	Summit silty clay loam	3-5	140	0.03
Washington & Benton, AR	320	Baxter cherty silt loam	3-8	118	0.03
	321	Baxter cherty silt loam	8-12	298	0.07
	322	Baxter cherty silt loam	12-20	240	0.06
	323	Baxter cherty silt loam	20-45	1,914	0.45
	335	Britwater gravelly silt loam	3-8	1,320	0.31
	336	Britwater gravelly silt loam	8-12	13	0.00
	345	Captina silt loam	1-3	17,124	4.07
	348	Captina silt loam	3-6	1,534	0.36
	349	Captina silt loam	3-6	5,587	1.33
	352	Craytown silt loam		204	0.05
	356	Clarksville cherty silt loam	12-50	11,213	2.67
	357	Clarksville cherty silt loam	12-60	10,874	2.58
	374	Elsah soils		1,988	0.47
	381	Fatima silt loam occasionally flooded		559	0.13
Washington & Benton, AR	401	Guin cherty silt loam	3-8	1,143	0.27
	402	Healing silt loam		473	0.11
	404	Healing silt loam occasionally flooded		1,949	0.46
	409	Jay silt loam	1-3	4,212	1.00

TABLE XII

ILLINOIS RIVER BASIN SOILS DATABASE

County	Soil Number	Soil Name/Classification	Slope Range (%)	Area (ha)	Watershed Coverage (%)
	410	Jay silt loam	3-8	951	0.23
	411	Johnsburg silt loam		3,553	0.84
	413	Johnsburg complex mounded		260	0.06
	414	Leaf silt loam		1,163	0.28
Washington & Benton, AR	415	Leaf complex mounded		573	0.14
	423	Mayes silty clay loam		267	0.06
	442	Newtonia silt loam	1-3	374	0.09
	443	Nixa cherty silt loam	3-8	22,615	5.38
	444	Nixa cherty silt loam	8-12	5,729	1.36
	445	Nixa very cherty silt loam	3-8	3	0.00
	453	Noark very cherty silt loam	8-12	370	0.09
	454	Noark very cherty silt loam	12-20	990	0.24
	455	Noark very cherty silt loam	20-45	1,524	0.36
	464	Pembroke silt loam	1-3	762	0.18
	465	Pembroke silt loam	3-6	1,065	0.25
	466	Pembroke gravelly silt loam	3-8	613	0.15
	467	Peridge silt loam	1-3	2,013	0.48
	469	Peridge silt loam	3-8	1,646	0.39
	471	Pickwick silt loam	1-3	844	0.20
	472	Pickwick silt loam	3-8	5,529	1.31
	473	Pickwick gravelly loam	3-8	150	0.04
	474	Pickwick gravelly loam	8-12	68	0.02
	489	Razort loam		679	0.16
	493	Razort silt loam occasionally flooded		1,726	0.41
	494	Razort gravelly silt loam occasionally flooded		2,182	0.52
	497	Rock land		191	0.05
	501	Secesh gravelly silt loam occasionally flooded		4,506	1.07
	506	Sloan silt loam		1,962	0.47
	507	Sogn rocky silt loam		573	0.14
	515	Summit silty clay	0-1	1,647	0.39
	516	Summit silty clay	1-3	325	0.08
	517	Summit silty clay	3-8	416	0.10
	518	Summit silty clay	3-15	21	0.01
	519	Summit silty clay	8-12	77	0.02
	520	Summit stony silty clay	3-12	335	0.08
	521	Summit stony silty clay	12-25	45	0.01
	522	Summit complex mounded		92	0.02
	523	Taloka silt loam	0-1	3,651	0.87
	524	Taloka silt loam	1-3	697	0.17

TABLE XII

ILLINOIS RIVER BASIN SOILS DATABASE

County	Soil Number	Soil Name/Classification	Slope Range (%)	Area (ha)	Watershed Coverage (%)
	525	Taloka complex mounded		531	0.13
	526	Tonti cherty silt loam	3-8	7,977	1.90
	533	Waben very cherty silt loam	3-8	781	0.19
	534	Waben very cherty silt loam	8-12	62	0.01
	601	Allegheny gravelly loam	3-8	138	0.03
Washington & Benton, AR	602	Allegheny gravelly loam	3-8	201	0.05
	603	3Allegheny gravelly loam	8-12	87	0.02
	604	Allegheny stony loam	8-12	235	0.06
	605	Allegheny stony loam	12-40	272	0.06
	611	Allen loam	3-8	238	0.06
	612	Allen loam	8-12	220	0.05
	613	Allen loam	12-20	127	0.03
	614	Allen stony loam	12-35	132	0.03
	615	Allen soils	8-20	36	0.01
	622	Allen-Hector complex	20-40	167	0.04
	627	Apison loam	1-3	113	0.03
	628	Apison loam	3-8	1,125	0.27
	629	Apison gravelly loam	3-8	203	0.05
	630	Cane loam	3-8	135	0.03
	638	Cherokee silt loam		2,031	0.48
	639	Cherokee complex mounded		244	0.06
	640	Cleora fine sandy loam		1,893	0.45
	645	Elsah gravelly soils		1,244	0.30
	646	Elsah cobbly soils		890	0.21
	655	Enders gravelly loam	3-8	106	0.03
	656	Enders gravelly loam	3-8	640	0.15
	657	Enders gravelly loam	3-12	398	0.09
	658	Enders gravelly loam	8-12	242	0.06
	659	Enders gravelly loam	8-12	204	0.05
	662	Enders stony loam	3-12	2,531	0.60
	664	Enders stony loam	12-30	132	0.03
	668	Enders-Allegheny Complex	8-20	8,062	1.92
	669	Enders-Allegheny Complex	20-40	10,162	2.42
	684	Fayetteville fine sandy loam	3-8	1,814	0.43
	685	Fayetteville fine sandy loam	8-12	471	0.11
	686	Fayetteville fine sandy loam	12-20	178	0.04
	687	Fayetteville stony fine sandy loam	12-35	340	0.08
	688	Fayetteville-Hector complex	20-40	782	0.19
	689	Hector-Mountainburg gravelly fine sandy loams	3-8	1,136	0.27

TABLE XII

ILLINOIS RIVER BASIN SOILS DATABASE

County	Soil Number	Soil Name/Classification	Slope Range (%)	Area (ha)	Watershed Coverage (%)
	690	Hector-Mountainburg gravelly fine sandy loams	8-12	285	0.07
	691	Hector-Mountainburg stony fine sandy loams	3-40	6,533	1.55
	708	Linker fine sandy loam	3-8	877	0.21
	712	Linker loam	1-3	284	0.07
	714	Linker loam	3-8	2,950	0.70
	716	Linker gravelly loam	3-8	851	0.20
Washington & Benton, AR	717	Linker gravelly loam	8-12	47	0.01
	724	Montevallo soils	3-12	308	0.07
	725	Montevallo soils	12-25	37	0.01
	726	Mountainburg stony sandy loam	3-12	29	0.01
	727	Mountainburg stony sandy loam	12-40	16	0.00
	791	Samba silt loam		63	0.15
	794	Samba complex mounded		118	0.03
	795	Savannah fine sandy loam	1-3	656	0.16
	796	Savannah fine sandy loam	3-8	3,893	0.93
Crawford, AR	834	Enders stony fine sandy loam	12-45	46	0.01
	852	Enders-Mountainburg Association rolling		70	0.02
	882	Linker fine sandy loam	3-8	22	0.01
	917	Mountainburg stony fine sandy loam	3-12	3	0.00
	931	Nella gravelly fine sandy loam	3-8	7	0.00
	938	Nella-Enders Association rolling		68	0.02
	939	Nella-Enders Association steep		204	0.05
	999	Water		490	0.12

TABLE XIII

USLE C FACTORS USED IN THE SIMPLE MODEL OF THE ILLINOIS RIVER BASIN

Land Use	Julian Day	USLE C Factor
Urban	---	0.003
Transportation, Communications, Utilities	---	0.003
Crop	1	0.4
	70	0.31
	90	0.24
	120	0.13
	150	0.1
	180	0.08
	210	0.08
	211	0.4
	300	0.2
	365	0.4
Pasture/Range	---	0.003
Orchards, Groves, Vineyards	---	0.3
Nurseries	---	0.3
Forest	---	0.003
Poultry Operations	---	0
Dairy	---	0
Hog Operations	---	0
Water	---	0

TABLE XIV

HYDROLOGIC SOILS GROUP AND CURVE NUMBER
FOR LAND USES IN THE ILLINOIS RIVER BASIN

Hydrologic Soil Group	Land Use Number	Land Use	Curve Number
A	1	Urban	71
B			78
C			84
D			86
A	2	Transportation	72
B			82
C			87
D			89
A	3	Crop	63
B			75
C			83
D			87
A	4	Pasture/Range	49
B			69
C			79
D			84
A	5	Orchards	41
B			55
C			69
D			71
A	6	Nurseries	69
B			75
C			82
D			86
A	7	Forest	36
B			60
C			73
D			79
A	8	Poultry Operations	100
B			100
C			100
D			100
A	9	Dairy	100
B			100
C			100
D			100
A	10	Hog Operations	100
B			100
C			100
D			100
A	11	Water	100
B			100
C			100
D			100

TABLE XV

OBSERVED SOIL TEST PHOSPHORUS STATISTICS FOR PASTURE IN THE UPPER ILLINOIS RIVER BASIN FROM 1992 TO 1995

County or Watershed Number	State	Number of Samples	Mean (lb/ac)	Median (lb/ac)	Standard Deviation (lb/ac)	Minimum (lb/ac)	Maximum (lb/ac)
Delaware	OK	370	93	56	80	7	520
Adair	OK	214	159	64	188	9	1,220
Cherokee	OK	109	52	41	35	9	167
Sequoyah	OK	0	-	-	-	-	-
10	AR	25	341	226	194	77	717
20	AR	37	297	203	231	45	999
30	AR	167	301	245	194	45	999
40	AR	25	239	127	233	54	883
50	AR	3	295[1]	-	-	-	-
60	AR	26	358	337	176	53	785
70	AR	54	227	161	194	31	999
80	AR	27	261	254	148	17	656
81	AR	0	242[2]	-	-	-	-

[1]Approximated as the average of watersheds 030, 060 and 070.

[2]Approximated as the average of watersheds 040, 070 and 080.

TABLE XVI

INITIAL SOIL TEST PHOSPHORUS BY LAND USE
FOR THE UPPER ILLINOIS RIVER BASIN

Land Use	Soil Test Phosphorus (lb/ac)	Area (ha)	Area (%)
Urban	60	14,500	3.5
Transportation, Communication, and Utilities	15	1,230	0.3
Crop	60	4,140	1
Pasture and Range	Variable[1]	212,000	49
Orchards, Groves, Vineyards	60	1,430	0.3
Nurseries	60	148	0.03
Forest	10	186,000	44
Poultry, Dairy, and Hog Houses	0	1,650	0.4
Water	0	6,910	1.6

[1]Defined as a function of distance from poultry house

TABLE XVII

POULTRY HOUSE AND AREA STATISTICS FOR THE
UPPER ILLINOIS RIVER BASIN FOR 1985

County or Watershed Number	State	Houses	Sites	Houses Per Site	Area (ha)
Delaware	OK	64	34	1.88	20,070
Adair	OK	313	158	1.98	102,960
Cherokee	OK	73	34	2.15	109,300
Sequoyah	OK	0	0	0	?
10	AR	214	102	2.1	24,230
20	AR	227	105	2.16	20,440
30	AR	751	306	2.45	58,430
40	AR	268	126	2.13	18,840
50	AR	95	37	2.57	16,030
60	AR	200	91	2.2	17,140
70	AR	111	49	2.27	12,390
80	AR	260	143	1.82	21,910
81	AR	141	61	2.31	5,710

TABLE XVIII

NUMBER OF POULTRY HOUSES, PASTURE APPLIED PHOSPHORUS AND PASTURE AREA BY WATERSHED

Watershed Number	Watershed Name	Number of Poultry Houses	Pasture Applied Litter (kg/ha)	Pasture Area (ha)
1	OSAGE	739	1,800	38,200
2	CLEAR	219	1,790	11,400
3	FORK	462	1,700	25,400
4	FLINT	280	1,350	19,400
5	BARON	412	2,030	19,000
6	CANEY	48	374	12,000
7	BENTON	286	1,180	22,700
8	RIVER	17	280	5,670
9	BORD	40	376	10,200
10	TYNER	17	294	5,400
11	WEST	143	958	14,900
12	BBARON	24	179	5,080
13	BILIN	5	124	3,780
14	LAKEUP	0	100	3,670
15	LAKE	0	100	5,760

TABLE XIX

WATERSHED NUMBERING CONVENTION WITH
WEATHER STATION AND WATERSHED AREA

Watershed Number	Watershed Name	Weather Station	Watershed Area (ha)
1	OSAGE	Bentonville	57,400
2	CLEAR	Fayetteville	20,900
3	FORK	Fayetteville	41,500
4	FLINT	Kansas	32,100
5	BARON	Odell	39,200
6	CANEY	Stilwell	31,600
7	BENTON	Siloam Springs	37,600
8	RIVER	Kansas	13,000
9	BORD	Kansas	33,000
10	TYNER	Kansas	10,900
11	WEST	Stilwell	30,500
12	BBARON	Tahlequah	13,000
13	BILIN	Tahlequah	10,200
14	LAKEUP	Tahlequah	5,380
15	LAKE	Webbers Fall	34,100

QUAL2EU

The QUAL2EU model (Brown and Barnwell, 1987) is a steady-state model for conventional pollutants in branching streams and well mixed lakes which permits simulation of up to 15 water quality constituents using a finite difference solution to the one-dimensional advective-dispersion mass transport and reaction equation. A sequence of stream reaches describing the stream of interest is divided into a number of subreaches or computational elements. For each computational element, a hydrologic balance of flow, a heat balance of temperature, and a materials balance of concentration is written. Mass can be gained or lost from the elements by transport processes, external sources and sinks, or by internal sources and sinks. The mass transport equation is solved for the steady flow, steady state condition in a classical implicit backward difference method (Brown and Barnwell, 1987). An enhancement to the model allows the user to perform uncertainty analysis. Sensitivity analysis, first-order error analysis, and Monte Carlo simulations are the three uncertainty analysis options which allow the user to assess the effect of model sensitivities and of uncertain input data on model forecasts.

The basic equation solved by QUAL2EU is a one-dimensional advection dispersion mass transport equation, which is numerically integrated over space and time for each water quality constituent of interest. For any constituent, C, this equation can be written as:

$$\frac{M}{t} = \frac{\left[A_x D_L \frac{C}{x}\right]}{x} dx \quad \frac{\left[A_x \bar{u} C\right]}{x} dx \quad (A_x dx) \frac{dC}{dt} \quad s$$

where

M = mass (M)

x = distance (L)

t = time (T)

C = concentration (ML^{-3})

A_x = cross-sectional area (L^2)

D_L = dispersion coefficient (L^2T^{-1})

$\bar{u}$ = mean velocity (LT^{-1})

s = external source or sink (MT^{-1})

Hydraulic characteristics in QUAL2EU assume that the stream hydraulic regime is steady state, and thus, the hydrologic balance can be written as:

$$\left(\frac{Q}{x}\right)_i = (Q_x)_i$$

where $(Q_x)_i$ is the sum of external inflows to the ith element (L^3T^{-1}). Using discharge coefficients, other hydraulic characteristics of the stream segments can be determined by equations of the form:

$$\bar{\ } = aQ^b$$

$$A_x = \frac{Q}{\bar{\ }}$$

and

$$d = \alpha Q^{\beta}$$

where a, b, α, and , are empirical constants, and d is the stream depth (L). These constants were developed from stage discharge rating curves.

Chlorophyll *a* is considered to be directly proportional to the concentration of phytoplanktonic algal biomass. Algal biomass is converted to chlorophyll *a* by the simple relationship:

$$Chl\ a = \alpha_o A$$

where

$Chl\ a$ = chlorophyll *a* concentration (µg/l),

A = algal biomass concentration (mg/l), and

α_o = a conversion factor (µg/mg).

The differential equation that governs the growth and production of algae is formulated as follows:

$$\frac{dA}{dt} = \mu A \quad \beta A \quad \frac{{}^{\beta}1}{d} A$$

where

A = algal biomass concentration (mg/l),

t = time (day),

μ = the specific growth rate of algae (day^{-1}),

ρ = the respiration rate of algae (day^{-1}),

${}^{\sigma}1$ = the settling rate for algae (ft/day), and

d = the average depth (ft).

The nitrogen cycle covers the step-wise transformation from organic nitrogen to ammonia, to nitrite, to nitrate. The differential equations governing these transformations are as shown below.

Organic Nitrogen:

$$\frac{dN_4}{dt} = \alpha_1 \beta A \quad \beta_3 N_4 \quad \beta_4 N_4$$

Ammonia Nitrogen:

$$\frac{dN_1}{dt} = \beta_3 N_4 \quad \beta_1 N_1 \quad \frac{\beta_3}{d} \quad F_1 \alpha_1 \mu A$$

Nitrite Nitrogen:

$$\frac{dN_2}{dt} = \beta_1 N_1 \quad \beta_2 N_2$$

Nitrate Nitrogen:

$$\frac{dN_3}{dt} = \beta_2 N_2 \quad (1 \quad F_1) \alpha_1 \mu A$$

$$F_1 = \frac{P_N N_1}{P_N N_1 \quad (1 \quad P_N) N_3}$$

where

A = algal biomass concentration (mg/l),

N_1 = the concentration of ammonia nitrogen (mg/l),

N_2 = the concentration of nitrite nitrogen (mg/l),

N_3 = the concentration of nitrate nitrogen (mg/l),

N_4 = the concentration of organic nitrogen (mg/l),

$_1$ = rate constant for biological oxidation of ammonia nitrogen (day^{-1}),

$_2$ = rate constant for oxidation of nitrite nitrogen (day^{-1}),

$_3$ = organic nitrogen hydrolysis rate (day^{-1}),

α_1 = fraction of algal biomass which is nitrogen (mg/mg),

$_3$ = the benthos source rate for ammonia nitrogen ($mg/ft^2/day$),

$_4$ = rate coefficient for organic nitrogen settling (day^{-1}),

= algal respiration rate (day^{-1}),

d = the mean depth of flow (ft),

F_1 = fraction of algal nitrogen uptake from ammonia pool,

μ = the specific growth rate of algae (day^{-1}), and

P_N = preference factor for ammonia nitrogen (0 to 1.0).

The phosphorus cycle operates much like the nitrogen cycle. Organic forms of phosphorus are generated by the death of algae which then convert to the dissolved inorganic state, where it is available to algae for primary production. The differential equations governing transformations of phosphorus from one form to the other follow the equations below:

Organic Phosphorus:

$$\frac{dP_1}{dt} = \alpha_2 \beta A \quad \beta_4 P_1 \quad \beta_5 P_1$$

Dissolved Phosphorus:

$$\frac{dP_2}{dt} = \beta_4 P_1 \quad \frac{\beta_2}{d} \quad \alpha_2 \mu A$$

where

P_1	=	the concentration of organic phosphorus (mg/l),
P_2	=	the concentration of inorganic or dissolved phosphorus (mg/l),
$_2$	=	the benthos source rate for dissolved phosphorus (mg/ft^2/day),
$_5$	=	organic phosphorus settling rate (day^{-1}),
	=	algal respiration rate (day^{-1}),
$_4$	=	organic phosphorus decay rate (day^{-1}),
μ	=	algal growth rate (day^{-1}),
A	=	algal biomass concentration (mg/l), and
d	=	mean stream depth (ft).

QUAL2EU Data Input

A model of the Illinois River was developed consisting of four headwaters.

USGS gaging stations were selected as headwater points because the historical data available at each of them would allow for the development of reasonably accurate flow conditions at a known point. USGS 07194800, on the Illinois River near Savoy, AR, was selected as the uppermost point on the mainstem of the Illinois River and is approximately 140 km above the Horseshoe Bend area of Lake Tenkiller. Major tributaries with USGS gages were selected as the remaining headwaters. USGS 07195000 is located on Osage Creek about 16 km above its confluence with the Illinois River. USGS 07196000 is located on Flint Creek near Kansas, OK about 5 km above its confluence with the Illinois River. And, finally, USGS 07196900 is located on Baron Fork Creek at Dutch Mills, AR about 59 km above its confluence with the Illinois River. Figure 8 shows these headwater locations, reach designations, and SIMPLE subbasins included in the model.

Distances from headwaters to confluences were estimated based on information downloaded from the STORET database for each gage, and were confirmed using USGS 1:100,000 scale hydrography DLGs. The DLGs were imported into GRASS as vector files and a measuring feature allowed for estimating stream segment lengths and the points of confluence. SIMPLE generated subbasins of the Illinois River Basin above lake Tenkiller were overlain on the vector map of streams to determine the stream segments that each of the subbasins would influence. Constrained by the number of elements QUAL2EU allows for each modeled stream reach, a model of 17 stream reaches was developed with USGS gaging stations and SIMPLE subbasin borders as

endpoints for most reaches. Selecting an element length of 1 km resulted in having a total of 220 elements which allowed for fairly accurate placement of confluences and point sources. Table XX describes the reaches, their endpoints, and lengths. Inputs modeled as point sources were located as close as possible to their actual points given the resolution of the model developed and are indicated in Table XXI below.

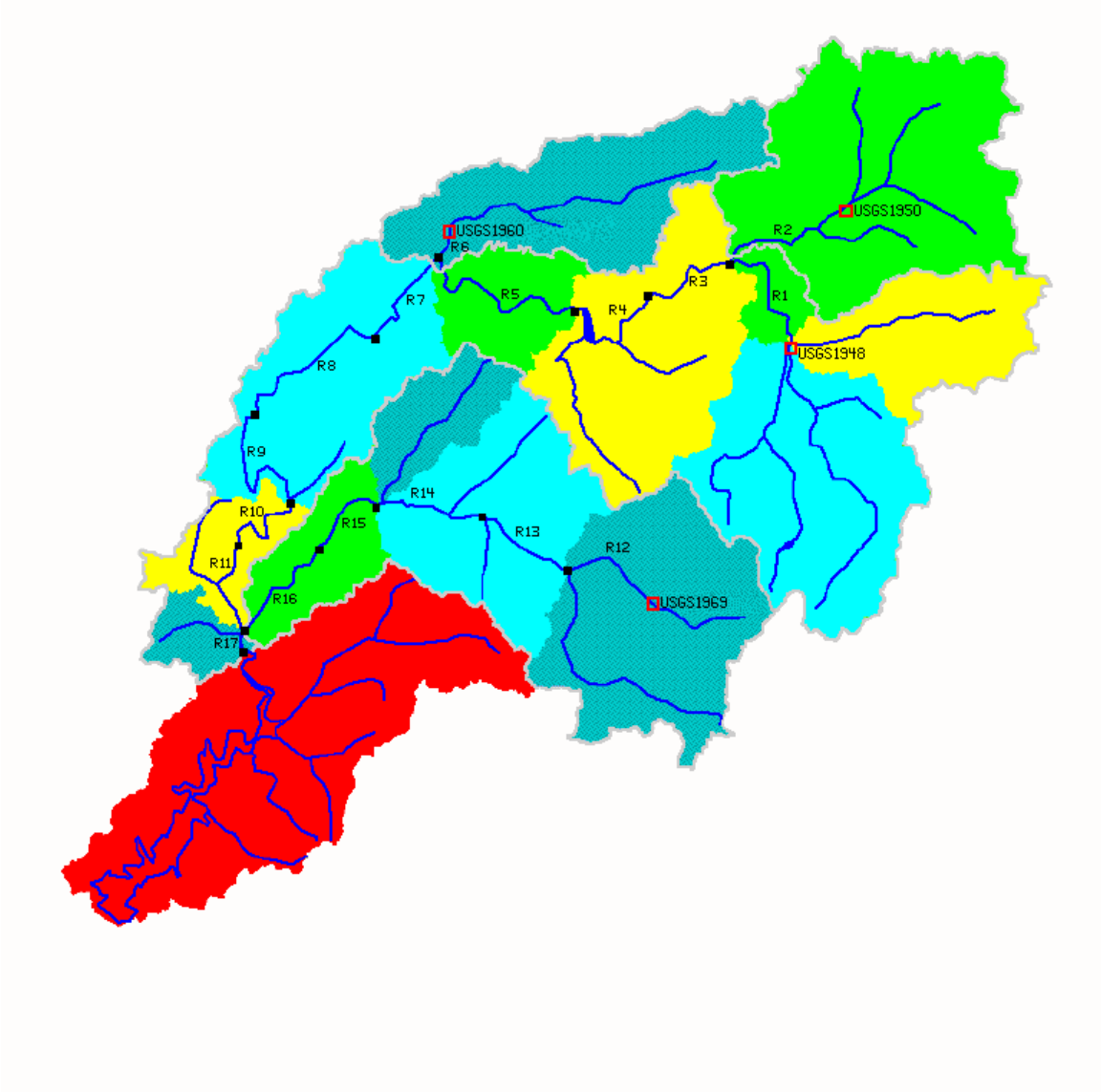

Figure 8. A Map of the Illinois River Basin Indicating the River and Tributaries (blue), Locations of the QUAL2EU Headwaters (red boxes), Reach Designations and Reach Endpoints (black squares), and SIMPLE Subbasins Included in the Model (all except red).

TABLE XX

REACHES DEVELOPED FOR THE QUAL2EU MODEL
OF THE ILLINOIS RIVER BASIN

Reach Number	Description	Number of 1 km Elements	Location in km above Horseshoe Bend or Confluence	Stream
1	GS48 to Osage Creek Confluence	18	140 to 122	Illinois River
2	GS50 to Illinois River Confluence	16	16 to 0	Osage Creek
3	Osage Creek confluence to GS54	10	122 to 112	Illinois River
4	GS54 to GS55	15	112 to 97	Illinois River
5	GS55 to Flint Creek Confluence	20	97 to 77	Illinois River
6	GS60 to Illinois River Confluence	5	5 to 0	Flint Creek
7	Flint Creek Confluence to SR3	11	77 to 66	Illinois River
8	SR3 to SR4.5	20	66 to 46	Illinois River
9	SR4.5 to Southern Border of BORD	19	46 to 27	Illinois River
10	Northern Border of BILIN to GS65	11	27 to 16	Illinois River
11	GS65 to Baron Fork Creek Confluence	12	16 to 4	Illinois River
12	GS69 to Western Border of BARON	12	59 to 47	Baron Fork Creek
13	Easter Border of WEST to 12 km Downstream	12	47 to 35	Baron Fork Creek
14	Endpoint of Reach 13 to Western Border of WEST	12	35 to 23	Baron Fork Creek
15	Eastern Border of BBARON to GS70	9	23 to 14	Baron Fork Creek
16	GS70 to Illinois River Confluence	14	14 to 0	Baron Fork Creek
17	Baron Fork Creek Confluence to Horseshoe Bend	4	4 to 0	Illinois River

The SIMPLE generated subbasins comprising the Illinois River Basin did not precisely correspond to the watersheds of the developed QUAL2EU model. The tributary headwaters at USGS 07195000, 07196000, and 07196900 fell within subbasins in their respective areas such that, in order to estimate headwater inputs at each, further division of individual subbasin SIMPLE output

was necessary. This was also the case for stream reaches other than those at the headwaters where the nonpoint source inputs of a particular subbasin would have to be divided between several stream reaches. The GRASS command *r.watershed* was used to determine subbasin areal frcctions which would influence specific stream reaches. SIMPLE subbasin output was then divided based on the percent area of the total subbasin to be directed to a specific stream reach. The results of this exercise are shown in Table XXI. SIMPLE nonpoint source subbasin inputs are indicated by capital letters with the fraction of that subbasin's output indicated in parentheses. Point source inputs are also indicated.

TABLE XXI

ILLINOIS RIVER BASIN QUAL2EU MODEL POINT AND NONPOINT SOURCE INPUTS TO REACHES

Reach Number	Headwater Inputs Point and Nonpoint Source	Point Source Inputs	Nonpoint Source Inputs
1	Prairie Grove, FORK (100%)	Fayetteville, CLEAR (100%)	OSAGE (8%)
2	Springdale, Rogers, OSAGE (57%)		OSAGE (35%)
3			BENTON (24%)
4			BENTON (76%)
5			RIVER (100%)
6	Gentry, Siloam Springs, FLINT (84%)		FLINT (16%)
7			BORD (14%)
8			BORD (54%)
9		Midwestern Nursery	BORD (32%)
10			BILIN (34%)
11		Tahlequah	BILIN (66%)
12	Lincoln, BARON (28%)		BARON (72%)
13			WEST (52%)
14		Westville, TYNER (100%)	WEST (48%)
15			BBARON (29%)
16			BBARON (71%)
17		Cherokee Nation	LAKEUP (100%)

Nutrient and discharge inputs at the four headwater elements were calculated by mass balancing fractions of subbasin SIMPLE outputs above the headwater location with any point sources entering the system above the headwater. The results of this mass balancing are given in Table XXII. Also indicated in Table XXII are estimates of base flow. Base flow discharge was calculated by differencing. SIMPLE runoff volumes were compared to average annual mean discharge measured at the headwater locations. Because the SIMPLE model predicts surface runoff, and predicted runoff will not exactly match observed runoff, the volumes generated by SIMPLE were less than the average annual discharge measured at the headwater gaging stations. Additionally, the SIMPLE model simulations were run on observed rainfall data from 1962 through 1986, while estimates of average annual discharge at the gaging stations were calculated from data for WTRs 1980 through 1986. SIMPLE runoff volumes converted into discharge rates were subtracted from annual average mean discharge calculated from USGS discharge data and this difference was assumed to be stream base flow. The nutrient concentrations in the base flow at the headwaters were assumed to be at background levels of 0.02 mg/l total phosphorus and 0.5 mg/l total nitrogen (Omernick, 1977).

Discharge coefficients describing hydraulic characteristics of stream reaches were developed using velocity data, and stage-discharge rating tables available from the USGS ADAPS database for each of the gaging stations. Coefficients for velocity and depth were developed by regression. Because the coefficients developed for the eight gaging stations did not describe hydraulic

characteristics for all reaches in the model, coefficients were estimated for several reaches based on coefficients of gaging stations above and below the reach. Data and estimates concerning the development of coefficients for velocity and depth are presented in Appendix A.

SIMPLE subbasin nonpoint loading to the stream reaches at points other than headwaters were distributed evenly along the stream reaches. The fraction of a subbasin's nutrient load influencing a particular stream reach was divided evenly among the elements of the reach.

Figure 9 is a schematic diagram of the QUAL2EU model developed for the Illinois River above Lake Tenkiller. The SIMPLE defined subbasins appear in bold and the numbers in parentheses indicate the percentage of that basins nutrient load entering the stream reach. Distances indicated next to USGS gaging station location imply distance to the confluence or the Horseshoe Bend area of Lake Tenkiller.

In order to answer questions about the sources of nutrients arriving at the Horseshoe Bend area of Lake Tenkiller, several different input data sets were created. The basic input data set was an estimate of average annual conditions constructed from average annual nutrient and discharge inputs. A second data set was created to simulate the average annual conditions without the influence of municipal point source inputs. This was created by reducing municipal point source nutrient inputs to zero, recalculating mass balances, and entering these data into the input file. A third data set was created which would simulate average annual background concentration inputs only. This data input file had

municipal point source nutrients reduced to zero, and nonpoint source inputs from the SIMPLE subbasins were reduced to background concentration levels.

Given these three input data sets it was possible to make estimates of the sources of nutrients at any given point in the model. Municipal point source contributions were assumed to be the difference between the average annual output and the output of the data set with municipal point sources reduced to zero. This method ignores potential nonlinearities in the modeled system but can provide reasonable estimates of point source contributions at any particular point in the system. Hence, at any given point in the modeled system, estimates could be made of the contributions of point, nonpoint, and background sources to total nutrient concentration and load.

In addition to the three input data sets described above, two other input data sets were created to simulate the effects of reducing nonpoint source nutrient loading by 25% and 50%. These were created by diminishing the SIMPLE subbasin nutrient output by those percentages and recalculating inputs at the headwaters, and recalculating nonpoint inputs along each reach.

SIMPLE output for the 15 defined subbasins within the Illinois River Basin were summarized on an annual basis. SIMPLE provided 25 years (1962-1986) of simulations given variable rainfall based on data from weather stations located within the basin.

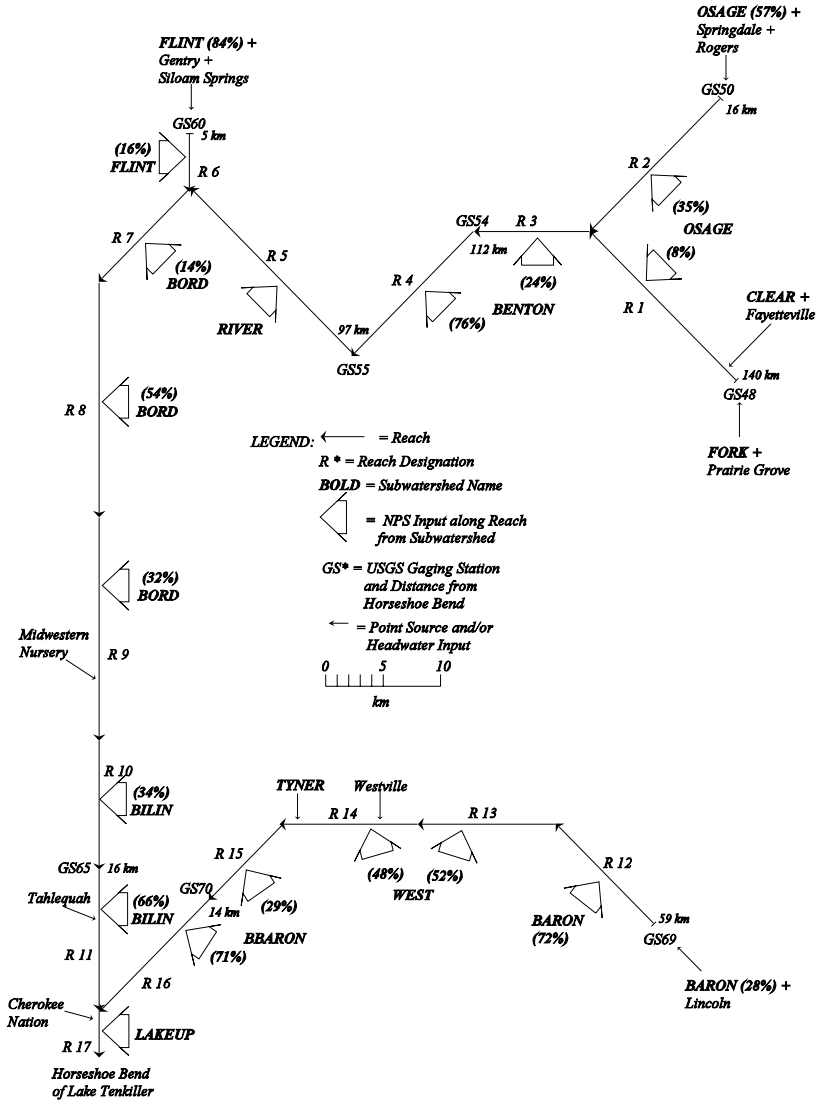

Figure 9. A Schematic Diagram of the QUAL2EU Model Developed for the Illinois River and Major Tributaries Showing Stream Reaches, Nonpoint Source Nutrient Loading Inputs from Defined Subbasins, Point Source Input Locations, and USGS Gaging Station Locations (Not to Scale).

To convert the SIMPLE output into a form appropriate for input into the QUAL2EU model some data manipulation was performed. Total annual runoff volume in cubic meters was calculated by converting runoff to meters, subbasin area to square meters, and obtaining the product, and this value was then divided by seconds/year to obtain the desired units of cubic meters per second for input of nonpoint source discharge volume into the QUAL2EU data sets. Concentration data were also extracted from the SIMPLE output in a similar manner. The total flux of phosphorus from a subbasin on an annual basis (kg/yr) was divided by the total annual runoff volume (m^3/yr) and converted into units of mg/l for input as the concentration of nonpoint source discharge from the subbasin. These calculations were performed for all 15 subbasins comprising the Illinois River Basin above Lake Tenkiller.

Nitrogen flux from each of the subbasins was calculated using unit-area loading methods based on the areas of defined land uses within the basin. Literature values for nitrogen loading based on land use in units of kg N/ha/yr were obtained from several sources.

TABLE XXII

FLOW AND NUTRIENT INPUTS AT THE HEADWATER LOCATIONS OF THE ANNUAL AVERAGE QUAL2EU MODEL OF THE ILLINOIS RIVER BASIN

Headwater	Source Name	Distance to Input (km)	Flow (cms)	Total Phosphorus (mg/l)	Total Nitrogen (mg/l)
Inputs @ GS48 (R1)	FORK	0	1.46	0.74	5.5
	Base Flow	0	1.60	0.02	0.5
	Prairie Grove	14.5	0.02	2.23	0.5
	Mass Balanced Input	0	3.07	0.37	2.9
Inputs @ GS50 (R2)	OSAGE above GS50	0	1.00	0.78	10.8
	Base Flow	0	5.09	0.02	0.5
	Rogers	14.5	0.18	3.05	0.3
	Springdale	13	0.35	3.14	1.2
	Mass Balanced Input	0	6.61	0.38	2.1
Inputs @ GS60 (R6)	FLINT above GS60	0	1.00	0.65	7.2
	Base Flow	0	2.51	0.02	0.5
	Gentry	16	0.02	3.03	8.1
	Siloam Springs	14.5	0.15	1.79	1.0
	Mass Balanced Input	0	3.68	0.28	2.4
Inputs @ GS69 (R12)	BARON above GS69	0	0.43	0.58	5.4
	Base Flow	0	0.74	0.02	0.5
	Lincoln	8	0.03	1.46	3.9
	Mass Balanced Input	0	1.19	0.25	2.4

Unlike the Beaver Lake Report (FTN Associates, Ltd., 1992), the export coefficient for pasture land use was held constant across the entire watershed. The land use category of confined animal was used to calibrate the nitrogen unit-area loading results with monitoring station water quality data based on data from USGS gaging stations 07196500 and 07197000. This nitrogen export estimation method was used to provide simple estimates of total nitrogen export from each of the subbasins compromising the Illinois River Basin. Confined

animal operations were heavily weighted to insure that wastes produced in any particular subbasin would retain that subbasin as the source. It was assumed that wastes produced at these locations are susceptible to transport due to rainfall events and erosion.

Export coefficients were chosen which were comparable to those used in the Beaver Lake Clean Lakes Report based on the assumption that land uses and soil types in the two basins are relatively similar. Weighting the confined animal operations more heavily, at 750 kg N/ha/yr, results in showing manure produced at these sites as a highly significant contributor to nitrogen transported to Lake Tenkiller, despite the limited area these operations occupy in the basin. In fact, animal manure and litter is applied to pasture and rangeland to promote forage growth, and this land use likely contributes the most significant fraction of runoff transported nitrogen. The method selected will give essentially the same exported quantity, but attribute its source erroneously.

The nitrogen unit-area loading results compared favorably to the results of the calculated average annual nonpoint source loading quantities calculated at the Horseshoe Bend area of Lake Tenkiller given in OWRB (1996). The seven major land uses defined for the watershed and the total nitrogen loading factors selected (in units of kg/ha/yr) were crop (5), confined animal (750), forest (2), pasture/range (2.4), roads and right-of-ways (2.5), urban (3), and water (0). Areas of each subbasin in each of the land use types were multiplied by the unit area loading factors and summed for each subbasin to obtain an annual nitrogen loading in kg/yr. This value was also converted into concentration units by

dividing it by the total annual runoff volume and adjusting to units of mg/l as appropriate for input into the QUAL2EU model.

The QUAL2EU model required phosphorus input in two fractions, dissolved and organic. USGS water quality monitoring data at the eight USGS gaging stations within the basin were examined to determine the average distribution of total phosphorus between these fractions. The data from SIMPLE was then adjusted into these two fractions based on basin wide in-stream averages of the distribution. Averaging 1,148 total phosphorus observations, 709 orthophosphate observations, and 81 organic phosphorus observations at the eight gaging stations, 69% of the in-stream total phosphorus was, on average, in the dissolved form and the remaining 31% in the organic form. Similarly, for total nitrogen, basin-wide average distributions of nitrogen forms were determined from data available at the eight USGS gaging stations. The nitrogen forms necessary for input into QUAL2EU, and their average percentage of total nitrogen, were organic (16%), ammonia (3.8%), and nitrate + nitrite (80.2%).

Point source loading information was obtained from the PCS database for dischargers in the Illinois River Basin. These data were incorporated into the QUAL2EU model as point sources unless the discharge was above the headwater locations. When discharges were above headwaters, the point source data were mass balanced with base flow and nonpoint source concentration and discharge, and input as headwater conditions. For simplicity, point sources were modeled as being static with regard to nutrient concentration

and discharge. The first-order kinetic assimilation rates calculated above were used to estimate the average annual quantity of nutrient loads from municipal point source dischargers in the Illinois River Basin arriving at the point where they were input into the QUAL2EU model. Point source discharges were distributed between low, medium, and high flows based on average frequency percent of flows in these strata at USGS gaging stations within the Illinois River Basin calculated as 58.1% low flow, 31.5 % medium flow, and 10.4% high flow, and reduced at rates corresponding to those calculated for each flow stratum.

Table XXIII lists municipalities and SIMPLE watersheds that were input into the QUAL2EU model as point sources along with estimates of average annual discharge, average annual concentration at the outfall, distance from the point source outfall to the input point in the model, and the estimated concentration at the input point. SIMPLE output from two subbasins, CLEAR and TYNER, were entered into the QUAL2EU model as point sources, but were treated as nonpoint sources in estimating sources of nutrient concentration and load contributions.

The model's nutrient, chlorophyll *a,* and discharge parameters were calibrated using average annual nutrient and discharge data measured at USGS gaging stations from water years 1980 - 1993. This resulted in only minor deviations from the defaults in the QUAL2EU model. Many parameters selected were suggested values provided by the user manual (Brown and Barnwell, 1987) and other sources. Appendix B contains results of the calibration of the QUAL2EU model developed for the Illinois River Basin above Lake Tenkiller.

The QUAL2EU model offers uncertainty analysis in the form of Monte Carlo simulations which were employed to estimate the frequency distributions of nutrient concentrations and loads at specific locations on the Illinois River and its tributaries, as well as the upper portion of Lake Tenkiller, based on variable nutrient concentrations and stream discharge.

An input variance file was created describing the coefficient of variation and distribution of point and nonpoint nutrient inputs, and discharge inputs. During the 2000 simulations, performed for each of the input data sets, input parameters were randomly selected from the distribution information provided.

TABLE XXIII

POINT SOURCE INPUTS TO THE QUAL2EU MODEL AFFECTING NUTRIENT LOADS AT THE HORSESHOE BEND AREA OF LAKE TENKILLER

Discharges	Flow (cms)	Total Phosphorus (mg/l)	Total Nitrogen (mg/l)	Distance to Input (km)	Input Conc. Total Phosphorus (mg/l)	Input Conc. Total Nitrogen (mg/l)	Location
Prairie Grove	0.015	2.93	0.6	14.5	2.23	0.5	Reach 1 Element 1
Fayetteville	0.300	0.41	0.4	18.5	0.29	0.3	Reach 1 Element 2
CLEAR	0.656	0.93	6.8	0	0.93	6.8	Reach 1 Element 2
Rogers	0.175	4.00	0.4	14.5	3.05	0.3	Reach 2 Element 1
Springdale	0.350	4.00	1.5	13	3.14	1.2	Reach 2 Element 1
Gentry	0.015	4.10	10.0	16	3.03	8.1	Reach 5 Element 1
Siloam Springs	0.150	2.35	1.2	14.5	1.79	1.0	Reach 5 Element 1
Midwest Nursery	0.020	1.17	9.9	0	1.17	9.9	Reach 9 Element 14
Tahlequah	0.150	2.70	3.4	6.5	2.39	3.1	Reach 11 Element 5
Lincoln	0.025	1.70	4.3	8	1.46	3.9	Reach 12 Element 1
Westville	0.025	4.29	10.0	6.5	3.80	9.2	Reach 14 Element 4
TYNER	0.308	0.33	3.2	0	0.33	3.2	Reach 14 Element 11
Cherokee Nation	0.005	3.15	8.9	0	3.15	8.9	Reach 17 Element 2

CHAPTER V

RESULTS

Autosampler Results

Autosampler data were collected at two sites in the Illinois River Basin. Four samplers were placed in the field but only those at USGS 07196500 and 07197000 provided results for three and two runoff events, respectively.

The samplers were placed in the field in July, 1993 at all four locations. A runoff event on August 1993 washed out and destroyed the sampler near USGS 07195500. Only one small part of the sampler was recovered and thus no data were recorded for that site. The sampler located near USGS 07196000 malfunctioned numerous times by drawing samples before the water reached the actuator, and again no reliable data were collected at this site. The sampler at USGS 07197000 successfully collected samples for two runoff events before being flooded out in November 1993. The sampler at USGS 07196500 successfully collected samples for three runoff events before being withdrawn from the field in December 1993.

The time and date recorded by the Automatic samplers were used to determine stream discharge, via the USGS ADAPS database, corresponding to

sample collection times. The results emphasize the fact that nonpoint transport of phosphorus and nitrogen occurs primarily during runoff events. Concentrations of both nutrients tend to increase with increasing stream discharge, and then taper off through the duration of the event. Figures 10 and 11 show the results of sampling at USGS 07196500 in September 1993. Total nitrogen concentration (Figure 10) appears to decrease with the initial increase in discharge as a dilution effect takes place and then there is an abrupt increase in concentration which abruptly tapers off to below the initial measured concentration.

Total phosphorus concentration (Figure 11) reacts in a somewhat different manner throughout the runoff event without the apparent dilution at the initial increase in discharge and a much more gradual decline to the initial measured concentration. The transport of phosphorus occurs in sediment bound and dissolved forms. The transport of phosphorus associated with soil particles and organic matter constitutes a major proportion of phosphorus transported from cultivated land. Runoff from grass and forested land generally carries less sediment and is dominated by dissolved forms (Sharpley, 1997). Appendix C includes several more graphics depicting the concentration of total phosphorus and total nitrogen during two other runoff events at USGS 07196500 and two runoff events at USGS 07197000 during the Fall of 1993. Also included in Appendix C are tables with the sampling times, concentration, and discharge data.

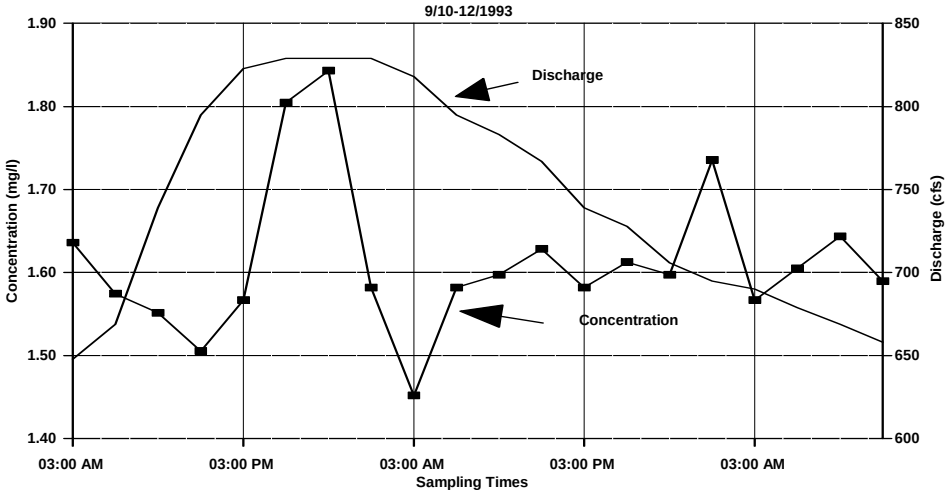

Figure 10. Measured Total Nitrogen Concentration in mg/l, and Discharge in cfs at USGS 07196500 September 10 - 12, 1993.

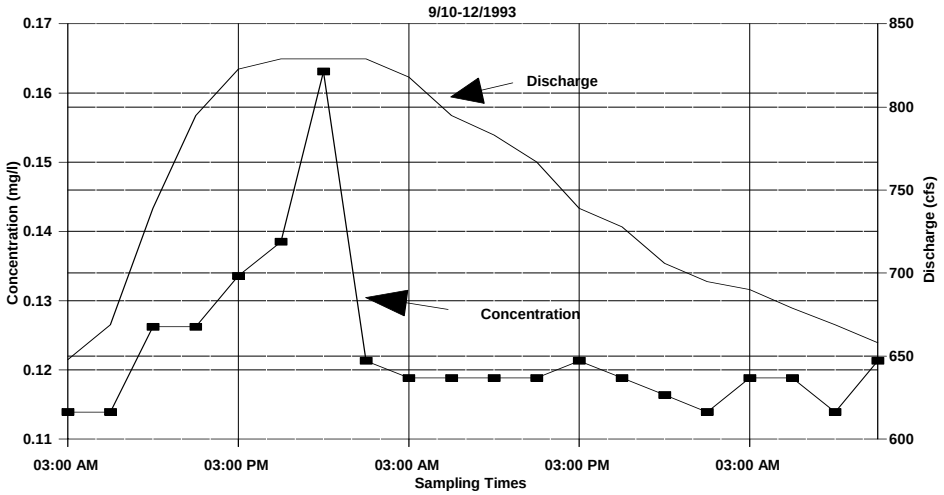

Figure 11. Measured Total Phosphorus Concentration in mg/l, and Discharge in cfs at USGS 07196500 September 10 - 12, 1993.

Historical Discharge and Water Quality

Water quality and discharge data were downloaded from U.S. EPA's STORET database for USGS gaging stations in the Illinois River Basin. Mean daily discharge data were downloaded from the USGS ADAPS database for USGS gaging stations in the basin. Available water quality data from all federal and state agencies for water years 1980 through 1993 were used to statistically describe total phosphorus and nitrogen concentrations. Instantaneous discharge from the STORET database and USGS ADAPS discharge data were used to calculate total phosphorus and nitrogen loads. When instantaneous discharge data were not recorded with concentration observations, the mean daily discharge recorded for that date was used for loading calculations.

Table XXIV contains selected descriptive statistics for average annual discharge at each of the eight gaging stations within the Illinois River Basin. Discharge records were complete at three gaging stations (07195500, 07196500, and 07197000), and adequate for the remainder with the possible exception of 07195000.

Tables XXV and XXVI contain selected descriptive statistics for average annual total phosphorus and nitrogen concentrations, respectively, at each of the eight gaging stations. Highest mean concentrations for total phosphorus were found at USGS 07195000, 07195400, and 07195500. The mean annual concentration of 0.96 mg/l total phosphorus in Osage Creek at 07195000, in the northeastern portion of the watershed, presumed to be heavily impacted by point source inputs from Rogers and Springdale WWTPs, seems to impact Illinois

River mainstem USGS gage concentrations for some distance downstream. Similarly for total nitrogen, the highest annual mean concentration is at 07195000 (4.6 mg/l), and this appears to affect concentrations on the Illinois River at least up to 07195500 at the Oklahoma/Arkansas border.

TABLE XXIV

STATISTICAL DESCRIPTIONS OF AVERAGE ANNUAL DISCHARGE AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980 - 1993

USGS Gaging Station Identification	Mean (cms)	Standard Deviation (cms)	Minimum (cms)	25th Percentile (cms)	50th Percentile (cms)	75th Percentile (cms)	Maximum (cms)	N
07194800	3.3	10.9	0.1	2.7	2.8	3.2	213	1,213
07195400	10.8	31.2	1.3	11.4	11.5	12.0	719	1,188
07195500	18.9	40.3	0.7	6.3	13.8	31.3	977	5,114
07196500	29.4	59.4	1.1	9.8	24.7	48.3	1,320	5,114
07195000	3.9	3.1	1.0	2.1	3.2	4.8	22.7	79
07196000	3.5	8.0	0.2	0.9	2.2	5.0	214	4,383
07196900	1.2	4.0	0.0	0.5	1.0	2.2	114	4,384
07197000	11.0	27.8	0.1	4.4	14.0	23.3	971	5,114

Tables XXVII and XXVIII contain selected descriptive statistics for total phosphorus and total nitrogen loads at the eight selected gaging stations. The highest mean total phosphorus load (272,000 kg/yr) was observed at USGS 07195500 near Watts, OK. Further down the mainstem at USGS 07196500, near Tahlequah, the mean total phosphorus load is reduced by about 31,000 kg/yr. It is assumed that this decrease is the result of a combination of factors. The data analysis may account for part of the difference based on sampling errors of observed concentrations and discharges at the two points. Also, in-stream settling and biological uptake may account for some of the decrease. A

third factor is the presence of Lake Frances just above the gage near Watts. Documented substantial seasonal algal blooms (Threlkeld, 1983) contribute to elevated nutrient concentrations for some distance below the dam (Burks et al., 1991), and high flow events probably flush significant quantities of nutrient laden sediments out of the lake which remain in the water column for some distance downstream. Mean total nitrogen loads are observed to be greatest at USGS 07196500, 07195400, and 07195500.

Table XXIX gives some additional information with regard to the frequencies of flows within each stratum, the percent total volume of water delivered by each flow stratum, and estimates of the percent total phosphorus and nitrogen load delivered by each stratum based on available samples. Averaging over all eight stations, over 57% of the daily flows are less than one-half of the mean, and over 89% are less than two times the mean flow. Flows categorized as low and medium combined transport about 46% and 39% of the average annual mass of the nutrients nitrogen and phosphorus, respectively. High flows are the most infrequent, just over 10% of daily flows, but deliver the greatest mass of nitrogen and phosphorus, about 54% and 61%, respectively. These results are not unexpected. High flows, related to rainfall runoff events, carry dissolved and suspended surface materials and eroded soil to the stream in great quantities. Table XXIX shows that the frequency of water quality samples taken in each of the flow strata were representative of the relative frequency of long-term flows in each stratum.

The Cox and Stuart trend test (Conover, 1980), a nonparametric test, was

applied to individual concentration and load observations for each nutrient to test for the presence of increasing trends over the period 1980 to 1993. This fairly simple trend test compares data from the first half of the period of record to that in the second half of the period of record. The null hypothesis tested in each case suggested that early concentrations or loads were equal to or greater than later concentrations or loads. The results of these tests, shown in Table XXX, indicate statistically significant increasing total phosphorus concentrations only at 07194800 (Illinois River near Savoy, AR), and 07197000 (Baron Fork Creek near Eldon, OK) over the period analyzed. Highly significant increasing total phosphorus load trends were indicated at 07194800, 07196500 (Illinois River near Tahlequah, OK), 07196000 (Flint Creek near Kansas, OK), and 07197000. Highly significant increasing total nitrogen concentration trends were indicated at 07194800, 07195500 (Illinois River near Watts, OK), 07196500, and 07196000. Highly significant increasing total nitrogen load trends were indicated at 07194800, 07195500, 07196500, 07196000, and 07197000.

TABLE XXV

STATISTICAL DESCRIPTIONS OF TOTAL PHOSPHORUS CONCENTRATIONS AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980 - 1993

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)	N
07194800	0.12	0.17	0.03	0.06	0.09	0.12	1.80	163
07195000	0.96	0.76	0.10	0.50	0.82	1.25	7.50	153
07195400	0.30	0.14	0.10	0.20	0.26	0.35	0.85	136
07195500	0.30	0.30	0.00	0.19	0.25	0.33	3.42	140
07196000	0.20	0.12	0.00	0.13	0.18	0.24	0.85	138
07196500	0.15	0.13	0.01	0.08	0.12	0.18	1.24	141
07196900	0.15	0.17	0.03	0.08	0.11	0.16	1.60	147
07197000	0.10	0.15	0.00	0.01	0.06	0.14	1.42	130

TABLE XXVI

STATISTICAL DESCRIPTIONS OF TOTAL NITROGEN CONCENTRATIONS AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980 - 1993

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)	N
07194800	2.0	1.0	0.1	1.3	1.9	2.5	7.3	162
07195000	4.6	1.6	0.2	3.6	4.8	5.6	10.8	151
07195400	3.0	3.3	0.9	2.2	2.7	3.2	39.9	132
07195500	2.0	1.2	0.0	1.0	2.1	2.7	5.2	103
07196000	1.7	0.9	0.2	0.9	1.5	2.4	4.9	109
07196500	1.3	0.9	0.0	0.5	1.2	2.0	4.1	101
07196900	2.2	1.6	0.0	0.8	2.2	3.2	6.8	147
07197000	1.1	0.8	0.2	0.5	1.0	1.3	4.2	99

TABLE XXVII

STATISTICAL SUMMARIES OF HISTORICAL TOTAL PHOSPHORUS LOADS AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980-1993

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)	N
07194800	17,200	52,800	107	826	3,660	10,600	477,000	128
07195000	72,300	45,300	16,600	46,600	63,000	83,800	341,000	72
07195400	122,000	169,000	15,000	50,200	72,700	104,000	1,170,000	98
07195500	272,000	1,190,000	290	33,700	59,900	117,000	9,800,000	136
07196000	48,200	342,000	241	4,470	8,670	17,200	3,820,000	124
07196500	241,000	792,000	286	21,600	50,900	144,000	5,750,000	141
07196900	3,950	9,760	27	429	1,610	3,260	84,200	140
07197000	76,800	360,000	17	496	6,310	21,800	3,220,000	130

TABLE XXVIII

STATISTICAL SUMMARIES OF HISTORICAL TOTAL NITROGEN LOADS AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980-1993

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)	N
07194800	289,000	484,000	111	22,000	106,000	326,000	3,050,000	129
07195000	581,000	507,000	111,000	283,000	462,000	722,000	3,470,000	66
07195400	1,530,000	2,190,000	79,500	378,000	880,000	1,760,000	17,500,000	100
07195500	1,280,000	3,500,000	14,200	141,000	430,000	966,000	32,000,000	100
07196000	237,000	1,040,000	5,120	17,400	54,100	140,000	10,100,000	97
07196500	1,590,000	3,120,000	20,000	87,400	444,000	1,290,000	15,100,000	99
07196900	82,400	160,000	13	2,650	30,700	88,200	1,140,000	137
07197000	417,000	1,090,000	1,030	15,200	78,300	250,000	7,610,000	99

TABLE XXIX

DESCRIPTIONS OF FLOW FREQUENCY, FLOW VOLUME, SAMPLING FREQUENCY AND FLUX, BY FLOW STRATA AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980-1993

USGS Gaging Station Identification	Flow Stratum	Stratum Frequency (%)	Volume (%)	Total Phosphorus Sampling Frequency (%)	Total Phosphorus Flux (%)	Total Nitrogen Sampling Frequency (%)	Total Nitrogen Flux (%)
07194800	low	68.5	10.9	53.5	8.2	50.8	10.4
	medium	20.4	21.5	32.3	11.9	33.6	28.1
	high	11.0	67.6	14.2	79.9	15.6	61.5
07195000	low	56.1	8.3	38.9	13.3	37.9	7.6
	medium	30.3	28.4	56.9	70.9	57.6	28.6
	high	13.6	63.2	4.2	15.8	4.5	63.8
07195400	low	56.1	8.3	30.6	17.8	27.0	7.6
	medium	30.3	28.4	50.0	43.9	55.0	28.6
	high	13.6	63.2	19.4	38.2	18.0	63.8
07195500	low	52.8	13.6	59.8	8.4	69.4	11.0
	medium	37.1	37.6	31.8	21.5	23.5	46.5
	high	10.1	48.8	8.3	70.1	7.1	42.5
07196000	low	54.6	13.5	62.0	8.1	70.1	10.2
	medium	35.0	35.7	29.8	25.1	23.7	45.0
	high	10.4	50.8	8.3	66.8	6.2	44.8
07196500	low	51.2	12.4	53.7	8.7	66.3	8.5
	medium	38.5	39.9	36.0	25.2	25.3	47.4
	high	10.2	47.7	10.3	66.1	8.4	44.2
07196900	low	61.6	9.4	64.3	8.5	59.9	9.9
	medium	28.3	28.5	28.6	18.5	32.1	34.2
	high	10.1	62.1	7.1	73.0	8.0	55.9
07197000	low	56.6	10.9	63.2	4.1	74.2	7.2
	medium	32.3	33.2	25.6	14.3	18.6	34.3
	high	11.1	56.0	11.2	81.6	7.2	58.5

TABLE XXX

COX AND STUART TREND TEST RESULTS ON HISTORICAL NUTRIENT WATER QUALITY MONITORING DATA AT USGS GAGING STATIONS IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980 - 1993

USGS Gaging Station Identification	Total Phosphorus Concentration (mg/l)	OSL	Total Phosphorus Load (kg/yr)	OSL	Total Nitrogen Concentration (mg/l)	OSL	Total Nitrogen Load (kg/yr)	OSL
07194800	***	0.035	***	0.023	***	0.008	***	0.016
07195400		0.992	*	0.159		0.984		0.611
07195500		0.981	**	0.073	***	0.000	***	0.000
07196500		0.500	***	0.000	***	0.042	***	0.000
07195000		0.977		0.977		1.000		0.569
07196000		0.551	***	0.000	***	0.019	***	0.000
07196900		0.997		0.684		0.643	*	0.113
07197000	***	0.002	***	0.000	**	0.095	***	0.001

*** = significant upward trend at alpha <= 0.05
** = significant upward trend at alpha <= 0.10
* = significant upward trend at alpha <= 0.20

As reported in the Phase I Diagnostic and Feasibility Study on Tenkiller Lake (OWRB, 1996), estimates were made on the contributions of point, nonpoint, and background loads reaching the Horseshoe Bend area of Lake Tenkiller based on water quality and discharge data at gages 0196500 (Illinois River near Tahlequah) and 07197000 (Baron Fork near Eldon) and point source information. Using first-order kinetic assimilation rates and calculated distances of monitoring stations and point sources from the Horseshoe Bend area, nutrient concentrations were decreased based on distance traveled to estimate the effect of biological uptake and settling of nutrients that reduces water column concentrations of nutrients moving downstream. First-order kinetic assimilation rates for total phosphorus were calculated as -0.022 , -0.016, and

-0.001 per kilometer for low, medium, and high flows, respectively. First-order kinetic assimilation rates for total nitrogen were calculated as -0.017, -0.009, and -0.006 per kilometer for low, medium, and high flows, respectively.

The first-order kinetic assimilation rates calculated above were used to estimate the average annual quantity of nutrient loads from the gaging stations and municipal point source dischargers arriving at the Horseshoe Bend area of Lake Tenkiller (see Tables XXXII and XXXIII). The method provides an estimate of 12,500 kg/yr total phosphorus and 61,600 kg/yr total nitrogen entering the Horseshoe Bend area of Lake Tenkiller from point sources.

The first-order assimilation rates were also used to estimate the in-stream reductions of nutrient loads calculated at USGS gaging stations 07196500 and 07197000, and point discharges from Tahlequah and Cherokee Nation facilities, which are several kilometers above the Horseshoe Bend area of Lake Tenkiller (see Tables XXXIV and XXXV). Estimates of mean annual adjusted loads entering Lake Tenkiller at Horseshoe Bend are 228,000 kg P/yr and 2,300,000 kg N/yr.

Background concentrations of phosphorus and nitrogen were assumed to be 0.02 mg/l and 0.5 mg/l, respectively, based on estimation methods discussed earlier. Background loads at Horseshoe Bend were estimated by multiplying the average annual runoff by the background concentration. The combined background load for gaging stations 07196500 and 07197000 was estimated as 25,000 kg/yr total phosphorus, and 550,000 kg/yr total nitrogen. Table XXXI describes background loading calculations for each of the USGS gaging stations

in the Illinois River Basin. Tables XXXVI and XXXVII summarize the estimated distribution of nutrient loads between background, point and nonpoint sources. For all flow regimes combined, nonpoint sources of nitrogen are estimated to represent 73.4% of the total nitrogen entering Lake Tenkiller at the Horseshoe Bend area. Point sources are estimated to contribute 2.7% of the total. Most stream flows fall within the medium flow stratum and here the results are similar with 73.9% from nonpoint sources and 2.2% from point sources.

TABLE XXXI

RESULTS OF EPA SCREENING METHODS FOR ESTIMATING BACKGROUND LOADING OF TOTAL PHOSPHORUS AND TOTAL NITROGEN

	USGS 07194800	USGS 07195000	USGS 07195400	USGS 07195500	USGS 07196000	USGS 07196500	USGS 07196900	USGS 07197000
Drainage Area (km^2)	433	337	1320	1650	285	2,480	105	795
Stream Flow (m)	0.368	0.381	0.368	0.327	0.373	0.327	0.369	0.349
Annual Runoff (m^3)	1.59E+08	1.28E+08	4.86E+08	5.38E+08	1.06E+08	8.13E+08	3.88E+07	2.78E+08
(m^3/s)	5.05	4.07	15.39	17.05	3.37	25.77	1.23	8.80
Estimated Background Concentration (mg/l)		Total Phosphorus		Total Nitrogen				
		0.020		0.501				
Estimated Background Load (kg/yr)								
Total Phosphorus	3,190	2,570	9,710	10,800	2,130	16,300	776	5,550
Total Nitrogen	79,800	64,300	243,000	270,000	53,200	407,000	19,400	139,000

As would be expected, the point source contribution is greatest at low flows and least under high flow conditions.

The estimates of total phosphorus load sources at the Horseshoe Bend area of Lake Tenkiller are similar. Considering all flow regimes combined, point sources contribute 5.5%, and nonpoint sources 83.5% of the total phosphorus load. Under low flow conditions it is estimated that point sources contribute 44.1% of the total phosphorus load, and this percent contribution declines dramatically with increasing stream flow rates.

TABLE XXXII

ESTIMATES OF POINT SOURCE TOTAL PHOSPHORUS TRANSPORTED TO THE HORSESHOE BEND AREA OF LAKE TENKILLER (1991 TO 1993 DATA)

Discharger	Estimated Load at Source (kg/yr)	Distance to Horseshoe Bend (km)	Estimated Corrected Load at Horseshoe Bend			Estimated Annual Total Load (kg/yr)
			Low Flow (kg/yr)	Medium Flow (kg/yr)	High Flow (kg/yr)	
Prairie Grove	1,200	161	19	28	23	70
Rogers	21,600	159	355	519	417	1,290
Fayetteville	4,500	156	80	114	90	284
Springdale	43,150	153	820	1,150	893	2,860
Lincoln	1,200	130	38	46	31	115
Gentry	1,700	109	85	91	56	232
Siloam Springs	10,000	100	623	628	362	1,610
Watts	500	100	31	31	18	81
Westville	2,900	45	615	441	187	1,240
Midwestern Nursery	600	23	211	131	49	391
Tahlequah	4,700	10	2,200	1,270	441	3,910
Cherokee Nation	530	8	257	147	51	455
Total	92,600		5,300	4,600	2,620	12,500

TABLE XXXIII

ESTIMATES OF POINT SOURCE TOTAL NITROGEN TRANSPORTED TO THE HORSESHOE BEND AREA OF LAKE TENKILLER (1991 - 1993 DATA)

Discharger	Estimated Load at Source (kg/yr)	Distance to Horseshoe Bend (km)	Estimated Corrected Load at			Estimated Annual Total Load (kg/yr)
			Low Flow (kg/yr)	Medium Flow (kg/yr)	High Flow (kg/yr)	
Prairie Grove	3,000	161	106	233	115	454
Rogers	54,000	159	1,960	4,250	2,090	8,300
Fayetteville	11,300	156	432	911	444	1,790
Springdale	108,000	153	4,380	8,990	4,340	17,700
Lincoln	3,000	130	180	304	139	623
Gentry	4,250	109	368	517	224	1,110
Siloam Springs	25,000	100	2,560	3,310	1,400	7,270
Watts	1,300	100	133	172	73	378
Westville	7,300	45	1,940	1,550	574	4,060
Midwestern Nursery	5,000	23	1,960	1,300	452	3,710
Tahlequah	17,000	10	8,350	4,930	1,670	15,000
Cherokee Nation	1,400	8	707	411	138	1,260
Total	241,000		23,100	26,900	11,700	61,500

TABLE XXXIV

ESTIMATED QUANTITIES OF TOTAL PHOSPHORUS LOADS AT THE HORSESHOE BEND AREA OF LAKE TENKILLER

Discharger	Estimated Load at Source (kg/yr)	Distance to Horseshoe Bend (km)	Estimated Corrected Load at Horseshoe Bend			Estimated Annual Total Load (kg/yr)
			Low Flow (kg/yr)	Medium Flow (kg/yr)	High Flow (kg/yr)	
USGS 07196500	198,000	16	12,000	38,600	111,000	162,000
USGS 07197000	72,600	15	2,100	8,130	51,200	61,400
Tahlequah	4,700	10	2,200	1,270	441	3,910
Cherokee Nation	530	8	257	147	51	455
Totals	276,000		16,600	48,100	163,000	228,000

TABLE XXXV

ESTIMATED QUANTITIES OF TOTAL NITROGEN LOADS AT THE HORSESHOE BEND AREA OF LAKE TENKILLER

Discharger	Estimated Load at Source (kg/yr)	Distance to Horseshoe Bend (km)	Estimated Corrected Load at Horseshoe Bend			Estimated Annual Total Load (kg/yr)
			Low Flow (kg/yr)	Medium Flow (kg/yr)	High Flow (kg/yr)	
USGS 07196500	1,950,000	16	110,000	685,000	925,000	1,720,000
USGS 07197000	631,000	15	35,400	181,000	348,000	564,000
Tahlequah	17,000	10	8,350	4,930	1,670	15,000
Cherokee Nation	1,400	8	707	411	138	1,260
Totals	2,600,000		154,000	871,000	1,270,000	2,300,000

TABLE XXXVI

ESTIMATED DISTRIBUTION OF TOTAL PHOSPHORUS LOAD BETWEEN BACKGROUND, POINT, AND NONPOINT SOURCES AT THE HORSESHOE BEND AREA OF LAKE TENKILLER

Source	Estimated Average Total Phosphorus Load at Horseshoe Bend (kg/yr)	Estimated Low Flow Contribution at Horseshoe Bend (kg/yr)	Estimated Medium Flow Contribution at Horseshoe Bend kg P/yr (%)	Estimated High Flow Contribution at Horseshoe Bend kg P/yr (%)
Background	25,000 (11.0%)	1,600 (9.7%)	5,230 (10.9%)	18,200 (11.2%)
Point Source	12,500 (5.5%)	7,290 (44.1%)	3,950 (8.2%)	1,300 (0.8%)
Nonpoint Source	190,000 (83.5%)	7,630 (46.2%)	39,000 (80.9%)	143,000 (88.0%)
Total	227,500	16,520 (7.3% of Total)	48,180 (21.2% of Total)	162,500 (71.4% of Total)

TABLE XXXVII

ESTIMATED DISTRIBUTION OF TOTAL NITROGEN LOAD BETWEEN BACKGROUND, POINT, AND NONPOINT SOURCES AT THE HORSESHOE BEND AREA OF LAKE TENKILLER

Source	Estimated Average Total Nitrogen Load at Horseshoe Bend (kg/yr)	Estimated Low Flow Contribution at Horseshoe Bend (kg/yr)	Estimated Medium Flow Contribution at Horseshoe Bend (kg/yr)	Estimated High Flow Contribution at Horseshoe Bend (kg/yr)
Background	550,000 (23.9%)	35,200 (22.8%)	208,000 (23.9%)	306,000 (24.0%)
Point Source	61,600 (2.7%)	35,800 (23.2%)	19,400 (2.2%)	6,400 (0.5%)
Nonpoint Source	1,690,000 (73.4%)	83,400 (54.0%)	644,000 (73.9%)	962,000 (75.5%)
Total	2,300,600	154,400 (6.7% of Total)	871,400 (37.9% of Total)	1,274,400 (55.4% of Total)

SIMPLE Model Results

The phosphorus transport model SIMPLE was used to determine estimates of nonpoint source phosphorus storm flow loading from fifteen subbasins comprising the entire Illinois River Basin above the Lake Tenkiller dam. The SIMPLE model does not account for gully and stream bank contributions or in-stream processes affecting nutrient loading. The Illinois River Basin was divided into fifteen subbasins to aid in the organization of results and to reduce the computer memory and hard disk requirements. The boundaries of the subbasins were determined using a digital terrain model. Figure 12 depicts the Illinois River Basin and the fifteen subbasins defined using the DTM.

Table XXXVIII describes the average annual results of the runs of the SIMPLE model on the subbasins comprising the Illinois River Basin. The model was run with 25 years of rainfall data (1962 through 1985) for each subbasin.

The SIMPLE model estimated runoff and total phosphorus export based on the rainfall data. The estimates of mean annual loading were converted to mean annual concentrations using mean annual runoff calculated by the model and the area of each basin. Basins with higher soil phosphorus levels show higher concentrations of nutrients in runoff. Unit area loading methods described above were used to estimate total nitrogen loading. Based on the 25 year simulation, the 13 subbasins above the Horseshoe Bend Area of Lake Tenkiller contribute an annual average 194,000 kg/yr total phosphorus to the system from anthropogenic nonpoint sources. This total is essentially the same as the 190,000 kg/yr nonpoint source total phosphorus load arriving at Horseshoe Bend estimated using historical data. Additionally, an annual average 4,850 kg/yr nonpoint total phosphorus is transported to the lake from the CANEY and LAKE subbasins at points below Horseshoe Bend.

Figure 13 shows SIMPLE predicted total phosphorus export in the Illinois River Basin. Areas highlighted in blue, magenta, and red, with export at or above 1.0 kg/ha/yr, cover a significant portion (23.9%) of the basin. Subbasins with 25% or more of their total area having predicted total phosphorus export values of 1.0 kg/ha/yr or greater are BENTON (25.9%), OSAGE (30.1%), FLINT (34.8%), CLEAR (42.2%), WEST (43.2%), BARON (44.3%), and FORK (49.1%). The remaining eight subbasins are predicted to have less than 1% of their total area with export values of 1.0 kg/ha/yr or greater.

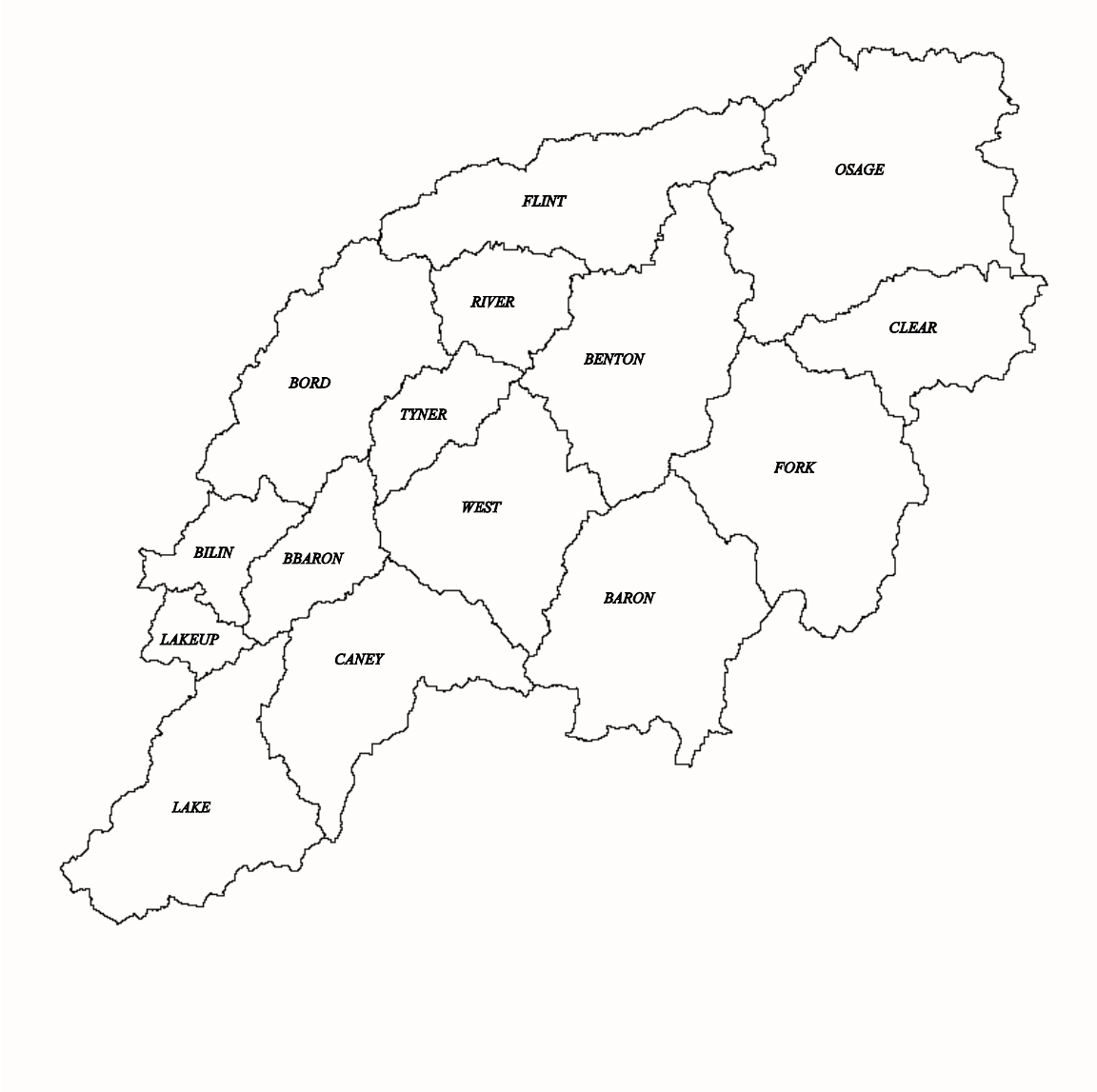

Figure 12. Identification of Subbasins in the Illinois River Watershed.

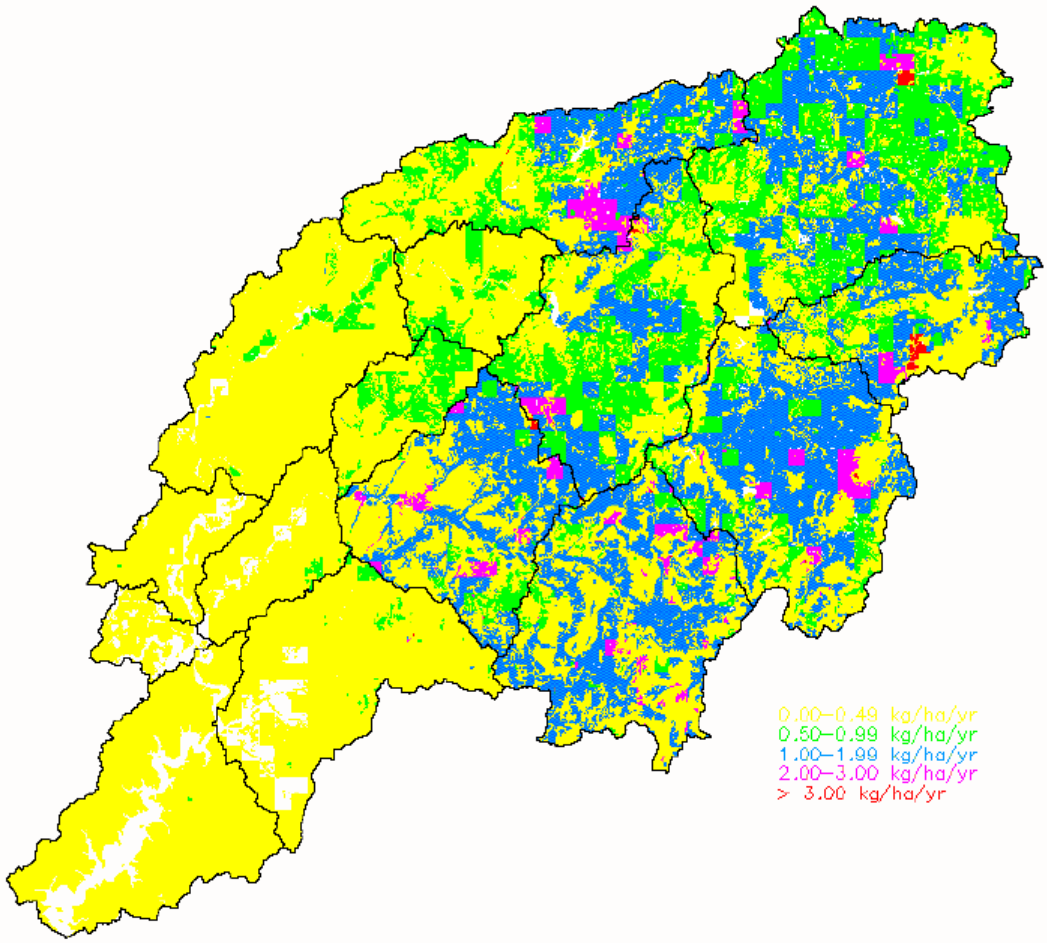

Figure 13. SIMPLE Predicted Average Annual Total Phosphorus Export in the Illinois River Basin.

Analysis shows that 47% of pasture and range land is predicted to have total phosphorus export values of 1.00 kg/ha/yr or greater. This accounts for 97.5% of all the area with high export values.

The box and whisker representations of subbasin discharge distributions, shown in Figure 14, identify the minimum, maximum, interquartile range, and the median. This will be the case for all box and whisker graphics unless otherwise

noted. Figure 15 shows the distributions of annual discharge from each of the 15 subbasins based on SIMPLE generated runoff for 25 years of rainfall data. Mean annual discharges calculated for each subbasin were used as nonpoint source inputs into the QUAL2EU model.

Figure 16 shows distributions of annual average total phosphorus loads for each of the 15 subbasins based on the 25 years of the SIMPLE simulation. OSAGE, FORK, BARON, BENTON, FLINT, and CLEAR subbasins show loadings substantially greater than the remaining subbasins based on the higher soil phosphorus levels in these subbasins. Total phosphorus loading for each subbasin was divided by the area of each subbasin to get annual average loading per hectare per year. Figure 16 shows that CLEAR, OSAGE, FORK, FLINT, BARON, and BENTON subbasins have loadings much greater than the others with median total phosphorus loadings greater than 0.50 kg/ha/yr. Concentrations of average annual total phosphorus in runoff from each of the subbasins are shown in Figure 17. The same subbasins stand out as having substantially greater concentrations. Figures 18 and 19 offer graphic representations of the relative contributions of SIMPLE subbasins to total phosphorus and nitrogen loads above Horseshoe Bend. Subbasins OSAGE, FORK, BARON, FLINT, BENTON, and CLEAR account for an estimated 89.2% of the nonpoint source total phosphorus load reaching Lake Tenkiller. These same subbasins account for 82.4% of the nonpoint source total nitrogen load to Lake Tenkiller.

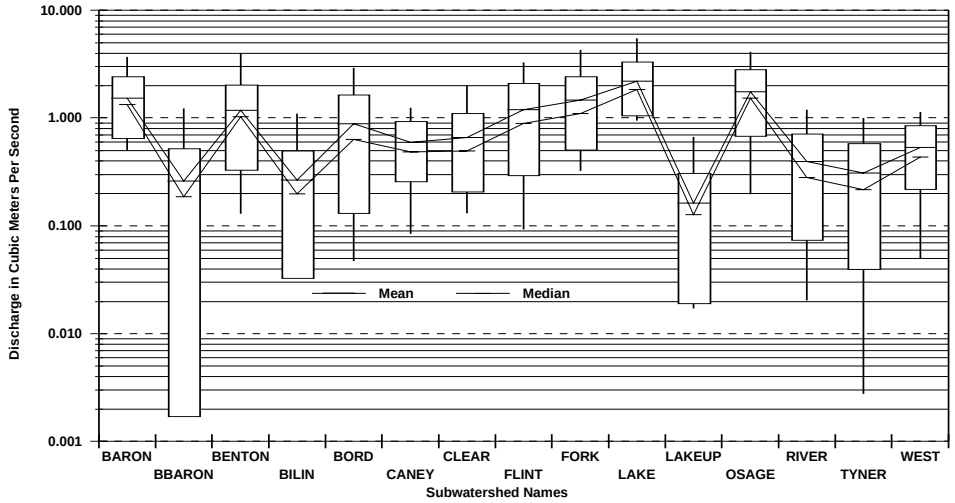

Figure 14. Distributions of Annual Average Discharge Based on SIMPLE Generated Runoff for Subbasins in the Illinois River Watershed.

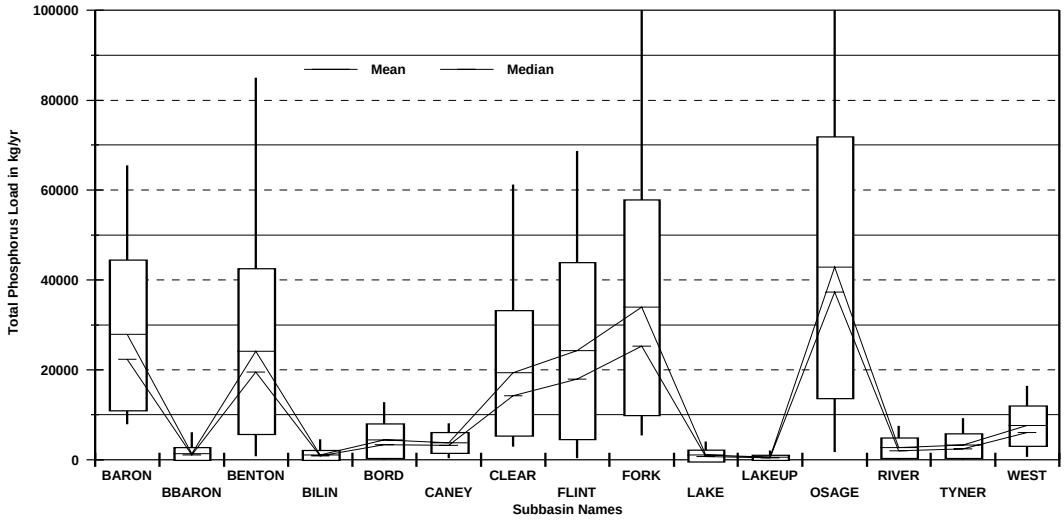

Figure 15. Distributions of SIMPLE Generated Annual Average Total Phosphorus Loads for Subbasins in the Illinois River Watershed.

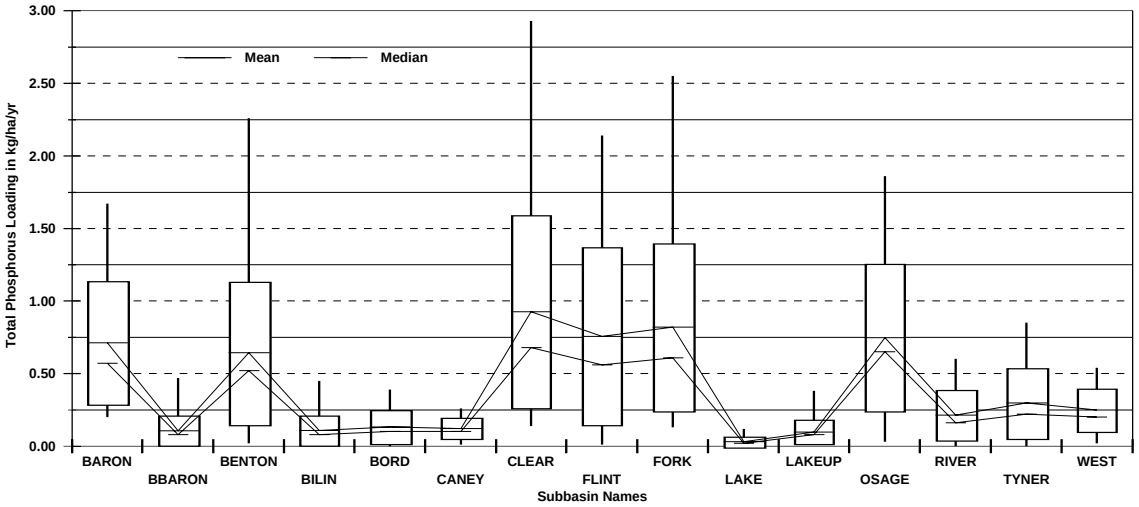

Figure 16. Distributions of SIMPLE Generated Annual Average Total Phosphorus Loading per Hectare for Subbasins in the Illinois River Watershed.

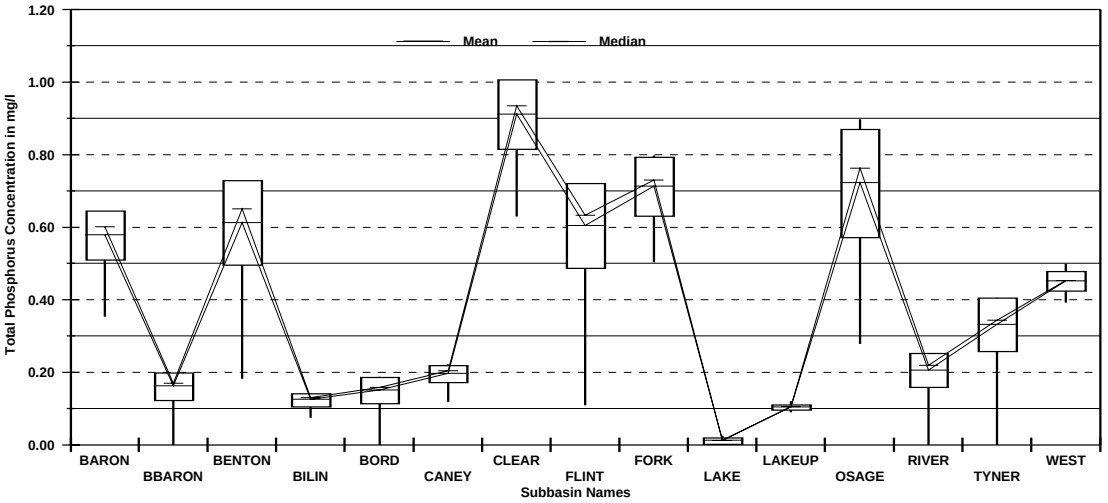

Figure 17. Distributions of Annual Average Total Phosphorus Concentration in Runoff from Subbasins in the Illinois River Watershed Based on SIMPLE Generated Loads, Runoff, and Subbasin Area.

TABLE XXXVIII

RESULTS OF SIMPLE SIMULATIONS OF TOTAL PHOSPHORUS TRANSPORT FROM ILLINOIS RIVER SUBBASINS BASED ON 25 YEARS OF RAINFALL DATA (1962-1986)

Subbasin Name	Statistics	Rainfall (cm)	Runoff (cm)	Total Phosphorus (kg/ha)	Total Phosphorus (kg/yr)	Total Phosphorus (mg/l)	Discharge (m^3/s)
BARON	Mean	123.5	12.3	0.71	27,900	0.58	1.53
	Median	115.4	10.8	0.57	22,400	0.60	1.34
	SD	28.3	7.0	0.42	16,500	0.07	0.88
	Minimum	69.8	4.0	0.20	7,840	0.35	0.50
	Maximum	207.3	29.5	1.67	65,500	0.63	3.67
BBARON	Mean	83.7	6.3	0.11	1,380	0.16	0.26
	Median	78.0	4.5	0.08	1,040	0.17	0.19
	SD	22.7	6.3	0.10	1,330	0.04	0.26
	Minimum	53.4	0.6	0.00	0	0.00	0.03
	Maximum	151.1	29.6	0.47	6,110	0.19	1.22
BENTON	Mean	113.2	9.9	0.64	24,200	0.61	1.18
	Median	105.9	8.6	0.52	19,600	0.65	1.02
	SD	28.8	7.1	0.49	18,300	0.12	0.84
	Minimum	59.5	1.1	0.02	752	0.18	0.13
	Maximum	192.9	33.6	2.26	85,000	0.70	4.01
BILIN	Mean	83.7	8.2	0.11	1,100	0.13	0.26
	Median	77.9	6.2	0.08	812	0.13	0.20
	SD	22.7	7.2	0.10	997	0.02	0.23
	Minimum	53.4	1.3	0.01	102	0.07	0.04
	Maximum	151.1	34.1	0.45	4,570	0.14	1.10
BORD	Mean	115.3	8.5	0.13	4,400	0.15	0.89
	Median	107.9	6.0	0.10	3,300	0.16	0.63
	SD	32.9	7.2	0.11	3,620	0.03	0.75
	Minimum	59.8	0.5	0.00	0	0.00	0.05
	Maximum	207.9	27.8	0.39	12,900	0.19	2.91
CANEY	Mean	84.1	6.0	0.12	3,820	0.20	0.59
	Median	84.6	4.8	0.10	3,150	0.20	0.48
	SD	19.6	3.3	0.07	2,230	0.02	0.33
	Minimum	54.6	0.8	0.01	314	0.12	0.08
	Maximum	126.0	12.4	0.26	8,180	0.22	1.24

TABLE XXXVIII

RESULTS OF SIMPLE SIMULATIONS OF TOTAL PHOSPHORUS TRANSPORT FROM ILLINOIS RIVER SUBBASINS BASED ON 25 YEARS OF RAINFALL DATA (1962-1986)

Subbasin Name	Statistics	Rainfall (cm)	Runoff (cm)	Total Phosphorus (kg/ha)	Total Phosphorus (kg/yr)	Total Phosphorus (mg/l)	Discharge (m^3/s)
CLEAR	Mean	108.6	9.9	0.93	19,300	0.91	0.66
	Median	106.1	7.4	0.68	14,200	0.93	0.49
	SD	24.0	6.7	0.66	13,800	0.09	0.44
	Minimum	54.8	2.0	0.14	2,930	0.63	0.13
	Maximum	165.4	30.2	2.93	61,200	1.00	2.00
FLINT	Mean	115.3	11.7	0.76	24,300	0.61	1.19
	Median	107.9	8.8	0.56	18,000	0.63	0.89
	SD	32.9	8.8	0.61	19,500	0.11	0.89
	Minimum	59.8	0.9	0.01	321	0.11	0.09
	Maximum	207.9	32.2	2.14	68,700	0.70	3.27
FORK	Mean	108.7	11.1	0.82	34,000	0.71	1.46
	Median	106.2	8.4	0.61	25,300	0.73	1.10
	SD	24.0	7.2	0.57	23,800	0.08	0.95
	Minimum	54.8	2.5	0.13	5,390	0.50	0.32
	Maximum	165.5	32.5	2.55	106,000	0.80	4.27
LAKE	Mean	93.0	20.3	0.03	1,030	0.01	2.19
	Median	89.3	17.0	0.02	680	0.01	1.83
	SD	25.6	10.4	0.03	1,080	0.01	1.12
	Minimum	49.6	8.8	0.00	0	0.00	0.95
	Maximum	168.3	50.8	0.12	4,080	0.03	5.47
LAKEUP	Mean	83.4	9.5	0.10	523	0.10	0.16
	Median	77.7	7.4	0.08	430	0.11	0.13
	SD	22.6	8.4	0.08	443	0.01	0.14
	Minimum	53.2	1.0	0.01	54	0.09	0.02
	Maximum	150.5	38.9	0.38	2,050	0.12	0.66
OSAGE	Mean	111.9	9.7	0.75	42,900	0.72	1.75
	Median	108.3	8.4	0.65	37,300	0.76	1.52
	SD	27.3	5.8	0.50	28,900	0.15	1.06
	Minimum	56.5	1.1	0.03	1,720	0.28	0.20
	Maximum	183.7	22.6	1.86	107,000	0.90	4.10

TABLE XXXVIII

RESULTS OF SIMPLE SIMULATIONS OF TOTAL PHOSPHORUS TRANSPORT FROM ILLINOIS RIVER SUBBASINS BASED ON 25 YEARS OF RAINFALL DATA (1962-1986)

Subbasin Name	Statistics	Rainfall (cm)	Runoff (cm)	Total Phosphorus (kg/ha)	Total Phosphorus (kg/yr)	Total Phosphorus (mg/l)	Discharge (m^3/s)
RIVER	Mean	115.1	9.9	0.21	2,670	0.21	0.39
	Median	107.7	7.0	0.16	2,010	0.22	0.28
	SD	32.8	8.0	0.17	2,150	0.04	0.32
	Minimum	59.7	0.5	0.00	0	0.00	0.02
	Maximum	207.6	29.9	0.60	7,540	0.23	1.19
TYNER	Mean	115.0	8.9	0.30	3,230	0.33	0.31
	Median	107.6	6.2	0.22	2,400	0.34	0.22
	SD	32.8	7.8	0.24	2,590	0.07	0.27
	Minimum	59.7	0.1	0.00	0	0.00	0.00
	Maximum	207.4	28.9	0.85	9,260	0.41	1.00
WEST	Mean	84.1	5.5	0.25	7,600	0.45	0.53
	Median	84.7	4.5	0.20	6,090	0.45	0.44
	SD	19.6	3.3	0.14	4,370	0.03	0.31
	Minimum	54.6	0.5	0.02	609	0.39	0.05
	Maximum	126.0	11.8	0.54	16,400	0.50	1.14

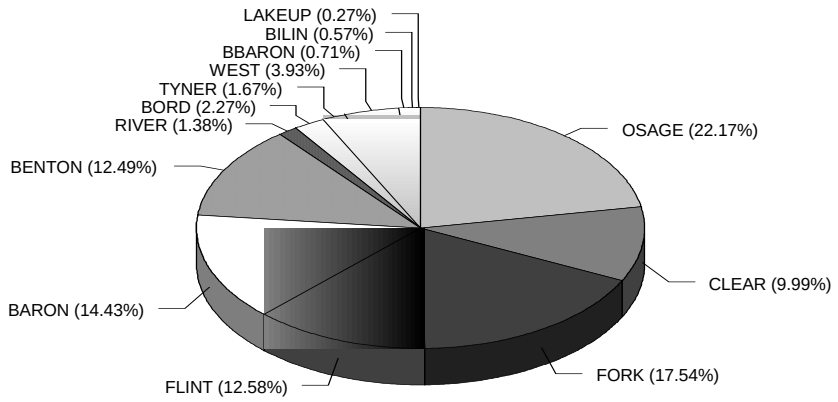

Figure 18. Relative Contributions of Average Annual Total Phosphorus Load

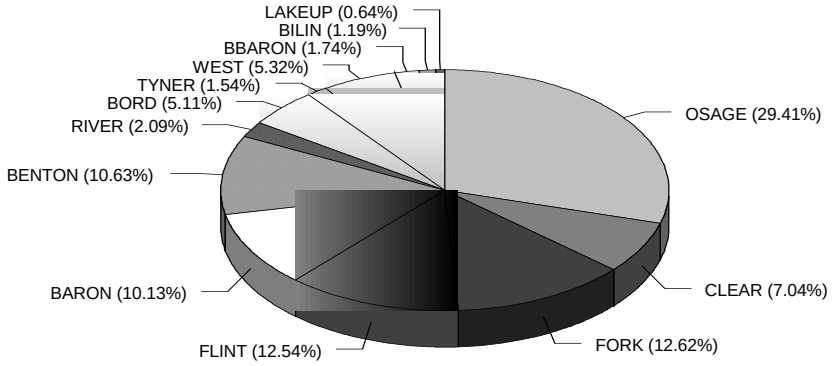

Figure 19. Relative Contributions of Average Annual Total Nitrogen Load from SIMPLE Subbasins at the Horseshoe Bend Area of Lake Tenkiller.

QUAL2EU Results

Two thousand Monte Carlo simulations were run on each of the QUAL2EU input data sets. The simulation representing average annual conditions was compared to historical data at the gaging stations in the basin. Figure 20 shows a comparison of simulated average annual flow versus historical flow. The simulation results match the historical data very closely due to the calibration of the QUAL2EU model with historical discharge data. Information concerning calibration of the QUAL2EU model can be found in Appendix B.

The 2000 simulations also allow for examination of the simulated frequency distributions of discharge compared to historical distributions. Figure 21 shows this comparison for average annual conditions where the simulated gaging station locations are indicated by reach (r) and element (e) numbers and historical data are described with gaging station abbreviations (GS*). Note that the medians are very close but the ranges differ in some cases. The Monte Carlo simulations varied the base input discharge based on a specified coefficient of variation. A single coefficient of variation was accepted by the model to describe discharge variation, and hence, at all gaging station locations within the basin. The coefficient of variation was calculated at all gaging stations, based on historical data, and averaged across all gaging stations to obtain one value to describe the entire basin. It should be noted that this may not

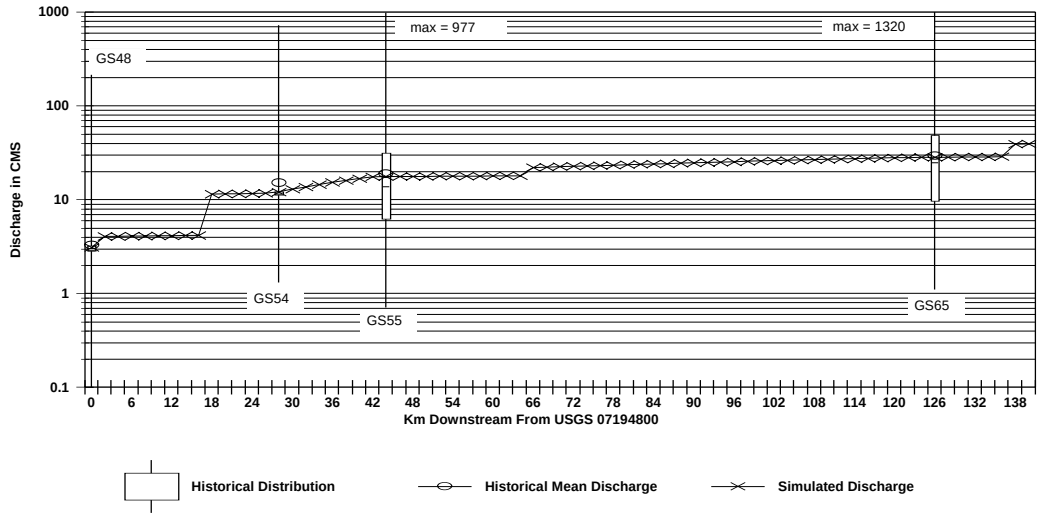

Figure 20. Simulated Annual Average Discharge Along the Mainstem of the Illinois River Compared with Historical Distributions of Discharge at USGS Gaging Stations.

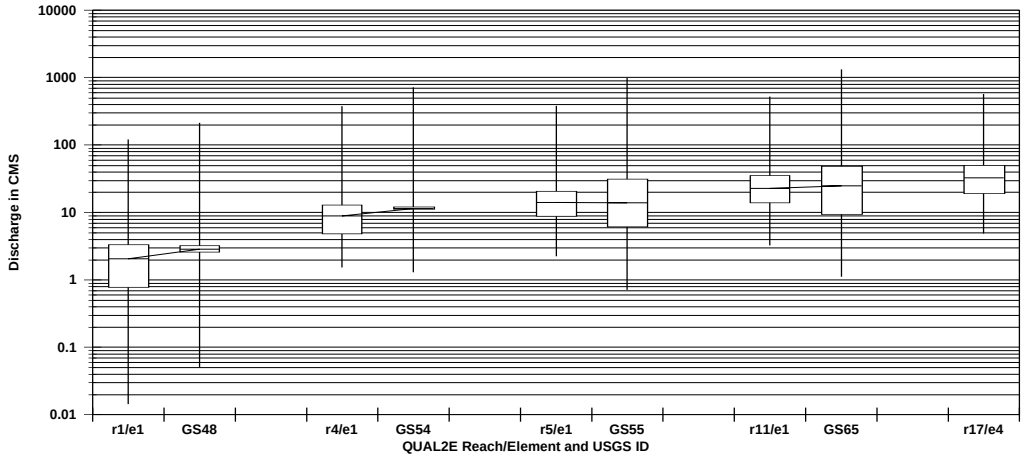

Figure 21. A Comparison of Simulated (r*/e*) and Historical (GS**)Discharge Distributions at Mainstem USGS Gaging Stations on the Illinois River.

accurately describe actual variation at each gage because of varying watershed size and characteristics.

Table XXXIX summarizes QUAL2EU simulation results for total phosphorus concentration at USGS gaging stations and the Horseshoe Bend area of Lake Tenkiller and these results are graphically compared to historical total phosphorus concentration distributions in Figure 22 . The model developed over-estimates average annual concentration but matches the historical data rather well on the mainstem of the Illinois River by maintaining average concentrations within the interquartile range of the historical data. An initial over-estimate occurs at USGS 07194800 (GS48 in Figure 22) as the SIMPLE model's predicted total phosphorus concentration is significantly greater than the historical annual average concentration.

Table XL summarizes average annual total nitrogen concentration results for the QUAL2EU model of the Illinois River basin. Figure 23 graphically compares mean simulated concentrations to historical distributions of total nitrogen at mainstem gaging stations. Again, the model over-estimates concentration when compared to the historical data, but in the mid and lower reaches of the mainstem, simulated concentrations do fall within the interquartile range of the historical data. The SIMPLE model's predicted total nitrogen concentration at USGS 07194800, the modeled Illinois River mainstem headwaters, is again significantly greater than the historical annual average concentration.

Figures 24 and 25 graphically compare distributions of simulated and

TABLE XXXIX

ESTIMATES OF AVERAGE ANNUAL TOTAL PHOSPHORUS CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.37	0.22	0.04	0.23	0.33	0.46	3.13
07195400	0.48	0.20	0.08	0.35	0.46	0.58	3.06
07195500	0.32	0.15	0.08	0.23	0.30	0.38	2.78
07196500	0.24	0.13	0.06	0.16	0.21	0.29	2.44
Horseshoe Bend	0.23	0.12	0.05	0.15	0.21	0.27	2.19
07195000	0.39	0.22	0.04	0.24	0.34	0.48	3.20
07196000	0.29	0.16	0.03	0.18	0.26	0.36	2.32
07196900	0.24	0.13	0.03	0.16	0.22	0.30	2.07
07197000	0.16	0.07	0.03	0.10	0.14	0.19	1.00

historical total phosphorus and total nitrogen concentrations, respectively. The model predicts a reasonable estimate of the measured distribution at most mainstem stations, with the possible exception of total phosphorus concentration at USGS 07194800.

Tables XLI through XLIII show the estimates of average annual total phosphorus concentrations, by source, at each of the USGS gages using simulated data. The simulation with zero point source nutrient inputs was subtracted from the average annual simulation to estimate point source contributions. In addition, a simulation was run estimating background concentrations of each nutrient. Nonpoint source nutrient concentrations were estimated by subtracting background simulation results from the simulation without point source nutrient inputs.

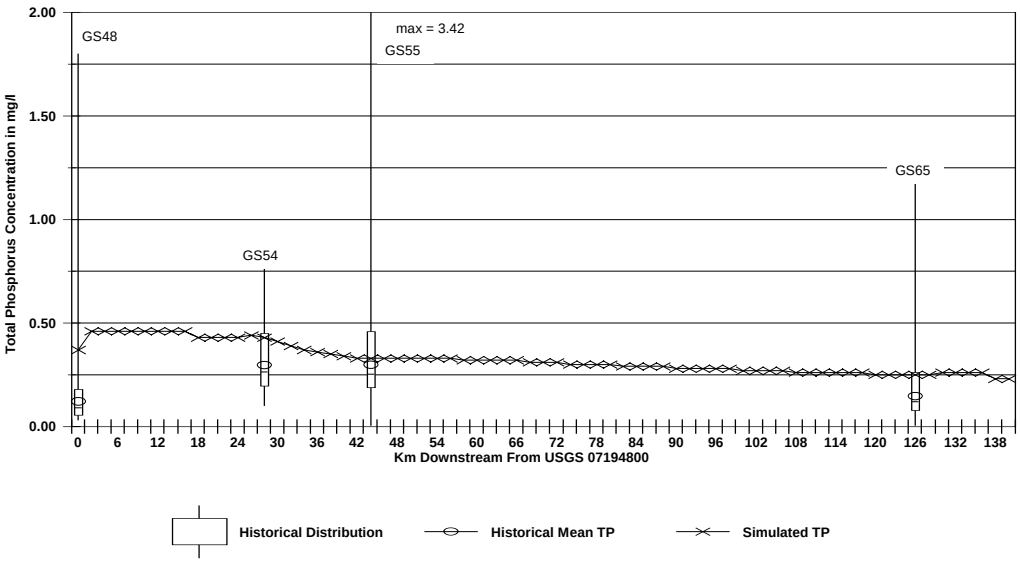

Figure 22. Simulated Average Annual Total Phosphorus Concentration Along the Mainstem of the Illinois River Compared to Historical Distributions at USGS Gaging Stations.

TABLE XL

ESTIMATES OF AVERAGE ANNUAL TOTAL NITROGEN CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	2.9	1.1	0.8	2.1	2.7	3.4	9.6
07195400	3.8	1.4	0.9	2.8	3.6	4.5	10.7
07195500	2.6	0.7	0.9	2.1	2.5	3.0	6.0
07196500	2.1	0.6	0.7	1.6	2.0	2.4	5.3
Horseshoe Bend	1.9	0.6	0.6	1.5	1.8	2.2	4.9
07195000	2.4	1.1	0.6	1.7	2.2	2.8	10.1
07196000	2.4	0.9	0.7	1.8	2.2	2.9	7.9
07196900	2.1	0.7	0.6	1.6	2.0	2.5	6.1
07197000	1.5	0.5	0.5	1.1	1.4	1.8	3.9

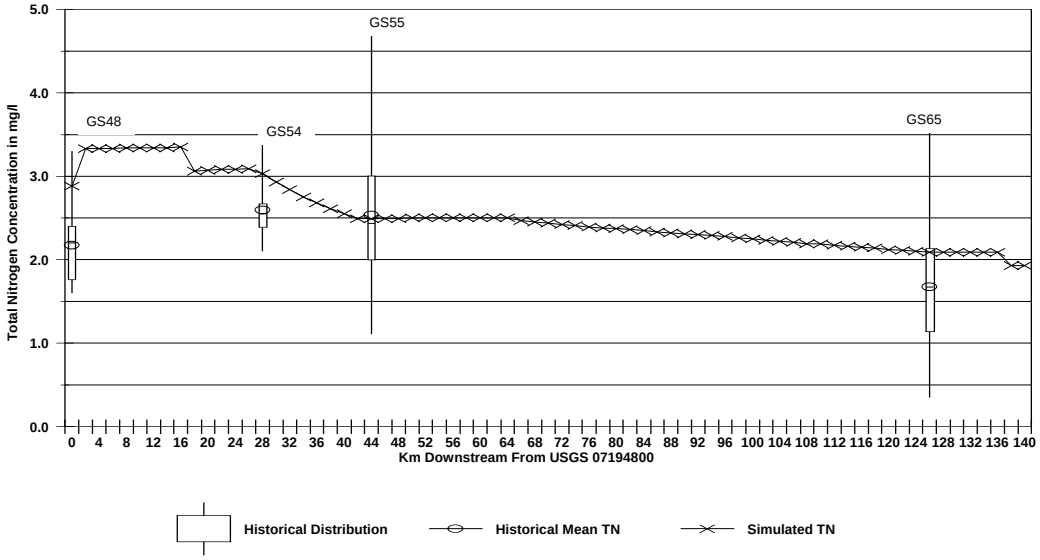

Figure 23. Simulated Average Annual Total Nitrogen Concentration Along the Mainstem of the Illinois River Compared to Historical Distributions at USGS Gaging Stations.

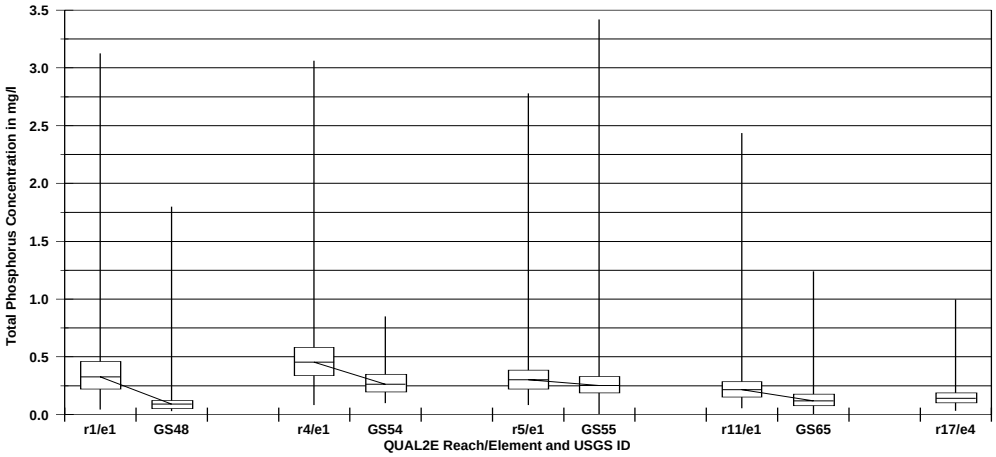

Figure 24. A Comparison of Simulated (r*/e*) and Historical (GS**) Total Phosphorus Distributions at Mainstem USGS Gaging Stations on the Illinois River.

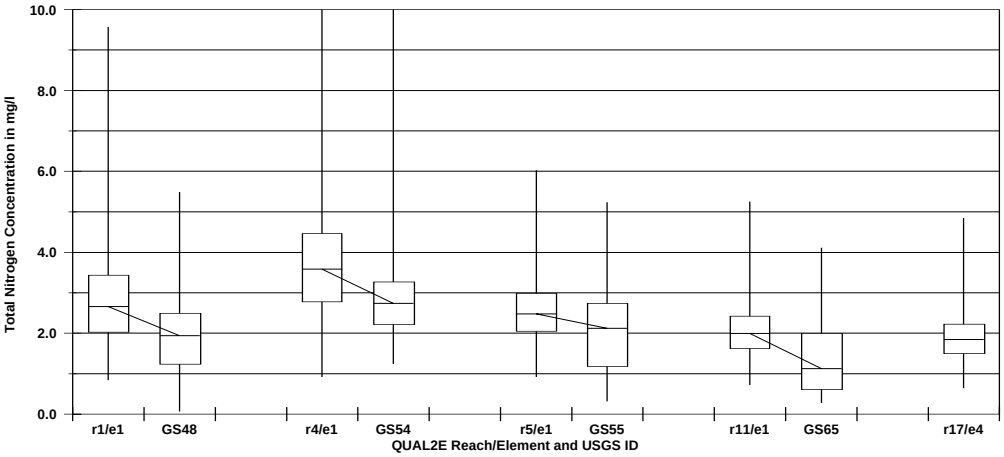

Figure 25. A Comparison of Simulated (r*/e*) and Historical (GS**)Total Nitrogen Distributions at Mainstem USGS Gaging Stations on the Illinois River.

This method may oversimplify the modeled system by ignoring probable nonlinearities, but it does provide useful estimates quantifying nutrients from each source. The simulated results then suggest that at Horseshoe Bend, where the mean annual concentration of total phosphorus was 0.23 mg/l, 0.15 mg/l (65.2%) are derived from nonpoint sources, 0.02 mg/l (8.7%) from background sources, and the remaining 0.06 mg/l (26.1%) from point sources.

TABLE XLI

ESTIMATES OF AVERAGE ANNUAL ANTHROPOGENIC NONPOINT SOURCE TOTAL PHOSPHORUS CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.34	0.20	0.04	0.21	0.30	0.42	2.87
07195400	0.35	0.16	0.04	0.24	0.33	0.43	1.52
07195500	0.23	0.09	0.04	0.16	0.21	0.27	1.39
07196500	0.16	0.07	0.04	0.11	0.15	0.20	1.28
Horseshoe Bend	0.15	0.07	0.04	0.10	0.14	0.18	1.20
07195000	0.13	0.08	0.01	0.08	0.12	0.16	0.94
07196000	0.20	0.11	0.02	0.12	0.17	0.24	1.43
07196900	0.21	0.11	0.03	0.13	0.19	0.25	1.66
07197000	0.12	0.06	0.02	0.08	0.11	0.15	0.80

TABLE XLII

ESTIMATES OF ANNUAL AVERAGE BACKGROUND TOTAL PHOSPHORUS CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.02	0.01	0.00	0.01	0.02	0.03	0.17
07195400	0.02	0.01	0.00	0.01	0.02	0.02	0.16
07195500	0.02	0.01	0.00	0.01	0.02	0.02	0.15
07196500	0.02	0.01	0.00	0.01	0.02	0.02	0.14
Horseshoe Bend	0.02	0.01	0.00	0.01	0.02	0.02	0.13
07195000	0.02	0.01	0.00	0.01	0.02	0.02	0.17
07196000	0.02	0.01	0.00	0.01	0.02	0.03	0.17
07196900	0.02	0.01	0.00	0.01	0.02	0.02	0.17
07197000	0.02	0.01	0.00	0.01	0.02	0.02	0.09

TABLE XLIII

ESTIMATES OF ANNUAL AVERAGE POINT SOURCE TOTAL PHOSPHORUS CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.01	0.01	0.00	0.01	0.01	0.01	0.09
07195400	0.12	0.03	0.04	0.10	0.11	0.13	1.38
07195500	0.08	0.05	0.04	0.05	0.07	0.09	1.25
07196500	0.06	0.05	0.01	0.03	0.05	0.07	1.02
Horseshoe Bend	0.06	0.04	0.01	0.04	0.06	0.08	0.87
07195000	0.24	0.13	0.03	0.15	0.21	0.29	2.09
07196000	0.08	0.04	0.01	0.05	0.07	0.10	0.72
07196900	0.02	0.01	0.00	0.01	0.02	0.02	0.25
07197000	0.02	0.00	0.01	0.01	0.02	0.02	0.11

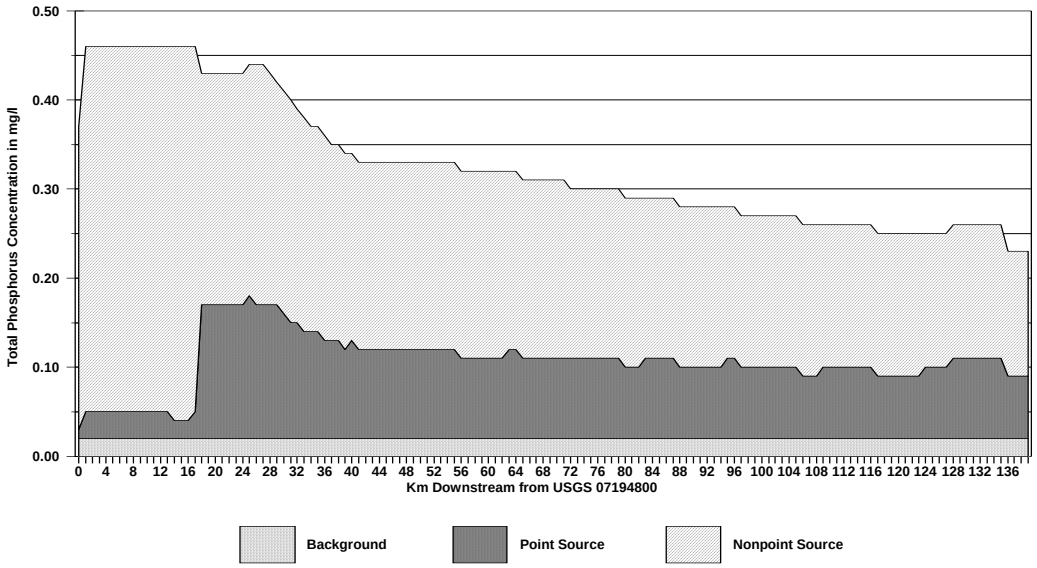

Figure 26. A Representation of the Fractions of Simulated Annual Average Total Phosphorus from Background, Point, and Nonpoint Sources Along the Mainstem of the Illinois River.

Figure 26 depicts graphically the simulated breakdown of the source of total phosphorus concentration on the mainstem of the Illinois River. The point source contribution jumps up at the point where Osage Creek enters the Illinois River, and gradually tapers downward proceeding toward Lake Tenkiller with the influence of Tahlequah's point source evident as another slight increase.

Similar tabular and graphical results were prepared for simulated total nitrogen concentration. Tables XLIV through XLVI describe the simulated breakdown of sources of total nitrogen concentration at USGS gaging station locations. Figure 27 graphically depicts the breakdown on the mainstem of the Illinois River. Again, using Horseshoe Bend as an example, the simulated average annual concentration of total nitrogen was 1.9 mg/l. Of this, 1.4 mg/l (71.8%) can be attributed to nonpoint sources, 0.5 mg/l (25.6%) to background sources, and only 0.05 mg/l (2.6%) attributable to point sources.

Table XLVII shows the relative errors of simulated versus historical total phosphorus and total nitrogen concentrations. It is evident that for total phosphorus there is consistent over-estimation of mean annual concentration at the gaging stations averaging just over 55%. The greatest difference is at USGS 07194800 (208%). The under-estimate of total phosphorus concentration at USGS 07195000 may be partly attributed to the point source in-stream assimilation method employed. Total nitrogen concentrations at the gaging stations was also over-estimated, although somewhat less, with an average relative error over all stations of nearly 24%. The greatest error occurs at USGS 07196500 with a 62% over-estimate. Again, USGS 07195000 shows a

significant under-estimate (-48%).

TABLE XLIV

ESTIMATES OF ANNUAL AVERAGE NONPOINT SOURCE TOTAL NITROGEN CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	2.4	0.9	0.7	1.7	2.2	2.8	7.9
07195400	3.2	1.2	0.7	2.4	3.1	3.9	9.3
07195500	2.1	0.6	0.7	1.7	2.0	2.4	4.8
07196500	1.6	0.5	0.6	1.3	1.5	1.8	3.9
Horseshoe Bend	1.4	0.4	0.5	1.1	1.3	1.6	3.4
07195000	1.8	0.8	0.5	1.2	1.7	2.2	8.4
07196000	1.9	0.7	0.5	1.4	1.7	2.2	6.0
07196900	1.6	0.5	0.5	1.2	1.5	1.8	4.6
07197000	1.0	0.3	0.3	0.7	0.9	1.1	2.4

TABLE XLV

ESTIMATES OF ANNUAL AVERAGE BACKGROUND TOTAL NITROGEN CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.5	0.2	0.1	0.4	0.5	0.6	1.7
07195400	0.5	0.1	0.2	0.4	0.5	0.6	1.3
07195500	0.5	0.1	0.2	0.4	0.5	0.6	1.2
07196500	0.5	0.2	0.2	0.4	0.4	0.6	1.2
Horseshoe Bend	0.5	0.2	0.2	0.4	0.4	0.6	1.3
07195000	0.5	0.2	0.1	0.4	0.5	0.6	1.7
07196000	0.5	0.2	0.1	0.4	0.5	0.6	1.6
07196900	0.5	0.2	0.2	0.4	0.5	0.6	1.3
07197000	0.5	0.2	0.2	0.4	0.5	0.6	1.5

TABLE XLVI

ESTIMATES OF AVERAGE ANNUAL POINT SOURCE TOTAL NITROGEN CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.00	0.00	0.00	0.00	0.00	0.00	0.00
07195400	0.05	0.00	0.02	0.04	0.05	0.05	0.06
07195500	0.03	0.01	0.02	0.02	0.03	0.04	0.02
07196500	0.03	0.01	0.01	0.03	0.04	0.03	0.12
Horseshoe Bend	0.05	0.01	0.02	0.05	0.06	0.07	0.14
07195000	0.07	0.02	0.02	0.05	0.06	0.08	0.06
07196000	0.07	0.03	0.02	0.05	0.06	0.08	0.24
07196900	0.04	0.02	0.00	0.03	0.04	0.06	0.18
07197000	0.04	0.01	0.01	0.03	0.04	0.05	0.02

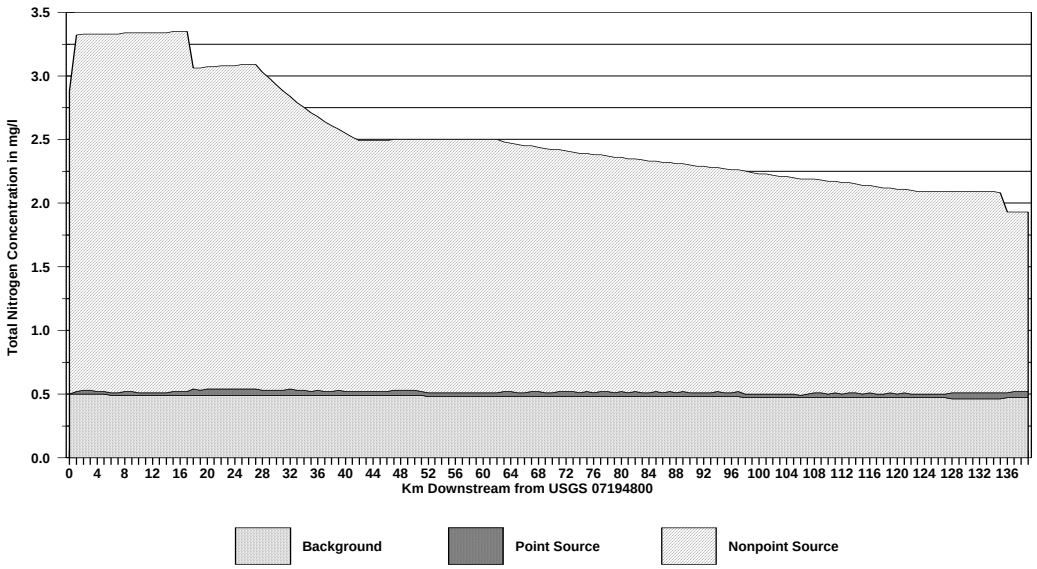

Figure 27. A Representation of the Fractions of Simulated Annual Average Total Nitrogen from Background, Point, and Nonpoint Sources Along the Mainstem of the Illinois River.

TABLE XLVII

RELATIVE ERRORS OF SIMULATED TOTAL PHOSPHORUS AND TOTAL NITROGEN CONCENTRATIONS COMPARED TO HISTORICAL DATA

USGS Gaging Station Identification	Historical Mean Total Phosphorus Conc. (mg/l)	Simulated Mean Total Phosphorus Conc. (mg/l)	Total Phosphorus Relative Error (%)	Historical Mean Total Nitrogen Conc. (mg/l)	Simulated Mean Total Nitrogen Conc. (mg/l)	Total Nitrogen Relative Error (%)
07194800	0.12	0.37	208	2.0	2.9	45
07195400	0.30	0.48	60	3.0	3.8	27
07195500	0.30	0.32	7	2.0	2.6	30
07196500	0.15	0.24	60	1.3	2.1	62
07195000	0.96	0.39	-59	4.6	3.8	-48
07196000	0.20	0.29	45	1.7	2.4	41
07196900	0.15	0.24	60	2.2	2.1	-5
07197000	0.10	0.16	60	1.1	1.5	36

Figure 28 compares simulated mean annual total phosphorus loading with historical distributions of annual average total phosphorus load at USGS gaging stations. The simulated mean loads fall outside the 75th percentiles of the historical loading data, but are relatively close when compared to historical mean values.

Tables L through LII describe simulated distributions of nonpoint, background, and point source total phosphorus loads, respectively. Using Horseshoe Bend as an example, where the simulated mean annual load was 291,000 kg/yr, 189,000 kg/yr (65%) can be attributed to nonpoint sources, 21,900 kg/yr (7.5%) to background sources, and the remaining 79,800 kg/yr (27.5%) to point sources. This compares reasonably well to historical data at Horseshoe Bend where the mean annual load was calculated to be 228,000 kg/yr, with 190,000 kg/yr (83.5%) coming from nonpoint sources, 25,000 kg/yr (11%) from background sources, and 12,500 kg/yr (5.5%) from point sources. The difference of 63,000 kg/yr in mean annual total phosphorus load can be explained mostly by the differing estimates in point source loads reaching Lake Tenkiller. On average, data at the eight USGS gaging stations correspond to that at Horseshoe Bend fairly well with an average 70% of the total phosphorus load coming from nonpoint sources, 7% from background sources, and 23% from point sources. Point sources contribute very little to the total phosphorus load at USGS 07194800 (2.6%), and a significant portion of the load at USGS 07195000 (60.2%).

Table XLIX summarizes the results for average annual total nitrogen

loading at USGS gaging stations and the Horseshoe Bend area of Lake Tenkiller. The simulation estimates an annual average total nitrogen load of 2,4800,000 kg/yr entering Lake Tenkiller with an interquartile range of 947,000 to 3,510,000 kg/yr. These estimates also compare very well with previous loading estimates of total nitrogen at the Horseshoe Bend location based on historical data.

Figure 29 compares simulated average annual total nitrogen loading with historical distributions of annual average total nitrogen load at USGS gaging stations. Again, the simulated mean loads overestimate average annual loading when compared to historical data.

Tables LIII through LV describe simulated distributions of nonpoint, background, and point source total nitrogen loads, respectively. Using Horseshoe Bend as an example, where the simulated mean annual average load was 2,480,000 kg/yr, 1,800,000 kg/yr (72.7%) can be attributed to nonpoint sources, 605,000 kg/yr (24.5%) to background sources, and the remaining 69,500 kg/yr (2.8%) to point sources. This compares reasonably well to the estimates at Horseshoe Bend, based on historical data, of a mean annual total nitrogen load 2,300,000 kg/yr, where 73.4 % (1,690,000 kg/yr) was from nonpoint sources, 23.9% (550,000 kg/yr) from background sources, and 2.7% (61,600 kg/yr) from point sources. On average, data at the eight gaging stations compare well with the results at Horseshoe Bend with an average 77% from nonpoint sources, 21% from background sources, and 2% from point sources.

Table LVI lists relative errors of simulated total phosphorus and total

nitrogen loads compared to historical mean loads at gaging stations and Horseshoe Bend. The greatest errors in simulated total phosphorus load are seen at USGS 07194800 (129%), a difference of 22,200 kg/yr, and USGS 07196900 (201%) with a difference of 7,950 kg/yr. Of the remaining seven sites, six were within 35% of the historical mean total phosphorus load. Simulated total nitrogen loading estimates corresponded fairly well with historical loading data with all nine sites within 30% of historical mean total nitrogen loading.

TABLE XLVIII

ESTIMATES OF ANNUAL AVERAGE TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	39,400	108,000	19	5,720	21,100	47,800	11,800,000
07195400	197,300	467,000	3,940	54,000	127,000	235,000	36,700,000
07195500	189,000	375,000	5,750	63,800	132,000	249,000	33,400,000
07196500	223,000	425,000	5,730	70,900	154,000	316,000	40,400,000
Horseshoe Bend	291,000	521,000	8,270	95,400	211,000	429,000	39,600,000
07195000	88,700	243,000	55	13,100	47,800	107,000	26,100,000
07196000	37,400	102,000	31	5,650	20,400	45,300	10,500,000
07196900	11,900	29,000	93	2,800	7,280	14,300	3,050,000
07197000	51,200	83,600	1,420	18,700	39,200	78,200	1,970,000

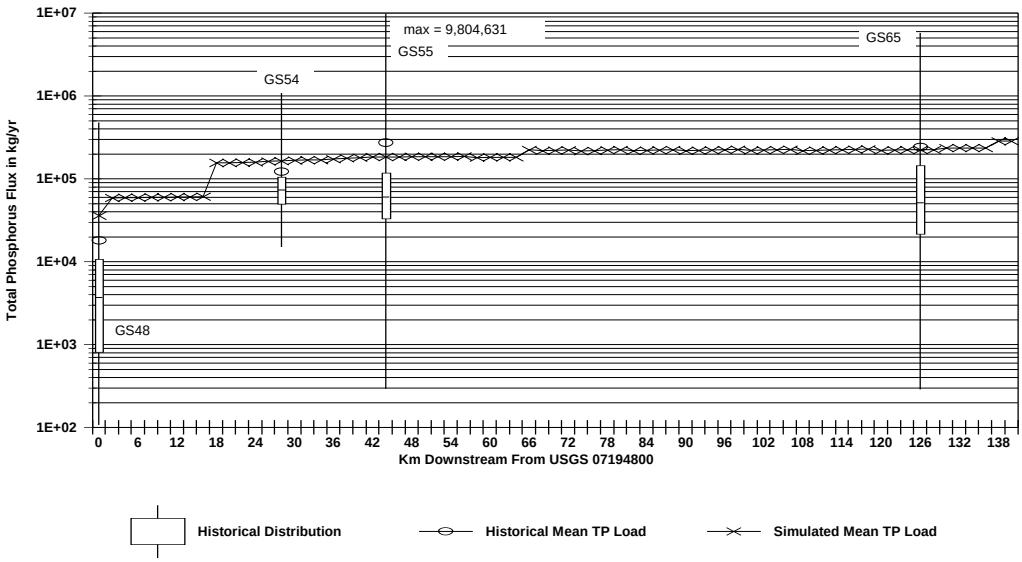

Figure 28. QUAL2EU Simulated Average Annual Total Phosphorus Load Compared to Historical Loading Distributions at Mainstem USGS Gaging Stations on the Illinois River.

TABLE XLIX

ESTIMATES OF ANNUAL AVERAGE TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	303,000	821,000	380	51,200	171,000	357,000	36,300,000
07195400	1,540,000	3,630,000	43,900	430,000	1,000,000	1,800,000	128,000,000
07195500	1,510,000	2,950,000	64,400	577,000	1,090,000	1,940,000	72,300,000
07196500	1,960,000	3,630,000	74,000	736,000	1,440,000	2,680,000	87,000,000
Horseshoe Bend	2,480,000	4,320,000	97,600	947,000	1,890,000	3,510,000	87,900,000
07195000	543,000	1,470,000	790	89,700	302,000	638,000	82,500,000
07196000	308,000	829,000	667	54,000	175,000	361,000	35,800,000
07196900	102,000	245,000	1,750	27,600	64,800	120,000	9,050,000
07197000	495,000	791,000	21,400	204,000	394,000	736,000	7,750,000

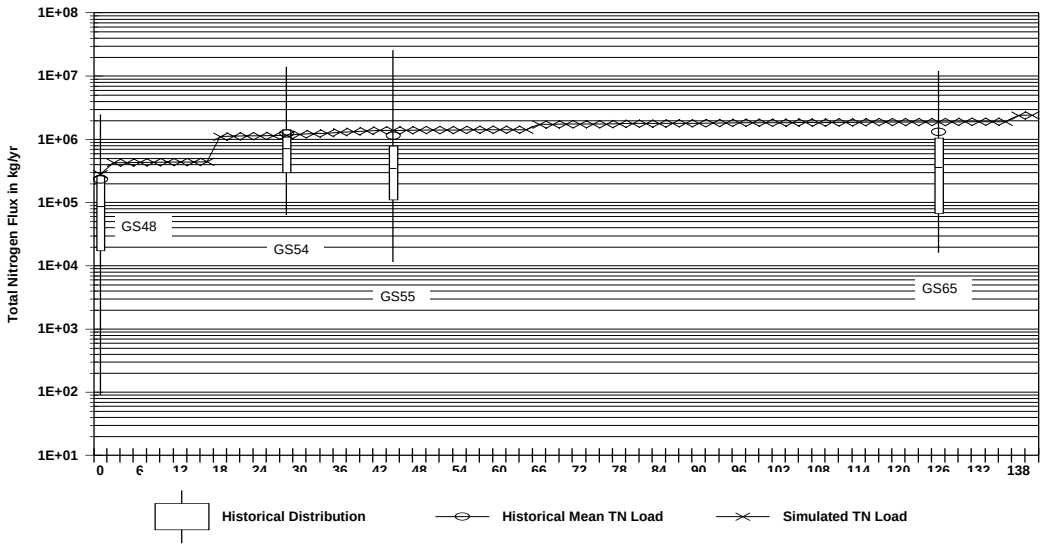

Figure 29. QUAL2EU Simulation Average Annual Total Nitrogen Load Compared to Historical Load Distributions at Mainstem USGS Gaging Stations on the Illinois River.

TABLE L

ESTIMATED AVERAGE ANNUAL NONPOINT SOURCE TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	36,300	99,400	18	5,280	19,500	43,900	10,900,000
07195400	142,000	338,000	2,020	36,300	90,700	174,000	18,200,000
07195500	133,000	262,000	2,940	45,300	93,400	176,000	16,600,000
07196500	154,000	289,000	3,990	51,200	110,000	216,000	21,300,000
Horseshoe Bend	189,000	336,000	5,670	63,100	139,000	277,000	21,600,000
07195000	30,000	83,600	18	4,370	16,100	36,600	7,680,000
07196000	24,800	67,600	19	3,690	13,400	29,800	6,480,000
07196900	9,990	24,300	74	2,350	6,110	12,100	2,440,000
07197000	39,300	64,500	1,020	14,000	29,900	60,800	1,580,000

TABLE LI

ESTIMATED AVERAGE ANNUAL BACKGROUND TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	2,110	5,790	1	297	1,120	2,570	648,000
07195400	7,740	18,400	144	2,060	4,880	9,100	1,940,000
07195500	11,100	22,100	280	3,590	7,570	14,900	1,790,000
07196500	15,900	30,300	409	4,880	10,800	23,000	2,250,000
Horseshoe Bend	21,900	39,300	613	6,780	15,400	32,500	2,260,000
07195000	4,550	12,400	3	685	2,460	5,480	1,390,000
07196000	2,540	6,920	2	388	1,380	3,070	775,000
07196900	975	2,370	8	232	597	1,170	244,000
07197000	6,270	10,300	177	2,150	4,670	9,760	178,000

TABLE LII

ESTIMATED ANNUAL AVERAGE POINT SOURCE TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	1,060	2,900	0	147	571	1,300	326,000
07195400	47,300	111,000	1,780	15,700	31,600	51,700	16,500,000
07195500	45,000	90,900	2,520	14,900	30,900	58,500	14,900,000
07196500	53,400	106,000	1,330	14,800	33,400	77,400	16,900,000
Horseshoe Bend	79,800	145,000	1,990	25,600	56,200	119,000	15,700,000
07195000	53,700	147,000	35	8,040	29,200	65,400	17,000,000
07196000	10,000	27,300	10	1,570	5,610	12,300	3,260,000
07196900	926	2,270	11	222	574	1,050	364,000
07197000	5,610	8,770	221	2,580	4,630	7,630	213,000

TABLE LIII

ESTIMATED AVERAGE ANNUAL NONPOINT SOURCE TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	251,000	680,000	315	42,500	142,000	295,000	30,000,000
07195400	1,320,000	3,120,000	34,800	364,000	857,000	1,560,000	112,000,000
07195500	1,210,000	2,360,000	50,300	464,000	877,000	1,550,000	57,600,000
07196500	1,480,000	2,750,000	57,200	561,000	1,090,000	2,030,000	64,600,000
Horseshoe Bend	1,800,000	3,140,000	71,100	689,000	1,380,000	2,540,000	62,300,000
07195000	415,000	1,130,000	582	67,800	230,000	489,000	68,400,000
07196000	237,000	637,000	509	41,600	135,000	277,000	27,200,000
07196900	75,700	182,000	1,320	20,500	48,100	88,300	6,830,000
07197000	320,000	511,000	13,500	132,000	256,000	478,000	4,830,000

TABLE LIV

ESTIMATED AVERAGE ANNUAL BACKGROUND TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	52,000	141,000	65	8,740	29,300	61,300	6,300,000
07195400	199,000	468,000	8,250	60,100	130,000	228,000	15,300,000
07195500	285,000	558,000	13,000	107,000	204,000	369,000	14,500,000
07196500	442,000	821,000	16,000	163,000	323,000	613,000	20,400,000
Horseshoe Bend	605,000	1,060,000	23,100	224,000	455,000	868,000	23,100,000
07195000	112,000	304,000	185	19,200	63,600	132,000	13,500,000
07196000	62,800	169,000	140	10,900	35,700	73,900	7,470,000
07196900	24,100	58,000	423	6,600	15,400	28,300	1,950,000
07197000	161,000	259,000	7,250	65,300	127,000	238,000	2,890,000

TABLE LV

ESTIMATED AVERAGE ANNUAL POINT SOURCE TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	0	0	0	0	0	0	0
07195400	18,300	42,500	816	6,060	13,700	18,000	659,000
07195500	15,800	30,600	1,050	6,050	11,900	23,900	240,000
07196500	31,800	58,900	1,020	12,500	25,500	34,600	2,040,000
Horseshoe Bend	69,500	120,000	3,370	33,200	56,200	102,000	2,460,000
07195000	15,200	40,900	23	2,640	8,610	17,300	513,000
07196000	8,390	22,600	18	1,450	4,830	9,780	1,090,000
07196900	2,150	5,210	8	506	1,270	3,000	269,000
07197000	12,900	20,200	619	6,020	11,100	19,700	37,500

TABLE LVI

RELATIVE ERRORS OF SIMULATED TOTAL PHOSPHORUS AND TOTAL NITROGEN LOADS COMPARED TO HISTORICAL DATA

USGS Gaging Station Identification	Historical Mean Annual Total Phosphorus Load (kg/yr)	Simulated Mean Annual Total Phosphorus Load (kg/yr)	Total Phosphorus Load Relative Error (%)	Historical Mean Annual Total Nitrogen Load (kg/yr)	Simulated Mean Annual Total Nitrogen Load (kg/yr)	Total Nitrogen Load Relative Error (%)
07194800	17,200	39,400	129	289,000	303,000	5
07195400	122,000	197,000	61	1,530,000	1,540,000	1
07195500	272,000	189,000	-31	1,280,000	1,510,000	18
07196500	241,000	223,000	-7	1,590,000	1,960,000	23
Horseshoe Bend	228,000	291,000	28	2,300,000	2,480,000	8
07195000	72,300	88,700	23	581,000	543,000	-7
07196000	48,200	37,400	-22	237,000	308,000	30
07196900	3,950	11,900	201	82,400	102,000	24
07197000	76,800	51,200	-33	417,000	495,000	19

Nonpoint sources were reduced by 25 and 50 percent to estimate the effects of this reduction on nutrient concentration and load at USGS gaging stations in the basin and the Horseshoe Bend area of Lake Tenkiller. Tables LVII - LXII describe statistically the estimated distributions of concentration and load for average annual conditions with a 25% decrease in nonpoint source loading. Only average annual and the nonpoint source data are given since the background and point source contributions were assumed to remain the same. The simulated annual mean total phosphorus concentration of 0.19 mg/l at

Horseshoe Bend with a 25% reduction of NPS input compares with 0.23 mg/l under annual average conditions, a 17% decrease (Table LXIII). This translates into a mean load reduction of 50,000 kg/yr. It is estimated that mean total nitrogen load at Horseshoe Bend could be reduced 21% or 510,000 kg/yr with a 25% reduction of NPS inputs. The estimates at the gaging stations showed similar reduction with an average 19% decrease in total phosphorus concentration and 18% decrease in load. Total nitrogen concentrations and loads at the eight gaging stations were reduced an average 21% (Table LXIV).

Tables LXV through LXX show the estimated distributions of nutrient concentrations and loads with a 50% reduction of NPS inputs into the model. The mean total phosphorus load at Horseshoe Bend would be reduced by an estimated 102,000 kg/yr (35%), and the estimated mean total nitrogen load reduced by 1,020,000 kg/yr (41%). Over all nine sites, a 50% reduction in nonpoint source inputs reduced total phosphorus concentration and load by 36% (Table LXXI). For total nitrogen, concentration was reduced by 43%, and load was reduced by 42% (Table LXXII).

TABLE LVII

ESTIMATED AVERAGE ANNUAL TOTAL NITROGEN CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH 25% REDUCTION OF NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	2.2	0.9	0.6	1.6	2.1	2.7	7.4
07195400	2.9	1.0	0.7	2.2	2.8	3.4	8.2
07195500	2.0	0.6	0.7	1.6	1.9	2.3	4.7
07196500	1.7	0.5	0.6	1.3	1.6	1.9	4.2
Horseshoe Bend	1.5	0.5	0.5	1.2	1.5	1.8	3.8
07195000	1.9	0.8	0.5	1.3	1.7	2.3	7.7
07196000	1.9	0.7	0.6	1.4	1.8	2.3	6.3
07196900	1.7	0.6	0.5	1.3	1.6	2.0	4.9
07197000	1.2	0.4	0.4	0.9	1.2	1.4	3.2

TABLE LVIII

ESTIMATED AVERAGE ANNUAL TOTAL PHOSPHORUS CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 25% REDUCTION OF NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.29	0.16	0.03	0.18	0.25	0.35	2.39
07195400	0.40	0.16	0.07	0.29	0.37	0.47	2.67
07195500	0.26	0.13	0.07	0.18	0.24	0.31	2.42
07196500	0.20	0.11	0.04	0.13	0.17	0.23	2.11
Horseshoe Bend	0.19	0.10	0.04	0.12	0.17	0.22	1.88
07195000	0.36	0.20	0.04	0.22	0.31	0.44	2.96
07196000	0.24	0.14	0.03	0.15	0.22	0.30	1.95
07196900	0.19	0.10	0.03	0.12	0.17	0.23	1.65
07197000	0.12	0.06	0.03	0.08	0.11	0.15	0.79

TABLE LIX

ESTIMATED AVERAGE ANNUAL TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 25% REDUCTION OF NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	234,000	634,000	293	39,500	132,000	276,000	28,100,000
07195400	1,180,000	2,800,000	34,500	334,000	770,000	1,380,000	97,600,000
07195500	1,180,000	2,320,000	50,600	453,000	857,000	1,520,000	57,000,000
07196500	1,550,000	2,870,000	58,000	580,000	1,140,000	2,120,000	68,800,000
Horseshoe Bend	1,970,000	3,440,000	77,500	753,000	1,510,000	2,790,000	69,500,000
07195000	433,000	1,170,000	634	72,000	242,000	511,000	62,700,000
07196000	243,000	655,000	528	42,700	139,000	286,000	28,400,000
07196900	81,000	195,000	1,400	21,900	51,400	95,000	7,180,000
07197000	404,000	646,000	17,600	166,000	322,000	599,000	6,370,000

TABLE LX

ESTIMATED AVERAGE ANNUAL TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 25% REDUCTION OF NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	30,200	82,600	15	4,390	16,200	36,600	9,070,000
07195400	161,000	381,000	3,410	44,500	103,000	190,000	32,000,000
07195500	154,000	308,000	4,840	51,200	106,000	203,000	29,100,000
07196500	183,000	350,000	4,400	56,900	125,000	258,000	34,900,000
Horseshoe Bend	241,000	432,000	6,590	78,300	174,000	354,000	34,000,000
07195000	80,800	221,000	51	11,900	43,400	98,000	24,100,000
07196000	31,000	84,500	26	4,700	16,900	37,600	8,860,000
07196900	9,310	22,700	71	2,180	5,690	11,200	2,420,000
07197000	40,900	66,900	1,150	15,000	31,500	62,500	1,560,000

TABLE LXI

ESTIMATED AVERAGE ANNUAL NONPOINT SOURCE TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 25% REDUCTION OF NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	182,000	494,000	227	30,800	103,000	214,000	21,800,000
07195400	964,000	2,280,000	25,400	268,000	626,000	1,140,000	81,600,000
07195500	884,000	1,730,000	36,600	340,000	641,000	1,130,000	42,200,000
07196500	1,070,000	1,990,000	41,200	404,000	788,000	1,470,000	46,400,000
Horseshoe Bend	1,300,000	2,260,000	51,000	495,000	996,000	1,820,000	43,900,000
07195000	306,000	829,000	426	50,100	170,000	362,000	48,700,000
07196000	172,000	463,000	370	30,300	98,200	202,000	19,900,000
07196900	54,700	132,000	968	14,800	34,700	63,500	4,960,000
07197000	230,000	367,000	9,690	95,200	184,000	342,000	3,450,000

TABLE LXII

ESTIMATED AVERAGE ANNUAL NONPOINT SOURCE TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 25% REDUCTION OF NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	27,000	74,000	14	3,950	14,500	32,700	8,090,000
07195400	106,000	252,000	1,490	26,700	67,000	129,000	13,500,000
07195500	98,100	195,000	2,030	32,700	67,700	130,000	12,400,000
07196500	113,000	214,000	2,660	37,200	81,000	157,000	15,800,000
Horseshoe Bend	139,000	248,000	3,980	46,000	102,000	203,000	16,100,000
07195000	22,500	61,900	14	3,140	11,700	27,200	5,670,000
07196000	18,400	50,300	14	2,740	9,900	22,200	4,820,000
07196900	7,410	18,100	52	1,720	4,520	9,000	1,810,000
07197000	29,100	47,800	752	10,300	22,200	45,100	1,170,000

TABLE LXIII

RELATIVE REDUCTION IN MEAN ANNUAL TOTAL PHOSPHORUS CONCENTRATION AND LOAD WITH A 25% REDUCTION IN NONPOINT SOURCE INPUTS

USGS Gaging Station Identification	Simulated Mean Annual Total Phosphorus Conc. (mg/l)	Simulated Mean Annual Total Phosphorus Conc. With 25% NPS Reduction (mg/l)	Change (%)	Simulated Mean Annual Total Phosphorus Load (kg/yr)	Simulated Mean Annual Total Phosphorus Load With 25% NPS Reduction (kg/yr)	Change (%)
07194800	0.37	0.29	-22	39,400	30,200	-23
07195400	0.48	0.40	-17	197,000	161,000	-18
07195500	0.32	0.26	-19	189,000	154,000	-19
07196500	0.24	0.20	-17	223,000	183,000	-18
Horseshoe Bend	0.23	0.19	-17	291,000	241,000	-17
07195000	0.39	0.36	-17	88,700	80,800	-9
07196000	0.29	0.24	-17	37,400	31,000	-17
07196900	0.24	0.19	-21	11,900	9,310	-22
07197000	0.16	0.12	-25	51,200	40,900	-20

TABLE LXIV

RELATIVE REDUCTION IN MEAN ANNUAL TOTAL NITROGEN CONCENTRATION AND LOAD WITH A 25% REDUCTION IN NONPOINT SOURCE INPUTS

USGS Gaging Station Identification	Simulated Mean Annual Total Nitrogen Conc. (mg/l)	Simulated Mean Annual Total Nitrogen Conc. With 25% NPS Reduction (mg/l)	Change (%)	Simulated Mean Annual Total Nitrogen Load (kg/yr)	Simulated Mean Annual Total Nitrogen Load With 25% NPS Reduction (kg/yr)	Change (%)
07194800	2.9	2.2	-24	303,000	234,000	-23
07195400	3.8	2.9	-24	1,540,000	1,180,000	-23
07195500	2.6	2.0	-23	1,510,000	1,180,000	-22
07196500	2.1	1.7	-19	1,960,000	1,550,000	-21
Horseshoe Bend	1.9	1.5	-21	2,480,000	1,970,000	-21
07195000	2.4	1.9	-21	543,000	433,000	-20
07196000	2.4	1.9	-21	308,000	243,000	-21
07196900	2.1	1.7	-19	102,000	81,000	-21
07197000	1.5	1.2	-20	495,000	404,000	-18

TABLE LXV

ESTIMATED AVERAGE ANNUAL TOTAL NITROGEN CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	1.6	0.5	0.6	1.1	1.4	1.9	5.2
07195400	2.0	0.5	0.7	1.5	1.9	2.4	5.6
07195500	1.5	0.5	0.4	1.2	1.4	1.7	3.5
07196500	1.2	0.4	0.4	0.9	1.2	1.4	3.0
Horseshoe Bend	1.1	0.4	0.4	0.9	1.1	1.3	2.8
07195000	1.4	0.4	0.6	1.0	1.3	1.7	5.3
07196000	1.4	0.4	0.5	1.0	1.3	1.7	4.6
07196900	1.2	0.4	0.4	0.9	1.2	1.4	3.6
07197000	0.9	0.3	0.3	0.7	0.9	1.1	2.5

TABLE LXVI

ESTIMATED AVERAGE ANNUAL TOTAL PHOSPHORUS CONCENTRATIONS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (mg/l)	Standard Deviation (mg/l)	Minimum (mg/l)	25th Percentile (mg/l)	50th Percentile (mg/l)	75th Percentile (mg/l)	Maximum (mg/l)
07194800	0.20	0.11	0.02	0.12	0.17	0.24	1.66
07195400	0.30	0.13	0.06	0.22	0.28	0.36	2.27
07195500	0.21	0.11	0.05	0.14	0.18	0.24	2.06
07196500	0.15	0.09	0.03	0.10	0.13	0.18	1.77
Horseshoe Bend	0.15	0.08	0.03	0.10	0.13	0.18	1.56
07195000	0.32	0.18	0.04	0.20	0.28	0.39	2.70
07196000	0.19	0.11	0.02	0.12	0.17	0.24	1.58
07196900	0.14	0.07	0.02	0.09	0.12	0.17	1.21
07197000	0.09	0.04	0.02	0.06	0.08	0.11	0.58

TABLE LXVII

ESTIMATED AVERAGE ANNUAL TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	165,000	448,000	207	27,900	93,300	194,000	19,800,000
07195400	822,000	1,940,000	25,000	23,600	538,000	959,000	67,100,000
07195500	854,000	1,670,000	36,800	325,000	616,000	1,100,000	41,500,000
07196500	1,130,000	2,100,000	42,000	422,000	830,000	1,560,000	50,200,000
Horseshoe Bend	1,460,000	2,550,000	57,300	555,000	1,110,000	2,080,000	50,800,000
07195000	321,000	870,000	482	53,900	180,000	380,000	42,800,000
07196000	177,000	478,000	388	31,100	101,000	208,000	20,900,000
07196900	59,700	143,000	1,040	16,200	37,900	69,700	5,270,000
07197000	312,000	499,000	13,600	129,000	248,000	461,000	5,000,000

TABLE LXVIII

ESTIMATED AVERAGE ANNUAL TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	20,900	57,200	10	3,030	11,200	25,300	6,290,000
07195400	124,000	294,000	2,830	34,300	79,400	145,000	27,200,000
07195500	112,000	240,000	3,650	38,700	81,300	157,000	24,700,000
07196500	142,000	275,000	3,070	43,100	96,600	202,000	29,300,000
Horseshoe Bend	189,000	342,000	4,900	60,500	136,000	279,000	28,300,000
07195000	72,600	199,000	46	10,600	39,100	88,200	22,000,000
07196000	24,500	66,800	20	3,710	13,300	29,700	7,140,000
07196900	6,730	16,400	52	1,570	4,110	8,110	1,780,000
07197000	30,400	49,600	884	11,200	23,600	46,300	1,150,000

TABLE LXIX

ESTIMATED AVERAGE ANNUAL NONPOINT SOURCE TOTAL NITROGEN LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	113,000	307,000	142	19,200	64,000	133,000	13,600,000
07195400	605,000	1,430,000	15,900	170,000	394,000	713,000	51,100,000
07195500	553,000	1,080,000	22,800	213,000	400,000	704,000	26,700,000
07196500	659,000	1,220,000	25,200	247,000	482,000	912,000	27,800,000
Horseshoe Bend	789,000	1,370,000	30,800	298,000	603,000	1,110,000	25,300,000
07195000	194,000	525,000	274	32,000	108,000	231,000	28,800,000
07196000	106,000	286,000	230	18,700	60,600	125,000	12,400,000
07196900	33,400	80,300	611	9,080	21,200	38,500	3,050,000
07197000	138,000	220,000	5,750	57,400	111,000	203,000	2,080,000

TABLE LXX

ESTIMATED AVERAGE ANNUAL NONPOINT SOURCE TOTAL PHOSPHORUS LOADS AT LOCATIONS CORRESPONDING TO USGS GAGING STATIONS AND HORSESHOE BEND WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS (N=2000)

USGS Gaging Station Identification	Mean (kg/yr)	Standard Deviation (kg/yr)	Minimum (kg/yr)	25th Percentile (kg/yr)	50th Percentile (kg/yr)	75th Percentile (kg/yr)	Maximum (kg/yr)
07194800	17,700	48,500	9	2,590	9,520	21,500	5,310,000
07195400	68,900	165,000	912	16,500	42,900	84,400	8,730,000
07195500	63,700	127,000	841	20,200	42,800	83,900	7,990,000
07196500	73,000	139,000	1,330	23,400	52,400	102,000	10,200,000
Horseshoe Bend	87,500	157,000	2,300	28,100	63,900	128,000	10,400,000
07195000	14,300	39,600	9	1,920	7,360	17,300	3,590,000
07196000	11,900	32,600	9	1,750	6,330	14,300	3,100,000
07196900	4,830	11,800	33	1,120	2,930	5,890	1,170,000
07197000	18,500	30,400	486	6,510	14,200	28,900	754,000

TABLE LXXI

RELATIVE REDUCTION IN MEAN ANNUAL TOTAL PHOSPHORUS CONCENTRATION AND LOAD WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS

USGS Gaging Station Identification	Simulated Mean Annual Total Phosphorus Conc. (mg/l)	Simulated Mean Annual Total Phosphorus Conc. With 50% NPS Reduction (mg/l)	Change (%)	Simulated Mean Annual Total Phosphorus Load (kg/yr)	Simulated Mean Annual Total Phosphorus Load With 50% NPS Reduction (kg/yr)	Change (%)
07194800	0.37	0.20	-46	39,400	20,900	-47
07195400	0.48	0.30	-38	197,000	124,000	-37
07195500	0.32	0.21	-34	189,000	120,000	-37
07196500	0.24	0.15	-38	223,000	142,000	-36
Horseshoe Bend	0.23	0.15	-35	291,000	189,000	-35
07195000	0.39	0.32	-18	88,700	72,600	-18
07196000	0.29	0.19	-34	37,400	24,500	-34
07196900	0.24	0.14	-42	11,900	6,730	-43
07197000	0.16	0.09	-44	51,200	30,400	-41

TABLE LXXII

RELATIVE REDUCTION IN MEAN ANNUAL TOTAL NITROGEN CONCENTRATION AND LOAD WITH A 50% REDUCTION IN NONPOINT SOURCE INPUTS

USGS Gaging Station Identification	Simulated Mean Annual Total Nitrogen Conc. (mg/l)	Simulated Mean Annual Total Nitrogen Conc. With 50% NPS Reduction (mg/l)	Change (%)	Simulated Mean Annual Total Nitrogen Load (kg/yr)	Simulated Mean Annual Total Nitrogen Load With 50% NPS Reduction (kg/yr)	Change (%)
07194800	2.9	1.6	-45	303,000	165,000	-46
07195400	3.8	2.0	-47	1,540,000	822,000	-47
07195500	2.6	1.5	-42	1,510,000	854,000	-43
07196500	2.1	1.2	-43	1,960,000	1,130,000	-42
Horseshoe Bend	1.9	1.1	-42	2,480,000	1,460,000	-41
07195000	2.4	1.4	-42	543,000	321,000	-41
07196000	2.4	1.4	-42	308,000	177,000	-43
07196900	2.1	1.2	-43	102,000	59,700	-41
07197000	1.5	0.9	-40	495,000	312,000	-37

CHAPTER VI

DISCUSSION

The focus of the study was on the quantities of nutrients delivered to the upper portion of Lake Tenkiller at the Horseshoe Bend area and the sources of those nutrients. The modeling scheme allowed for the estimation of frequency distributions of average annual nutrient concentrations and loadings at this point, as well as, estimates of contributions by point, nonpoint, and background sources.

The SIMPLE model provided estimates of runoff generated nonpoint source phosphorus delivered to the stream. Based on the SIMPLE model, areas with high soil phosphorus levels, due to long-term application of chicken litter, were shown to be the greatest contributors to nonpoint phosphorus at the Horseshoe Bend Area. Figure 30 shows estimated annual average total phosphorus loading from the subbasins above Lake Tenkiller based on the SIMPLE model. Subbasins OSAGE, FORK, BARON, FLINT, BENTON, and CLEAR were shown to deliver the greatest quantities. The SIMPLE model did not predict nonpoint nitrogen export from the subbasins. Unit-area loading methods were applied to estimate average annual total nitrogen export from each of the subbasins. Figure 31 indicates that total nitrogen export from the

subbasins above Lake Tenkiller is dominated by seven subbasins (OSAGE, FORK, BARON, BENTON, BORD, and FLINT).

The QUAL2EU model used SIMPLE phosphorus export estimates, and unit area total nitrogen loading estimates from each of the subbasins and routed them downstream to Lake Tenkiller. The following pie graphs describe the estimated mean relative distributions of sources of nutrient concentrations at Horseshoe Bend under annual average, and 25% and 50% reductions of nonpoint source inputs (Figures 32 - 37) based on the QUAL2EU model. For obvious reasons, as nonpoint source contributions are decreased, point and background contributions increase.

Nonpoint contributions to total phosphorus and total nitrogen concentration are estimated to be 60.9% and 73.1%, respectively (Figures 32 and 37). With a 25% reduction of nonpoint source inputs, these percentages drop to 52.6% and 66% of total phosphorus and total nitrogen concentrations, respectively (Figures 34 and 35). A 50% reduction of nonpoint source total phosphorus and total nitrogen indicates a potential decrease in nonpoint contributions to concentration at Horseshoe Bend down to 40% and 54% of total phosphorus and total nitrogen, respectively (Figures 36 and 37). Total phosphorus point source contributions are estimated to contribute 30.4% to total phosphorus concentration under average annual conditions, and this percentage increases to 46.7% with a 50% decrease in nonpoint sources. Point sources are estimated to contribute 2.6% of the total nitrogen concentration under average

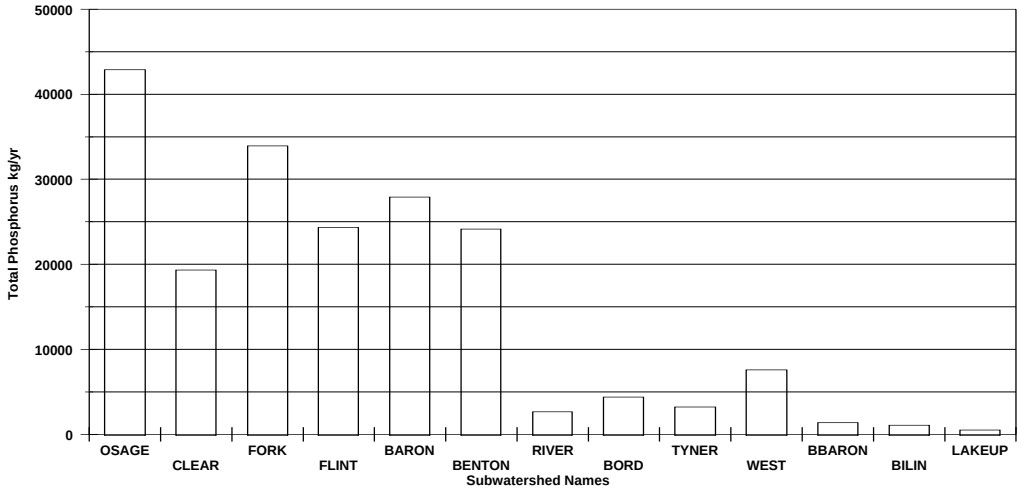

Figure 30. SIMPLE Estimated Average Annual Total Phosphorus Loads From Subbasins Above the Horseshoe Bend Area of Lake Tenkiller.

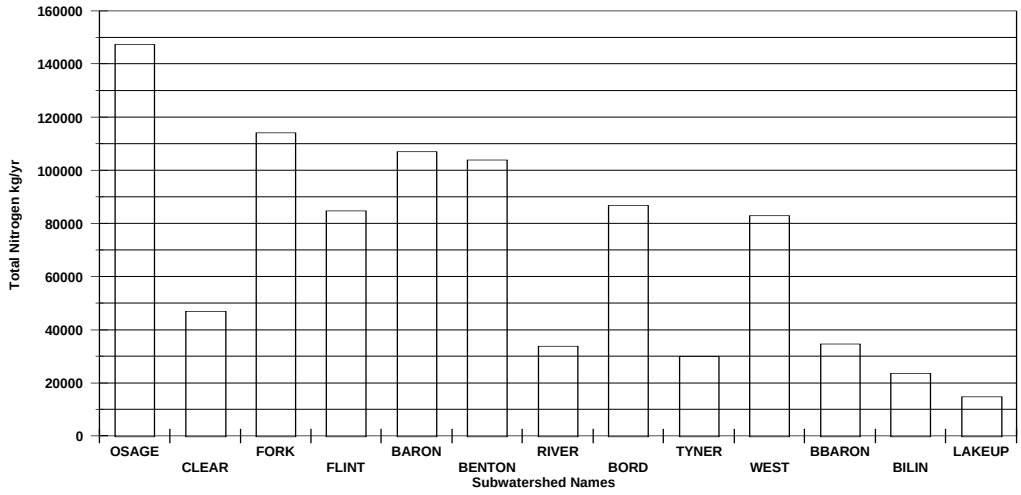

Figure 31. Unit Area Loading Estimates of Average Annual Total Nitrogen Loads From Subbasins Above the Horseshoe Bend Area of Lake Tenkiller.

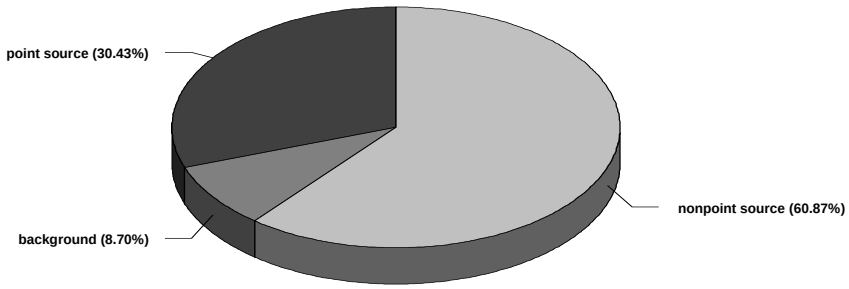

Figure 32. Estimates of Percent Contributions of Point, Nonpoint, and Background Sources to Average Annual Total Phosphorus Concentration at Horseshoe Bend Under Average Annual Conditions.

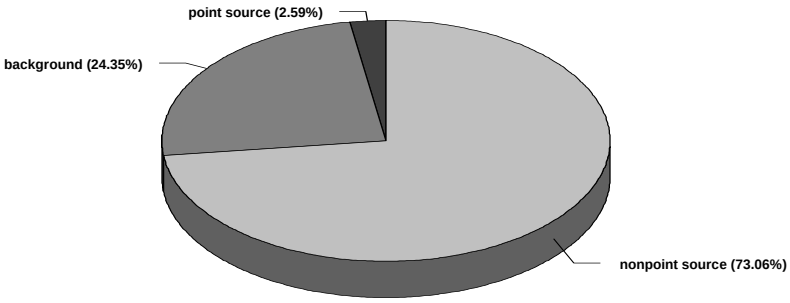

Figure 33. Estimates of Percent Contributions of Point, Nonpoint, and Background Sources to Average Annual Total Nitrogen Concentration at Horseshoe Bend Under Average Annual Conditions.

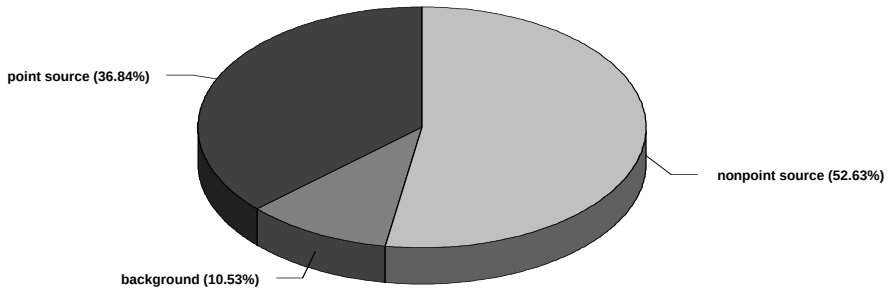

Figure 34. Estimates of the Percent Contributions of Point, Nonpoint, and Background Sources to Total Phosphorus Concentration at Horseshoe Bend with a 25% Reduction of Nonpoint Source Total Phosphorus.

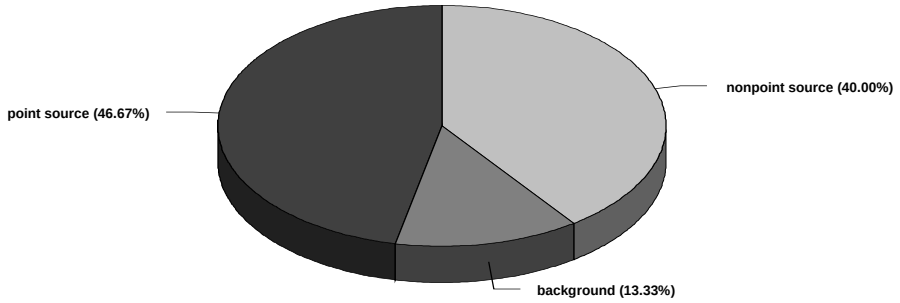

Figure 35. Estimates of the Percent Contributions of Point, Nonpoint, and Background Sources to Total Phosphorus Concentration at Horseshoe Bend with a 50% Reduction of Nonpoint Source Total Phosphorus.

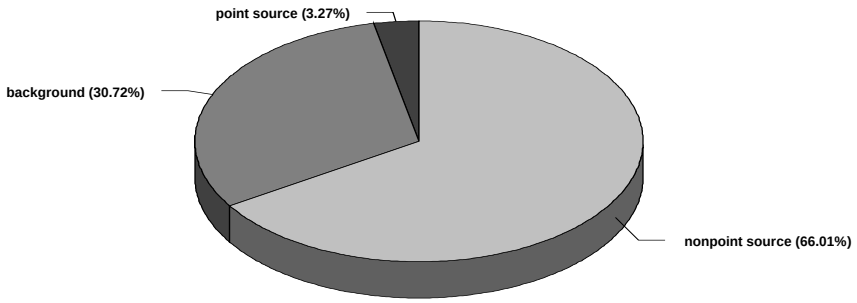

Figure 36. Estimates of the Percent Contributions of Point, Nonpoint, and Background Sources to Total Nitrogen Concentration at Horseshoe Bend with a 25% Reduction of Nonpoint Source Total Nitrogen.

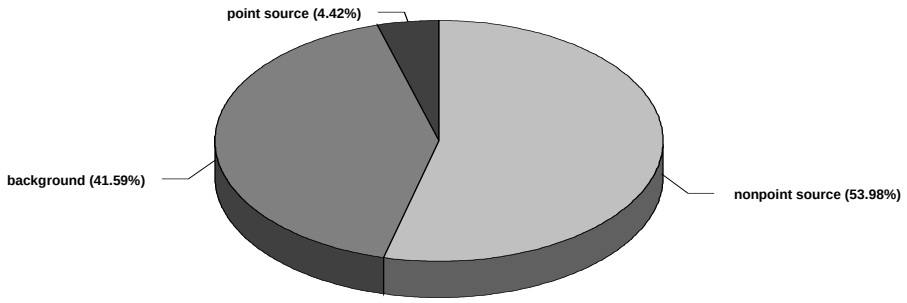

Figure 37. Estimates of the Percent Contributions of Point, Nonpoint, and Background Sources to Total Nitrogen Concentration at Horseshoe Bend with a 50% Reduction of Nonpoint Source Total Nitrogen.

annual conditions, and this increases to 4.4% with a 50% decrease in nonpoint source inputs.

Pie graphs (Figures 38 through 43) were prepared to depict the contributions of different sources of nutrient loads at gaging stations on the mainstem of the Illinois River and Horseshoe Bend under average annual conditions and with reductions of nonpoint source inputs. Figure 38 shows that under average annual conditions, in the upper part of the Illinois River Basin (USGS 07194800) nonpoint sources of total phosphorus dominate total load. USGS 07195400, 07195500, and 07196500 all show relatively similar contributions of nonpoint sources to total phosphorus load at close to 70%. Horseshoe Bend shows the effects of Tahlequah point source influence and the confluence of the Baron Fork Creek with the Illinois River, in that the nonpoint contribution is reduced to nearly 65% while the point source contribution increases.

Figure 39 shows the same type of information for total nitrogen loading. The upper reaches of the basin, represented by USGS 07194800, show little influence of point sources with nonpoint sources comprising over 80% of the total load. Along the mainstem of the Illinois River represented by USGS 07195400, 07195500, and 07196500, point sources contribute just over 1% of total load while nonpoint sources contribute near 80% of the total load. Again, the Horseshoe Bend area of Lake Tenkiller shows the influence of Tahlequah's point source input and the confluence of the Baron Fork Creek with the Illinois River. Figures 40 and 41 show estimates of contributions of point, nonpoint, and

background sources to total phosphorus loads with reductions in nonpoint source inputs. With a 25% reduction in nonpoint source inputs, point source contributions to total phosphorus loads increase about 5% while nonpoint source contributions decrease 6 to 7%. Fifty percent reductions in nonpoint source inputs result in even greater shifts with point sources contributing over 35% of the total phosphorus load at stations 07195400, 07195500, and 07196500, and over 40% at Horseshoe Bend.

A similar analysis of total nitrogen loads with reductions in nonpoint source inputs (Figures 42 and 43) show that point sources of total nitrogen contribute significantly less to total nitrogen loads. Significant changes are only seen in the relative contributions of background and nonpoint sources. As nonpoint source inputs are decreased, background contribution increase.

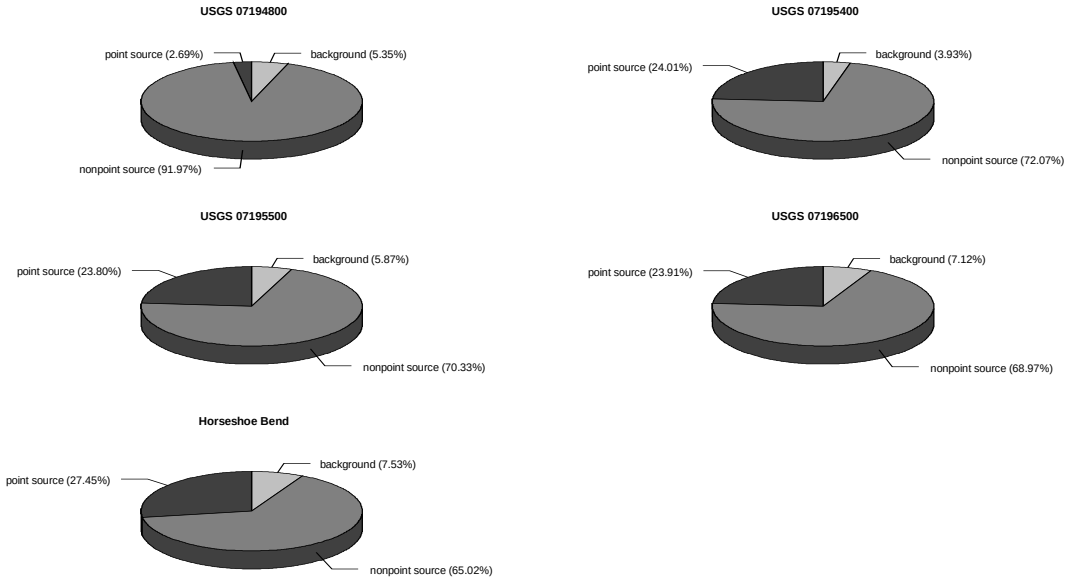

Figure 38. Estimates of Contributions of Nonpoint, Point, and Background Sources to Total Phosphorus Load at Mainstem USGS Gaging Stations and Horseshoe Bend Under Average Annual Conditions.

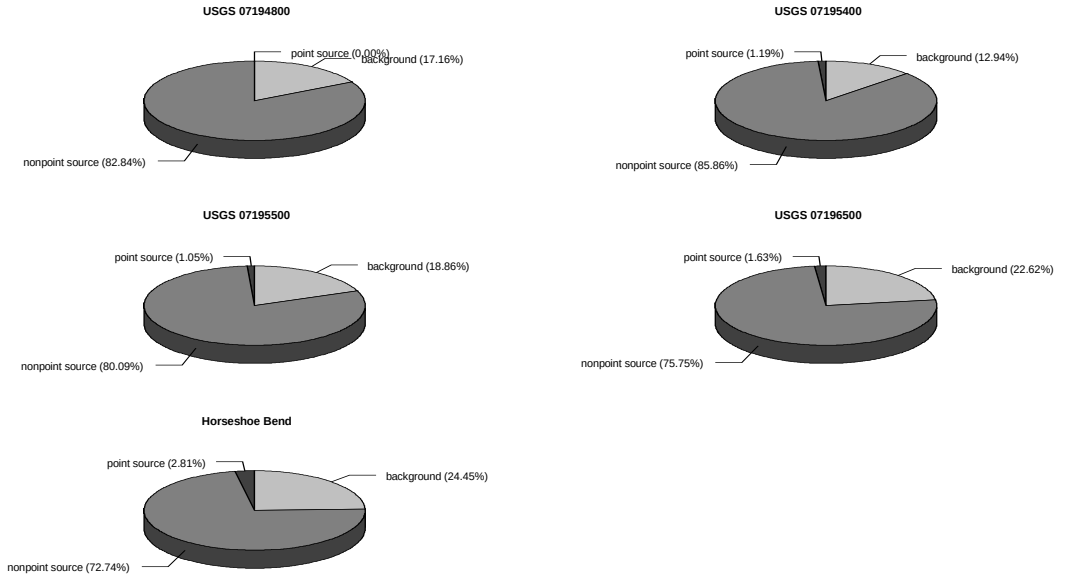

Figure 39. Contributions of Nonpoint, Point, and Background Sources to Total Nitrogen Load at Mainstem USGS Gaging Stations and Horseshoe Bend Under Average Annual Conditions.

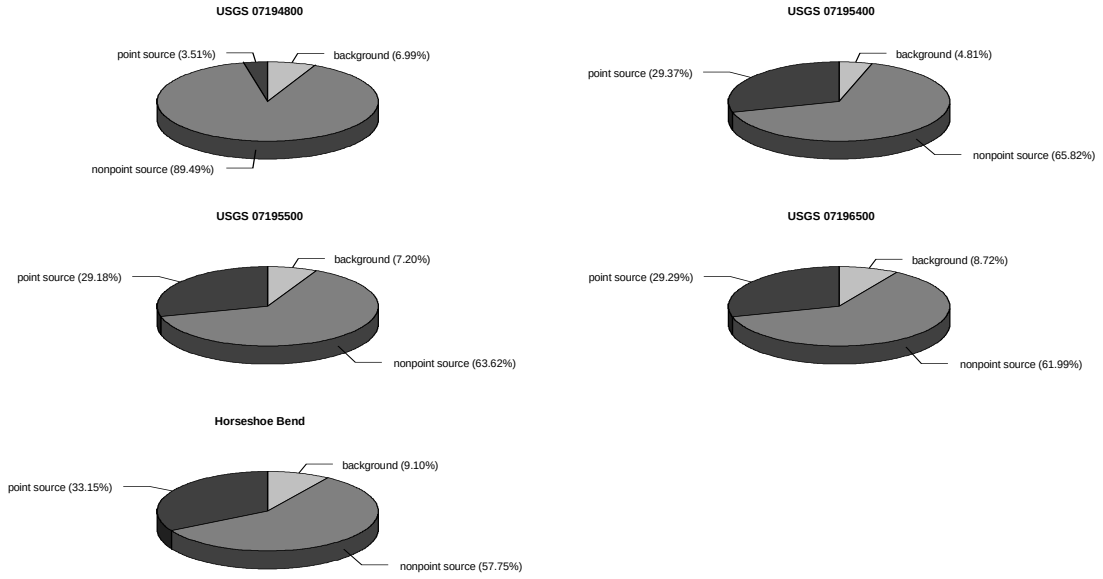

Figure 40. Estimates of Contributions of Point, Nonpoint, and Background Sources to Total Phosphorus Loads at Mainstem USGS Gaging Stations and Horseshoe Bend with a 25% Reduction of Nonpoint Source Inputs.

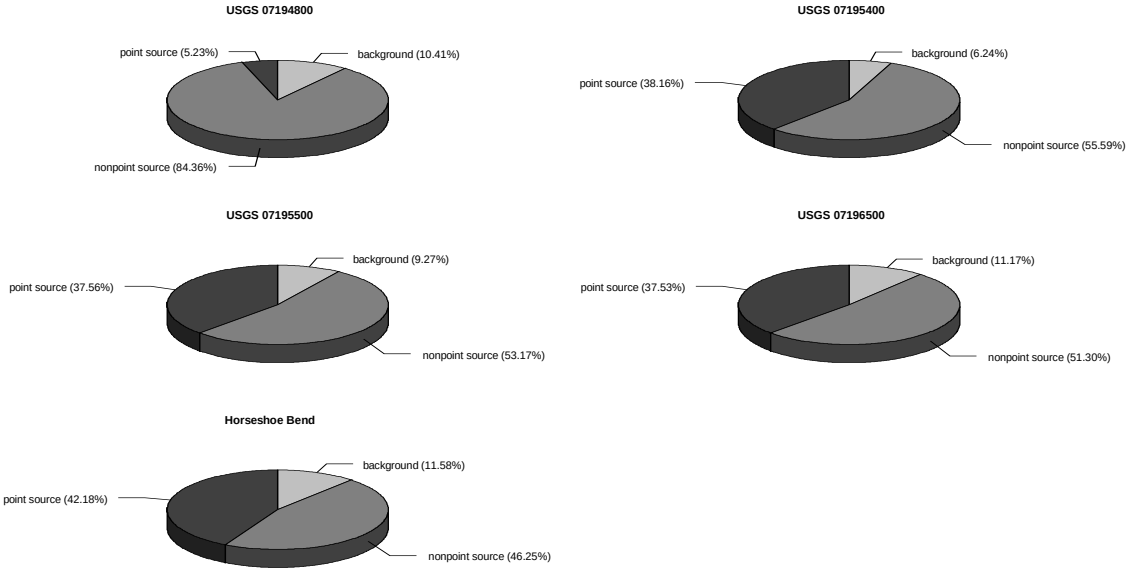

Figure 41. Estimates of Contributions of Point, Nonpoint, and Background Sources to Total Phosphorus Loads at Mainstem USGS Gaging Stations and Horseshoe Bend with a 50% Reduction of Nonpoint Source Inputs.

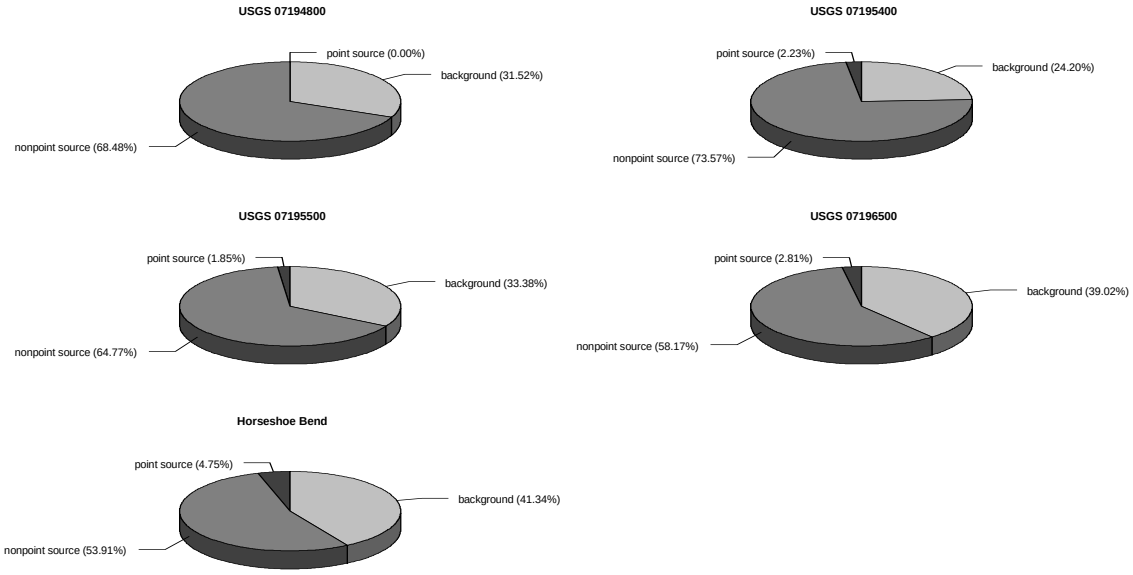

Figure 42. Estimates of Contributions of Point, Nonpoint, and Background Sources to Total Nitrogen Loads at Mainstem USGS Gaging Stations and Horseshoe Bend with a 50% Reduction of Nonpoint Source Inputs.

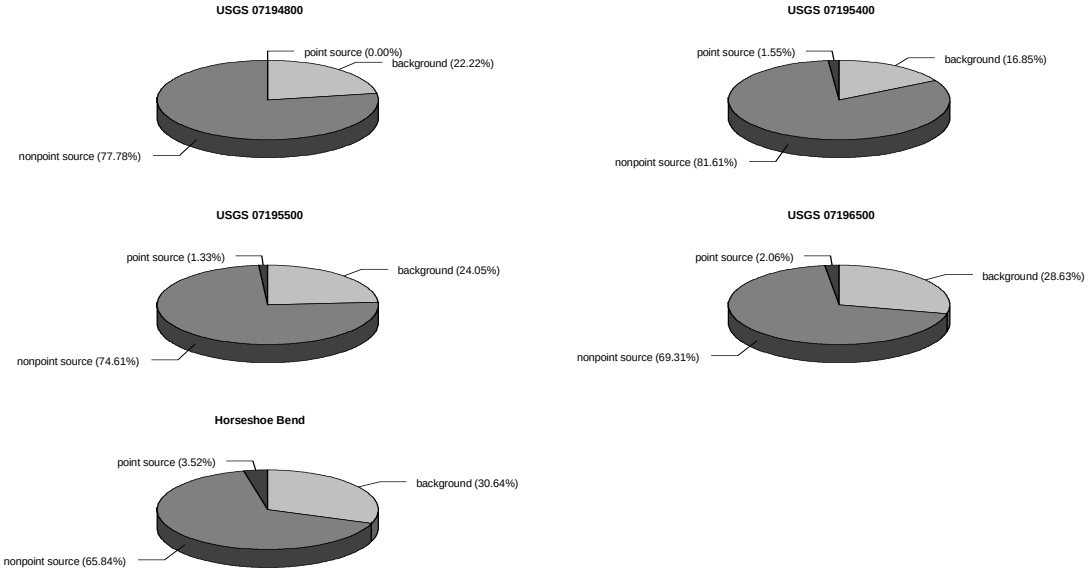

Figure 43. Estimates of Contributions of Point, Nonpoint, and Background Sources to Total Nitrogen Loads at Mainstem USGS Gaging Stations and Horseshoe Bend with a 25% Reduction of Nonpoint Source Inputs.

Vollenweider (1976, in Wetzel, 1983) and others have attempted to relate phosphorus loading in freshwater lakes to lake trophic status. Lakes with mean total phosphorus concentrations around 0.01 mg/l would be classified as oligotrophic, and concentrations around 0.1 mg/l would lead to a classification of potentially eutrophic. The critical concentrations vary with respect to hydraulic residence time and mean lake depth. Lakes with larger volumes and shorter residence times are be able to assimilate greater phosphorus loads without showing the effects of eutrophication. Based on the table of suggested loadings given earlier in the section on phosphorus (Table II), distributions of simulated total phosphorus loads under annual average, 25% and 50% reduction of nonpoint sources were plotted using Lake Tenkiller's mean depth and hydraulic residence time (Figure 44). All cases, including the absence of all nonpoint source loadings, would suggest that enough phosphorus in entering Lake Tenkiller to exacerbate eutrophic conditions. It is important to note that the relationship was developed for natural lakes and that reservoirs tend to act differently in that they have riverine attributes in the upper portions (Thornton, et al., 1990).

Wetzel (1983) developed a similar relationship for total nitrogen based on the work of Vollenweider and others. Using a similar graphing technique, Figure 45 depicts the results of total nitrogen loading to lake Tenkiller under simulated annual average, 25% reduction of NPS, and a 50% reduction of NPS. Again, all cases would suggest that total nitrogen loading to Lake Tenkiller is excessive.

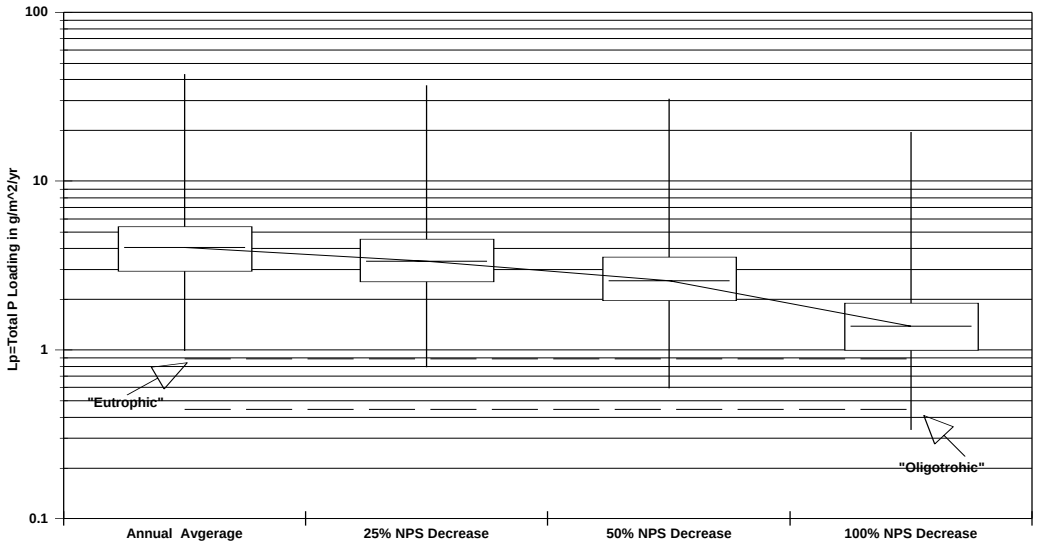

Figure 44. Simulated Distributions of Total Phosphorus Loading at Horseshoe Bend Plotted Against Vollenweider's (1976) Trophic Classification Model.

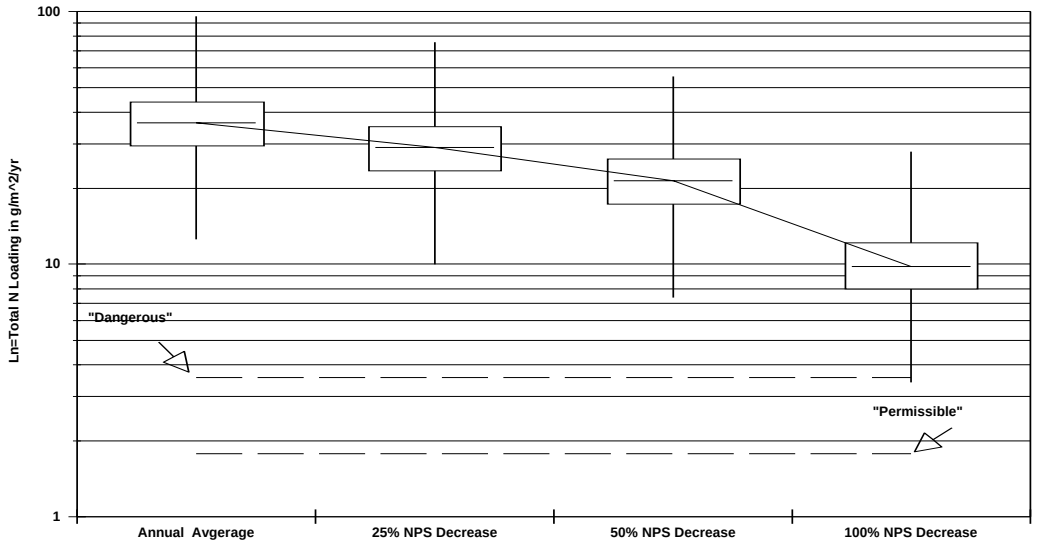

Figure 45. Simulated Distributions of Total Nitrogen Loading at Horseshoe Bend Plotted Against Wetzel (1983) Trophic Classification Model.

Carlson (1977) developed a numerical trophic state index for lakes that

ranges from 0 to 100 where each major division (10, 20, 30, etc.) represents a potential doubling in algal biomass. There are no set delineations describing oligo-, meso-, or eutrophic conditions, rather the index describes a trophic continuum. Based solely on total phosphorus loading to Lake Tenkiller, Table LXXIII shows the results of using this index. The median annual average simulated total phosphorus load corresponding to an index number of 81 would suggest that Lake Tenkiller would be experiencing loading sufficient to result in eutrophic conditions. Decreases in nonpoint source total phosphorus loads result in decreases in the index number. A 100% reduction of nonpoint sources of phosphorus would bring the lake closer to a mesotrophic classification.

TABLE LXXIII

CARLSON'S TROPHIC STATE INDEX RESULTS FOR LAKE TENKILLER (BASED ON ESTIMATED TOTAL PHOSPHORUS CONCENTRATION)

	Annual Average	25% Nonpoint Source Reduction	50% Nonpoint Source Reduction	100% Nonpoint Source Reduction
Minimum	61	57	53	45
25% Percentile	76	74	71	61
50% Percentile	81	78	74	65
75% Percentile	85	83	79	70
Maximum	115	113	110	104

Control of pollutants of agricultural origin presently focuses on their sources. A combination of methodologies collectively known as Best Management Practices (BMPs) is expected to reduce the amount of potential pollutant applied to the land and the amount of pollutant entering streams and ground water. These practices include controlling erosion and drainage,

conscientious crop management, prudent management of animals and associated pasture and rangeland, conservative utilization of nutrients, and precise methods for controlling pests. BMPs are expected to be adapted to specific site conditions rather than applied across the board. This requires close cooperation between the land manager who will implement the BMPs and the "experts" who will help to decide on the set of practices which should provide the most advantageous result.

General erosion and sediment control practices would include conservation tillage, contour farming, critical area planting, crop residue use, perennial vegetation at field borders, vegetated filter strips within fields, grassed waterways, water and sediment control basins, terracing, and riparian zone protection.

Some practices generally applicable include: 1) agricultural BMPs which reduce erosion and runoff; 2) efficient use of irrigation to minimize leaching of nitrate from the root zone before plant uptake can occur; 3) consideration of cropping sequences or cover crops that might utilize residual nutrients; 4) improved fertilizer management (timing, rates, and evaluation of soil nitrogen and phosphorus availability); 5) use of slow release N and P fertilizers for long-season crops; and, 6) use of nitrification and urease inhibitors where appropriate.

Management practices specifically designed to reduce nutrient export from land surfaces to be considered would include:

1) the use of soil surveys in determining soil productivity and

identifying environmentally sensitive sites and areas;

2) the use of producer documented yield histories to determine realistic yield expectations;

3) the use of soil testing for pH, phosphorus, and nitrogen;

4) testing manures and compost for nutrient content;

5) the use of proper timing, formulation, and application rates of fertilizers to maximize plant utilization while minimizing loss to the environment; and,

6) the use of buffer areas.

Grazing management practices, with regard to potential effects of animals grazing near streams, consider excluding animals from critical streambank areas and by providing stream crossings and/or hardened or alternative water access. Additionally, overgrazing of pasture and rangelands should be minimized to reduce degradation of vegetative cover and aid in the reduction of erosion and sediment loss.

Thoughtful selection of applicable BPMs, selectively applied first to those subbasins delivering the greatest nutrient loads, and best available technologies to reduce nutrient loads from point sources, would seem necessary to prolong the life of Lake Tenkiller.

The poultry industry is economically important to the region and it would be necessary to work with members of the industry to implement strategies which reduce nutrient loading but do not inhibit the growth and expansion of this industry. Methods for exporting animal wastes out of the basin in the form of

composted fertilizer product should be vigorously investigated and pursued. Poultry farmers in the region who produce more litter than they can responsibly apply to pasture and range lands on their property might consider selling the litter to other landowners in the region who have a need for a good fertilizer product. State agencies should also consider creating avenues for distributing the litter on an even wider scale to allow for removal of significant quantities of the litter out of the watershed.

As indicated in the graphics depicting sources of nutrients reaching Lake Tenkiller, point sources of phosphorus are not insignificant. In the analysis of historical water quality data, point source discharges were reduced over distance traveled to Lake Tenkiller using first-order kinetic assimilation coefficients. These coefficients were calculated based on available water quality data near the mid-point of the modeled system, and were assumed to represent the entire system. Point source discharges, such as that of the City of Fayetteville, traveling close to 160 kilometers to the lake, were shown contribute minimally to total phosphorus load. In the simulations, point source discharges were reduced at calculated rates only until they were input into the modeled system. This resulted in showing the simulated point source total phosphorus load reaching Horseshoe Bend (79,800 kg/yr) as being only slightly less (14%) than the sum of total phosphorus loads from each of the point sources (92,600 kg/yr).

As explained by Newbold (in Calow and Petts, 1992), biota remove nutrients from the water, but also regenerate nutrient to the water. The cycling of nutrients in a river may be intensive and still produce negligible effects on

nutrient concentrations. For phosphorus, which does not exchange with the atmosphere, biota cannot alter the log-run total transport substantially. Biota can influence chemical and physical forms of nutrients, and thus the timing of nutrient transport.

Allen (1995) discusses evidence that variations in discharge on seasonal and annual time scales influence whether nutrients are stored or exported. Allen discusses a phosphorus budget study by Meyers and Likens (1979) which estimated that 48% of annual inputs and 67% of exports of nutrients occurred during 10 days of the water year. Phosphorus accumulated in stream sediments and biota for 319 days of the year of study. In years of low flow, Meyer and Likens reported exports of phosphorus to be less than imports with export/import ratios as low as 0.6. In years of high flow the reverse occurred with export/import ratios as high as 1.5.

Different areas of a system may act differently in the rates with which nutrients are incorporated into biota and settle. Smaller streams may act as more efficient buffers to the transport of nutrients downstream, while larger riverine systems may simply act as through-put systems (Allen, 1995). In the long-term, small amounts of nutrients may be removed from the system, but would be transported downstream in pulses corresponding to flood events which would scour bottom sediments and organisms and transport them some distance downstream before they again settle to the bottom. Some nutrients would settle in flood plains and perhaps be effectively removed from the water system, but most would likely eventually reach a downstream reservoir. Modeling the river

system as steady-state may not offer the ability to accurately account for this complex and dynamic process.

In light of the previous discussion, it seems necessary to include some simple information regarding the total quantity of nutrients being input into the system. Using average annual total background loads as calculated with the U.S. EPA screening methods (Mills et al., 1985), a sum of estimated average annual point source loads at the source, average annual nonpoint source total phosphorus as estimated by SIMPLE, average annual total nitrogen loads estimated using unit-area loading methods, and QUAL2EU modeling results of average annual loads of these nutrient at the Horseshoe Bend Area of Lake Tenkiller, a simple calculation was made to estimate the amounts of each nutrient being assimilated by the system. Table LXXIV shows the results of this calculation. If nutrients are not removed from the system by settling and biotic uptake, and are eventually transported downstream to Lake Tenkiller, then the estimated average annual loads of total phosphorus and total nitrogen are 322,300 and 2,820,000 kg/yr, respectively. These values are 31,300 and 340,000 kg/yr, respectively, higher than modeled annual averages and represent assimilated nutrients. In terms of total inputs to the QUAL2EU model, employing first-order kinetic assimilation rates results in a reduction of total phosphorus from 103,300 kg/yr at the source to 82,000 kg/yr where input into the model. This accounts for about 67% all total assimilation of total phosphorus. The QUAL2EU model reduces the total phosphorus load by only 10,000 kg/yr for the remainder of the flow distance from the point source input point down to Lake

Tenkiller. Thus, the developed assimilation rates, and the settling coefficients for QUAL2EU, would seem to differ significantly. If we assume that all phosphorus input into the system moves down to the reservoir, then point source contributions to total phosphorus load increases to 32% from the 27.5% based on QUAL2EU results. For nitrogen, the point sources represent a small fraction of total inputs and no significant change in relative contribution is seen.

TABLE LXXIV

ESTIMATES OF TOTAL AVERAGE ANNUAL LOADS OF PHOSPHORUS AND NITROGEN ENTERING THE ILLINOIS RIVER BASIN ABOVE THE HORSESHOE BEND AREA OF LAKE TENKILLER INCLUDING SOURCES AND SINKS

Total Inputs by Source	Estimated Average Annual Total Phosphorus Load (kg/yr)	Estimated Average Annual Total Nitrogen Load (kg/yr)
a) Background	25,000 (7.8 %)	610,000 (21.6%)
b) Point Source	103,300 (32.0%)	70,000 (2.5%)
c) Nonpoint Source	194,000 (60.2%)	2,140,000 (75.9%)
d) Total	322,300 (100%)	2,820,000 (100%)
QUAL2EU Estimates at Horseshoe Bend		
e) Background	21,900	605,000
f) Point Source	79,800	69,500
g) Nonpoint Source	189,000	1,800,000
h) Total	291,000	2,480,000
i) QUAL2EU Point Source Input After Application of First-Order Assimilation Rate	82,000 79% of Total Point Source (b)	60,200 86% of Total Point Source (b)
j) Point Source Assimilation Before QUAL2EU (b - i)	21,300 6.6% of Total (d)	9,800 0.3%of Total (d)
Assimilation in QUAL2EU (d - h - j)	10,000 3.1% of Total (d)	330,200 11.7% of Total (d)
Total Assimilation (d-h)	31,300 9.7% of Total (d)	340,000 12.1% of Total (d)

CHAPTER VII

CONCLUSIONS AND RECOMMENDATIONS

Based on the information available, we can draw several conclusions in regard to nutrient transport in the Illinois River Basin. First, there has been a dramatic increase in the number of poultry houses in the basin since 1980. The birds raised in these houses produce significant quantities of manure, the bulk of which is applied to pasture and rangeland within the basin. The automatic samplers gathered limited information with respect to the typical runoff concentration at only two points in the basin. However, the results were what would be expected. During runoff events there is an increase in nutrient concentration which would correspond to surface nutrients carried to the stream in either dissolved or sediment bound forms. Ultimately these nutrients reach the lower portion of the basin at Lake Tenkiller.

Historical water quality and discharge analysis suggest that there are increasing trends at some locations with regard to phosphorus concentration and load, and more evident increasing trends with regard to nitrogen concentration and load. The discharge and concentration data combined suggest that a great majority of the nutrient load carried to the Illinois River and its tributaries occurs during "high" flows, again suggesting that these nutrients are mobilized during

runoff events.

Data constructed for use with the SIMPLE model used the locations of poultry houses to estimate soil phosphorus distributions in pasture lands. Runoff, calculated from rainfall data, then transports a portion of the nutrients in the soil to the river and its tributaries. Subbasins with the greatest densities of poultry houses, and hence high estimated soil phosphorus levels, generally delivered the greatest quantities of nutrients to the Illinois River and its tributaries.

These estimates of nutrient loading, when routed through the riverine model QUAL2EU, closely approximate the nutrient loading at the Horseshoe Bend area of Lake Tenkiller based on historical data. Estimates of point and background nutrient loading in the basin allowed for approximations of the distribution of the total nutrient load between nonpoint, point, and background sources. Nonpoint sources account for the majority of nutrient loading. Point source nutrient loading is also significant, especially for phosphorus.

Finally, models of lake trophic status suggest that the average annual loading of both phosphorus and nitrogen at the Horseshoe Bend area of Lake Tenkiller is excessive and will likely lead to a continued deterioration of lake water quality. In fact, even with 25%, 50%, and 100% reductions of nonpoint source nutrient loading, the lake would still be receiving loads of phosphorus and nitrogen which would exacerbate the eutrophic characteristics already documented.

The modeling study performed on the Illinois River Basin, utilizing GIS for

estimation of nutrient export in combination with a stream routing model, provides results which can be used to develop management strategies for the reduction of nutrient loading to Lake Tenkiller. Critical areas can be identified which would be obvious starting points to mitigate nutrient export from the basin. The stream routing portion of the modeling scheme allowed for the estimation of annual contributions of nutrients from various sources and can be employed to determine the effectiveness of various management strategies applied.

This study did not test the effectiveness of specific management strategies but rather the effects of specified reductions in nonpoint nutrient loading. The modeling scheme could be easily modified to test the effects of certain specific management practices. The SIMPLE model can provide estimates of nutrient export under specific management schemes which would be accounted for in the land use/land cover data layer and associated parameters, as well as alterations in the ways fertilizer is applied to fields, and the initial levels of phosphorus in fields related to land use.

There are areas where model confidence could be enhanced. The initial phosphorus data layer for the study was based on county or watershed level soil testing data which may not be accurate. A more representative soil testing technique using samples from each of the various land uses weighted by the areal extent of each land use may provide a more accurate picture of present initial phosphorus levels.

The digital elevation data for the basin was not all at the most desirable resolution of 30 meters. The use of 80 meter resolution data in a significant

portion of the Arkansas (seven 7.5 minute quadrangles) part of the Illinois River Basin affected the output of the digital terrain model in that area. Since this portion of the basin is generally more topographically uniform, certain areas may not be modeled as accurately as those with 30 meter resolution data.

The land use/land cover data used in this study was from 1985. The use of more recent information via satellite imagery or some other more current source would be valuable. Confined animal sites were also obtained from this 1985 data and have likely changed significantly over the last decade. More current information in these areas would lead to greater confidence in the results of the modeling scheme.

While the focus of many nonpoint nutrient studies focus on phosphorus, information on nitrogen loading is useful and important in considerations of lake management. Nitrogen loading values in this study were based on unit loading assumptions. The SIMPLE model does not model the export of nitrogen. A more refined technique for accounting for nitrogen would enhance these results.

The riverine model could be expanded to travel further up the Illinois River and its tributaries to more accurately model both point and nonpoint nutrient inputs. This would allow for a more precise modeling of nutrient input to specific points within the basin. Obtaining more basin specific parameters in terms of hydrologic characteristics and nutrient and algal reaction coefficients would improve confidence in modeled predictions.

The sources of nutrients were divided into point (municipalities), nonpoint, and background. The nutrient loads determined to be of nonpoint origin could be

further evaluated to include other sources. A survey of septic systems in the basin and approximations of the nutrient contribution from them could be valuable. Range cattle were essentially disregarded in the modeling effort and some estimates of the impact of these animals on the system should be included.

Overall, the modeled system corresponds reasonably well with observed historical data. It can be an effective tool in analyzing the dynamics of nutrient transport in a large watershed, and it can provide useful information for directing management schemes to improve water quality.

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APPENDIX A

DEVELOPMENT OF DISCHARGE COEFFICIENTS FOR THE QUAL2EU MODEL

The QUAL2EU model offers two options for describing the hydraulic characteristics of the system, a geometric representation and a functional representation. Data were readily available to determine discharge coefficients for the functional representation. The USGS ADAPS Database offers access to stage-discharge rating tables for gaging stations, and also to discharge measurement data. Stage-discharge rating tables with depth and discharge data were available for each of the eight gaging stations in the basin. Discharge measurement data, also available at all eight gaging stations, which includes periodic observations of stream velocity and discharge, among other parameters, were downloaded for WTRs 1980 - 1993.

Regression was used to solve :

$$\bar{\ } = aQ^{b}$$

and

$$d = \alpha Q^{\beta}$$

for a, b, α, and ,, the empirical constants. Tables LXXV and LXXVI include the coefficients developed for each of the eight gaging stations in the Illinois River Basin. Reaches in the developed QUAL2EU model of the Illinois River, and its tributaries, which did not have a USGS gaging station at their starting location, were assigned interpolated velocity and discharge coefficients based on coefficients determined for gaging stations above and/or below the reach.

TABLE LXXV

COEFFICIENTS FOR VELOCITY DEVELOPED FOR
USGS GAGING STATIONS IN THE
ILLINOIS RIVER BASIN

USGS Gaging Station Identification	Coefficient for Velocity (*a*)	Exponent for Velocity (*b*)	Velocity/ Discharge Observations (n)	Velocity/ Discharge r^2
07194800	0.248	0.338	28	0.73
07195400	0.375	0.140	26	0.19
07195500	0.269	0.248	27	0.46
07196500	0.148	0.286	36	0.58
07195000	0.283	0.499	52	0.68
07196000	0.371	0.276	35	0.68
07196900	0.313	0.291	56	0.79
07197000	0.306	0.316	39	0.60

TABLE LXXVI

COEFFICIENTS FOR DEPTH DEVELOPED FOR
USGS GAGING STATIONS IN THE
ILLINOIS RIVER BASIN

USGS Gaging Station Identification	Coefficient for Depth (α)	Exponent for Depth ( )	Depth/ Discharge Observations (n)	Depth/ Discharge r^2
07194800	0.203	0.558	34	0.99
07195400	0.255	0.481	19	0.99
07195500	0.123	0.634	20	0.99
07196500	0.122	0.561	18	0.99
07195000	0.102	0.793	21	0.98
07196000	0.120	0.468	19	0.98
07196900	0.227	0.456	29	0.99
07197000	0.132	0.587	26	0.99

APPENDIX B

QUAL2EU MODEL CALIBRATION

TECHNIQUES AND RESULTS

The QUAL2EU model, designed to simulate phosphorus and nitrogen transport from the Illinois River Basin down to the Horseshoe Bend Area of Lake Tenkiller, was designed to accommodate SIMPLE outputs from the various subbasins comprising the basin as a whole. The model was then calibrated using water quality data (water years 1980 - 1993) from USGS gaging stations and scenic river stations within the basin. Mean annual statistics were calculated for discharge, dissolved, organic and total phosphorus, and for organic nitrogen, ammonia, nitrate + nitrite, and total nitrogen. The calculated mean annual values were incorporated into the QUAL2EU model as initial conditions at sites in the model corresponding to the water quality monitoring sites. Mean annual discharges were used at each of the four headwater locations, and gaging station data not at headwater locations were used to calibrate flow proceeding down toward the lake. Model runs were performed until average predicted nutrient concentrations at monitoring stations were within 15% of historical annual average concentrations.

Previous experience with the QUAL2E model has shown that accurate hydrologic characteristics are necessary for the model. USGS ADAPS data available for each of the gaging stations was used to calculated coefficients of depth and velocity at each station. Rating tables at each station were used to calculate coefficients of depth. Also available for each station were station reports with velocity measurements recorded with discharge.

Table LXXVII lists statistical summaries of nutrient water quality data for scenic river monitoring sites in the Illinois River Basin. Table LXXVIII lists final

QUAL2EU reaction coefficients and information on sources. Figures 56 - 67 compare simulated discharge and water quality to historical means at USGS gaging stations and scenic river sites.

TABLE LXXVII

STATISTICAL SUMMARIES OF NUTRIENT DATA AT SCENIC RIVER MONITORING SITES IN THE ILLINOIS RIVER BASIN FOR WATER YEARS 1980 - 1993

Station	Statistic	Total Nitrogen (mg/l)	Ammonia (mg/l)	Total Kjeldahl Nitrogen (mg/l)	Nitrate + Nitrite (mg/l)	Total Phosphorus (mg/l)	Ortho-Phosphate as P (mg/l)
SR1	Mean	2.3	0.3	0.5	1.9	0.33	0.16
	Standard Deviation	1.1	0.4	0.3	0.9	0.40	0.12
	Count	116	5	121	121	121	64
SR2	Mean	2.0	0.1	0.4	1.7	0.24	0.13
	Standard Deviation	1.4	0.1	0.3	1.2	0.22	0.10
	Count	115	22	134	134	137	80
SR3	Mean	1.8	0.1	0.4	1.5	0.23	0.14
	Standard Deviation	1.0	0.1	0.3	0.8	0.18	0.12
	Count	107	29	133	133	136	79
SR4	Mean	1.8	0.1	0.4	1.5	0.21	0.13
	Standard Deviation	1.0	0.1	0.4	0.8	0.18	0.12
	Count	107	28	133	133	135	78
SR4.5	Mean	---	0.1	0.4	1.3	0.19	0.12
	Standard Deviation	---	0.1	0.2	0.7	0.09	0.07
	Count	0	30	27	27	30	29
SR5	Mean	1.8	0.1	0.5	1.4	0.19	0.10
	Standard Deviation	1.2	0.1	0.6	0.8	0.26	0.10
	Count	117	27	142	142	144	87

TABLE LXXVIII

REACTION COEFFICIENTS USED IN THE QUAL2EU MODEL OF THE ILLINOIS RIVER BASIN

Parameter	Units	Value	Source
Ratio of chlorophyll *a* to Algal biomass	µg Chl *a*/mg A	10	U.S. EPA (1991b)
Fraction of algal biomass that is nitrogen	mg N/mg A	0.08	Bowie et al. (1985), U.S. EPA (1991b)
Fraction of algal biomass that is phosphorus	mg P/mg A	0.011	Bowie et al. (1985), U.S. EPA (1991b)
Maximum algal growth rate	day^{-1}	2.5	Bowie et al. (1985), U.S. EPA (1991b)
Algal respiration rate	day^{-1}	0.15	U.S. EPA (1991b)
Michaelis-Menton half-saturation constant for nitrogen	mg N/l	0.03	U.S. EPA (1991b)
Michaelis-Menton half-saturation constant for phosphorus	mg P/l	0.005	U.S. EPA (1991b)
Non-algal light extinction coefficient	ft^{-1}	0.3	
Linear algal self-shading coefficient	ft^{-1}/((µg Chl *a*)/l)	0.0432	U.S. EPA (1991b)
Nonlinear algal self-shading coefficient	ft^{-1}/(λg Chl *a*/l)$^{2/3}$	0.003	U.S. EPA (1991b)
Algal preference for ammonia		0.7	U.S. EPA (1991b)
Algal settling rate	ft/day	0.5	Bowie et al. (1985), U.S. EPA (1991b)
Benthos source rate for dissolved phosphorus	mg P/ft^2-day	0	U.S. EPA (1991b)
Benthos source rate for ammonia	mg O/ft^2-day	0	U.S. EPA (1991b)

TABLE LXXVIII

REACTION COEFFICIENTS USED IN THE QUAL2EU MODEL OF THE ILLINOIS RIVER BASIN

Parameter	Units	Value	Source
Organic nitrogen settling rate	day^{-1}	0.01	Bowie et al. (1985), U.S. EPA (1991b)
Organic phosphorus settling rate	day^{-1}	0.01	Bowie et al. (1985), U.S. EPA (1991b)
Rate constant for biological oxidation of ammonia to nitrite	day^{-1}	0.2	Bowie et al. (1985), U.S. EPA (1991b)
Rate constant for biological oxidation of nitrite to nitrate	day^{-1}	2.0	Bowie et al. (1985), U.S. EPA (1991b)
Rate constant for hydrolysis of organic nitrogen to ammonia	day^{-1}	0.01	Bowie et al. (1985)
Rate constant for decay of organic phosphorus to dissolved phosphorus	day^{-1}	0.1	Bowie et al. (1985), U.S. EPA (1991b)

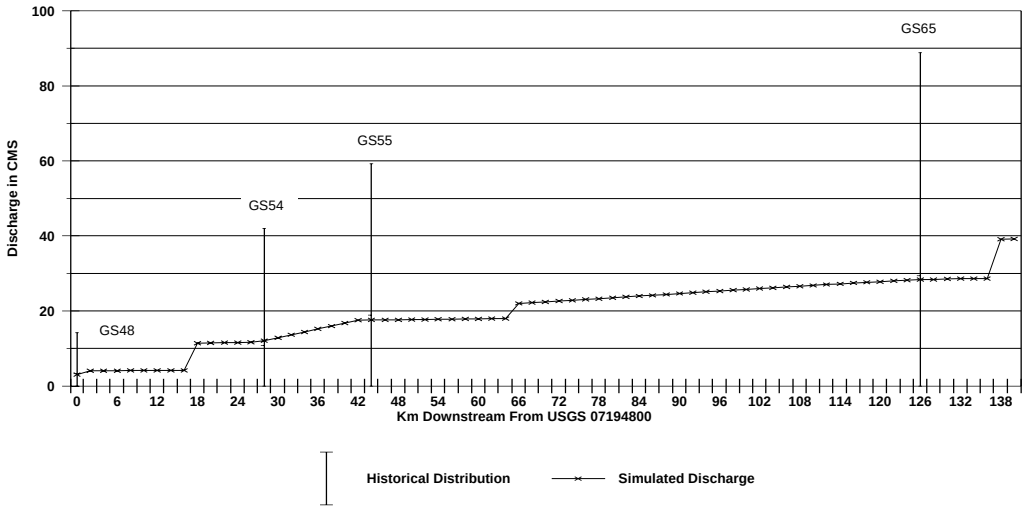

Figure 46. Results of Calibration of QUAL2EU Model with Historical Discharge at Mainstem Stations on the Illinois River. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

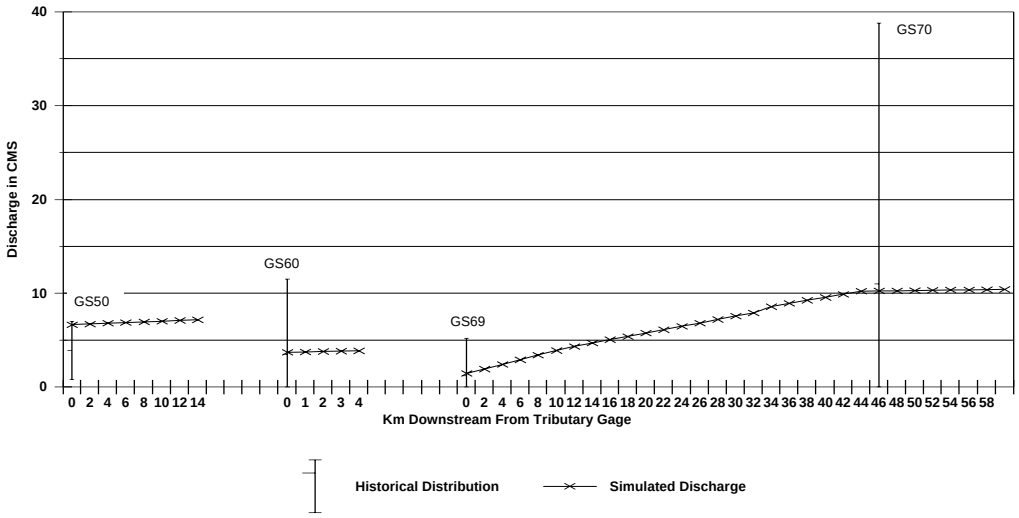

Figure 47. Results of Calibration of QUAL2EU Model with Historical Discharge at Tributary Stations in the Illinois River Basin. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

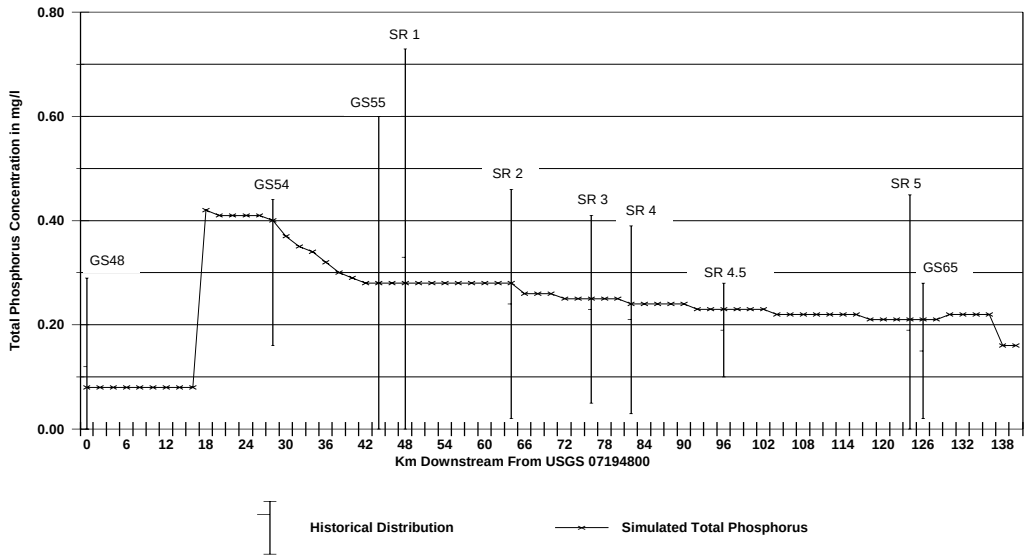

Figure 48. Results of Calibration of QUAL2EU Model with Historical Total Phosphorus Data at Mainstem USGS and Water Quality Monitoring Stations. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

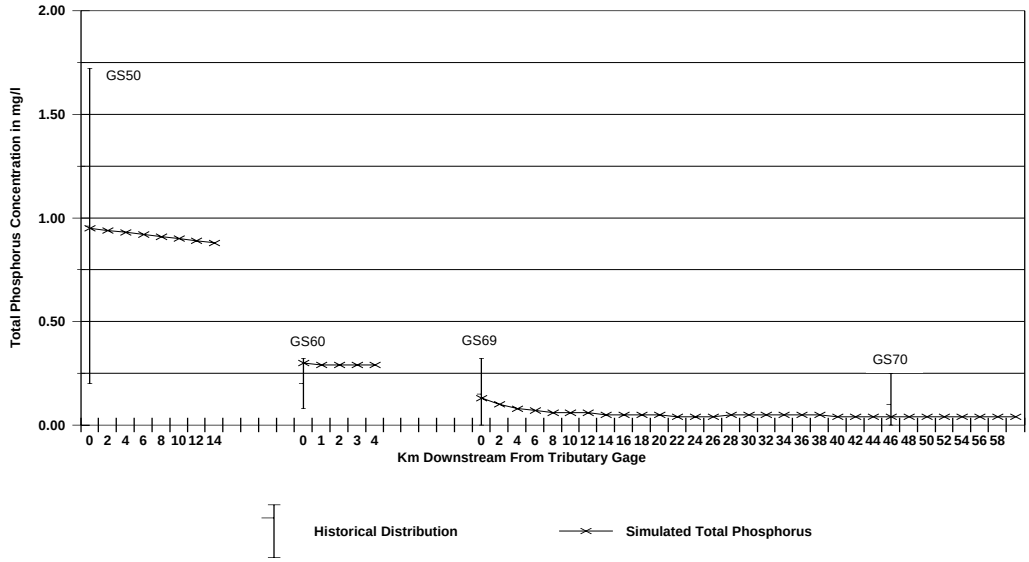

Figure 49. Results of Calibration of QUAL2EU Model with Historical Total Phosphorus Data at Tributary Stations in the Illinois River Basin. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

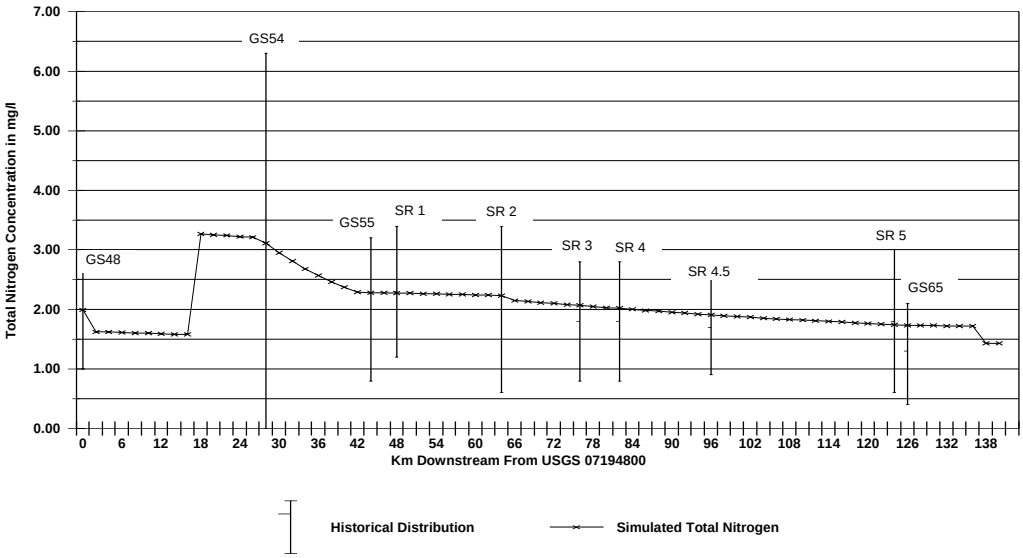

Figure 50. Results of Calibration of QUAL2EU Model with Historical Total Nitrogen Data at Mainstem Stations on the Illinois River. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

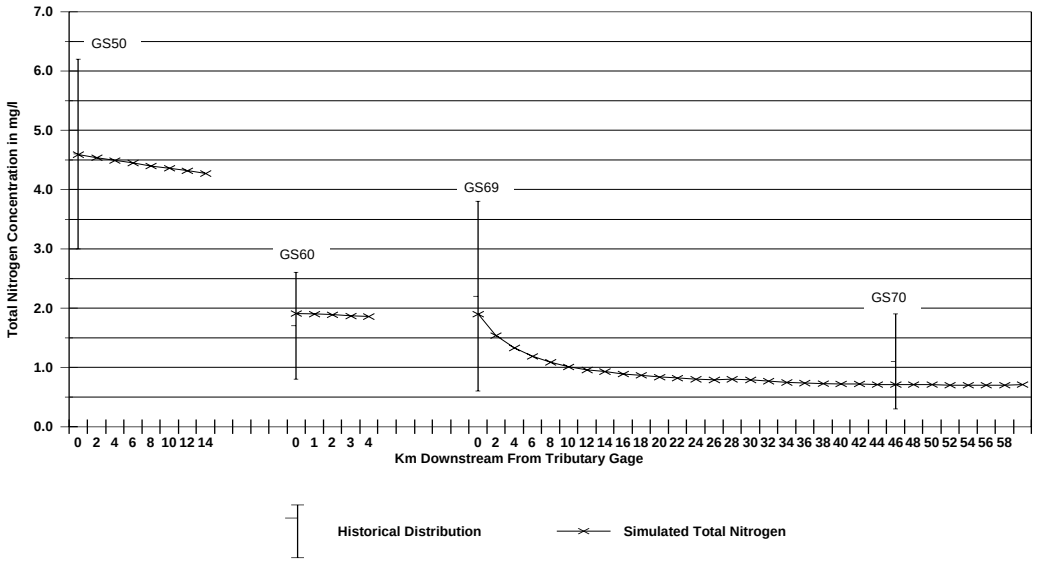

Figure 51. Results of Calibration of QUAL2EU Model with Historical Total Nitrogen Data at Tributary Stations in the Illinois River Basin. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

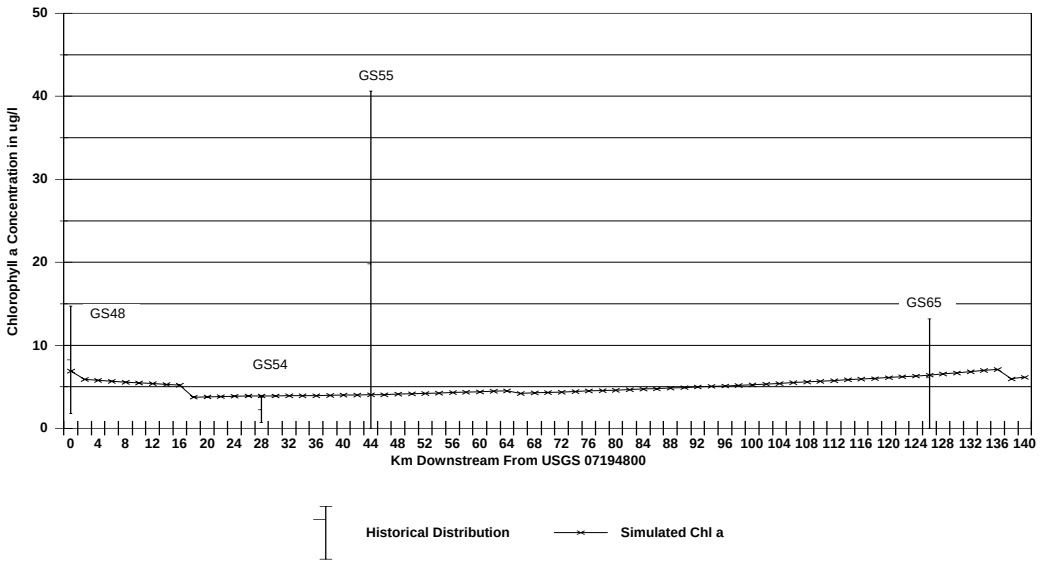

Figure 52. Results of Calibration of QUAL2EU Model with Historical Chlorophyll *a* Data at Mainstem Stations on the Illinois River. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

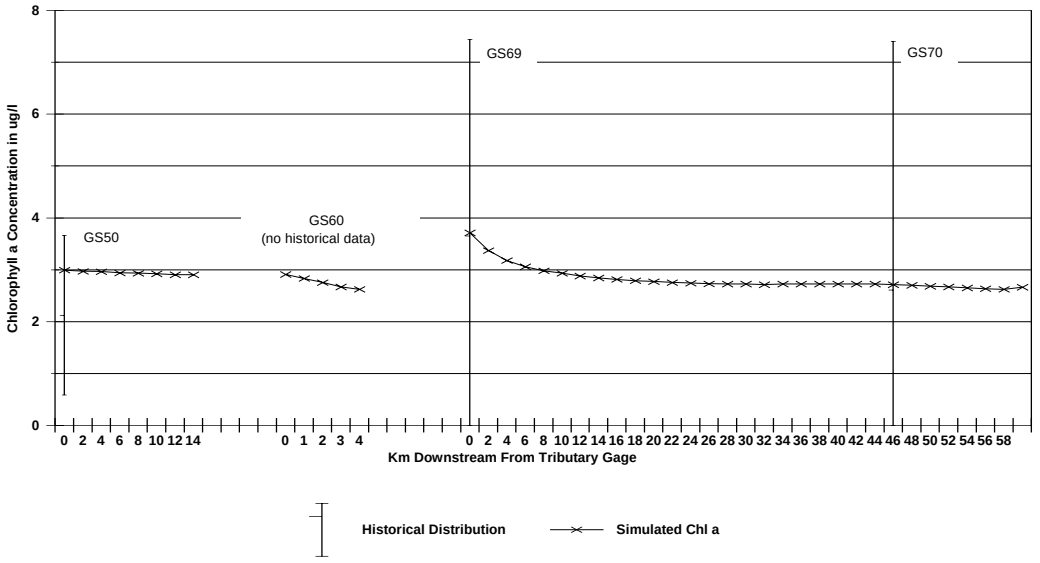

Figure 53. Results of Calibration of QUAL2EU Model with Historical Chlorophyll *a* Data at Tributary Stations in the Illinois River Basin. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

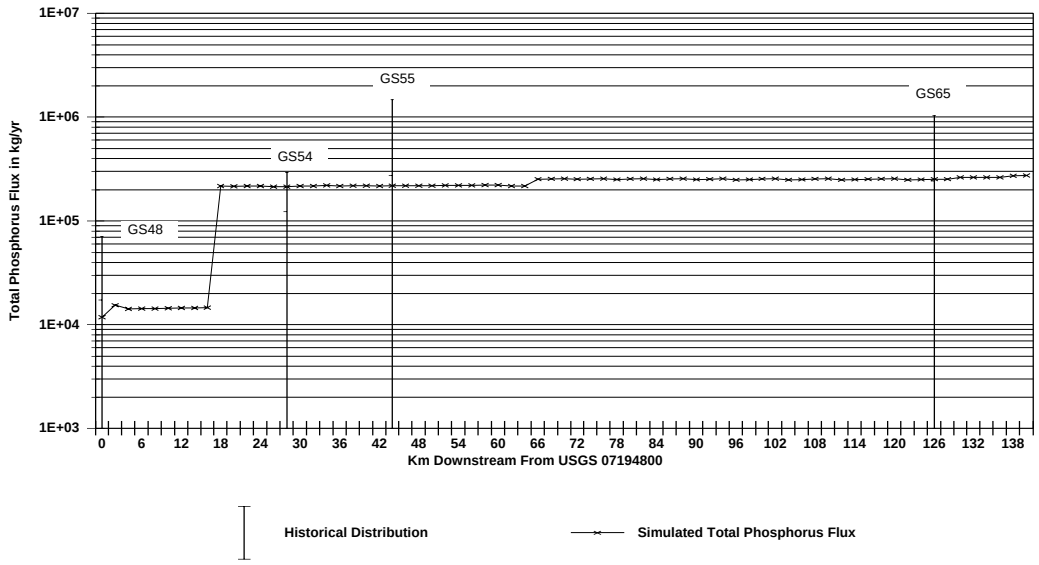

Figure 54. Results of Calibration of QUAL2EU Model with Historical Total Phosphorus Loading Data at Mainstem Stations on the Illinois River. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

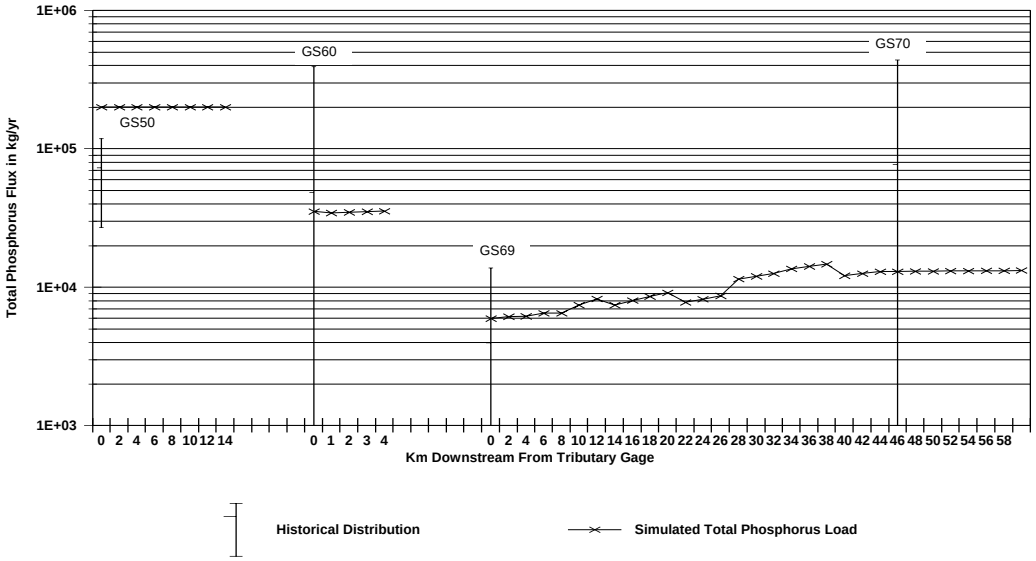

Figure 55. Results of Calibration of QUAL2EU Model with Historical Total Phosphorus Loading Data at Tributary Stations in the Illinois River Basin. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

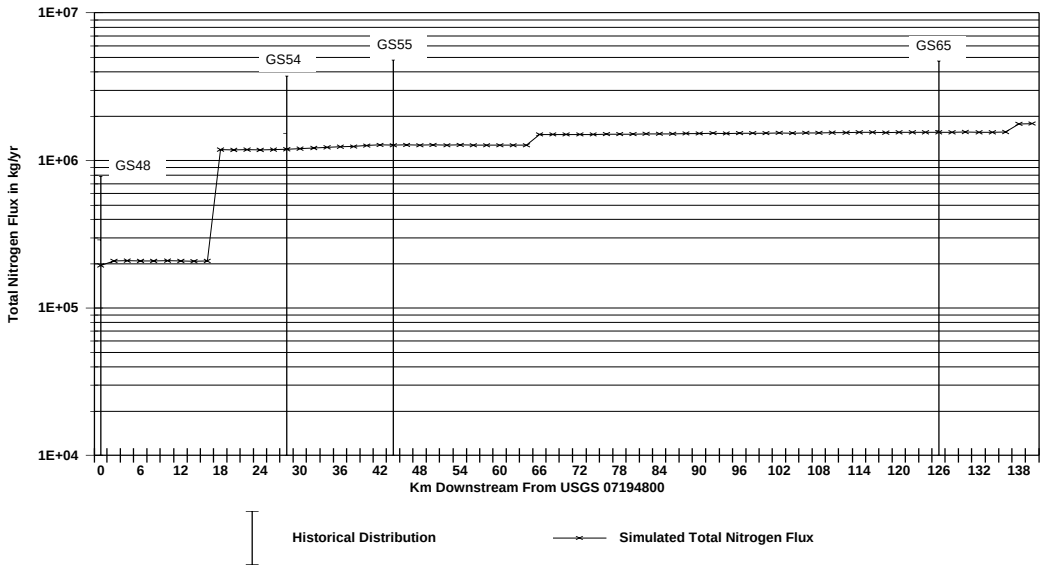

Figure 56. Results of Calibration of QUAL2EU Model with Historical Total Nitrogen Loading Data at Mainstem Stations on the Illinois River. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

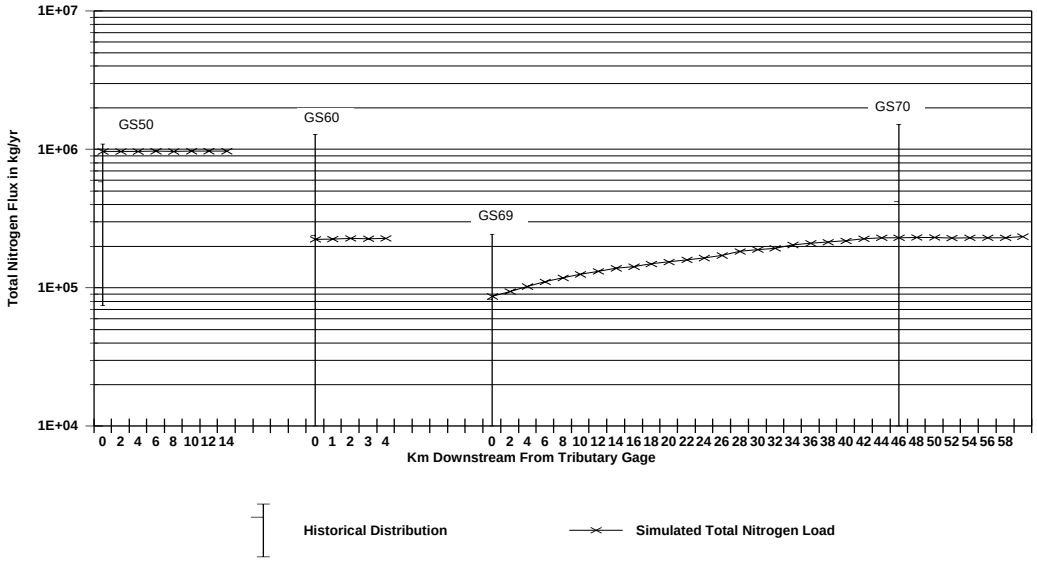

Figure 57. Results of Calibration of QUAL2EU Model with Historical Total Nitrogen Loading Data at Tributary Stations in the Illinois River Basin. Historical Data (WTRs 1980 - 1993) is Shown as the Mean Plus and Minus One Standard Deviation.

APPENDIX C

STANDARD CURVES, DATA, AND FIGURES ASSOCIATED WITH AUTOMATIC SAMPLING OF THE ILLINOIS RIVER AND BARON FORK CREEK

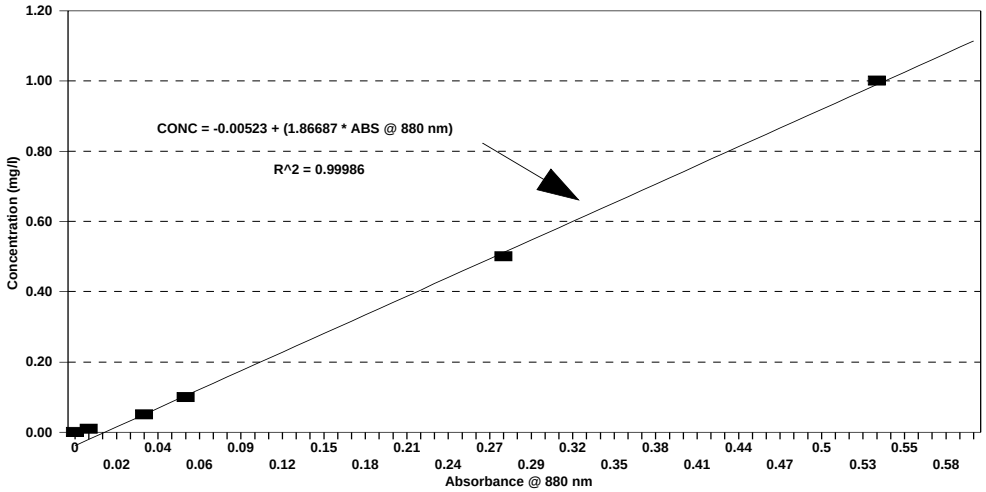

Figure 58. Standard Curve Developed for Analysis of Total Phosphorus Concentration in Automatic Sampler Observations.

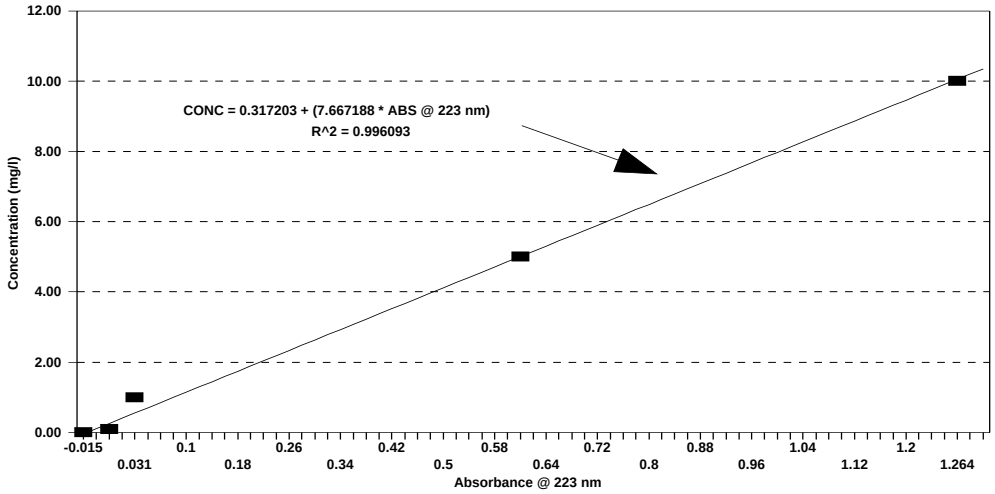

Figure 59. Standard Curve Developed for Analysis of Total Nitrogen Concentration in Automatic Sampler Observations.

TABLE LXXIX

MEASURED CONCENTRATION AND DISCHARGE DATA FOR
AUTOMATIC SAMPLING AT USGS 07196500
SEPTEMBER 10 - 12, 1993

Time	Discharge (cfs)	Total P (mg/l)	Total N (mg/l)
3:00 AM	648	0.114	1.64
6:00 AM	669	0.114	1.58
9:00 AM	739	0.126	1.55
12:00 PM	795	0.126	1.51
3:00 PM	823	0.134	1.57
6:00 PM	829	0.139	1.81
9:00 PM	829	0.163	1.84
12:00 AM	829	0.121	1.58
3:00 AM	818	0.119	1.45
6:00 AM	795	0.119	1.58
9:00 AM	783	0.119	1.60
12:00 PM	767	0.119	1.63
3:00 PM	739	0.121	1.58
6:00 PM	728	0.119	1.61
9:00 PM	706	0.116	1.60
12:00 AM	695	0.114	1.74
3:00 AM	690	0.119	1.57
6:00 AM	679	0.119	1.61
9:00 AM	669	0.114	1.64
12:00 PM	658	0.121	1.59

TABLE LXXX

MEASURED CONCENTRATION AND DISCHARGE DATA FOR
AUTOMATIC SAMPLING AT USGS 07196500
SEPTEMBER 26 - 27, 1993

Time	Discharge (cfs)	Total P (mg/l)	Total N (mg/l)
3:36 AM	1,422	0.150	1.83
3:41 AM	1,470	0.150	1.63
5:41 AM	1,720	0.156	2.35
7:41 AM	1,930	0.160	1.91
9:41 AM	2,060	0.162	1.99
11:41 AM	2,130	0.168	1.86
1:41 PM	2,160	0.168	1.84
3:41 PM	2,170	0.173	1.77
5:41 PM	2,190	0.282	1.60
7:41 PM	2,220	0.173	1.95
9:41 PM	2,230	0.191	1.99
11:41 PM	2,230	0.185	1.89
1:41 AM	2,230	0.264	2.06
3:41 AM	2,220	0.244	1.72
5:41 AM	2,170	0.185	1.81
7:41 AM	2,120	0.168	1.84
9:41 AM	2,050	0.152	1.72
11:41 AM	2,000	0.162	1.75
1:41 PM	1,950	0.146	1.78
3:41 PM	1,910	0.150	1.59
5:41 PM	1,870	0.156	1.56
7:41 PM	1,830	0.148	1.67
9:41 PM	1,810	0.140	1.62

TABLE LXXXI

MEASURED CONCENTRATION AND DISCHARGE DATA FOR
AUTOMATIC SAMPLING AT USGS 07196500
OCTOBER 21 - 22, 1993

Time	Discharge cfs	Total P mg/l as P	Total N mg/l as N
12:00 AM	1,450	0.113	2.37
12:30 AM	1,470	0.115	2.19
1:00 AM	1,490	0.113	2.46
3:00 AM	1,570	0.125	2.28
5:00 AM	1,670	0.125	2.72
7:00 AM	1,890	0.131	2.55
9:00 AM	2,180	0.151	3.17
11:00 AM	2,450	0.179	2.81
1:00 PM	2,630	0.201	2.55
3:00 PM	2,780	0.209	2.81
5:00 PM	2,850	0.215	2.90
7:00 PM	2,910	0.227	2.72
9:00 PM	2,930	0.223	2.64
11:00 PM	2,930	0.233	2.46
1:00 AM	2,930	0.233	2.81
3:00 AM	2,850	0.221	2.10
5:00 AM	2,780	0.274	3.02
7:00 AM	2,670	0.167	2.46
9:00 AM	2,590	0.195	2.64
11:00 AM	2,500	0.181	2.46
1:00 PM	2,410	0.171	3.26
3:00 PM	2,320	0.169	2.99

TABLE LXXXII

MEASURED CONCENTRATION AND DISCHARGE DATA FOR
AUTOMATIC SAMPLING AT USGS 07197000
AUGUST 25 - 26, 1993

Time	Discharge (cfs)	Total P (mg/l)	Total N (mg/l)
8:00 AM	84	0.660	3.67
11:00 AM	106	0.178	2.81
2:00 PM	108	0.156	3.52
5:00 PM	106	0.129	3.09
8:00 PM	99	0.104	2.99
11:00 PM	90	0.111	2.91
2:00 AM	86	0.104	2.95
5:00 AM	82	0.099	2.89
8:00 AM	78	0.092	2.90
11:00 AM	78	0.097	2.97
2:00 PM	72	0.075	2.12
5:00 PM	67	0.072	2.08

TABLE LXXXIII

MEASURED CONCENTRATION AND DISCHARGE DATA FOR AUTOMATIC SAMPLING AT USGS 07197000 OCTOBER 16 - 18, 1993

Time	Discharge (cfs)	Total P (mg/l)	Total N (mg/l)
7:30 PM	1,250	0.280	1.66
7:45 PM	1,660	0.369	2.00
8:00 PM	2,050	0.615	2.32
8:15 PM	2,425	1.327	3.59
9:30 PM	3,965	1.002	4.41
11:30 PM	4,805	0.576	3.68
1:30 AM	4,420	0.505	3.39
3:30 AM	3,875	0.400	2.65
5:30 AM	3,985	0.436	2.44
7:30 AM	4,155	0.392	2.46
9:30 AM	4,030	0.227	2.49
11:30 AM	3,650	0.268	1.78
1:30 PM	3,300	0.233	2.07
3:30 PM	3,000	0.215	2.06
5:30 PM	2,760	0.179	1.97
7:30 PM	2,585	0.168	2.25
10:30 PM	2,350	0.162	2.04
12:30 AM	2,225	0.150	1.92

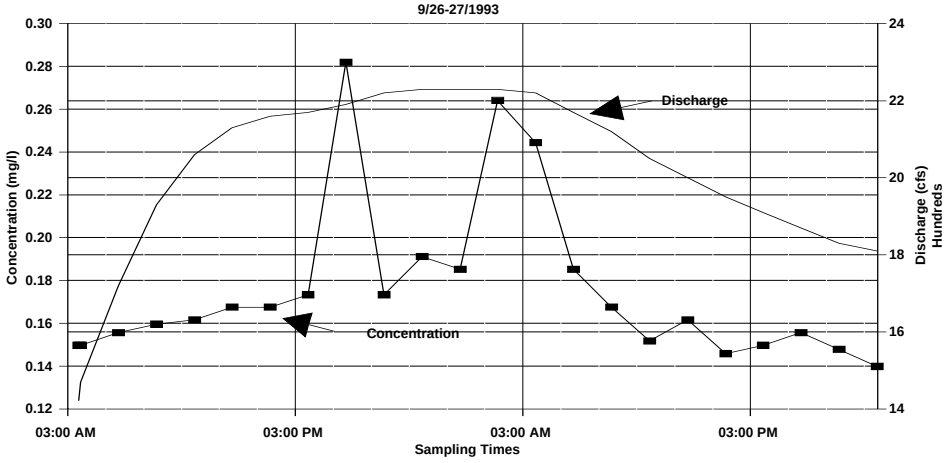

Figure 60. Measured Total Phosphorus Concentration and Discharge at USGS 07196500 September 26 - 27, 1993.

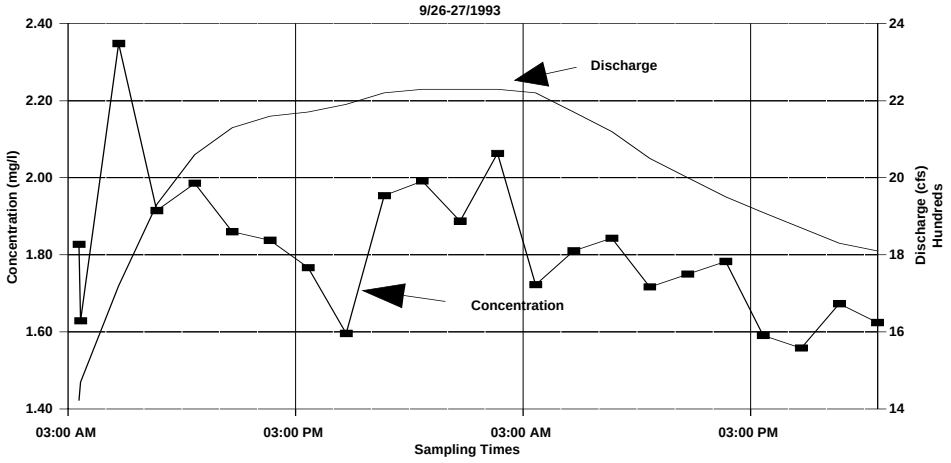

Figure 61. Measured Total Nitrogen Concentration and Discharge at USGS 07196500 September 26 - 27, 1993.

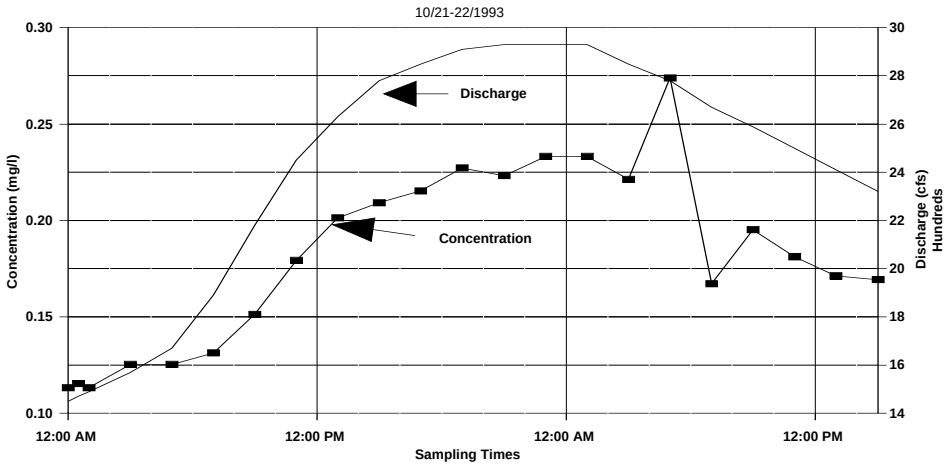

Figure 62. Measured Total Phosphorus Concentration and Discharge at USGS 07196500 October 21 - 22, 1993.

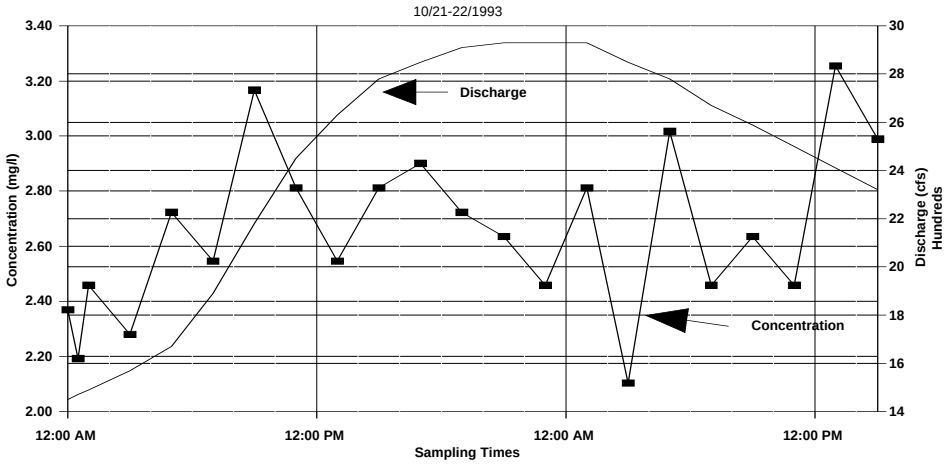

Figure 63. Measured Total Nitrogen Concentration and Discharge at USGS 07196500 October 21 - 22, 1993.

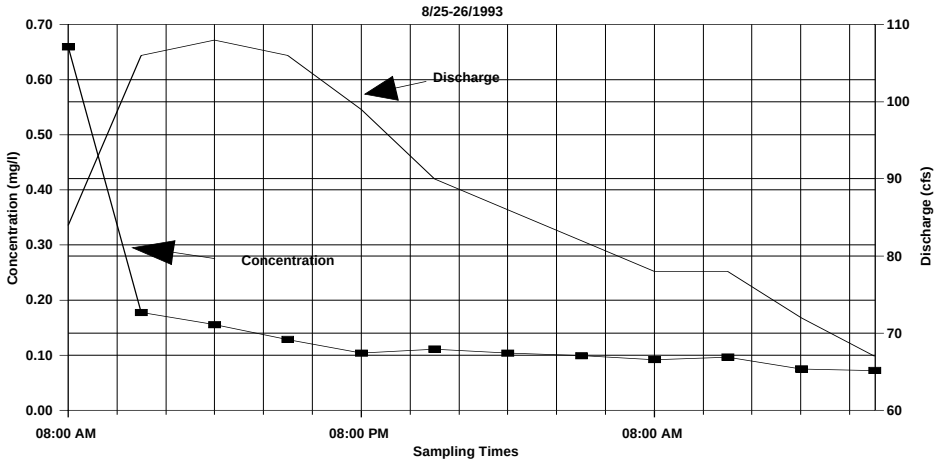

Figure 64. Measured Total Phosphorus Concentration and Discharge at USGS 07197000 August 25 - 26, 1993.

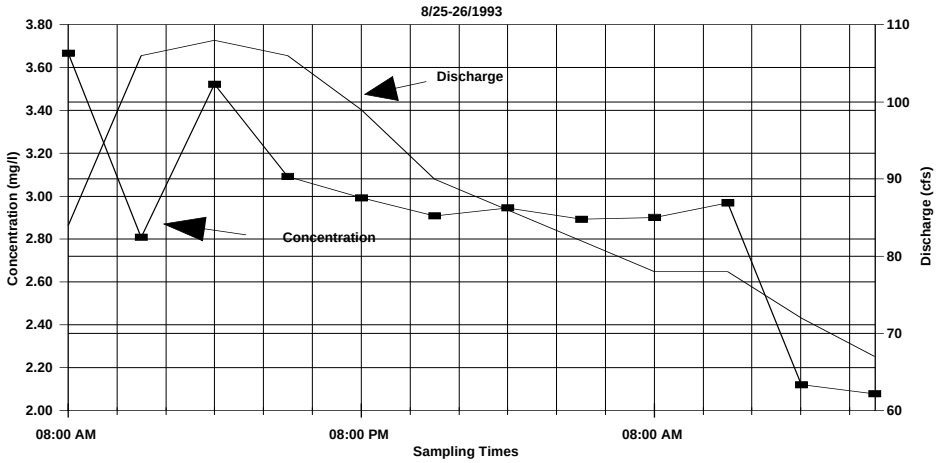

Figure 65. Measured Total Nitrogen Concentration and Discharge at USGS 07197000 August 25 - 26, 1993.

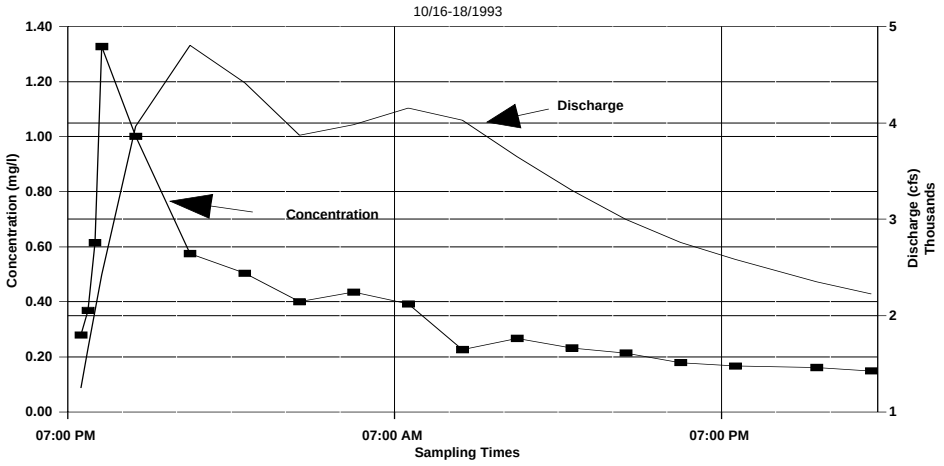

Figure 66. Measured Total Phosphorus Concentration and Discharge at USGS 07197000 October 16 - 18, 1993.

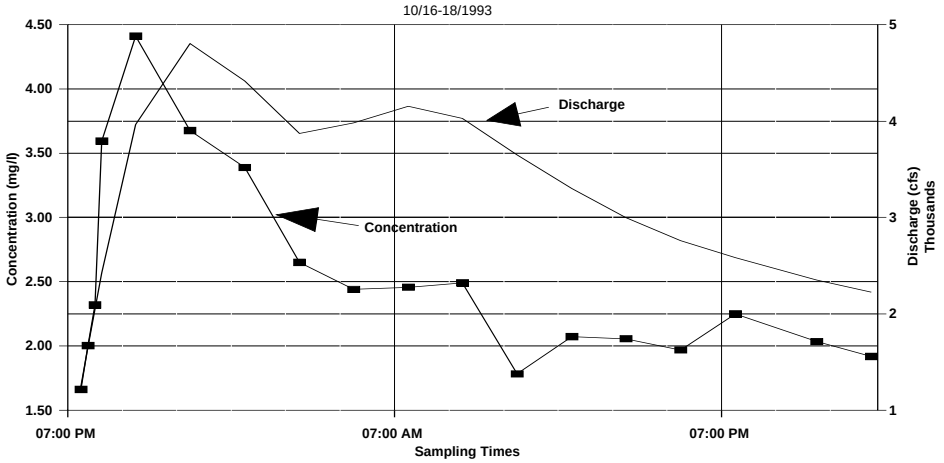

Figure 67. Measured Total Nitrogen Concentration and Discharge at USGS 07197000 October 16 - 18, 1993.